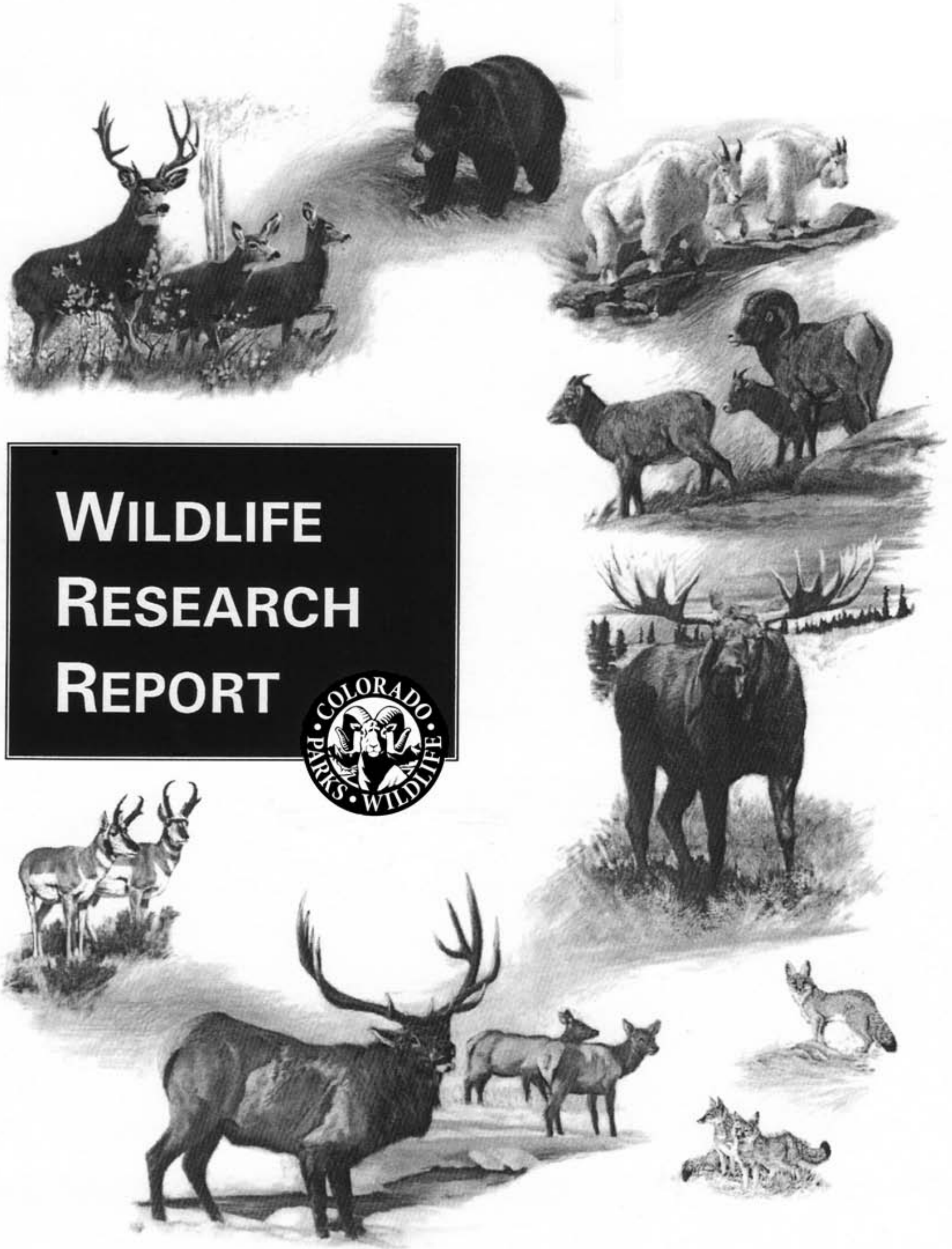
MAMMALS - JULY 2012



WILDLIFE RESEARCH REPORTS

JULY 2011 – JUNE 2012

MAMMALS PROGRAM

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

The Wildlife Reports contained herein represent preliminary analyses and are subject to change.
For this reason, information MAY NOT BE PUBLISHED OR QUOTED
without permission of the Author(s).

STATE OF COLORADO
John Hickenlooper, *Governor*

DEPARTMENT OF NATURAL RESOURCES
Mike King, *Executive Director*

PARKS AND WILDLIFE COMMISSION

Tim Glenn, *Chair*	Salida
Gary Butterworth, *Vice Chair*	Colorado Springs
Mark Smith, *Secretary*	Center
David Brougham	Lakewood
Chris Castilian	Denver
Dorothea Farris	Carbondale
Allan Jones	Meeker
Bill Kane	Basalt
Gaspar Perricone	Denver
James Pribyl	Boulder
John Singletary	Vineland
Robert Streeter	Fort Collins
Lenna Watson	Grand Junction
Dean Wingfield	Vernon
Mike King, Executive Director, *Ex-officio*	Denver
John Salazar, Dept. of Agriculture, *Ex-officio*	Lakewood

DIRECTOR'S LEADERSHIP TEAM

Rick Cables, Director
Ken Brink, Steve Cassin, Marilyn Gallegos Ramirez, Heather Dugan,
Gary Thorson, Jeff Ver Steeg, Craig McLaughlin (acting)
Ron Velarde, Steve Yamashita, Tom Speeze, John Geerdes
Kurt Mill, Dan Prenzlow

MAMMALS RESEARCH STAFF

Chad Bishop, Mammals Research Leader
Mat Alldredge, Wildlife Researcher
Chuck Anderson, Wildlife Researcher
Eric Bergman, Wildlife Researcher
Jake Ivan, Wildlife Researcher
Heather Johnson, Wildlife Researcher
Ken Logan, Wildlife Researcher
Kay Knudsen, Librarian
Margie Michaels, Program Assistant

Colorado Division of Parks and Wildlife
July 1, 2011 – June 30, 2012

TABLE OF CONTENTS
MAMMALS WILDLIFE RESEARCH REPORTS

LYNX / WOLVERINE CONSERVATION

DEER / ELK CONSERVATION

PREDATORY MAMMALS CONSERVATION

SUPPORT SERVICES

Colorado Division of Parks and Wildlife
July 2011–June 2012

WILDLIFE RESEARCH REPORT

State of:	Colorado	:	Division of Parks and Wildlife
Cost Center:	3430	:	Mammals Research
Work Package:	0638	:	Wolverine Conservation
Task No.:	N/A	:	Assessing the efficacy of monitoring wolverine on a regional scale using occupancy and abundance estimation
Federal Aid Project No.	N/A		

Period Covered: July 1, 2011 – June 30, 2012

Author: J. S. Ivan*****

Personnel: M. Schwartz, USFS Rocky Mountain Research Station; M. Ellis, University of Montana

***(**J. S. Ivan was the sole Colorado Parks and Wildlife contributor for this work and is thus listed as "author." However, the draft manuscript included here was a collaborative effort and all personnel listed are co-authors on the manuscript. M. Ellis is the first author.)

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

ABSTRACT

Conservation biologists and resource managers are often faced with the task of designing monitoring programs for species that are rare, diffuse, or patchily distributed across large landscapes. These efforts are frequently very expensive and seldom can be conducted by one entity. It is essential that a power analysis is undertaken to ensure stated goals are feasible. We developed a spatial-based simulation, which accounts for natural history, habitat use, and sampling scheme, to investigate power for monitoring wolverines in two areas of the U.S. Rocky Mountains. The first area is a well-established metapopulation of wolverine in the northern Rocky Mountain states of Montana, Idaho, and Wyoming, where the current population is approximately 350 individuals and there are concerns of population decline. Based on current population size estimates and detection probabilities in the northern U.S. Rockies, most sampling schemes are likely to only detect large declines in population sizes (i.e. 50% decline over 10 years). In general, increasing the number of grids sampled or the per visit detection probability had a much greater effect on power than increasing the number of visits per year. For small populations, we found very low power to detect declines. The second analysis was a forecast of the effort required to monitor an increasing population in the southern U.S. Rockies, given recolonization or reintroduction. Occupancy-based methods can only produce enough power to detect population trends if populations are increasing dramatically (i.e. doubling or tripling in 10 years), regardless of the sampling effort. In sum, our approach provides a spatially based framework to evaluate monitoring protocols and objectives by explicitly incorporating the link between changes in population size and estimated occupancy, all while accounting for natural history of the species in question. These analyses were specific to wolverines, but our approach could easily be adapted to other species.

WILDLIFE RESEARCH REPORT

ASSESSING THE EFFICACY OF MONITORING WOLVERINE ON A REGIONAL SCALE USING OCCUPANCY AND ABUNDANCE ESTIMATION.

JACOB S. IVAN

P. N. OBJECTIVE

Assess power for detecting trends in wolverine population growth using occupancy.

SEGMENT OBJECTIVES

1. Build code to simulate realistic distribution and space use of wolverine on the landscape.
2. Build code to realistically simulate sampling the wolverine population using an occupancy framework.
3. Build code to analyze data “collected” via occupancy surveys.
4. Summarize results of 1000s of iterations of randomly generated wolverine distributions and subsequent occupancy surveys; plot power to detect trends against various scenarios intended to reflect the range of conditions expected for both the sampling and process portions of the simulation.
5. Prepare manuscript for publication

INTRODUCTION

Wildlife populations worldwide have faced major population reductions in abundance and geographic range due to both natural and anthropogenic causes (Butchart et al. 2010, Hoffmann et al. 2010, Rands et al. 2010, Inman et al. 2011). Currently, many populations are facing multiple threats including habitat fragmentation and loss, climate change, direct and indirect exploitation, disease, invasive species, and the interaction among these threats (Primack 2006, Laurance et al. 2008, Povilitis and Suckling 2010). Responding to these major threats to wildlife and fish populations worldwide, many countries have adopted legislations aimed at affording protection to species of conservation concern (Hutchins et al. In Press, Waples et al. In Review). Two of the more powerful pieces of legislation are Canada’s Species at Risk Act (SARA) and the United States’ Endangered Species Act (ESA). These acts not only identify species at risk and aim to protect them from additional harm, but also stipulate and provide mechanisms for recovery. For example, in the United States approximately half of the annual budget spent on threatened and endangered species is designated for recovery (GAO 2005, Male and Bean 2005). However, determining when a species of concern is declining or subsequently recovering requires information about trend.

The majority of studies that have examined trends in fish and wildlife were historically based in assessments of population abundance (Dennis et al. 1991, Bart et al. 2007, Foster et al. 2009, Broms et al. 2010). While estimates of abundance are important, other measures such as changes in genetic or demographic parameters or changes in geographic range size have been used to infer trend (Gaston 1991, Schwartz et al. 2007, Marucco et al. 2009, Broms et al. 2010). Recently, more attention has been placed on estimating changes in occupancy of a species geographic range (Joseph et al. 2006, MacKenzie et al. 2006). Occupancy estimation generally requires multiple surveys to a set of sample units, noting on each survey whether the species of interest was detected or not. Subsequently, these repeat-visit data are used to estimate the probability of detecting the species of interest if it was present, and then adjusting the raw presence-absence data in light of this probability to estimate the proportion of area occupied (MacKenzie et al. 2006). If occupancy estimation is conducted over multiple time intervals, trend in occupancy is obtained (Field et al. 2005, MacKenzie 2005, Marsh and Trenham 2008).

Before launching an occupancy study, power analysis should be conducted to allocate monitoring effort efficiently (Field et al. 2005, MacKenzie 2005, Rhodes et al. 2006). Most studies base power analyses for occupancy estimation on detecting declines in occupancy over time; however, these simulations rarely consider spatial dynamics. Also, monitoring trends in occupancy is often used as a surrogate for trends in abundance, but this link is rarely evaluated (e.g. Field et al. 2005, Finley et al. 2005, Otto and Roloff 2011). Rhodes et al. (2006) and Rhodes and Jonzén (2011) modeled spatial and temporal correlations in population dynamics to account for spatial structure in populations and provide allocation recommendations in occupancy studies. They find, when spatial correlation is low and temporal correlation is high, it is most efficient to sample many sites infrequently. In the opposite situation, when spatial correlation among population dynamics is high and temporal correlation is low, they recommend sampling few sites often. Finally, when there is a decoupling of abundance and space, they suggest maximizing spatial replication (Rhodes and Jonzén 2011). Furthermore, if interest is in detecting declines in occupancy, they suggest sampling high quality habitats, whereas if the objective is to detect an increase, sampling intermediate-quality habitats is the best strategy. We extended their work by building a species-specific model of a population changing over time. We then sampled from this population using a multi-season occupancy framework to determine power to detect population trends under various scenarios. This approach allows us to optimally allocate scarce monitoring resources for designing an occupancy-based monitoring effort.

Our model was designed to optimize sampling allocation for a large-scale wolverine monitoring effort. Wolverines are a Holarctic carnivore species known for their large home ranges, low densities, and occasional long distance movements (Lofroth and Krebs 2007, Squires et al. 2007, Inman et al. 2012). The species is currently under consideration for listing under the ESA (USFWS 2010) largely due to the fact that their numbers were greatly reduced (possibly eliminated) in the contiguous United States in the early 20th century. Wolverine populations have recolonized Idaho, Montana, Washington, and Wyoming and single male wolverines have recently dispersed to California and Colorado (Aubry et al. 2007, Moriarty et al. 2009). Yet, they are still absent from significant portions of their historical range and their current abundance in the contiguous United States is still likely to be at most 500 individuals.

Recent research by Aubry et al. (2007) and Copeland et al. (2011) has shown that the historical distribution of wolverine was consistent with the distribution of persistent spring snow. Copeland et al. (2011) characterized persistent spring snow cover in the entire northern hemisphere based on a 21-day composite (24 April–15 May) of images from 2000-2006 at a 0.5km^2 resolution using moderate resolution imaging spectroradiometer (MODIS) satellite images (Hall et al. 2006). They found that >99% of wolverine den sites and >89% year-round telemetry locations were located within areas that were classified as having persistent spring snow in at least one of the seven years for which data were available. Schwartz et al. (2009) demonstrated that wolverine gene flow was facilitated by areas with persistent spring snow compared to areas that were snow free.

In this paper we use habitat (i.e., persistent spring snow), movement, and home range data to build a spatially based model for assessing the power for monitoring wolverine in their current range and in areas where they may eventually recolonize either naturally or through reintroduction.

METHODS

Study area

There are two study areas for this project. The primary study area consists of the U.S. Rocky Mountains in northern and central Idaho, western Montana, and northwest Wyoming (“Northern

Rockies", Figure 1). The area is composed of individual mountain ranges each characterized by high alpine areas (maximum elevation 3900 m) and surrounded by wide areas of semiarid grasslands and irrigated agriculture (elevation ~1400 m). This area is known to be occupied by wolverines, with current population estimates ranging from 200-500 individuals (USFWS 2010). We removed from our analyses mountain ranges on the edge of this range, including the Wallowa Mountains of Eastern Oregon, the Bighorn Mountains of Eastern Montana and Wyoming, and the Bear River Range on the Idaho/Utah border; all three of which have no historical records of wolverines (Aubry et al. 2007) and do not contain continuous patches of persistent spring snow cover (Schwartz et al. 2009, Copeland et al. 2011). We allowed areas 'used' by simulated wolverines to extend up to 50 km into Alberta and British Columbia, Canada to account for continuous wolverine populations in the Northern Rockies, but excluded these areas from occupancy analyses.

The second study area is the mountainous region of the Southern U.S. Rockies ("Southern Rockies"). This area is characterized by high, steep mountains (max elevation 4,400 m). As a result, there are strong gradients in physical attributes of the landscape, which lead to heavily dissected vegetation types. In the Southern Rockies, alpine and subalpine zones can be relatively narrow and give way to montane forests, montane shrublands, and semiarid grassland or sagebrush communities over relatively short distances. This area does not currently have a population of wolverines, although wolverines are thought to have occurred there historically (Aubry et al. 2007), and there seems to be adequate habitat, including persistent spring snow (Aubry et al. 2007, McKelvey et al. 2011). Areas of persistent spring snow are more patchily distributed in the Southern Rockies landscape, and separated from areas of persistent spring snow in the Northern U.S. Rockies by >200km. Most mountain ranges in this study area occur in Colorado, but we included the Medicine Bow and Sierra Madre ranges in southern Wyoming, as well as the southern San Juan Mountains in northern New Mexico.

Individual Utilization Distributions

We randomly selected points within areas of persistent spring snow (using Copeland et al. 2010) for the center of individual home ranges for adult female, adult male, and transient male wolverines. Among these three groups, locations were chosen independently to allow for overlapping home ranges (Copeland 1996, Inman et al. 2011); however, within each group, selection of home range centers was constrained to reflect territoriality. The buffer distances required between home ranges centers were at least 16 km for adult females, reflecting a 225 km^2 home range, and at least 25.2 km for adult and transient males, reflecting 500 km^2 home ranges (Banci 1994, Krebs et al. 2007, Schwartz et al. 2009). We also required that all home range centers were located in snow patches large enough to support at least one resident female wolverine (Krebs et al. 2007). Within each group (adult females, adult/resident males, adult/transient males), locations for home range centers were randomly selected in an iterative fashion until no additional individuals could be placed in the landscape or until the desired number of individuals was met.

Once home range centers were established for a given simulated landscape, we assigned a bivariate normal utilization distribution for each individual. For resident females, we assumed that an individual spends 90% of her time within their 225 km^2 home range (radius = 8.5 km). For resident males, we assumed individuals spend 90% of their time within their 12.6 km home range radius, but we allowed for larger sizes and greater overlap among transient male home ranges by assuming individuals only spend 70% of their time in the original 500 km^2 home range. Each of these distributions produced a surface with decreasing probability of use with increasing distance from the home range center. To make these bivariate normal utilization distributions more realistic, we overlaid them on the persistent spring snow layer and multiplied the layers together. In the persistent spring snow layer, areas of non-snow were weighted as having 1/20 the probability of use compared to snow areas, based on resistance values found for models of genetic least cost paths (Schwartz et al. 2009). We standardized the product of the two layers to transform it back into a probability density. Thus, each individual utilization distribution takes a unique shape based on availability of snow.

In this approach, it is possible for individuals to make short term, long distance movements during a given study period. The tails of the bivariate normal utilization distribution allow for a very small, but non-zero, probability of reaching any point on the landscape. In preliminary analyses, we tested for the effect of excluding these long distance movement events by cutting off the tails of the bivariate normal, such that the probability of an individual being more than 1-2 standard deviations away from its home range center was set to 0, compared to a situation with no limit on movement. Although allowing short-term, long-distance movements did affect the estimated occupancy of the landscape, the effect on power was minor. Occasional long-distance movements are possible in wolverine ecology, especially by males and transients (Moriarty et al. 2009). For territorial males and females, we would expect these movements to be less likely over the course of the relatively short survey period. Thus we based our power analyses on a 'mixed' scenario in which long distance movements were possible for transient males (i.e. no limit), resident males were allowed some larger movement events (limited to within 2 s.d. of home range center), and movements of females, which may have dens, were limited 1 sd from their home range center.

Following the rules state above, our program, SPACE (Spatially-based Power Analyses for Conservation and Ecology), created 1000 surfaces for N=500 or N=200 individuals on the Northern Rockies landscape, reflecting high and low estimates of wolverine population size in the study area. We then simulated 10%, 20%, or 50% declines in our simulated populations over a decade (λ = 0.989, 0.977, 0.933) by randomly removing an appropriate number of individuals at each time step. We also simulated scenarios (n_{sim} = 1000) for a hypothetical reintroduced or recolonizing population in the Southern Rockies. These populations were started with N=30 individuals then allowed to increase by 50%, 100%, or 200% over a decade (λ = 1.041, 1.072, 1.0116). We initiated all populations with a 2:1:2 ratio of females:resident males:transient males.

Sampling

To estimate occupancy, we sampled from our simulated landscapes during each time step or "year" of the simulation. We divided the study area into 225km^2 sample units (cells), matching home range sizes for resident females, a strategy widely used for monitoring carnivores (e.g., Zielinski and Stauffer 1996). We excluded cells that did not overlap the persistent snow layer by ≥50%. This resulted in 388 cells for the main Northern Rockies study region, and 128 cells for the Southern Rockies. For each cell, the probability of at least one wolverine being present (hereafter, 'probability of presence' was Eqn 3):

$$\mathrm{P}(\#\,wolverines \geq 1)_j = 1 - \mathrm{P}(wolverines\ absent)_j = 1 - \prod_{i=1}^{N}\left(1 - \iint_{(x,y)\in\Omega_j} f_i(x,y)\,dx\,dy\right)$$

where N is the number of wolverines in the simulated study area, $f_i(x,y)$ is the probability density function (i.e., utilization distribution) describing the use surface for the i^{th} wolverine, and Ω_j represents the area included in the j^{th} grid. We approximated integral values by summing pixel values in the raster, assuming equal pixel areas.

To construct a simulated encounter history (i.e., the data necessary for occupancy estimation) for cell *j* in year *k*, we assigned a 1 (present) or 0 (absent) for each visit by comparing a random draw from Uniform (0,1) with the probability of presence for that cell (draws less than the probability of presence resulted in a detection, and a 1 in the encounter history for that visit). Thus, a cell with simulated encounter history "010" indicates that 3 visits were made to the cell in a given year, and wolverines were detected on the second visit only. After initial construction, we used progressively reduced versions of the encounter histories to explore the effect of changes in parameters associated with sampling on power to detect population changes. For example, we omitted data from even numbered years (i.e., inserted "." for each "0" or "1" of the omitted years) to examine the effect of sampling every other year; we tested the effects of smaller sample sizes by reducing the number of cells or visits included in the encounter histories; and we reduced the number of detections to simulate imperfect detection (See Table 1). To create encounter histories with lower detection probability, we randomly removed an appropriate proportion of 1s from each encounter history. Thus, to go from a detection probability of 1.0 to 0.8, we retained 0.8/1.0 = 80% of the 1s; for each 1 (wolverine detected) in a given encounter history, we conducted a random draw from uniform (0,1) and compared this draw against 0.8. We retained the 1 if the draw was ≤ 0.8, and changed it to a 0 (wolverine not detected) otherwise. Similarly, to go from encounter histories reflecting detection probability = 0.8 to detection probability = 0.2, we evaluated each 1 in a given history, retaining it if the random draw was ≤ 0.25 (0.2/0.8), changing it to 0 otherwise.

We used these encounter histories to obtain annual estimates of occupancy and detection probability for each simulated landscape and parameter set. Note that the subject of our simulations is a mobile carnivore capable of moving freely between sample cells, and our simulation setup reflected this reality. Therefore, interpretation of estimated occupancy parameters was different than the usual context in which the status (occupied or not) of a given cell is assumed static over the course of a survey. Specifically, the estimate of occupancy (Ψ) generated under this context is the probability that any given cell is *used* rather than *occupied*, and any reference to Ψ or "occupancy" from here forward refers to probability of use. Furthermore, the estimate of detection probability generated in this context is actually the product of true detection probability (i.e., probability of detection given that the species of interest is present; this quantity is specified directly for any given simulation) and a landscape-wide probability that an individual is present and available for detection (i.e., probability of presence; see above). We refer to the detection probability estimated by the model as p_{est}, and the actual detection probability specified for the simulations as p_{sim}, such that $p_{est} = p_{sim}$ × probability of presence.

We used the R (R Development Core Team 2011) package RMark to input the encounter histories and construct models to fit in Program MARK (White and Burnham 1999). Specifically, we employed the 'Robust Design Occupancy' data type (MacKenzie et al. 2006) in which colonization (γ) could vary through time but was constrained to be the complement of extinction (ε; i.e., changes in occupancy were considered random rather than Markovian or static) and detection probability (p) varied with time. This model structure is appropriate because: 1) we were interested primarily in the occupancy estimates themselves; we had no interest in modeling occupancy dynamics (colonization, extinction) explicitly, 2) the simulation specifications allowed "movement" in and out of adjacent cells, thus mimicking random changes in occupancy, and 3) "movement" between adjacent cells forced p_{est} to be a function of probability of presence, which changed through time depending on the simulated landscape and birth/death of individuals. Thus, p_{est} should have varied through time as well.
We extracted the 10 occupancy estimates and the variance-covariance matrix for these estimates from each simulation, then used the variance components procedure in RMark to fit a linear random effects trend model to the estimates. A trend was 'detected' if the 95% confidence interval of the trend parameter (on the logit scale) from the random effects model excluded zero and was in the correct direction (e.g., <0 for declining trends; Tallmon et al. 2010). Thus, we computed the statistical power produced by a sampling scenario, i.e. the probability that we detect a significant trend given that there is a trend in the underlying data, as the percentage of simulations in which a trend was detected.

For datasets in which we simulated sampling every other year, we fit models in which we fixed $\gamma^1=\gamma^2$, $\gamma^3=\gamma^4$, etc. such that the product of these parameters were estimated, and we bridged years in which no data were collected to produce valid estimates of occupancy for those years where data were collected.

In these scenarios, only 5 occupancy estimates were generated, and we fit random effects models to those 5 estimates.

We repeated these analyses for each combination of population growth or decline, simulated detection probability (p_{sim}), number of visits, cell size, number of cells sampled, and annual or alternating year sampling schemes that were applied to the 1000 simulated landscapes of N = 30, 200, or 500 (Table 1). Where applicable, all sampling was cumulative to facilitate the most meaningful contrasts between levels of a parameter; for example, a sample of n = 50 cells would include the same cells as an n = 25 sample with 25 additional cells included. Similarly, an encounter history with 4 visits would include the same string of 0s and 1s as a 3-visit history, with one additional visit included. Our simulations were designed to be generalizations in that we do not attempt to define when a sampling season might begin, what the sampling mechanism was, or what constitutes a visit. Thus, these simulations could represent flying over selected cells in the study area to search for tracks in the snow, in which case a 'visit' is a single flight (Gardner et al. 2010), or they could reflect the use of hair snag devices in which a 'visit' is 1 month of continuous sampling (Magoun et al. 2011). We bracketed the sampling parameters (cell size, detection probability, visits) based on previous efforts described in the literature (Magoun et al. 2007, Gardner et al. 2010, Magoun et al. 2011).

RESULTS

Effects of home range parameters

Due to the spacing rules among individuals that we used to reflect wolverine territoriality, the Northern Rockies landscape becomes saturated with approximately 850 individuals (420 ± 6 females, 219 ± 4 resident males, 219 ± 4 transient males; mean ± s.d. across 100 simulated landscapes). For N=800, the median probability of at least one wolverine per cell across the landscape was 0.47. This value reflects the availability of individuals on the landscape, yielding on average 280.4 cells in which wolverine were available for detection per sampling occasion across the 388 cells in the grid. As the population size decreased, the average probability of at least one wolverine per cell fell to 0.74 (212.4 detections per occasion) for N=500 and 0.05 (18.9 detections per occasion) for N=30 across simulations. With perfect detection associated with sampling ($p_{sim} = 1$), these cell-based probabilities for use translate to an estimated occupancy (Ψ) of 0.99±0.01 for the entire landscape for populations with N = 500 individuals and 0.06±0.01 for N = 30.

Effects of population size and trend

We investigated the upper limits of power with occupancy estimation by examining the ability to detect trends when the simulated detection probability was perfect ($p_{sim} = 1$) and with a large number of visits (5) to each unit. We focused these analyses on the U.S. Northern Rockies landscape and a quickly declining population ($\lambda = 0.933$). Even with perfect detection and intense sampling, detecting a large decline (50% over 10 years) in a large starting population (N = 500) with adequate power (>80% chance of detecting the trend) required a sample of 50 out of 388 cells (Figure 2). As the population size decreased, the amount of sampling needed to detect a 50% decline even under this 'best case' scenario with perfect detection increased dramatically. For example, when N=200, achieving 80% power required sampling approximately 75 to 150 cells. Detecting trends in small populations (N=30) was difficult; even if we included the entire grid (388 cells) in the sample and assumed perfect detection, we had less than 70% power to detect a trend.

Regardless of the starting sample size, power to detect trends was lower for increasing populations compared to the decreasing scenarios described above. For example, to detect a 50% increase ($\lambda = 1.041$) with >80% confidence, the amount of the total sampling grid that would need to be included in the sample increased to ~25% of the grid ($n_{cells} \approx 60$) for N=500 or ~ 50% ($n_{cells} \approx 125$) for

N=200. For N=30, sampling the entire grid, assuming perfect detection probability, and with an intense sampling effort (5 visits), we were able to detect a 50% increase in <40% of the simulations.

With current population sizes (N=500) in the Northern Rockies, the ability to detect declines fell dramatically as the strength of the decline decreased (Figure 3). We found a reasonable chance (≥80%) of detecting a 50% decline in population size over a 10-year period, depending on the combination of sample size and detection probability. However, for a 10% decline in population size over the 10 year period, no amount of sampling could yield enough power to detect the trend. Similarly, even with a large sample size and high detection probability, a 20% decline was detected in <60% of the simulations (Figure 3). With either population increases or declines, sampling every other year substantially increased the number of cells and visits that would need to be sampled.

Trade-offs in sampling methodology

After the strength of the population decline or increase, the parameter that most influences power to detect change was the simulation detection probability (p_{sim}). In nearly all scenarios relatively large gains in power were realized when p_{sim} increased from 0.2 to 0.8. For instance, a monitoring scheme that called for 2 visits to each of 100 sample units would have ~25% chance of detecting a 50% decline over 10 years when $p_{sim} = 0.2$. Power for detecting that same decline under the same sampling regime increased to 80% when $p_{sim} = 0.8$ (Figure 3, upper left panel). By comparison, an increase in sample size from $n_{cells} = 50$ to $n_{cells} = 300$ resulted in only a doubling in power (25% to ~50%). In fact, when $p_{sim} = 0.2$, 80% power cannot be achieved even if the entire grid is sampled. Similar gains in power relative to simulation detection probability and sample were realized in other scenarios we simulated. The exceptions to this result were when the goal was to detect a 10% decline over 10 years or to detect a 20% decline when sampling was only conducted every other year. Both scenarios yield very low power and negligible improvement with increased p_{sim} or sample size (Figure 3, middle panels).

The number of visits to each sample unit influenced power as well, although generally to a lesser degree than magnitude of population change, simulation detection probability, and sample size. Even with perfect simulation detection probability ($p_{sim} = 1$), the power to detect a trend increased with the number of visits at each grid cell due to the number of opportunities for an individual to be present. When simulation detection probability is high but imperfect (i.e., $p_{sim} = 0.8$), some gain in power could be realized by visiting each sampled cell 3 times vs. visiting them only twice (Figure 3, separation between the two lightest dotted lines). However, the gain realized for making 4 visits rather than 3 is small, and there is no appreciable difference in power for 4, 5, 6, or 7 visits under the scenarios we simulated. When simulated detection probability was low (i.e., $p_{sim} = 0.2$), potentially greater gains in power could be realized by making more visits, but it depends on the scenario (Figure 3, in some cases there is a moderate amount of separation in the solid lines, in other cases there is not). Note that at low detection probabilities ($p_{sim} = 0.2$), it is often inadvisable to make more visits to each sampled cell because such an approach actually decreases power (See discussion).

Effect of Cell Size

In order to achieve the threshold of 80% power to detect a 50% population decline, changing cell sizes in the grid had implications for both the number of cells and the total area that would need to be sampled. (Figure 5). Grids of 100km^2 and 225km^2 cells yielded similar power in terms of the percent of the grid that would need to be included in the sample, although the smaller cell size requires sampling more cells (i.e., the total grids were comprised of 887 100km^2 cells versus 388 225km^2 cells). Assuming 3 visits and high detection, getting 80% power for detecting a 50% decline required 120 cells (12,000km^2) from the small grid versus 70 cells (15,750km^2) for the medium sized grid. As the size of the grid increased, the power to detect trends in occupancy decreased. The 1000km^2 grid produced very low power to detect population trends. In this case, the grid in the Northern Rockies comprised only 76 cells. Including every cell in the sample, with seven visits and high detection probability, we detected a 50%

population decline in <20% of the runs. The phenomenon in which power is actually reduced with a high number of visits occurs for the 225km^2 cell size at low p_{sim}, and for the 500km^2 and 1000km^2 size at high p_{sim}.

Power to detect increases in small populations

For small populations (N=30), power for detecting population trends was limited except for situations with large population increases and high detection probability (Figure 4). For the purposes of comparison, there was greater power for detecting trends in the Southern Rockies landscape than in the Northern Rockies, although the total sampling area in the Southern Rockies landscape is approximately only a third of the Northern Rockies. For both landscapes, a doubling of the population over ten years ($\lambda = 1.072$) could be detected with >80% power in scenarios where a large proportion of the landscape was included with relatively high capture probability. If simulation detection probability is low, then adequate power can only be achieved via sampling a large portion of the available landscape, and making a large number (≥5) of visits to each sampled cell.

DISCUSSION

Monitoring population trends over time is one of the most common goals for management of endangered species. Using a spatially explicit simulation for wolverine in the U.S. Rocky Mountains, we were able to test the ability of occupancy-based approaches to detect trends in population size under a range of monitoring scenarios. Even for large changes in population size (e.g. 50% declines over 10 years), we found that detecting population trends required large-scale, intensive sampling. In many scenarios, no amount of sampling could produce sufficient power to achieve monitoring goals. Our results highlight the importance of analyzing the statistical power of monitoring schemes and using approaches that incorporate the effect of sampling and power over the course of multiple steps in a monitoring protocol.

In the case of the wolverine, work has commenced to evaluate the effectiveness of various approaches for detecting presence. These range from using fix-winged aircraft to find tracks in 100-km^2 (Magoun et al. 2007) or 1000-km^2 (Gardner et al. 2010) sampling cells, to using cameras at bait stations (Mulders et al. 2007, Magoun et al. 2011), to the use of non-invasive genetic sampling (Ulizio et al. 2006, Schwartz and Monfort 2008, Magoun et al. 2011). These efforts produce varying detection probabilities from 0.2 to 0.8 as bracketed in our simulations.

However, matching estimates from field studies to our results, is not straightforward. It is important to note that detection probability estimated from pilot analysesesis not the same as the p_{sim} input in our analyses. Due to the 'mobile animal' phenomenon, animals are capable of moving freely between sample cells and therefore can be detected in multiple cells during one sampling occasion. As a result, occupancy models cannot separate the effects of true detection probability (p_{sim}) and probability of presence (See Methods). Consequently, p_{est} returned from pilot studies will be smaller than the detection probabilities used in our simulations (p_{sim}). For example, if pilot work indicates that $p_{est} = 0.2$, power can be assumed to be slightly better than the curves shown for $p_{sim} = 0.2$ in our figures. The exact correspondence between p_{est} and p_{sim} is dependent on cell size, population size, and home range size of the species in question. Thus, no rule of thumb holds for converting between the two. However, matching p_{est} derived from pilot work to curves for p_{sim} can still be useful as it will result in conservative estimates of power, which would be a prudent way to design monitoring schemes.

In the case of wolverines, pilot work specific to occupancy monitoring in the Northern Rockies has been carried out using camera stations (B. Inman, Wildlife Conservation Society, unpublished data) and hair snags (J. Waller, Glacier National Park, unpublished data) in 100-km^2 sample units. Initial results from this work suggest p_{est} is approximately 0.25 – 0.3, which in our simulations corresponded to

$p_{sim} \approx 0.8$ (i.e., $p_{est} = p_{sim} \times$ probability of presence, where our mean probability of presence was 0.33; thus $0.25/0.33 \approx 0.8$). It's important to note that the mean probability of presence depends on assumptions about the number of animals, the landscape, and home range configurations. Based on this estimate, and assuming 3-4 visits to each sample unit (sampling occurred during 3-4 months over winter for each pilot study), our research suggests that roughly 100-150 100-km^2 cells would need to be sampled per year to attain an 80% probability of detecting a 50% decline in the Northern Rockies population (Figure 5). Thus, intensive sampling over a small area is unlikely to be a viable solution for detecting population trends. To accomplish anything meaningful, monitoring will require well-coordinated surveys across multiple entities and jurisdictions. Anything less than a large-scale, coordinated effort will likely be of limited or no value.

The spatially explicit nature of our approach is especially important in linking changes in occupancy to population trends. Our results demonstrate that the underlying landscape can influence power to detect population changes. Specifically, in the comparison of power for populations with N=30 in the Northern versus Southern Rockies, power to detect trends in occupancy was similar in terms of percent of the total study area included in the sample, but very different in terms of the absolute area that needs to be sampled. For example, to detect a 3x increase of the N=30 populations with a 225km^2 grid and >80% power required sampling ~20% of either landscape, which translates to sampling 16000km^2 in the northern landscape versus 6000km^2 in the south. Note, however, that the scenarios in this comparison, populations of N=30 in the Northern versus Southern U.S. Rockies, are intended to illustrate the effect of underlying landscape for a fixed population size. In reality, changing the size of a study area would generally also change the size of the population included, which we found to substantially affect power to detect trends.

Previous recommendations for selecting cell sizes have been *ad hoc*. In some cases, our results indicate a relatively straightforward relationship between cell size and the number of cells needed or the total area sampled to achieve a given power threshold. Between a 100km^2 grid and a 225km^2 grid, with high detection probability, 80% power can be obtained either by sampling many small cells or fewer of the larger cells. However, by the time cell sizes reach 1000km^2 for wolverine, the home ranges for multiple individuals are included in the cell, such that occupancy-based methods alone will only pick up changes once a much larger population decline has occurred. The point at which this switch occurs will likely depend on an interaction of the population size, landscape, home range sizes, and cell size.

We also discovered a counterintuitive anomaly when computing power under scenarios in which cell size is equal to home range size, as is often advised for occupancy surveys of mobile carnivores. Specifically, we noted that when detection probability is low, power generally increases with increasing visits to each sample unit, but there is a point at which conducting more visits actually decreases power. We offer the following explanation for this phenomenon: When the cell size is equal to the home range size, the interplay between p_{sim} (i.e., 0.2) and availability is such that the p_{est} is fairly low and makes a substantial upward adjustment on the count of cells (c) in which wolverines were actually detected. As we make more visits we detect wolverine use in cells that are seldom used, so c increases, but p_{est} from the model does not (only the precision on p_{est} improves). After about 6 or 7 visits c increases enough that the occupancy estimates resulting from upward adjustments on c approach 1.0. If estimates for all years are at or near 1.0, then there is no trend and we have no power to detect declines. This does not occur when cell sizes are small, because c will also be small, and any upward adjustments will not approach 1.0. A similar phenomenon occurs if cells are large and p_{sim} is high. In that case, most cells are used, and c will be large, especially with a large number of visits. Thus, even a small upward adjustment on c pushes the estimates to close to 1.0, which again makes detecting trends difficult. Thus, if maximizing power is a goal, increasing visits beyond a certain threshold may not be helpful depending on cell size, availability of animals, and the probability of detecting them given their presence.

Our simulations currently do not include cost functions, so trade-offs between cell size, number of cells to sample, number of visits at each cell, and detection probability have been conducted absent an important real-world consideration. For instance, in a given situation, it may be easy to complete more visits to a site (e.g., leave camera sets out 1 more month), but extremely costly to improve capture probability (e.g, purchase an entire set of new cameras with improved functionality). Therefore, managers may opt to make more visits to improve power even though intensifying effort (visits) by a given percent may be inferior to improving detection probability by a similar percentage. Future simulation work should include cost as a factor in weighing the importance of the design factors we considered here.

Most studies base power analyses for occupancy estimation solely on detecting various simulated declines in occupancy. Here, we employed a more mechanistic, spatially-based approach in which we simulated animals on a landscape, accounted for their natural history (territoriality, difference between sexes), tied their space use to key habitat features (persistent spring snow), and forced declines or increases in the real parameter of interest (abundance) to determine whether occupancy estimation could detect those changes. Thus, our approach is a direct test of the link between occupancy and abundance, providing a more meaningful examination of whether real-world changes of interest in population size can actually be detected using occupancy estimation. It also sets the stage for direct comparisons between occupancy and estimation of other metrics (e.g., abundance) that could potentially be used to monitor populations. That is, we have established the machinery necessary to simulate ‘truth’ (the configuration of animals on the landscape and changes in that configuration and/or number) and can then sample from that true population in various ways to simulate data gathering under different monitoring approaches. While results from this analysis can be used directly to guide the monitoring of wolverine or similar species, the largest contribution is the framework which can be used for making decisions about the design of a large scale monitoring effort provided information on movement and habitat use is available. Our goals were to establish this framework to encourage cost-effective decisions in designing monitoring programs and to inspire well-coordinated surveys across multiple entities and jurisdictions. Without such coordination our analyses convincingly show that most efforts for species like wolverine will be wasted.

ACKNOWLEDGMENTS

We thank Paul Lukacs, Gary White, and Larissa Bailey for providing invaluable technical advice, and Jeff Laake for implementing the “random occupancy dynamics” model into RMark so it could be used in this analysis. We thank the RMRS and a PECASE award to MKS for providing the initial funding for this effort.

LITERATURE CITED

Aubry, K. B., K. S. McKelvey, and J. P. Copeland. 2007. Distribution and broadscale habitat relations of the wolverine in the contiguous United States. Journal of Wildlife Management 71:2147–2158.

Banci, V. 1994. Wolverine. Pages 99–122 *in* L. F. Ruggiero, K. B. Aubry, S. W. Buskirk, L. J. Lyone, andW. J. Zielinski, editors. The scientific basis for conserving forest carnivores in the western United States. U.S. Forest Service, General Technical Report, RM-254.

Bart, J., S. Brown, B. Harrington, and R. I. G. Morrison. 2007. Survey trends of North American shorebirds: Population declines or shifting distributions? . Journal of Avian Biology 38:73–82.

Broms, K., J. R. Skalski, J. J. Millspaugh, C. A. Hagen, and J. H. Schulz. 2010. Using Statistical Population Reconstruction to Estimate Demographic Trends in Small Game Populations. The Journal of Wildlife Management 74:310–317.

Butchart, S. H. M., M. Walpole, B. Collen, A. van Strien, J. P. W. Scharlemann, R. E. A. Almond, J. E. M. Baillie, B. Bomhard, C. Brown, J. Bruno, K. E. Carpenter, G. M. Carr, J. Chanson, A. M. Chenery, J. Csirke, N. C. Davidson, F. Dentener, M. Foster, A. Galli, J. N. Galloway, P.

Genovesi, R. D. Gregory, M. Hockings, V. Kapos, J.-F. Lamarque, F. Leverington, J. Loh, M. A. McGeoch, L. McRae, A. Minasyan, M. H. Morcillo, T. E. E. Oldfield, D. Pauly, S. Quader, C. Revenga, J. R. Sauer, B. Skolnik, D. Spear, D. Stanwell-Smith, S. N. Stuart, A. Symes, M. Tierney, T. D. Tyrrell, J. Vié, and R. Watson. 2010. Global Biodiversity: Indicators of Recent Declines. Science 328:1164–1168.

Copeland, J. P. 1996. Biology of wolverine in Central Idaho. University of Idaho, Moscow, ID.

Copeland, J. P., K. S. McKelvey, K. B. Aubry, A. Landa, J. Persson, R. M. Inman, J. Krebs, E. Lofroth, H. Golden, J. R. Squires, A. Magoun, M. K. Schwartz, J. Wilmot, C. L. Copeland, R. E. Yates, I. Kojola, and R. May. 2010. The bioclimatic envelope of the wolverine (Gulo gulo): do climatic constraints limit its geographic distribution? Canadian Journal of Zoology-Revue Canadienne De Zoologie 88:233–246.

_____. 2011. The bioclimatic envelope of the wolverine (Gulo gulo): do climatic constraints limit its geographic distribution? Canadian Journal of Zoology-Revue Canadienne De Zoologie 88:233–246.

Dennis, B., P. L. Munholland, and J. M. Scott. 1991. Estimation of growth and extinction parameters for endangered species. Ecological Monographs 61:115–143.

Devineau, O., T. M. Shenk, G. C. White, P. F. Doherty, P. M. Lukacs, and R. H. Kahn. 2010. Evaluating the Canada lynx reintroduction programme in Colorado: patterns in mortality. Journal of Applied Ecology 47:524–531.

Field, S. A., A. J. Tyre, and H. P. Possingham. 2005. Optimizing allocation of monitoring effort under economic and observational constraints. The Journal of Wildlife Management 69:473–482.

Finley, D. J., G. C. White, J. P. Fitzgerald, and Vojta. 2005. Estimation of swift fox population size and occupancy rates in eastern Colorado. Journal of Wildlife Management 69:861–873.

Foster, C. R., A. F. Amos, and L. A. Fuiman. 2009. Trends in Abundance of Coastal Birds and Human Activity on a Texas Barrier Island Over Three Decades. Estuaries and Coasts 32:1079–1089.

GAO. 2005. Fish and Wildlife Service Generally Focuses Recovery Funding on High Priority Species, but Needs to Periodically Assess its Funding Decisions. US Government Accountability Office, Washington, DC, US-GAO Report No. GAO-05-211. GAO-05-211.

Gardner, C. L., J. P. Lawler, J. M. V. Hoef, A. J. Magoun, and K. A. Kellie. 2010. Coarse-Scale Distribution Surveys and Occurrence Probability Modeling for Wolverine in Interior Alaska. Journal of Wildlife Management 74:1894–1903.

Gaston, K. J. 1991. How large is a species' geographic range? Oikos 61:434–438.

Getz, W. M., S. Fortmann-Roe, P. C. Cross, A. J. Lyons, S. J. Ryan, and C. C. Wilmers. 2007. LoCoH: Nonparameteric Kernel Methods for Constructing Home Ranges and Utilization Distributions. Plos One 2.

Getz, W. M., and C. C. Wilmers. 2004. A local nearest-neighbor convex-hull construction of home ranges and utilization distributions. Ecography 27:489–505.

Hall, D. K., G. A. Riggs, and V. V. Salomonson. 2006. MODIS/Terra snow cover daily L3 global 500m grid. Version 4, 24 April – 21 May 2000–2006. *in* National Snow and Ice Data Center, Boulder, CO.

Hoffmann, M., C. Hilton-Taylor, A. Angulo, M. Böhm, T. M. Brooks, S. H. M. Butchart, K. E. Carpenter, J. Chanson, B. Collen, N. A. Cox, W. R. T. Darwall, N. K. Dulvy, L. R. Harrison, V. Katariya, C. M. Pollock, S. Quader, N. I. Richman, A. S. L. Rodrigues, M. F. Tognelli, J.-C. Vié, J. M. Aguiar, D. J. Allen, G. R. Allen, G. Amori, N. B. Ananjeva, F. Andreone, P. Andrew, A. L. A. Ortiz, J. E. M. Baillie, R. Baldi, B. D. Bell, S. D. Biju, J. P. Bird, P. Black-Decima, J. J. Blanc, F. Bolaños, W. Bolivar-G, I. J. Burfield, J. A. Burton, D. R. Capper, F. Castro, G. Catullo, R. D. Cavanagh, A. Channing, N. L. Chao, A. M. Chenery, F. Chiozza, V. Clausnitzer, N. J. Collar, L. C. Collett, B. B. Collette, C. F. C. Fernandez, M. T. Craig, M. J. Crosby, N. Cumberlidge, A. Cuttelod, A. E. Derocher, A. C. Diesmos, J. S. Donaldson, J. W. Duckworth, G. Dutson, S. K. Dutta, R. H. Emslie, A. Farjon, S. Fowler, J. Freyhof, D. L. Garshelis, J. Gerlach, D. J. Gower, T. D. Grant, G. A. Hammerson, R. B. Harris, L. R. Heaney, S. B. Hedges, J.-M. Hero, B. Hughes, S.

A. Hussain, J. Icochea M, R. F. Inger, N. Ishii, D. T. Iskandar, R. K. B. Jenkins, Y. Kaneko, M. Kottelat, K. M. Kovacs, S. L. Kuzmin, E. La Marca, J. F. Lamoreux, M. W. N. Lau, E. O. Lavilla, K. Leus, R. L. Lewison, G. Lichtenstein, S. R. Livingstone, V. Lukoschek, A. M. Alonso, D. P. Mallon, P. J. K. McGowan, A. McIvor, P. D. Moehlman, S. Molur, A. M. Alonso, J. A. Musick, K. Nowell, R. A. Nussbaum, W. Olech, N. L. Orlov, T. J. Papenfuss, G. Parra-Olea, W. F. Perrin, B. A. Polidoro, M. Pourkazemi, P. A. Racey, J. S. Ragle, M. Ram, G. Rathbun, R. P. Reynolds, A. G. J. Rhodin, S. J. Richards, L. O. Rodríguez, S. R. Ron, C. Rondinini, A. B. Rylands, Y. Sadovy de Mitcheson, J. C. Sanciangco, K. L. Sanders, G. Santos-Barrera, J. Schipper, C. Self-Sullivan, Y. Shi, A. Shoemaker, F. T. Short, C. Sillero-Zubiri, D. L. Silvano, K. G. Smith, A. T. Smith, J. Snoeks, A. J. Stattersfield, A. J. Symes, A. B. Taber, B. K. Talukdar, H. J. Temple, R. Timmins, J. A. Tobias, K. Tsytsulina, D. Tweddle, C. Ubeda, S. V. Valenti, P. Paul van Dijk, L. M. Veiga, A. Veloso, D. C. Wege, M. Wilkinson, E. A. Williamson, F. Xie, B. E. Young, H. R. Akçakaya, L. Bennun, T. M. Blackburn, L. Boitani, H. T. Dublin, G. A. B. da Fonseca, C. Gascon, T. E. Lacher, G. M. Mace, S. A. Mainka, J. A. McNeely, R. A. Mittermeier, G. M. Reid, J. P. Rodriguez, A. A. Rosenberg, M. J. Samways, J. Smart, B. A. Stein, and S. N. Stuart. 2010. The Impact of Conservation on the Status of the World's Vertebrates. Science 330:1503–1509.

Inman, R. M., M. L. Packila, K. H. Inman, A. J. McCue, G. C. White, J. Persson, B. C. Aber, M. L. Orme, K. L. Alt, S. L. Cain, J. A. Fredrick, B. J. Oakleaf, and S. S. Sartorius. 2011. Spatial ecology of wolverines at the southern periphery of distribution. The Journal of Wildlife Management 76:778–792.

Inman, R. M., M. L. Packila, K. H. Inman, A. J. Mccue, G. C. White, J. Persson, B. C. Aber, M. L. Orme, K. L. Alt, S. L. Cain, J. A. Fredrick, B. J. Oakleaf, and S. S. Sartorius. 2012. Spatial ecology of wolverines at the southern periphery of distribution. The Journal of Wildlife Management 76:778–792.

Joseph, L. N., S. A. Field, C. Wilcox, and H. P. Possingham. 2006. Presence-absence versus abundance data for monitoring threatened species. Conservation Biology 20:1679–1687.

Krebs, J., E. C. Lofroth, and I. Parfitt. 2007. Multiscale habitat use by wolverines in British Columbia, Canada. Journal of Wildlife Management 71:2180–2192.

Laurance, W. F., S. G. Laurance, and D. W. Hilbert. 2008. Long-term dynamics of a fragmented rainforest mammal assemblage. Conservation Biology 22:1154–1164.

Lofroth, E. C., and J. Krebs. 2007. The abundance and distribution of wolverines in British Columbia, Canada. Journal of Wildlife Management 71:2159–2169.

MacKenzie, D. I. 2005. What Are the Issues with Presence-Absence Data for Wildlife Managers? The Journal of Wildlife Management 69:849–860.

MacKenzie, D. I., J. D. Nichols, J. A. Royle, K. H. Pollock, L. L. Bailey, and J. E. Hines. 2006. Occupancy Estimation and Modeling: Inferring Patterns and Dynamics of Species Occurrence. Elsevier, Amsterdam.

Magoun, A. J., C. D. Long, M. K. Schwartz, K. L. Pilgrim, R. E. Lowell, and P. Valkenburg. 2011. Integrating motion-detection, cameras, and hair snags for wolverine identification. Journal of Wildlife Management 75:731–739.

Magoun, A. J., J. C. Ray, D. S. Johnson, P. Valkenburg, F. N. Dawson, and J. Bowman. 2007. Modeling wolverine occurrence using aerial surveys of tracks in snow. Journal of Wildlife Management 71:2221–2229.

Male, T. D., and M. J. Bean. 2005. Measuring progress in US endangered species conservation. Ecology Letters 8:986–992.

Marsh, D. M., and P. C. Trenham. 2008. Current trends in plant and animal population monitoring. Conservation Biology 22:647–655.

Marucco, F., D. H. Pletscher, L. Boitani, M. K. Schwartz, K. L. Pilgrim, and J.-D. Lebreton. 2009. Wolf survival and population trend using non-invasive capture-recapture techniques in the Western Alps. Journal of Applied Ecology 46:1003–1010.

Moriarty, K. M., W. J. Zielinski, A. G. Gonzales, T. E. Dawson, K. M. Boatner, C. A. Wilson, F. V. Schlexer, K. L. Pilgrim, J. P. Copeland, and M. K. Schwartz. 2009. Wolverine Confirmation in California after Nearly a Century: Native or Long-Distance Immigrant? Northwest Science 83:154–162.

Mulders, R., J. Boulanger, and D. Paetkau. 2007. Estimation of population size for wolverines Gulo gulo at Daring Lake, Northwest Territories, using DNA based mark-recapture methods. Wildlife Biology 13:38–51.

Otto, C. R. V., and G. J. Roloff. 2011. Using multiple methods to assess detection probabilities of forest-floor wildlife. The Journal of Wildlife Management 75:423–431.

Povilitis, A., and K. Suckling. 2010. Addressing Climate Change Threats to Endangered Species in U.S. Recovery Plans. . Conservation Biology 24:372–376.

Primack, R. B. 2006. Essentials of Conservation Biology. Fourth edition. Sinauer Associates, Sunderland.

R Development Core Team. 2011. R Foundation for Statistical Computing, Vienna, Austria.

Rands, M. R. W., W. M. Adams, L. Bunnen, S. H. M. Butchart, A. Clements, D. Coomes, A. Entwistle, I. Hodge, V. Kapos, J. P. W. Scharlemann, W. J. Sutherland, and B. Vira. 2010. Biodiversity Conservation: Challenges Beyond 2010 Science 329:1298–1303.

Rhodes, J. R., and N. Jonzén. 2011. Monitoring temporal trends in spatially structured populations: how should sampling effort be allocated between space and time?. . Ecography 34:1040–1048.

Rhodes, J. R., A. J. Tyre, N. Jonzén, C. A. McAlpine, and H. P. Possingham. 2006. Optimizing Presence-Absence Surveys For Detecting Population Trends. The Journal of Wildlife Management 70:8–18.

Schwartz, M. K., J. P. Copeland, N. J. Anderson, J. R. Squires, R. M. Inman, K. S. McKelvey, K. L. Pilgrim, L. P. Waits, and S. A. Cushman. 2009. Wolverine gene flow across a narrow climatic niche. Ecology 90:3222–3232.

Schwartz, M. K., G. Luikart, and R. S. Waples. 2007. Genetic monitoring as a promising tool for conservation and management. Trends in Ecology & Evolution 22:25–33.

Schwartz, M. K., and S. L. Monfort. 2008. Genetic and endocrine tools for carnivore surveys. *in* R. A. Long, P. MacKay, J. C. Ray, andW. J. Zielinski, editors. Noninvasive survey methods for North American carnivores. Island Press, Washington D.C.

Shenk, T. M. 2009. Post–Release Monitoring of Lynx Reintroduced to Southwestern Colorado. Colorado Division of Wildlife.

Shenk, T. M., and R. H. Kahn. 2010. The Colorado lynx reintroduction program. Colorado Division of Wildlife.

Squires, J. R., J. Copeland, T. J. Ulizio, I. K. Schnvartz, and L. F. Ruggiero. 2007. Sources and patterns of wolverine mortality in western Montana. Journal of Wildlife Management 71:2213–2220.

Theobald, D. M., and T. M. Shenk. 2011. Areas of high habitat use from 1999–2010 for radio-collared Canada lynx reintroduced to Colorado. Colorado State University.

Ulizio, T. J., J. R. Squires, D. H. Pletscher, M. K. Schwartz, J. J. Claar, and L. F. Ruggiero. 2006. The Efficacy of Obtaining Genetic-Based Identifications from Putative Wolverine Snow Tracks. Wildlife Society Bulletin 34:1326–1332.

USFWS. 2010. Endangered and threatened wildlife and plants; 12-month finding on a petition to list the North American wolverine as endangered or threatened.

White, G. C., and K. P. Burnham. 1999. Program MARK: survival estimation for populations of marked animals. Bird Study 46:120–138.

Zielinski, W. J., and H. B. Stauffer. 1996. Monitoring Martes Populations in California: Survey Design and Power Analysis. Ecological Applications 6:1254–1267.

Prepared by ____________________________
Jacob S. Ivan

Table 1. A summary of variables and ranges of those variables tested in our simulations using program SPACE.

Variable	Values tested		
Population size	N = 30, 200, 500		
Population growth rates	λ =	0.933, 1.041, 1.072, 1.116	N = 30
		0.933, 1.041	N=200
		0.933, 0.977, 0.989, 1.041	N=500
Limit on movement	none; 1, 2 s.d. from home range center		
Simulated detection probability	0.2, 0.8		
Number of cells sampled	10 - 90% of grid		
Number of visits	2 - 7		
Cell size	100, 225, 500, 1000 km^2		
Sampling	Annual or alternating years (every other year)		

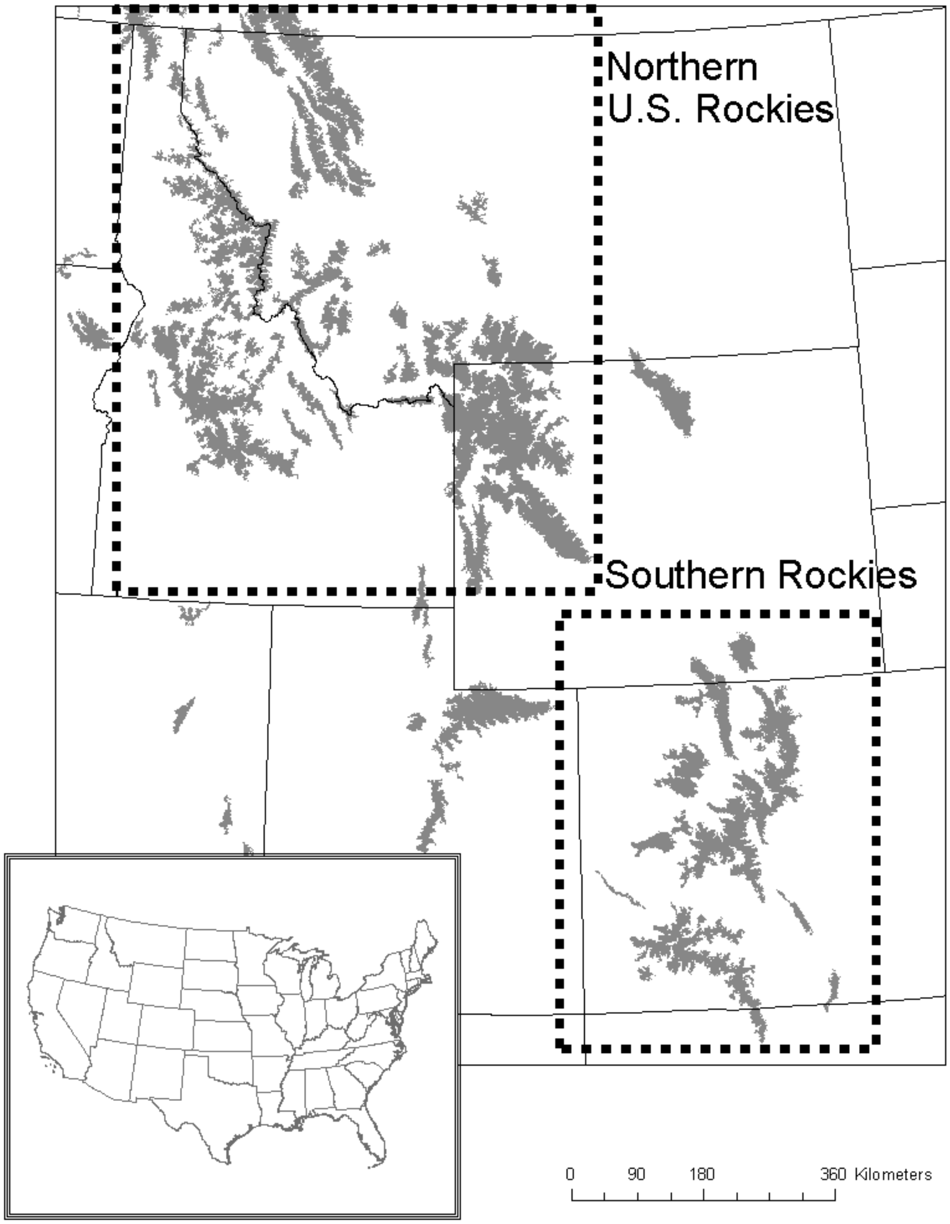


Figure 1. Map of study area. Distribution of persistent spring snow in the U.S. Rocky Mountains. Two separate landscapes were included in this study: one corresponding to fairly continuous habitat in the U.S. Northern Rockies, which is currently occupied by wolverines, and a second area in the Southern Rockies, where wolverines may recolonize or be reintroduced.

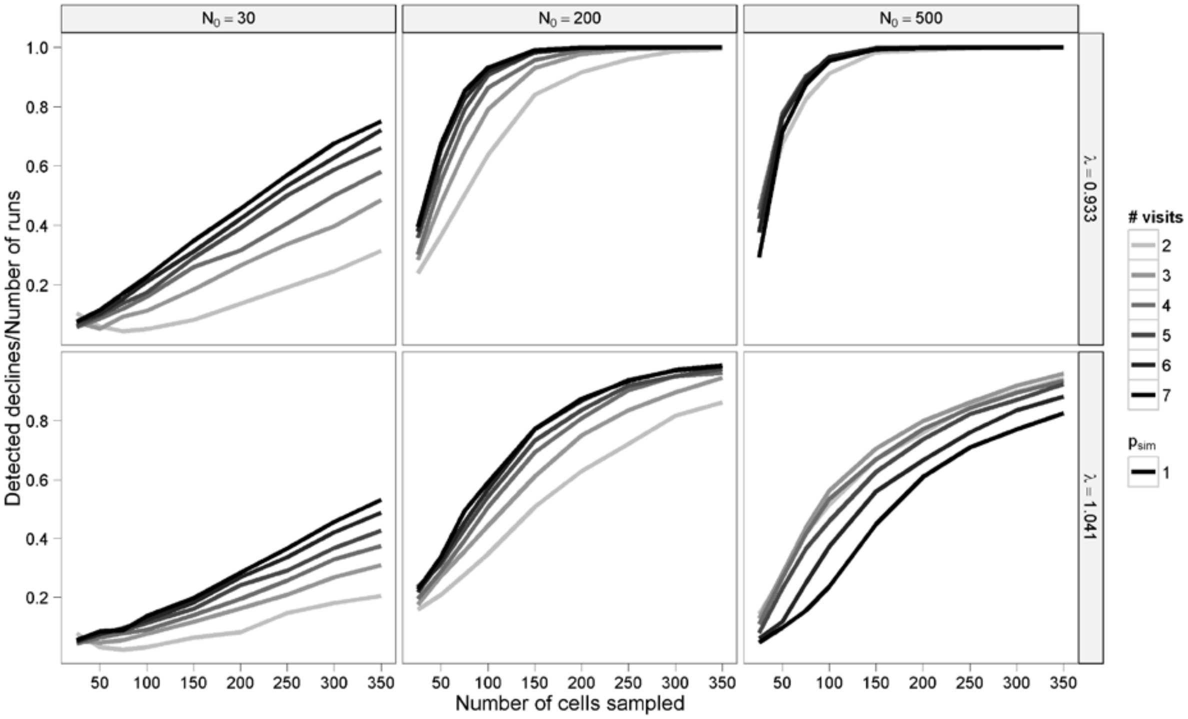


Figure 2. Effect of population size. Effect of population size on power to detect trends in the Northern U.S. Rockies. Assumes perfect detection associated with sampling for a 50% decline (λ = 0.933) or a 50% increase (λ = 1.041) from initial population sizes of 30, 200, and 500 individuals in the Northern Rockies. Simulated populations were sampled using a grid of 225km^2 cells overlaid on the landscape.

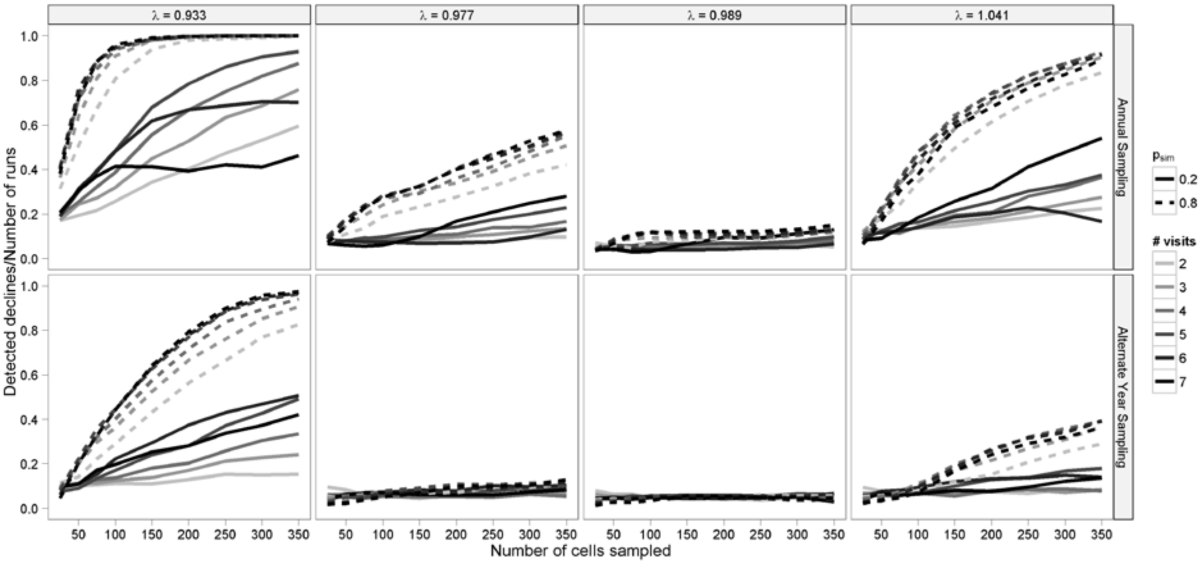


Figure 3. Power for detecting trends in the U.S. Northern Rockies. Results from a power analysis for assessing the feasibility of using occupancy to monitor trend in the population of wolverines in the U.S. Northern Rockies, assuming N=500 individuals and a cells size of 225km^2. Results are parsed by population growth rate (λ = 0.933, 0.977, 0.989, 1.041 corresponding to 50%, 20%, and 10% declines over 10 years or a 50% increase), sampling effort (whether sampling occurred annually or every other year), detection probability for sampling, number of visits per year, and number of grid cells sampled from a total of 388. Power is based on number of detected trends in 1,000 simulated populations.

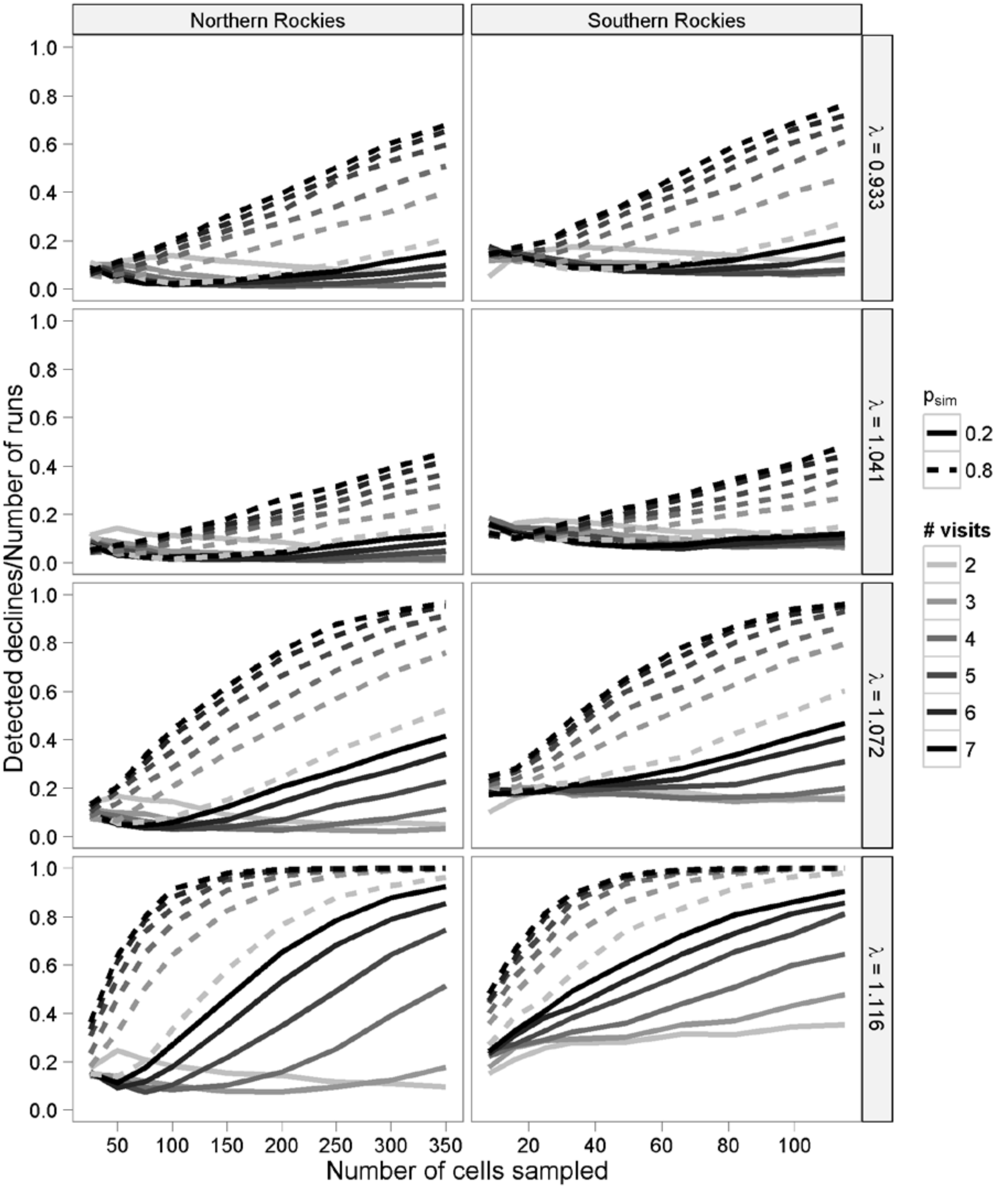


Figure 4. Comparison of N=30 populations in Northern and Southern Rockies. Power to detect population trends for populations of 30 wolverines in U.S. Northern Rockies compared to the same population size in the Southern Rockies landscape. Ability to detect a population decline depends on population growth rate (λ = 0.933, 1.041, 1.072, 1.116 corresponding to a 50% decline over 10 years or 50%, 2-fold, or 3-fold increases in population size over 10 years) and sampling effort (detection probability for sampling, number of visits per year, number of grid cells sampled from a total of 388 for the Northern Rockies or 128 for Colorado). Power is based on number of detected trends from 1,000 simulated populations.

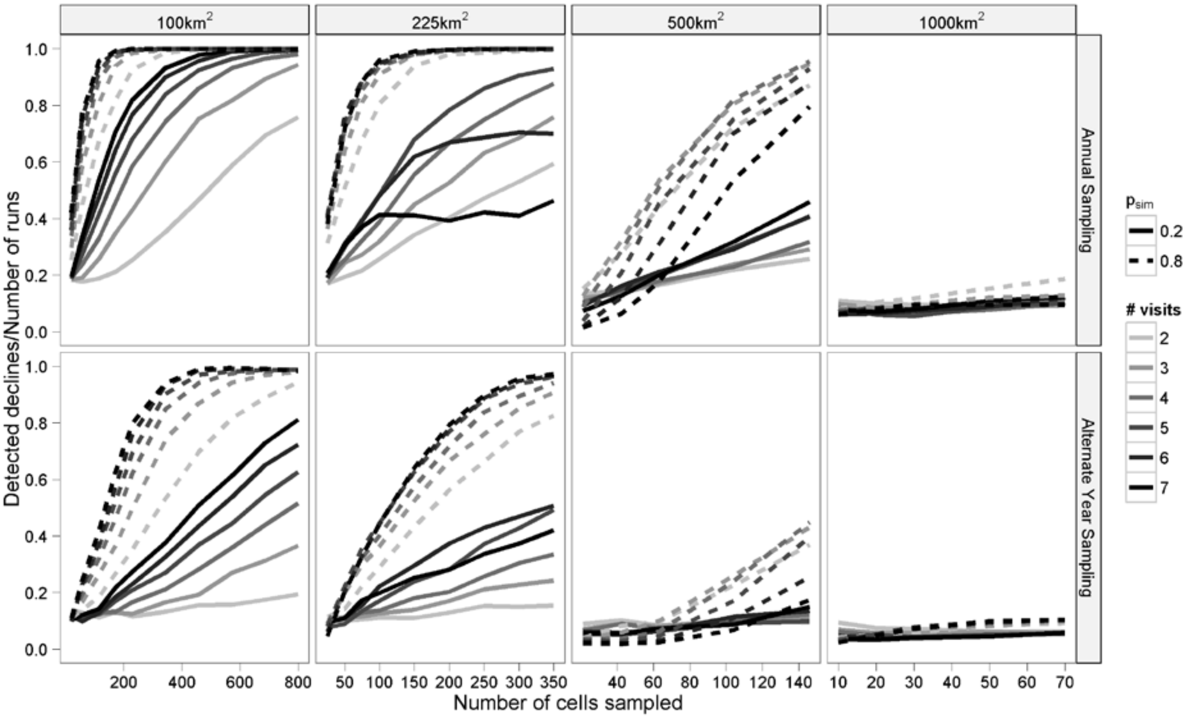


Figure 5. Effect of cell size on power. Effect of grid size on power to detect population trends in the wolverine population (N=500) in the Northern Rockies using occupancy. As grid size changes, the total number of grid cells on the landscape also changes from 887 for a 100km^2 grid, 388 for a 225 km^2 grid, to 76 for a 1000km^2 grid.

Colorado Division of Parks and Wildlife
July 2011–June 2012

WILDLIFE RESEARCH REPORT

State of:	Colorado	:	Division of Parks and Wildlife
Cost Center:	3430	:	Mammals Research
Work Package:	0670	:	Lynx Conservation
Task No.:	N/A	:	Monitoring Canada Lynx in Colorado Using Occupancy Estimation: Initial Implementation in the Core Lynx Research Area
Federal Aid Project No.	N/A		

Period Covered: July 1, 2011 – June 30, 2012

Author: J. S. Ivan

Personnel: T. Shenk, G. Merrill, E. Newkirk

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

ABSTRACT

In an effort to restore a viable population of Canada lynx (*Lynx canadensis*) to the southern portion of their former range, 218 individuals were reintroduced into Colorado from 1999–2006 (Devineau et al. 2010). In 2010, the Colorado Division of Wildlife (now Colorado Parks and Wildlife [CPW]) determined that the reintroduction effort met all benchmarks of success, and that a viable, self-sustaining population of Canada lynx had been established. The purpose of this project was to develop a scientifically rigorous statewide plan to monitor this newly established population. Occupancy estimation, the use of presence/absence data to estimate the proportion of sample units used by a species within a study area, is appropriate for such a program. To evaluate this approach and provide initial estimates of occupancy and detection probability for planning purposes, we conducted a pilot occupancy estimation project in the core reintroduction area in the San Juan Mountains of southwestern Colorado. Lynx habitat in the study area was divided into 75-km^2 sample units (8.66 km x 8.66 km cells), and we stratified the units into those accessible for snow tracking and “inaccessible” units, which were sampled via remote cameras. We randomly sampled 30 units from each stratum. A summary of snow tracking results can be found in Ivan (2011). Of the 120 cameras we deployed in late fall to survey the 30 inaccessible units, 113 were still operational when retrieved in early summer; 6 had memory cards that reached capacity in either May or June; 1 was stolen. We obtained 151,191 photos (min = 90, max = 6,948 per camera) from this effort. We determined species for each photo and checked our work using multiple observers. Average agreement between observers was 96%. We estimated that approximately 25% of inaccessible cells were used by lynx. Detection probability was 0.43. These pilot data are currently being used to conduct simulations and power analyses to determine how many sample units will be required to detect population changes of interest in Colorado.

WILDLIFE RESEARCH REPORT

MONITORING CANADA LYNX IN COLORADO USING OCCUPANCY ESTIMATION: INITIAL IMPLEMENTATION IN THE CORE LYNX RESEARCH AREA

JACOB S. IVAN

P. N. OBJECTIVE

Assess the use of occupancy estimation as a means of monitoring Canada lynx in Colorado using the Core Research Area in the San Juan Mountains as a test site.

SEGMENT OBJECTIVES

1. Obtain initial estimates of occupancy and detection probability from units where remote cameras were the primary detection method.
2. Determine covariates and covariate structures that will be most useful for modeling occupancy and detection probability for camera surveys.
3. Combine these results with those obtained via previous work (snow tracking) to inform simulation work aimed at determining the number of sample units, and visits to each unit, required to detect changes of interest in the lynx population in Colorado.

INTRODUCTION

The Canada lynx (*Lynx canadensis*) occurs throughout the boreal forests of northern North America. While Canada and Alaska support healthy populations of the species, the lynx is currently listed as threatened under the Endangered Species Act (ESA) of 1973, as amended (16 U. S. C. 1531 et. seq.; U. S. Fish and Wildlife Service 2000) in the conterminous United States. Colorado represents the southern-most historical distribution of naturally occurring lynx, where the species occupied the higher elevation, montane forests in the state (U. S. Fish and Wildlife Service 2000). Lynx were extirpated or reduced to a few animals in Colorado, however, by the late 1970’s (U. S. Fish and Wildlife Service 2000), most likely due to multiple human-associated factors, including predator control efforts such as poisoning and trapping (Meaney 2002). Given the isolation of and distance from Colorado to the nearest northern populations of lynx, the Colorado Division of Wildlife (now Colorado Parks and Wildlife [CPW]) considered reintroduction as the best option to reestablish the species in the state. Therefore, a reintroduction effort was begun in 1997, and 218 lynx were released into Colorado from 1999 – 2006 (Devineau et al. 2010). The goal of the Colorado lynx reintroduction program was to establish a self-sustaining, viable population of lynx. Progress toward this goal was tracked via evaluation of critical criteria related to lynx survival, fidelity, and recruitment. Recently, CPW determined that the criteria had been met and a viable Canada lynx population currently exists in Colorado (Shenk and Kahn 2010).

In order to track the distribution, stability, and persistence of this new lynx population, a minimally-invasive, long-term, statewide monitoring program is required. Abundance estimation is not feasible logistically and presents statistical difficulties even when field logistics can be managed. However, occupancy estimation, which uses detection/non-detection survey data to estimate the proportion of area occupied in a study area, is appropriate and feasible. In short, such a monitoring scheme requires multiple visits to a sample of survey units, and on each visit observers record whether a lynx was detected or not. Such information can be used to compute the probability of detecting a lynx given that it is present on a unit, which can in turn be used to estimate the proportion (ψ) of all survey units that are occupied. This metric can be tracked through time and is assumed to be closely tied to the size and extent of the lynx population. That is, if the proportion of survey units occupied by lynx declines through time, we assume this is due to a decline in the lynx population itself. Additionally, occupancy surveys can provide information relative to the distribution of lynx in the state.

CPW initiated work to evaluate detection methods for occupancy estimation in 2009-2010 (Shenk 2009). Three methods of detecting lynx were tested in sample units where lynx were known to occur: snow tracking surveys, remote camera surveillance, and hair snags. The best method for detecting lynx was snow-tracking (daily detection probability = 0.70). Camera surveillance was far less efficient (daily detection probability = 0.085), and hair snares were ineffective (daily detection probability = 0.0; Ivan and Shenk 2010). Snow tracking, however, requires safe and extensive access to a survey unit via truck and/or snowmobile. Therefore, it cannot be used in roadless or wilderness areas, which may provide important lynx habitat. Here we build on this work to test occupancy estimation on a large scale using snow tracking where accessibility permitted it, and remote cameras in areas that were not accessible.

METHODS

Study Area

The study area consisted of the 20,684 km^2 "Lynx Core Research Area" in southwest Colorado. The Core Research Area is defined as areas >2591 m (>8500 ft) in elevation within the area bounded by New Mexico to the south, Taylor Mesa to the west, and Monarch Pass on the north and east (Figure 1). Topography in this area is characterized by wide plateaus, river valleys, and rugged mountains that reach elevations over 4200 m. Engelmann spruce (*Picea engelmanii*) - subalpine fir (*Abies lasiocarpa*) is the most widely distributed coniferous forest type at elevations most typically used by lynx (2591-3353 m, 8500-11,000 ft).

Sampling

The study area was divided into 75 km^2 (8.66 km × 8.66 km) sample units, which reflects the mean annual home range size of reproductively active female lynx in Colorado (Shenk 2007) and Montana (Squires and Laurion 1999). Sample units that did not meet the following criteria were discarded as they did not represent potential lynx habitat that could be surveyed.

1. ≥50 % of the cell contained conifer or montane/alpine habitat, as identified by the SWReGAP LandCover Dataset (http://earth.gis.usu.edu/swgap/swregap_landcover_report.pdf) and
2. ≥ 50 % of the cell was located on public land (tribal, NGO, city, and county lands were considered private) as determined by COMaP (Theobald, D.M., G. Wilcox, S.E. Linn, N. Peterson, and M. Lineal. 2008. Colorado Ownership, Management, and Protection v7 database. Human Dimensions of Natural Resources and Natural Resource Ecology Lab, Colorado State University, Fort Collins, CO, www.nrel.colostate.edu/projects/comap).

Each of the remaining sample units was assigned a random number resulting from a spatially balanced sampling scheme (RRQRR; Theobald et al. 2007) and units were stratified by accessibility for snow tracking or camera surveys. The cells with the lowest 30 random numbers for each stratum were selected for sampling during the pilot work. A few cells in both strata were discarded once field work began due to access issues and these were replaced with cells 31, 32, etc.

Snow tracking Surveys

A detailed discussion of the methods and results associated with snow tracking surveys appears in Ivan (2011). We do not repeat that discussion here. Instead we focus on methods and results from the remote cameras, as those data were unavailable for the 2011 report.

Camera Surveys

Four remote camera sets (RECONYX RapidFire^{TM} Professional PC85) were placed within each selected "inaccessible" sample unit during September and October. Placement of camera sets was not random within the unit; they were placed strategically on the landscape to maximize coverage of the sample unit and exploit microsites most likely to be used by lynx. Camera sets consisted of 1) a remote camera mounted to a tree using a Master Lock ^{TM} Python^{TM} cable lock, 2) a target tree at which the camera was pointed, generally about 5–10m away, 3) a compact disc strung from a nearby branch to visually attract lynx from a distance, 4) 2 feathers strung up in such a manner as to entice lynx to walk between the camera and the target tree, and 5) wool soaked in commercial scent lure that was packed into the bark of the target tree to hold lynx in front of the camera (Figure 2). Cameras were placed higher than usual, about head-height, and pointed slightly downward at the target tree so photos could be obtained during both snow-free periods and during periods of accumulating snow. Cameras were collected during June and July at which time the number of photos, percent of memory card used, percent battery life remaining, and condition of visual/scent lures was recorded. All photo attributes were imported into a database and species was assessed for each photo based on review by at least 2 observers.

Analysis

Assumptions inherent in occupancy estimation are 1) surveyed sites are either occupied or not occupied by the species of interest throughout the duration of the study; no sites change status during the survey period (i.e., the system is closed), 2) the probability of occupancy is constant across sites or can be modeled using covariates, 3) the probability of detection is constant across sites or can be modeled using site-specific covariates, and 4) species detection at a site is assumed to be independent of species detection at other sites (MacKenzie et al. 2006). Sampling mobile carnivores such as lynx presents a clear violation of the first assumption as individuals undoubtedly move into and out of sample units routinely. Fortunately, estimation can proceed, but the quantities estimated are different from traditional occupancy estimation. Rather than estimating the probability that a unit is *occupied* by lynx, we now estimate the probability that a sample unit is *used* by lynx. Also, the estimated detection parameter is not the probability of detection given a site is occupied, it is the product of a) the probability of detection given the species is available for detection, and b) the probability that the species was available. These subtleties aside, the procedure still gives a metric (use) that can be monitored through time to detect trends.

We used the "Occupancy Estimation" data type in Program MARK to produce initial estimates of occupancy (i.e., use, ψ) and detection probability (p) for the camera stratum. Photos were grouped by month (November to March) for each sample unit such that encounter histories included 5 "visits." Due to this grouping, there were no meaningful covariates for p. Individual cameras recorded moon phase and temperature for each photo, but aggregated over a month, these data were not helpful. Some camera sets used different scent lures than others, but aggregating by unit negates the utility of this information as well.

We hypothesized that the proportion of spruce/fir and/or willow (*Salix* spp.) cover in each unit may affect the probability of use and/or probability of detection. Thus, we considered these covariates as potentially important for explaining variability in ψ and p. We held ψ constant and built an additive model for each detection covariate (one at a time) to determine the best structure for p. We then held p at the best structure as determined by AICc (Burnham and Anderson 2002) and fit additive models using the covariates for ψ. We also ran a model where both p and ψ were held constant as a baseline for comparison. We report estimates of p and ψ from the AICc top model.

RESULTS

Of the 120 cameras deployed during Fall 2010, 113 were still operational when retrieved in Summer 2011 after 234-309 days of deployment. Six had memory cards that reached capacity in either May or June, and one camera was stolen. On average, we obtained 1,260 photos per camera (min = 90, max = 6,948) for a total of 151,191 photos. At the time of retrieval, compact discs were still operational for 46% of camera sets, feathers were operational at 64% of sets, and remnants of scent lure were detected at 55% of sets. We obtained 445 photos of lynx and detected them in 7 of the 30 units sampled (Figure 1). Average agreement between photo reviewers was 96%.

Of the model structures we fit, none was clearly better than the others as AICc weight was distributed fairly evenly (Table 1). Beta estimates for fitted models suggested that ψ was positively associated with both percent spruce/fir and percent willow in a given unit. Spruce/fir was also positively associated with detection probability, whereas willow was negative associated with detection probability. However none of these models were as well supported by the data as the null model in which ψ and p were considered constant across cells. Thus, results generally followed our expectations, but the null model came out on top likely due to sparse data and small samples in this pilot study. Model-averaged estimates for ψ and p were 0.25 and 0.42, respectively. Detection probability using cameras was about the same as for snowtracking (Ivan 2011), but estimated probability of use for inaccessible sampling units was about half that estimated for accessible cells sampled via snow tracking.

DISCUSSION

Initial results indicate that occupancy (use) can be adequately modeled using data collected via snow tracking. Precision on estimates of ψ and p was relatively poor, but this can be addressed by sampling more units and/or making more visits. Modeling p and ψ as functions of the covariates (spruce/fir and willow) was not as well supported as specifying them to be constant across units. However, we recommend continuing to record and use these covariates and others in future surveys as it seems reasonable that these covariates should impact detection probability and/or use, and their effects may be important as sample size increases.

We estimated that lynx used approximately 25% of the sample units available in the Core Research Area. However, for this pilot study, lynx habitat was coarsely defined as units with >50% conifer and/or montane cover and >50% public land. In several cases, sampled units met these criteria, but field crews that actually made visits indicated these units did not appear to include much lynx habitat. CPW recently finished an analysis to produce a map of predicted lynx habitat throughout the state. In the future, we expect to use this map to frame the population of units to sample for lynx monitoring. This more refined population of sample units should reduce time wasted surveying units that do not include good lynx habitat, and will result in an increased estimate of probability of use. Indeed, re-running the analysis using only those cells ($n = 24$) within the top 40% of predicted lynx habitat in the state increased the occupancy estimate to 0.31.

Roughly half of the visual attractants we used did not operate through the entirety of the study. These attractants are important for drawing lynx to the set from a distance and their failure diminishes the utility of the cameras for detecting lynx. If cameras are to be used in the future, design changes will be necessary to ensure that most of these visual attractants operate throughout the sampling season. We suggest that attractants be attached via wire rather than fishing line. We also suggest that auditory attractants may be helpful. In a recent study on cougars (*Puma concolor*) in the Front Range of Colorado, visitation rates at camera sites increased dramatically when auditory attractants were used in addition to scent lures and visual attractants (Kirstie Yeager, personal communication).

ACKNOWLEDGMENTS

We thank Britta Schielke, Cate Brown, Wendy Lanier, Joan Meiners, Shane McKenzie, Nick Burgmeier, Doug Clark, Bob Peterson, Tim Hanks, Kei Yasuda, Ashley Bies, Tyler Kelly, Alyssa Winkler, and Carolyn Shores for their efforts in the field. Dale Gomez and Rhandy Ghormley (USFS) graciously coordinated housing for seasonal crews. We thank various personnel from both the Rio Grande and San Juan National Forests for logistical help in the field. Funding was provided by a U.S. Fish and Wildlife Service Section 6 Grant.

LITERATURE CITED

Burnham, K. P., and D. R. Anderson. 2002. Model Selection and Multimodel Inference: A practical information-theoretic approach. Springer, New York, New York, USA.

Devineau, O., T. M. Shenk, G. C. White, P. F. Doherty Jr., P. M. Lukacs, and R. H. Kahn. 2010. Evaluating the Canada lynx reintroduction programme in Colorado: patterns in mortality. Journal of Applied Ecology 47:524–531.

Ivan, J. S., and T. M. Shenk. 2010. Estimating the Extent, Stability and Potential Distribution of Canada Lynx (*Lynx canadensis*) in Colorado: initial implementation in the core lynx research area. Wildlife Research Report, Colorado Division of Wildlife, Fort Collins, Colorado.

Ivan, J. S. 2011. Monitoring Canada lynx in Colorado using occupancy estimation: Initial implementation in the Core Lynx Research Area. Wildlife Research Report. Colorado Division of Parks and Wildlife, Fort Collins, CO, USA. Pages 11–20.

MacKenzie, D. I., J. D. Nichols, J. A. Royle, K. H. Pollock, L. L. Bailey, and J. E. Hines. 2006. Occupancy Estimation and Modeling: Inferring Patterns and Dynamics of Species Occurrence. Elsevier Academic Press, Oxford, UK.

McKelvey, K. S., J. von Kienast; K.B. Aubry; G. M. Koehler; B. T. Maletzke; J. R. Squires; E. L. Lindquist; S. Loch; M. K. Schwartz. 2006. DNA analysis of hair and scat collected along snow tracks to document the presence of Canada lynx. Wildlife Society Bulletin 34: 451-455.

Meaney C. 2002. A review of Canada lynx (*Lynx canadensis*) abundance records from Colorado in the first quarter of the 20Th century. Colorado Department of Transportation Report.

Shenk, T. M. 2005. Post-release monitoring of lynx reintroduced to Colorado. Job Progress Report, Colorado Division of Wildlife, Fort Collins, Colorado.

__________. 2007. Post-release monitoring of lynx reintroduced to Colorado. Wildlife Research Report, Colorado Division of Wildlife, Fort Collins, Colorado.

__________. 2009. Post-release monitoring of lynx reintroduced to Colorado. Wildlife Research Report, Colorado Division of Wildlife, Fort Collins, Colorado.

Shenk, T.M., and R. H. Kahn. 2003. Post-release monitoring of lynx reintroduced to Colorado. Wildlife Research Report, Colorado Division of Wildlife, Fort Collins, Colorado.

__________. 2010. The Colorado lynx reintroduction program. Report to the Colorado Division of Wildlife, Fort Collins, Colorado.

Squires, J. R. and T. Laurion. 1999. Lynx home range and movements in Montana and Wyoming: preliminary results. Pages 337–349 in L. F. Ruggiero, K. B. Aubry, S. W. Buskirk, G. M. Koehler, C. J. Krebs, K. S McKelvey, and J. R. Squires, editors. Ecology and Conservation of Lynx in the United States. General Technical Report for U. S. D. A. Rocky Mountain Research Station. University Press of Colorado, Boulder, Colorado.

Theobald, D.M., D.L. Stevens, Jr., D. White, N.S. Urquhart, A.R. Olsen, and J.B. Norman. 2007. Using GIS to generate spatially balanced random survey designs for natural resource applications. *Environmental Management* 40(1): 134–146.

U. S. Fish and Wildlife Service. 2000. Endangered and threatened wildlife and plants: final rule to list the contiguous United States distinct population segment of the Canada lynx as a threatened species. Federal Register 65, Number 58.

Prepared by ____________________________
Jacob S. Ivan

Table 1. Model selection results for estimating lynx occupancy of sample units surveyed via remote camera in the Core Research Area, San Juan Mountains, Colorado, Winter 2010–2011.

Model	AIC_c	ΔAIC_c	AIC_c Wt	Num Par
ψ(.)*p*(.)	84.54	0.00	0.29	2
ψ(.)*p*(willow)	85.06	0.52	0.22	3
ψ(.)*p*(SprFir)	85.37	0.83	0.19	3
ψ(SprFir)*p*(.)	85.73	1.19	0.16	3
ψ(willow)*p*(.)	85.92	1.38	0.14	3

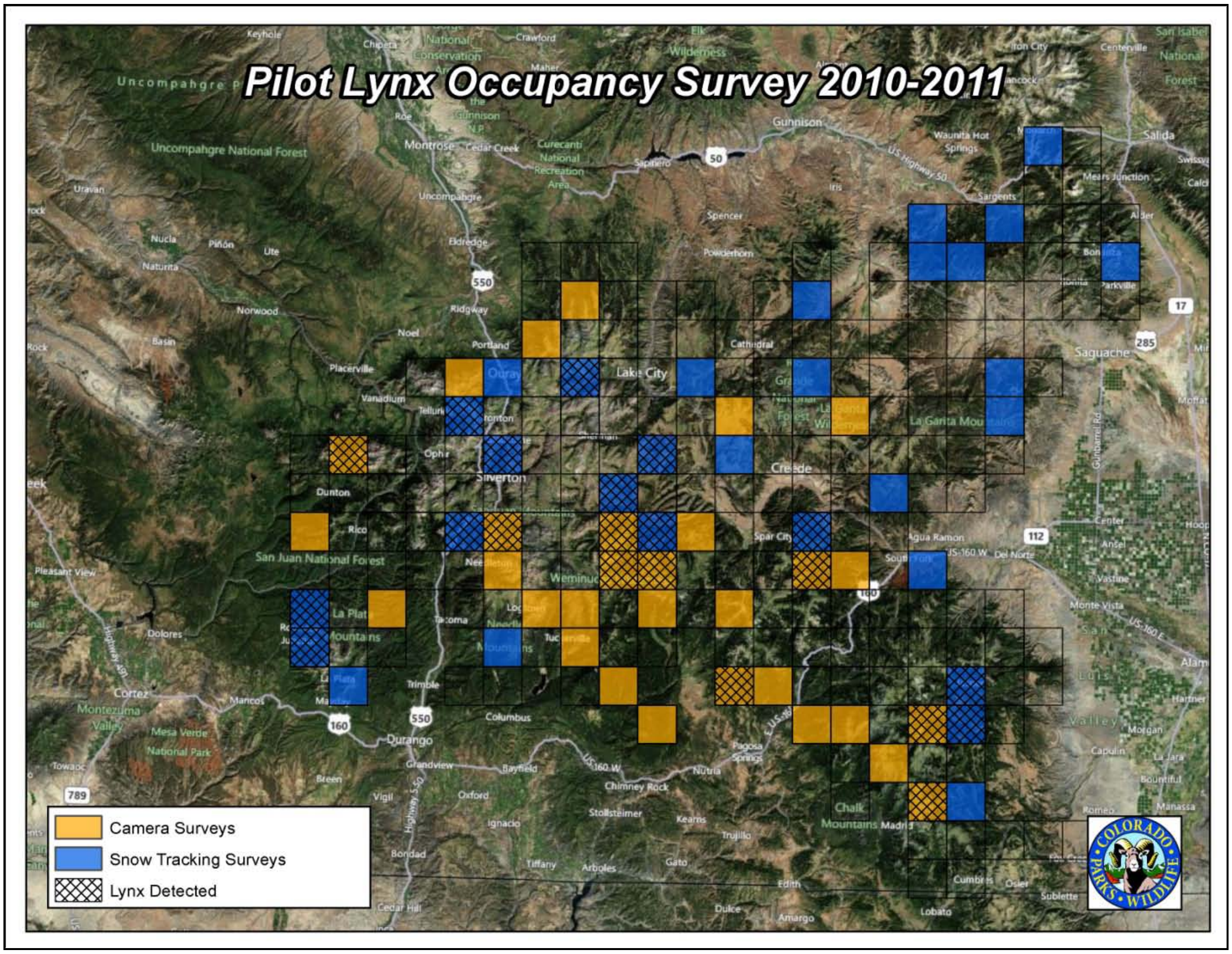


Figure 1. Canada lynx Core Research Area in southwest Colorado. Squares are 75km^2 sample units available for occupancy surveys. Blue represents the sample of 30 “accessible” units selected for snow tracking surveys. Orange are “inaccessible” units selected for surveys using remote cameras. Cross-hatching indicates units where lynx were detected.

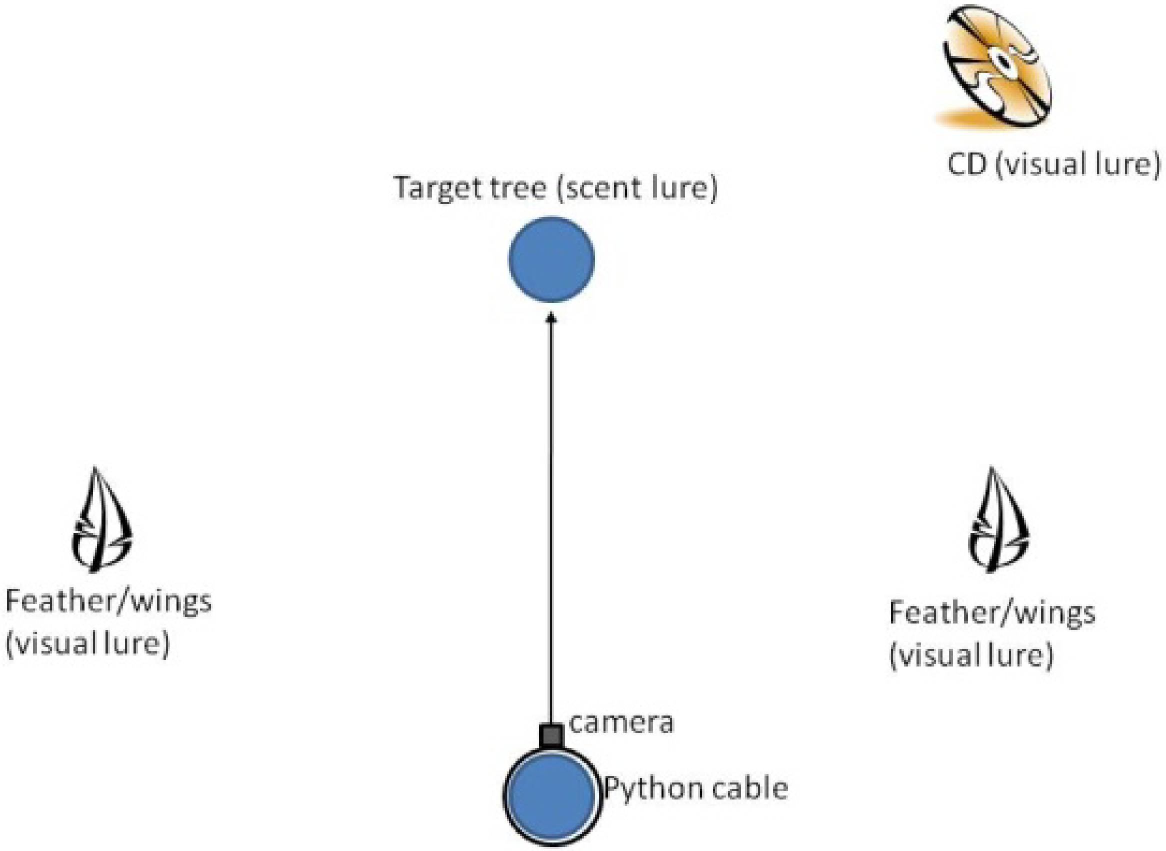


Figure 2. General configuration of remote camera sets for detecting Canada lynx. Four such sets were deployed in each of 30 inaccessible sample units from Fall 2010 to Summer 2011.

Colorado Division of Parks and Wildlife
July 2011–June 2012

WILDLIFE RESEARCH REPORT

State of:	Colorado	:	Division of Parks and Wildlife
Cost Center:	3430	:	Mammals Research
Work Package:	0670	:	Lynx Conservation
Task No.:	N/A	:	Predicted lynx habitat in Colorado
Federal Aid Project No.	N/A		

Period Covered: July 1, 2011 – June 30, 2012

Author: J. S. Ivan

Personnel: M. Rice, P. Lukacs, T. Shenk (National Park Service), D. Theobald (Colorado State University), E. Odell

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

ABSTRACT

In an effort to restore a viable population of federally threatened Canada lynx (*Lynx canadensis*) to the southern portion of their former range, 218 individuals were reintroduced into Colorado from 1999–2006 (Devineau et al. 2010). In 2010, the Colorado Division of Wildlife (now Colorado Parks and Wildlife [CPW]) determined that the reintroduction effort met all benchmarks of success, and that a viable, self-sustaining population of Canada lynx had been established (Shenk and Kahn 2010). The purpose of this project was to develop a statewide predictive map of relative lynx use based upon location data collected during the reintroduction period. To build the map, we divided the state into 1.5 km × 1.5 km cells and tallied the number of locations in each cell. We then fit models to these count data using vegetation, elevation, slope, wetness, and degree of human development in each cell as predictor variables. We produced models for both summer and winter habitat use. We found that regardless of season, lynx were positively associated with spruce/fir (*Picea engelmannii*/*Abies lasiocarpa*), mixed spruce/fir, aspen (*Populus tremuloides*), elevation and slope; they were negatively associated with distance to large forest patches. During summer, lynx use of lodgepole pine (*Pinus contorta*) stands was predicted to increase. Lynx were predicted to avoid montane forest (Douglas-fir [*Pseudotsuga menziesii*], Ponderosa pine [*Pinus ponderosa*]), and areas near high traffic volume road segments, especially during summer. These maps of predicted lynx use should aid land managers in prioritizing areas for conservation, development, and resource extraction with respect to potential impacts to lynx and lynx habitat.

WILDLIFE RESEARCH REPORT

PREDICTED LYNX HABITAT IN COLORADO

JACOB S. IVAN

P. N. OBJECTIVE

Use location data collected during Canada lynx (*Lynx canadensis*) reintroduction to build a model of relative use, then apply this model statewide to produce a predictive map of relative lynx use for Colorado.

SEGMENT OBJECTIVES

1. Prepare manuscript for submission to *Journal of Wildlife Management*.

INTRODUCTION

In an effort to restore a viable population of federally threatened Canada lynx (*Lynx canadensis*) to the southern portion of their former range, 218 individuals were reintroduced into Colorado from 1999–2006 by the Colorado Division of Wildlife (now Colorado Parks and Wildlife [CPW], Devineau et al. 2010). In 2010, CPW determined that the reintroduction effort met all benchmarks of success, and that a viable, self-sustaining population of Canada lynx had been established (Shenk and Kahn 2010). Attainment of this goal is a conservation success, but it has also created a series of issues for land management agencies to consider as they plan changes to the landscape. These issues require knowledge of the types of landscapes and forest stands important for reproduction, movement, dispersal, and general home range use by lynx.

As a first step toward providing this information, Theobald and Shenk (2011) conducted an analysis to describe the types of areas that were known to be used by re-introduced lynx. Specifically, they used LoCoH (Getz and Wilmers 2004, Getz et al. 2007) methods to create a population-level utilization distribution (UD, a probability surface of lynx occurrence) for lynx in Colorado. They then summarized landscape attributes within the 90% isopleth (i.e., polygon(s) containing 90% of the probability surface) of this UD. This work provides valuable information regarding the types of areas that were *known* to be used by lynx from 1999 to 2010. By nature of the data collection and research focus, most of this “use” information was derived from core areas in the San Juan Mountains of southwest Colorado and Sawatch Range in the central part of the state.

The purpose of the current project is to extend the work of Theobald and Shenk (2011) by producing a map of *predicted* lynx use on a *statewide* scale. Such an exercise will identify areas within Colorado that should contain high quality lynx habitat, regardless of whether or not it was used by the sample of radio-telemetered individuals tracked during reintroduction research. Both works have strengths and weaknesses, but together they provide tools for prioritizing areas for conservation, development, and resource extraction with respect to potential impacts to lynx.

METHODS

While this worked was completed in January 2012, the final report was included in revisions to the previous annual report and is not repeated here. We refer the reader to Ivan (2011) for details regarding methods and results from this work. Our intent is to work this report into a manuscript submission to *Journal of Wildlife Management* by Fall 2012.

SUMMARY

As expected, relative predicted use by lynx during winter months was negatively associated with distance to large patches of conifer (D50HA) and positively associated with spruce/fir (SF), mixed spruce/fir (MIXSF), elevation (ELEV) and slope . Of these associations, the relationship with spruce/fir was strongest. Predicted use was also positively associated with topographic wetness and aspen cover. We projected these associations (and other more minor associations included in competing models) onto a map of the state and arbitrarily defined the top 20% of predictions as high quality lynx habitat. There are 1,869,975 ha of such habitat in Colorado. Most of this high quality habitat was predicted to occur in the southern part of the state in the San Juan, Culebra, and Wet Mountain Ranges. In the central portion of the state, high predicted use is expected in the northern Sawatch and West Elk Ranges, along with Grand Mesa. The Park Range and Flat Tops comprised the best predicted winter lynx habitat farther north

Associations between relative predicted summer use and SF, MIXSF, ELEV, slope, and D50HA were similar to those observed during winter. However, the associations with D50HA and slope were stronger during summer. We also found positive associations between lodgepole pine, aspen, and distance to high volume road segments. The summer predictive map reflects more dispersed predicted use by lynx with the lodgepole playing a larger role, especially farther north. The central and southern Sawatch Range in central Colorado is predicted to have more use than during winter, whereas use on Grand Mesa is predicted to decline. In the northern part of the state, lynx use was predicted to shift more toward the Medicine Bow and Front Ranges.

LITERATURE CITED

Ivan, J. S. 2011. Predicted lynx habitat in Colorado. Wildlife Research Report. Colorado Division of Parks and Wildlife, Fort Collins, CO, USA. Pages 21–35.

Prepared by ___________________________
Jacob S. Ivan

Colorado Division of Parks and Wildlife
July 2011–June 2012

WILDLIFE RESEARCH REPORT

State of: Colorado : Division of Parks and Wildlife
Cost Center: 3430 : Mammals Research
Work Package: 0670 : Lynx Conservation
Task No.: N/A : Density, demography, and seasonal movements of snowshoe hares in central Colorado.
Federal Aid Project No. N/A

Period Covered: July 1, 2011 – June 30, 2012

Author: J. S. Ivan

Personnel: G. White, T. Shenk

ABSTRACT

To improve understanding of snowshoe hare ecology in the southern portion of their range, and enhance the ability of agency personnel to manage subalpine landscapes for snowshoe hares (*Lepus americanus*) and lynx (*Lynx canadensis*) in Colorado, we estimated snowshoe hare density, survival, recruitment, and movement in west-central Colorado, USA from July 2006–March 2009. We sampled 3 types of forest stands that purportedly provide good habitat for hares: 1) mature Engelmann spruce (*Picea engelmannii*)/subalpine fir (*Abies lasiocarpa*), 2) early seral, even-aged lodgepole pine (*Pinus contorta*), and 3) mid-seral, even-aged lodgepole pine that had been pre-commercially thinned. In all stand types and all seasons, snowshoe hare densities were <1.0 hares/ha. During summer, hare densities [±SE] were highest in early seral lodgepole pine (0.20 [0.01] to 0.66 [0.07] hares/ha), lowest in mid-seral lodgepole pine (0.01 [0.04] to 0.03 [0.03] hares/ha), and intermediate in mature spruce-fir (0.01 [0.002] to 0.26 [0.08] hares/ha). During winter, densities were similar among the 3 stand types. Annual survival of hares was highest in mature spruce-fir (0.14 [0.05] to 0.20 [0.07]) and similar between the 2 lodgepole stand types (0.10 [0.03] to 0.16 [0.06]). Stand attributes indicative of dense cover were positively correlated with density estimates and explain relatively more process variance in hare densities than other attributes. These same attributes were not positively correlated with hare survival. Both density and survival of hares in early seral lodgepole stands were positively correlated with the occurrence of similar stands in the surrounding landscape. Recruitment of juvenile hares occurred during all 3 summers in early seral lodgepole stands, 2 of 3 summers in mature spruce-fir stands, and in only 1 of 3 summers in mid-seral lodgepole. Within-season movements of hares were larger during winter than during summer and tended to be larger in early seral lodgepole stands. Hares in both early and mid-seral lodgepole stands tended to make larger movements between seasons than hares in spruce-fir stands, possibly reflecting the variable value of these stands as mediated by snow depth. Based on stand-specific estimates of density, demography, and movement, we conclude that thinned, mid-seral lodgepole stands are less important than mature spruce-fir and small lodgepole stand types. Management for snowshoe hares (and lynx) in central

Colorado should focus on maintaining the latter. Given the more persistent nature of spruce-fir compared to early seral lodgepole, and the fact that such stands cover considerably more area, mature spruce-fir may be the most valuable stand type for snowshoe hares in the region.

We used simulation to compare relative performance of the method we developed to estimate density for this project (TELEM) to other contemporary methods that are widely used (i.e., spatially explicit capture-recapture (SECR), and mean maximum distance moved (MMDM)). We evaluated performance (percent error) under all combinations of 3 levels of detection probability (0.2, 0.4, 0.6), 3 levels of occasions (5, 7, 10), and 3 levels of abundance (10, 20, 40 animals). We also tested each estimator using 5 different models for animal home ranges. TELEM performed best across most combinations of capture probabilities, sampling occasions, true densities, and home range configurations, and performance was unaffected by home range shape. SECR outperformed MMDM estimators in nearly all comparisons and may be preferable to TELEM at low capture probabilities, but performance varied with home range configuration. MMDM estimators exhibited substantial positive bias for most simulations, but performance improved for elongated or infinite home ranges.

WILDLIFE RESEARCH REPORT

DENSITY, DEMOGRAPY, AND SEASONAL MOVEMENTS OF SNOWSHOE HARES IN CENTRAL COLORADO

JACOB S. IVAN

P. N. OBJECTIVE

Assess the relative value of 3 stand types (mature spruce/fir, early seral lodgepole pine, and thinned, mid-seral lodgepole pine) that purportedly provide high quality hare habitat by estimating density, survival, recruitment, and movements of hares in such stands during summer and winter.

SEGMENT OBJECTIVES

1. Publish manuscripts in peer-reviewed scientific journals.

INTRODUCTION

Snowshoe hares (*Lepus americanus*), their famous 10-year population cycle, and close association with Canada lynx (*Lynx canadensis*) have been well-studied in boreal Canada for decades. Snowshoe hare range, however, extends south into the Sierra Nevada, Southern Rockies, upper Lake States, and Appalachian Mountains. Ecology of snowshoe hares in these more southerly regions is not as well understood, though hare research in the U.S. Rocky Mountains has accelerated over the past decade. Through this recent work, biologists have identified stands of young, densely-stocked conifers and those of mature, uneven-aged conifers as primary hare habitat in the region. Both stand types are characterized by dense understory vegetation that provides both browse and protection from elements and predators.

From 1999 to 2006, Canada lynx were reintroduced into Colorado in an effort to restore a viable population to the southern portion of their former range. Snow tracking of released individuals and their progeny indicated that the majority of lynx winter diet in Colorado was comprised of snowshoe hares. Thus, long-term success of the lynx reintroduction effort hinges, at least partly, on maintaining adequate and widespread populations of snowshoe hares in the state. To improve our understanding of snowshoe hare ecology in the southern portion of their range, and enhance the ability of agency personnel to manage subalpine landscapes for snowshoe hares and lynx in Colorado, we conducted an observational study to evaluate purported primary hare habitat in the state. Specifically, we estimated snowshoe hare density, survival, recruitment, and movement indices in mature, uneven-aged spruce/fir (*Picea engelmannii/Abies lasiocarpa*) and 2 classes of young, even-aged lodgepole pine: 1) "small" lodgepole pine (*Pinus contorta*) stands, which were clear cut 20–25 years prior to this study and had regenerated into densely stocked stands with trees 2.54–12.69 cm in diameter, and 2) "medium" lodgepole pine stands (tree diameter = 12.70–22.85 cm) which were clear cut 40-60 years prior to this study and pre-commercially thinned ~20 years prior.

Animal density is one of the most common and fundamental parameters in wildlife ecology and was the first metric we used to evaluate the stand types. However, density can be difficult to estimate from mark-recapture data because animals move on and off of a trapping grid during a sampling session (i.e., lack of geographic closure). Thus, we first developed a density estimator that uses ancillary radio telemetry locations, in addition to mark-recapture information, to account for lack of geographic closure resulting in relatively unbiased estimates of density. We also completed a series of simulations to test the performance of this "telemetry" estimator over a range of sampling parameters (i.e., capture probabilities, sampling occasions, densities, and home range configurations) likely to be encountered in the field, and compared its performance to two other commonly used, contemporary estimators: spatially explicit capture-recapture (SECR), and mean maximum distance moved (MMDM).

STUDY AREA

The study area encompassed roughly 1200 km^2 around Taylor Park and Pitkin, Colorado, USA (39°50'N, 106°34'W; Figure 1), and included a portion of the "Core Reintroduction Area" occupied by reintroduced Canada lynx (Shenk 2009). Open sagebrush (*Artemisia tridentata*) parks dissected by narrow riparian zones of willow (*Salix* spp.) and potentilla (*Potentilla* spp.) dominated the relatively low elevation (~2800–3000 m) parts of the study area. Extensive stands of lodgepole pine occupied low and mid-elevation slopes (~3000–3300 m), giving way to narrow bands of Engelmann spruce/subalpine fir in the sub-alpine zone (~3200–3600 m). Alpine tundra topped the highest parts of the study area (~3300–4200 m). Moist spruce-fir forests also occurred on north-facing slopes at mid-elevations.

Climate was typical of continental, high-elevation zones with relatively short, mild summers and long, harsh winters. Mean July temperature was 14 °C; mean January temperature was −11 °C (Ivan 2011). Maximum snow depth on the study area averaged 80 cm but ranged from 22–163 cm depending on year, elevation, and aspect (Ivan 2011). Snowpack generally persisted from November through May (low elevations) or June (high elevations and north-facing slopes).

Some human habitation occurred in the study area, mostly in the form of seasonal residences. Considerable recreational use occurred during summer in the form of dispersed camping and off-highway vehicle traffic. A suite of native predators were present within the study area including lynx, cougar (*Puma concolor*), coyote (*Canis latrans*), red fox (*Vulpes vulpes*), pine marten (*Martes Americana*), Great Horned Owl (*Bubo virginianus*) and Northern Goshawk (*Accipiter gentilis*).

METHODS

Refer to Ivan (2011) for methods associated with fieldwork conducted during 2006–2009 and subsequent statistical analyses. During fiscal year 2011–2012 we completed work on 2 manuscripts submitted as a pair to the journal *Ecology*. The first of these manuscripts lays out an approach to estimating animal density using auxiliary telemetry information to improve estimates. The second manuscript uses simulation to compare performance of this new estimator to other contemporary estimators. We have just completed what we believe to be final revisions to these papers. Additionally, we spent much of year combining the demography and movement chapters of the primary author's dissertation into a single, comprehensive treatment of snowshoe hare ecology in central Colorado that includes analyses on hare density, survival, recruitment, and movement. This manuscript was recently submitted to the *Journal of Wildlife Management* for consideration as either a research article or monograph.

RESULTS AND DISCUSSION

A comprehensive treatment of the results is widely available in dissertation form (Ivan 2011), so we do not repeat that here. We are currently in the process of publishing results in the peer-reviewed literature. Below is list of manuscripts that have been submitted for publication (abstracts are provided in Appendix I):

Ivan, J. S., G. C. White, and T. M. Shenk. In Review. Using auxiliary telemetry information to estimate animal density from capture-recapture data. Ecology.

Ivan, J. S., G. C. White, and T. M. Shenk. In Review. Using simulation to compare methods for estimating density from capture-recapture data. Ecology.

Ivan, J. S., G. C. White, and T. M. Shenk. In Review. Density, Demography, and Seasonal Movements of Snowshoe Hares in Central Colorado. Journal of Wildlife Management.

SUMMARY

In all stand types and all seasons, snowshoe hare densities were <1.0 hares/ha. During summer, hare densities [±SE] were highest in early seral lodgepole pine (0.20 [0.01] to 0.66 [0.07] hares/ha), lowest in mid-seral lodgepole pine (0.01 [0.04] to 0.03 [0.03] hares/ha), and intermediate in mature spruce-fir (0.01 [0.002] to 0.26 [0.08] hares/ha). During winter, densities were similar among the 3 stand types. Annual survival of hares was highest in mature spruce-fir (0.14 [0.05] to 0.20 [0.07]) and similar between the 2 lodgepole stand types (0.10 [0.03] to 0.16 [0.06]). Stand attributes indicative of dense cover were positively correlated with density estimates and explain relatively more process variance in hare densities than other attributes. These same attributes were not positively correlated with hare survival. Both density and survival of hares in early seral lodgepole stands were positively correlated with the occurrence of similar stands in the surrounding landscape. Recruitment of juvenile hares occurred during all 3 summers in early seral lodgepole stands, 2 of 3 summers in mature spruce-fir stands, and in only 1 of 3 summers in mid-seral lodgepole. Within-season movements of hares were larger during winter than during summer and tended to be larger in early seral lodgepole stands. Hares in both early and mid-seral lodgepole stands tended to make larger movements between seasons than hares in spruce-fir stands, possibly reflecting the variable value of these stands as mediated by snow depth. Based on stand-specific estimates of density, demography, and movement, we conclude that thinned, mid-seral lodgepole stands are less important than mature spruce-fir and small lodgepole stand types. Management for snowshoe hares (and lynx) in central Colorado should focus on maintaining the latter. Given the more persistent nature of spruce-fir compared to early seral lodgepole, and the fact that such stands cover considerably more area, mature spruce-fir may be the most valuable stand type for snowshoe hares in the region.

The estimator we developed is based on a modified Huggins closed capture estimator. It directly accounts for lack of geographic closure (animals moving on and off of the sampling grid during the sampling period) using telemetry data, and this auxiliary information is used to compute estimates of density. Contrary to other approaches, this method is free from assumptions regarding the distribution of animals on the landscape, the stationarity of their home ranges, and biases induced by abnormal movements in response to baited detectors. The estimator is freely available in Program MARK. We found that our approach performed best across most combinations of capture probabilities, sampling occasions, true densities, and home range configurations, and performance was unaffected by home range shape. Spatially explicit capture-recapture methods outperformed “mean maximum distance moved” (MMDM) estimators in nearly all comparisons and may be preferable to our telemetry estimator at low capture probabilities, but performance varied with home range configuration. MMDM estimators exhibited substantial positive bias for most simulations, but performance improved for elongated or infinite home ranges.

LITERATURE CITED

Ivan, J. S. 2011. Density, demography, and seasonal movement of snowshoe hares in central Colorado. Dissertation, Colorado State University, Fort Collins, Colorado, USA.

Prepared by ____________________________
Jacob S. Ivan

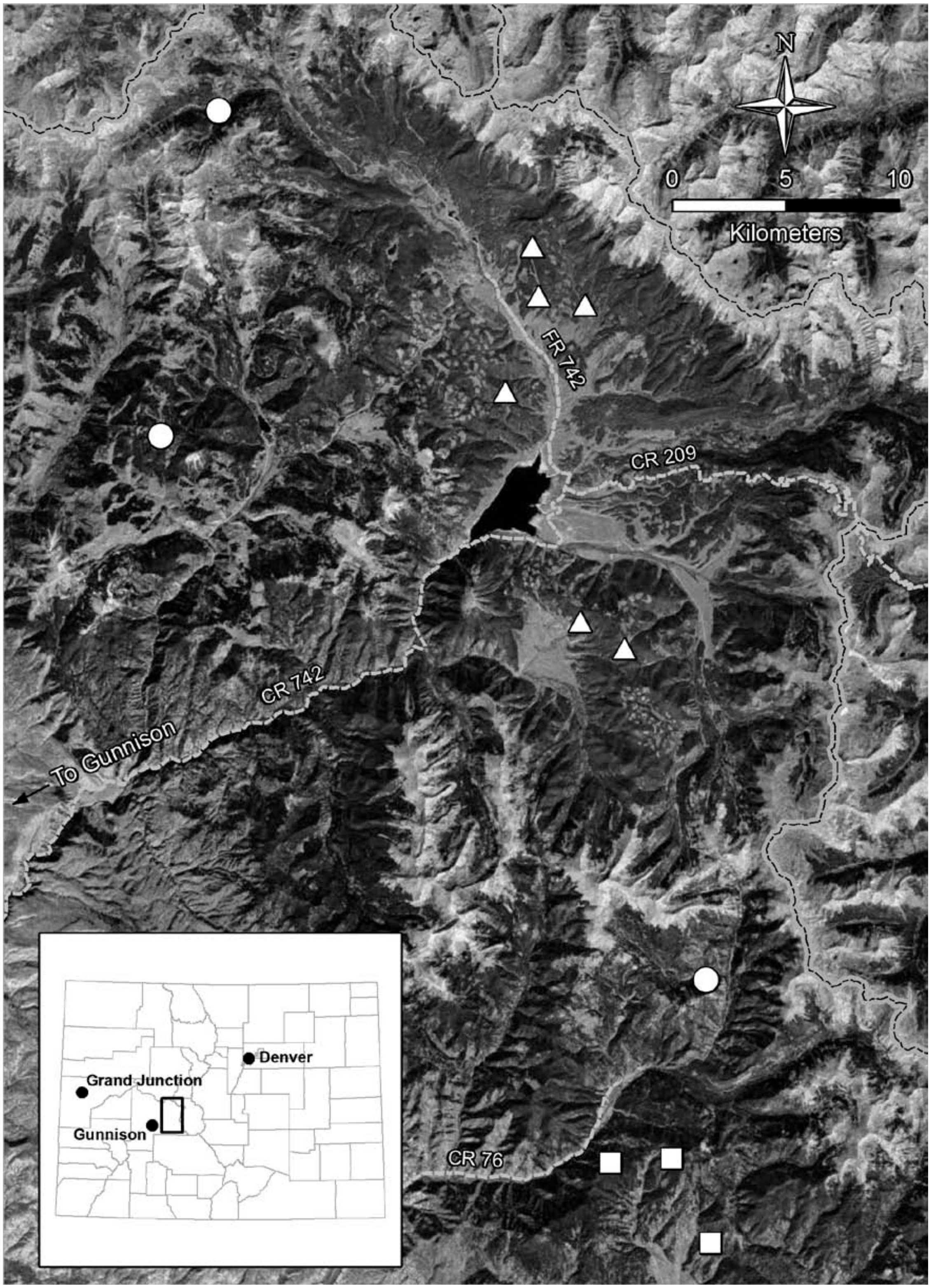


Figure 1. Study area near Taylor Park and Pitkin, central Colorado. We estimated snowshoe density, demography, and movement in 3 late-seral Engelmann spruce/subalpine fir stands (circles), 3 mid-seral lodgepole stands (squares), and 6 early-seral lodgepole stands (triangles) from summer 2006 through winter 2009.

APPENDIX I

PROJECT PAPERS

The following manuscript (referenced here by abstract) is currently in review at the journal *Ecology*.

USING AUXILIARY TELEMETRY INFORMATION TO ESTIMATE ANIMAL DENSITY FROM CAPTURE-RECAPTURE DATA

JACOB S. IVAN, GARY C. WHITE, AND TANYA M. SHENK

ABSTRACT

Estimation of animal density is fundamental to ecology, and ecologists often pursue density estimates using grids of detectors (e.g., cameras, traps, hair snags) to sample animals. However, under such a framework, reliable estimates can be difficult to obtain because animals move on and off of the study site during the sampling session (i.e., the site is not closed geographically). Generally, practioners address lack of geographic closure by a) inflating the area sampled by the detectors based on the mean distance individuals moved between trapping events, or b) invoking hierarchical models in which animal density is assumed to be a spatial point process, and detection is modeled as a declining function of distance to a detector. We provide an alternative in which lack of geographic closure is sampled directly using telemetry, and this auxiliary information is used to compute estimates of density based on a modified Huggins closed capture estimator. Contrary to other approaches, this method is free from assumptions regarding the distribution of animals on the landscape, the stationarity of their home ranges, and biases induced by abnormal movements in response to baited detectors. The estimator is freely available in Program MARK.

The following manuscript (referenced here by abstract) is currently in review at the journal *Ecology*.

USING SIMULATION TO COMPARE METHODS FOR ESTIMATING DENSITY FROM CAPTURE-RECAPTURE DATA

JACOB S. IVAN, GARY C. WHITE, TANYA M. SHENK

Estimation of animal density is fundamental to wildlife research and management, but estimation is often complicated by lack of geographic closure of sampling grids. Contemporary methods for estimating density using mark–recapture data include: 1) approximating the effective area sampled by an array of detectors based on the mean maximum distance moved (MMDM) by animals during the sampling session, 2) spatially explicit capture–recapture (SECR) methods that formulate the problem hierarchically with a process model for animal density and an observation model in which detection probability declines with distance from a detector, and 3) a telemetry estimator (TELEM) that uses auxiliary telemetry information to estimate the proportion of animals on the study site. We used simulation to compare relative performance (percent error) of these methods under all combinations of 3 levels of detection probability (0.2, 0.4, 0.6), 3 levels of occasions (5, 7, 10), and 3 levels of abundance (10, 20, 40 animals). We also tested each estimator using 5 different models for animal home ranges. TELEM performed best across most combinations of capture probabilities, sampling occasions, true densities, and home range configurations, and performance was unaffected by home range shape. SECR outperformed MMDM estimators in nearly all comparisons and may be preferable to TELEM at low capture probabilities, but performance varied with home range configuration. MMDM estimators

exhibited substantial positive bias for most simulations, but performance improved for elongated or infinite home ranges.

The following manuscript (referenced here by abstract) is currently in review at the *Journal of Wildlife Management*.

Density, Demography, and Seasonal Movements of Snowshoe Hares in Central Colorado

JACOB S. IVAN, GARY C. WHITE, TANYA M. SHENK

ABSTRACT

To improve understanding of snowshoe hare ecology in the southern portion of their range, and enhance the ability of agency personnel to manage subalpine landscapes for snowshoe hares (*Lepus americanus*) and lynx (*Lynx canadensis*) in Colorado, we estimated snowshoe hare density, survival, recruitment, and movement in west-central Colorado, USA from July 2006–March 2009. We sampled 3 types of forest stands that purportedly provide good habitat for hares: 1) mature Engelmann spruce (*Picea engelmannii*)/subalpine fir (*Abies lasiocarpa*), 2) early seral, even-aged lodgepole pine (*Pinus contorta*), and 3) mid-seral, even-aged lodgepole pine that had been pre-commercially thinned. In all stand types and all seasons, snowshoe hare densities were <1.0 hares/ha. During summer, hare densities [±SE] were highest in early seral lodgepole pine (0.20 [0.01] to 0.66 [0.07] hares/ha), lowest in mid-seral lodgepole pine (0.01 [0.04] to 0.03 [0.03] hares/ha), and intermediate in mature spruce-fir (0.01 [0.002] to 0.26 [0.08] hares/ha). During winter, densities were similar among the 3 stand types. Annual survival of hares was highest in mature spruce-fir (0.14 [0.05] to 0.20 [0.07]) and similar between the 2 lodgepole stand types (0.10 [0.03] to 0.16 [0.06]). Stand attributes indicative of dense cover were positively correlated with density estimates and explain relatively more process variance in hare densities than other attributes. These same attributes were not positively correlated with hare survival. Both density and survival of hares in early seral lodgepole stands were positively correlated with the occurrence of similar stands in the surrounding landscape. Recruitment of juvenile hares occurred during all 3 summers in early seral lodgepole stands, 2 of 3 summers in mature spruce-fir stands, and in only 1 of 3 summers in mid-seral lodgepole. Within-season movements of hares were larger during winter than during summer and tended to be larger in early seral lodgepole stands. Hares in both early and mid-seral lodgepole stands tended to make larger movements between seasons than hares in spruce-fir stands, possibly reflecting the variable value of these stands as mediated by snow depth. Based on stand-specific estimates of density, demography, and movement, we conclude that thinned, mid-seral lodgepole stands are less important than mature spruce-fir and small lodgepole stand types. Management for snowshoe hares (and lynx) in central Colorado should focus on maintaining the latter. Given the more persistent nature of spruce-fir compared to early seral lodgepole, and the fact that such stands cover considerably more area, mature spruce-fir may be the most valuable stand type for snowshoe hares in the region.

Colorado Division of Parks and Wildlife
July 1, 2011 – June 30, 2012

WILDLIFE RESEARCH REPORT

State of Colorado : Division of Parks and Wildlife
Cost Center 3430 : Mammals Research
Work Package 3001 : Deer Conservation
Task No. 6 : Population Performance of Piceance Basin Mule Deer in Response to Natural Gas Resource Extraction and Mitigation Efforts to Address Human Activity and Habitat Degradation
Federal Aid Project: W-185-R :

Period Covered: July 1, 2011 – June 30, 2012

Authors: C. R. Anderson and C. J. Bishop

Personnel: E. Bergman, T. Bryan, A. Burleson, B. deVergie, D. Finley, M. Fisher, L. Gepfert, C. Harty, D. Johnston, A. Jones, T. Knowles, J. Lewis, H. MacIntyre, J. Matijas, B. Panting, T. Parks, B. Petch, J. Rivale, J. Simpson, S. Singleton, M. Trump, B. Tycz, R. Velarde, L. Wolfe, CPW; E. Hollowed, L. Belmonte, BLM; S. Monsen, Western Ecological Consulting, Inc.; D. Freddy, Hoch Berg Enterprises; T. Graham, Ranch Advisory Partners; M. Wille, T & M Contractors.; P. Lendrum, T. Bowyer, Idaho State University; P. Doherty, J. Northrup, M. Peterson, G. Wittemyer, K. Wilson, G. White, Colorado State University; R. Swisher, S. Swisher, Quicksilver Air, Inc.; D. Felix, Olathe Spray Service, Inc.; L. Coulter, Coulter Aviation. Project support received from Federal Aid in Wildlife Restoration, Colorado Mule Deer Association, Colorado Mule Deer Foundation, Colorado State Severance Tax Fund, EnCana Corp., ExxonMobil Production Co./XTO Energy, Marathon Oil Corp., Shell Petroleum, and Williams Production LMT Co.

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

ABSTRACT

We propose to experimentally evaluate winter range habitat treatments and human-activity management alternatives intended to enhance mule deer (*Odocoileus hemionus*) populations exposed to energy-development activities. The Piceance Basin of northwestern Colorado was selected as the project area due to ongoing natural gas development in one of the most extensive and important mule deer winter and transition range areas in Colorado. The data presented here represent the first 4 pretreatment years of a long-term study addressing habitat improvements and evaluation of energy development practices intended to improve mule deer fitness in areas exposed to extensive energy development. We monitored 4 winter range study areas representing varying levels of development to serve as treatment (Ryan Gulch, North Magnolia, South Magnolia) and control (North Ridge) sites and recorded habitat use and movement patterns using GPS collars (≥5 location attempts/day), estimated overwinter fawn and annual adult female survival, estimated early and late winter body condition of adult females using ultrasonography, and estimated abundance using helicopter mark-resight surveys. We targeted 260 fawns (60–80/study area) and 140 does (30–40/study area) in early December 2011 for VHF and GPS radiocollar attachment, respectively, and 140 does in March 2012 (30–40/study area) for late winter body condition assessment and to increase our GPS radiocollar sample in 1 of the 4 areas (24 in Ryan Gulch) to address neonate

survival. Based on the data collected since January 2008, deer from all areas appear to be in reasonably good condition and have exhibited relatively high survival rates 3 of the 4 years (mean fawn $\hat{S} > 0.65$) with lower winter fawn survival during 2010/11 in 3 of 4 study areas (mean $\hat{S} = 0.49$ excluding North Ridge), and winter range deer densities appear to be stable. More extreme winter conditions during 2010/11 likely contributed to the observed decline in fawn survival rates. Pilot habitat treatments in North and South Magnolia (116 acres total) were completed January 2011 (Anderson and Bishop 2011), another 54 acres were treated January 2012 to assess mechanical treatment methods (hydro-ax, roller-chop, chain), and all required NEPA surveys were completed this summer for the remaining sites (Fig. 6). The Biological Assessment should be completed during September 2012 allowing the remaining 1,030 acres to be treated using hydro-ax this winter. We will continue to collect the various population and habitat use data across all study sites to evaluate the effectiveness of habitat treatments (North and South Magnolia) scheduled for fall/winter 2012–2013 (1,200 acres total). This evaluation will allow us to determine whether it is possible to effectively mitigate development disturbance in highly developed areas, or whether it is better to allocate mitigation dollars toward less or non-impacted areas. In collaboration with Colorado State University, we are also evaluating deer behavioral responses to varying levels of development activity in the Ryan Gulch study area and neonate survival in relation to energy development from all study areas. This will allow us to assess the effectiveness of certain Best Management Practices (BMPs) for reducing disturbance to deer and include neonatal data to other demographic parameters for evaluation of mule deer/energy development interactions. The study is slated to run through at least 2017, and preferably 2019, to adequately measure mule deer population responses to landscape level manipulations.

WILDLIFE RESEARCH REPORT

POPULATION PERFORMANCE OF PICEANCE BASIN MULE DEER IN RESPONSE TO NATURAL GAS RESOURCE EXTRACTION AND MITIGATION EFFORTS TO ADDRESS HUMAN ACTIVITY AND HABITAT DEGRADATION

CHARLES R. ANDERSON, JR and CHAD J. BISHOP

PROJECT NARRITIVE OBJECTIVES

1. To determine experimentally whether enhancing mule deer habitat conditions on winter range elicits behavioral responses, improves body condition, increases fawn survival, or ultimately, population density on mule deer winter ranges exposed to extensive energy development.
2. To determine experimentally to what extent modification of energy development practices enhance habitat selection, body condition, fawn survival, and winter range mule deer densities.

SEGMENT OBJECTIVES

1. Collect and reattach GPS collars to maintain sample sizes for addressing mule deer habitat use and behavior patterns in 4 study areas experiencing varying levels of energy development of the Piceance Basin, northwest Colorado.
2. Estimate early and late winter body condition of adult female mule deer in each of the 4 winter herd segments using ultrasound techniques.
3. Monitor over-winter fawn and annual adult female mule deer survival by daily ground tracking and bi-weekly aerial tracking.
4. Conduct Mark-Resight helicopter surveys to estimate mule deer abundance in each study area.
5. Complete NEPA surveys to allow future habitat treatments for assessing efficacy of habitat improvement projects to mitigate energy development disturbances to mule deer.
6. Initiate neonate survival evaluations to complete demographic parameters for assessing mule deer/energy development interactions.

INTRODUCTION

Extraction of natural gas from areas throughout western Colorado has raised concerns among many public stakeholders and Colorado Parks and Wildlife (CPW) that the cumulative impacts associated with this intense industrialization will dramatically and negatively affect the wildlife resources of the region. Concern is especially high for mule deer due to their recreational and economic importance as a principal game species and their ecological importance as one of the primary herbivores of the Colorado Plateau Ecoregion. Extraction of natural gas will directly affect the potential suitability of the landscape used by mule deer through conversion of native habitat vegetation with drill pads, roads, or noxious weeds, by fragmenting habitat because of drill pads and roads, by increasing noise levels via compressor stations and vehicle traffic, and by increasing the year-round presence of human activities. Extraction will indirectly affect deer by increasing the human work-force population of the region resulting in the need for additional landscape for human housing, supporting businesses, and upgraded road/transportation infrastructure. Additionally, increased traffic on rural roads will raise the potential for vehicle-animal collisions and additive direct mortality to mule deer populations. Thus, research documenting these relationships and evaluating the most effective strategies for minimizing and mitigating these activities will greatly enhance future management efforts to sustain mule deer populations for future recreational and ecological values.

The Piceance Basin in northwest Colorado contains one of the largest migratory mule deer populations in North America and also exhibits some of the largest natural gas reserves in North America. Projected energy development throughout northwest Colorado within the next 20 years is expected to reach about 15,000 wells, many of which will occur in the Piceance Basin, which currently supports over 250 active gas well pads (http://cogcc.state.co.us; Fig. 1). Anderson and Freddy (2008*a*) in their long-term research proposal identified 6 primary study objectives to assess measures to offset impacts of energy extraction on mule deer population performance. During the past 4 years, we have gathered baseline habitat utilization data from GPS-collared deer across the Piceance Basin to allow assessment of mitigation approaches that will be implemented over the next 1–2 years and evaluated for another 4-6 years. We are currently monitoring 1 control area without development (North Ridge), 2 areas with relatively high development activity (0.6–0.9 well pads & facilities/km^2; Ryan Gulch and South Magnolia), and another area with relatively minor development activity (0.1 well pads & facilities/km^2; North Magnolia). In comparison to the un-manipulated control area (North Ridge), the North and South Magnolia areas will receive similar levels of mechanical habitat treatments to evaluate this mitigation strategy relative to differing development intensities, and deer behavior patterns relative to differing development activities in the Ryan Gulch area will be monitored to identify effective Best Management Practices (BMPs) for future application. This progress report describes the previous 4.5 years (Jan 2008–June 2011) of addressing mule deer population performance during the pretreatment phase on 4 winter range herd segments, which includes monitoring habitat selection and behavior patterns of adult female mule deer; spring/summer neonate, overwinter fawn and adult female survival; estimates of adult female body condition during early and late winter, and annual late-winter abundance estimates.

STUDY AREAS

The Piceance Basin, located between the cities of Rangely, Meeker, and Rifle in northwest Colorado, was selected as the project area due to its ecological importance as one of the largest migratory mule deer populations in North America and because it exhibits one of the highest natural gas reserves in North America (Fig. 1). Historically, mule deer numbers on winter range were estimated between 20,000–30,000 (White and Lubow 2002), and the current number of well pads (Fig.1) and projected number of gas wells in the Piceance Basin over the next 20 years is about 250 and 15,000, respectively. Mule deer winter range in the Piceance Basin is predominantly characterized as a topographically diverse pinion pine (*Pinus edulis*)-Utah juniper (*Juniperus osteosperma*; pinion-juniper) shrubland complex ranging from 1,675 m to 2,285 m in elevation (Bartmann and Steinert 1981). Pinion-juniper are the dominant overstory species and major shrub species include Utah serviceberry (*Amelanchier utahensis*), mountain mahogany (*Cercocarpus montanus*), bitterbrush (*Purshia tridentata*), big sagebrush (*Artemisia tridentata*), Gamble's oak (*Quercus gambelii*), mountain snowberry (*Symphoricarpos oreophilus*), and rabbitbrush (*Chrysothamnus* spp.; Bartmann et al. 1992). The Piceance Basin is segmented by numerous drainages characterized by stands of big sagebrush, saltbush (*Atriplex* spp.), and black greasewood (*Sarcobatus vermiculatus*), with the majority of the primary drainages having been converted to mixed-grass hay fields. Grasses and forbs common to the area consist of wheatgrass (*Agropyron* spp.), blue grama (*Bouteloua gracilis*), needle and thread (*Stipa comata*), Indian rice grass (*Oryzopsis hymenoides*), arrowleaf balsamroot (*Balsamorhiza sagittata*), broom snakeweed (*Gutierrezia sarothreae*), pinnate tansymustard (*Descurainia pinnata*), milkvetch (*Astragalus* spp.), Lewis flax (*Linum lewisii*), evening primrose (*Oenothera* spp.), skyrocket gilia (*Gilia aggregata*), buckwheat (*Erigonum* spp.), Indian paintbrush (*Castilleja* spp.), and penstemon (*Penstemon* spp.; Gibbs 1978). The climate of the Piceance Basin is characterized by warm dry summers and cold winters with most of the annual moisture resulting from spring snow melt.

Wintering mule deer population segments we investigated in the Piceance Basin include: North Ridge (53 km^2) just north of the Dry Fork of Piceance Creek including the White River in the northeastern portion of the Basin, Ryan Gulch (141 km^2) between Ryan Gulch and Dry Gulch in the southwestern portion of the Basin, North Magnolia (79 km^2) between the Dry Fork of Piceance Creek and Lee Gulch in the north-central portion of the Basin, and South Magnolia (83 km^2) between Lee Gulch and Piceance Creek in the south-central portion of the Basin (Fig. 1). Each of these wintering population segments has received varying levels of natural gas development: no development in North Ridge, light development in North Magnolia (0.14 pads & facilities/km^2), and relatively high development in the Ryan Gulch (0.60 pads & facilities/km^2) and South Magnolia (0.86 pads & facilities/km^2) segments (Fig. 1). Among the 4 study areas, North Ridge will serve as an unmanipulated control site, Ryan Gulch will serve to address human-activity management alternatives (BMPs) that benefit mule deer exposed to energy development, and North and South Magnolia will serve to address the utility of habitat treatments intended to enhance mule deer population performance in areas exposed to light (North Magnolia) and heavy (South Magnolia) energy development activities.

METHODS

Tasks addressed this period included mule deer capture and collaring efforts, monitoring overwinter fawn and annual adult female survival, estimating adult female body condition during early and late winter using ultrasonography, estimating mule deer abundance applying helicopter mark-resight surveys, working with BLM to complete NEPA surveys to proceed with mechanical habitat treatments fall/winter 2012, and initiation of evaluating neonate survival in developed and undeveloped landscapes. We employed helicopter net-gunning techniques (Barrett et al. 1982, van Reenen 1982) to capture 60–80 fawns and 30–40 adult females during early December 2011 and early March 2012 in each of the 4 study areas. Once netted, all deer were hobbled and blind folded. Fawns were weighed, radio-collared and released on site, and adult females were transported to localized handling sites for recording body measurements and fitted with GPS collars (30–40/area during December 2011, primarily recaptures during March 2012; 5 or 24 fixes/day; G2110D, Advanced Telemetry Systems, Isanti, MN, USA) and released. To provide direct measures of decline in overwinter body condition, 30 does were recaptured in each study area that were captured the previous December; 24 uncollared does were also captured in Ryan Gulch to achieve a desired sample size of 30/study area for monitoring neonate survival. Fawn collars were spliced and fitted with rubber surgical tubing to facilitate collar drop between mid-summer and early autumn, and GPS collars were supplied with timed drop-off mechanisms scheduled to release early in April of the year following deployment. All radio-collars were equipped with mortality sensing options (i.e., increased pulse rate following 4–8 hrs of inactivity).

Mule Deer Habitat Use and Movements

We downloaded and summarized data from GPS collars deployed December 2010 following collar drop and retrieval in early April 2012. GPS collars deployed maintained the same fix schedule of attempting fixes every 5 hours except in Ryan Gulch where fix rates were programmed for 1/hour to increase resolution of GPS data for evaluation of deer behavior patterns in relation to differing development activities. We plotted deer locations and recorded timing and distance of spring and fall 2011 migrations for each study area. Mule deer winter concentration areas were created using composite GPS data (March 2010 through April 2011 from all deer; 5 location attempts/day) from each study area and mapped in ArcGIS (ver. 9.3) using Spatial Analyst (kernel probability density functions separated by quantiles). Mule deer resource selection analyses are pending completion of high resolution habitat data layers currently being developed by BLM.

Mule Deer Survival

Mule deer mortality monitoring consisted of daily ground telemetry tracking and aerial monitoring approximately every 2 weeks from fixed-wing aircraft on winter range and bi-weekly aerial monitoring on summer range. Once a mortality signal was detected, deer were located and necropsied to assess cause of death. We estimated weekly survival using the staggered entry Kaplan-Meier procedure (Kaplan and Meier 1958, Pollock et al. 1989). Capture-related mortalities (any mortalities occurring within 10 days of capture) and collar failures were censored from survival rate estimates. We estimated survival rates from 1 July 2011 through 30 June 2012 for adult females and from early December 2011–mid June 2012 for fawns.

Adult Female Body Measurements

We applied ultrasonography techniques described by Stephenson et al. (1998, 2002) and Cook et al. (2001) to measure maximum subcutaneous rump fat (mm), loin depth (longissimus dorsi muscle, mm), and to estimate % body fat. We estimated a body condition score (BCS) for each deer by palpating the rump (Cook et al. 2001, 2007, 2009). We examined differences ($P < 0.05$) in nutritional status among study areas and between years using a two-sample *t*-test. We considered differences in body condition meaningful when mean rump fat or % body fat differed statistically between comparisons. Other body measurements recorded included pregnancy status (pregnant, barren) via blood samples, weight (kg), chest girth (cm), and hind-foot length (cm).

Abundance Estimates

We conducted 4 (North Ridge, North Magnolia) or 5 (Ryan Gulch, South Magnolia) helicopter mark-resight surveys (2 observers and the pilot) during late March/early April, 2012 to estimate deer abundance in each of the 4 study areas. We delineated each study area from GPS locations collected on winter range during the first 3 years of the study (Jan 2008 through April 2011). Two aerial fixed-wing telemetry surveys/study area were conducted during helicopter mark-resight surveys to determine which marked deer were within each survey area, and we confirmed adult female locations during surveys from GPS data acquired April 2012. We delineated flight paths in ArcGIS 9.3 prior to surveys following topographic contours (e.g., drainages, ridges) and approximating 500–600 m spacing throughout each study area; flight paths during surveys were followed using GPS navigation in the helicopter. Two approximately 12 x 12 cm pieces of Ritchey livestock banding material (Ritchey Livestock ID, Brighton, CO USA) were uniquely marked using color, number, and symbol combinations and attached to each radio-collar to enhance mark-resight estimates. Each deer observed during surveys was recorded as mark ID#, unmarked, or unidentified mark.

We used program MARK (White and Burnham 1999), applying the immigration-emigration mixed logit-normal model (McClintock et al. 2008), to estimate mule deer abundance and confidence intervals. For mark-resight model evaluations, we examined parameter combinations of varying detection rates with survey occasion and whether individual sighting probabilities (i.e., individual heterogeneity) were constant or varied ($\sigma^2 = 0$ or $\neq 0$). Model selection procedures followed the information-theoretic approach of Burnham and Anderson (2002).

RESULTS AND DISCUSSION

Deer Captures and Survival

The helicopter crew captured 264 fawns and 138 does in Dec and Jan 2011 and 142 does during March 2012. Seventeen fawn mortalities (6.4%; ultimate cause = 6 capture myopathy, 10 predation, 1 vehicle collision) occurred within the 10 day myopathy period. Doe mortalities totaled 5 (3.1%; ultimate cause = 4 capture myopathy, 1 vehicle collision) and 7 (4.9%; all capture myopathy) within 10 days of the Dec and Jan and March capture periods, respectively. Mortality rates 10 days post capture have varied between 2–3% for fawns and 0–3% for does since Jan 2008, but were higher this year. Dry conditions and abnormally high dust from pipeline construction relative to previous years may be related.

Fawn survival from early December 2010 through mid June 2011 was similar ($P > 0.05$) among study areas ranging from 0.60 to 0.75 (Table 1; all areas combined = 0.69, 95% CI = 0.63–0.74, $n = 247$). General comparisons to previous years suggest relatively high fawn survival during winter 2009–2010 and relatively low survival during winter 2010–2011 (Fig. 2), which correlates to some degree to winter severity. Exceptions include North Ridge, which has been stable throughout, and Ryan Gulch where relatively low precision of estimates do not allow statistical discrimination (Fig. 2). Annual adult female survival varied from 0.68 (North Magnolia) to 0.93 (Ryan Gulch; Table 1) this year and was comparable among study areas during 2011/12 and to previous years ($P > 0.05$) with the exception of North Magnolia deer exhibiting lower survival this year than during 2009/10 (Anderson and Bishop 2010) and lower than Ryan Gulch this year. The relatively low adult female survival from North Magnolia may result in declining population trends if low survival persists.

Spring Migration Patterns

Collaboration with Idaho State University to direct a graduate student to address mule deer migration patterns in developed and undeveloped landscapes (funded from energy company contributions) has recently been completed. Two manuscripts have been prepared for publication; one is in review and the second has recently been accepted for publication (Lendrum et al. 2012). In addressing habitat selection during spring migration, Lendrum et al. (2012; Fig. 3) noted that mule deer migrating through the most developed landscapes exhibited longer step lengths (straight line distance between GPS locations) and selected habitats providing greater security cover versus more open areas with increased foraging opportunities through undeveloped landscapes. Migrating deer also selected areas closer to well pads, but avoided roads except in the highest developed areas where road densities may be too high for avoidance without significant deviations from traditional migration routes. These results suggest that deer may avoid disturbance where feasible or increase their rate of travel through highly developed landscapes where the energetic cost of avoidance may be too high.

Mule Deer Body Condition

Early-winter body condition measurements of adult female mule deer December 2011 were higher for deer from Ryan Gulch and North Ridge than previous years ($P < 0.05$), but were comparable for the North and South Magnolia deer ($P > 0.05$; Table 2). Comparisons among study areas in December suggested Ryan Gulch deer were in better condition that the other 3 areas. By late winter, however, body condition declined and deer from all study areas exhibited similar condition (Table 2). Improved condition of deer arriving on winter range was expected in December because of improved moisture conditions during spring and summer 2011. We were surprised that condition of North and South Magnolia deer did not mimic deer from the other 2 study areas, especially since there is summer ranges overlap with North Ridge and North Magnolia and Ryan Gulch and South Magnolia, respectively (Fig. 3). It was also surprising that deer from all study areas did not maintain higher condition by late winter given the mild winter conditions that were evident during 2011–2012, as was the case for North and South Magnolia deer during the mild winter of 2009–2010 (Table 2). Slightly higher late winter condition estimates were evident from all areas compared to 2009 and 2011, but these differences were not statistically significant ($P > 0.05$). December fawn weights were comparable to previous years and among study areas last year, with the exception of Ryan Gulch females which showed improvement over the previous year (Fig. 4). More detailed analyses will be conducted to identify factors potentially attributing to these observations.

Neonate Survival

To complete demographic parameters addressing mule deer–energy development interactions, CPW, Colorado State University, and ExxonMobil Production entered into a collaborative agreement to investigate neonate mule deer survival in developed and undeveloped landscapes (funded by ExxonMobil Production Co.). Mark Peterson (GRA) and Paul Doherty (CSU professor) will be assisting with this research, which began March 2012 and will continue for 3 years. To initiate this component of the study, we targeted 30 adult female mule deer/study area to receive Vaginal Implant Transmitters (VITs) during March 2012. Pregnancy rates during March were normal ranging from 96% to 98%/study area (n = 28–46/area). March fetal counts ranged from 1.54 in South Magnolia to 1.92 in North Magnolia. We located 100 does with VITs and 97 neonates at parturition sites, with 85 neonates receiving radiocollars. Neonate survival will be monitored from June through December each year and compared among study areas relative to energy development activities.

Mule Deer Population Estimates

Mark-resight models that best predicted abundance estimates (lowest AIC_c; Burnham and Anderson 2002) exhibited variable sightability across surveys (P_t) for all study areas and homogenous individual sightability ($\sigma^2 = 0$) for North Ridge and South Magnolia deer and variable individual sightability ($\sigma^2 \neq 0$) for North Magnolia and Ryan Gulch deer. North Ridge exhibited the highest deer density (18.3/km^2), with comparably lower deer densities in the other 3 areas (7.4–9.2/km^2; Table 3, Fig. 5). Populations appear stable over the 4 year monitoring period exhibiting annual variation less than the error around point estimates, with the exception of North Magnolia which exhibited a positive increase in 2011 from the previous 2 years (Fig. 5). Abundance estimates from 2012 were similarly precise from all 4 study areas with the mean Confidence Interval Coefficient of Variation (CICV) ranging from 0.13–0.17.

Magnolia Habitat Treatments

In proceeding with mule deer habitat improvements in heavy (South Magnolia) and light developed areas (North Magnolia), we completed pilot habitat treatments in January 2011 (116 acres total; Anderson and Bishop 2011) and January 2012 (54 acres) to assess mechanical treatment methods (hydro-ax, roller-chop, chain). All required NEPA surveys were completed this summer for the remaining sites (Fig. 6). The Biological Assessment should be completed by September 2012, allowing the remaining 1,030 acres to be treated using hydro-ax during fall–winter 2012–2013. Vegetation response in the pilot treatment sites was promising by fall 2011 (Fig. 6), likely due to the moist conditions present during the previous spring and summer. Dryer conditions this spring inhibited a similar response, but treatments completed last January exhibited surprisingly good grass and forb growth; shrub response wasn’t as vigorous as the previous year. All expenses addressing these habitat treatments will be covered through a Wildlife Management Plan agreement between CPW and ExxonMobil Production/XTO energy.

SUMMARY AND COLLABORATIONS

The goal of this study is to investigate habitat treatments and energy development practices that enhance mule deer populations exposed to extensive energy development activity. The information presented here provides data describing mule deer population parameters from the first 4.5 years of the pre-treatment period of a long-term study intended to address how mule deer react to landscape scale habitat and human activity modifications. The pretreatment period will continue through this fall to provide baseline data to compare against intended improvements in habitat conditions and evaluation of concentration and/or reduction in human development activities. Post-treatment monitoring will continue for 4–6 years to provide sufficient time to measure how deer respond to these changes. Based on the data collected thus far, deer from all areas appear to be in reasonably good condition and are exhibiting expected survival rates relative to changes in winter severity. We will continue to collect the various population and habitat use data across all study sites to evaluate the effectiveness of habitat improvements on winter range. This approach will allow us to determine whether it is possible to effectively mitigate development impacts in highly developed areas, or whether it is better to allocate mitigation dollars toward less or non-impacted areas. In a recent project conducted on the Uncomphahgre Plateau, Bergman et al. (2009) found that habitat treatments implemented in pinyon-juniper habitat in undeveloped areas were effective for deer. We are also evaluating deer behavioral responses to varying levels of development activity. This will allow us to assess the effectiveness of certain BMPs for reducing disturbance to wintering mule deer.

Hay field improvements have been completed in the North Magnolia study area by Williams Production LMT Co. to fulfill a Wildlife Management Plan agreement with CPW; elk response has already been evident and mule deer response will continue to be monitored. Additional collaboration with Williams Production LMT Co. has produced a clustered development plan recently implemented in the Ryan Gulch study area and new technologies will be implemented to reduce human activity through remote monitoring of well pads and fluid collection systems. Recent collaboration agreements with ExxonMobil Production Co. and Colorado State University have provided graduate research opportunities to enhance data collection and inference about mule deer–energy development interactions. Additional funding and cooperative agreements will be necessary to sustain this project through completion (at least 2017 and preferably through 2019). We optimistically anticipate the opportunity to work cooperatively toward developing solutions for allowing the nation's energy reserves to be developed in a manner that benefits wildlife and the people who value both the wildlife and energy resources of Colorado.

LITERATURE CITED

Anderson, C. R., Jr. 2009. Population performance of Piceance Basin mule deer in response to natural gas resource extraction and mitigation efforts to address human activity and habitat degradation. Job Progress Report, Colorado Division of Wildlife, Ft. Collins, CO, USA.

Anderson, C. R., Jr., and D. J. Freddy. 2008*a*. Population performance of Piceance Basin mule deer in response to natural gas resource extraction and mitigation efforts to address human activity and habitat degradation. Final Study Plan, Colorado Division of Wildlife, Ft. Collins, CO, USA.

Anderson, C. R., Jr., and D. J. Freddy. 2008*b*. Population performance of Piceance Basin mule deer in response to natural gas resource extraction and mitigation efforts to address human activity and habitat degradation—Stage I, Objective 5: Patterns of mule deer distribution & movements. Pilot Study, Colorado Division of Wildlife, Ft. Collins, CO, USA.

Anderson, C. R., Jr., and C. J. Bishop. 2010. Population performance of Piceance Basin mule deer in response to natural gas resource extraction and mitigation efforts to address human activity and habitat degradation. Job Progress Report, Colorado Division of Wildlife, Ft. Collins, CO, USA.

Anderson, C. R., Jr., and C. J. Bishop. 2011. Population performance of Piceance Basin mule deer in response to natural gas resource extraction and mitigation efforts to address human activity and habitat degradation. Job Progress Report, Colorado Division of Wildlife, Ft. Collins, CO, USA.

Bartmann, R. M. 1975. Piceance deer study—population density and structure. Job Progress Report, Colorado Divison of Wildlife, Fort Collins, Colorado, USA.

Bartmann, R. B., and S. F. Steinert. 1981. Distribution and movements of mule deer in the White River Drainage, Colorado. Special Report No. 51, Colorado Division of Wildlife, Fort Collins, Colorado, USA.

Bartmann, R. M., G. C. White, and L. H. Carpenter. 1992. Compensatory mortality in a Colorado mule deer population. Wildlife Monograph No. 121.

Barrett, M. W., J. W. Nolan, and L. D. Roy. 1982. Evaluation of a hand-held net-gun to capture large mammals. Wildlife Society Bulletin 10:108-114.

Bergman, E. J., C. J. Bishop, D. J. Freddy, and G. C. White. 2009. Evaluation of winter range habitat treatments on over-winter survival and body condition of mule deer. Job Progress Report, Colorado Division of Wildlife, Ft. Collins, USA.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multi-model inference: a practical information-theoretic approach. Second edition. Springer-Verlag, New York, New York, USA.

Cook, R. C., J. G. Cook, D. L. Murray, P. Zager, B. K. Johnson, and M. W. Gratson. 2001. Development of predictive models of nutritional condition for rocky mountain elk. Journal of Wildlife Management 65:973-987.

Cook, R. C., T. R. Stephenson, W. L. Meyers, J. G. Cook, and L. A. Shipley. 2007. Validating predictive models of nutritional condition for mule deer. Journal of Wildlife Management 71:1934-1943.

Cook, R. C., J. G. Cook, T. R. Stephenson, W. L. Meyers, S. M. McCorquodale, D. J. Vales, L. L. Irwin, P. Briggs Hall, R. D. Spencer, S. L. Murphie, K. A. Schoenecker, P. J. Miller. 2009. Revisions of rump fat and body scoring indices for deer, elk, and moose. Journal of Wildlife Management 74:880-896.

Gibbs, H. D. 1978. Nutritional quality of mule deer foods, Piceance Basin, Colorado. Thesis, Colorado State University, Fort Collins, Colorado, USA.

Kaplan, E. L., and P. Meier. 1958. Nonparametric estimation from incomplete observations. Journal of the American Statistical Association 52:457-481.

Lendrum, P. E., C. R. Anderson, Jr., R. A. Long, J. K. Kie, and R. T. Bowyer. 2012. Habitat selection by mule deer during migration: effects of landscape structure and natural gas development. Ecosphere *In press*.

McClintock, B. T., G. C. White, K. P. Burnham, and M. A. Pride. 2008. A generalized mixed effects model of abundance for mark—resight data when sampling is without replacement. Pages 271-289 *in* D. L. Thompson, E. G. Cooch, and M. J. Conroy, editors, Modeling demographic processes is marked populations. Springer, New York, New York, USA.

Pollock, K. H., S. R. Winterstein, C. M. Bunck, and P. C. Curtis. 1989. Survival analysis in telemetry studies: the staggered entry design. Journal of Wildlife Management 53:7-15.

Stephenson, T. R., V. C. Bleich, B. M. Pierce, and G. P. Mulcahy. 2002. Validation of mule deer body composition using in vivo and post-mortem indices of nutritional condition. Wildlife Society Bulletin 30:557-564.

Stephenson, T. R., K. J. Hundertmark, C. C. Swartz, and V. Van Ballenberghe. 1998. Predicting body fat and mass in moose with untrasonography. Canadian Journal of Zoology 76:717-722.

Unsworth, J. W., D. F. Pack, G. C. White, and R. M. Bartmann. 1999. Mule deer survival in Colorado, Idaho, and Montana. Journal of Wildlife Management 63:315-326.

Van Reenen, G. 1982. Field experience in the capture of red deer by helicopter in New Zealand with reference to post-capture sequela and management. Pages 408-421 *in* L. Nielsen, J. C. Haigh, and M. E. Fowler, editors. Chemical immobilization of North American wildlife. Wisconsin Humane Society, Milwaukee, USA.

White, G. C., and K. P. Burnham. 1999. Program MARK: survival estimation from populations of marked individuals. Bird Study 46:120-139.

White, C. C., and B. C. Lubow. 2002. Fitting population models to multiple sources of observed data. Journal of Wildlife Management 66:300-309.

Prepared by____
Charles R. Anderson, Jr., Wildlife Researcher

Table 1. Survival rate estimates ($\hat{S}$) of fawn (3 Dec. 2011–18 June 2012) and adult female (1 July 2011–30 June 2012) mule deer from 4 winter range study areas of the Piceance Basin in northwest Colorado.

Cohort Study area	Initial sample size (*n*)	March doe sample[a] (*n*)	$\hat{S}$ (95% CI)
Fawns			
Ryan Gulch	57		0.600 (0.466–0.734)
South Magnolia	55		0.745 (0.630–0.861)
North Magnolia	56		0.721 (0.601–0.842)
North Ridge	73		0.681 (0.578–0.784)
Adult females			
Ryan Gulch	44	67	0.927 (0.858–0.997)
South Magnolia	30	45	0.903 (0.810–0.997)
North Magnolia	31	49	0.683 (0.536–0.830)
North Ridge	35	60	0.803 (0.698–0.908)

[a] Adult female sample sizes following capture and radio-collaring efforts March, 2012.

Table 2. Mean rump fat (mm), Body Condition Score (BCS[a]), and % body fat (% fat) of adult female mule deer from 4 study areas in the Piceance Basin of northwest Colorado, March and December, 2009–2012. Values in parentheses = SD.

	March 2009			December 2009			March 2010		
Study Area	Rump fat	BCS	% fat	Rump fat	BCS	% fat	Rump fat	BCS	% fat
Ryan Gulch	1.73 (1.78)	2.66 (0.55)	7.54 (1.80)	8.35 (6.36)	4.06 (1.13)	12.96 (4.53)	2.31 (1.44)	2.35 (0.48)	6.69 (1.58)
South Magnolia	1.47 (0.68)	2.50 (0.60)	7.26 (1.82)	10.05 (6.19)	4.07 (1.21)	13.46 (4.96)	3.12 (2.20)	2.64 (0.59)	7.70 (2.01)
North Magnolia	1.30 (0.79)	2.56 (0.68)	6.96 (2.23)	10.67 (5.76)	4.25 (0.96)	13.92 (3.92)	3.15 (2.34)	2.85 (0.53)	8.28 (1.86)
North Ridge	1.57 (1.22)	2.60 (0.56)	7.28 (1.66)	5.25 (5.65)	3.63 (1.11)	11.02 (4.54)	1.77 (1.11)	2.42 (0.49)	6.83 (1.50)

Table 2. Continued.

	December 2010			March 2011			December 2011		
Study Area	Rump fat	BCS	% fat	Rump fat	BCS	% fat	Rump fat	BCS	% fat
Ryan Gulch	7.75 (6.15)	3.34 (0.98)	10.82 (4.32)	1.55 (0.60)	2.53 (0.42)	7.05 (1.20)	13.41 (6.93)	4.21 (1.17)	13.17 (3.64)
South Magnolia	9.85 (6.78)	3.30 (0.61)	11.21 (3.32)	1.65 (0.75)	2.35 (0.50)	6.56 (1.49)	7.53 (4.66)	3.37 (0.76)	9.95 (2.73)
North Magnolia	9.55 (6.49)	2.56 (0.68)	11.65 (4.86)	1.65 (0.67)	2.53 (0.49)	7.06 (1.35)	9.43 (6.41)	3.79 (0.93)	11.15 (3.57)
North Ridge	6.14 (5.29)	3.32 (0.82)	10.32 (3.39)	1.45 (0.76)	2.24 (0.49)	6.24 (1.45)	9.81 (5.81)	3.62 (1.00)	11.22 (3.38)

Table 2. Continued.

	March 2012		
Study Area	Rump fat	BCS	% fat
Ryan Gulch	2.15 (1.44)	2.74 (0.44)	7.22 (1.16)
South Magnolia	1.71 (0.76)	2.58 (0.36)	6.97 (1.12)
North Magnolia	1.87 (0.78)	2.85 (0.33)	7.65 (0.94)
North Ridge	2.24 (1.58)	2.70 (0.35)	7.26 (1.05)

[a] Body condition score taken from palpations of the rump following Cook et al. (2009).

Table 3. Mark-resight abundance (*N*) and density estimates of mule deer from 4 winter range herd segments in the Piceance Basin, northwest Colorado, 27 March–4 April 2012. Data represent 4 helicopter resight surveys from North Ridge and North Magnolia and 5 resight surveys from Ryan Gulch and South Magnolia.

Study area	Mean No. sighted	Mean No. marked	*N* (95% CI)	Density (deer/km^2)
Ryan Gulch	268	24	1,048 (897–1,243)	7.4
South Magnolia	161	25	630 (556–724)	7.6
North Magnolia	267	32	727 (648–840)	9.2
North Ridge	319	34	972 (862–1,113)	18.3

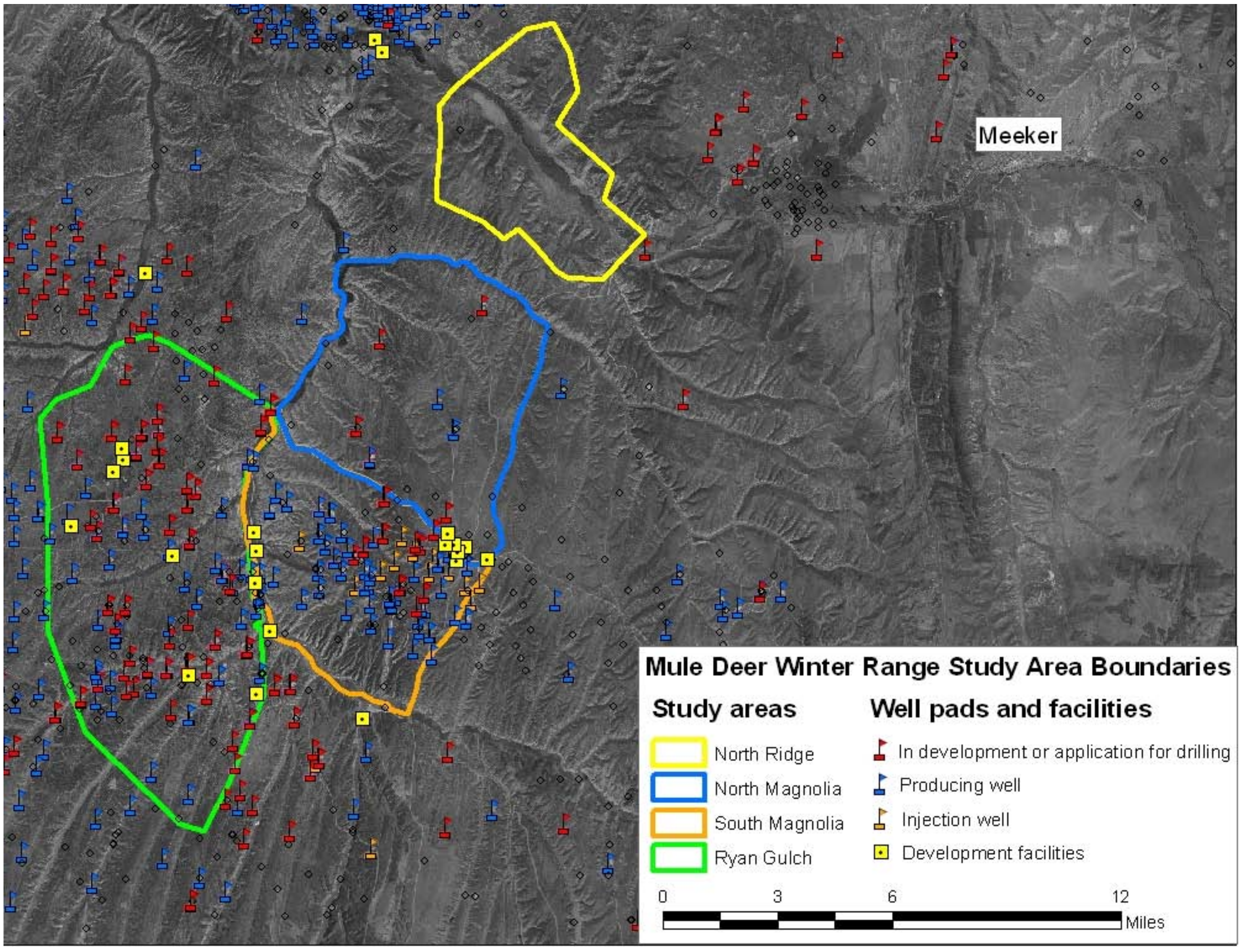


Figure 1. Mule deer winter range study areas relative to active natural gas well pads and energy development facilities in the Piceance Basin of northwest Colorado, summer 2012 (Accessed http://cogcc.state.co.us/ Aug. 8, 2012).

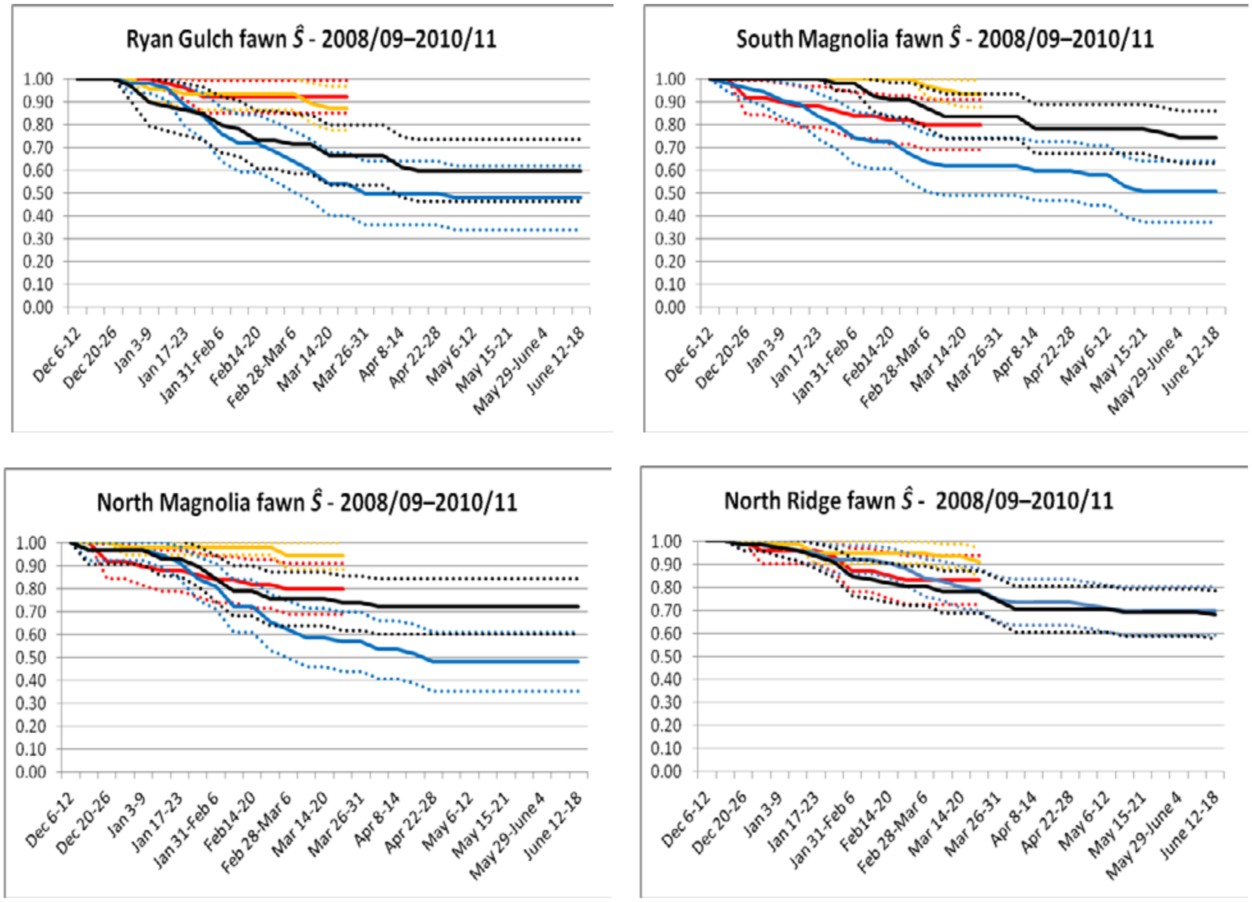


Figure 2. Over-winter (Dec–Mar & June) mule deer fawn survival ($\hat{S}$) from 4 study areas in the Piceance Basin, northwest Colorado, 2008/09 (red lines), 2009/10 (orange lines), 2010/11 (blue lines), and 2011/12 (black lines). Solid lines = $\hat{S}$ and dashed lines = 95% CI. Comparable data among years December–March 2008–2009 and 2009–2010 due to premature collar drop and December–mid-June 2010–2011 and 2011–2012.

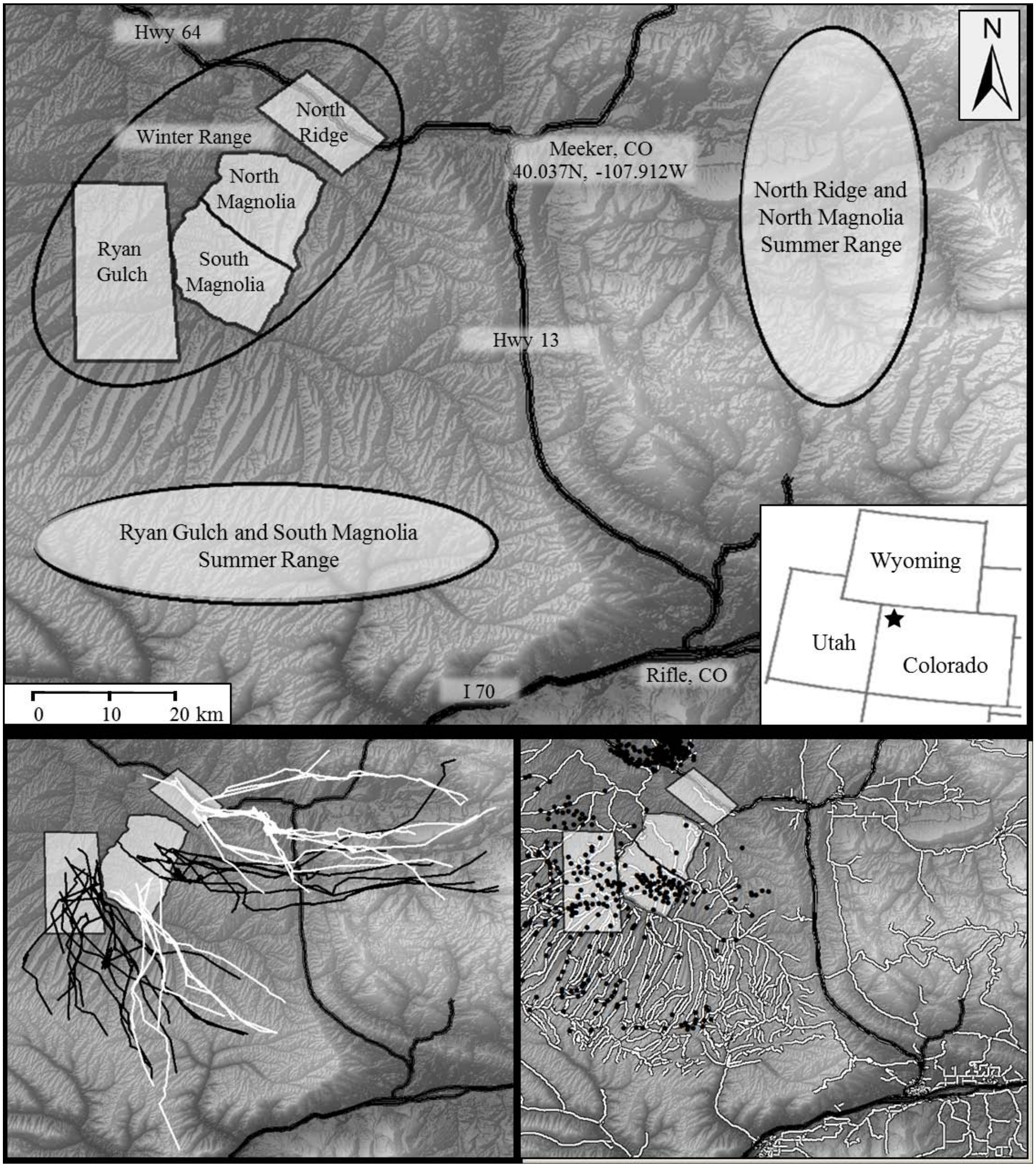


Figure 3. Mule deer study areas in the Piceance Basin of northwestern Colorado, USA (Top), spring 2009 migration routes of adult female mule deer (*n* = 52; Lower left), and active natural-gas well pads (black dots) and roads (state, county, and natural-gas; white lines) from May 2009 (Lower right; from Lendrum et al. 2012).

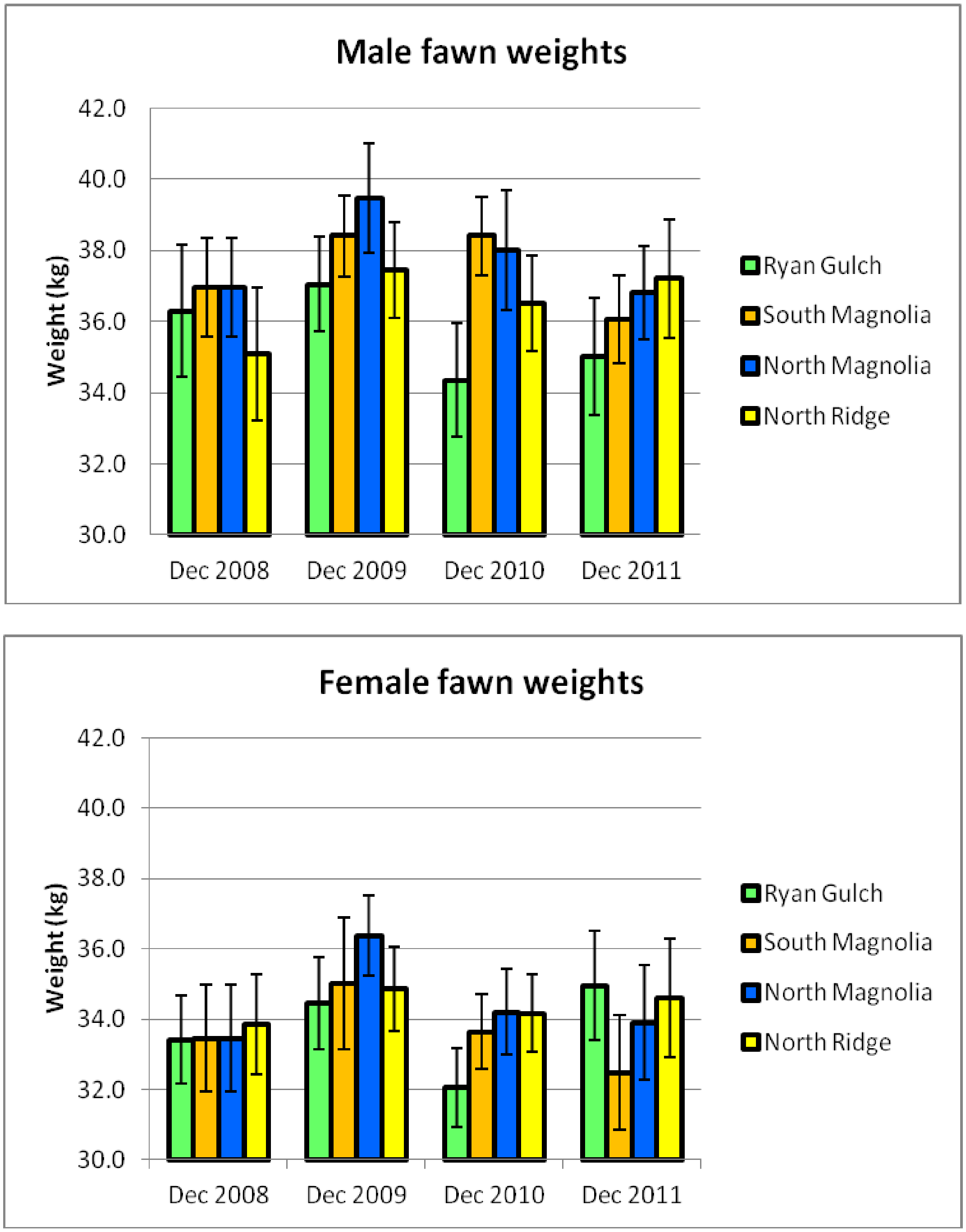


Figure 4. Mean male and female fawn weights and 95% CI (error bars) from 4 mule deer study areas in the Piceance Basin, northwest Colorado, December 2008–2011.

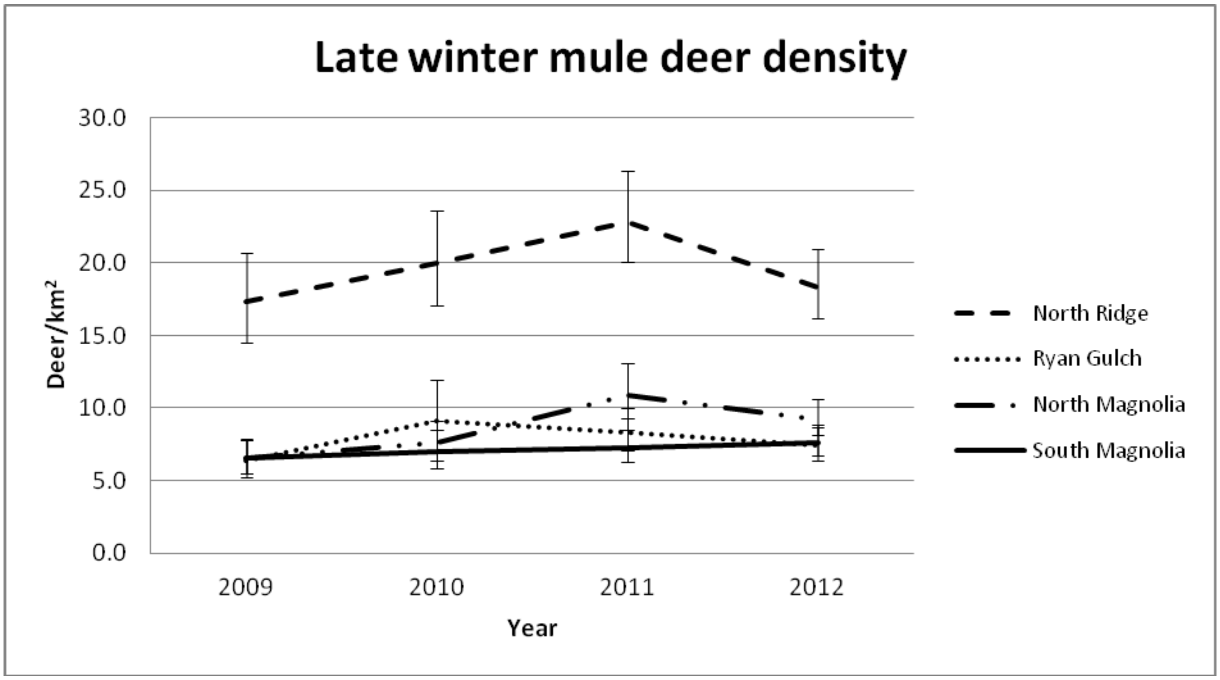


Figure 5. Mule deer density estimates and 95% CI (error bars) from 4 winter range herd segments in the Piceance Basin, northwest Colorado, late winter 2009–2012.

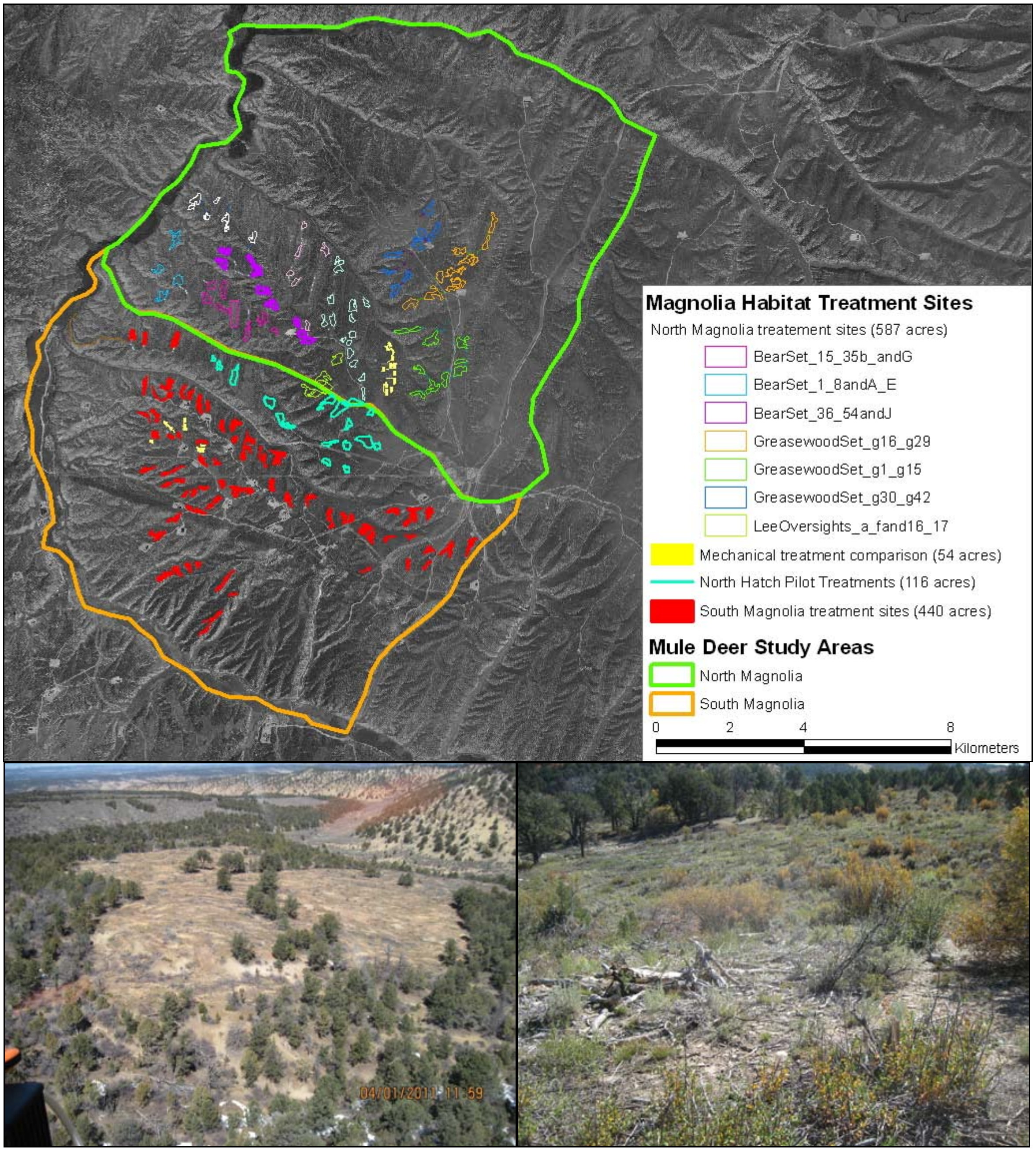


Figure 6. Habitat treatment site delineations in 2 mule deer study areas (600 acres each) of the Piceance Basin, northwest Colorado (Top; cyan and yellow polygons have been completed and remaining sites are scheduled for treatment fall/winter 2012/13). January 2011 hydro-ax treatment-site photos from North Hatch Gulch during April (Lower left, aerial view) and October, 2011 (Lower right, ground view).

Colorado Parks and Wildlife
July 2011 – June 2012

WILDLIFE RESEARCH REPORT

State of:	Colorado	:	Parks and Wildlife
Cost Center:	3430	:	Mammals Research
Work Package:	3001	:	Deer Conservation
Task No.:	2	:	Evaluation of Winter Range Habitat Treatments On Over-winter Survival and Body Condition of Mule Deer
Federal Aid Project No.	W-185-R		

Period Covered: July 1, 2011 - June 30, 2012

Author: E.J. Bergman; project cooperators, C.J. Bishop, D.J. Freddy, G.C. White and P. Doherty

Personnel: C. Anderson, L. Baeten, D. Baker, B. Banulis, J. Boss, A. Cline, D. Coven, M. Cowardin, K. Crane, R. Del Piccolo, B. deVergie, B. Diamond, K. Duckett, S. Duckett, J. Garner, D. Hale, C. Harty, A. Holland, E. Joyce, D. Kowalski, B. Lamont, R. Lockwood, S. Lockwood, D. Lucchesi, D. Masden, J. McMillan, M. Michaels, G. Miller, Mike Miller, Melody Miller, C. Santana, M. Sirochman, T. Sirochman, M. Stenson, R. Swygman, C. Tucker, D. Walsh, S. Waters, B. Watkins, P. Will, L. Wolfe, V. Yavovich, K. Yeager, M. Zeaman, CPW, L. Carpenter - Wildlife Management Institute, D. Felix, L. Felix - Olathe Spray Service, P. Johnston, M. Keech, D. Rivers, J. Rowe, L. Shelton, M. Shelton, R. Swisher, S. Swisher - Quicksilver Air

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

ABSTRACT

Between November 2004 and June 2009 we conducted a five year, multi-area study to assess the impacts of landscape level winter range habitat improvement efforts on mule deer population performance. This study took place on the Uncompahgre Plateau and in adjacent valleys in southwest Colorado. We measured over-winter fawn survival and deer abundance annually on 5 study areas. Four study areas were permanently located, whereas location of the fifth area varied each year to accommodate the variability in habitat treatments over the southern half of the Uncompahgre Plateau. Additionally, on 2 of the study areas we estimated late winter body condition of adult female deer. Compared to results from other research throughout the West, as well as on the Uncompahgre Plateau, survival estimates for 6-month old mule deer fawns were highly variable between areas, and tended to be near published long term averages. Estimated survival rates from this study ranged between 0.359 (SE = 0.0950) and 0.933 (SE = 0.0648). Survival models confirmed that areas that have received advanced habitat treatments have higher fawn survival. Deer abundance on the study areas varied between winters, but in general abundance estimates did not show increasing trends. A slight decrease in density between the first and last years of the study was observed in reference study units. Major fluctuations within abundance and density estimates were attributed to animal movements and winter severity. Based on estimates of total body fat for adult female deer, a distinction between treatment and reference study areas did occur, with body condition parameters indicating that late winter body condition of adult female deer on the treated

study area was higher. Results from overwinter fawn survival work have been submitted to the *Journal of Wildlife Management* for peer review. Final revisions of density and abundance results will be completed during the fall of 2012 and spring of 2013 and submitted for peer-reviewed publication upon completion.

WILDLIFE RESEARCH REPORT

EVALUATION OF WINTER RANGE HABITAT TREATMENTS ON OVER-WINTER SURVIVAL AND BODY CONDITION OF MULE DEER

ERIC J. BERGMAN

P. N. OBJECTIVES

To determine whether mechanical/chemical treatments of native habitat vegetation increases over-winter mule deer fawn survival, adult doe body condition, and localized deer densities on the Uncompahgre Plateau in southwest Colorado and to conduct a simulation based optimization study to determine optimal management strategies of deer under variable environmental, habitat and harvest conditions.

SEGMENT OBJECTIVES

1. Complete all portions of dissertation requirements of PhD through Colorado State University.
2. Complete revision for density and body condition components of the study.
3. Submit and revise 3 chapters of dissertation work as part of the professional peer review process.

INTRODUCTION

A common trend among many terrestrial, mammalian systems is a tendency to cycle between population highs and lows (Jedrzejewska and Jedrzejewski 1998, Krebs et al. 2001, Clutton-Brock and Pemberton 2004). While the true cause of these cycles is likely a merger of habitat quality, weather, disease, predation, hunting, competition and community population dynamics, it is often necessary or intriguing for wildlife managers and ecologists to identify the primary limiting factor to population growth. Without exception, mule deer populations have also demonstrated a tendency to show large fluctuations. Several dramatic declines have been observed since the turn of the 19th century (Connolly 1981, Gill 2001, Hurley and Zager 2004). However, only one period of increase, a general trend during the 1940's and 1950's, has been noted. The most recent and pressing decline took place during the 1990's (Unsworth et al. 1999). Colorado has not escaped these tendencies, with certain parts of the state experiencing population declines by as much as 50% between the 1960's and present time (Gill 2001, B. Watkins personal communication). Primarily due to the value of mule deer as a big game hunting species, wildlife managers' challenges are two-fold: understanding the underlying causes of mule deer population change and managing populations to dampen the effects of these fluctuations.

In Colorado, the role of habitat as the limiting factor for mule deer populations was recently tested. Specifically, the role of forage quality and quantity on over-winter fawn survival was tested using a treatment/reference cross-over design with *ad libitum* pelleted food supplements as a substitute for instantaneous high quality habitat improvements (Bishop et al. 2009). The primary hypothesis behind this research concerned the interaction between predation and nutrition. If supplemental forage treatments improved over-winter fawn survival (i.e., if predation did not prevent an increase), then it could be concluded that over-winter nutrition was the primary limiting factor on populations. As such, nutrition enhancement treatments increased the fawn survival rate by 0.22 (Bishop et al. 2009). This research effectively identified some of the underlying processes in mule deer population regulation, but did not test the effectiveness of acceptable habitat management techniques. Due to the undesirable effects of feeding wildlife (e.g., artificially elevating density, increased potential for disease transmission and cost), a more appropriate technique for achieving a high quality nutrition enhancement needed to be assessed.

We completed a multi-year, multi-area study to assess the impacts of landscape level winter range treatments on mule deer population performance. We conducted the study on the Uncompahgre Plateau and adjacent valleys in southwest Colorado because this area had an active history of habitat treatments that were implemented in part to enhance deer populations. To assess the impacts of habitat treatments on mule deer in these areas, we measured over-winter fawn survival, mule deer density and late winter body condition.

STUDY AREA

At the onset of this study (Bergman et al. 2005), we identified 2 pairs of treatment/reference study areas, stratified into historically known high and low deer density areas. The selection process for these pairs of experimental units followed several strict guidelines:

1) Treatment/reference units could not be further than 10km apart, but needed to have adequate buffer to minimize the movement of animals between the treatment and reference areas.
2) Reference study areas could not have received any mechanical treatment during the past 30 years.
3) Strata were defined by winter range type (all experimental units had to be in pinyon/juniper winter range) and deer density.
4) Treatment units needed to have received mechanical treatment in the past, but also had to be capable of receiving further treatments during the study period.

Each winter a 5th study area was added to increase the level of inference that could be drawn from this study. For each of the 4 winters covering the study period, this 5th study area shifted between 4 randomly selected areas. The treatment history on each of these additional study areas varied, but was representative of what can be expected of typical winter-range treatments. During the first winter of this study, this 5th study area fell on Shavano Valley. Treatments on Shavano Valley were primarily composed of roller-chopping in the higher pinyon/juniper range and were reseeded with browse species. During the second winter of the study, the 5th study area fell on the Colona Tract (~5km^2) of Billy Creek State Wildlife Area (approximately 15km south of Montrose, CO). The treatment history of Colona Tract was primarily composed of brush mowing and chemical control of weeds and dry land fertilization of preferred species. During the third winter of the study, the 5th study area was located at McKenzie Buttes. The treatments at McKenzie Buttes were slightly older (10-15 years) and were also composed of roller-chopping. During the final year of the study, the 5th study area was located at Transfer Road. The treatments available to deer at Transfer Road were younger (1-2 years) and were composed of hydro-ax and some roller-chopping.

The high density treatment area was located on the Billy Creek tract of Billy Creek State Wildlife Area (approximately 20km south of Montrose, CO). The high density reference area was located around Beaton Creek (approximately 15km south of Montrose, CO and approximately 5km north of Billy Creek State Wildlife Area). Both of the high density study areas were located in GMU 65 (DAU D-40). The low density treatment area was located on Peach Orchard Point, on/near Escalante State Wildlife Area (approximately 25km southwest of Delta, CO). The low density reference area was located on Sowbelly and Tatum draws (approximately 25km west of Delta, CO and approximately 8km from Peach Orchard Point). Both of the low density study areas were located in GMU 62 (DAU D-19). All of the other study areas, mentioned above, were also located in GMU 62 (DAU D-19) to the west of Montrose, CO.

METHODS

Twenty-five mule deer fawns were captured and radio-collared in each of the 5 study areas. Fawns were captured via baited drop-nets (Ramsey 1968, Schmidt et al. 1978, Bartmann et al. 1992) and helicopter net-gunning (Barrett et al. 1982, van Reenen 1982) between mid-November and late-

December. To make fawn collars temporary, one end of the collar was cut in half and reattached using rubber surgical tubing; fawns shed the collars after approximately 6 months.

On a daily basis, from December through May, we monitored the radioed fawns in order to document live/death status. This allowed us to determine accurately the date of death and estimate the proximate cause of death. Daily monitoring was done from the ground to maximize efficient collection of mortalities and assessment of cause specific mortality. Weekly aerial telemetry flights were conducted to insure that all deer were heard at least once a week, allowing weekly survival estimates for each study area.

To estimate body condition, an additional 30 adult female deer were captured via helicopter net-gunning and fitted with temporary neckbands in late-February within each of the 2 high density study areas. For body condition work, we relied on methods that employed the use of ultrasonography to estimate total body fat (Stephenson et al. 1998, Cook 2000, Stephenson et al. 2002). Blood samples were also collected for endocrinology and pregnancy tests.

During late winter (early-March) we estimated deer density on each of our study areas. Helicopter based mark-resight techniques were used for density estimation (Gill 1969, Bartmann et al. 1986, Kufeld et al. 1980, Freddy et al. 2004).

Survival analyses were conducted on all years of data. In addition to including individual covariates (fawn sex and mass), we tested the role of habitat treatment history on survival. Estimating survival for study areas took place in several different forms. The simplest form was constant survival where all study areas were pooled and survival was estimated using a single parameter. The second simplest form was to estimate survival for each unique study area (i.e., 8 survival estimates were generated, hereafter “Area”). The remaining model structures allowed study areas to be partitioned according to treatment history. Derivations of these models that included year as either an additive or multiplicative effect were then built.

All survival models were evaluated in program MARK using the known-fate model type with logit link function (White and Burnham 1999). All models were compared using Akaike's Information Criterion corrected for small sample size (Burnham and Anderson 2003). All abundance and density estimates were also computed using program MARK (White and Burnham 1999). Abundance models varied via the process used to estimate the detection probability of deer, but abundance estimates across areas and years were not pooled.

RESULTS AND DISCUSSION

Survival models indicate that advanced landscape treatments do benefit deer. Model structures that incorporate the landscape treatment history of an area outperformed those that did not accommodate treatment history (Table 1, Appendix 1). The top performing model allowed year and week to vary as an additive effect and incorporated fawn mass. Fawn sex did not add much additional strength to any given model. Of particular interest to this study is that models incorporating study area treatment level consistently improved the performance of simpler models that had identical structure, save this one aspect. Not surprisingly, allowing survival rates to vary by year was fundamental for a model to receive any model weight.

Density and abundance estimates were collected during March for all study areas during the last four years of the study. Abundance estimates tended to fluctuate by year in each area, but no discernable trends were observed (Fig. 1, Appendix 1). Fluctuations were likely due to localized winter conditions and the concentrating or diluting of deer on our study areas. Overall, no major changes in density were overwhelmingly evident, although habitat treatments may have arrested population declines that were observed in reference areas (Fig. 1).

Late winter body condition estimates for adult females were consistent during all years of this study. For the two study areas where body condition estimates were measured, multiple linear regression model results reflected the same trends that were observed in survival estimates. A distinction between treatment and reference study areas, based on body condition parameters, indicated that late winter body condition of adult female deer on the treated study area was higher (Table 2, Fig. 2).

Progress towards completion of the requirements for a PhD was also made during the 2011-12 year. As of summer 2012, all coursework needed to meet scholastic requirements has been completed. Additionally, 1 dissertation chapter has been submitted to a peer reviewed journal and 2 additional chapters are in the internal review process.

SUMMARY

Survival rates for mule deer fawns across our study areas and across years ranged between 36% and 93%. Estimates of total deer density across our study areas continued to reflect historical estimates, but annual variation was observed. Overall, a consistent trend of higher survival of fawns and higher body condition of adult female deer was observed in treated study areas, indicating winter range treatments have a positive effect on survival.

LITERATURE CITED

Barrett, M.W., J.W. Nolan, and L.D. Roy. 1982. Evaluation of a hand-held net-gun to capture large mammals. Wildlife Society Bulletin 10:108-114.

Bartmann, R.M., L.H. Carpenter, R.A. Garrott, and D.C. Bowden. 1986. Accuracy of helicopter counts of mule deer in pinyon-juniper woodland. Wildlife Society Bulletin 14:356-363.

——————, G.C. White, and L.H. Carpenter. 1992. Compensatory mortality in a Colorado mule deer population. Wildlife Monographs 121:5-39.

Bishop, C.J., G.C. White, D.J. Freddy, B.E. Watkins and T.R. Stephenson. 2009. Effect of enhanced nutrition on mule deer population rate of change. Wildlife Monographs 172.

Burnham, K.P. and D.R. Anderson. 2003. Model selection and multi-model inference. Springer, New York, USA.

Clutton-Brock, T., and J. Pemberton, editors. 2004. Soay sheep: dynamics and selection in an island population. Cambridge University Press, UK.

Connolly, G. E. 1981. Limiting factors and population regulation. Pages 245-285 *in* O. C. Wallmo, editor. Mule and black-tailed deer of North America. University of Nebraska Press, Lincoln, USA.

Cook, R. C. 2000. Studies of body condition and reproductive physiology in Rocky Mountain Elk. Thesis, University of Idaho, Moscow, USA.

Freddy, D.J., G.C. White, M.C. Kneeland, R.H. Kahn, J.W. Unsworth, W.J. DeVergie, V.K. Graham, J.H. Ellenberger, and C.H. Wagner. 2004. How many mule deer are there? Challenges of credibility in Colorado. Wildlife Society Bulletin 32:916-927.

Gill, R.B. 1969. A quadrat count system for estimating game populations. Colorado Division of Game, Fish and Parks, Game Information Leaflet 76. Fort Collins, USA.

—————— 2001. Declining mule deer populations in Colorado: reasons and responses. Colorado Division of Wildlife Special Report Number 77.

Hurley, M., and P. Zager. 2004. Southeast mule deer ecology - Study I: Influence of predators on mule deer populations. Progress Report, Idaho Department of Fish and Game, Boise, USA.

Jedrzejewska, B., and W. Jedrzejewski. 1998. Predation in Vertebrate Communities: the Białowieża Primeval Forest as a case study. Springer-Verlag, Berlin, Germany.

Krebs, C.J., S. Boutin, and R. Boonstra, editors. 2001. Ecosystem dynamics of the boreal forest: the Kluane project. Oxford University Press, New York, New York, USA.

Kufeld, R.C., J.H. Olterman, and D.C. Bowden. 1980. A helicopter quadrat census for mule deer on Uncompahgre Plateau, Colorado. Journal of Wildlife Management 44:632-639.

Ramsey, C.W. 1968. A drop-net deer trap. Journal of Wildlife Management 32:187-190.

Schmidt, R.L., W.H. Rutherford, and F.M. Bodenham. 1978. Colorado bighorn sheep-trapping techniques. Wildlife Society Bulletin 6:159-163.

Stephenson, T.R., V.C. Bleich, B.M. Pierce, and G.P. Mulcahy. 2002. Validation of mule deer body composition using *in vivo* and post-mortem indices of nutritional condition. Wildlife Society Bulletin 30:557-564.

—————, T. R., K. J. Hundertmark, C. C. Schwartz, and V. Van Ballenberghe. 1998. Predicting body fat and body mass in moose with ultrasonography. Canadian Journal of Zoology 76:717-722.

Unsworth, J.W., D.F. Pac, G.C. White, and R.M. Bartmann. 1999. Mule deer survival in Colorado, Idaho, and Montana. Journal of Wildlife Management 63:315-326.

Van Reenen, G. 1982. Field experience in the capture of red deer by helicopter in New Zealand with reference to post-capture sequela and management. Pages 408-421 *in* L. Nielsen, J. C. Haigh, and M. E. Fowler, editors. Chemical immobilization of North American wildlife. Wisconsin Humane Society, Milwaukee, USA.

White, G.C., and K. P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46 Supplement:120-138.

Prepared by______________________________
Eric J. Bergman, Wildlife Researcher

Table 1. Model selection results of overwinter survival analysis of 6-month old mule deer fawns from different study units in southwestern Colorado. Model selection is based on Akaike's Information Criterion corrected for small sample size (AIC_c). Models were constructed with an intercept (Int) and year (Yr) as a 3-parameter offset. Models could be comprised of effects including year, week, mass, traditional treatments (Trt), advanced treatments (Ad. Trt) and individual study units (Area).

Model #	**Model Structure**	**Δ AIC_c**	**AIC_c Weight**	**Model Likelihood**	***k*[a]**
1	Int + Yr + Week + Mass + Ad. Trt	0.00[b]	0.376	1.00	29
2	Int + Yr + Week + Mass + Sex + Ad. Trt	1.34	0.193	0.51	30
3	Int + Yr + Week + Mass + Trt + Ad. Trt	1.91	0.145	0.38	30
4	Int + Yr + Week + Mass + Sex + Trt + Ad. Trt	3.17	0.077	0.21	31
5	Int + Yr + Week + Mass	3.18	0.077	0.20	28
6	Int + Yr + Week + Mass + Sex	4.66	0.037	0.10	29
7	Int + Yr + Week + Mass + Trt	4.83	0.034	0.09	29
8	Int + Yr + Week + Mass + Area	5.16	0.028	0.08	35
9	Int + Yr + Week + Mass + Sex + Trt	6.39	0.015	0.04	30
10	Int + Yr + Week + Mass + Sex + Area	6.73	0.013	0.03	36

[a] Accounting for parameters is as follows: Int = 1, Yr = 3, Week = 23, Mass = 1, Trt = 1, Ad. Trt = 1, Sex = 1, Area = 7

[b] AIC_c value for the top model was 1404.77

Table 2. Cumulative model weights for body condition response variables from multiple linear regression models for adult female mule deer. Data from southwest Colorado during early March, 2007–2009. Model weights are based on Akaike's Information Criterion corrected for small sample size (AIC_c).

Response	Cumulative Covariate AIC_c Weight					
Variable	Unit	Year	Chest	Age	Foot	Pregnant
%IFBF	0.722	0.823	0.966	0.363	0.293	0.260
TT4	0.998	0.998	0.511	0.998	0.489	0.802
FT4	1.000	1.000	0.412	0.890	0.533	0.827
TT3	0.278	1.000	0.271	0.989	0.633	0.274
FT3	0.265	0.999	0.317	0.982	0.504	0.452

Figure 1. Mule deer density estimates, with 95% confidence intervals, for 8 study units on the Uncompahgre Plateau in southwest Colorado. Northern study units (Sowbelly, Peach and Transfer) are depicted in panel A, whereas southern study units (Shavano, Colona, McKenzie, Buckhorn and BCSWA) are depicted in panel B.

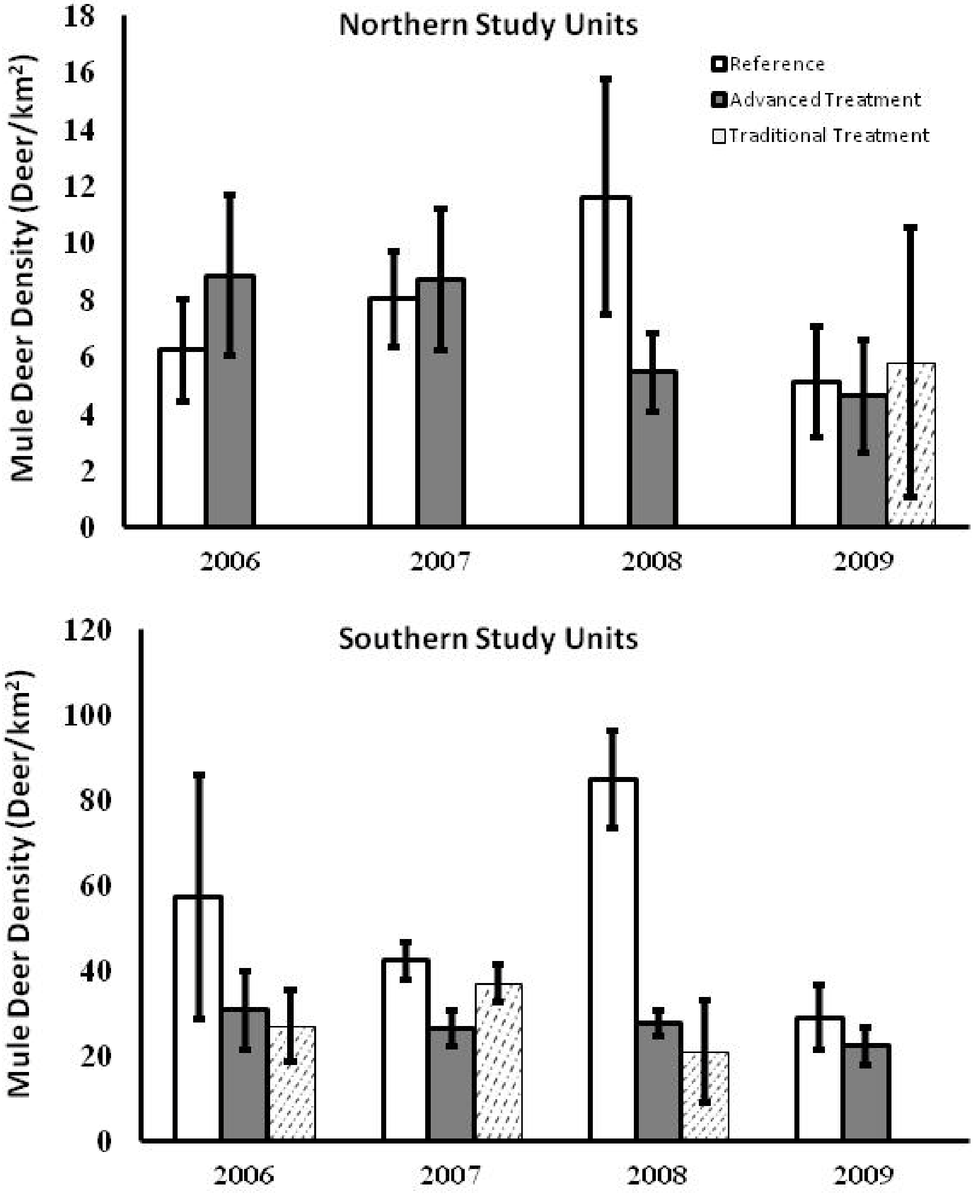
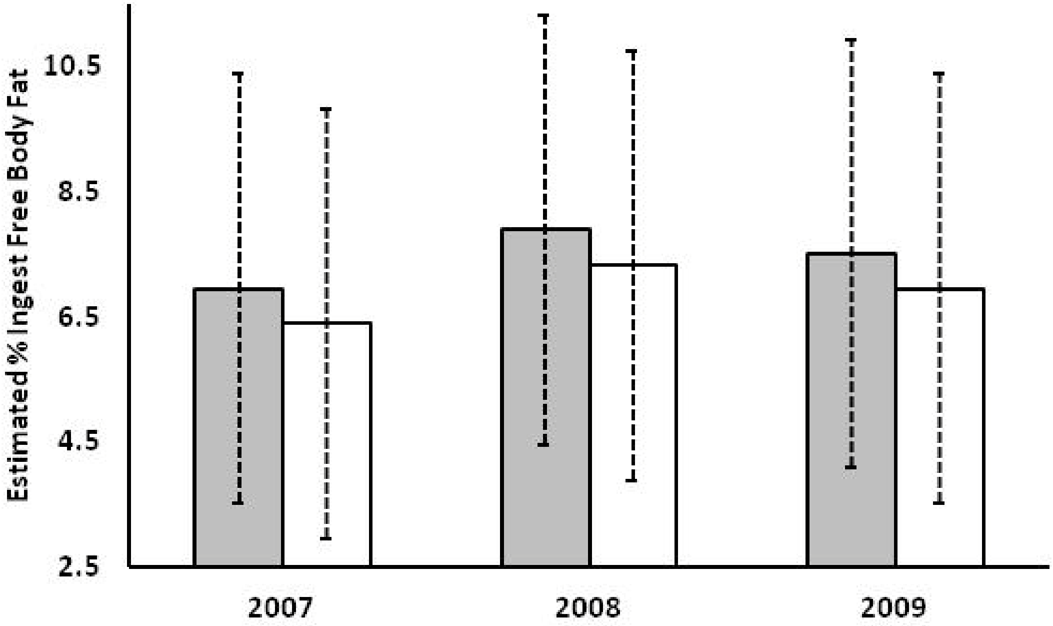


Figure 2. Scaled estimates of late winter percent ingest free body fat (%IFBF), with 95% confidence intervals, for adult female mule deer in southwest Colorado. Solid gray bars reflect estimates for my treatment study area (Billy Cree State Wildlife Area) and white bars reflect estimates for my reference study area (Buckhorn). Estimates and projection intervals were generated according to the model $\widehat{\%IFBF} = -2.514 - 0.562 \times Buckhorn + 0.940 \times Year^{2008} + 0.563 \times Year^{2009} + 0.100 \times Chest$ in which chest girth was held constant at the observed mean of 95.476 cm.



APPENDIX I

ABSTRACTS FROM DISSERTATION CHAPTERS EXPECTED FOR PUBLICATION

The following abstracts have either been submitted to the Journal of Wildlife Management for publication, or they will be submitted during FY 2012-2013.

EFFECT OF HABITAT MANAGEMENT ON OVERWINTER SURVIVAL OF MULE DEER FAWNS IN COLORADO

Eric J. Bergman, Chad J. Bishop, David J. Freddy, Gary C. White, and Paul F. Doherty, Jr.

ABSTRACT

Wildlife managers and ecologists are often compelled to identify the primary limiting factor to population growth in order to facilitate population management. Due to their iconic status and economic value, mule deer (*Odocoileus hemionus*) are not exempt from this need. Habitat management, in the form of mechanical or chemical manipulation of the vegetative landscape, has been utilized as a population management strategy to bolster mule deer populations. Yet evaluation of this strategy in the form of deer population response has been lacking. To address a knowledge gap and to evaluate the effectiveness of habitat management as a deer population management strategy, we conducted a 4-year study that measured the overwinter survival of mule deer fawns on study units that had experienced different levels of habitat management efforts. Mule deer fawns that overwintered on areas that received both a traditional treatment as well as follow-up treatments experienced increased survival ($\hat{S}$= 0.768, SE = 0.0851) over fawns on winter range that had only received traditional treatments or no habitat treatments at all ($\hat{S}$ = 0.675, SE = 0.112). When partitioned into different levels of treatment intensity, mule deer fawns inhabiting winter range that received both traditional treatments and follow-up treatments experienced higher survival ($\hat{S}$ = 0.768, SE = 0.0849) than fawns on units that experienced only traditional treatments ($\hat{S}$ = 0.687, SE = 0.108), which in turn experienced higher survival than fawns in areas that had received no habitat treatments ($\hat{S}$ = 0.669, SE = 0.113). Our study provides evidence supporting the long-held view that habitat management is a viable and economically feasible population management strategy for mule deer in pinyon pine (*Pinyon edulis*) - Utah juniper (*Juniperus osteosperma*) ecosystems.

RESPONSE OF MULE DEER DENSITY TO HABITAT MANAGEMENT IN COLORADO

Eric J. Bergman, Chad J. Bishop, David J. Freddy, Gary C. White, and Paul F. Doherty, Jr.

ABSTRACT

The suite of demands competing for wildlife management funds necessitates direct assessment of management decisions, especially when these decisions have direct costs, as well as tangible opportunity costs. A specific example of such a decision includes habitat management for mule deer (*Odocoileus hemionus*), for which estimating direct effects on abundance has been difficult. However, recent advancements in abundance estimation methodologies have made estimating abundance more possible than in the past. We conducted a mark-resight study that estimated mule deer abundance and density across multiple study units that had been exposed to different intensities of habitat treatments on the eastern slope of the Uncompahgre Plateau and in neighboring drainages of the San Juan mountain range in SW Colorado. Our treatments were comprised on common habitat management techniques including hyrdo-axe and roller-chopper disturbances, as well chemical control of weeds and reseeding of desirable mule deer browse species. Reference study units received no habitat management treatments. Based on model selection strategies, resighting probabilities (range 0.070–0.567) were best modeled as an interactive function of study unit and year, although sampling method proved to also be important in estimating resight probabilities. Abundance estimates across study areas were variable, although annual variation in estimates was greatest in reference study units. Total deer densities varied between 20–84 deer/km^2 in southern study units and 4–12 deer/km^2 in northern study units. A consistent pattern of higher deer density on advanced treatment study units was not observed despite its being the primary hypothesis of the study. We recommend that if population abundance and density are to be used as population response variables, they only be used in tandem with other, more sensitive parameters such as overwinter survival or late winter body condition.

Colorado Division of Parks and Wildlife
July 2011 – June 2012

WILDLIFE RESEARCH REPORT

State of:	Colorado	:	Division of Parks and Wildlife
Cost Center:	3430	:	Mammals Research
Work Package:	3001	:	Deer Conservation
Task No.:	4	:	Development of an Automated Device
		:	for Collaring and Weighing Mule Deer Fawns
Federal Aid Project No.	W-185-R		

Period Covered: July 1, 2011 – June 30, 2012

Authors: C. J. Bishop, M. W. Alldredge, D. P. Walsh, E. J. Bergman, and C. R. Anderson.

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

ABSTRACT

We designed and produced a trap-like device for mule deer that would automatically attach a radio collar to a ≥6-month-old fawn and record the fawn’s weight and sex, without requiring physical restraint or handling of the animal. Our passive collaring device is designed to allow biologists and researchers to radio-collar, weigh, and identify sex of ≥6-month-old mule deer fawns with minimal expense and labor when compared to traditional mule deer capture techniques. This technique should significantly reduce stress that is typically associated with capture and handling and eliminate capture-related mortality. We collaborated with students and faculty in the Mechanical Engineering Department at Colorado State University to produce a conceptual model and early prototype. We then worked with professional engineers at Dynamic Group Circuit Design in Fort Collins, Colorado, to produce a fully-functional prototype of the device. We conducted an extensive field evaluation of the device with free-ranging mule deer during October-March, 2010-11, and January-March, 2012. We successfully collared, weighed, and identified sex of 6 different mule deer fawns across 4 winter range locations along Colorado’s northern Front Range during winter 2010-11. Collars were purposefully made to shed from deer within several weeks or months of being collared. Two fawns were successfully re-collared after they shed the first collars they received. Thus, we observed 8 successful collaring events involving 6 different fawns in 2010-11. Most fawns demonstrated minimal response to collaring events, either remaining in the device or calmly exiting. We successfully collared, weighed, and identified sex of 2 different mule deer fawns in the Piceance Basin of northwest Colorado during February-March 2012. We collared fewer fawns in winter 2011-12 than the previous winter in part because of a shortened evaluation period (i.e., 3 instead of 6 months). Winter conditions were mild overall during 2011-12, which likely contributed to the lower collaring rate since deer had ample foraging options and may not have been as strongly attracted to bait. During 2010-11, certain components of the collaring device failed to function optimally when temperatures dropped below approximately –15° C, while other components did not adequately withstand mule deer use under field conditions. Also, certain behaviors of mule deer when approaching and using the device created circumstances where it was possible to collar the same animal twice, which happened on one occasion. We incorporated a series of device modifications during summer-fall 2011 necessary to address these various issues. The device functioned well under field

conditions during January-March 2012, indicating the modifications were effective. Our automated collaring device allowed mule deer fawns to be remotely collared, weighed, and sexed with minimal or no stress to the animals. However, fawns typically required one or more weeks of exposure to the device before they entered and accessed the bait. This slow acclimation period limited utility of the device when compared to traditional capture techniques used to collar fawns. Future work will focus on additional device modifications and altered baiting strategies that decrease fawn acclimation period, and in turn, increase collaring rates.

WILDLIFE RESEARCH REPORT

DEVELOPMENT OF AN AUTOMATED DEVICE FOR COLLARING AND WEIGHING MULE DEER FAWNS

CHAD J. BISHOP, MATHEW W. ALLDREDGE, ERIC J. BERGMAN, DANIEL P. WALSH, AND CHARLES R. ANDERSON, JR.

P. N. OBJECTIVE

To develop and evaluate a trap-like device for mule deer that will automatically attach a radio collar to a ≥6-month-old deer fawn and record the fawn's weight and sex, without requiring physical restraint or handling of the animal.

SEGMENT OBJECTIVES

1. Evaluate effectiveness and functionality of an automated collaring device for collaring, weighing, and identifying sex of mule deer fawns during winter under free-ranging conditions.

INTRODUCTION

Colorado Parks and Wildlife (CPW) captures and radio-marks 6-month-old mule deer (*Odocoileus hemionus*) fawns each year to support research and management of mule deer. Approximately 240–300 fawns are captured annually to monitor survival among 4–5 populations distributed across western Colorado and an additional 100–350 fawns are captured as part of ongoing research studies. Other state agencies in the western United States capture large numbers of mule deer fawns annually also. Most capture is accomplished with net-guns fired from helicopters (Barrett et al. 1982, van Reenen 1982, Webb et al. 2008), which is becoming increasingly expensive (i.e., >$500 per captured deer). Also, net gunning is inherently dangerous with a small market, which at times limits availability of contractors. Drop nets (Ramsey 1968, Schmidt et al. 1978), clover traps (Clover 1956), drive nets (Beasom et al. 1980), and darting (Wolfe et al. 2004) are used occasionally in the western United States to capture deer, but these techniques can be time consuming and labor intensive. Many biologists lack time and resources given other job requirements to conduct such capture operations for any length of time. The increasing cost of helicopter net-gun capture coupled with increasing demand for capturing and radio-collaring 6-month-old fawns has created a need for another capture alternative. Specifically, there is need for a capture technique that is relatively inexpensive to employ considering both operating and personnel costs.

In response to CPW's capture needs, we conceived the idea of an automated marking device for ≥6-month-old deer fawns that would attach a radio collar and record weight and sex without physically restraining the animal or requiring handling. The idea of automatically attaching radio transmitters to animals is not new, although to our knowledge, there are no proven methods or devices for use on deer or other ungulates. Even a relatively expensive trap or device (e.g., >$5,000 ea.) would reduce CPW's capture costs assuming the device could be reused over time with few maintenance expenses. Such a device would enable seasonal wildlife technicians or graduate students to radio-collar samples of deer fawns independently or with little assistance from researchers and biologists because no animal handling would be required. We want the device to record weight and sex because these variables are useful covariates in survival analyses and are typically measured when fawns are captured and handled.

A passive marking device would minimize animal stress associated with capture and should not cause capture-related mortality. The large-mammal capture techniques described above place considerable, temporary stress on animals as part of netting and handling. Roughly 2-3% of animals typically die from capture-related injuries or stresses under routine capture conditions. Thus, successful development of a passive marking system would reduce CPW's operating expenses and improve animal welfare. Therefore, we designed, produced, and evaluated an automated device for collaring, weighing, and identifying sex of mule deer fawns during winter under free-ranging conditions.

STUDY AREA

We worked with captive deer at the Foothills Wildlife Research Facility (FWRF) in Fort Collins, Colorado, when designing the device and evaluating initial prototypes. We conducted subsequent evaluations of the collaring device with free-ranging deer in various field locations. During 2010-11, we conducted field evaluations with free-ranging deer at 5 sites along Colorado's northern Front Range: 1) Horsetooth Reservoir, west of Fort Collins, private land 2) Masonville, southwest of Fort Collins, private land, 3) Red Feather, northwest of Fort Collins, private land, 4) Hall Ranch, west of Lyons, Boulder County Parks and Open Space, and 5) Heil Valley Ranch, southwest of Lyons, Boulder County Parks and Open Space. During 2012, we conducted field evaluations with free-ranging deer at Hall Ranch (Jan) and in the Piceance Basin southwest of Meeker, Colorado (Feb-Mar).

METHODS

We identified detailed specifications to guide the design and development of an automated collaring device and sought assistance from Colorado State University's Mechanical Engineering Department. The collaring device became a senior design project for 6 CSU engineering students during the 2008-09 school year. We met with the students weekly and provided them a materials budget of $10,000 to produce a prototype device. We conducted staged evaluations of device components during the year by working with captive deer at FWRF. We also conducted limited evaluations with free-ranging deer during spring 2009. Field evaluations focused primarily on how deer utilized and interacted with the device to guide subsequent design and development decisions. We documented utilization and interactions using direct observation and motion-sensor digital cameras. We relied exclusively on digital cameras when we were not on-site during an evaluation. Automation of the collaring device was disabled any time we were not present to prevent any potential harm to deer.

Following preliminary field evaluations, we refined our design specifications and developed a contract with Dynamic Group Circuit Design (DGCD), Fort Collins, Colorado, to produce a fully-functional prototype device. We routinely met with electrical engineers from DGCD, and a mechanical engineer subcontracted by DGCD, during 2009-10. These meetings ensured that our device specifications were being satisfactorily met from both engineering and deer biology perspectives. Working with DGCD, we produced a fully-functional prototype device in 2010 that met our design specifications as set forth in the contract.

The prototype device comprises an aluminum cage attached to a bait compartment (Fig. 1). Deer enter the device through an adjustable opening at the front of the cage. The adjustable opening can be used to deter entry of larger animals by adjusting both width and height. The sides of the cage comprise one-way gates that prevent entry into the device but allow an animal to exit the device at any point. The bait compartment is accessed through an opening positioned at the rear of the cage. An expandable radio collar is placed in this opening by extending it around four rectangular, aluminum plates that hold the collar in the fully-expanded position (Fig. 2). Radio collars are made expandable by attaching springs to each end of the transmitter; that is, springs are used in place of belting on standard radio collars. Clear plexiglass separates the cage from the bait compartment to maximize visibility. A deer is able to extend its head and neck through the expanded radio collar positioned in the rear opening to access the bait in the bait compartment, which is the only access point to the bait (i.e., it cannot be reached by an animal outside of the device). The floor of the cage is a scale that continuously records weight and informs device operation. Only animals in a specified weight range can be collared, which allows the user to target fawns and avoid collaring adult deer. Specifically, the mechanism that releases the collar around a deer's neck will not trigger when an animal is too heavy or too light. Also, an actuator moves a plexiglass plate into the space between the rear cage opening and the bait pan, preventing animals outside of the weight range from accessing the bait. Shortly after a non-target animal exits the device, the collar release mechanism is once again able to be released (when triggered) and the actuator lowers the plexiglass plate so that the bait is accessible. To prevent an animal from being collared twice, a loop antenna is placed around the entrance to the cage and connected to a radio frequency identification (RFID) reader. All collars used with the device include a small RFID transponder sewn into the collar material. If a previously-collared fawn enters the cage, the RFID transponder is detected, which in turn prevents the collar from being released and activates the actuator to block access to the bait.

If a deer enters the cage that is in the specified weight range and has not been previously collared, the collar will release around the deer's neck once it accesses the bait. The collar release is triggered when a deer's head breaks an infrared beam positioned immediately above the bait pan. The collar is released by activating a solenoid, which in turn releases a lever or trigger that causes the upper 2 aluminum plates holding the expanded collar in place to collapse (Figs. 3 and 4). The collar is then situated around the deer's neck. In 2011, we replaced the release lever with an archery caliper release in an attempt to improve the release mechanism. When the collar is released, 2 different cameras are immediately activated to take a series of 3 photographs each. One camera is positioned in the back of the bait compartment and set to take a close-up photo of the top of the deer's head. The second camera is positioned in the floor of the cage and set to take a photo of the deer's abdomen and groin. These cameras are activated only when a collar is released and facilitate determination of deer sex. In 2011, we removed the floor camera after determining it was not necessary or effective for identifying deer sex. Last, when a collar is released, the device records and stores the weight of the deer.

An external computer can be hooked up to the device to change program settings, remotely operate the device, and upload weight data. The device is powered by a 12 volt battery that must be recharged every 2-3 days assuming continuous operation. DGCD prepared a user's manual that explains device operation and detailed schematics to allow future production.

We evaluated effectiveness of the device in the field during October-March 2010-11 and January-March 2012. Initially, we only set the device with a collar in place when we were present and able to directly observe deer interactions with the device. After collaring several animals in this manner and troubleshooting problems with the device, we set the device to operate remotely without an observer on-site, which is how it was intended to be used.

RESULTS AND DISCUSSION

2010-2011 Field Evaluation

We began baiting sites at Horsetooth Reservoir and Masonville on October 21, 2010, to attract deer for evaluating the device. We baited sites with alfalfa hay, apple pulp, dried fruit, and cereal. We baited several other sites briefly but discontinued baiting due to lack of deer use. Deer immediately responded to bait at Horsetooth Reservoir and began accessing the bait daily. On October 26, we placed the collaring device on site and began encouraging deer to walk into the device by placing bait on the scale inside the cage. On October 29, we documented a deer accessing the bait pan within the bait compartment for the first time. In the following weeks, we continued to periodically document deer entering the device and accessing the bait pan, although malfunctioning of the device prevented deer from being collared. One malfunction occurred because an electrical signal emitted from a camera placed at the entry of the device interfered with the RFID reader, which ultimately prevented fawns from being collared. It took roughly a week to diagnose the problem, which was corrected by simply removing the camera from the entry of the device. This particular camera was not wired into the device and was not critical to device functioning. We deemed that this camera was unnecessary and would be more useful if placed approximately 5 meters away from the trap to better document deer use and behavior. A second malfunction occurred because the scale did not have adequate support underneath and touched the ground, thereby giving inaccurate weight readings, which also prevented deer from being collared. We corrected this particular problem by welding an aluminum frame to better support the scale. Once these problems were corrected and other adjustments were made, we remotely collared our first fawn (female) on November 17, 2010. The fawn showed little reaction to the collaring event, calmly exiting the trap shortly after receiving the collar. The fawn's weight and sex were successfully recorded. Sex was positively confirmed based on a photograph of the fawn's head taken by the camera positioned in the bait compartment.

We continued to monitor the device at Horsetooth Reservoir because there were adequate numbers of uncollared fawns in the area. However, we continued to encounter various problems with the device that affected functionality. Most notably, the collar release mechanism began failing to release the collar when a fawn was in position. We quickly determined that device controls were working properly and that an electrical signal was successfully being sent to the solenoid when an uncollared fawn was in position accessing the bait. The source of the problem was a mechanical failing associated with the release mechanism itself. When an expanded collar was in place (i.e., in a fully-expanded state), the tension of the collar sometimes prevented the release lever from moving enough to release the aluminum plates holding the collar in position. Once aware of the problem, we began making adjustments to the release mechanism to improve its functionality. Another problem we identified was that fawns were placing their front hooves on a piece of metal trim at the front of the cage when accessing the bait, which led to inaccurate weight readings and missed opportunities to collar fawns. We corrected this problem by placing a plastic shield above the metal trim so that deer could no longer place hooves on the metal trim. Following this modification, the entire floor surface of the cage comprised only the scale. We also noted that small fawns accessing the bait sometimes failed to break the infrared beam extending across the center of the bait pan, thereby failing to be collared. Thus, we adjusted the positioning of the bait pan to make sure that fawns successfully broke the infrared beam when accessing the bait, regardless of size. Once these changes were made, we successfully collared two more fawns (1 male and 1 female) on successive days, December 13 and 14, 2010. Also, the female fawn that was collared on November 17 shed its collar on December 13 and was successfully recollared on December 20.

On December 21, the actuator that opens and closes the bait door short-circuited in response to cold, snowy weather and damaged the circuit board that controls operation of the device. The actuator was positioned such that moisture could enter it. The moisture, in combination with cold temperatures, caused the failure. It became evident at this point that future device modifications would likely require a heavier-duty actuator. However, until a new actuator could be researched, tested, and installed, DGCD used the same actuator and positioned it differently so that it was less likely to take on moisture. DGCD also replaced the circuit board to restore functionality of the collaring device. Several weeks were required to make these modifications, causing the device to be inoperable from December 21, 2010, through January 15, 2011. On January 20, we recollared the female fawn that was initially collared on December 14 (it shed the first collar on January 13). We then moved the device to the Masonville bait site on January 21, after documenting 5 successful collaring events at Horsetooth Reservoir.

The Masonville bait site was regularly visited by 4 bucks, 3 does, and 2 fawns. The fawns were aggressively chased by the 4 bucks once we put the collaring device in place and restricted the amount of bait available outside of the collaring device. We solved this problem by creating a separate bait site for the bucks a short distance away. It took one week before the fawns at Masonville became comfortable entering the collaring device and accessing the bait in the bait pan. We did not put a collar in place initially because we speculated that the fawns would be more likely to access the bait pan for the first time if they were not required to extend their head through the collar. Once one of the fawns became acclimated and we put a collar in the device, the bait door/actuator began malfunctioning again, preventing the fawn from being collared. The malfunctioning was apparently related to cold temperatures. The bait door/actuator began functioning correctly again several days later and we collared a male fawn on February 4, 2011. The only other fawn on site showed no interest in accessing the bait in the bait pan during the ensuing week. Thus, we stopped baiting the site on February 12 and moved the device to the Red Feather site on February 14.

Several of the gate arms that prevent deer entry into the sides of the device had been damaged by deer over the course of the winter. During February 14–20, as deer became accustomed to the collaring device, we replaced all gate arms with a new, more durable hinge system. We then resumed normal operations and collared our 7th fawn (female) on February 27, 2011. Unfortunately, the RFID reader failed to detect this collared fawn the following day, allowing the fawn to receive a second collar on February 28. We suspended collaring efforts for several days evaluating the RFID failure. It became evident that if a collared fawn entered the device quickly, it could go undetected by the RFID reader. We were aware of this potential problem, but this was the first time it actually occurred. We documented no ill effects of the second collar on the fawn. Realizing the odds of a double-collaring event were low, we resumed collaring efforts on approximately March 6. Incidentally, the odds of the double-collared fawn receiving a third collar were essentially zero because the fawn now had two RFID transponders. We made note that the RFID problem would need to be resolved with a device modification during the following year. The other couple of fawns routinely visiting the site were reluctant to access the bait pan. On March 17, we moved the collaring device to the Heil Valley Ranch site on Boulder County Parks and Open Space land.

Deer regularly visiting the Heil site included 4 bucks, 2 does, and 1 fawn. We were unable to keep the bucks from being aggressive toward the does and fawn around the collaring device, which prevented the fawn from entering the device. In response, we moved the device to the Hall Ranch bait site on March 24, 2011, where 3-4 bucks, 2-3 does, and 1-3 fawns were using the site. Deer acclimated quickly to the collaring device and we collared our 8th fawn on March 28th, immediately after placing the collar in the device. A few days later we concluded the field evaluation because weather was turning warm, green forage was abundant, and bears were coming out of hibernation.

2011 Device Modifications

During our 2010-11 winter field evaluation, we documented a number of issues with the collaring device that needed resolved. During summer-fall 2011, working with DGCD, we made several modifications to the device to address these issues.

- Issue: The solenoid release mechanism occasionally failed to release the collar even when the solenoid was triggered. Modification: We evaluated and incorporated an alternative release mechanism that used an archery caliper release instead of the existing metal, latch system.
- Issue: We documented several scenarios that could allow a fawn to receive a second collar. First, if a collared fawn extended its head through the entry to the device and was detected by the RFID reader but failed to move forward onto the scale for ≥30 seconds, the bait door moved back into the open position. Second, if a collared fawn was on the scale for >15 minutes (i.e., bedded down on the scale), the scale re-zeroed and the door moved back into the open position. At this point another fawn could step into the device, which would indicate a correct weight range, and the collared fawn could receive a second collar if it then accessed the bait. Third, as we directly witnessed, if a collared fawn entered the device quickly, the RFID reader sometimes failed to detect the RFID transponder in the fawn's collar. Modifications: We resolved these issues by reprogramming the device and increasing sensitivity of the RFID reader/antenna.

- Issue: The actuator that controls the bait door commonly malfunctioned in cold temperatures (i.e., ≤ –12 °C). We intend for the device to be fully functional at –32 °C. Modification: We researched other actuators and selected a higher quality unit that would be more likely to perform adequately under the desired conditions. We then evaluated the actuator under controlled temperature settings in a freezer to confirm functionality before installation in the collaring device.
- Issue: The camera mounted on the floor of the device commonly failed to provide useful images for identifying sex. The camera in the bait compartment positioned to take pictures of a fawn's head provided conclusive evidence of sex, indicating the floor camera was unnecessary. Modification: We removed the floor-mounted camera from the device and eliminated the associated wiring and programming.

2012 Field Evaluation

We made considerable progress evaluating and subsequently modifying the collaring device during 2010-2011, and therefore, we believed that a 3-month evaluation period during January-March 2012 would be sufficient for a follow-up field evaluation. We initially evaluated the collaring device during January 2012 at Hall Ranch near Lyons, Colorado. We began evaluating the collaring device on January 10. Unfortunately, the bait sites were visited primarily by adult males, limiting any opportunities to collar fawns. These adult males also appeared to prevent regular attendance at the bait site/collaring device by adult females and fawns that were in the area. The problem of adult males dominating a bait site is not unique to this study and has been documented over time when attempting to capture fawns where bait is used to attract animals to a trap (Colorado Parks and Wildlife, unpublished data). Given the challenges posed by adult males near Lyons, we moved the collaring device at the end of January to Piceance Basin, southwest of Meeker, Colorado, where a separate deer study was underway and could benefit from additional collared fawns. We also believed deer densities would be higher near our bait sites in Piceance Basin than our sites along Colorado's northern Front Range, potentially offering more opportunities to collar fawns.

We initiated our evaluation of the collaring device in Piceance Basin on February 5, 2012. We were unable to monitor the collaring device in the field on a daily basis given the distance from the original study area along the northern Front Range. Our first monitoring period occurred during February 5-10. We collared a male fawn weighing 66 lbs on February 7 during early evening. During February 11-18, we baited the collaring device but did not monitor deer activity on site. We resumed direct field monitoring of the device during February 19-24. During this time, we consistently observed a mixture of does, fawns, and bucks on site but did not successfully collar a fawn. The previously collared fawn was routinely on site and often entered the collaring device. During February 25-March 3, we again baited the collaring device but did not conduct field observations. We resumed field monitoring during March 4-9. Deer consistently accessed the bait site during this period, typically with a group size of 6-7 deer that included 3-4 fawns. On March 9, we collared a female fawn weighing 63 lbs. We once again ceased direct field monitoring during March 10-18 and completed our final monitoring period during March 19-21. Deer were active on site during this final evaluation period, including the previously collared fawns, although we were unable to collar any new fawns. We then ceased our evaluation of the collaring device for 2012, having collared 2 fawns during an approximately 1.5-month evaluation period in Piceance Basin.

Our modifications to the collaring device in 2011 appeared to have improved functionality of the device. The only problem we documented during our field evaluation in 2012 was that the bait door often remained open when the first collared fawn reentered the device on subsequent occasions. While initially of concern, the bait door always closed when tested with other collars. Additionally, the bait door closed each time the second collared fawn reentered the device after having been collared. Thus, we concluded there was a problem with the RFID transponder placed in the first collar rather than a problem with the collaring device itself.

Although the device functioned well, the rate at which deer were collared was particularly slow. Our field observations indicate that fawns typically required one or more weeks of exposure before they entered the device and accessed the bait in the bait pan. Some fawns were reluctant to enter the device even after days or weeks of exposure. We tried various baiting strategies in an attempt to maximally encourage fawns to enter the device to access bait in the bait pan. If bait were only placed in the bait pan inside the device, deer groups were not attracted and retained at the site, and therefore, no fawns were present. If too much bait were placed outside the device, there was no incentive for fawns to enter the device and extend their head through the expanded collar to access bait in the bait pan. Generally speaking, we learned that some bait has to be placed outside the device to attract deer groups to the site and that some bait should be placed on the floor/scale to lure fawns into the device. We also tried placing all bait external to the device in buckets to limit the number of animals accessing bait at any given time and to make them more accustomed to placing their heads in an enclosed area to obtain the bait. However, our observations did not suggest this technique was any more effective at encouraging fawns to enter the device and extend their heads through the expanded collar to access bait in the pan. Winter weather conditions were overall mild during our study, particularly during winter 2011-12, which may partly explain the slow rate at which fawns were collared. During winter conditions exhibiting greater snow depths and lower temperatures with less forage available, we would expect fawns to have greater incentive to enter the device and extend their head through the expanded collar to access bait.

SUMMARY

We developed a fully-functional prototype of an automated collaring device for mule deer in collaboration with professional engineers. The automated collaring device is designed to allow biologists and researchers to radio-collar portions of their deer samples with minimal time and expense because no animal handling is required and deer can be collared at any time. Primary time commitments include baiting sites, moving the device(s) among sites, and adding collars to the device. The collaring device should also have distinct benefits for studies in urban environments by providing a non-invasive technique for collaring deer. We successfully collared 6 different fawns during Nov–Mar, 2011–12, along Colorado’s northern Front Range. We recollared 2 of these fawns after they shed their initial collars, resulting in 8 successful collaring events. Fawns generally showed minimal reaction to being collared. It was evident that fawns did not experience the type of stress that is associated with typical capture and handling techniques. We documented a number of functional issues with the collaring device in 2010-11, which we resolved through design modifications during summer-fall 2011. We conducted a follow-up field evaluation during January-March 2012 and collared 2 additional fawns during February and March in Piceance Basin. The largest drawback of the collaring device is the slow rate at which fawns were collared. Fawns typically required one or more weeks of exposure to the device before fully entering the device and extending their head through the expanded collar to access bait in the bait pan. This slow acclimation period limited utility of the device when compared to traditional capture techniques used to collar fawns. In the future, additional design modifications or more clever baiting strategies will be necessary to improve collaring rates. We also plan to evaluate placement of ≥ 2 collaring devices at the same site once a second collaring device is produced. With more collaring devices, potentially less bait would need to be placed external to the devices and deer might be more inclined to access bait in the bait pans within the collaring devices.

LITERATURE CITED

Barrett, M. W., J. W. Nolan, and L. D. Roy. 1982. Evaluation of a hand-held net-gun to capture large mammals. Wildlife Society Bulletin 10:108–114.

Beasom, S. L., W. Evans, and L. Temple. 1980. The drive net for capturing western big game. Journal of Wildlife Management 44:478–480.

Clover, M. R. 1956. Single-gate deer trap. California Fish and Game 42:199–201.

Ramsey, C. W. 1968. A drop-net deer trap. Journal of Wildlife Management 32:187–190.

Schmidt, R. L., W. H. Rutherford, and F. M. Bodenham. 1978. Colorado bighorn sheep-trapping techniques. Wildlife Society Bulletin 6:159–163.

van Reenen, G. 1982. Field experience in the capture of red deer by helicopter in New Zealand with reference to post-capture sequela and management. Pages 408–421 *in* L. Nielsen, J. C. Haigh, and M. E. Fowler, editors. Chemical immobilization of North American wildlife. Wisconsin Humane Society, Milwaukee, USA.

Webb, S. L., J. S. Lewis, D. G. Hewitt, M. W. Hellickson, and F. C. Bryant. 2008. Assessing the helicopter and net gun as a capture technique for white-tailed deer. Journal of Wildlife Management 72:310–314.

Wolfe, L. L., M. W. Miller, and E. S. Williams. 2004. Feasibility of “test-and-cull” for managing chronic wasting disease in urban mule deer. Wildlife Society Bulletin 32:500–505.

Prepared by ____________________________________

Chad J. Bishop, Mammals Research Leader

Figure 1. Automated collaring device for mule deer, comprising an aluminum cage and a bait compartment. Deer become collared by entering the cage and extending their head through an expanded radio collar when accessing bait.

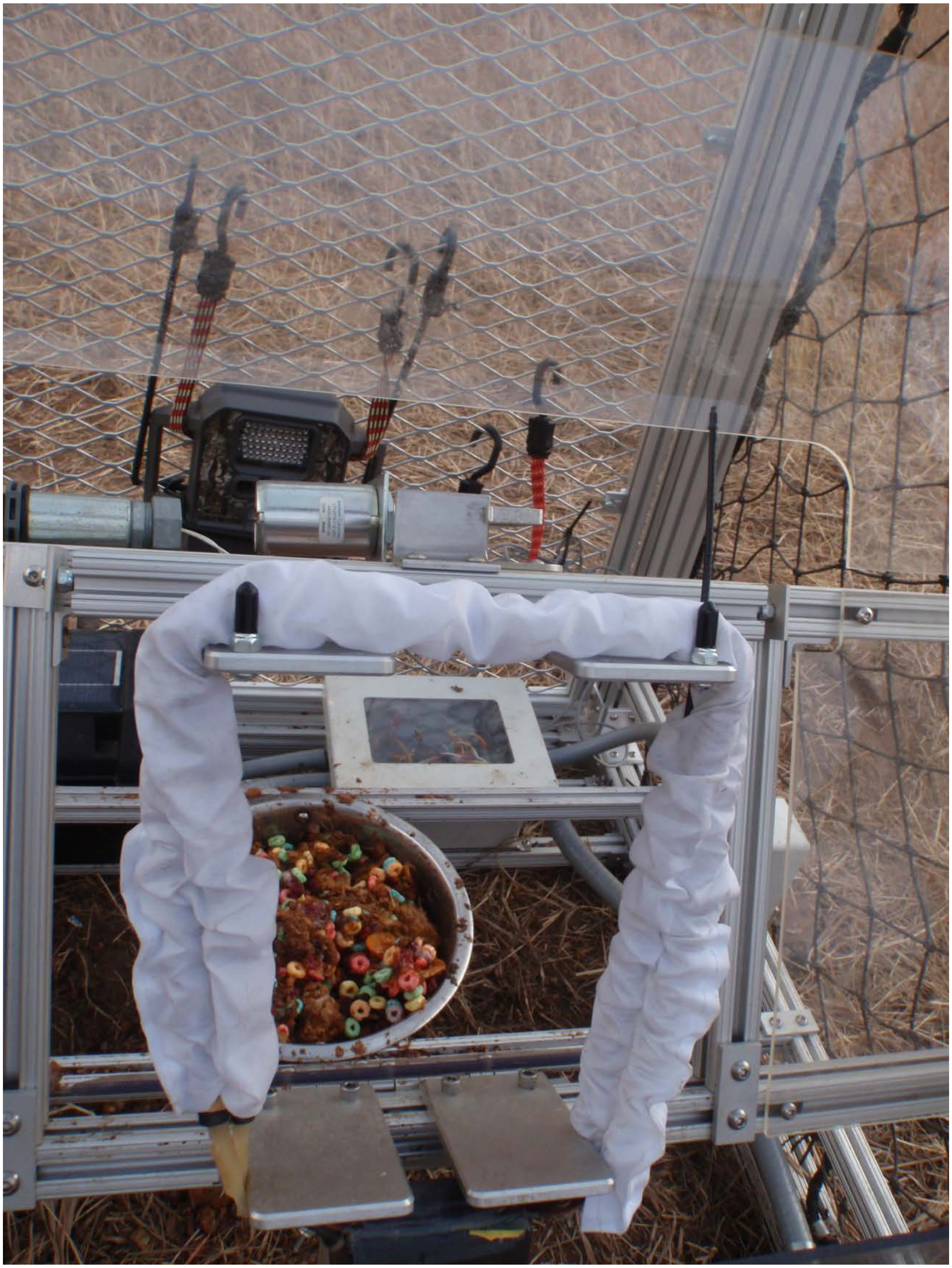


Figure 2. View of the radio collar and bait compartment of an automated collaring device for mule deer. To reach bait, deer must extend their head and neck through the expanded radio collar.

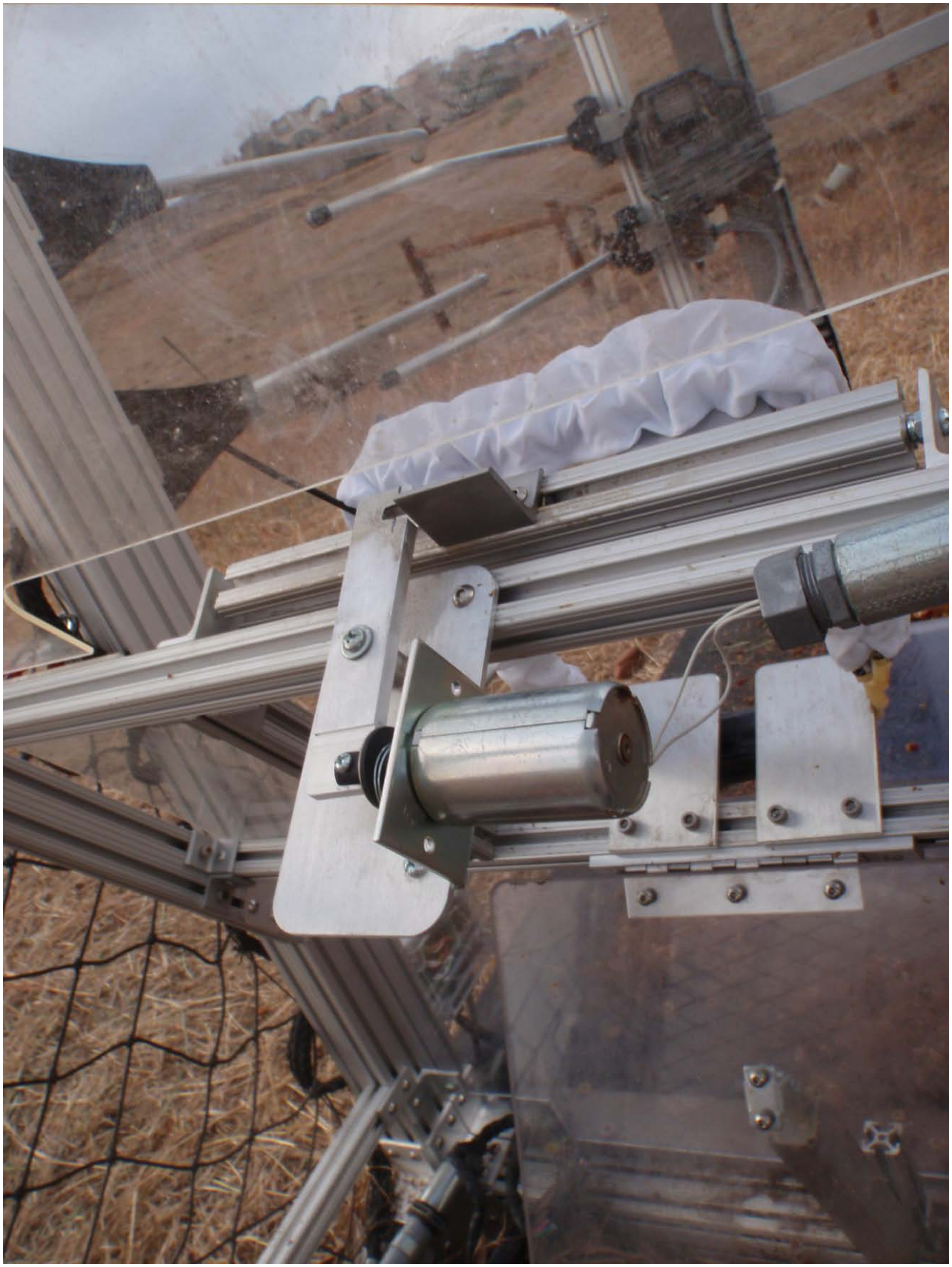


Figure 3. View of the collar release mechanism in an automated collaring device for mule deer.

Figure 4. Female mule deer fawn accessing bait by extending her head through an expanded radiocollar.

Colorado Parks and Wildlife
July 2011 – June 2012

WILDLIFE RESEARCH REPORT

State of: Colorado : Parks and Wildlife
Cost Center: 3430 : Mammals Research
Work Package: 3001 : Deer Conservation
Task No.: : Assessment of survival and optimal harvest Strategies of adult male mule deer in Middle Park, Colorado

Federal Aid
Project No. W-185-R

Period Covered: July 1, 2011 - June 30, 2012

Author: E.J. Bergman; Project Cooperators; C.J. Bishop, K. Oldham, and L. Sidener

Personnel: G. Abram, G. Birch, J. Broderick, M. Crosby, B. Davies, T. Elm, D. Gillham, K. Holinka, A. Holland P. Lukacs, B. Manly, S. Murdoch, S. Schwab, S. Shepherd
Colorado Parks and Wildlife

R. Swisher, S. Swisher, T. McKendrick
Quicksilver Air

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

ABSTRACT

We continued field work on a study designed to assess the survival and optimal harvest strategies of adult male mule deer in Middle Park, Colorado. Two years of baseline survival data for adult (≥ 1 yr. old) male deer were collected prior to termination of this project. During December (2011), 49 additional adult (≥ 1.5 years old) male deer were captured and radio collared, returning the radio collared sample to 100 animals at the start of the 2012 calendar year. The natural, annual survival rate for all deer for the period ending on the 14th of December (2011) was estimated at 0.820 (SE = 0.0394). From the 15th of December (2011) through the 31st of July (2012), survival was estimated at 0.919 (SE = 0.0275). Due to consternation expressed by a very select group of trophy hunters, the project was terminated in July of 2012.

WILDLIFE RESEARCH REPORT

ASSESSMENT OF SURVIVAL AND OPTIMAL HARVEST STRATEGIES OF ADULT MALE MULE DEER IN MIDDLE PARK, COLORADO

ERIC J. BERGMAN

P.N. OBJECTIVES

SEGMENT OBJECTIVES

1. Continue field work in the form of capturing and radio collaring animals.
2. Collect survival data on radio collared deer and provide preliminary survival estimates for adult male mule deer.

INTRODUCTION

Historically, management of big game species has focused on the performance of adult females and the young of the year segments of the population. In the case of mule deer, this has been further refined to the aspects of annual (for adult females) and overwinter (for young of the year) survival. The performance of the male component of populations was deemed less important because it takes few males to provide adequate breeding coverage for the population, and historic harvest management objectives were set to maximize hunting opportunities. As long as sufficient numbers of males were available to breed females there was no desire to restrict hunting opportunity. However, during the past 10-15 years, the management of big game populations, and mule deer populations in particular, has shifted from the objective of providing maximal opportunity towards providing higher quality opportunities (Bishop et al. 2005b, Bergman et al. 2010). High quality opportunities are typically defined by hunters as a combination of the chance to see a greater number of male deer during the hunt, increased potential to harvest an older age class animal (i.e., an animal with more developed antler morphometry), but also reduced interaction and competition with other hunters. In response to this shift in hunter desires and concerns over declining mule deer numbers, the Colorado Division of Wildlife (now Colorado Parks and Wildlife [CPW]) implemented a statewide limitation in deer hunting in 1999. This statewide limitation gave CPW the ability to reduce total hunter numbers and to control the distribution of hunters throughout the state. Since 1999 Colorado's deer herds have become composed of a greater number of males, yet little biological data on them exist. Also stemming from this change in harvest management was a new responsibility for Colorado's terrestrial biologists and wildlife managers. Prior to 1999, licenses were sold over-the-counter and were not limited in number (i.e., any hunter who wished to purchase one was able to do so), and the decision of how many licenses to make available did not need to be considered. Since 1999, CPW has the added responsibility of deciding how many licenses should be allocated in each Data Analysis Unit (DAU). This decision must reflect a balance between meeting DAU population performance objectives, and maximizing hunter opportunity.

Big game populations in Colorado are currently modeled using multiple sources of biological data (White and Lubow 2002). Model inputs include harvest estimates, young recruitment to December, and measured rates of survival of adult females and fawns. Also, the ratio of adult males to adult females is estimated and used to align models by minimizing the difference between observed and modeled values. Only rarely have the survival rates of adult males been measured. This gap in knowledge has historically been viewed as trivial and adult male survival rates have been assumed to be similar to the rates of females. Similarly, it has been assumed that natural survival rates (i.e., post hunt survival) of males do not geographically vary. However, model performance under these assumptions has been poor and the need to measure adult male survival as a parameter has increased. Presently, a number of population models in Colorado suggest that natural adult male survival may be lower than adult female survival, yet empirical data is lacking to verify these suppositions.

Despite this apparent lack of information, survival of adult male mule deer and adult male black-tail deer are not completely novel parameters of interest (Pac and White 2007, Bender et al. 2004, Bleich et al. 2006, Bishop et al. 2005a, McCorquodale 1999). These studies also suggest that adult male mule deer survival tends to be lower than adult female survival when differences occur, further emphasizing the need to rigorously evaluate adult male survival rates. Bishop et al. (2005a) observed lower natural survival rates of adult males than adult females in southwest Idaho: differences were most apparent during winter in 2 of 3 study areas. Pac and White (2007) found that natural survival rates of yearling males in Montana were lower than the average adult female survival rate documented by Unsworth et al. (1999). Finally, Miller et al. (2008) found that adult male survival rates were lower than adult female survival rates in Colorado in response to chronic wasting disease (CWD). In particular to the population modeling interests of Colorado outside the CWD endemic area, the work conducted by Pac and White (2007) has had the greatest utility. This work focused on the survival of males under differing management scenarios and showed a shift in cause-specific mortality of males in areas where harvest was more restricted. It is currently unknown if survival rates would be similar between Montana and Colorado. Similarly, the likelihood of observing shifts in mortality sources is unknown. It has been demonstrated that adult female deer herds in Colorado tend to be habitat limited (Bishop et al. 2009, Bartmann et al. 1992), but the trade-off between harvest, habitat and survival in adult male mule deer has not been explored.

An additional need in Colorado pertains to the harvest management of adult male mule deer. As discussed above, a large shift in mule deer herd size and structure occurred as a result of changes in harvest management. Overall, this shift has been viewed as positive by both CPW as well as the public. However, CPW maintains the responsibility of optimally managing the deer of Colorado and maximizing hunting opportunity under this new set of constraints. To date, CPW has had limited biological information and data to guide harvest management decisions. In particular for this issue, as Data Analysis Units (DAUs) reach and surpass their adult male: adult female ratio objectives, CPW typically responds by increasing the number of available hunting licenses. In situations where herds are continually lower than DAU objectives, available hunting licenses are reduced. What remains unknown about survival of adult male deer is at what level natural survival is reduced due to intraspecific competition (i.e., increased density of adult male deer). If, or when deer herds exceed the adult male: adult female objectives for DAUs, it is often assumed that the surplus of male deer remain in the population into perpetuity. However, this assumption is based on the premise that compensatory mortality does not occur. Similarly, it assumes that annual variation in survival is negligible. However, these assumptions are not biologically realistic. It is possible that herds with large post-hunt populations of adult males experience higher levels of non-harvest mortality. Under this scenario, harvest has not been optimized and more hunters could have been afforded the opportunity to hunt with no effect on post hunting season ratios of adult males to adult females. The most effective way to learn about the mortality process is via manipulative experimentation, but to date this topic has not been deemed a high enough priority to pursue.

STUDY AREA

This study took place in Middle Park, Colorado, within DAU D-9. Within D-9 are 6 Game Management Units (27,181, 18, 37, 371, and 28; Fig. 1).

METHODS

Capture of adult male deer was conducted in January and December of 2011. Capture was conducted via helicopter net-gunning (Webb et al. 2008, Potvin and Breton 1988, White and Bartmann

1994, Barrett et al. 1982). All captures occurred after the completion of the 4th rifle hunting season, eliminating conflicts between capture efforts and hunting. All deer were fitted with expandable radio collars. All radio collars were equipped with mortality sensors that doubled in pulse rate after remaining motionless for 4 hours. Between the time of capture and mid-June, we used ground-based monitoring to determine the live/dead status of deer 3-4 times per week. Additionally, every 5-10 days we conducted a telemetry flight to detect any animals that hadn't been heard from the ground during the preceding week. A general location was collected for each radio marked deer in early-March to determine if it had departed the GMU in which it had originally been captured. From mid-June through remainder of the summer, deer were monitored from the ground weekly and from the air once per month. When detected, all mortalities were investigated as quickly as possible to determine cause of death and to get an accurate estimate of the date of death.

RESULTS AND DISCUSSION

In December (2011), 49 deer were captured and radio collared. On one occasion, an animal suffered a fractured leg as part of the capture process and was subsequently euthanized at the capture site via gunshot to the head. No other capture related injuries or mortalities occurred, although one animal was killed via vehicular collision 2 days post capture. This animal was subsequently censored from survival analysis due to uncertainty if stress related to the capture process had influenced its fate.

Survival of adult male deer between January 2011 and the 14th of December (2011) was estimated to be 0.820 (SE = 0.0394). There was no apparent difference between the north half the D-9 and the south half of D-9 (Table 1), validating the assumptions of the original study plan design. During the 2011 hunting seasons, a total of 31 radio collared bucks were harvested (Fig. 1). Due to mild winter conditions, survival from the 15th of December (2011) through the 31st of July (2012) was very high (0.909, SE = 0.0275).

SUMMARY

Project efforts were successful during the first two years of the study, although local resistance to the project remained high. Based on public meetings, the majority of hunters in the Middle Park area supported the project, although a very select group of trophy hunters remained opposed. Ultimately, CPW leadership determined that it "could not overcome the current opposition" and the study should not proceed. Thus, this research project has been terminated as of 7/12/2012.

LITERATURE CITED

Barrett, M.W., J.W. Nolan, and L.D. Roy. 1982. Evaluation of a hand-held net-gun to capture large mammals. Wildlife Society Bulletin 10:108-114.

Bartmann, R.M., G.C. White, and L.H. Carpenter. 1992. Compensatory mortality in a Colorado mule deer population. Wildlife Monographs 121:1-39.

Bender,L.C., G.A. Schirato, R.D. Spencer, K.R. McAllister, and B.L. Murphie. 2004. Survival, cause-specific mortality, and harvesting of male black-tailed deer in Washington. Journal of Wildlife Management 68:870-878.

Bergman, E.J., B.E. Watkins, C.J. Bishop, P.M. Lukacs, and M. Lloyd. 2010. Biological and socio-economic effects of statewide limitation of deer licenses in Colorado. Journal of Wildlife Management. *In Review.*

Bishop, C.J., J.W. Unsworth and E.O. Garton. 2005a. Mule deer survival among adjacent populations in southwest Idaho. Journal of Wildlife Management 69:311-321.

Bishop, C.J., G.C. White, D.J. Freddy and B.E. Watkins. 2005b. Effect of limited antlered harvest on mule deer sex and age ratios. Wildlife Society Bulletin 33:662-668.

Bishop, C.J., G.C. White, D.J. Freddy, B.E. Watkins, and T.R. Stephenson. 2009. Effect of enhanced nutrition on mule deer population rate of change. Wildlife Monographs 172:1-28.

Bleich, V.C., B.M. Pierce, J.L. Jones and R.T. Bowyer. 2006. Variance in survival of young mule deer in the Sierra Nevada, California. California Fish and Game. 92:24-38.

Miller, M.W., H.M. Swanson, L.L. Wolfe, F.G. Quatarone, S.L. Huwer, C.H. Southwick and P.M. Lukacs. 2008. Lions and prions and deer demise. Plos One 3:1-7.

McCorquodale, S.M. 1999. Movements, survival, and mortality of black-tailed deer in the Klickitat Basin of Washington. Journal of Wildlife Management 63:861-871.

Pac, D.F., and G.C. White. 2007. Survival and cause-specific mortality of male mule deer under different hunting regulations in the Bridger Mountains, Montana. Journal of Wildlife Management 71:816-827.

Potvin, F., and L. Breton. 1988. Use of a net gun for capturing white-tailed deer, Odocoileus-virginianus, on Anticosti Island, Quebec. Canadian Field Naturalist 102:697-700.

Unsworth, J.W., D.F. Pac, G.C. White, and R.M. Bartmann. 1999. Mule Deer Survival in Colorado, Idaho, and Montana. Journal of Wildlife Management 63:315-326.

Webb, S.L., J.S. Lewis, D.G. Wewitt, M.W. Hellickson, and F.C. Bryant. 2008. Assessing the helicopter and net gun as a capture technique for white-tailed deer. Journal of Wildlife Management 72:310-314.

White, G.C. and R.M. Bartmann. 1994. Drop nets versus helicopter net guns for capturing mule deer fawns. Wildlife Society Bulletin 22:248-252.

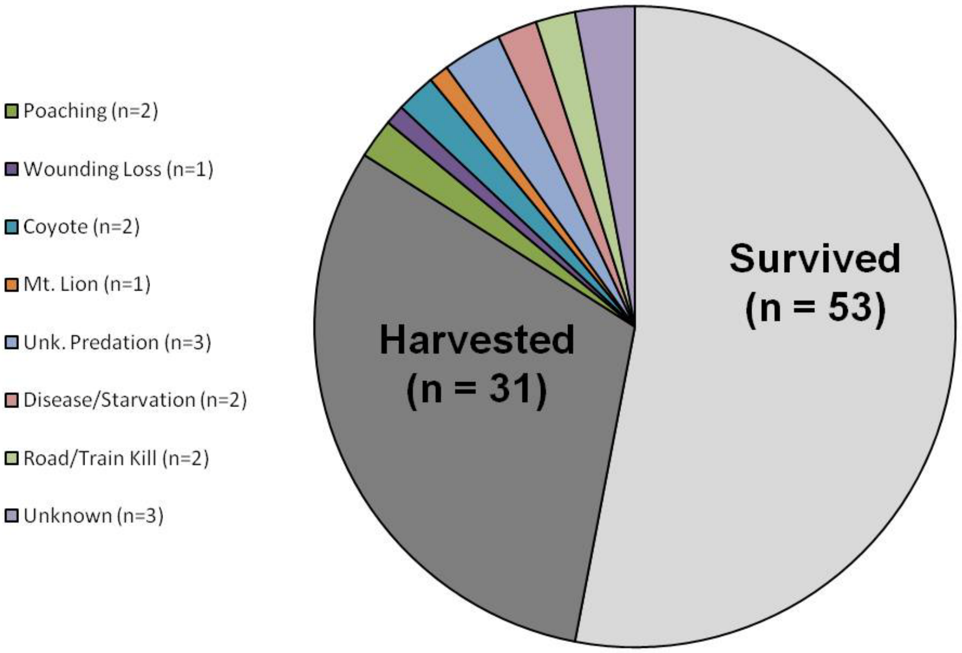
White, G.C. and B.C. Lubow. 2002. Fitting population models to multiple sources of observed data. Journal of Wildlife Management 66:300-309.

Prepared by______________________________
Eric J. Bergman, Wildlife Researcher

Table 1. Model results for known-fate survival models based on mule deer buck data collected in Middle Park, Colorado. Model comparison is made via Akaike's Information Criterion corrected for small sample size (AIC$_c$). Of interest to the original study design, there was no strong evidence that there was a difference in survival between the northern and southern halves (Area) of the study area.

Model	**ΔAIC$_c$**	**AIC$_c$ Weight**	**Likelihood**	**Parameters**
$\hat{S}$ (Constant)	0.00	0.69	1.00	1
$\hat{S}$ (Area)	1.56	0.31	0.46	2
$\hat{S}$ (Week)	49.77	0.00	0.00	48
$\hat{S}$ (Area + Week)	51.37	0.00	0.00	49
$\hat{S}$ (Area*Week)	131.16	0.00	0.00	96

Figure 1. Fate and associated cause of death of 100 mule deer bucks between January 2011 and December 2011 in Middle Park, Colorado.



Colorado Division of Parks and Wildlife
July 2011 – June 2012

WILDLIFE RESEARCH REPORT

State of:	Colorado	:	Division of Parks andWildlife
Cost Center:	3430	:	Mammals Research
Work Package:	3002	:	Elk Conservation
Task No.		:	Evaluating solutions to reduce elk and mule deer damage on agricultural resources.
Federal Aid Project No.			

Period Covered: July 1, 2011 – June 30, 2012

Author: H.E. Johnson; project cooperators, P. Dorsey, M. Hammond, C. Bishop, K. VerCauteren, D. Walter, C. Anderson, and J. Fischer.

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

ABSTRACT

Elk and mule deer provide important recreational, ecological, and economic benefits, but they can also cause substantial damage to agricultural resources in rural environments. This situation has generated significant challenges for wildlife agencies that are responsible for maintaining viable ungulate populations while also minimizing crop damage. One of the most severe areas of ungulate damage in Colorado has been the sunflower fields around Dove Creek. In this region, roughly a quarter of million dollars were annually paid to farmers between 2007 and 2009 for depredation caused by elk and deer. The main management tool used by Colorado Parks and Wildlife (CPW) to reduce ungulate damage has been the allocation of kill permits, distribution hunts, and private land only doe/cow hunts; however, tolerance for these permits has been low among local sportsman and the general public. Pressure from local sunflower growers over crop damage, and frustration from the general public over kill permits, generated the need for CPW to evaluate other management options for reducing elk and deer crop depredation. As a result, CPW partnered with wildlife damage researchers from the National Wildlife Research Center to find science-based solutions for reducing crop damage. Collaboratively, our goals are to 1) examine elk and deer distribution and migration patterns around agricultural areas to design public hunting opportunities to reduce depredation, 2) experimentally test a suite of non-lethal fencing techniques to minimize crop damage, and 3) map and model landscape characteristics associated with ungulate damage to specify more effective site-specific management techniques to minimize depredation. During FY11-12 we focused on collecting field data to meet project objectives. Specifically, we constructed experimental fence plots and monitored their effectiveness in reducing elk and deer damage (objective 2) and collared elk and deer to collect information about local movement and distribution patterns (data required to meet objectives 1 and 3).

WILDLIFE RESEARCH REPORT

EVALUATING SOLUTIONS TO REDUCE ELK AND MULE DEER DAMAGE ON AGRICULTURAL RESOURCES

HEATHER E. JOHNSON

P.N. OBJECTIVES

To conduct a study on elk and mule deer around the agricultural fields of Dove Creek that 1) examines wild ungulate distribution patterns to design public hunting opportunities to reduce crop damage, 2) experimentally tests a suite of non-lethal fencing techniques to minimize crop depredation, and 3) maps and models landscape characteristics associated with damage to specify more effective site-specific management practices.

SEGMENT OBJECTIVES

1. Implement the construction of experimental fence plots on sunflower fields in the vicinity of Dove Creek, including electric fences, temporary winged fences, and chemical repellent fences.
2. Collect field data on elk and deer damage to sunflowers in experimental fence plots throughout the growing season.
3. Capture and collar adult female elk and mule deer around agricultural fields in the vicinity of Dove Creek.

INTRODUCTION

Elk and deer provide important recreational, ecological, and economic benefits, but they can also cause substantial damage to agricultural resources in rural areas (Austin et al. 1998, Wisdom and Cook 2000). In Colorado, elk and deer crop depredation accounts for a majority of the wildlife damage claims in the state, and CPW is obligated to pay for those lost resources. In recent years, the agency has spent approximately $500,000 on an annual statewide basis to compensate farmers for ungulate depredation. This situation has generated significant challenges for CPW and other wildlife agencies that are responsible for maintaining viable ungulate populations while also minimizing crop damage (Van Tassell et al. 1999, Wagner et al. 1997, Hegel et al. 2009, Walter et al. 2010).

Elk and deer crop depredation results from a combination of factors including the seasonal distribution and abundance of local forage resources, landscape configuration, and herd density patterns (Vecellio et al. 1994; Yoder 2002; Hegel et al. 2009). Damage can be highly variable within and among growing seasons, as local patterns in precipitation and temperature will alter the availability of native forage and the motivation of ungulates to feed on agricultural fields (Walter et al. 2010). The juxtaposition of cropland and wildland has also been found to be particularly important in driving damage rates, as those cultivated fields closer to cover experience more damage (Nixon et al. 1989, Hegel et al. 2009). Additionally, studies have found that ungulate damage is often caused by only a subset of individuals in the population, depending on the spatial and social structuring of the herd. These observations have critical implications for wildlife managers, as 1) management practices may be differentially effective based on the variability of native forage conditions and the spatial juxtaposition of other habitat features, and 2) management techniques targeted at specific animals may be more effective than implementing those techniques on the population at large (Blejwas et al. 2002, Hegel et al. 2009). As a result, it is important to understand both the spatial configuration of seasonal resources and the resource selection patterns of different segments of local ungulate populations to successfully identify strategies to reduce elk and deer crop damage (Hegel et al. 2009).

One of the most significant hotspots of elk and mule deer depredation in Colorado has been in the vicinity of Dove Creek, where CPW paid roughly a quarter of million dollars annually to farmers between 2007 and 2009. High damage in this region has been primarily attributed to a recent switch in the crops that are locally grown. Farmers traditionally grew beans, spring and winter wheat, oats, alfalfa and grass hay which had minimal damage by wild ungulates. In recent years, however, local growers have planted sunflowers, a high-value seed oil crop used to produce biofuels, and a crop that is highly desirable to wild ungulates. In addition to this recent switch in crops, ungulate damage around Dove Creek is exacerbated by the spatial distribution of sunflower fields in relation to the surrounding wildlands (e.g., sagebrush-mixed shrublands and piñon-juniper woodlands). The region is fractured with deep canyons that provide refugia for elk and deer, and fields adjacent to the canyon rims experience the greatest amount of depredation. With the substantial increase in biofuel production in the U.S. (World Resources Institute 2008), the agricultural conversion observed around Dove Creek will likely become common, as high-priced, crops replace more traditionally-grown, lower-cost crops (Walter et al. 2009).

The main management tool available to CPW to reduce ungulate sunflower damage has been to increase harvest through the use of kill permits, distribution hunts, and private land only (PLO) doe/cow hunts, however tolerance for these permits has been low among local sportsman and the general public. Permits are typically allocated to farmers between June and August, when calves and fawns are still dependent on their mothers, reducing the acceptability of female hunts. Additionally, local elk and deer populations are near or below management objectives, creating a paradox where CPW ultimately wants to increase ungulate herds, but reduce crop depredation. Hunting is also economically important around Dove Creek, so there is a strong desire in the local community to have increased public hunting opportunities and reduced PLO damage hunts.

Given pressure by farmers over elk and deer sunflower damage, and frustration by sportsmen and the public over kill permits, CPW wildlife managers were interested in finding alternative solutions for reducing sunflower depredation. As a result, personnel from CPW partnered with wildlife-damage researchers from the National Wildlife Research Center (NWRC) to find non-lethal science-based solutions for reducing sunflower depredation. Collaboratively, we developed a proposal to 1) identify public hunting strategies that reduce crop damage, 2) test a suite of non-lethal fencing techniques to minimize crop depredation, and 3) map and model landscape characteristics associated with damage behavior to specify more effective management practices (Johnson et al. 2011). Results from this study should enable CPW and local growers to reduce ungulate crop depredation, leading to a decrease in compensation payments, a decrease in kill permits/distribution hunts, and an increase in public hunting opportunities.

In FY11-12 we focused on collecting field data to meet project objectives. Specifically, we constructed experimental fence plots and monitored their effectiveness in reducing elk and deer damage (objective 2) and collared elk and deer to collect information about local movement and distribution patterns (data required to meet objectives 1 and 3).

STUDY AREA

The area around Dove Creek, Colorado (Montezuma, San Miguel and Dolores counties) is comprised of a mixture of agricultural and public lands. This project focuses on the north half of CPW Game Management Unit 72 and the west half of 711 (the portion west of the Dolores River). The area is generally characterized as mountain shrubland interspersed with irrigated and dryland agricultural fields, ranging from 1,981 to 2,590 m in elevation. The mountain shrub habitat type is primarily composed of serviceberry (*Amelanchier alnifolia*), bitterbrush (*Purshia tridentata*), mountain mahogany (*Cercocarpus montanus*), squaw apple (*Peraphyllum ramosissimum*) and black sagebrush (*Seriphidium novum*).

Sunflower fields around Dove Creek are spatially juxtaposed to deep canyons that provide refugia for elk, exacerbating ungulate damage on agricultural crops (Figure 1).

METHODS

Testing the effectiveness of different fence types for reducing ungulate damage

During FY11-12 we constructed experimental fence plots to test the effectiveness of three non-lethal exclusionary fences for reducing elk and deer damage: a polyrope electric fence, a temporary "winged" fence, and an organic repellent "fence." These differ from traditional exclusionary fencing for elk and deer, in that they are cheaper to construct and can be easily moved among fields over time, as farmers grow sunflowers on a rotational basis. Each fence type is described below:

- Polyrope electric fence – The polyrope electric fence acts primarily as psychological barrier for elk and deer based on learned behavioral, avoidance conditioning (McKillop and Sibly 1988). The fences consists of conductive wires which are woven into synthetic electric "ropes" that are more durable, visible, and easy to install than traditional electric fences (Figure 2; Hygnstrom and Craven 1988, VerCauteren et al. 2006). Avoidance conditioning occurs when an animal contacts the fence, often with the nose or tongue, and receives a powerful electric shock. Polyrope fences have had success in reducing deer damage (Hygnstrom and Craven 1988, Seamans and VerCauteren 2006), but have not been experimentally tested for reducing elk damage. For the 5 randomly selected polyrope treatment plots, we constructed a fence approximately 1.8 meter tall with 5 strands to discourage passage under, through, or over the fence. The polyrope was powered by a Speedrite™ 3000 energizer (Tru-Test Incorporated, San Antonio, Texas) using a 12-volt deep-cycle battery with a solar-panel recharger.
- Temporary winged fence - For seasonal agricultural resources, such as sunflowers, temporary fences may be sufficient to provide protection from wild ungulates and are inexpensive, lightweight, and easy to erect and remove (Rosenberry et al. 2001, VerCauteren et al. 2006). We tested the effectiveness of a temporary "winged" fence made of polypropylene mesh (Figure 3). The fence is installed completely on one side of the target field, and partially installed on two other sides having 50-100 meter "wings" that extend perpendicular from the full fence line. This design was found to reduce deer damage in corn fields (Hildreth et al. In Review) but has not yet been tested on elk or on deer with crops other than corn. On those plots receiving winged-fence treatments, we installed the fence such that the side receiving complete protection was along the crop/wildland interface. The fence was made of 2.4 meter tall black barrier material (e.g., Guardian Warning Barrier, Tenax Corporation, USA, Baltimore, Maryland) for increased height and visual deterrence.
- Plantskydd - Repellents are nonlethal substances that can be used to deter ungulates by decreasing a plant's palatability (Walter et al. 2010). We will test the effectiveness of a relatively new product, Plantskydd, for reducing sunflower damage around Dove Creek. This product was developed in Sweden to decrease mammalian wildlife damage on commercial forests. It works by emitting an odor that animals associate with predator activity, repelling the animal before it forages on crop plants. There is great interest in the success of this product as it can be easily applied to vegetation by ground and aerial spraying, used on both organic and conventionally grown sunflowers, and is cost-effective for growers. That said, the effectiveness of Plantskydd has not been experimentally tested, only anecdotally reported. To test this method, 5 Plantskydd treatment plots were ground sprayed in a ~20 ft swath around the plot perimeters after germination had begun (as directed by the manufacturer). Plantskydd was reapplied to treatment plots once/month throughout the growing season as the repellent may wash off or decompose over time and needs to be reapplied to new plant material.

We constructed the fence plots based on a randomized block design. We identified 5 different sunflower fields to serve as replicates (~160-200 acres in size); all fields had previously suffered high ungulate crop damage. Within each field we specified 4 10-acre plots, one for each experimental fence treatment type (polyrope fence, temporary winged fence, chemical repellent fence) and a control (Figure 1). The 4 plots were randomly assigned within each field, such that each field (block) contained one replicate of all treatments (Gotelli and Ellison 2004). This design allows us to statistically account for environmental heterogeneity, as we expect that damage will be variable among fields. Within the fields, study plots were spaced as far apart as possible, to account for plot independence. Plots were also placed along the agriculture/wildland boundary, where depredation is expected to be concentrated. Fences were installed by Dillon Fencing (Naturita, CO) during the end of June and early July 2011, after sunflowers had germinated.

The 20 plots (experimental and control plots) were delineated were monitored from mid-July through mid-October (time of harvest). Treatment and control plots were examined for 2 key response variables: elk/deer incursion and sunflower damage. We quantified incursion by elk and deer into our plots on a biweekly basis, assessing the permeability of the different fence types. To do this, an observer walked the perimeter of each plot, counting the number of elk and deer tracks entering and exiting the field. Tracks were raked out between observations so they were not double-counted. Differences in the number of elk and mule deer tracks into treatment/control fields were tested using repeated measures ANOVA.

In addition to quantifying incursion into experimental plots, we also quantified direct damage to sunflower plants. We assessed damage every 2 weeks using the variable-area-transect (VAT) method for estimation of crop depredation (Engeman and Sterner 2002, Gilsdorf et al. 2004a, Gilsdorf et al. 2004b). In each plot, we conducted 15 VAT transects at random starting points, inspecting a row of sunflowers, and counting the total number of sunflowers that were damaged and undamaged. If 5 cervid-damaged sunflowers were tallied in 100 meters, we recorded the distance traveled and the total number of sunflowers on the transect. If 5 cervid-damaged sunflowers were not tallied in 100 meters, the observer recorded the total number of sunflowers and any cervid-damaged sunflowers observed in that distance. We calculated the percentage of sunflowers damaged per transect using the equation ~ damage = (number of damaged sunflowers) / (number of damaged sunflowers+number of undamaged sunflowers) (Gilsdorf et al. 2004a, Gilsdorf et al. 2004b). Additionally, at the end of the season, we had an agricultural assessor evaluate game damage and year-end yields between treatment and control plots, the ultimate measure of success for each management technique.

Just prior to the sunflower harvest in mid-October 2011, we removed all fencing materials from our study fields. The materials were stored over the winter by CPW and re-deployed to 4 different sunflower fields in June 2012 for the second year of testing.

Collaring elk and deer to collect information on movement and distribution

To obtain data on ungulate movement and distribution patterns we contracted Quicksilver Air to capture and collar 20 adult female elk and 20 adult female deer using a net gun from a helicopter (Krausman et al. 1985). Females were the target of collaring efforts because they cause a majority of the crop depredation and should provide valuable insight into herd distributions. Helicopter captures were scheduled from 11-13 October 2011, just prior to the start of first rifle season. There was a narrow window in which to capture animals, as helicopter operations could only occur after the heat of the summer had passed, but before rifle season had begun (to minimize impacts to hunters). Captured elk and deer were hobbled and blindfolded, fitted with a global positioning system (GPS) collar, aged, measured and released. GPS collars were programmed to collect a location every 4 hours for 2 years, and then drop off the animals in fall 2013. The collars are “store-on-board,” meaning that the data can only be downloaded once the collar is retrieved from the field. Until collars drop-off, we are conducting monthly aerial telemetry flights to monitor survival and obtain some general location information.

Once GPS collar data has been retrieved, elk and mule deer locations will be used to map seasonal distribution and migration patterns in ArcGIS. This should allow CPW to design public hunts that will target conflict elk and mule deer, while minimizing the need for PLO hunts and kill permits. For example, the Utah Division of Wildlife Resources is willing to consider special elk hunts south of Hwy 491 if we find that any or all of the resident elk herds (causing damage) spend portions of the year in Utah. Locations will also allow us to determine the amount of use of crop fields by elk and deer, and the proportion of animals using crop fields (whether it is only certain segments of the population, or the population at large).

Animal location data will also be used to model ungulate damage potential in relation to field locations, surrounding habitat types, human development, and topography. These variables have been important in explaining rates of ungulate depredation, as damage tends to increase closer to cover, further from roads, and depending on crop palatability (Grover and Thompson 1986, Nixon et al. 1989, Hegel et al. 2009). Information about the location of a crop field in the context of the overall landscape will allow CPW to work with local growers to identify appropriate management tools, and the timing of their implementation, to reduce game damage. Such a model will serve as a powerful tool for CPW managers, as they will be able to predict the likelihood of depredation for different fields, depending on location, the surrounding environment, and the crop type, and therefore help landowners specify crop choice or management actions to reduce damage.

RESULTS AND DISCUSSION

Between 20 July 2011 and 20 October 2011 we conducted 137 incursion surveys of the 20 fence plots and 2,100 sunflower ungulate damage assessments. We used repeated measures ANOVA to determine whether there were statistically significant differences in elk and deer incursion into each fence treatment type. We found that incursion varied by treatment for both elk and deer (Elk: $F_{20,\,116} = 6.84$, $P < 0.001$; Deer: $F_{20,\,116} = 6.24$, $P < 0.001$) such that the electric fence plots had the fewest elk and deer tracks entering the plot, followed by the winged fences, the Plantskydd repellant fences, and the control plots (Figure 4).

Biweekly damage assessments of the sunflower fields showed that crop damage followed the same general trends as the frequency of elk and deer entering the treatment plots. Generally, the electric fence plots received the least elk and deer damage, followed by the winged fence, and then by the Plantskydd treatment; the control fields had the highest levels of ungulate damage (Table 1, Figure 5). As expected, the percentage of damaged plants/plot generally increased throughout the growing season, except during the last two assessments. This pattern may have resulted from differences in an observer’s ability to detect damage at different stages of sunflower growth. At the end of the growing season, when the plants are dry and sunflower heads are bent over, damage may be harder to detect than at earlier stages of growth (i.e., when the heads are upright and the leaves are erect). Damage, however, was generally minimal in 2011 across all fields and plots (<1% in plots with electric fences and ~4% in control plots). We suspect that minimal damage was the result of abundant natural forage for elk and deer, as late spring rains in 2011 generated more forage than is typically observed in the vicinity of Dove Creek during summer. Indeed, CPW did not pay out any damage claims to farmers for elk and deer crop depredation in 2011, as wild ungulates were not readily observed on fields. We plan on constructing and monitoring experimental fence plots for a second year in 2012, to test the effectiveness of these treatments when sunflower fields experience more typical rates of damage.

Quicksilver Air captured and collared 20 adult female deer and 9 adult female elk. Although deer were readily available for capture throughout the study area, the helicopter crew had a difficult time finding elk in the study area. Wildlife managers suspect that the elk had already left the agriculture areas around Dove Creek, and had potentially crossed the Utah border by that point in the fall. We plan on trying to ground dart elk during summer 2012 to deploy the remaining elk collars.

We conducted monthly aerial telemetry flights for collared animals to track survival and general movement patterns. Four deer died during winter 2011. One deer died in late October, likely due to capture related causes (D12). The other 3 mortalities occurred in February and March 2011, one from a vehicle collision (D4) and the other two from unknown causes (D8 and D19). GPS collars were retrieved from all mortalities so that the data could be downloaded and processed (Figure 6). This information will be used during FY13-14 to map and model seasonal ungulate distributions, game damage potential, and management options for farmers.

SUMMARY AND FUTURE PLANS

During FY11-12 we constructed the experimental fence plots for the first year of fieldwork, quantified elk and deer damage across our different fencing treatment types, and collared elk and deer in the study area. In FY12-13 we will conduct the fencing experiments for a second field season, and attempt to deploy our remaining elk collars via ground darting. We will continue to monitor the survival and movements of collared animals on a monthly basis using aerial telemetry, until collars detach from the animals in fall 2013. The benefits of this project include gaining knowledge about local elk and deer movements and distribution relative to agricultural fields, identifying non-lethal techniques for reducing ungulate damage to sunflowers and other crops, the development of models to identify areas highly susceptible to damage based on landscape characteristics, and the potential to redesign public hunting opportunities to increase opportunity while reducing those resident animals causing a majority of the damage.

LITERATURE CITED

Austin, D.D., P.J. Urness, and D. Duersch. 1998. Alfalfa hay crop loss due to mule deer depredation. Journal of Range Management 51:29-31.

Blejwas, K.M., B.J. Sacks, M.M. Jaeger, and D.R. McCullough. 2002. The effectiveness of selective removal of breeding coyotes in reducing sheep predation. Journal of Wildlife Management 66:451-462.

Engeman, R.M., R.T. Sterner. 2002. A comparison of potential labor-saving sampling methods for assessing large mammal damage in corn. Crop Protection 21:101-105.

Gilsdorf, J.M., S.E. Hygnstrom, K.C. VerCauteren, G.M. Clements, E.E. Blankenship, and R.M. Engeman. 2004a. Evaluation of a deer-activated bio-acoustic frightening device for reducing deer damage in cornfields. Wildlife Society Bulletin 32:515-523.

Gilsdorf, J.M., S.E. Hygnstrom, K.C. VerCauteren, G.M. Clements, E.E. Blankenship, and R.M. Engeman. 2004b. Propane exploders and Electronic Guards were ineffective at reducing deer damage in cornfields. Wildlife Society Bulletin 32:524-531.

Gotelli, N.J., and A.M. Ellison. 2004. A primer of ecological statistics. Sinauer Associates, Sunderland, Massachusetts.

Hegel, T.M., C.C. Gates, and D. Eslinger. 2009. The geography of conflict between elk and agricultural values in the Cypress Hills, Canada. Journal of Environmental Management 90:222-235.

Hildreth, A.M., S.E. Hygnstrom, E.E. Blankenship, and K.C. VerCauteren. In Review. Efficacy of a partial poly-mesh fence with wings to reduce deer damage to corn.

Hygnstrom, S.E. and S.R. Craven. 1988. Electric fences and commercial repellents for reducing deer damage in cornfields. Wildlife Society Bulletin 16:291-296.

Johnson, H.E., P.Dorsey, M. Hammond, C. Bishop, K. VerCauteren, D. Walter, and C. Anderson. 2011. Evaluating solutions to reduce elk and mule deer damage on agricultural resources. Study Proposal, Colorado Division of Parks and Wildlife, Fort Collins, USA.

Krausman, P. R., J. J. Hervert, and L. L. Ordway. 1985. Capturing deer and mountain sheep with a net-gun. Wildlife Society Bulletin 13:71–73.

McKillop, I.G., and R.M. Sibly. 1988. Animal behaviour at electric fences and implications for management. Mammal Review 18:91-103.

Nixon, C.M., L.P. Hansen, P.A. Brewer, J.E. Chelsvig. 1989. Ecology of white-tailed deer in an intensively farmed region of Illinois. Wildlife Monographs 118:1-77.

Rosenberry, C.S., L.I. Muller, and M.C. Conner. 2001. Movable, deer-proof fencing. Wildlife Society Bulletin 29:754-757.

Seamans, T.W., and K.C. VerCauteren. 2006. Evaluation of ElectroBraid™ fencing as a white-tailed deer barrier. Wildlife Society Bulletin 34:8-15.

Van Tassell, L.W., C. Phillips, B. Yang. 1999. Depredation claim settlements in Wyoming. Wildlife Society Bulletin 25:886-894.

Vecellio, G.M., R.H. Yahner, and G.L. Storm. 1994. Crop damage by deer at Gettysburg Park. Wildlife Society Bulletin 22:89-93.

VerCauteren, K.C., M.J. Lavelle, and S. Hygnstrom. 2006. Fences and deer-damage management: a review of designs and efficacy. Wildlife Society Bulletin 34:191-200.

Wagner, K.K., R.H. Schmidt, and M.R. Conover. 1997. Compensation programs for wildlife damage in North America. Wildlife Society Bulletin 25:312-319.

Walter, W.D., M.J. Lavelle, J.W. Fischer, T.L. Johnson, S.E. Hygnstrom, and K.C. VerCauteren. 2010. Management of damage by elk (*Cervus elaphus*) in North America: a review. Wildlife Research 37:630-646.

Walter, W.D., K.C. VerCauteren, J.M. Gilsdorf, and S.E. Hygnstrom. 2009. Crop, native vegetation, and biofuels: response of white-tailed deer to changing management priorities. Journal of Wildlife Management 73:339-344.

World Resources Institute. 2008. WRI EarthTrends monthly update page. http://earthtrends.org/updates/node/180.

Wisdom, M.J., and J.G. Cook. 2000. North American Elk. Pages 694-735 *in* S. Demarais and P.R. Krausman, editors. Ecology and management of large mammals in North America. Prentice Hall, Upper Saddle River, New Jersey, USA.

Yoder, J. 2002. Deer-inflicted crop damage and crop choice in Wisconsin. Human Dimensions of Wildlife 7:179-196.

Prepared by ____

Heather E. Johnson, Wildlife Researcher

Table 1. Average percentage of damaged plants/transect during successive damage assessments across the 2011 sunflower growing season. Averages are displayed by field and treatment plot.

	ASSESSMENT						
FIELD/PLOT	1	2	3	4	5	6	7
Guynes							
Control	1.7	5.2	5.6	6.9	9.2	8.9	1.5
Electric	0.0	0.1	0.1	0.2	0.4	0.2	0.2
Plantskydd	0.8	4.5	6.6	2.5	3.9	7.4	1.5
Winged	0.0	0.3	0.9	0.6	1.0	1.0	0.7
Schear-Brewer							
Control	0.3	1.1	0.7	1.0	2.6	1.2	1.4
Electric	0.0	0.1	0.7	0.4	0.6	0.6	0.4
Plantskydd	0.0	6.5	2.3	1.9	4.5	1.0	2.2
Winged	0.0	0.3	1.8	0.7	1.1	1.5	1.1
Schear-Homestead							
Control	0.0	7.1	3.0	7.7	3.1	7.5	2.5
Electric	0.0	0.1	0.2	0.1	0.2	0.1	0.2
Plantskydd	0.0	0.4	3.2	2.4	9.4	5.3	1.6
Winged	0.1	0.3	5.6	3.1	8.1	5.6	2.1
Schear-Hudgeons							
Control	0.1	0.6	0.7	1.3	0.5	0.7	0.6
Electric	0.0	0.5	0.9	0.8	0.8	0.5	0.7
Plantskydd	0.0	0.3	0.4	0.6	0.4	0.3	0.4
Winged	0.1	0.5	1.1	0.5	2.5	1.1	0.5
Warren							
Control	0.0	0.0	0.1	0.1	0.1	0.2	0.3
Electric	0.0	0.0	0.1	0.0	0.1	0.1	0.2
Plantskydd	0.0	0.0	0.1	0.1	0.0	0.1	0.0
Winged	0.0	0.0	0.1	0.0	0.1	0.2	0.1

Figure 1. Placement of experimental fence plots within the 5 replicate sunflower fields during the 2011 growing season (July – October). Fields are located adjacent to wildland canyons.

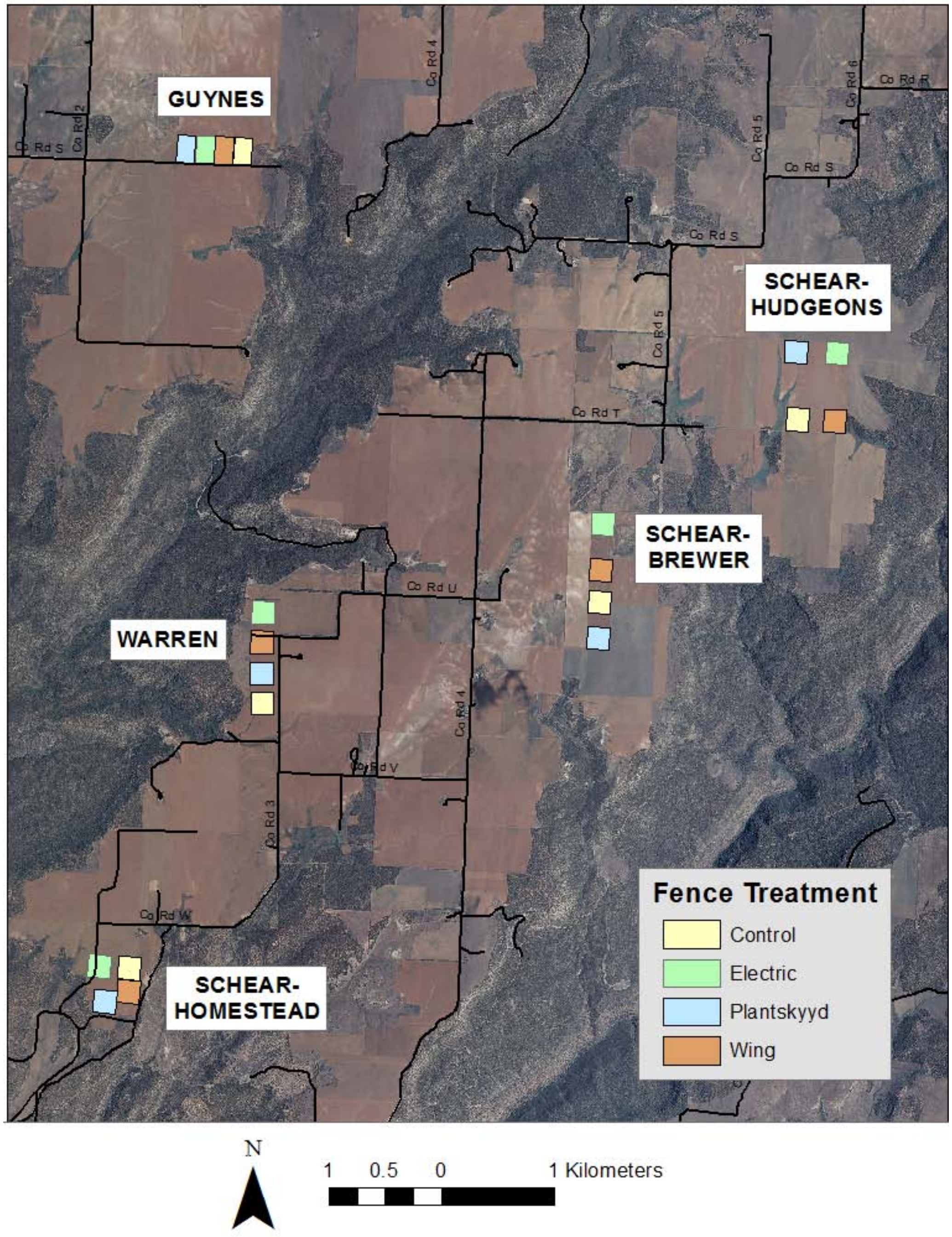


Figure 2. Photo of a polyrope electric fence constructed in a sunflower field south of Dove Creek.

Figure 3. Photo along a winged temporary fence constructed in a sunflower field south of Dove Creek.

Figure 4. Mean number of deer and elk that crossed into experimental fence plots on a biweekly basis, by treatment type, during damage surveys throughout the growing season (results generated from a repeated measures ANOVA).

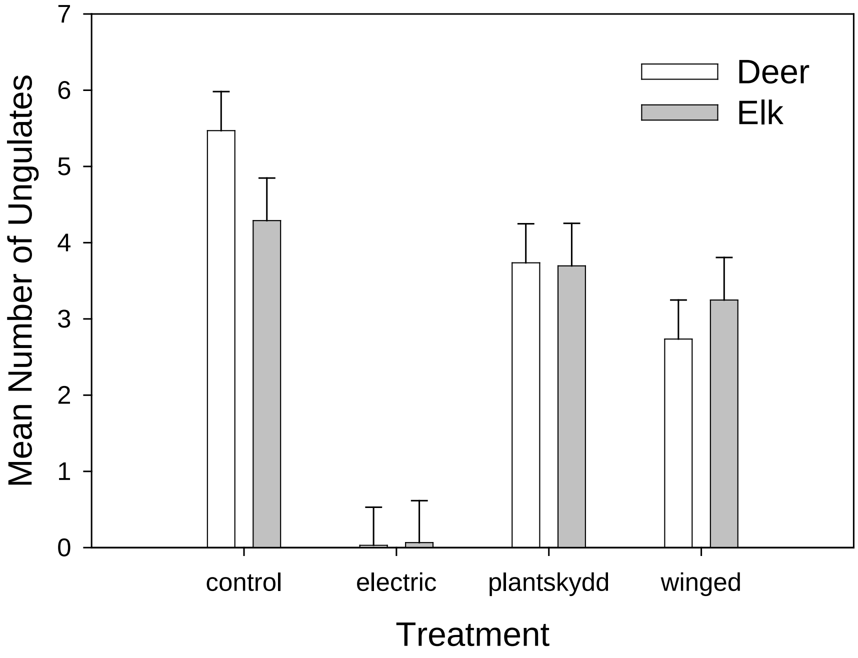


Figure 5. Average percentage of damaged plants/transect for biweekly assessments across all fields for different treatment types.

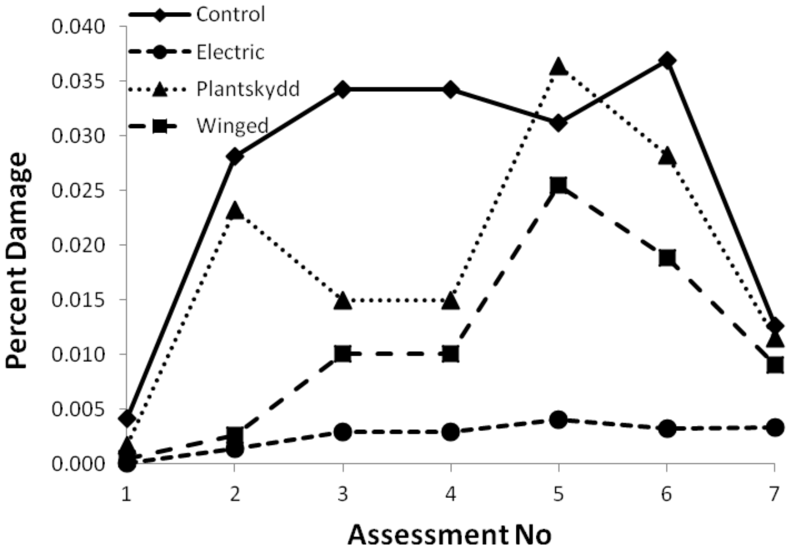
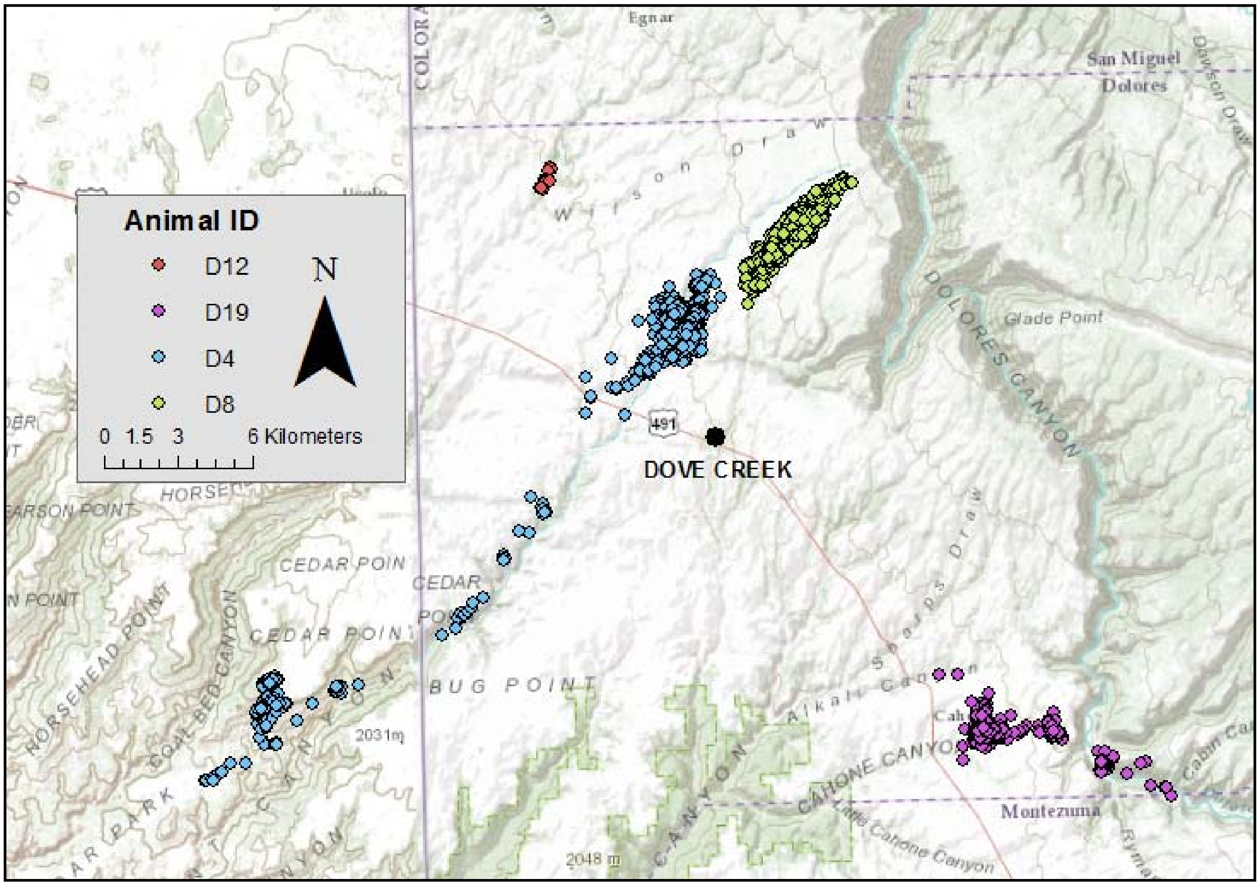


Figure 6. GPS collar locations from deer mortalities during FY11-12 in the vicinity of Dove Creek, CO.



Colorado Division of Parks and Wildlife
July 2011 – June 2012

WILDLIFE RESEARCH REPORT

State of:	Colorado	:	Division of Wildlife
Cost Center:	3430	:	Mammals Research
Work Package:	3003	:	Predatory Mammal Conservation
Task No.		:	Black bear exploitation of urban environments:
		:	finding management solutions and assessing
			regional population effects
Federal Aid Project No.			

Period Covered: July 1, 2011 – June 30, 2012

Author: H.E. Johnson; project cooperators, C. Bishop, J. Broderick, J. Apker, S. Lischke, M. Alldredge, S. Breck, J. Beckmann, K. Wilson, M. Reynolds-Hogland, T. Spezze, and P. Dorsey.

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

ABSTRACT

Across the country conflicts among people and black bears are increasing in frequency and severity, and have become a high priority wildlife management issue. Whether increases in conflicts reflect recent changes in bear population trends or just bear behavioral shifts to anthropogenic food resources, is largely unknown, with key implications for bear management. This issue has generated a pressing need for bear research in Colorado and has resulted in a unique collaboration that builds on the resources and abilities of personnel from 5 entities: the Colorado Parks and Wildlife (CPW), the USDA National Wildlife Research Center, Colorado State University, Wildlife Conservation Society, and Bear Trust International. Collectively, we completed year 1 of a 5-year study on black bears that 1) tests management strategies for reducing bear-human conflicts, 2) determines the influence of urban environments on bear habitat-use patterns and demography, 3) identifies public attitudes and perceptions about bears, bear management and bear-human encounters, and 4) develops population and habitat models to support the sustainable monitoring and management of bears in Colorado. This project was initiated in FY10-11; during this past fiscal year we have primarily focused on coordinating research logistics and collecting field data in the vicinity of Durango, Colorado. Specifically, we obtained data on garbage-related bear-human conflicts, trapped and marked black bears, monitored the vital rates of collared bears (survival, fecundity and cub survival) through telemetry and winter den visits, collected data on the availability of late summer/fall mast, tracked human-related bear mortalities and removals, performed non-invasive genetic mark-recapture surveys, and conducted a survey of public attitudes and perceptions about bear-human encounters. Project collaborators will continue to seek additional funding to implement the remaining activities outlined in the research proposal (i.e., purchase additional containers for an urban-food-removal experiment, increase the sample size of collared bears, and acquire telemetry collars to test a translocation model). Information from this study will provide solutions for sustainably managing black bears *outside* urban environments, while reducing bear-human conflicts *within* urban environments; knowledge that is critical for wildlife managers in Colorado and across the country.

WILDLIFE RESEARCH REPORT

BLACK BEAR EXPLOITATION OF URBAN ENVIRONMENTS: FINDING MANAGEMENT SOLUTIONS AND ASSESSING REGIONAL POPULATION EFFECTS

HEATHER E. JOHNSON

P.N. OBJECTIVES

To conduct a study on black bears in Colorado that 1) tests management strategies for reducing bear-human conflicts, 2) determines the influence of urban environments on bear habitat-use patterns and demography, 3) identifies public attitudes and perceptions about bears, bear management and bear-human encounters, and 4) develops population and habitat models to support the sustainable monitoring and management of bears.

SEGMENT OBJECTIVES

1. Work with personnel from CPW Area 15, CPW Southwest Region, the City of Durango, La Plata County, US Forest Service (Columbine and Pagosa Ranger Districts), Bureau of Land Management (BLM; Tres Rio Field Office), and private landowners on field research logistics.
2. Collect pre-treatment data on the frequency of bears accessing human garbage in preparation for an urban bear-proofing experiment.
3. Trap and collar adult female black bears in the vicinity of Durango to collect data on bear habitat-use patterns and demography.
4. Monitor bear survival via global position system (GPS) collar locations.
5. Obtain data on summer/fall natural food availability for bears based on the phenology and abundance of gambel oak, serviceberry, chokecherry, hawthorne, pinyon pine and squaw apple.
6. Investigate the winter dens of collared female bears to collect data on fecundity and cub survival, inspect collar fit, and replace collar spacers and batteries.
7. Track human-related bear mortalities and removals around Durango from lethal conflict mortalities, vehicle collisions, harvest, and translocations.
8. Perform non-invasive genetic mark-recapture surveys to estimate bear density and population size around Durango (urban site) and in the Piedra watershed (wildland site).
9. Conduct a survey of public attitudes and perceptions about bears, the local bear population, bear management and bear-human encounters.

INTRODUCTION

In Colorado and across the country, conflicts among people and black bears (*Ursus americanus*) appear to be increasing in number and severity (Hristienko and McDonald 2007, Baruch-Mordo et al. 2008, CPW unpublished data). Bear-human conflicts can result in public safety concerns, property damage, bear mortality (i.e. euthanasia), and high management costs, and thus, have become a critical wildlife management issue. While wildlife agencies have used a variety of tools to try to minimize bear-human conflicts (i.e., education, aversive conditioning of bears, and modifications to harvest), conflict rates have continued to rise. Whether increases in bear-human conflicts reflect recent changes in the bear population or just behavioral shifts to anthropogenic food resources, is largely unknown, as bear population parameters have been exceeding difficult to estimate (Garshelis and Hristienko 2006). Without a thorough understanding of the relationship between conflict rates and bear behavior and population dynamics, it has been difficult for wildlife agencies to successfully reduce conflicts through bear management.

While there is uncertainty about how to reduce bear-human conflicts, two key factors thought to exacerbate this problem are expanding human development and climatic variation. Colorado has had one of the highest rates of exurban development in the nation (Theobald and Romme 2007), and this development has resulted in additional human food on the landscape in the form of garbage, agricultural resources, fruit trees, etc. The availability of human food to bears has been identified as the primary cause of bear-human conflicts (Spencer et al. 2007, Beckmann et al. 2008, Greenleaf et al. 2009), as bears are opportunistic foragers that will readily take advantage of this resource. Bear-use of human food not only increases interactions between bears and people but has been found to alter bear activity patterns, foraging behavior, movement rates, and even survival and reproductive rates (Beckmann and Berger 2003a, Beckmann and Berger 2003b, Hostetler et al. 2009), having the potential to significantly alter both bear behavior and demography. This phenomenon is further complicated by variation in annual weather patterns, as bear-use of human development appears to increase when natural foods are in short supply (Zack et al. 2003, Baruch-Mordo et al. 2010). Because bears predominately consume vegetation, recent patterns of drought in Colorado have caused natural food failures in some years. As a result, bears may be increasing their reliance on human foods, with associated behavioral and demographic impacts. While the effects of urbanization and climate have critical implications for modifying bear-habitat relationships, they also have critical implications for increasing rates of bear-human conflicts. To develop successful strategies to reduce conflicts while maintaining viable bear populations, wildlife agencies must understand how factors such as climate, natural food availability, human food ability, and management influence the behavior and dynamics of bear populations.

To address these questions, Colorado Parks and Wildlife has partnered with the USDA National Wildlife Research Center, Wildlife Conservation Society, Colorado State University, and Bear Trust International. Collectively, we initiated a project in FY10-11 to 1) test management strategies for reducing bear-human conflicts, 2) determine the influence of urban environments on bear habitat-use patterns and demography, 3) identify public attitudes and perceptions about bears, bear management and bear-human encounters, and 4) develop population and habitat models to support the sustainable monitoring and management of bears in Colorado (Johnson et al. 2011). This information should provide solutions for sustainably managing black bears *outside* urban environments, while reducing bear-human conflicts *within* urban environments; knowledge that is critical for wildlife managers in Colorado and across the west.

During FY11-12 we worked with internal and external stakeholders on field research logistics, obtained data on garbage-related bear-human conflicts, trapped and marked black bears, monitored the vital rates of collared bears (survival, fecundity and cub survival) through telemetry and winter den visits, collected data on the availability of late summer/fall mast, tracked human-related bear mortalities and removals, performed non-invasive genetic mark-recapture surveys, and conducted a survey of public attitudes and perceptions about bear-human encounters. Our efforts focused largely on collecting field data to meet research objectives 1-3, information which will eventually be used to address objective 4. We report general summary information from field activities over the past year; detailed analyses of field data will occur in future years.

STUDY AREA

To meet study objectives, a combination of site-specific field data and statewide data will be required. Site-specific field data is being collected in the vicinity of Durango, and is the focus of this progress report. Regional and statewide analyses will be conducted in future years. The town of Durango contains ~17,000 people (within city limits) and sits at 1,985 m along the Animas river valley. The town is surrounded by mountainous terrain ranging in elevation from ~1,930 to ~3,600 m, and is generally characterized by mild winters and warm summers that experience monsoon rains. Vegetation in the region is dominated by ponderosa pine, oak, pinyon-juniper, aspen, mountain shrub, and agricultural communities. Key forage species for black bears include gambel oak (*Quercus gambelii*), chokecherry (*Padus virginiana*), serviceberry (*Amelanchier alnifolia*), hawthorne (*Crataegus spp*), squaw apple (*Peraphyllum ramosissimum*), angelica (*Angelica spp*), sweet cicily (*Osmorhiza spp*), cow parsnip (*Heracleum sphondylium*) and waterleaf (*Hydrophyllum spp*). Durango is predominately surrounded by public land managed by the San Juan National Forest, BLM, CPW, La Plata County and the City of Durango. The vicinity of Durango is considered high quality bear habitat, and the town has consistently experienced high rates of bear-human conflicts.

METHODS

Objective 1: Testing management strategies to reduce bear-human conflicts

Given that the primary cause of black bear-human conflicts has been attributed to the availability of human foods to bears, it has been suggested that the most effective strategy to reduce conflicts is to reduce the availability of that food source (Peine 2001, Beckmann et al. 2004, Gore et al. 2005, Spencer et al. 2007). This strategy has had some success within national parks (Greenleaf et al. 2009), and anecdotally in some communities (Mammoth Lakes CA, Juneau AK, Whistler BC), but no research has ever scientifically tested the benefits of “cleaning up” a town. Given the high price to operationally “bear-proof” a community, municipalities must have definitive evidence that such an effort would significantly decrease conflict activity before initiating major changes to waste storage and collection practices.

As part of this project we will be implementing the first experimental test of wide-scale urban bear-proofing for reducing bear-human conflicts. To do this, we will drastically reduce the accessibility of anthropogenic foods known to attract bears (garbage, bird-feeders, pet food, etc) within 2 designated ‘treatment’ areas, while simultaneously monitoring 2 comparable ‘control’ areas where no action will occur. In the treatment areas we will provide bear-proof garbage containers, canvass citizens to discourage food outside of secure structures (bird-feeders, pet food, etc), conduct daily patrols to remove human foods, and provide strict enforcement. Each area will contain approximately 500 homes in residential neighborhoods. Treatment and control areas will be monitored for 3 years after the experiment has commenced, and we will track the number of conflicts and their severity among our experimental units. Conflicts will be recorded from weekly monitoring and from calls received by CPW, the City of Durango, and Bear Smart Durango (local non-profit organization).

During summer 2011 project personnel collected pre-treatment data (data collection for 2012 is ongoing) on bears accessing garbage in Durango. In July and August, months that experience the highest numbers of bear-human conflicts (CPW unpublished data), technicians patrolled each street within proposed treatment/control areas on the day waste removal was scheduled to occur (when maximum human food was assumed to be available to bears). Technicians conducted patrols from ~05:30 - 06:30 AM and recorded the locations where there was evidence that bears had obtained garbage or other human food sources. Additionally, during late July, we quantified the “availability” of garbage to bears, by documenting the location and container type (wildlife-resistant or regular) of every garbage receptacle in the survey area. These data will allow us to track changes in the number of wildlife-resistant containers over the course of the study, and provide an estimate of the amount of human food available to bears in town. In addition to collecting pre-treatment data, we worked with the City of Durango to coordinate the logistics of implementing the bear-proofing experiment in spring 2013.

Objective 2: Determining the influence of urban environments on bear behavior and demography

To sustainably manage bears in the face of a growing human population and changing landscape conditions, it is critical to elucidate the drivers and dynamics of bear populations. Of those factors that influence bear populations, the expansion of human development is the least understood, most contentious, and has the greatest potential to elicit major population change. To elucidate the influence of human development on bear habitat-use patterns and demography, we are collecting a suite of data types including locations from collared bears on the urban-wildland interface, survival and reproductive rates of those bears in conjunction with their habitat-use patterns, information on annual summer/fall mast production, and genetic data to estimate bear density in urban and wildland habitat types. We briefly describe data collection methods for this portion of the study below; detailed information is available in Johnson et al. (2011).

Collaring and Marking Bears – To assess bear movement and habitat-use patterns with respect to human development, we are capturing and collaring adult female bears. We are specifically targeting adult females as they represent the reproductive segment of the population and allow us to obtain information on multiple key vital rates. For example, in addition to being able to track adult female survival, the vital rate with the highest elasticity (Beston 2011), we can use collared females to track fecundity and cub survival, vital rates that are often associated with variation in bear population growth rates (Mitchell et al. 2009, Beston 2011).

We have targeted summer trapping efforts within ~10 km of the center of Durango to collar a cohort of bears that experience similar natural food availability, have anthropogenic food resources readily available, and encompass a range of behaviors and habitat-use patterns relative to the urban-wildland interface. Bears are trapped with box traps, which are baited with fish, fruit, human foods (at urban locations) and manufactured scents. Traps are set in the evening and checked the following morning. Adult female bears are fitted with a GPS collar manufactured by Vectronics, and a tooth is pulled for age verification. A collar records a bear's location every hour, and uploads a location to a central database via satellite system every 6 hours. Although trapping efforts are focused on adult females, all bears that are trapped (i.e., males, subadults, yearlings) are uniquely marked with a PIT and ear-tag and are weighed, measured, and sampled for blood and hair.

Evaluating Bear Movement and Habitat-Use Relative to the Urban-Wildland Interface – To examine movement and habitat-use patterns of bears along the urban-wildland interface we will use GPS collar location data from adult female bears. We will assess the influence of factors such as natural food availability, human food availability, weather, habitat covariates, and individual bear attributes (i.e., age, reproductive status) on bear behavior. During winter 2012, we downloaded hourly GPS location data from the collars during winter den checks, and will continue to download and process this data on an annual basis. We will use locations in conjunction with various types of spatial data to conduct a suite of movement and resource selection analyses (Manly et al. 2002, McLoughlin et al. 2010, Morales et al. 2010). In terms of spatial data, we will use satellite imagery to track annual spring/early summer forage availability, and ground surveys to track late summer/fall mast availability (see details below). Weather information will be modeled using PRISM spatial data (www.prism.oregonstate.edu/) which interpolates monthly temperature and precipitation patterns across landscapes, accounting for elevation and topography. Covariates related to human development will be derived from existing CPW digital data layers such as parcel density, road density, and census population size.

While most habitat and human development information can be extracted from existing spatial data sources, there is no existing data layer that tracks annual variation in late summer/fall hard and soft mast for bears. The abundance of acorn and berry resources for bears is known to be highly variable, depending on annual trends in precipitation and temperature (Noyce and Coy 1989). To account for variation in the availability of natural forage for bears around Durango we conducted weekly mast surveys. Surveys were performed between mid-August and mid-September in 2011, when fruits and nuts should reach peak maturation. In the Durango region, the key mast species for bears are gambel oak, chokecherry, serviceberry, hawthorne, squaw apple, and pinyon pine (Beck 1991, Tom Beck, personal communication). We randomly selected 12 transects on public lands to evaluate bear natural food availability. Each transect was 1 km in length and was situated along an existing trail or stream drainage. For each transect, field technicians recorded the phenological stage and the percentage of plants of each species that exhibited mast in different abundance categories (mast failure, <25% of plants with mast, 25 – 50% of plants with mast, etc).

Estimating Demographic Rates – To assess the influence of human development on bear demographic rates and population trends we are using the following data types: 1) survival and reproduction of collared adult female bears, 2) mortalities and removals from marked and unmarked bears in the vicinity of Durango, and 2) samples from non-invasive genetic surveys of bears around Durango and in the Piedra watershed.

Collared female bears allow us to track annual survival, fecundity and cub survival, parameters we monitored in FY11-12 and which we will continue to monitor for the next 4 years. We used real-time GPS collar locations to assess adult female survival, investigating mortalities and slipped collars when GPS locations were stationary for multiple sampling points. Fecundity and cub survival were monitored from den checks of collared females. Numbers of newborn cubs provide information to estimate fecundity rates, while repeated annual den checks of collared females allow us to estimate cub survival. Yearlings hibernate with their mothers, so we can observe the number of cubs alive in the den in year t that have survived their first year of life to $t+1$. Adult female survival, fecundity and cub survival will be collectively used in projection models to assess population performance in future analyses (Caswell 2001).

In addition to tracking survival and reproduction of collared bears, we are also tracking survival and cause-specific mortality of marked and unmarked bears in the study area. All bears that are trapped are marked with an ear-tag and PIT tag, unique identifiers that we are using to collect data on human-related bear mortalities and removals. Mortalities and removals primarily occur from translocations, vehicle collisions, conflict-related euthanasia and harvest. For all bears removed from the study area we collected a hair and tooth sample and recorded the date, mortality/removal cause, location, bear age, sex, weight, and morphological measurements. We will use mark-recapture and recovery analyses to estimate adult male survival and subadult survival, while also gaining valuable information on cause-specific bear mortality around human development.

To better understand the influence of urban environments on bear density and population sizes, we are employing non-invasive genetic sampling (Woods et al. 1999, Mowat and Strobeck 2000) to compare these parameters between a bear population around the urban center of Durango and in a nearby "wildland" area. For each area we identified a 36 cell grid (576 km^2) where each cell was 4 x 4 km in size; we constructed 1 snare in each cell. Snares consisted of a scented bait hanging high in a tree, surrounded barbed wire around a cluster of trees encircling the bait. When the bears climbed over or under the wire to investigate the bait, they left a hair sample on the barbed wire. In summer 2011 we hung a single strand of barbed wire (50 cm high), and on the other half of the snares we hung two strands (50 and 20 cm high). Our goal with this design was to determine whether the additional strand of wire increased capture probability. In summer 2012 all sites were strung with a single strand of wire. Snares were deployed during the first 2 weeks of June, and we conducted 6 weekly sampling occasions thereafter. On each occasion, we randomly re-baited the snare with anise, strawberry, fish, maple or bacon scent, and collected hair samples off all barbs. Each hair sample was uniquely catalogued according to the site, date, occasion, and barb number.

In 2011, we sampled a total of 31 grid cells in Durango (dropping 5 cells where public or motorized access was prohibited) and 9 cells in the Piedra watershed. We did not have the logistical capability to sample both grids in their entirety, so we ran a pilot study on the Piedra to determine whether twice/month sampling (as opposed to weekly) would have significant impacts on DNA quality, DNA contamination (hair samples from >1 bear/barb), and recapture rate. In 2012, we constructed 35 snares in the Durango grid and 34 snares in the Piedra grid. The layout of the Piedra grid had to be modified in

2012 to account for closures associated with the Little Sand fire, which began burning on May 13th 2012 (Figure 1). This modification can be easily accounted for in future analyses with spatially-explicit mark recapture statistics (Efford et al. 2009, Gardner et al. 2010).

In fall 2011, all hair samples were sent to the laboratory at Wildlife Genetics International for genotyping; genetic results were returned at the end of June 2012. Summary data from the Durango grid is provided, and the remainder of the analyses will occur during FY12-13. Samples collected in 2012 will be sent to the laboratory this fall.

Objective 3: Identifying public attitudes about bear-human encounters

Wildlife management agencies must identify the biological factors driving increases in bear-human conflicts, but they also must identify and incorporate human attitudes and perceptions about this issue into management strategies. This is particularly critical for black bears, as increasing bear-human conflicts around urban development have simulated significant public interest and concern. It is also critical because bear-human conflicts typically arise over bear-use of human foods, prompting investigators to suggest that a critical component of reducing conflicts is managing human behavior (Beckmann et al. 2004, Gore et al. 2008, Baruch-Mordo et al. 2011). Thus, in conjunction with Stacy Lischka, Human Dimensions Specialist for CPW, we have initiated a public survey to 1) better understand public perceptions about bears, bear management, and bear-human encounters and 2) explore motivations for compliance and non-compliance with wildlife ordinances designed to reduce bear-human conflicts. To meet those objectives, we developed a three part public mail survey to be conducted in conjunction with our urban bear-proofing experiment. Residents will be surveyed pre-, during, and post-implementation of the experiment, in treatment and control areas, as well as across a larger portion of the community. Surveys will be mailed to all residents within Durango city limits, and a subset of La Plata county residents within the study area. Survey responses will allow us to quantify current public attitudes and perceptions about bears, and how those perceptions change over time in association with a management effort such as wide-scale urban bear-proofing. The survey will also determine the number of residents that have had interactions with bears, the acceptability of management actions by CPW, and factors that promote or inhibit residents from complying with wildlife ordinances.

The pre-treatment survey was mailed to 5,852 residents; 4,352 residents in Durango city limits and 1,500 in surrounding areas of La Plata county (Appendix 1). The total valid sample, once surveys mailed to incorrect addresses were returned, was 5,329. Surveys were mailed on January 17th 2012, a reminder postcard was mailed on February 2nd 2012, and a second survey was mailed to non-respondents on February 29th 2012. For those people that did not send back a completed survey, we mailed a non-response postcard on May 18th 2012. The postcard had a few background questions so that any systematic biases in respondents could be assessed and incorporated into analyses (Appendix 2).

RESULTS AND DISCUSSION

Objective 1: Testing management strategies to reduce bear-human conflicts

During summer 2011 we collected pre-treatment data for the proposed bear-proofing experiment. We observed 129 instances of bears accessing garbage during our weekly surveys in July and August; observations peaked the first week of August. Of those garbage containers accessed by bears, 10% were wildlife-resistant and 90% were regular containers. Bears accessed human food from wildlife-resistant containers when they were not closed properly or the locking mechanism on the lid was broken. In quantifying the availability of garbage to bears, we recorded the location and container type of 1,167 garbage cans in the proposed treatment and control areas. Of those containers, 14% were wildlife resistant and 86% were regular (non-wildlife resistant). This demonstrates the limited residential bear-proofing that currently exists in Durango, and the relevance of conducting an experimental test of wide-scale urban bear-proofing in this community.

This past year, we have worked on the logistics of conducting a wide-scale urban bear-proofing experiment that should commence in spring 2013. A majority of the necessary funds were secured through CPW and the Summerlee Foundation; we are still seeking funds to complete project needs. With funds currently dedicated to the project, we purchased 760 wildlife-resistant containers from Solid Waste Systems (Parker, CO), a company that manufactures products certified by the Living with Wildlife Foundation. This fall, those containers will be fitted with electronic chips and entered into the Durango's Solid Waste Program database. Because all residential waste is removed by the City of Durango, city staff will replace regular garbage containers with the newly purchased wildlife-resistant containers according to CPW's study design. The wildlife-resistant containers will be distributed in late fall and winter after the bears have hibernated so that they are in place for the experiment in spring 2013.

Objective 2: Determining the influence of urban environments on bear behavior and demography

Between May 15th 2011 and August 15th 2012, a total of 162 different bears were marked as part of this study, during 287 bear captures. Information about these captures is described below for each discrete capture season: summer 2011, winter 2012, and summer 2012 (ongoing; Table 1).

During summer 2011 we conducted 92 total bear captures; 71 captures were unique individuals and 21 were recaptures. Of the unique individuals captured, there were 30 females, 38 males, and 3 cubs of unidentified sex (cubs were released without being immobilized and thus, gender was not determined; Table 1). We collared a total of 26 adult females, however two bears slipped out of their collars and were not recaptured, leaving 24 collared bears at the end of the field season. The mean estimated age of bears ≥1 year-old on their initial capture date was 5.3 (5.7 for females and 5.1 for males), and the mean weight was 80.8 kg (59.9 kg for females and 97.4 kg for males). The mean age of collared females, based on tooth cementum, was 6 years, and estimated ages ranged from 2.5 to 23. In total, we placed traps/snares at 102 different locations (26 on public land and 76 on private land) and we had 1,253 trap nights. Capture success generally peaked during the first couple weeks of June and was highly variable throughout the remainder of the summer (Figure 2).

We visited the winter dens of 22 collared females between January and March 2012. Although we had 24 adult female bears collared in fall 2011, 1 female was harvested (B49), and we could not locate the den of 1 bear wearing a Lotek collar (B51). Nine females did not have any cubs or yearlings, 3 bears had yearlings (6 yearlings in total) and 10 had newborn cubs (21 cubs in total; 11 females and 10 males). Of those females with yearlings, 1 bear had 1 yearling, 1 bear had 2 yearlings, and 1 bear had 3 yearlings. Of those females with newborn cubs, 1 had only 1 cub, 7 bears had twins, and 2 bears had triplets. We PIT and ear-tagged yearlings in the den, recorded information on weight and body size, and collected hair and blood samples. We also PIT tagged newborn cubs, and recorded their sex and weight. One collared bear (B43) died during the immobilization process in the den.

Between May 15th and August 15th 2012, we conducted summer captures to obtain a sample of 40 GPS collared adult females (captures are currently ongoing). During that time there were 153 total captures; 74 were unique individuals and 79 were recaptures (Table 1). Of the unique individuals captures, 33 bears were females, 37 were males, and 4 cubs were of unidentified sex (cubs were not immobilized). The mean estimated age of bears ≥1 year-old on their initial capture date was 4.7 (5.3 for females and 4.3 for males) and the mean weight was 72.3 kg (58.3 kg for females and 83.5 kg for males). This summer, to date, 22 new adult females have been collared. Given mortalities and slipped collars (3 collars were slipped in spring/summer 2012), 37 females were collared as of August 15th, and trapping will continue through mid-September or until 40 GPS collars have been deployed. To date, traps have been placed in ~78 different trap locations (26 on public land and 52 on private land) for approximately 1,024 trap nights. Capture success generally climbed each week until the second week of July, and has remained high (Figure 2). The increase in capture success in 2012 is likely due to extra trapping effort, as we increased our weekly trap nights from 5 nights/week to 7 nights/week and had a higher number of traps that were baited and set on a consistent basis.

Although we are still working to deploy collars for the study, the Vectronics GPS technology has been highly efficient at tracking collared bears for movement rates, habitat-use patterns, den site locations, and daily survival. To date, we have obtained >60,000 locations from 48 different female bears (Figure 3). Additionally, the GPS collar technology has allowed us to observed long-distance movements by females, particularly during the estrous period; data which has been rarely collected and reported. For example, this past June, 3 different collared bears traveled between ~60 and ~320 km in different directions from the study area (Figure 4). Two of the bears returned to their original home ranges, and 1 died in a vehicle collision as she appeared to be returning to Durango (B35).

In 2011, mast surveys revealed that the peak timing for serviceberry maturation was in mid-August, for chokecherries it was during the last week of August and first week of September, for squaw apples it was around September 1st, and for acorns it was during in the first two weeks of September. Hawthorne berries and pinyon cones were only observed on 2 of 12 transects; neither had reached peak maturation by mid-September. Across the transects, on average, <25% of gambel oak, chokecherry, squaw apple, and hawthorn plants had mast production. Serviceberry and pinyon production was categorized as a complete failure for the year.

Between 1 May 2011 and August 15th 2012, 25 bears were removed from the vicinity of Durango due to non-harvest, human-related causes. Of those bears that were removed, 9 were lethally removed due to nuisance behavior (breaking into houses, killing livestock, etc), 10 were killed in vehicle collisions (including 2 collared females), 4 were translocated due to conflicts with people (including 1 collared female), and 2 died from research activities (including 1 collared female). Of those mortalities and removals, 17 bears were unmarked and 8 were marked/collared for the research project (1 marked bear was a lethal conflict removal outside the study area); there were 8 adult females, 4 adult males, 2 subadult females, 7 subadult males, and 4 cubs. In addition, approximately 20 bears were harvested in the greater Durango area (GMUs 74, 75, and 751), three of which were marked by the research project (1 collared female and 2 adult males).

In summer 2011, we collected 998 hair samples from the Durango and Piedra hair-snare grids; 743 samples from Durango and 255 from Piedra. Over the 6 sampling occasions from 31 snares around Durango we collected 224, 167, 138, 77, 68, and 69 hair samples, respectively. Over the 3 sampling occasions from 9 snares in the Piedra we collected 127 samples; 46, 50, and 31 samples/occasion, respectively. We also collected 128 additional samples from 10 snares in the Piedra watershed that were only checked on a single occasion. We received the genetic results back from Wildlife Genetics International at the end of June 2012, and have summarized the Durango data. Of the 743 hair samples submitted to the laboratory, good genotypes were obtained for 438 samples. Of the remaining samples that did not produce a valid genotype, 193 did not contain enough genetic material, 104 failed during analyses for other reasons, 4 samples were not black bear, and 2 were contaminated (hair from >1 bear in the sample). Across the 438 valid samples there were 107 different individuals (61 females and 47 males) detected during 192 “captures” (multiple hair samples from a single bear during 1 sampling occasion were considered 1 “capture”). Of the different individuals, 21 were only detected in 1 sampling occasion and 86 were detected in >1 occasion (recaptures). The probability of detecting a bear within any single sampling occasion was ~0.21, and across all sampling occasions was ~0.76. More detailed analyses of these data will be included in the FY12-13 report.

In summer 2012, we collected 1,367 hair samples from the Durango and Piedra grids; 586 samples from Durango and 781 samples from Piedra. Over the 6 sampling occasions from 35 snares around Durango we collected 92, 136, 59, 55, 142, and 102 samples, respectively. Over the 6 sampling occasions from 34 sites in the Piedra watershed we collected 73, 135, 142, 118, 144, and 169 samples respectively. Samples will be sent to Wildlife Genetics International this fall for genetic analysis.

Objective 3: Identifying public attitudes about bear-human encounters

Of the 5,334 valid surveys that were mailed to residents, we received 2,947 completed surveys; 2,170 from Durango residents and 777 from La Plata county residents. The overall response rate was 55%. Non-response postcards were mailed to 2,375 residents and 354 postcards were returned (15%). Survey results are being electronically recorded so this data can be analyzed in FY12-13.

SUMMARY AND FUTURE PLANS

During FY11-12 we successfully coordinated field logistics and conducted several aspects of data collection (monitoring garbage-related bear-human conflicts, trapping and collaring bears, tracking human-related bear mortalities, assessing summer/fall forage availability, implementing DNA hair-snare surveys, and conducting a public survey). We will continue these field activities through 2015, and begin data analyses as field data are compiled. Project collaborators will continue to seek additional funding to implement the remaining activities outlined in the research proposal. These activities include the implementation of an urban bear-proofing experiment, increasing the number of GPS collared female bears, and purchasing telemetry collars for a translocation study. In addressing the objectives of this project we hope to better understand the influence of urban environments on bear populations, elucidate the relationship between bear-human conflicts and bear behavior and population trends, develop tools to promote the sustainable management of bears in Colorado, and ultimately, identify solutions for reducing bear-human conflicts in urban environments.

LITERATURE CITED

Baruch-Mordo, S., S.W. Breck, K.R. Wilson, and J. Broderick. 2011. The carrot or the stick? Evaluation of education and enforcement as management tools for human-wildlife conflicts. Plos One 6: e15681.

Baruch-Mordo, S., S.W. Breck, K.R. Wilson, and D.M. Theobald. 2008. Spatiotemporal distribution of black bear-human conflicts in Colorado, USA. Journal of Wildlife Management 72:1853-1862.

Baruch-Mordo, S., K.R. Wilson, D. Lewis, J. Broderick, J. Mao, and S.W. Breck. 2010. Roaring Fork Valley urban black bear ecology study: progress report to the Colorado Division of Wildlife. Contact Sharon Baruch-Mordo for copy. Email: sharonb_m@yahoo.com

Beck, T.D.I. 1991. Black bears of west-central Colorado. Technical Publication No. 39, Colorado Division of Wildlife, Colorado.

Beckmann, J.P., and J. Berger. 2003a. Using black bears to test ideal-free distribution models experimentally. Journal of Mammalogy 84:594-606.

Beckmann, J.P., and J. Berger. 2003b. Rapid ecological and behavioural changes in carnivores: the response of black bears (*Ursus americanus*) to altered food. Journal of Zoology 261:207-212.

Beckmann, J.P., L. Karasin, C. Costello, S. Matthews, and Zoë Smith. 2008. Coexisting with black bears: perspectives from four case studies across North America. Wildlife Conservation Society Working Paper No 33.

Beckmann, J.P., C.W. Lackey, and J. Berger. 2004. Evaluation of deterrent techniques and dogs to alter behavior of “nuisance” black bears. Wildlife Society Bulletin 32: 1141-1146.

Beston, J.A. 2011. Variation in life history and demography of the American black bear. Journal of Wildlife Management 75:1588-1596.

Caswell, H. 2001. Matrix population models: construction, analysis, and interpretation. Second Edition, Sinauer Associates, Sunderland, Massachussetts.

Efford, M.G., D.K. Dawson, and D.L. Borchers. 2009. Population density estimated from locations of individuals on a passive dector array. Ecology 90:2676-2682.

Garshelis, D.L., and H. Hristienko. 2006. State and provincial estimates of American black bear numbers versus agency assessments of population trend. Ursus 17:1-7.

Gardner, B., J.A. Royle, M.T. Wegan, R.E. Rainbolt, P.D. Curtis. 2010. Estimating black bear density using DNA data from hair snares. Journal of Wildlife Management 74:318-325.

Gore, M.L., W.F. Siemer, J.E. Shanahan, D. Schuefele, and D.J. Decker. 2005. Effects on risk perception of media coverage of a black bear-related human fatality. Wildlife Society Bulletin 33:507-516.

Gore, M.L., B.A. Knuth, C.W. Scherer, and P.D. Curtis. 2008. Evaluating a conservation investment designed to reduce human-wildlife conflicts. Conservation Letters 1:136–145.

Greenleaf, S.S., S.M. Matthews, R.G. Wright, J.J. Beecham, and H.M. Leithead. 2009. Food habitat of American black bears as a metric for direct management of human-bear conflict in Yosemite Valley, Yosemite National Park, California. Ursus 20:94-101.

Hostetler, J.A., J.W. McCown, E.P. Garrison, A.M Neils, M.A. Barrett, M.E. Sunquist, S.L. Simek, and M.K. Oli. 2009. Demographic consequences of anthropogenic influences: Florida black bears in north-central Florida. Biological Conservation 142:2456-2463.

Hristienko, H., and J.E. McDonald Jr. 2007. Going into the 21st century: a perspective on trends and controversies in the management of the American black bear. Ursus 18:72-88.

Johnson, H.E, C.J. Bishop, M.W. Alldredge, J. Brodrick, J. Apker, S. Breck, K. Wilson, and J. Beckmann. 2011. Black bear exploitation of urban environments: finding management solutions and assessing regional population effects. Research Proposal, Colorado Division of Parks and Wildlife, Fort Collins, USA.

Manly, B.F.J., L.L. McDonald, D.L. Thomas, T.L. McDonald, and W.P. Erickson. 2002. Resource selection by animals; statistical design and analysis for field studies. Second Edition. Kluwer Academic Publishers, Boston, Massachussetts.

McLoughlin, P.D., D.W. Morris, D. Fortin, E. Vander Wal, and A.L. Contasti. 2010. Considering ecological dynamics in resource selection functions. Journal of Animal Ecology 79:4-12.

Mitchell, M.S., L.B. Pacifici, J.B. Grand, and R.A. Powell. 2009. Contributions of vital rates to growth of a protected population of American black bears. Ursus 20:77-84.

Morales, J.M., P.R. Moorcroft, J. Matthiopoulos, J.L. Frair, J.G. Kie, R.A. Powell, E.H. Merrill, and D.T. Haydon. 2010. Building the bridge between animal movement and population dynamics. Philosophical Transactions of the Royal Society Series B 365:2289-2301.

Mowat, G., and C. Strobeck. 2000. Estimating population size of grizzly bears using hair capture, DNA profiling, and mark-recapture analysis. Journal of Wildlife Management 64:183-193.

Noyce, K.V., and P.L. Coy. 1989. Abundance and productivity of bear food species in different forest types of northcentral Minnesota. International Conference on Bear Research and Management 8:169-181.

Peine, J.D. 2001. Nuisance bears in communities: Strategies to reduce conflicts. Human Dimensions of Wildlife 6:223-237.

Spencer, R.D., R.A. Beausoleil, and D.A. Martorello. 2007. How agencies respond to human–bear conflicts: a survey of wildlife species in North America. Ursus 18: 217–229.

Theobald, D.M., and W.H. Romme. 2007. Expansion of the US wildland-urban interface. Landscape and Urban Planning 83:340-354.

Woods, J.G., D. Paetkau, D. Lewis, B.N. McLellan, M. Proctor, and C. Strobeck. 1999. Genetic tagging of free-ranging black and brown bears. Wildlife Society Bulletin 27:616-627.

Zack, C.S., B.T. Milne, and W.C. Dunn. 2003. Southern oscillation index as an indicator of encounters between humans and black bears in New Mexico. Wildlife Society Bulletin 31:517-520.

Prepared by ____
Heather E. Johnson, Wildlife Researcher

Table 1. Capture information for black bears that have been marked in the vicinity of Durango, CO (collared adult females are identified with an "*"). Only information from the initial capture of each individual is shown (no recaptures).

Bear ID	Capture Date	UTM Easting	UTM Northing	Sex	Age	Kg
B1	5/10/2011	246233	4142768	M	1	35.4
B2	5/12/2011	271495	4130889	M	9	144.2
B3	5/13/2011	271495	4130894	M	6	130.2
B4	5/16/2011	270950	4127914	M	3	84.4
B5	5/16/2011	270227	4139984	M	6	135.2
B6*	5/17/2011	243210	4128716	F	4	62.6
B7*	5/17/2011	243225	4133053	F	4	63.5
B8*	5/18/2011	271478	4130892	F	4	51.7
B9	5/26/2011	238803	4126790	M	1	34.9
B10*	5/26/2011	269869	4139040	F	7	80.7
B11	6/3/2011	252163	4137968	M	8	130.2
B12	6/2/2011	253216	4137387	M	5	103.4
B13	6/3/2011	253216	4138868	M	3	59.0
B14*	6/6/2011	252157	4137967	F	7	58.1
B15	6/6/2011	253216	4138868	M	3	58.1
B16	6/7/2011	253216	4138868	M	7	117.0
B17*	6/7/2011	256936	4134633	F	4	51.7
B18*	6/8/2011	256918	4134625	F	8	61.7
B19	6/9/2011	235193	4128894	M	9	146.5
B20	6/9/2011	243258	4133040	M	10	131.5
B21*	6/10/2011	252298	4136435	F	8	69.4
B22	6/10/2011	252163	4137968	M	8	87.5
B23	6/13/2011	246350	4135617	M	3	64.9
B24*	6/14/2011	243252	4133030	F	7	64.9
B25*	6/15/2011	239003	4134158	F	4	63.5
B26	6/15/2011	252164	4137966	M	10	108.9
B27*	6/16/2011	243252	4133030	F	23	75.3
B28	6/16/2011	253233	4138873	M	6	101.2
B29	6/21/2011	239840	4126949	M	1	49.0
B30*	6/22/2011	235911	4128916	F	6	60.3
B31	6/24/2011	239840	4126949	M	4	85.3
B32	6/24/2011	243252	4133030	F	1	19.1
B33	6/28/2011	239294	4133260	M	1	34.9
B34	6/28/2011	239001	4134154	M	3	85.3
B35*	7/5/2011	246350	4135617	F	3	44.5
B36	7/6/2011	239840	4126949	M	4	67.1
B37	7/7/2011	243252	4133030	M	1	39.0
B38	7/13/2011	243236	4128710	M	8	145.1
B39	7/13/2011	251222	4133120	M	6	149.7
B40*	7/21/2011	248550	4131645	F	4	80.7

B41	7/22/2011	237368	4132272	M	3	67.1
B42*	7/26/2011	245945	4141391	F	8	69.9
B43*	8/3/2011	246183	4142791	F	11	85.3
B44	8/3/2011	765141	4132487	M	1	35.4
B45	9/3/2011	245965	4139587	M	6	176.0
B46*	8/5/2011	243435	4128720	F	4	58.1
B47*	8/8/2011	251783	4131581	F	6	53.5
B48	8/10/2011	245914	4139620	F	1	26.3
B49*	8/11/2011	243435	4128720	F	4	55.3
B50*	8/11/2011	245965	4139587	F	11	101.2
B51*	8/12/2011	249049	4130370	F	12	61.7
B52*	8/12/2011	245965	4139587	F	4	65.3
B53	8/15/2011	243435	4128720	M	7	163.3
B54	8/16/2011	251898	4130516	M	2	52.6
B55*	8/18/2011	251464	4134423	F	3	49.0
B56	8/29/2011	246321	4132993	M	10	166.9
B57*	8/30/2011	243374	4135903	F	3	46.3
B58	8/31/2011	243374	4135903	M	3	48.1
B59	9/1/2011	243952	4132935	M	15	153.3
B60	9/2/2011	242187	4133020	F	2	35.4
B61	9/3/2011	244602	4130321	M	7	213.6
B62	9/6/2011	245790	4128530	M	1	22.7
B63	9/7/2011	248612	4131251	M	2	37.2
B64	8/6/2011	245850	4141969	M	1	28.6
B65*	9/15/2011	243948	4134848	F	4	89.8
B66	9/20/2011	240731	4130163	F	1	39.9
B67*	9/21/2011	256930	4134626	F	3	52.6
B68	9/21/2011	249067	4133006	M	8	207.7
B95	1/19/2012	247647	4142276	M	1	44.0
B96	1/19/2012	247647	4142276	M	1	44.0
B97	1/19/2012	247647	4142276	M	1	49.0
B98	1/26/2012	257183	4134879	M	1	20.9
B99	2/27/2012	249929	4137615	M	cub	1.1
B100	2/27/2012	249929	4137615	F	cub	1.2
B101	2/29/2012	248090	4126214	M	cub	1.8
B102	2/29/2012	248090	4126214	F	cub	1.7
B103	3/1/2012	243713	4120831	M	cub	1.9
B104	3/1/2012	243713	4120831	M	cub	1.7
B105	3/6/2012	240146	4132714	F	cub	2.0
B106	3/8/2012	268055	4139774	F	cub	2.7
B107	3/8/2012	268055	4139774	F	cub	2.7
B108	3/14/2012	245785	4138519	F	cub	1.4
B109	3/14/2012	245785	4138519	M	cub	1.4
B110	3/15/2012	244547	4140021	M	cub	2.5
B111	3/15/2012	244547	4140021	F	cub	2.5

B112	3/15/2012	244547	4140021	F	cub	2.6
B113	3/17/2012	245228	4143164	F	cub	2.5
B114	3/17/2012	245228	4143164	M	cub	2.0
B115	3/17/2012	245228	4143164	M	cub	2.0
B116	3/20/2012	240909	4134002	M	cub	2.0
B117	3/20/2012	240909	4134002	F	cub	2.0
B118	3/22/2012	243524	4146585	M	cub	2.5
B119	3/22/2012	243524	4146585	F	cub	2.9
B120	5/27/2012	254732	4133249	F	1	20.9
B121*	5/29/2012	251670	4132767	F	4	76.2
B122*	5/30/2012	249059	4132998	F	5	66.2
B123	6/5/2012	240102	4128939	M	2	48.1
B124*	6/6/2012	249158	4127065	F	7	80.7
B125*	6/8/2012	244618	4132132	F	8	98.9
B126	6/8/2012	251670	4132767	F	1	15.9
B127*	6/10/2012	239005	4134459	F	10	58.1
B128*	6/11/2012	239005	4134159	F	8	56.2
B129*	6/14/2012	254576	4135043	F	6	54.4
B130	6/22/2012	250152	4127691	M	1	12.7
B131	6/23/2012	765047	4131635	M	6	111.6
B132	6/28/2012	765047	4131635	M	1	20.4
B133*	6/29/2012	765932	4127651	F	3	49.0
B134*	6/30/2012	765932	4127651	F	8	90.7
B135	6/30/2012	252014	4133509	M	1	20.9
B136	7/1/2012	765047	4131635	F	2	46.3
B137	7/5/2012	249059	4132998	M	2	26.8
B138	7/5/2012	254997	4135825	F	2	30.8
B139	7/5/2012	238245	4131204	M	1	30.4
B140	7/5/2012	763921	4132873	M	3	67.1
B141*	7/5/2012	765132	4132506	F	3	55.3
B142	7/6/2012	254997	4135825	M	3	37.2
B143*	7/6/2012	241210	4137115	F	3	45.4
B144*	7/7/2012	238245	4131204	F	9	72.6
B145*	7/7/2012	763921	4132873	F	6	70.3
B146	7/7/2012	254739	4133234	M	5	110.2
B147	7/10/2012	241334	4138018	M	1	43.5
B148	7/11/2012	255983	4135921	M	3	73.9
B149	7/15/2012	244618	4132132	M	1	45.4
B150	7/16/2012	241334	4138018	M	2	53.5
B151	7/26/2012	243888	4129546	M	3	60.3
B152*	7/17/2012	241210	4137114	F	8	99.8
B153	7/17/2012	249059	4132998	M	2	49.9
B154	7/17/2012	253439	4134693	M	3	63.5
B155	7/19/2012	241334	4138018	F	2	30.8
B156	7/19/2012	252621	4130532	F	1	26.3

B157	7/19/2012	248417	4144294	M	6	136.1
B158	7/20/2012	252546	4134789	M	3	50.8
B159	7/21/2012	242236	4127920	F	2	39.9
B160	7/24/2012	249059	4132998	M	2	64.0
B161*	7/25/2012	242546	4134789	F	5	76.2
B162	7/25/2012	249059	4132998	M	10	108.0
B163	7/26/2012	243954	4134875	F	1	37.6
B164	7/28/2012	242611	4133863	M	6	117.0
B165*	7/29/2012	251815	4133706	F	12	85.3
B166	7/29/2012	252621	4130532	M	8	121.6
B167*	7/31/2012	248578	4139143	F	18	58.1
B168	7/31/2012	253439	4134693	M	4	87.5
B169	7/31/2012	249059	4132998	F	2	44.5
B170	8/1/2012	249059	4132998	M	4	119.3
B171	8/2/2012	248192	4137051	M	2	26.8
B172	8/2/2012	248578	4139143	F	2	28.1
B173*	8/3/2012	248578	4139143	F	5	73.9
B174*	8/3/2012	253341	4128740	F	3	43.5
B175*	8/3/2012	254916	4128609	F	10	76.2
B176	8/3/2012	252621	4130532	M	2	54.4
B177	8/4/2012	248578	4139143	M	7	93.9
B178	8/4/2012	249059	4132988	M	2	35.4
B179	8/5/2012	248578	4139143	M	1	35.4
B180*	8/5/2012	248939	4141533	F	3	71.7
B181*	8/5/2012	247127	4138557	F	3	58.1
B182	8/8/2012	259049	4132998	M	2	70.8
B190	8/9/2012	245293	4128959	M	12	153.3
B191	8/11/2012	249059	4132998	M	4	60.3
B192	8/11/2012	245293	4128959	M	9	148.8
B193	8/12/2012	243652	4129360	M	5	87.5
B194	8/12/2012	243218	4128712	M	5	137.0
B195	8/13/2012	259049	4132998	M	6	151.0
B196	8/14/2012	249059	4132998	F	2	42.2

Figure 1. Map of the locations of the 2012 hair snare sites for the Durango and Piedra non-invasive genetic sampling grids.

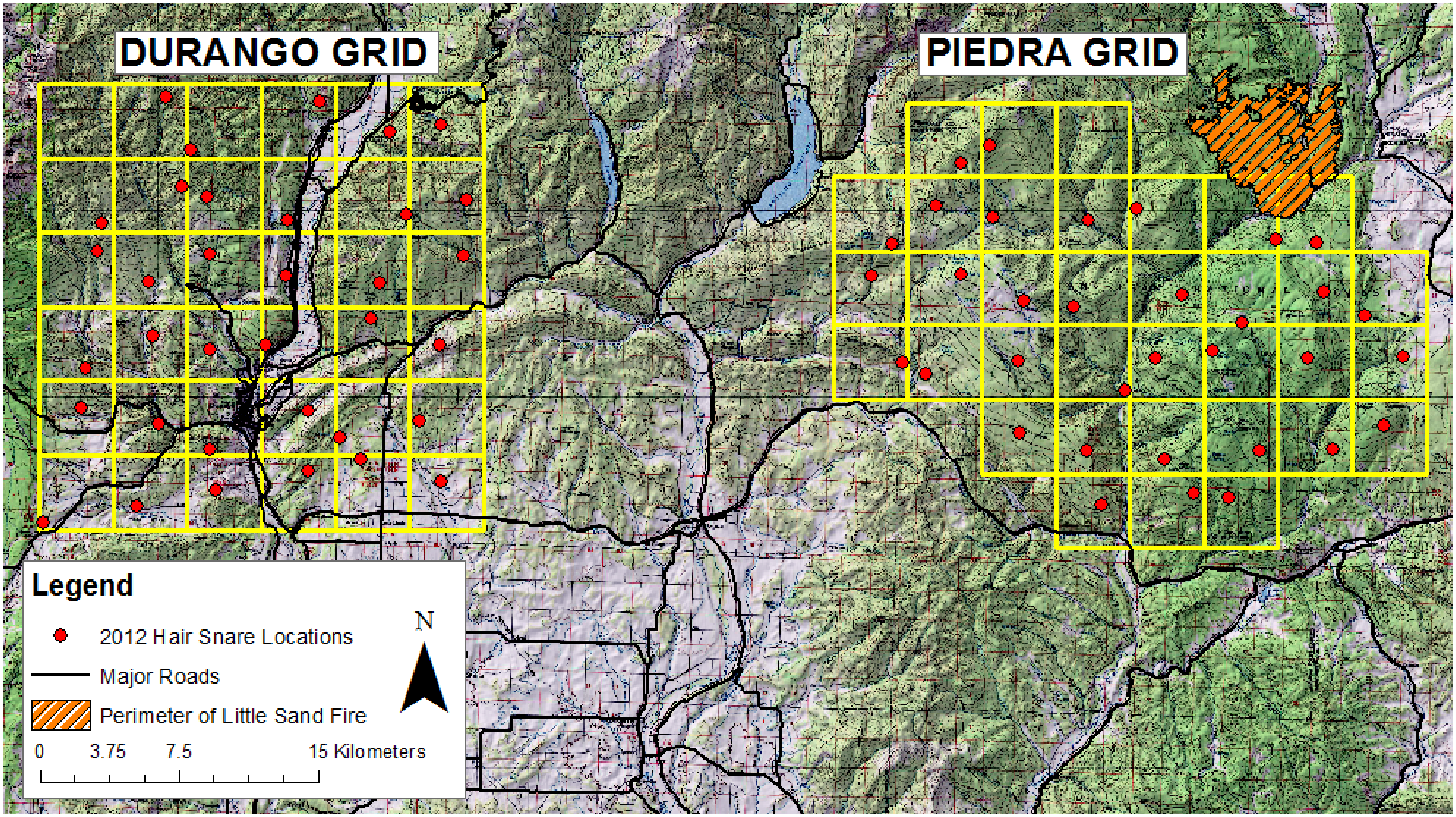


Figure 2. Number of black bear captures by week from May 15th through September 3rd for the 2011 and 2012 summer trapping seasons (2012 is currently ongoing).

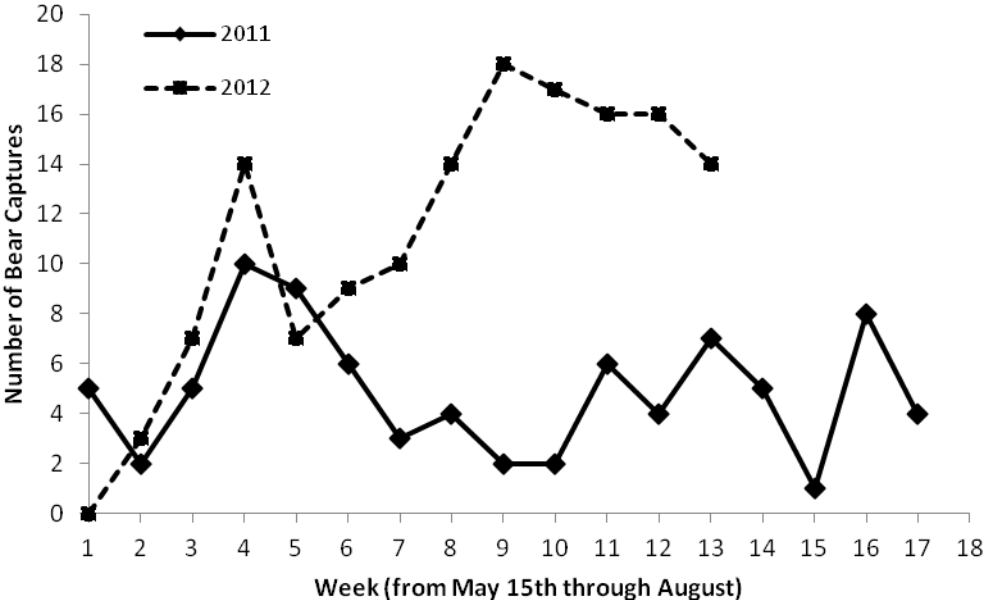


Figure 3. Locations from 48 adult female black bears collected with GPS collars from May 2011 to present in the vicinity of Durango, CO (different colored clusters of points represent different individual bears).

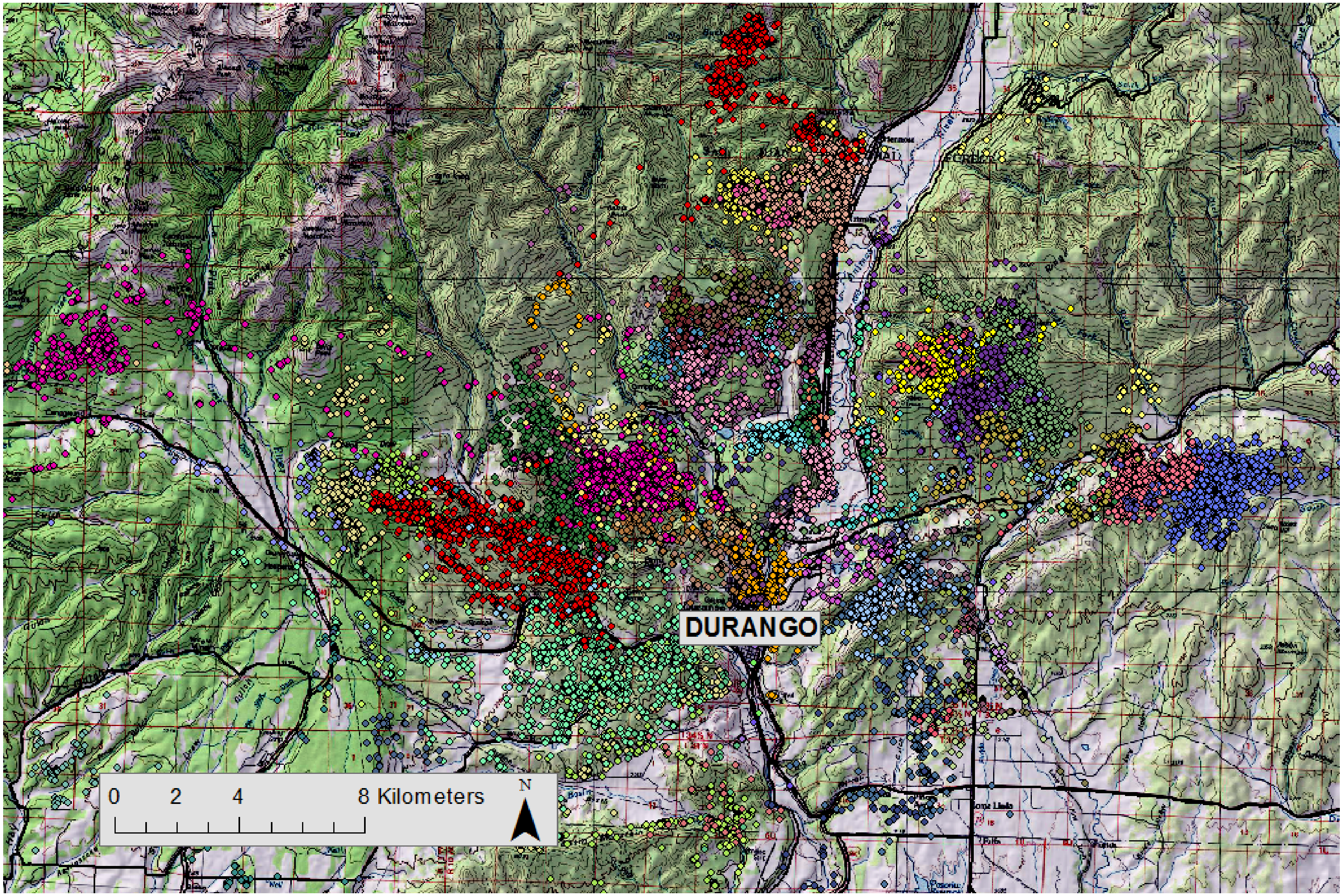
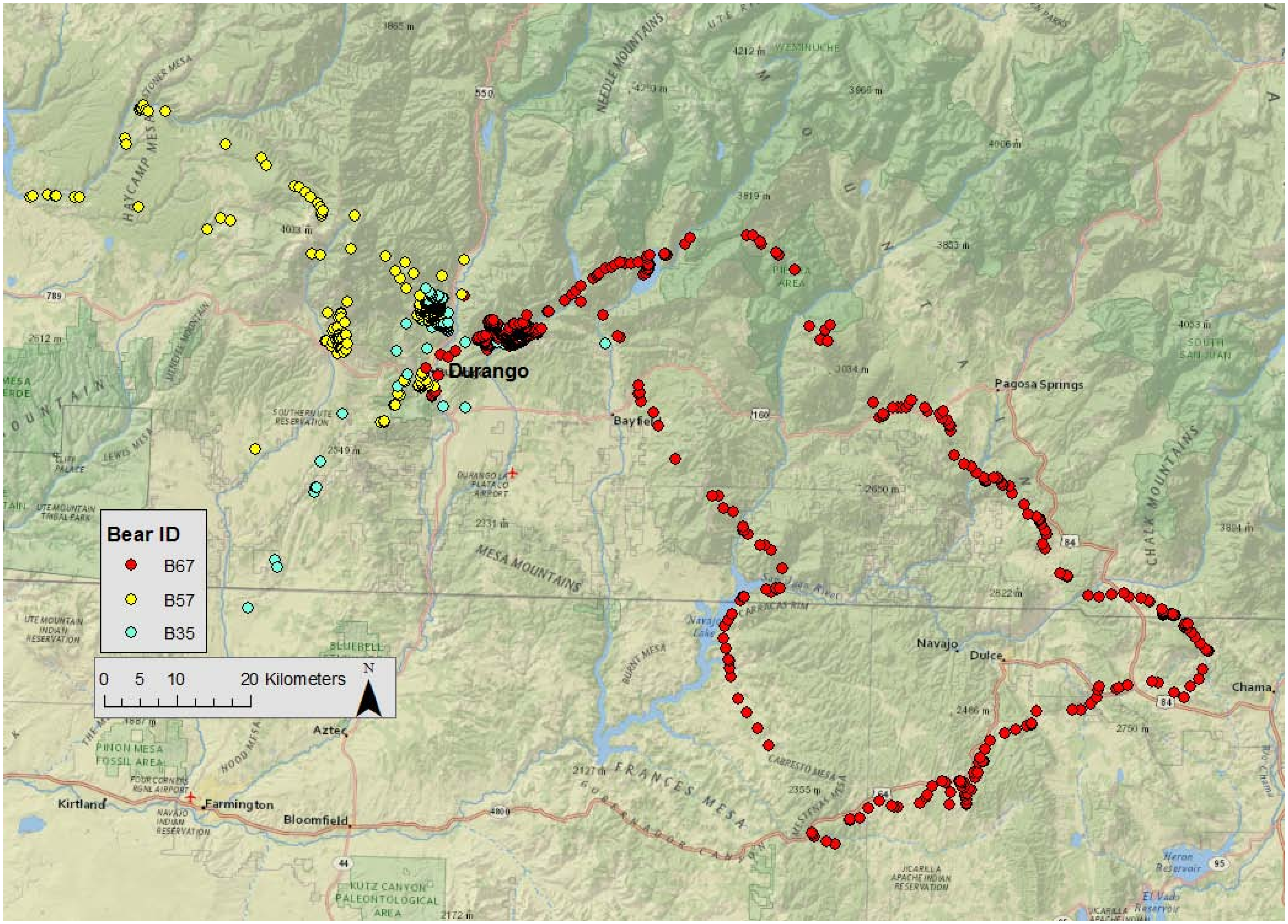


Figure 4. Long distance movements of three collared adult female bears during the breeding season in June 2012.



Appendix 1

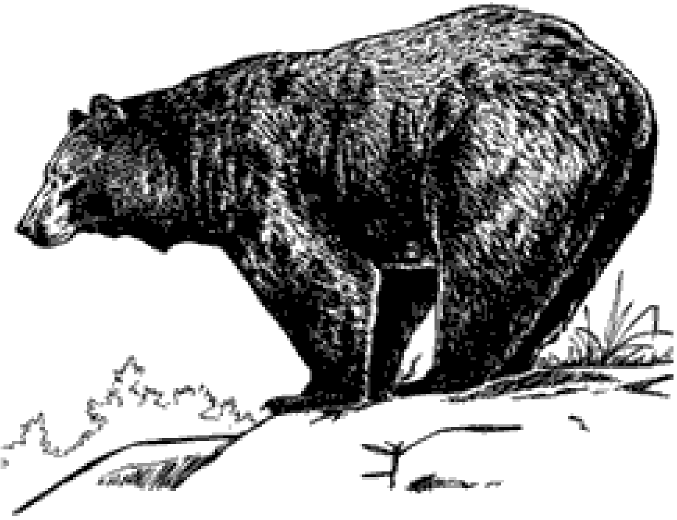
Living with Black Bears in Colorado:
A survey of your views

COLORADO PARKS & WILDLIFE

6060 Broadway • Denver, Colorado 80216
Phone (303) 297-1192 • FAX (303) 291-7109
wildlife.state.co.us • parks.state.co.us

Living with Black Bears in Colorado:

A survey of your views



THANK YOU FOR YOUR COOPERATION!
All of your responses will be kept confidential.
Please return this survey in the postage-paid return envelope provided.

Living with Black Bears in Colorado

This questionnaire is part of a study to help wildlife managers learn what residents think about black bears in Colorado. This survey is your chance to tell Colorado Parks and Wildlife (CPW) how you interact with black bears and how you would like to see black bear populations managed. Results of this study will be used to help wildlife managers address black bear-human interactions while sustaining Colorado's black bear populations. Your views are important and give us a better understanding of how residents feel about this issue. Please keep in mind that we are interested in everyone's responses, because the opinions of all Colorado residents living in black bear country are important.

You are part of a sample of Durango residents we have selected to provide opinions about black bear management. Your input is crucial for this evaluation. Even if you do not see black bears regularly or hold strong opinions about black bears, we still need to hear from you. Please complete this survey as soon as possible. When you are finished, please return it in the postage-paid envelope provided, **no later than March 30, 2012**. The survey should take about 20 minutes to complete. The final question provides you with an opportunity to share with us any additional comments you may have about black bears in Colorado.

Your responses will remain confidential and at no time will your name be associated with any of your responses.

If you have any questions or comments about this study, please contact Stacy Lischka at 303/291-7279 or by email at stacy.lischka@state.co.us.

THANK YOU FOR YOUR ASSISTANCE!

If you choose not to complete the questionnaire, please make a note in question 28 and return the survey in the postage-paid envelope included.

Wildlife and You. In its efforts to improve management of black bears in Colorado, Colorado Parks and Wildlife (CPW) wants to learn about how black bears affect the lives of Coloradans. In this section, please tell us a little about ways you interact with wildlife, black bears and land in Colorado.

1. The following are some ways that Coloradans interact with bears and other wildlife. Have you participated in these activities in the past 3 years? *(Please check one for each item.)*

	Yes	**No**		**Yes**	**No**
a. Read about bears or other wildlife	[ ]1	[ ]2	**e**. Hunted any species of wildlife	[ ]1	[ ]2
b. Photographed bears or other wildlife	[ ]1	[ ]2	**f.** Grew food or flowers in a garden	[ ]1	[ ]2
c. Closely observed or tried to identify birds or other wildlife	[ ]1	[ ]2	**g.** Worked on a farm or ranch	[ ]1	[ ]2
d. Hiked, biked, camped or backpacked in a natural area	[ ]1	[ ]2			

2. Have you ever hunted black bears in Colorado? *(Please check one.)*

[ ]1 Yes
[ ]0 No

3. Based on your experience, how has the number of black bears in the area where you live changed over the last 3 years? *(Please circle only one.)*

Increased greatly		Stayed the same		Decreased greatly	Not sure	No opinion
1	2	3	4	5	6	7

4. How would you like to see the number of black bears in the area where you live change in the next 3 years? *(Please circle only one.)*

Increase greatly		Stay the same		Decrease greatly	Not sure	No opinion
1	2	3	4	5	6	7

5. How important is it to you that the change in black bear populations you indicated in Question 4 occur over the next 3 years? *(Please circle only one.)*

Very important		Slightly important		Not at all important	Not sure	No opinion
1	2	3	4	5	6	7

6. People relate to wildlife in many ways; some of these relationships are listed below. Please indicate how strongly you agree or disagree with the following statements by checking one box for each item.

	Strongly agree	**Agree**	**Neither agree, nor disagree**	**Disagree**	**Strongly disagree**	**I am not sure.**
a. I tolerate most wildlife nuisance problems.	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
b. It is important to me to hunt game animals for recreation.	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
c. It is important to me to observe or photograph wildlife.	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
d. I appreciate the role wildlife plays in the environment.	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
e. I express opinions about wildlife and their management to wildlife managers or public officials.	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
f. It is important to me to know wildlife exist in Colorado.	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
g. It is important to me that wildlife are included in educational materials to learn about nature.	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
h. It is important to me to understand the behavior of wildlife.	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
i. It is important to me to hunt game animals for food.	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
j. I tolerate personal safety hazards associated with some wildlife.	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
k. It is important to me to talk about wildlife with family and friends.	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
l. Local economies benefit from the sale of equipment, supplies, or services related to wildlife recreation.	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
m. It is important to me to see wildlife in books and movies.	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
n. It is important to me that game animals are managed for harvest without risking the future of populations.	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$

Your Experiences with Black Bears. Coloradans come into contact with black bears in many ways and these interactions mean different things to each person. Please tell us about your interactions with black bears and what they mean to you.

7. How important is it to you to know that that bears live in this area and that their populations will continue to exist in the future? *(Please circle only one.)*

Very important		Slightly important		Not at all important	Not sure	No opinion
1	2	3	4	5	6	7

8. In your opinion, how important of an issue are negative interactions between humans and black bears where you live? *(Please circle only one.)*

Very important		Slightly important		Not at all important	Not sure	No opinion
1	2	3	4	5	6	7

9. Overall, how would you rate management of black bears and bear-human interactions in the area where you live? *(Please circle only one.)*

Excellent	Above average	Average	Below average	Poor	Not sure	No opinion
1	2	3	4	5	6	7

10. How often have you experienced the following interactions with black bears in the past 3 years in the area where you live? *(Please check one for each item.)*

	0 times	**1-2 times**	**3-4 times**	**More than 5 times**	**I am not sure.**
a. Saw black bears in the wild, parks or preserves	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$
b. Saw black bears in urban or suburban areas of town	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$
c. Saw black bears near home	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$
d. Had a black bear break in to or attempt to break into my garbage	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$
e. Had a black bear damage my garden or fruit trees	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$
f. Had a black bear damage my bird feeder, pet feeder, or grill	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$
g. Had a black bear cause damage to other property (e.g. fences, car, garage, etc.)	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$
h. Had a black bear attack or harass my pets or livestock	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$
i. Had a black bear enter or attempt to enter my home	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$
j. Knew someone who was attacked or harassed by a black bear	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$
k. Was attacked or harassed by a black bear myself	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$

11. For those interactions you have experienced, how acceptable was it to you that these events occurred? *(Please check one for each item that you have experienced in the last 3 years.* ***Please skip any items you have not experienced****.)*

	Very acceptable	**Somewhat acceptable**	**Neither acceptable, nor unacceptable**	**Somewhat unacceptable**	**Very unacceptable**	**I am not sure.**
a. Saw black bears in the wild, parks or preserves	[ ]1	[ ]2	[ ]3	[ ]4	[ ]5	[ ]6
b. Saw black bears in urban or suburban areas of town	[ ]1	[ ]2	[ ]3	[ ]4	[ ]5	[ ]6
c. Saw black bears near home	[ ]1	[ ]2	[ ]3	[ ]4	[ ]5	[ ]6
d. Had a black bear break in to or attempt to break into my garbage	[ ]1	[ ]2	[ ]3	[ ]4	[ ]5	[ ]6
e. Had a black bear damage my garden or fruit trees	[ ]1	[ ]2	[ ]3	[ ]4	[ ]5	[ ]6
f. Had a bear damage my bird feeder, pet feeder, or grill	[ ]1	[ ]2	[ ]3	[ ]4	[ ]5	[ ]6
g. Had a black bear cause damage to property (e.g. fences, car, garage, etc.)	[ ]1	[ ]2	[ ]3	[ ]4	[ ]5	[ ]6
h. Had a black bear attack or harass my pets or livestock	[ ]1	[ ]2	[ ]3	[ ]4	[ ]5	[ ]6
i. Had a black bear enter or attempt to enter my home	[ ]1	[ ]2	[ ]3	[ ]4	[ ]5	[ ]6
j. Knew someone who was attacked or harassed by a black bear	[ ]1	[ ]2	[ ]3	[ ]4	[ ]5	[ ]6
k. Was attacked or harassed by a black bear myself	[ ]1	[ ]2	[ ]3	[ ]4	[ ]5	[ ]6

12. Did you report negative interactions with bears you've experienced in the past 3 years to any authorities? *(Please check one.)*

[ ]1 Yes [ ]0 No ***Please skip to question 14.***

13. To whom did you report your negative interactions with black bears in or around Durango? *(Please check all that apply.)*

[ ]1 Durango police department or LaPlata County sherriff's department
[ ]2 City of Durango
[ ]3 Colorado Parks and Wildlife
[ ]4 Bearsmart Durango
[ ]5 USDA Wildlife Services

14. How likely do you believe it is that you will experience the following interactions with black bears **next year** where you live? *(Please check one for each item.)*

	Very likely	**Somewhat likely**	**Neither likely, nor unlikely**	**Somewhat unlikely**	**Very unlikely**	**I am not sure.**
a. See black bears in the wild, parks or preserves	[ ]1	[ ]2	[ ]3	[ ]4	[ ]5	[ ]6
b. See black bears in urban or suburban areas of town	[ ]1	[ ]2	[ ]3	[ ]4	[ ]5	[ ]6
c. See black bears near my home	[ ]1	[ ]2	[ ]3	[ ]4	[ ]5	[ ]6
d. Have a black bear break in to or attempt to break into my garbage	[ ]1	[ ]2	[ ]3	[ ]4	[ ]5	[ ]6
e. have a black bear damage my garden or fruit trees	[ ]1	[ ]2	[ ]3	[ ]4	[ ]5	[ ]6
f. Have a black bear damage my bird feeder, pet feeder, or grill	[ ]1	[ ]2	[ ]3	[ ]4	[ ]5	[ ]6
g. Have a black bear cause damage to other property (e.g. fences, car, garage, etc.)	[ ]1	[ ]2	[ ]3	[ ]4	[ ]5	[ ]6
h. Have a black bear attack or harass my pets or livestock	[ ]1	[ ]2	[ ]3	[ ]4	[ ]5	[ ]6
i. Have a black bear enter or attempt to enter my home	[ ]1	[ ]2	[ ]3	[ ]4	[ ]5	[ ]6
j. Know someone who will be attacked or harassed by a black bear	[ ]1	[ ]2	[ ]3	[ ]4	[ ]5	[ ]6
k. Be attacked or harassed by a black bear myself	[ ]1	[ ]2	[ ]3	[ ]4	[ ]5	[ ]6

15. Who do you believe is **most** responsible for limiting the number of negative interactions between humans and black bears in this area? *(Please check only one.)*

[ ]1 Individual residents and landowners
[ ]2 Durango police department
[ ]3 City of Durango and the City Council
[ ]4 Citizens' groups or other non-profit organizations
[ ]5 Colorado Parks and Wildlife
[ ]6 Another Colorado state agency
[ ]7 USDA Wildlife Services
[ ]8 U.S. Fish and Wildlife Service or another federal agency
[ ]9 Other *(Please indicate.____)*

Addressing Human-Black Bear Interactions. CPW is dedicated to working to address interactions between black bears and people, for the good of both the human and black bear populations. Please tell us about what you do to address bear-human interactions in your life and why you choose to take those actions.

16. The number of interactions between humans and black bears has been increasing in Colorado. Please tell us how much of a role you believe each of the following items play in this increase. *(Please rank the following items from most to least important, where 1 is the most important item and 5 is the least important.)*

_____ The number of black bears is increasing.

_____ The number of humans is increasing.

_____ Individual black bears have lost their fear of humans.

_____ Human development is expanding in bear habitat.

_____ Black bears live in a larger area of Colorado than they did in the past.

17. Do you take any of the following actions yourself to attempt to minimize your risk of having a negative interaction with black bears in the area where you live? *(Please check one for each item.)*

	Yes	**No**		**Yes**	**No**
a. Use a wildlife-resistant garbage container or dumpster	[ ]$_1$	[ ]$_2$	**g**. Not using composters, planting gardens or fruit trees	[ ]$_1$	[ ]$_2$
b. Fence my garden and/or fruit trees	[ ]$_1$	[ ]$_2$	**h.** Remove bird, squirrel and other wildlife feeders	[ ]$_1$	[ ]$_2$
c. Carry bear spray when walking or recreating	[ ]$_1$	[ ]$_2$	**i.** Fence beehives, chickens or other livestock	[ ]$_1$	[ ]$_2$
d. Avoid hiking or recreating in areas where bears have been seen	[ ]$_1$	[ ]$_2$	**j.** Keep my pets indoors	[ ]$_1$	[ ]$_2$
e. Put my garbage out on the morning of pickup day, rather than the night before	[ ]$_1$	[ ]$_2$	**k.** Keep the doors and windows of my house and car closed	[ ]$_1$	[ ]$_2$
f. Feed my pets indoors	[ ]$_1$	[ ]$_2$	**l.** Other (*Please indicate. ______________________)*	[ ]$_1$	[ ]$_2$

18. Individuals decide whether to invest their time, effort and money to reduce their chance of having an encounter with a black bear for many reasons. Which of the following is the **most** important reason you would decide to take action to prevent or reduce your risk of negative interactions with black bears? *(Please check only one.)*

[ ]$_{1}$ I want to protect my property (e.g. sheds, fences, etc.) from bears.
[ ]$_{2}$ I want to protect my pets and livestock from bears.
[ ]$_{3}$ I want to protect myself and my family from bears.
[ ]$_{4}$ It is easy to prevent or reduce negative interactions with bears.
[ ]$_{5}$ I want to keep bears acting wild and eating natural foods.
[ ]$_{6}$ City ordinances require me to take action.
[ ]$_{7}$ I can receive financial assistance to take actions to prevent or reduce conflicts.
[ ]$_{8}$ I want to prevent bears from being killed or re-located because they caused conflicts.
[ ]$_{9}$ My neighbors expect me to take action.
[ ]$_{10}$ Other *(Please indicate. ____)*

19. Similarly, some individuals decide it is not worth their time, effort and money to take actions to prevent or reduce their risk of negative interactions with black bears. Which of the following items is the **most** important reason you decide **not to take actions** to prevent or reduce your risk of negative interactions with black bears? *(Please check only one.)*

[ ]$_{1}$ I have never experienced negative interactions with bears.
[ ]$_{2}$ I believe bear conflicts are part of the cost of living where I do.
[ ]$_{3}$ I do not believe my actions will prevent bear conflicts at my home.
[ ]$_{4}$ I do not believe I am at risk for bear conflicts.
[ ]$_{5}$ I believe bears that come into areas where people live should be removed or killed.
[ ]$_{6}$ I do not think it will harm bears to eat human food or other waste.
[ ]$_{7}$ It is too difficult for me to take actions to prevent or reduce conflicts.
[ ]$_{8}$ It is too expensive for me to take actions to prevent or reduce conflicts.
[ ]$_{9}$ I believe someone else (e.g. the city, CPW, etc.) is responsible for preventing bear conflicts.
[ ]$_{10}$ I believe bears are being well managed by someone else (e.g. the city, CPW, etc.).
[ ]$_{11}$ No one else in my area takes action to prevent bear conflicts.
[ ]$_{12}$ Black bears only cause problems during a short period of the year.
[ ]$_{13}$ Other *(Please indicate. ____)*

20. Which of the following events would motivate you to take actions to reduce your risk of future negative interactions with bears? *(Please check **all** that apply.)*

[ ]$_1$ Receiving educational materials telling me how to avoid interactions with bears
[ ]$_2$ Knowing that I may recieve a fine for not bear-proofing my garbage, yard or home
[ ]$_3$ Feeling that my neighbors and community expect me to bear-proof my garbage, yard and home
[ ]$_4$ Having access to cheap or free materials for bear-proofing
[ ]$_5$ Having a bear break in or attempt to break in to my car or home
[ ]$_6$ Being threatened by a bear or having a friend or family member threatened by a bear
[ ]$_7$ Other *(Please indicate. ____)*

21. CPW takes actions to attempt to reduce or prevent negative interactions between black bears and people. How acceptable is it to you that CPW takes the following actions to manage black bears in the area where you live? *(Please check one for each item.)*

	Acceptable	**Neither acceptable, nor unacceptable**	**Unacceptable**	**I am not sure.**
a. Educate citizens about how to coexist with bears in their area	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$
b. Support city ordinances that require citizens to use bear-resistant garbage containers	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$
c. Provide financial assistance to residents for bear-proofing garbage, gardens and fruit trees	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$
d. Increase hunting licenses to increase bear harvest in areas with conflicts	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$
e. Fine individuals who are feeding bears intentionally or unintentionally	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$
f. Trap and relocate bears that cause conflict	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$
g. Kill bears that cause multiple conflicts	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$
h. Other *(Please indicate. __________________________)*	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$

22. How effective do you believe the following actions taken by government agencies can be to reduce or prevent negative interactions with bears? *(Please check one for each item.)*

	Very effective	**Somewhat effective**	**Neither effective, nor ineffective**	**Somewhat ineffective**	**Very ineffective**	**I am not sure.**
a. Educate citizens about how to coexist with bears in their area	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
b. Back city ordinances that require bear-resistant garbage containers	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
c. Provide financial assistance to residents for bear-proofing garbage, gardens and fruit trees	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
d. Increase hunting licenses to increase bear harvest	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
e. Fine individuals who fee bears intentionally or unintentionally	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
f. Trap and relocate bears that cause conflict	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
g. Kill bears that cause multiple conflicts	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
h. Other *(Please indicate.__________________)*	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$

23. How effective do you believe the following actions taken by private citizens can be in reducing or preventing negative interactions with bears? *(Please check one for each item.)*

	Very effective	**Somewhat effective**	**Neither effective, nor ineffective**	**Somewhat ineffective**	**Very ineffective**	**I am not sure.**
a. Use wildlife-resistant garbage containers or dumpsters	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
b. Fence gardens or fruit trees	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
c. Put garbage out on the morning of pickup, rather than the night before	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
d. Feed pets indoors	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
e. Not use a composter, planting gardens or fruit trees	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
f. Remove bird, squirrel and other wildlife feeders	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
g. Fence beehives, chickens or other livestock	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$
h. Keep the doors and windows of houses and cars closed	[ ]$_1$	[ ]$_2$	[ ]$_3$	[ ]$_4$	[ ]$_5$	[ ]$_6$

Background Information. The following questions will help us understand more about the people affected by black bear management in Colorado. All responses are confidential.

24. For how many years have you lived in Colorado? *(Please indicate.)*

_______________ years

25. Are you [ ]$_1$ male or [ ]$_2$ female? *(Please check one.)*

26. In what year were you born? *(Please indicate.)* 19 ________

27. How would you describe the area where you currently live? *(Please check one.)*

[ ]$_1$ Rural setting, on a farm/ranch
[ ]$_2$ Rural setting, not on a farm/ranch
[ ]$_3$ Rural subdivision
[ ]$_4$ Residential area at the edge of Durango or adjacent to a natural area
[ ]$_5$ Residential Durango, not near a natural area

28. Please use the space below to provide any additional comments you may have about black bears and their management in this area.

THANK YOU FOR YOUR TIME AND ASSISTANCE!
Please return this survey in the postage-paid envelope provided.

Appendix 2

Bear Survey
NRS

April 18, 2012

Recently you were mailed a questionnaire seeking your views about your interactions with black bears in Durango. Our response rate to this survey was lower than we needed to be meaningful. We would like to ask you a few questions so we can understand the nature of this non-response.

We are not asking you to fill out anything like the survey we previously sent you. Rather, we have attached a postage-paid, addressed postcard for you to fill out, detach, and drop in the mail. It should take no more than a minute or two to fill out the postcard. We would sincerely appreciate your taking the time to get this back to us soon, as it will provide valuable information for our study.

As before, your response to this is voluntary. Nevertheless, your input is important to ensuring wildlife managers have the very best information on which to base decisions. You may be assured of complete confidentiality. The postcard has an identification number for mailing purposes only. Your name will never be linked to your responses. Your cooperation is greatly appreciated. Thank you in advance for taking the time to help us in this matter.

Sincerely,

Stacy Lischka
Human Dimensions Specialist
Colorado Parks and Wildlife

↓ Tear here and return the bottom half. ↓

Colorado Parks and Wildlife
Public Involvement Unit
6060 Broadway
Denver, CO 80216

DIVISION OF CENTRAL SERVICES
1001 E 62nd Ave
DENVER, CO 80216-9914

Colorado Parks and Wildlife
Public Involvement Unit
6060 Broadway
Denver, CO 80216

1. How do you believe the number of black bears in the area where you live has changed over the last 3 years? *(Circle one.)* **n=351**

Increased greatly		Stayed the same		Decreased greatly	Not sure	No opinion
5%	13%	46%	10%	3%	19%	5%

2. How would you like to see the number of black bears in the area where you live change in the next 3 years? *(Circle one.)* **n=352**

Increase greatly		Stay the same		Decrease greatly	Not sure	No opinion
5%	8%	42%	13%	15%	12%	6%

3. How important is it to you to know that bears live in this area and that their populations will continue to exist in the future? *(Circle one.)* ***n=354***

Very Important		Neither important, nor unimportant		Very unimportant	Not sure	No opinion
51%	19%	10%	2%	12%	1%	2%

4. How important to you are negative interactions between humans and black bears where you live? *(Circle one.)* **n=347**

Very Important		Neither important, nor unimportant		Very unimportant	Not sure	No opinion
41%	20%	15%	7%	10%	6%	2%

5. Do you take any of the following actions to minimize your risk of a negative interaction with black bears? *(Check one for each item.)*

	n	**Yes**	**No**
a. Use a wildlife-resistant garbage container or dumpster	354	47%	53%
b. Fence my garden and/or fruit trees	314	49%	50%
c. Carry bear spray when walking or recreating	328	10%	90%
d. Avoid hiking or recreating in areas where bears have been seen	335	34%	65%
e. Put my garbage out on the morning of pickup day, rather than the night before	344	84%	16%
f. Feed my pets indoors	304	89%	12%
g. Not using composters, planting gardens or fruit trees	321	42%	57%
h. Remove bird, squirrel and other feeders	328	63%	34%

6. Do you currently own or rent the home that you live in? *(Please check one.)* ***n=347***

76% Own 24% Rent

7. In what year were you born? **n=341**

19 _ $\bar{x}$ = 61_

8. Are you male or female? **n=344**

47% Male 53% Female

Colorado Division of Parks and Wildlife
July 2011 –June 2012

WILDLIFE RESEARCH REPORT

State of:	Colorado	:	Division of Parks and Wildlife
Cost Center:	3430	:	Mammals Research
Work Package:	3003	:	Carnivore Conservation
Task No.:	1	:	Puma Population Structure and Vital Rates
		:	on the Uncompahgre Plateau
Federal Aid Project No.	W-204-R1		

Period covered: July 31, 2011−June 30, 2012

Author: Kenneth A. Logan.

Personnel: K. Logan, S. Bard, B. Dunne, W. Hollerman, W. Jesson, R. Navarrete, B. Nay, H. Taylor, S. Waters, B. Banulis, T. Bonacquista, K. Crane, J. Koch, E. Phillips, and G. Watson of CPW; volunteers and cooperators including: private landowners, Bureau of Land Management, Ridgway State Park, Colorado State University, Oklahoma State University, and U.S. Forest Service. Supplemental financial support received in previous years from The Howard G. Buffett Foundation and Safari Club International Foundation.

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

ABSTRACT

The Colorado Division of Wildlife (now Colorado Parks and Wildlife [CPW]) initiated a 10-year study on the Uncompahgre Plateau in 2004 to quantify puma population characteristics in the absence (*reference period*, years 1-5) and presence (*treatment period*, years 6-10) of sport-hunting. The purpose of the study is to evaluate assumptions underlying the Colorado Parks and Wildlife model-based approach to managing pumas with sport-hunting in Colorado. The *reference period* began December 2004 and ended July 2009, during which we captured, sampled, and marked 109 pumas for population research purposes on the Uncompahgre Plateau (Logan 2009). This report provides information on the third year of the *treatment period* (TY3), August 2011 through July 2012, on puma population characteristics and dynamics with hunting as a mortality factor.

Puma sport-hunting opened November 21 and closed December 23, 2011 after a quota of 8 independent pumas was harvested. The harvest was designed to test the management assumption that a 15% harvest of independent pumas results in a stable-to-increasing population. A total of 8 pumas were killed: 3 adult females, 1 adult male, and 4 subadult males. The harvest of 8 independent pumas represented 16.7% of the 48 independent pumas in our *minimum count* during November 2011 to April 2012. Independent females and males comprised 37.5% and 62.5% of the harvest, respectively. Four other radio-collared independent pumas (2 adult females, 2 adult males) and 3 non-collared adults (1 female, 2 males) in the study area population died during the Colorado puma hunting season. Of those, 2 adult females died of natural causes and the remainder was killed by puma hunters in GMUs adjacent to

the study area. The total mortality of 15 independent pumas during the TY3 hunting season represented 31.2% of the 48 *minimum count* of independent pumas on the study area. Seventy-four hunters requested mandatory permits with an attached voluntary hunter survey in TY3. Thirty-six of the hunters provided responses to written (n = 31) or telephone call follow-up contact (n = 5). An estimated 49 hunters actually hunted on the study area, of which about 16.3% harvested pumas and 26.5% captured pumas (i.e., harvested plus treed and released). Twenty-four of 26 answering hunters responded that they were selective hunters, and the capture, tracking, and population data indicated that most hunters practiced selection. Puma tracks < 1 day old encountered by hunters and pumas captured by hunters indicated that independent female pumas were detected more frequently than males by hunters.

From August 2011 to July 2012 twenty-eight individual pumas were captured 35 times by research teams. Two capture teams with dogs operated over 79 search days from December 27, 2011 through April 12, 2012 to find 268 puma tracks, pursue pumas 89 times, and capture 21 pumas 26 times. Capture efforts with cage traps resulted in the capture of 1 adult female for the first time. Nine new cubs were captured and radio-collared. A total of 42 pumas were monitored by radio-telemetry in TY3. Search efforts also revealed the presence of at least 26 other independent pumas. Our *minimum count* of 48 independent pumas from November 2011 to April 2012 included: 31 females and 17 males. The minimum count of 48 independent pumas in TY3 was lower than 52 in TY2 and 55 in TY1. A preliminary *minimum* estimated density of independent pumas was 2.87/100 km^2. The proportion of radio-collared adult females giving birth in the August 2011 to July 2012 biological year was 0.19 (3/16). Three litters that could be dated to month of birth were produced in August. Since 2005 a birth peak has occurred from May through August, involving 86% of births. We monitored 20 female and 7 male adult radio-collared pumas for survival and agent-specific mortality. Survival rates in TY3 for adult females (0.548, *SE*=0.1063) and males (0.167, *SE*=0.1076) were lower than in TY1 and TY2. A preliminary assessment is that hunting mortality is additive to natural mortality. Of 12 cubs monitored with radio-telemetry in TY3, 6 died. Three died of starvation after their mothers were killed by puma hunters. Three others died of natural-related causes, including 2 that starved after their mother died of a natural cause. One non-marked male cub was struck and killed by a vehicle on state highway 62. Puma harvest, capture, and radio-telemetry data from the beginning of this study to the present provided information on dispersals of 33 pumas initially marked on the study area. Those pumas moved from about 18.2 to 370 km from initial capture sites. We investigated the prevalence of *Trichinella* spp. in pumas killed in southwest Colorado in collaboration with Dr. Mason Reichard, Oklahoma State University. Twelve of 14 (85.7%) puma tongues were infected with *Trichinella*. The apparent decline in the puma population on the study area during TY1 to TY3 necessitates a reduction in the harvest quota to continue to test the harvest assumption for a stable-to-increasing puma population. This change will be pursued for TY4 and the results of the harvest monitored through the end of the *treatment period*.

WILDLIFE RESEARCH REPORT

PUMA POPULATION STRUCTURE AND VITAL RATES ON THE UNCOMPAHGRE PLATEAU, COLORADO

KENNETH A. LOGAN

P. N. OBJECTIVE

Quantify puma population sex and age structure; estimate puma population vital rates, including: reproduction of females, stage-specific survival, and immigration and emigration; quantify agent-specific mortality rates; model puma population dynamics; develop and execute the puma harvest manipulation to begin the population-wide test of Colorado Parks and Wildlife (CPW) puma management assumptions in the third year of a five-year *Treatment Period* of the Uncompahgre Plateau Puma Project― all to improve the CPW model-based approach to managing pumas in Colorado.

SEGMENT OBJECTIVES

1. Execute the third year of the five-year *treatment period* by working with CPW biologists and managers to manipulate the puma population with sport-hunting and to survey hunters.
2. Continue gathering data on puma population sex and age structure.
3. Continue gathering data for estimates of puma reproduction rates.
4. Continue gathering data to estimate puma sex and stage-specific survival rates.
5. Continue gathering data on agent-specific mortality.
6. Explore frequency of *Trichinella* ssp. in pumas harvested in southwest Colorado in collaboration with Dr. Mason Reichard, Veterinary Health Science, Oklahoma State University.

INTRODUCTION

Colorado Parks and Wildlife managers need reliable information on puma biology and ecology in Colorado to develop sound management strategies that address diverse public values and the CPW objective of "achieving healthy, self-sustaining populations" through management (Colorado Division Of Wildlife 2002-2007 Strategic Plan:9). Although 4 puma research efforts have been made in Colorado since the early 1970s and puma harvest data is compiled annually, reliable information on certain aspects of puma biology and ecology, and management tools that may guide managers toward effective puma management is lacking.

Mammals Research staff held scoping sessions with a number of the CPW's wildlife managers and biologists prior to initiating the project. In addition, we consulted with other agencies, organizations, and interested publics either directly or through other CPW employees. In general, CPW staff in western Colorado highlighted concern about puma population dynamics, especially as they relate to their abilities to manage puma populations through regulated sport-hunting. Secondarily, they expressed interest in puma―prey interactions. Staff on the Front Range placed greater emphasis on puma―human interactions. Staff in both eastern and western Colorado cited information needs regarding effects of puma harvest, puma population monitoring methods, and identifying puma habitat and landscape linkages. Management needs identified by CPW staff and public stakeholders form the basis of Colorado's puma research program, with multiple lines of inquiry (i.e., projects):

Improve our ability to manage puma hunting with enhanced scientific bases, strategies, and tools―

● Puma population characteristics (i.e., density, sex and age structure).
● Puma population dynamics and vital rates (i.e., birth rates, survival rates, emigration rates, immigration rates, population growth rates).

- Field methods and models for assessing and tracking changes in puma populations.
- Relative vulnerability of puma sex and age classes to hunter harvest.

<u>Improve our understanding of puma habitat needs and interrelationships of puma management units—</u>

- Puma habitat use, movements, and use of landscape linkages.
- Puma recruitment patterns (i.e., progeny, immigration, emigration).
- Models for identifying puma habitat and landscape linkages.

<u>Improve our understanding of the puma's role in the ecology of other species</u>

- Relationships of puma to mule deer, elk, and other natural prey.
- Relationships of puma to species of special concern, e.g., desert bighorn sheep.

<u>Improve our understanding of puma-human interactions and abilities to manage them</u>

- Behavior of puma in relation to people and human facilities.
- Puma predation on domestic animals.
- Effects of translocating nuisance pumas.
- Effects of aversive conditioning on pumas.

While all projects cannot be addressed concurrently, understanding their relationships to one another is expected to help individual projects maximize their benefits to other projects that will assist the CPW to achieve its strategic goal in puma management (Fig.1). This project has been addressing all of the gray-shaded components on the left side of the conceptual model in Figure 1.

Management issues identified by managers translate into researchable objectives, requiring descriptive studies and field manipulations. Our goal is to provide managers with reliable information on puma population biology and to develop useful tools for their efforts to *adaptively manage* puma in Colorado to maintain healthy, self-sustaining populations.

The highest-priority management needs are being addressed with this intensive population study that focuses on puma population dynamics using sampled, tagged, and GPS/VHF-radio-collared pumas to investigate the effects of sport-hunting and other causes of mortality on puma population dynamics. Those objectives include:

Describe and quantify puma population sex and age structure.
Estimate puma population vital rates, including: reproduction rates, age-stage survival rates, emigration rates, immigration rates.
Estimate agent-specific mortality rates.
Improve the CPW's puma model-based management and attendant assumptions with Colorado-specific data from objectives 1—3. Consider other useful models.
Conduct a pilot study to develop methods that yield reliable estimates of puma population abundance.
Investigate diseases in pumas.

A descriptive and manipulative study will estimate population parameters in an area that appears typical of puma habitat in western Colorado and will yield defensible population parameters based upon contemporary Colorado data. This study will be conducted in two 5-year periods. A completed 5-year *reference period*, 2004-09, (i.e., absence of recreational hunting) allowed puma life history traits to interact with the main habitat factors that influenced puma population growth (e.g., prey availability and vulnerability, Pierce et al. 2000, Logan and Sweanor 2001, Logan 2009). A subsequent 5-year *treatment period* started in 2009-10 which involves the use of controlled recreational hunting to manipulate the puma population.

TESTING ASSUMPTIONS AND HYPOTHESES

Hypotheses associated with main objectives 1—5 of this puma population research are structured to test assumptions guiding puma management in Colorado.

1. Considering limitations (i.e., methods, number of years, assumption violations) to the previous Colorado-specific studies on puma populations (Currier et al. 1977, Anderson et al. 1992, Koloski 2002), managers assume that puma population densities in Colorado are within the range of those quantified in more intensively studied populations in Wyoming (Logan et al. 1986), Idaho (Seidensticker et al. 1973), Alberta (Ross and Jalkotzy 1992, and New Mexico (Logan and Sweanor 2001). The CPW assumes density ranges of 2.0−4.6 puma/100 km^2 (i.e., includes pumas of all stage classes - adults, subadults, and cubs, J. Apker, CPW Carnivore Biologist, person. commun. Nov. 19, 2003) to extrapolate to Data Analysis Units (DAUs) to guide the model-based quota-setting process. Likewise, managers assume that the population sex and age structure is similar to puma populations described in the intensive studies. Using intensive efforts to capture, mark, and estimate non-marked animals developed and refined during the study to estimate the puma population, the following will be tested:

   H_1: Puma densities during the 5-year *reference period* (absence of recreational puma hunting) in conifer and oak communities with deer, elk and other prey populations typical of those communities in Colorado will vary within the range of 2.0 to 4.6 puma/100 km^2 and will exhibit a sex and age structure similar to puma populations in Wyoming, Idaho, Alberta, and New Mexico.

2. Recreational puma hunting management in Colorado DAUs is guided by a model to estimate allowable harvest quotas to achieve one of two puma population objectives: 1) maintain puma population stability or growth, or 2) cause puma population decline (CDOW, Draft L-DAU Plans, 2004, CDOW 2007). These objectives are expected to provide both the capacity for puma population resiliency to achieve a state-wide goal of a healthy, self-sustaining puma population while managing the puma population to provide sport-hunting opportunity and population control in some DAUs (even though puma population dynamics in any DAUs are not known). Basic model parameters are: puma population density, sex and age structure, annual population growth rate, and relative puma habitat quality and quantity. Parameter estimates are currently chosen from literature on studies in western states that are judged to provide reliable information. Background material used in the model assumes a moderate annual rate of growth of 15% (*i.e.,* $\lambda = 1.15$) for the adult and subadult puma population (CDOW 2007). This assumption is based upon information with variable levels of uncertainty (e.g., small sample sizes, data from habitats dissimilar to Colorado). Parameters influencing λ include population density, sex and age structure, female age-at-first-breeding, reproduction rates, sex- and age-specific survival, immigration and emigration.

   H_2: Population parameters estimated during a 5-year *reference period* (in absence of recreational puma hunting) in conifer and oak communities with deer, elk and other prey populations typical of those communities in Colorado will yield an estimated annual adult plus subadult population growth rate that will match or exceed $\lambda = 1.15$.

3. An assumption is that the CPW can manage puma population growth through recreational hunting on the basis that for a stable puma population hunting removes the annual increment of population growth (i.e., from current judgments on population density, structure, and λ) Puma harvest rate formulations for DAUs assumes that total mortality (i.e., harvest plus other detected deaths) in the range of 8 to 15% of the harvest-age population (i.e., independent pumas comprised of adults plus subadults) with the total mortality comprised of 35 to 45% females (i.e., adults and subadults) is acceptable to manage for a stable-to-increasing puma population (CDOW 2007). This assumption is

vital to providing the capacity for resiliency in the state-wide puma population which is hunted by applying this assumption to about three-quarters of the puma GMUs in the state.

H_3: Total mortality of an estimated 15% of the adults and subadults with no more than 45% of the total mortality comprised of females will not result in a declining trend of the harvest-age segment of the population.

4. To reduce a puma population, hunting must remove more than the annual increment of population growth. For DAUs with the objective to suppress the puma population, the total mortality guide of *greater than* 15 to 28% of the harvest-age population with greater than 45% comprised of females is suggested (CDOW 2007). This assumption is applied to about one-quarter of the GMUs in the state.

H_4: Total mortality of an estimated 16% or greater of the harvestable population with greater than 45% females will cause a declining trend in the abundance of harvest-age pumas (i.e., adults and subadults).

5. The increase and decline phases of the puma population make it possible to test hypotheses related to shifts in the age structure of the population which have been linked to harvest intensity in Wyoming and Utah.

H_5: The puma population on the Uncompahgre Plateau study area will exhibit a young age structure after hunting prohibition at the beginning of the *reference period*. During the 5 years of hunting prohibition, greater survival of independent pumas will cause an older age structure in harvest-age pumas (i.e., adults and subadults) as suggested by the work of Anderson and Lindzey (2005) in Wyoming and Stoner (2004) in Utah. As hunting is re-instated in the *treatment period*, the age structure of harvested pumas and the harvest-age pumas in the population will decline as observed by Anderson and Lindzey (2005) in Wyoming and Stoner (2004) in Utah.

Researchers in Wyoming (Anderson and Lindzey 2005) concluded that sex and age composition of the harvest varies predictably with puma population size because the likelihood of a specific sex or age class of puma being harvested with the use of hounds is a product of the relative abundance of particular sex and age classes in the population and their relative vulnerability to harvest. Results of that study suggest that managers could use sex and age composition of the harvest to infer puma population changes (Anderson and Lindzey 2005). The CPW currently uses this approach as one tool to infer potential DAU puma population dynamics (CDOW 2007). This assumes no purposeful selection by hunters for any particular sex or age-stage other than the puma must be legal (i.e., independent subadult or adult, not a lactating female or a female in association with spotted cubs) and that changes in the sex and age structure of the harvested pumas is due solely to changes in the relative abundance of particular sex and age classes in the population and their relative vulnerability to harvest. Theoretically, pumas that travel longer distances with movements that intercept access routes used by hunters (i.e., roads, trails) should be more exposed to detection by hunters and thus more vulnerable to harvest. A key assumption to this method is that pumas are killed as they are encountered and the harvest sex and age composition will reliably indicate whether a population is stable, increasing, or declining even if harvest intensity does not vary. Thus, an alternate view is that a population segment, such as independent females, may be more abundant and have shorter movement lengths, yet be detected more frequently by hunters. However, because the same intensively studied Wyoming puma population was manipulated over 6 years with varying intensities of harvest (Anderson and Lindzey 2005), variations in harvest structure using the same harvest level over a period of years could not be examined. This is a property we will investigate during the *treatment period* on the Uncompahgre Plateau puma study. Moreover, we will directly evaluate to what extent puma harvest might be influenced by hunter selection. A hunter survey is intended to reveal puma hunter behavior, detection of different classes of pumas, and lack of or presence of hunter selection. These data should allow us to examine the credibility of the assumption of non-selection by hunters and the robustness of this technique in gauging puma population dynamics relative to harvest.

We want to examine the usefulness of this approach in Colorado. CPW managers attempt to weight sport-harvest toward male pumas in GMUs with the stable-to-increasing population objective with an active educational program (i.e., mandatory hunter exam, brochure, workshops). Thus, there is a need to test assumptions associated with the Anderson and Lindzey (2005) method.

H_6: No hunter selection is practiced so that the sex and age structure of pumas harvested by hunters in this population protected from hunting during a 5-year *reference period* and subsequently managed for stability or increase with conservative harvest levels will reflect the relative vulnerabilities to detection and capture with dogs during each year in the 5-year *treatment period* in this order from high to low vulnerabilities: subadult males, adult males, subadult females, adult females without cubs or with cubs >6 months old, and adult females with cubs ≤6 months old (Barnhurst 1986, Anderson and Lindzey 2005). In each of the 5 years of the *treatment period,* subadults and adult males should comprise the majority of the harvest and reflect the assumed sex and age structure (Anderson and Lindzey 2005) of a puma population managed for a stable to increasing phase and not hunted for 5 previous years (i.e., a puma population source).

Desired outcomes and management applications of this research include:

1. Quantification of variations in puma population density, sex and age structure, growth rates, vital rates, and an understanding of factors affecting them will aid adaptive puma management by yielding population parameters and tools useful for assessing puma population dynamics, evaluation of management alternatives, and effects of management prescriptions.
2. Testing assumptions about puma populations, currently used by CPW managers, will help managers to biologically support and adapt puma management based on Colorado-specific estimated puma population characteristics, parameters, and dynamics.
3. Methods for assessing puma population dynamics will allow managers to evaluate modeled populations and estimate effects of management prescriptions designed to achieve specified puma population objectives in targeted areas of Colorado. Ascertaining puma numbers and densities during the project will allow assessment of monitoring techniques. Potential methods include use of harvest sex and age structure and photographic and DNA genotype capture-recapture. Study plans to develop and test feasible field and analytical methods will be developed as we learn the logistics of performing those methods, after we have preliminary data on puma demographics and movements which will inform suitable sampling designs, and if we have adequate funding.
4. Information which will be disseminated to citizen stakeholders interested in pumas in Colorado, and thus contribute to informed public participation in puma management.

STUDY AREA

The study area for the puma population research is on the Uncompahgre Plateau (in Mesa, Montrose, Ouray, and San Miguel Counties; Fig. 2). The study area includes about 2,253 km^2 (870 mi.^{2}) of the southern halves of Game Management Units (GMUs) 61 and 62, and about 155 km^2 (60 mi.^{2}) of the northern edge of GMU 70 (between state highway 145 and San Miguel River). The area is bounded by state highway 348 at Delta, 25 Mesa road and Forest Service road FS503 to Nucla, state highway 97 to state highway 141 to state highway 145 to Placerville, state highway 62 to Ridgeway, U.S. highway 550 to Montrose, and U.S. highway 50 to Delta.

The study area seems typical of puma habitat in Colorado that has vegetation cover that varies from the pinion-juniper covered foothills starting from about 1,700 m elevation to the spruce-fir and aspen forests growing to the highest elevations of about 3,000 m. Mule deer (*Odocoileus hemionus*) and elk (*Cervus elaphus*) are the most abundant wild ungulates available for puma prey. Cattle and domestic sheep are raised on summer ranges on the study area. People reside year-round along the eastern and western fringe of the area, and there is a growing residential presence especially on the southern end of the plateau. A highly developed road system makes the study area easily accessible for puma research efforts. A detailed description of the Uncompahgre Plateau is in Pojar and Bowden (2004).

METHODS

Reference and Treatment Periods

This research was structured in two 5-year periods: a *reference period* (years 1—5) and a *treatment period* (years 6—10). The *reference period* was closed to puma hunting on the study area and was expected to cause a population increase phase. The *treatment period* (starting in November 2009) involves manipulation of the puma population with sport-hunting structured to achieve a management objective for a stable to increasing population. In both phases, puma population structure, and vital rates are being quantified, and management assumptions and hypotheses regarding population dynamics and effects of harvest are being tested. Contingent upon results of pilot studies, we will also assess enumeration methods for estimating puma population abundance.

The *reference period*, without recreational puma hunting as a major limiting factor, was consistent with the natural history of the current puma species in North America which evolved life history traits during the past 10,000 to 12,000 years (Culver et al. 2000) that enable pumas to survive and reproduce (Logan and Sweanor 2001). In contrast, puma hunting, with its modern intensity and ingenuity, might have influenced puma populations in western North America for at least the past 100 years. Hence, the *reference period*, years 1 to 5, provided conditions where individual pumas in this population expressed life history traits interacting with the environment without recreational hunting as a limiting factor. Theoretically, the main limiting factor was vulnerable prey abundance (Pierce et al. 2000, Logan and Sweanor 2001). This allowed researchers to understand basic system dynamics *before* manipulating the population with controlled recreational hunting. In the *reference period*, all pumas in the study area were protected, except for individual pumas involved in depredation on livestock or human safety incidents. In addition, all radio-collared and ear-tagged pumas that ranged in a buffer zone in the northern halves of GMUs 61 and 62 were protected from recreational hunting mortality.

The *reference period* allowed researchers to quantify baseline demographic data on the puma population to estimate parameters useful for assessing the CPW's assumptions for its model-based approach to puma management. The *reference period* also facilitated other operational needs (because hunters did not kill the animals) including the marking of a large proportion of the puma population for parameter estimates and gathering movement data from GPS-collared pumas.

During the *treatment period*, years 6 to10, recreational puma hunting is occurring on the same study area using management prescriptions structured from information learned during previous years. Using recreational hunting for the treatment is consistent with the CPW's objectives of manipulating natural tendencies of puma populations, particularly survival, to maintain either population stability or increase or suppression (CDOW, Draft L-DAU Plans, 2004). Theoretically, survival of independent pumas is being influenced mainly by recreational hunting, which is being quantified by agent-specific mortality rates of radio-collared pumas. Dynamics of the puma population are being manipulated to evaluate hypotheses that are related to effects of hunting (i.e.,: effects of harvest rates, relative vulnerability of puma sex and age classes to hunting, variations in puma population structure due to hunting). The killing of tagged and collared pumas during the *treatment period* is not hampering operational needs (as it would have during the start-up years), because a majority of independent pumas in the population have already been marked, and sampling methods formalized.

Pumas on the study area that may be involved in depredation of livestock or human safety incidences may be lethally controlled. Researchers that find that GPS-collared pumas have killed domestic livestock will record such incidents to facilitate reimbursement to the property owner for loss of the animal(s). In addition, researchers will notify the Area Manager of the CPW if they perceive that an individual puma may be a threat to public safety.

Field Methods

<u>Puma Capture</u>: Realizing that pumas live at low densities and capturing pumas is difficult, as a starting point, our logistical aim was to have a *minimum* of 6 puma in each of 6 categories (36 total) radio-tagged in any year of the study if those or greater numbers are present. The 6 categories are: adult female, adult male, subadult female, subadult male, female cub, male cub. Our aim was to provide more quantitative and precise estimates of puma demographics than were achieved in earlier Colorado puma studies. This relatively large number of pumas might represent the majority of the puma population on the study area, and would provide the basic data for age- and sex-specific reproductive rates, survival rates, agent-specific mortality rates, emigration, and other movement data.

Puma capture and handling procedures were approved by the CPW Animal Care and Use Committee (file #08-2004). All captured pumas were examined thoroughly to ascertain sex and describe physical condition and diagnostic markings. Ages of adult pumas were estimated initially by the gum-line recession method (Laundre et al. 2000) and dental characteristics of known-age puma (Logan and Sweanor, unpubl. data). Ages of subadult and cub pumas were estimated initially based on dental and physical characteristics of known-age pumas (Logan and Sweanor unpubl. data). Body measurements recorded for each puma included at a minimum: mass, pinna length, hind foot length, plantar pad dimensions. Tissue collections included: skin biopsy (from the pinna receiving the 6 mm biopsy punch for the ear-tags), and blood (30 ml from the saphenous or cephalic veins) for genotyping individuals, parentage and relatedness analyses, and disease screening; hair (from various body regions) for genotyping tests of field gathered samples. Universal Transverse Mercator Grid Coordinates on each captured puma were recorded via Global Positioning System (GPS, North American Datum 27).

Pumas were captured year-round using 4 methods: trained dogs, cage traps, foot-hold snares, and by hand (for small cubs). Capture efforts with dogs were conducted mainly during the winter when snow facilitated thorough searches for puma tracks and enabled dogs to follow puma scent. The study area was searched systematically multiple times per winter by four-wheel-drive trucks, all-terrain vehicles, snow-mobiles, and on foot. When puma tracks ≤1 day old were detected, trained dogs were released to pursue pumas for capture.

Pumas usually climbed trees to take refuge from the dogs. Adult and subadult pumas captured for the first time or requiring a change in telemetry collar were immobilized with Telazol (tiletamine hydrochloride/zolazepam hydrochloride) dosed at 5 mg/kg based on estimated body mass (Lisa Wolfe, DVM, CPW, attending veterinarian, pers. comm.). The immobilizing agent was delivered into the caudal thigh muscles via a Pneu-Dart® shot from a CO_2-powered pistol. Immediately, a 3m-by-3m square nylon net was deployed beneath the puma to catch it in case it fell from the tree. A researcher climbed the tree, fixed a Y-rope to two legs of the puma and lowered the cat to the ground with an attached climbing rope. Once the puma was on the ground, its head was covered, its legs tethered, and vital signs monitored (Logan et al. 1986). Normal signs include: pulse ~70 to 80 bpm, respiration ~20 bpm, capillary refill time ≤2 sec., rectal temperature ~101^{o}F average, range = 95 to 104^{o}F (Kreeger 1996). Pumas that climbed trees too dangerous for the pumas or researchers for capture were released without handling, or we encourage the animals to leave the tree by heaving snowballs toward them. If the pumas climbed a safe tree, then we handled them as described above.

A cage trap was used to capture adults, subadults, and large cubs when pumas were lured into the trap using road-killed or puma-killed ungulates (Sweanor et al. 2008). A cage trap was set only if a target puma scavenged on the lure (i.e., an unmarked puma, or a puma requiring a collar change). Researchers continuously monitored the set cage trap from about 1 km distance by using VHF beacons on the cage and door. Researchers handled captured pumas within 30 minutes of capture. Puma were immobilized with Telazol injected into the caudal thigh muscles with a pole syringe. Immobilized pumas were restrained and monitored as described previously. If non-target animals were caught in the cage trap, we opened the door and allowed the animal to leave the trap.

Small cubs (≤10 weeks old) were captured using our hands (covered with clean leather gloves) or with a capture pole. Cubs were restrained inside new burlap bags during the handling process and were not administered immobilizing drugs. Cubs at nurseries were approached when mothers were away from nurseries (as determined by radio-telemetry). Cubs captured at nurseries were removed from the nursery a distance of 30 to 100 m to minimize disturbance and human scent at nurseries. Immediately after handling processes were completed, cubs were returned to the exact nurseries where they were found (Logan and Sweanor 2001).

Marking, Global Positioning System- and Radio-telemetry: Pumas did not possess easily identifiable natural marking, such as tigers (see Karanth and Nichols 2002), therefore, the capture, marking, and GPS- or VHF- collaring of individual pumas was essential to a number of project objectives, including estimating numbers, vital rates, and gathering movement data relevant to population dynamics (i.e., emigration and movement across Data Analysis Unit boundaries). Adults, subadults, and cubs were marked 3 ways: GPS/VHF- or VHF-collar, ear-tag, and tattoo. The identification number tattooed in the pinna was permanent and could not be lost unless the pinna was severed. A colored (bright yellow or orange), numbered rectangular (5 cm x 1.5 cm) ear-tag (Allflex USA, Inc., DFW Airport, TX) was inserted into each pinna to facilitate individual identification during direct recaptures. Cubs ≤10 weeks old were ear-tagged in only one pinna.

Adult and subadult female pumas were fitted with GPS collars (approximately 400 g each, Lotek Wireless, Canada) if available. Initially, GPS-collars were programmed to fix and store puma locations at 4 times per day to sample daytime, nighttime, and crepuscular locations (i.e., 0:00, 06:00, 12:00, 19:00). GPS locations for pumas provided precise, quantitative data on movements to assess the relevance of puma DAU boundaries, our search efforts, and to evaluate puma behavior and social structure. The GPS-collars also provided basic information on puma movements and locations to design other pilot studies in this program on vulnerability of puma to sport-harvest, habitat use, and enumeration methods (e.g., photographic or DNA mark-recapture).

Subadult male pumas were fitted initially with conventional VHF collars (Lotek, LMRT-3, ~400 g each) with expansion joints fastened to the collars, which allowed the collar to expand to the average adult male neck circumference (~46 cm). If subadult male pumas reached adulthood on the study area, we would recapture them and fit them with GPS collars. In addition, other adult and female subadult pumas were fitted with VHF collars when GPS collars were not available.

VHF radio transmitters on GPS collars enabled researchers to find those pumas on the ground in real time to acquire remote GPS data reports, facilitate recaptures for re-collaring, and to determine their reproductive and survival status. VHF transmitters on GPS- and VHF-collars had a mortality mode set to alert researchers when pumas were immobile for 3 to 24 hours so that dead pumas could be found to quantify survival rates and agent-specific mortality rates by gender and age. Locations of GPS- and VHF-collared pumas were identified about once per week (as flight schedules and weather allowed) from light fixed-wing aircraft (e.g., Cessna 185) fitted with radio signal receiving equipment (Logan and Sweanor 2001). GPS- and VHF-collared pumas were located from the ground opportunistically using a hand-held yagi antenna. At least 3 bearings on peak aural signals were mapped to fix locations and estimate location error around those locations (Logan and Sweanor 2001). Aerial and ground locations were plotted on 7.5 minute USGS maps (NAD 27) and UTMs along with location attributes were recorded on standard forms. GPS and aerial locations were mapped using GIS software.

We attempted to collar all cubs in observed litters. Cubs were fit with small VHF transmitters mounted on expandable collars that expand to adult neck size (Wildlife Materials, Murphysboro, Illinois, HLPM-2160, 47g, Telonics, Inc., Mesa, Arizona MOD 080, 62g, or Telonics MOD 205, 90g,) when cubs weighed 2.3—11 kg (5—25 lb). Cubs could wear these small expandable collars until they were over 12 months old. Cubs were recaptured to replace collars as opportunities allowed. Monitoring radio-collared cubs allowed quantification of survival rates and agent-specific mortality rates (Logan and Sweanor 2001).

Analytical Methods

Population characteristics each year were tabulated with the number of individuals in each sex and age category. Age categories, as mentioned, include: adult (puma ≥24 months old, or younger breeders), subadults (young puma independent of mothers, <24 months old that do not breed), cubs (young dependent on mothers, also called kittens) (Logan and Sweanor 2001). When data allowed, age categories were further partitioned into months or years.

Reproductive Rates: Reproductive rates were estimated for GPS- and VHF-collared female pumas directly (Logan and Sweanor 2001). Genetic paternity analysis will be used to ascertain paternity for adult male pumas (Murphy et al. 1998).

Survival and Agent-specific Mortality Rates: Radio-collared pumas provided known fate data used to estimate survival rates for each age stage using the Kaplan-Meier procedure to staggered entry (Pollock et al. 1989). A binomial survival model was also used for crude estimates of survival during the subadult age stage (Williams et al. 2001:343-344). In addition, when data collection is complete, survival rates will be modeled in program MARK (White and Burnham 1999, Cooch and White 2004) where effects of individual (e.g., sex, age stage, reproductive stage) and temporal (i.e., *reference period*, *treatment period*) covariates to survival can be examined. Agent-specific mortality rates can also be analyzed using proportions and Trent and Rongstad procedures (Micromort software, Heisey and Fuller 1985).

Population Inventory: The population of interest was independent pumas (i.e., adults and subadults) mainly during November to March which corresponds with the Colorado puma hunting season. Independent pumas were those that could be legally killed by recreational hunters. Initially, we estimated the *minimum* number of independent pumas and puma density (i.e., number of independent puma/100 km^2) each winter. The *minimum* number of independent pumas included all marked pumas known to be present on the study area during the period, plus individuals thought to be non-marked and detected by visual observation or tracks that were separated from locations of radio-collared pumas. Furthermore, adults comprised the breeding segment of the population and subadults were non-breeders that are potential recruits into the adult population in ≤1 year. The sampling unit was the individual independent puma (~≥1 yr. old).

Puma Population Dynamics: A deterministic, discrete time model parameterized with population characteristics and vital rates from this research was used to assess puma population dynamics (Logan 2008).

Functional Relationships: Once data collection is complete, a variety of analyses will be conducted to estimate parameters and examine functional relationships. Graphical methods will be used to initially examine functional relationships among puma population parameters. Linear regression procedures and coefficients of determination will be used to assess functional relationships if data for the response variable are normally distributed and the variance is the same at each level. If the relationship is not linear, data is non-normal, and variances are unequal, we will consider appropriate transformations of the data for regression procedures (Ott 1993). Non-parametric correlation methods, such as Spearman's rank correlation coefficient, will also be used where appropriate to test for monotonic relationships between puma abundance and other parameters of interest (Conover 1999). Relationships of explanatory variables to survival parameters will be modeled in MARK. Statistical analyses can be performed in a variety of software (e.g., SYSTAT, R, and MARK).

RESULTS AND DISCUSSION

Segment Objective 1

Puma harvest: This biological year, August 2011 to July 2012, was the third year of the *treatment period* (TY3) in this study of puma population dynamics on the Uncompahgre Plateau. The hunting season on the study area began on November 21, 2011 and was scheduled to extend to January 31, 2011, unless the harvest quota was taken before then. The harvest design quota was 8 pumas (i.e., 15% harvest of the estimated minimum number of independent pumas), with the objective to manage for a stable to increasing population. This harvest design tests the CPW's current assumption that total mortality (i.e., harvest plus other natural deaths) in the range of 8 to 15% of the harvest-age population (i.e., independent pumas comprised of adults plus subadults) with the total mortality comprised of 35 to 45% females (i.e., adults and subadults) is acceptable to manage for a stable-to-increasing puma population (Assumption and Hypothesis 3 p.5-6 this report). The initial quota of 8 pumas for TY1 was based on the projected minimum number of 53 independent pumas expected on the study area in winter 2009-10, modeled from a *minimum count* of pumas during winter 2007-08 (Table 1; Logan 2010). The quota of 8 pumas for TY3 was based on the observed *minimum count* of 52 independent pumas during November 2010 to April 2011 in TY2 and that approximately the same number of independent pumas was expected during the puma hunting season for TY3.

The hunting structure in TY3 was the same as in TY1 and TY2. The number of puma hunters on the study area was not limited. Each hunter on the study area was required to obtain a hunting permit from the CPW Montrose Service Center. Permits were free and unlimited. Each permit allowed the individual hunter with a legal puma hunting license in Colorado to hunt in the puma study area for up to 14 days from the issue date. Unsuccessful hunters that wanted to continue hunting past the permit expiration date requested a new permit for another 14 days, or until the hunter killed a puma within the season, or the season on the study area closed due to the quota being reached, or the end of the hunting season. This permit system allowed the CPW to monitor the number of hunters on the study area and to contact each hunter for survey information (see later in this section).

All pumas harvested on the study area were examined by principal investigator K. Logan or a wildlife research technician and sealed as mandated by Colorado statute. All successful hunters reported their puma kill and presented the puma carcass for inspection by CPW within 48 hours of harvest. Upon inspection, the following data were recorded: sex, age, and location of harvest. In addition, an upper premolar tooth was collected for aging (i.e., mandatory) and a tissue sample was collected for DNA genotyping. Each successful hunter was also asked at that time to complete a one-page hunter survey form. All other hunters that did not report a puma kill on the study area were asked to complete the survey form and return it in a stamped envelope that was provided. An attempt was made to contact other hunters by telephone if they did not mail in surveys.

The puma hunting season occurred on the study area from November 21 to December 23, 2011, taking 33 days to fill the quota of 8 pumas. This was 12 days more than it took to harvest 8 pumas in TY2 (i.e., 21 days, Nov. 22 to Dec. 12, 2010) and 7 more days than it took to harvest 8 pumas in TY1 (i.e., 26 days, Nov. 16 to Dec. 11, 2009). Eight pumas were killed on the study area, including: 3 adult females, 1 adult male, and 4 subadult males (Table 2). Of the 8 harvested pumas, 6 were marked: F3, F70, F75, M120, M138, and M141. In addition to the pumas killed on the study area during the Colorado puma hunting season, adult males M67 and M87 were killed by hunters in north GMU61, and adult females F104 and F119 died of natural causes. In addition, 3 non-marked adult pumas that apparently ranged on the study area were killed by hunters (Table 3). Of those, one adult male was pursued across 25 Mesa Road, the north study area boundary, and killed in north GMU62; another adult male was snow-tracked across Colorado state highway 145, a south study area boundary, and was killed in east GMU70; and an adult female with a radio-collared cub that ranged on the study area was killed adjacent to the study area in north GMU61. All these pumas were included in the *minimum count* of pumas for TY3.

The harvest of 8 independent pumas on the study area was 16.7% (8/48*100) of the *minimum count* of 48 independent pumas counted on the study area, including 31 females and 17 males, determined by the research team during November 2011 to April 2012 (Table 4). Independent females and males comprised 37.5% (3/8*100) and 62.5% (5/8*100) of the harvest, respectively. This harvest structure was 9.7% (3/31*100) of the independent females and 29.4% (5/17*100) of the independent males.

Considering the mortality of 4 other radio-collared adults (F104, F119, M67, M87) and 3 non-collared adults (1 female, 2 males) (Table 3), the mortality of 15 independent pumas was 31.2% (15/48*100) of the *minimum* number of independent pumas. The mortality composition of 6 females and 9 males was comprised of 40.0% (6/15*100) females and 60.0% (9/15*100) males. This harvest structure was 19.4% (6/31*100) of the independent females and 52.9% (9/17*100) of the independent males in the *minimum count*.

The *minimum count* of 48 independent pumas in TY3 was lower than the minimum count of 52 independent pumas in TY2 and 55 independent pumas in TY1 (Table 4, Fig.3.). *Minimum count* TY3 = 48 independent pumas, including 31 females and 17 males. This count reflected the relatively low adult female and low adult male survival rates (see Table 15, later). Because the harvest quota of 8 independent pumas in TY1 resulted in a *minimum count* of 52 independent pumas in TY2 and was expected to result in a stable-to-increasing population trend, we decided to set the quota to harvest 8 independent pumas in the TY3 (2011-12) hunting season to emulate an approximate 15% harvest of independent pumas to achieve a stable to increasing population objective while also considering that a number of independent pumas in the study area population might be killed outside of the study area as in the TY1 and TY2 hunting seasons. However, the additional pumas killed by hunters outside of the study area and natural mortality occurring during the hunting season and other parts of the biological year has apparently resulted in a declining population trend (Fig. 3).

<u>*Hunter permits and survey:*</u> In TY3 mandatory permits with the voluntary survey attached were requested by 74 individual puma hunters. This number is up from 64 hunters in TY2 and down from 79 individual hunters in TY1. Twenty-three of the hunters requested a second permit, 13 hunters requested a third permit, and one hunter requested a fourth permit after a previous permit expired after 14 days. Thirty-six hunters (48.6%) provided responses to the voluntary survey either by turning in the printed survey (n = 31) or providing information during follow-up telephone calls (n = 5) by principal investigator K. Logan. The remaining 38 hunters could not be contacted because either they did not have working phone numbers or they did not return calls. Of the respondents, 12 hunters indicated that they did not hunt on the study area. The proportion of the 36 respondents that hunted extrapolated to the total of 74 hunters (24/36 = 0.666) indicated that about 49 hunters took to the field for pumas on the study area during the 33-day TY3 hunting season. This was up from 42 hunters in TY2, but down from 67 hunters in TY1 (Logan 2010, 2011). Considering that 49 hunters were estimated to be afield, then 16.3% of the hunters harvested pumas (8/49*100) and 26.5% of hunters captured pumas (13/49*100; see captured and released pumas below and in Table 5).

The 31 puma hunters that turned in the written volunteer survey were asked to answer, “Do you consider yourself a *selective* or *non-selective* hunter?” A *selective* hunter is one that purposely is hunting for a specific type of legal puma, such as a male, large male, or large female. A *non-selective* hunter is one that intends to take whatever legal puma is first encountered or caught, with no desire for sex or size. *Selective* hunter was indicated by 24 respondents (92.3%; 24/26 = 0.923). Of the remaining 7 hunters, 2 indicated they were non-selective (7.7%), and 5 did not answer the question because they indicated that they did not hunt on the study area. The volunteer hunter survey also revealed that hunters treed pumas on the study area, but they chose not to kill them (Table 5). Those hunters reported they treed pumas 4 times and observed one, including 2 adult females (1 of them twice), 1 female of unspecified age-class, and 2 "young" males (1 male treed by 2 hunters). Two of the females were marked with collars and ear-tags. Hunters gave various reasons for not wanting to kill the pumas, including reasons based on puma sex, size, and one hunter did not want to kill a puma (Table 5).

In an effort to better ascertain the vulnerability of sexes and age-stages (i.e., adult, subadult) of independent pumas to detection by puma hunters and hunter selection to address assumption 6 and hypothesis 6 (previously), the survey was changed in TY2 to ask hunters, "What was the sex of the lion that made the first set of tracks you encountered that were less than one day old?". This question pertained to pumas that could be pursued by dogs and captured with a relatively high probability to allow the hunter an opportunity to harvest the puma. Associated with the question, we asked, "Did you pursue the lion to harvest it?" Hunters' responses in TY3 showed they encountered 21 puma tracks less than one day old. Of those, 15 tracks were of females, and 6 tracks were of males, indicating that during the hunting season females are more detectable than males by a ratio of 2.5:1, consistent with the sex structure of independent pumas in the *minimum* count on the study area which was 31 females and 17 males (Table 4). Of the 15 female tracks, 1 female puma was pursued by a hunter with intent to harvest it, and that female was killed. Nine hunters indicated they observed female tracks as their first tracks <1day old, but did not pursue the puma with intent to harvest it. Another 4 hunters did not answer the question, "Did you pursue the lion to harvest it?" Six hunters indicated they observed male tracks as their first track <1 day old; 4 indicated they pursued the puma to harvest it, and 3 male pumas were killed. Two hunters indicated they did not pursue male pumas to harvest them.

These preliminary survey and harvest data for TY3 indicate that hunters detect independent females more frequently than male pumas and females were captured by hunters slightly less than or about the same frequency as independent males by 6 to 7 (i.e., females = 3 harvested + 3 captured and released; males = 5 harvested + 2 captured and released). Moreover, hunters were choosing to kill males more frequently than females. Results in TY3 indicated selection for male pumas by hunters was consistent with TY1 and TY2 results where hunters caught females slightly more frequently than males, yet the males were selected for harvest. This preliminary assessment from years TY1, TY2, and TY3 puma harvest and hunter survey data suggests that female pumas were detected by hunters more frequently than male pumas, most puma hunters were selective, and hunter choices influenced harvest sex and age composition.

Segment Objective 2

After the harvest quota was filled, puma research teams immediately initiated capture operations with trained dogs. Two fully-staffed capture teams, one each detailed on the east and west slopes, systematically and thoroughly searched the study area to capture, sample, and GPS/VHF radio-collar pumas the remainder of winter and early spring 2011-12. These efforts along with cage trap efforts and hand-capturing cubs at nurseries maintained samples to quantify population sex and age structure, survival, and agent-specific mortality, and allowed determination of *minimum* population size on the study area.

We made 35 puma captures of 28 individuals from August 2011 to July 2012 (Tables 6-11); 21 individual pumas were captured with dogs 26 times. One puma was captured in a cage trap. Six cubs were captured at nurseries by hand. A total of 42 individual pumas were monitored with radio-telemetry from

August 2011 to July 2012 (some of these had been collared in previous years), representing sex and age classes including: 19 adult females, 7 adult males, 4 subadult females, 4 subadult males, and 12 cubs (i.e., 1 cub and 2 subadult males survived to older age classes during the biological year).

Trained dogs were used as our main method to capture, sample, and mark pumas from December 27, 2011 to April 12, 2012. Those efforts resulted in 79 search days, 268 total puma tracks detected of which 138 were ≤1 day old, 89 pursuits, and a total of 26 puma captures of 21 individual pumas (Table 6). This was the third year we deployed 2 fully-staffed hound capture teams in the *treatment period.* Search days with dogs in TY3 (79 days) were similar to TY2 (81 days), but slightly lower than TY1 (86 days) (Table 12). The frequency of tracks (tracks/day) encountered in TY3 was slightly lower than TY2, but slightly higher than TY1. The number of pursuits in TY3 was 10 less than in TY2 and 4 less than in TY1. The capture rate in TY3 was less than half that in TY2, but similar to TY1. The number of new pumas captured for the first time in TY3 was 4 less than TY2, but 2 more than TY1 (Table 12).

Researchers in the two hound capture teams from December 27, 2011 to April 12, 2012 also recorded instances when the *first* tracks ≤1 day old of independent pumas were encountered on each search route each day to represent encounters with puma tracks that could be detected and pursued by puma hunters. The count was: 70 tracks of females, including 17 associated with cubs; 2 of 2 orphaned cubs; 12 tracks of males; and 2 tracks of unspecified sex. These tracks ≤ 1 day old were found after the TY3 puma hunting season when 3 independent females and 5 independent males were harvested (Table 2). Therefore, the harvested pumas were not present to make tracks for our researchers to observe. The loss of the 3females and 5 males may be reflected in the substantially higher ratio of female:male tracks post-hunting season. By comparison, the number of female to male tracks reported by puma hunters in TY3 was 15 females and 6 males (Segment Objective 1 above).

Puma capture efforts using ungulate carcasses and cage traps was sporadic from October 5, 2011 to April 11, 2012 (Table 10). We used 21 road-killed mule deer at 18 different sites. One independent adult female puma, F172, was captured for the first time. Pumas scavenged at 3 of 21 (14.29%) sites where deer carcasses were used for bait.

We sampled 9 new cubs, including 2 females and 7 males (Table 11). All were radio-collared to monitor survival and agent-specific mortality (Appendix A).

In addition to our direct puma captures with dogs December through April, we detected 17 radio-collared pumas that we were able to identify with GPS or VHF telemetry 40 times, thus, negating the need to capture those pumas directly with dogs (Table 6). Upon detecting puma tracks that were aged at ≤1 day old, we followed the tracks with a radio receiver in an effort to detect if the tracks might be of a puma wearing a functional collar. We assigned tracks to a collared individual if we received radio signals from a puma that we judged to be <1 km from the tracks and in direction of travel of the tracks. This approach allowed us to more efficiently allocate our capture efforts toward pumas of unknown identity on the study area, particularly unmarked pumas or pumas with non-functioning GPS- or VHF- radiocollars.

In addition to the harvest and capture data (previously), our search efforts also revealed the presence of at least 26 other pumas which we included in our *minimum count* November 2011 through April 2012 (Table 4). We classified those pumas as: 10 adult females, 4 adult males, 1 subadult female, 1 subadult male, and 10 cubs. Two adult females and 2 cubs were treed by our hounds, but we could not handle the pumas because they climbed dangerous trees (Table 8). Of those, 2 adult females were sampled with biopsy darts to obtain a tissue sample for genotyping the individuals. We could separate the activity of the other pumas from the GPS- and VHF- collared pumas in time, space, and track size differences between females, males, and numbers of cubs with females. Moreover, of the 26, 4 non-marked independent pumas (3 females, 1 male) and 4 non-marked cubs were confirmed with photographs from digital trail cameras.

Our search and capture efforts during December 2011 through April 2012 and information from the puma hunting season in TY3 enabled us to quantify a *minimum count* of 48 independent pumas detected on the Uncompahgre Plateau study area, including 31 independent females and 17 independent males (Table 4). This count was based on the number of known radio-collared pumas, non-marked pumas harvested by hunters on the study area, observations of marked and non-marked pumas observed by researchers or pursued, treed and released by hunters on and adjacent to the study area, and puma tracks observed by researchers that could not be attributed to pumas with functioning radiocollars. Of the 48 independent pumas, 27 (56%) were marked and 21 (44%) were assumed to be non-marked animals (i.e., some may have ear-tags and tattoos).

The abundance and sex structure of independent pumas on the east and west slopes of the study area were similar. The east slope count included 21 independent pumas (14 females, 7 males). The west slope count included 27 independent pumas (17 females, 10 males). A decline in the study area puma population is evident on the east slope. Considering the *minimum count* of 48 independent pumas, a preliminary *minimum* density for the winter puma habitat area estimated at 1,671 km^2 on the Uncompahgre Plateau study area was 2.87 independent pumas/100 km^2.

The TY3 *minimum count* of 48 independent pumas is lower than the two previous treatment years TY1 and TY2 and appears to signal a declining trend in the puma population on the Uncompahgre Plateau study area (Fig. 3). The declining trend is further supported by declining survival rates of adult pumas on the study area (see Segment Objective 4&5 below). Taking into account the apparent declining trend in the number of independent pumas, a simple linear regression model of minimum counts of independent pumas in TY1, TY2, and TY3 on year projected that a minimum of 45 independent pumas could be expected in TY4 if the population decline continues. The recommended puma harvest for TY4 will be 5 pumas, representing 11.1% of the 45 expected number of independent pumas. This harvest rate is in the mid-range of the 8-15% test assumption for a stable to increasing population.

The estimated age structure of independent pumas in November 2011 at the beginning of the puma hunting season in TY3 on the Uncompahgre Plateau study area is depicted in Figure 4. The male age structure has declined when compared with TY1 and TY2 (Logan 2010, 2011). The female age structure is more evenly distributed and does not yet reflect a decline in survival rates of adult females in TY3 (Logan 2010, 2011). In addition to the independent pumas, we counted a minimum of 19 cubs in TY3 (Table 4).

Segment Objective 3

During the past 7.7 years of this work we compiled data on puma reproduction that was not previously available on pumas in Colorado (Table 13). Puma reproduction data (i.e., litter size, sex structure, gestation, birth interval, proportion of females giving birth per year) were summarized for the *reference period* in Logan (2009). In TY3 we directly observed 4 litters in nurseries which were born in June 2012 (1; F118's cub not marked), July 2011 (1) and August 2011 (2), each with 1 to 3 cubs born to radio-collared females. Data on reproduction we observed in TY1, TY2, and TY3 were added to Table 13 which gives the reproductive chronology and information on mates of reproducing females. But those data will not be summarized again until the end of the *treatment period*. The proportion of radio-collared adult females giving birth from August 2011 to July 2012 biological year (TY3) was 0.19 (3/16), substantially lower than TY1 (0.53, 8/15) and TY2 (0.56, 9/16), further evidence for a declining puma population.

Considering our 46 total litters from 24 females, including 44 observed with cubs 26 to 42 days old and 2 other litters confirmed by nurseries and nursling cub tracks with GPS-collared females (the latter include F111's cubs caught later when 8.5 months old) (Table 13), the distribution of puma births by month since 2005 indicate births extending from March into September (Fig. 5). Births are high in May and June, peak in July, and decline in August and September. Births during late spring to late summer (May to August) involve 86% of the births (Fig. 5). The data indicate that the large majority of puma breeding activity occurred February through May (i.e., gestation averages about 90-92 days, Logan 2009). In comparison, Anderson et al. (1992:47-48) found on the Uncompahgre Plateau during 1982-1987 that of 10 puma birth dates 7 were during July, August, and September, 2 in October, and 1 in December, with most breeding occurring April through June. The 2 data sets indicated puma births on the Uncompahgre Plateau have occurred in every month except January and November (so far). As we gather more data on the puma births during the *treatment period*, we will examine the distributions of births in the *reference* and *treatment periods* separately for a treatment effect on timing of breeding and births.

Segment Objectives 4 & 5

From December 8, 2004 (capture and collaring of the first adult puma M1) to July 31, 2012, we radio-monitored 21 adult male and 34 adult female pumas to quantify survival and agent-specific mortality rates (Table 14). Survival and agent-specific mortality of adult pumas were summarized for the *reference period* in Logan (2009). Preliminary estimates of adult puma survival rates in the absence of sport-hunting during the *reference period* indicated high survival, with adult male survival generally higher than adult female survival (Table 15).

We monitored 20 adult females and 7 adult males for annual survival and agent-specific mortality in TY3. Annual survival rate for adult females was 0.548 (*SE*=0.1063) and for males was 0.167 (*SE*=0.1076). Preliminary adult puma survival for TY1, TY2, and TY3 are also shown in Table 15. So far, adult male survival is substantially lower in the *treatment period* than in the *reference period.* Adult female survival is lower in TY1 and TY3, with marked decline in TY3. Yet, female survival is generally higher than male survival. These characteristics are probably indicative of hunter selection for male pumas (previously in Segment Objective 1). The lower adult puma survival rates were consistent with an observed decline in the puma population on the study area (see Segment Objective 2, previously).

Human-related factors caused 8 deaths of radio-marked adult pumas in TY3, including: sport-hunting harvest (3 males- M67, M87, M138; 3 females- F3, F70, F75), illegal shooting (M73), and depredation control (1 male- M153) (Tables 2, 3, 14). In addition, 6 adult female pumas died of natural causes: F23 and F24 were killed by a male puma; F104 apparently died of starvation associated with senescence; F116 apparently died of complications associated with pregnancy and parturition; F119 died of a ruptured uterus and internal bleeding associated with pregnancy, and F135 died of unknown natural cause (Table 14). The occurrence of an increasing frequency of natural deaths and declining adult survival rates in this hunted puma population suggests that sport-hunting causes additive mortality.

We have radio-monitored 27 subadult pumas (i.e., independent pumas <24 months old), including 11 females and 16 males (Table 16). We lost contact with 2 males that probably dispersed from the study area unknown distances. Of the remaining 25 subadults (females and males combined), 6 (2 females, 4 males) died before reaching adulthood, indicating a preliminary binomial survival rate of 0.76 (i.e., 19/25). F66 died at 23 months old of trauma to internal organs that caused massive bleeding attributed to trampling by an elk or mule deer. M99 died at about 16 months old; punctures to his skull were consistent with canine bites from another puma and suggested intra-species strife as cause of death. M115 died at about 14 months old due to complications of a broken left foreleg, cause unknown. This injury probably affected his ability to efficiently kill prey. F143 was killed and eaten by a male puma while in competition for an elk carcass that one of the pumas killed. Two subadult males were killed by puma hunters. We need to increase our efforts to acquire larger samples of male and female radio-monitored subadult pumas to acquire reliable estimates of their survival.

Harvest data along with our capture and radiotelemetry data provided dispersal and fate information on 33 marked pumas, 25 males and 8 females. Of those, 25 (4 females, 21 males) were initially captured and marked as cubs, and 8 (4 females, 4 males) were captured and marked in the subadult life-stage on the Uncompahgre Plateau puma study area (Table 17). Twenty males were killed away from the study area by hunters at linear distances (i.e., from initial capture sites to kill sites) ranging from about 20 to 370 km. Two males with extreme moves were killed in the Snowy Range of southeastern Wyoming (369.6 km) and the Cimarron Range of north-central New Mexico (329.8 km). Four females were killed by puma hunters; 3 off the study area ranging from 24.0 to 74.5 km from initial capture sites; 1 on the study area 18.2 km from her initial capture site. Female F52 was treed and released by hunters in December 2008 and 2009 south of Powderhorn, Colorado, indicating that she established an adult home range there before she was killed by a puma hunter in that area on Jan. 9, 2012. Three males marked initially as cubs born on the study area (M67, M87, M92) dispersed from their natal ranges and were recaptured as adults on the study area. All were born on the east slope of the Uncompahgre Plateau and moved to the west slope. Twenty-three of the 33 pumas had reached adult ages ranging from 24 to 79 months old.

A preliminary estimate of cub survival during the *reference period* was summarized in Logan 2009 using 36 radio-collared cubs (16 males, 20 females) marked at nurseries when they were 26 to 42 days old. In that summary, estimated survival of cubs to one year of age was 0.53. [The estimated minimum survival rate using the Kaplan-Meier procedure was 0.5285 (SE = 0.1623). The maximum estimated cub survival was practically the same, 0.5328 (SE = 0.1629).] The major natural cause of death in cubs, where cause could be determined, was infanticide and cannibalism by other, especially male, pumas.

In TY3 we monitored the fates of 12 radio-collared cubs (Appendix A). We lost contact with one (M156) after he shed his expandable radio-collar; he was 59 days old. Of the remaining 11 collared cubs, 6 died. Cubs M154 and M155 died probably of starvation after their mother died of an unknown natural cause; they were 77-81 days old. M159 died of an unknown natural cause when he was about 105 days old. His siblings F157 and F158 died of starvation after their mother F70 was killed by a puma hunter; they were 150 days old. M162 died of starvation after his mother was killed by a puma hunter; he was about 10.6 months old. Three other cubs that were orphaned at older ages survived to the subadult life stage. F147 was orphaned at 12 months old when her mother F24 was killed by a male puma. F147 continued to range on her natal area until her radiocollar quit functioning when she was 19 months old. Siblings F149 and M161 were orphaned at 13.5 months old when their mother F23 was killed by a male puma. Both siblings dispersed to the east slope of the study area when they were 14 to 15 months old. Another cub, F152, offspring of F93, survived to at least 25 month old in July 2012 and ranged on her natal area. A greater number of cubs over a longer period of time must be sampled before estimating cub survival and agent-specific mortality rates in the *treatment period*.

In addition, a non-marked male puma cub was struck and killed by a vehicle on state highway 62 in Leopard Creek on the south boundary of the study area on October 7, 2011. This mortality made the fourteenth puma death recorded due to vehicle collision on the study area since 2004 (Table 18). Five of the 14 pumas were marked, including 3 adults with GPS/VHF collars. Those 3 adults died during the first year of the *treatment period*.

Thirty-five adult pumas (26 females, 9 males) have worn GPS collars since this project began in 2004 (Table 19). Over 60 thousand GPS locations have been obtained and will be used for studies on puma behavior, social organization, population dynamics, movements, habitat use and puma-human relations in collaboration with colleagues in Mammals Research and Colorado State University.

Segment Objective 6

A pilot survey of prevalence of *Trichinella* spp. in puma from southwest Colorado was initiated by Mammals Researcher Ken Logan and Dr. Mason Reichard of Center for Veterinary Health Science, Oklahoma State University, Stillwater, OK.

Summary: The current pilot study documented the occurrence and high prevalence of *Trichinella* spp. in *Puma concolor* from Colorado. Twelve of 14 (85.7%) puma tongues were infected with *Trichinella*. The high prevalence of the zoonotic nematode in Colorado pumas justifies expansion of the sampling area to include pumas from a broader geographical scale.

Background: *Trichinella* spp. are zoonotic nematodes capable of infecting humans and other animals. Wild animals and humans throughout the world become infected when they ingest infected tissue containing the parasite. Infection in humans of *Trichinella* spp. may result in nausea, diarrhea, vomiting, fatigue, fever, abdominal discomfort, headaches, chills, cough, eye swelling, and may even lead to heart and breathing problems. In severe cases, infection of *Trichinella* spp. may result in death.

Hunting of pumas in Colorado has substantial historical, cultural, recreational, and economic importance. However, little current research and literature (either public or peer-reviewed) is available regarding the prevalence of *Trichinella* in Colorado puma and the potential for human infection. In 1995, an outbreak of trichinellosis in 10 people from Idaho County, Idaho was reported from the consumption of improperly prepared cougar jerky (Vollbrecht et al. 1996). The outbreak of trichinellosis in Idaho stresses the importance of wild carnivores as reservoirs of *Trichinella* spp. infections to humans (Kennedy et al. 2009). In addition to Idaho, pumas infected with *Trichinella* spp. have been reported from Montana (Worley et al. 1974; Winters 1969), Oregon (Rausch et al. 1983), Wyoming (Worley et al. 1974), and British Columbia, Canada (Gajadhar and Forbes 2010).

The purpose of the current pilot study was to determine if puma from southwest Colorado were infected with *Trichinella* spp. The specific objectives were to:

1. Determine the prevalence of *Trichinella* spp. in *P. concolor* from southwest Colorado.
2. Determine which species of *Trichinella* is/are present in *P. concolor* from southwest Colorado.
3. Establish baseline data on the occurrence, prevalence, and distribution of *Trichinella* spp. in southwest Colorado.

Pilot Project Design: Tongues from hunter-killed pumas were artificially digested to detect *Trichinella* spp. larvae. Infection with *Trichinella* spp. was assessed according to sex, age class, and geographic location of capture.

Collection of Tissue from Pumas

Tongues from dead pumas were collected by Mammals Researcher Ken Logan from pumas that were killed by sport-hunters (n = 12) and for depredation control (n = 2) in GMUs 61, 62, 64, 65, 66, and 521 representing Delta, Gunnison, Montrose, and Ouray counties in southwest Colorado. Jaws of the cats were opened, tongue firmly grasped, and pulled out of the mouth. One-half to three-quarters of the puma’s tongue was cut from the carcass using a clean knife or sterile scalpel. Excised tongues were placed in zip-top bags, labelled with sex, age estimate, and unique identifiers according to the host puma and location of where and when the sample was collected. Tongues samples were then frozen (-20 C) until they were shipped to Oklahoma State University for analysis.

Determination of *Trichinella* Infection

Infection with *Trichinella* sp. was determined by tissue digestion of tongues from puma (Webster et al., 2006). Puma tongues were weighed and homogenized in a Polytron (Kinematica GmbH, Kriens-Luzern, Switzerland). Ground samples were mixed with 10 ml of artificial digestive fluid (1% pepsin [1:3,000 IU] and 1% hydrochloric acid) per 1 gram of tissue. Digests were then mixed vigorously on magnetic stir plats at 37° C for 3 hours. Digests were allowed to settle for 20 min and the sediment containing *Trichinella* larvae were washed with tap water and enumerated under 40x magnification. Results were recorded as the number of larvae recovered per gram of tongue tissue digested.

Results: Fourteen pumas were tested for infection with *Trichinella*. Twelve of the 14 (85.7%) were infected with *Trichinella* (Table 20). Because the prevalence of *Trichinella* infection was high and the sample size was relatively small, additional statistical comparisons of sex and age classes were not made as they were unlikely to yield useful information. Based on previously published data on the prevalence of *Trichinella* spp. in pumas from other locations (Winters 1969; Worley et al. 1974; Vollbrecht et al. 1996), we anticipated that approximately 50% of the pumas from southwest Colorado would have been infected. However, the prevalence of *Trichinella* in pumas was much higher than originally thought. The common occurrence of the zoonotic parasites in pumas from southwest Colorado coupled with the fact that consumption of improperly prepared meat from wild felids can infect humans (Vollbrecht et al. 1996) necessitates continued sampling from a broader geographical area in Colorado to determine infection risk to humans.

Project Continuation: Trichinella larvae recovered from these twelve pumas will be submitted to the International *Trichinella* Reference Center (ITRC, www.iss.it/site/Trichinella/) in Rome, Italy for genotyping to identify the species of *Trichinella*. Individual *Trichinella* larvae will be identified by a multiplex PCR analysis following the protocol described by Zarlenga et al. (1999) and modified by Pozio and La Rosa (2003). Briefly, DNA will be extracted from individual worms and PCR will be performed using ExTaq DNA polymerase (Takara) in 50 ml containing 1.5 mM MgCl2, 200 mM dNTPs, 50 pmol of each primer and 0.5 unit of ExTaq DNA polymerase. The PCR-amplified fragments from purified DNA will be visualised by agarose gel electrophoresis (2.0% standard agarose).

When *Trichinella* larvae were counted from infected pumas, we noticed that the majority of the worms were still alive after being frozen for at least 6 months or longer. The trait of freeze resistance suggests the *Trichinella* in pumas from southwestern Colorado are either *T. nativa* or *Trichinella* genotype T6 (i.e., the two freeze resistant species in North America). However, *T. murrelli*, not freeze resistant, is the species of *Trichinella* most commonly recovered from wild animals in temperate areas across North America.

Fifteen additional puma tongues from southwest Colorado were collected to accumulate a larger sample size during the 2011 to 2012 puma hunting season. Those tissues will be analyzed for prevalence of *Trichinella* by Dr. Mason's laboratory in 2013.

SUMMARY

Manipulative, long-term research on puma population dynamics, effects of sport-hunting, and development and testing of puma enumeration methods began in December 2004. After 7.7 years of effort 168 unique pumas have been captured, sampled, marked, and released. Using these animals, we monitored fates of pumas in all sexes and age stages, including: 34 adult females, 21 adult males, 11 subadult females, 16 subadult males, 45 female cubs, 71 male cubs, and 1 cub of undetermined sex (some individuals occur in more than one stage class). Data from marked animals were used to quantify puma population characteristics and vital rates in a *reference period* without sport-hunting off-take as a mortality factor from December 2004 to July 2009. Puma population characteristics and vital rates in a reference condition allowed us to develop a puma population model, and to use population data and modeling scenarios to conduct a preliminary assessment of CPW puma management assumptions and guide directions for the remainder of the puma research on the Uncompahgre Plateau. Moreover, our data and model provide tools currently useful to CPW wildlife biologists and managers for assessing puma harvest strategies. The 5-year *treatment period* began August 2009 in which sport-hunting is a mortality factor. The *treatment period* will be a population-wide test of CPW puma management assumptions. Now 3 years of the *treatment period* are complete (TY1, TY2, TY3). Although data support some CPW puma management assumptions (e.g., population structure, density, how sport-harvest can cause population decline), it is still too early in this research to adequately test all the assumptions and attendant hypotheses. Although the assumption and hypothesis on harvest structure and hunter selection is not supported with the first 3 years of data in the *treatment period*, this could change with a substantial change in abundance and sex structure of independent pumas available for hunting in TY4 and TY5. The puma harvest quota for TY4 is recommended to be 5 independent pumas to align with the research design and harvest objective, and the hunters will be surveyed again. To improve data on puma population vital rates, attention will be given to increasing radio-collared sample sizes across the various life stages and sexes. We will continue to explore methods for estimating puma abundance with accurate and affordable methods. Furthermore, we will continue collaboration with colleagues on investigations of puma population parameter estimation, puma movements, puma habitat modeling and mapping, puma-human relations, and *Trichinella* prevalance. All of these efforts should enhance the Colorado puma research and management programs.

LITERATURE CITED

Anderson, A. E, D. C. Bowden, and D. M. Kattner. 1992. The puma on Uncompahgre Plateau, Colorado. Technical Publication No. 40. Colorado Division of Wildlife, Denver.

Anderson, C. R., Jr., and F. G. Lindzey. 2005. Experimental evaluation of population trend and harvest composition in a Wyoming cougar population. Wildlife Society Bulletin 33:179-188.

Barnhurst, D. 1986. Vulnerability of cougars to hunting. Master's Thesis. Utah State University.

Colorado Division of Wildlife 2002-2007 Strategic Plan. 2002. Colorado Department of Natural Resources, Division of Wildlife. Denver.

Colorado Division of Wildlife. 2007. Colorado mountain lion management data analysis unit revision and quota development process. Colorado Division of Wildlife, Denver.

Conover, W. J. 1999. Practical nonparametric statistics. John Wiley & Sons, Inc., New York.

Cooch, E., and G. White. 2004. Program MARK- a gentle introduction, 3rd edition. Colorado State University, Fort Collins.

Culver, M., W. E. Johnson, J. Pecon-Slattery, and S. J. O'Brien. 2000. Genomic ancestry of the American puma (Puma concolor). The Journal of Heredity 91:186-197.

Currier, M. J. P., and K. R. Russell. 1977. Mountain lion population and harvest near Canon City, Colorado, 1974-1977. Colorado Division of Wildlife Special Report No. 42.

Gajadhar, A. A., and L. B. Forbes. 2010. A 10-year wildlife survey of 15 species of Canadian carnivores identifies new hosts or geographical locations for *Trichinella* genotypes T2, T4, T5, and T6. Veterinary Parasitology 168: 78-83.

Heisey, D. M., and T. K. Fuller. 1985. Evaluation of survival and cause specific mortality rates using telemetry data. Journal of Wildlife Management 49:668-674.

Karanth, K. U., and J. D. Nichols. 2002. Monitoring tigers and their prey: A manual for researchers, managers and conservationists in tropical Asia. Centre for Wildlife Studies, Bangalore, India.

Kennedy, E. D., R. L. Hall, S. P. Montgomery, D. G. Pyburn, and J. L. Jones. 2009. Trichinellosis surveillance – United States, 2002-2007. Morbidity and Mortality Weekly Report 58: 1-7.

Koloski, J. H. 2002. Mountain lion ecology and management on the Southern Ute Indian Reservation. M. S. Thesis. Department of Zoology and Physiology, University of Wyoming, Laramie.

Kreeger, T. J. 1996. Handbook of wildlife chemical immobilization. Wildlife Pharmaceuticals, Inc., Fort Collins, Colorado.

Laundre, J. W., L. Hernandez, D. Streubel, K. Altendorf, and C. L. Lopez Gonzalez. 2000. Aging mountain lions using gum-line recession. Wildlife Society Bulletin 28:963-966.

Logan, K. A., E. T. Thorne, L. L. Irwin, and R. Skinner. 1986. Immobilizing wild mountain lions (*Felis concolor*) with ketamine hydrochloride and xylazine hydrochloride. Journal of Wildlife Diseases. 22:97-103.

_____, and L. L. Sweanor. 2001. Desert puma: evolutionary ecology and conservation of an enduring carnivore. Island Press, Washington, D.C.

_____. 2008. Puma population structure and vital rates on the Uncompahgre Plateau, Colorado. Wildlife Research Report. Colorado Division of Wildlife, Fort Collins.

_____. 2009. Puma population structure and vital rates on the Uncompahgre Plateau, Colorado. Wildlife Research Report. Colorado Division of Wildlife, Fort Collins.

_____. 2010. Puma population structure and vital rates on the Uncompahgre Plateau, Colorado. Wildlife Research Report. Colorado Division of Wildlife, Fort Collins.

_____. 2011. Puma population structure and vital rates on the Uncompahgre Plateau, Colorado. Wildlife Research Report. Colorado Division of Parks and Wildlife, Fort Collins.

Murphy, K., M. Culver, M. Menotti-Raymond, V. David, M. G. Hornocker, and S. J. O'Brien. 1998. Cougar reproductive success in the Northern Yellowstone Ecosystem. Pages 78-112 *in* The ecology of the cougar *(Puma concolor)* in the Northern Yellowstone ecosystem: interactions with prey, bears, and humans. Dissertation, University of Idaho, Moscow.

Ott, R. L. 1993. An introduction to statistical methods and data analysis. Fourth edition. Wadsworth Publishing Co., Belmont, California.

Pierce, B. K., V. C. Bleich, and R. T. Bowyer. 2000. Social organization of mountain lions: does land a tenure system regulate population size? Ecology 81:1533-1543.

Pollock, K. H., S. R. Winterstein, C. M. Bunck, and P. D. Curtis. 1989. Survival analysis in telemetry studies: the staggered entry design. Journal of Wildlife Management 53:7-15.

Pojar, T. M., and D. C. Bowden. 2004. Neonatal mule deer fawn survival in west-central Colorado. Journal of Wildlife Management 68:550-560.

Pozio, E., and G. La Rosa. 2003. PCR-derived methods for the identification of *Trichinella* parasites from animal and human samples. Methods in Molecular and Cellular Biology 216: 299-309.

Rausch, R. L., C. Maser, and E. P. Hoberg. 1983. Gastrointestinal helminths of the cougar, *Felis concolor* L. in northeastern Oregon. Journal of Wildlife Diseases 19: 14-19.

Ross, P. I., and M. G. Jalkotzy. 1992. Characteristics of a hunted population of cougars in southwestern Alberta. Journal of Wildlife Management 56:417-426.

Seidensticker, J. C., M. G. Hornocker, W. V. Wiles, and J. P. Messick. 1973. Mountain lion social organization in the Idaho Primitive Area. Wildlife Monographs No. 35.

Stoner, D. C. 2004. Cougar exploitation levels in Utah: implications for demographic structure, metapopulation dynamics, and population recovery. Master of Science Thesis. Utah State University.

Sweanor, L. L., K. A. Logan, J. W. Bauer, B. Milsap, and W. M. Boyce. 2008. Puma and human spatial and temporal use of a popular California state park. Journal of Wildlife Management 72:1076-1084.

Vollbrecht, A., D. Sokolowski, W. Hollipeter, R. Sigler, J. Greenblatt, D. E. Anderson, P. J. Tennican, H. R. Gamble, and D. Zarlenga. 1996. Oubreak of trichinellosis associated with eating cougar jerky – Idaho, 1995. Morbidity and Mortality Weekly Report 45: 205-206.

Webster, P., C. Maddox-Hyttel, K. Nockler, A. Malajauskas, J. van der Giessen, E. Pozio, P. Boireau, C. M. O. Kapel. 2006. Meat inspection for *Trichinella* in pork, horsemeat and game within the EU: available technology and its present implementation. Eurosurveillance 11:50-55.

Williams, B. K., J. D. Nichols, and M. J. Conroy. 2001. Combining closed and open mark-recapture models: the robust design. Pages 523-554 *In* Analysis and management of animal populations. Academic Press, New York.

White, G. C., and K. P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46 (Suppl):S120-S139.

Winters, J. B. 1969. Trichiniasis in Montana mountain lions. Bulletin of the Wildlife Disease Association 5: 400.

Worley, D. E., J. C. Fox, J. B. Winters, and K. R. Greer. 1974. Prevalence and distribution of *Trichinella spiralis* in carnivorous mammals in the United States northern Rocky Mountain region. Proceedings of the 3rd International Conference on Trichinellosis, Florida, USA, November 2-4, 1972.

Zarlenga, D.S., Chute, M.B., Martin, A. and Kapel, C.M. 1999. A multiplex PCR for unequivocal differentiation of all encapsulated and nonencapsulated genotypes of *Trichinella*. International Journal for Parasitology 29:1859–1867.

Prepared by: ________________________
Kenneth A. Logan, Wildlife Researcher

Table 1. Projected puma population growth modeled from a *minimum count* of independent pumas during winter 2007-08 *reference period* year 4 (RY4). *Treatment period* year 1 (TY1), shaded in gray, indicates the results used to derive a quota of 8 independent pumas, representing 15% of the independent pumas (from Logan 2009).

Harvest Level	**Year**	**Projected Minimum Puma Population Size**					**Independent Pumas**	
		Adult		**Subadult**				
		Female	**Male**	**Female**	**Male**	**Cub**	**Total**	**Lambda**
No harvest.	RY4	16	8	5	4	20	33	
	RY5	18	10	9	8	33	45	1.37
	TY1	23	14	8	8	42	53	1.17

Table 2. Pumas harvested by sport-hunters in Treatment Year 3 (TY3) on the Uncompahgre Plateau Study Area, Colorado, November 21 to December 23, 2011.

Puma sex	**Age (yr.)**	**Previous M/F I.D. or specimen P no. if not marked**	**Date of kill**	**Location/UTM NAD27 Zone, Easting, Northing**	**Hunter/status**
M	1.6	P1038	12/5/2011	Cottonwood Fork of Dry Creek 12, 756280, 4250547	Ray David/resident
M	1.4	M120	12/6/2011	Spring Creek 13, 238681, 4249866	Gary Gleason/resident
F	9.5	F3	12/11/2011	Lindsay Creek 13, 238911, 4252542	Kari McClanahan/ resident
F	8	F75	12/14/2011	Cottonwood Creek (W) 12, 732894, 4239423	Joe Gray/non-resident
F	7	F70	12/22/2011	Spring Creek, Puma Fork 13, 239323, 4243719	Dustin Gleason/resident
M	1.1	P1049	12/23/2011	Hills west of Colona, CO 13, 256132, 4245751	Dawson Flowers/ resident
M	2.5	M138	12/23/2011	Horsefly Creek (E) 13, 249592, 4240770	Darren Reed/resident
M	1.3	M141	12/23/2011	Little Bucktail Creek 12, 752201, 4239371	Kenneth Sowell/ resident

Table 3. Four other independent VHF/GPS-collared and 3 non-collared adult pumas in the *minimum count* for the Uncompahgre Plateau Study Area that died during the 2011-2012 Colorado puma hunting season.

Puma sex (M or F)	**Age (yr.)**	**Date of kill/death**	**Place of kill/UTM NAD27 Zone, Easting, Northing**	**Hunter/status/other cause**
M87	3.4	12/6/2011	Forty-seven Canyon, Tabaguache Canyon 6.5 km N of study area	John Elmer/rresident/S. Garvey Outfitter
M67	4.4	12/18/2011	Lower Tabaguache Canyon 12.61 km NW of study area 12, 707031, 4247827	Karl Red/resident
F119	7.5	1/28/2012	Clay Creek 12, 743719, 4228535	Ruptured uterus and blood loss associated with pregnancy
F104	11	1/31/2012	Lower Roubideau Creek, died 1.73 km N of study area 12, 748282, 4288223	Starvation, probably associated with senescence
M	5.5	12/6/2011	Cottonwood Creek north of Roubideau Canyon 0.66 km N of study area 12, 736764, 4274349	Brett Merritt/non-resident Trailed from study area/R. Navarrete Outfitter
M	3	1/17/2012	Specie Creek 1.65 km S of study area 12, 752861, 4211534	Alan Hatfield/resident Trailed from study area
F	6	1/18/2012	Pinto Mesa 1.02 km N of study area 12, 721658, 4247479	James Williams/non-resident/S. Garvey Outfitter Radio-collared cub M162 ranged on study area

Table 4. *Minimum count* of pumas based on numbers of known radio-collared pumas, visual observations of non-marked pumas, harvested non-marked pumas, and track counts of suspected non-marked pumas on the study area during September 2009 to April 2010 of Treatment Year 1 (TY1), November 2010 to April 2011 (TY2), and November 2011 to April 2012 (TY3) Uncompahgre Plateau study area, Colorado.

Treatment Year (TY)	**Study Area region**	**Adults**		**Subadults**		**Cubs**		
		Female	**Male**	**Female**	**Male**	**Female**	**Male**	**Unknown sex**
TY1	East slope	16	10	1	1	1	4	4-8*
	West slope	14	10	0	3	3	3	5-6
	subtotals	30	20	1	4	4	7	9-14
	Total Independent Pumas = 55, including 31 females, 24 males. Cubs = 20-25							
TY2	East slope	15	5	3	2	7	9	7
	West slope	15	7	2	3	2	5	9
	subtotals	30	12	5	5	9	14	16
	Total Independent Pumas = 52, including 35 females, 17 males. Cubs = 39							
TY3	East slope	13	4	1	3	4	2	4
	West slope	14	5	3	5	1	2	6
	subtotals	27	9	4	8	5	4	10
	Total Independent Pumas = 48, including 31 females, 17 males. Cubs = 19							

*One adult non-marked female puma was killed by a hunter in Roubideau Canyon. The female puma was lactating, indicating she had nurslings. Up to 4 cubs were assumed to be in the litter.

Table 5. Pumas captured and released by sport-hunters in Treatment Year 3 (TY3) on the Uncompahgre Plateau Study Area, Colorado, November 21 to December 23, 2011. Data are from puma hunter responses in 36 voluntary surveys, including: 31 original surveys on printed voluntary permits and 5 telephone contacts with hunters that did not return printed surveys on permits. Total response rate from 74 individual permitted hunters was 48.6 % (36/74 = 0.486*100).

Puma sex/age stage/mark	**Date of capture**	**Capture location**	**Hunter name**	**Reason for releasing the puma given by hunter**
F/adult/collar/eartags	12/4/2011	Transfer Rd., Roubideau Cyn.	George Quintana	Did not want to kill a female. Observed the puma on the road.
M/young/no marks	11/30/2011 to 12/5/2011	Dry Park.	Eric Franklin	Did not want to kill a small male.
F/adult/F75/ collar/eartags/caught twice	12/14 to 16/2011	Cottonwood Cr.	Thomas Barnes	Caught F75 twice. Did not want to kill a puma. Wanted to take photos.
M/young/no marks	11/30/2011 to 12/5/2011	Dry Park.	Ross Ward	Did not want to kill a small male. Same male caught with Eric Franklin (above).
F/no marks	12/20 to 23/2011	Loghill Mesa	Zachary Prock	Did not want to kill a female.

Table 6. Summary of puma capture efforts with dogs from December 27, 2011 to April 12, 2012, Uncompahgre Plateau, Colorado.

Month	No. Search Days	No. & type of puma tracks found[a,b]	No. & type of pumas pursued	No. & I.D. or type of pumas captured, observed, or identified
December	4	16 tracks: 4 male, 10 female, 1 cub, 1 undetermined independent puma Tracks ≤1 day old: 2 male, 8 female, 0 cub	4 pursuits: 2 male, 2 female, 0 cub	2 pumas captured 2 times: adult female F137 and dependent young F152 (of F93). In addition, adult females F93 and F111 were associated with tracks by VHF telemetry.
January	25	103 tracks: 12 male, 56 female, 31 cub, 4 undetermined independent pumas Tracks ≤1 day old: 4 male, 31 female, 23 cub, 1 undetermined	39 pursuits: 3 male, 19 female, 16 cub, 1 undetermined	12 pumas captured 14 times: F93, F96, M170 (cub of F171), F171, cub (not handled, of F171), F8, F140, F149 (cub of F23), M160, M161 twice (cub of F23), F163 twice, M162 (orphaned cub). In addition, adult females F23 (3 times), F93 (2 times), F136, F137, F149 (2 times), F152 (2 times), M170, and F171 were located by VHF telemetry in association with tracks.
February	20	71 tracks: 9 male, 44 female, 17 cub, 1 undetermined independent puma Tracks ≤1 day old: 7 male, 26 female, 9 cub, 1 undetermined	27 pursuits: 6 male, 13 female, 8 cub	9 pumas captured 9 times: F28 (not handled in hole), F129 (not handled, dangerous tree), M131 (not handled, dangerous tree), F163, M164, M165, PF1051 (biodarted, not handled in dangerous tree), PF1052 (biodarted, not handled in dangerous tree), cub (not handled in dangerous tree, of non-marked female). In addition, adult females F23 (2 times), F93, F95, F96 (2 times), F136, F137, F171, adult male M164, subadult females F147 and F163 (3 times), and cubs F152 and M162 (2 times) were associated with tracks by VHF telemetry.
March	22	66 tracks: 12 male, 39 female, 14 cub, 1 undetermined independent puma Tracks ≤1 day old: 2 male, 16 female, 4 cub	18 pursuits: 2 male, 12 female, 4 cub	2 pumas captured 2 times: F149, M161. In addition, adult females F23, F95, F96, F171, subadult females F140 and F149, cubs M161 and M170 were associated with tracks by VHF telemetry.
April	8	12 tracks: 6 male, 5 female, 1 cub Tracks ≤1 day old: 1 male, 2 female, 1 cub	1 pursuit: 1 male	0 pumas captured. None associated with tracks with VHF telemetry.
TOTALS	79	268 tracks: 43 male, 154 female, 64 cub, 7 undetermined Tracks ≤1 day old: 16 male 83 female 37 cub 2 undetermined	89 pursuits: 14 male, 46 female, 28 cub 1 undetermined	21 individual pumas were captured 26 times with aid of dogs. In addition, 17 radio-collared pumas were detected 40 times by tracks and identified with VHF telemetry ≤1 km from the tracks.

[a] Puma hind-foot tracks with plantar pad widths >50 mm wide are assumed to be male; ≤50 mm are assumed to be female (Logan and Sweanor 2001:399-412).

[b] Each capture season researchers also recorded instances when the *first* puma tracks ≤1 day old were encountered on each search route each day to gather data on vulnerability to detection using methods similar to puma hunters. For 2011-2012 (TY3) the count was: 70 tracks of females, including 17 of those associated with cubs; 2 tracks of 2 orphaned cubs; 12 tracks of males; and 2 tracks of undetermined sex.

Table 7. Adult and subadult pumas captured for the first time, sampled, tagged, and released from December 2011 to April 2012, Uncompahgre Plateau, Colorado.

Puma I.D.	Sex	Estimated Age (mo.)	Mass (kg)	Capture date	Capture method	Location
M160	M	19	46	1/18/2012	Dogs	Sanborn Park, head of Albin Draw
F163	F	18	43	1/26/2012	Dogs	San Miguel Canyon E of Pinyon
M164	M	19	56	2/14/2012	Dogs	Pinto Mesa, moved from Big Bucktail Canyon
M165	M	19	56	2/24/2012	Dogs	Head of Coal Canyon
F171	F	27	45	1/20/2012	Dogs	McKenzie Butte
F172	F	33	NM*	3/28/2012	Cage trap	Monitor Canyon, Roubideau Canyon

*Not measured.

Table 8. Pumas that were captured and observed with aid of dogs, some of which were biopsy-darted and given specimen numbers (e.g., P1051), but were not handled at that time for safety reasons, and a puma killed for depredation control, December 2011 to April 2012, Uncompahgre Plateau, Colorado.

Puma sex & I.D.	Age stage or months	Capture date	Location	Comments
F P1035	18	10/22/2011	Dallas Creek, Pleasant Valley.	Puma killed by Wildlife Services agent for killing a domestic llama. Puma not previously marked.
F P1051	adult	2/13/2012	Potter Canyon, Roubideau Canyon.	Puma climbed dangerous tree. Biopsy-darted to obtain tissue sample for genotype. In association with a single cub about 8 to 10 months old, which also could not be handled due to dangerous tree.
F P1052	adult	2/29/2012	Monitor Canyon, Roubideau Canyon.	Puma climbed dangerous tree Biopsy-darted to obtain tissue sample for genotype.
Unknown none	cub 4 to 5	1/12/2012	E Loghill Mesa.	Puma cub climbed high in dangerous tree. Probably 1 of 2 cubs of F171.
Unknown none	cub 8 to 10	2/29/2012	Monitor Canyon, Roubideau Canyon.	Puma cub climbed high in dangerous tree. Not handled. In association with P1051 above.

Table 9. Pumas recaptured with dogs (none in cage traps) December 2011 to April 2012, Uncompahgre Plateau, Colorado.

Puma I.D.	Recapture Date	Mass (kg)	Estimated Age (mo.)	Capture Method/ Location	Process
F152	12/27/2011	Observed	18	Dogs/Dry Cr. Basin	F152 climbed dangerous tree. Could not be handled to fit with radiocollar.
F137	12/28/2011	Observed	35	Dogs/W Fk. Dry Cr.	None.
F96	1/9/2012	44	72	Dogs/lower Delores Cr.	Replaced faulty GPS collar.
F140	1/13/2012	47	17	Dogs/Tomcat Cr.	Fitted with VHF collar.
F93	1/17/2012	Observed	109	Dogs/Lower Linscott Cyn.	None.
F8	1/17/2012	Observed	104	Dogs/Coal Canyon	None.
F152	1/18/2012	43	19	Dogs/Shavano Mesa	Fitted with GPS collar.
F149	1/24/2012	Observed	9	Dogs/Big Bucktail Cyn.	None.
M161	1/24/2012	Observed	9	Dogs/Big Bucktail Cyn.	None.
F163	1/27/2012	Observed	18	Dogs/Maverick Draw	None.
F129	2/2/2012	Observed	18	Dogs/Dolores Cyn.	F129 climbed dangerous tree. Could not be handled to fit with radiocollar.
M131	2/2/2012	Observed	18	Dogs/Dolores Cyn.	M131 climbed dangerous tree. Could not be handled to fit with radiocollar.
F163	2/9/2012	Observed	19	Dogs/San Miguel Cyn.	None.
F28	2/16/2012	Observed	107	Dogs/San Miguel Cyn.	F28 took refuge in hole. Could not be handled.
F149	3/5/2012	29	11	Dogs/Tomcat Cr.	Replaced VHF collar on F149.
M161	3/5/2012	Observed	11	Dogs/Tomcat Cr.	None.

Table 10. Summary of puma capture efforts with cage traps from October 5, 2011 to April 11, 2012, Uncompahgre Plateau, Colorado.*

Month	No. of Sites	Carnivore activity & capture effort results
October	6	No pumas scavenged 7 mule deer carcasses used at the sites.
November	7	Unknown male puma scavenged deer carcass on SE Loghill Rim 11/9/2011; attempted capture with cage trap; male puma did not return. Unknown male puma walked ~100 m from deer carcass at same bait site above 11/22/2011, but did not scavenge the bait. Unknown male puma walked ~1.5 m from a deer carcass at same bait site as above 11/29/2012, but did not scavenge the bait.
March	2	Puma F172 captured in cage trap baited with mule deer carcass 3/28/2012. Unknown female puma and lone cub scavenged mule deer carcass at another bait site 4/6-8/2012. The pumas were pursued with dogs on 4/9/2012, but were not captured.
April	5	No pumas scavenged 5 mule deer carcasses used at the sites.

* We used 21 road-killed mule deer at 18 different sites. Of the road-killed deer baits, 3 of 21 (14.29%) were scavenged by pumas.

Table 11. Puma cubs sampled August 2011 to July 2012 on the Uncompahgre Plateau Puma Study area, Colorado.

Cub I.D.	Sex	Estimated birth date[a]	Estimated age at capture (days)	Mass (kg)	Mother	Estimated age of mother at birth of this litter (mo)
M154	M	7/6/2011	42	2.6	F135	33
M155	M	7/6/2011	42	3.0	F135	33
M156[b]	M	8/20/2011	43	3.25	F137	30
F157	F	8/18/2011	40	2.5	F70	76
F158	F	8/18/2011	40	2.5	F70	76
M159	M	8/18/2011	40	2.5	F70	76
M161[c]	M	4/22/2011	276	24	F23	80
M162[d]	M	7/2011	183	12	Nonmarked	Adult
M170[e]	M	8/2011	137	9	F171	22

[a] Estimated age of cubs sampled at nurseries is based on the starting date for GPS location and radio-telemetry foci for mothers at nurseries, and development characteristics of cubs caught with mothers without radiocollars or mothers with non-functioning radiocollars.

[b] Probably more than one cub in F137's litter; others probably hiding in a hole at the nursery.

[c] M161 is sibling of F149; birth date known from radio-telemetry on mother F23.

[d] M162 was observed with one non-marked sibling on 2/7/2012. Both cubs were orphans; their mother non-marked mother apparently killed by a hunter on 1/18/ 2012 on Pinto Mesa.

[e] M170 was observed with one sibling on 1/12,13/2012. Mother F171 was captured for first time on 1/20/2012.

Table 12. Summary of puma capture efforts with dogs, December 2004 to April 2012, Uncompahgre Plateau, Colorado.

Period	**Track detection effort**	**Pursuit effort**	**Puma capture effort**	**Effort to capture an independent puma for the first time**
Dec. 2, 2004 to May 12, 2005	109/78 = 1.40 tracks/day	35/78 = 0.45 pursuit/day 78/35 = 2.23 day/pursuit	14/78 = 0.18 capture/day 78/14 = 5.57 day/capture	11 pumas captured for first time 11/78 = 0.14 capture/day 78/11 = 7.09 day/capture
Nov. 21, 2005 to May 26, 2006	149/82 = 1.82 tracks/day	43/82 = 0.52 pursuit/day 82/43 = 1.91 day/pursuit	14/82 = 0.17 capture/day 82/14 = 5.86 day/capture	7 pumas captured for first time 7/82 = 0.08 capture/day 82/7 = 11.71 day/capture
Nov. 13, 2006 to May 11, 2007	177/78 to 182/78 = 2.27-2.33 tracks/day	45/78 to 47/78 = 0.58-0.60 pursuit/day 78/47 to 78/45 = 1.66-1.73 day/pursuit	22/78 = 0.28 capture/day 78/22 = 3.54 day/capture	7 pumas captured for first time 7/78 = 0.09 capture/day 78/7 = 11.14 day/capture
Nov. 19, 2007 to April 24, 2008	217/77 to 218/77 = 2.82-2.83 tracks/day	49/77 = 0.64 pursuit/day 77/49 = 1.57 day/pursuit	20/77 = 0.26 capture/day 77/20 = 3.85 day/capture	7 pumas captured for first time 7/77 = 0.09 capture/day 77/7 = 11.00 day/capture
Dec. 9, 2008 to April 30, 2009	198/71 to 202/71 = 2.79-2.84 tracks/day	75/71 to 78/71 = 1.06-1.10 pursuit/day 71/75 to 71/78 = 0.91-0.95 day/pursuit	24/71 = 0.34 capture/day 71/24 = 2.96 day/capture	9 pumas captured for first time 9/71 = 0.13 capture/day 71/9 = 7.89 day/capture
Dec. 15, 2009 to April 30, 2010	266/86 = 3.09 tracks/day	93/86 = 1.08 pursuit/day 86/93 = 0.92 day/pursuit	26/86 = 0.30 capture/day 86/26 = 3.31 day/capture	9 pumas captured for first time 9/86 = 0.11 capture/day 86/9 = 9.56 day/capture
Nov. 16 and Dec. 14, 2010 to April 22, 2011	300/81 = 3.70 tracks/day	99/81 = 1.22 pursuit/day 81/99 = 0.82 day/pursuit	52/81 = 0.64 capture/day 81/52 = 1.56 day/capture	15 pumas captured for first time 15/81 = 0.18 capture/day 81/15 = 5.40 day/capture
Dec. 27, 2011 To April 12, 2012	268/79 = 3.39 tracks/day	89/79 = 1.13 pursuit/day 79/89 = 0.89 day/pursuit	26/79 = 0.28 capture/day 79/26 = 3.04 day/capture	11 pumas captured for first time 11/79 = 0.14 capture/day 79/11 = 7.18 day/capture

Table 13. Individual puma reproduction histories, Uncompahgre Plateau, Colorado, 2005-2012.

Consort pairs and estimated ages[a]				Dates pairs consorted[b]	Estimated birth date[c]	Estimated birth interval (mo.)	Estimated gestation (days)	Observed number of cubs[d]
Female	**Age (mo.)**	**Male**	**Age (mo.)**					
F2	53				05/28/05			3
F2	67				07/29/06	14.0		2
F2	89				05/19/08	22.0		4
F3	36				08/01/04			1
F3	50	M6	37	06/22-24/05	09/26/05	13.8	93-95	2
F3	62				09/17/06	11.7		3
F3	84	M51	60	03/31/08	07/03/08	21.5	94	3
F3	107	M55	69	03/28-31/10	06/28/10	23.8	89-92	2
F7	67				05/19/05			2
F7	82				08/13/06	14.9		4
F7	106				07/10/08	23.9		3
F8*[e]	24				06/26/05			2
F8	37				08/13/06	13.4		4
F8	60	M73	49	02/28-29/08	05/29/08	22.5	90-91	2
F8	95				04/18/11	34.7		2
F16	32				09/22/05			4
F16	52				05/24/07	19.9		4
F16	75	M6	80	01/13-14/09	04/15/09	22.7	91-92	3
F23*	21				05/30/06			3
F23	45	M27 or M29[f]	78 107	02/19-25/08	05/23/08	23.8	87-93	3
F23	80	M67	53	01/28-31/11	04/22/11	Non-funct.GPS	84-86	2
F24	75	M29	92	04/12-15/07	06/14/07		90-93	4
F24	114				09/10	Non-funct.GPS		3
F25	74				08/01/05			1
F25	94				04/16/07	20.5		1
F25	110				08/19/08	16.1		2
F25	129				3/10	Non-funct.GPS		3
F28*	36				06/09/06			2
F28	48	M29	88	12/27-29/06	03/30/07	11.7	92-93	≥2 tracks
F28	68				11/08			1
F30*	48	M55	34	04/16-20/07	07/17/07		88-92	3
F50	21				07/01/06			1
F54	24				07/01/06			1
F70*	38	M51	60	03/10/08	06/05/08		87	3
F70	52				08/31/09	14.8		3
F70	76				08/18/11	23.6		3
F72*	28				07/09/08			1
F72	51				06/12/10	23.1		2
F72	64				07/15/11	13		3 photographed
F75	32				06/01/07			1
F75	55	M73	61	02/11/09	05/07/09	23.2	93	2
F93	56				08/07			2
F93	90				06/16/10			2
F94*	46				05/27/09			3
F94	60	M55	70	04/15/10	07/15/10	13.3	91	3
F96	55	M55	71	05/21/10	08/21/10			4
F104	110				07/08/10			3
F111*	32				06/16/10			2
F116[g]	36				2009			2

Table 13 continued.

Consort pairs and estimated ages[a]				Dates pairs consorted[b]	Estimated birth date[c]	Estimated birth interval (mo.)	Estimated gestation (days)	Observed number of cubs[d]
Female	Age (mo.)	Male	Age (mo.)					
F118[h]	50				06/20/2012	23		≥1 observed
F119	66				08/09			2
F119[i]	96 expected				02/12 expected	29 expected		1 plus 1-2 uterine scars
F135	33				07/06/11			2
F136[j]	39				07/10/11			≥1 remains
F136	51				07/05/12	12		2
F137	30				07/08/11			≥1
F171	22				08/11			2

[a] Ages of females were estimated at litter birth dates. Ages of males were estimated around the dates the pairs consorted.

[b] Consort pairs indicate pumas that were observed together based on GPS data or VHF location data.

[c] Estimated birth dates were indicated by GPS data of mothers at nurseries or by back-aging cubs to approximate birth date.

[d] Observed number of cubs do not represent litter sizes as some cubs were observed when they were 5 to 16 months old after postnatal mortality could have occurred in siblings. Only cub tracks were observed with F28.

[e] Asterisk (*) indicates first probable litter of the female, based on nipple characteristics noted at first capture of the female.

[f] A radio-collared, ear-tagged male puma was visually observed with F23 on 2/25/08. Both M27 and M29 wore non-functional GPS collars in that area at the time.

[g] When captured on 1/20/10, puma F116 was in association with 2 large cubs which were not captured.

[h] One cub observed with F118 in Maverick Draw 7/19/2012.

[i] F119 died of a ruptured uterus and internal bleeding on 1/28/12. Cub in uterus in third trimester; 1-2 uterine scars indicated expulsion of 1-2 fetuses.

[j] Remains of F136's cubs found 8/9/11. Cause of death predation by puma or black bear.

Table 14. Summary for individual adult puma survival and mortality, December 8, 2004 to July 31, 2012, Uncompahgre Plateau, Colorado.

Puma I.D.	Monitoring span	Status: Alive/Lost contact/Dead; Cause of death
M1	12-08-04 to 08-16-06	Dead. Lost contact− failed GPS/VHF collar. M1 ranged principally north of the study area as far as Unaweep Canyon. M1 was killed by a puma hunter on 01-02-10 west of Bang's Canyon, north of Unaweep Canyon, GMU 40. M1 was about 97 months old at death.
M4	01-28-05 to 12-28-05	Dead; killed by a male puma. Estimated age at death 37−45 months.
M5	08-01-06 to 02-20-09	Dead. Born on study area; offspring of F3. M5 was independent of F3 by 13 months old, and dispersed from his natal area at about 14 months old. Established adult territory on northwest slope of Uncompahgre Plateau at the age of 24 months (protected from hunting mortality in buffer area) and ranged into the eastern edge of Utah (vulnerable to hunting). Killed by a puma hunter on 02-20-09 in Beaver Creek, Utah at age 54 months.
M6	02-18-05 to 05-21-10	Dead. M6 was struck and killed by a vehicle on highway 550 south of Colona, CO on 05-21-10. M6 was about 99 months old at death.
M27	03-10-06 to 05-07-09	Dead. Lost contact− failed GPS/VHF collar. Recaptured 12-02-07 & 01-22-08 by puma hunter/outfitter north of the study area. Possibly visually observed on study area with F23 on 02-25-08. Recaptured by a puma hunter/outfitter 12-11-08 & 12-28-08 north of the study area. Photographed by a trail camera on the study area (Big Bucktail Canyon) on 5 occasions: 03-27-09, 04-02-09, 04-15-09, 04-24-09, & 05-07-09. M27 was killed by a puma hunter on 12-09-09 in the North Fork Mesa Creek, Uncompahgre Plateau, GMU 61 North. M27 was about 100 months old at death.
M29	04-14-06 to 02-25-09	Dead. Lost contact− failed GPS/VHF collar. Possibly visually observed on study area with F23 on 02-25-08. Recaptured on study area 02-25-09, but could not be safely handled to change faulty GPS collar. M29 was killed by a puma hunter on 11-16-09 in Beaver Canyon, GMU 70 East. M29 was about 121 months old at death.
M32	04-26-06 to 12-02-10	Dead. Killed by a puma hunter on 12-02-10 in McKenzie Creek on the Uncompahgre Plateau study area. M32 was about 112 months old at death.
M51	01-07-07 to 03-20-09	Dead. Lost contact− failed GPS/VHF collar after 03-20-09. Killed by a puma hunter on 12-11-09 in Shavano Valley, Uncompahgre Plateau study area. M51 was about 77 months old at death.
M55	01-21-07 to 07-31-10	Dead. Killed by a puma hunter on 11-25-10 in Spring Creek Canyon on the Uncompahgre Plateau study area. M55 was about 77 months old at death.
M67	08-23-07 to 12-18-11	Dead. M67 is offspring of F30. Dispersed natal area. Established territory on W side U.P. study area. Killed by a puma hunter in Tabaguache Creek 12-18-2011 at age 52.9 months.
M71	01-29-08 to 11-12-09	Dead. Lost contact– M71 shed his VHF collar with an expansion link on about 11-12-09. He was killed by a puma hunter on 12-09-09 on the west rim of Spring Creek Canyon, Uncompahgre Plateau study area. M71 was about 47 months old at death.
M73	02-21-08 to 10-26-11	Dead. Illegally killed 10-26-2011 in Bear Pen Gulch, upper East Fork Escalante Canyon; shot through abdomen during second rifle season. M73 was about 80 months old at death.
M87	02-09-11 to 12-06-11	Dead. M87 is offspring of F3. Dispersed from natal area. Established territory on W side of U.P. study area. Killed by a puma hunter in 47 Canyon, Tabaguache Canyon 12-06-2011. M87 was 41 months old at death.
M90	11-16-10 to 11-23-10	Dead. M90 was killed by a hunter on 11-23-10 on McKenzie Butte. M90 was offspring of F72, born 07-09-08. He was 28 months old at death.
M100	03-27-09 to 07-31-09	Dead. M100 was killed by a puma hunter on 01-16-10 in Naturita Canyon, GMU 70 East. M100 was about 63 months old at death.
M114	02-27-10 to 03-10-12	Dispersed from U.P. study area after 06-23-10. Killed by a puma hunter in Beaver Creek, NE of Canyon City, GMU59, 03-10-12. M114 was about 55 months old at death.
M133	11-12-10 to 12-01-10	Dead. M133 was killed by a puma hunter on 12-01-10 in Dry Fork Escalante Canyon north of the study area. M133 was about 43 months old at death.

Table 14. Continued.

Puma I.D.	Monitoring span	Status: Alive/Lost contact/Dead; Cause of death
M134	06-01-11 to 06-10-11	Dead. M134 was offspring of unmarked female puma in Roubideau Canyon. Independent by about 03-28-11. Shot dead by USDA, APHIS, WS agent while in the act of attacking domestic sheep on 06-10-11 when he was 24 months old at start of adult life stage.
M138	07-01-11 to 12-23-11	Dead. Killed by a puma hunter in Horsefly Canyon (E) 12/23/11. M138 was about 29 months old at death.
M144	09-01-11 to 07-31-12	Alive. Initially captured as 18 mo. old subadult on W side U.P. study area 03-07-11. Established adult territory on NW U.P.
M153	09-01-11 to 09-13-11	Dead. Killed for depredation control; killed an alpaca in Pleasant Valley 09-13-11.
M165	07-01-12 to 07-31-12	Alive. Initially captured as 19 mo. old subadult on W side U.P. study area 02-24-12. Moved to Escalante Creek drainage by adult age 07-31-12.
F2	01-07-05 to 08-14-08	Dead; killed by another puma (sex of puma unknown; male suspected) 08-14-08. F2 was about 92 months old at death.
F3	01-21-05 to 12-11-11	Dead. Killed by a puma hunter in Lindsay Creek 12-11-11. F3 was about 120 months old at death.
F7	02-24-05 to 08-03-08	Dead. Killed by U.S. Wildlife.Services agent 08-03-08 for predator control of depredation on domestic sheep. F7 was about 107 months old at death.
F8	03-21-05 to 07-31-12	Alive.
F16	10-11-05 to 09-11-09	Dead. F16 was struck and killed by a vehicle on Ouray County Road 1 southwest of Colona, CO on 09-11-09. F16 was about 80 months old at death.
F23	02-05-06 to 06-06-12	Dead. Killed by a male puma about 06-06-12. F23 was about 94 months old at death. F23 may have attempted to defend 2 cubs (F149, M161; 13.5 months old) and/or calf elk kill.
F24	01-17-06 to 07-31-11	Dead. Killed by a male puma in Logging Camp Draw about 09-16-11. F24 was about 126 months old at death. F24 may have attempted to defend ≥2 cubs (F147, non-marked siblings; 12 mo. old).
F25	02-08-06 to 02-03-11	Dead. Lost radio contact after 09-04-09– failed GPS/VHF collar. Photographed alive with three ~9 month old cubs on 12-03-10 on Loghill Mesa. F25 shot dead by a ranch hand on 02-03-11 in Pleasant Valley, Dallas Creek because she was seen among cattle. F25 was about 138 months old at death and in excellent physical condition (49 kg).
F28	03-23-06 to 02-16-12	Lost radio contact after 09-25-07− failed GPS/VHF collar. Recaptured F28 on the study area 02-01-10 and 01-01-11 and 02-16-12, but could not be handled to replace non-functional GPS collar.
F30	04-15-06 to 07-29-08	Dead. Killed by another puma (sex of puma unknown) 07-29-08. F30 was about 60 months old at death.
F50	12-14-06 to 03-26-07	Dead of natural causes 03-26-07; probably injury or illness-related; exact agent unknown. F50 was about 30 months old at death.
F54	01-12-07 to 08-18-07	Dead; killed by a male puma while in direct competition for prey (i.e., mule deer fawn) 08-18-07. F54 was about 49 months old at death.
F70	01-14-08 to 12-22-11	Dead. Killed by a puma hunter Spring Creek 12-22-11. F70 was 80 months old at death. Her death orphaned 2 cubs, F157 and F158, at 4 months old; both starved to death about 01-15-12 at about 5 months old.
F72	02-12-08 to 12-21-11	Lost radio contact after 12-02-10. F72 recaptured in Fisher Creek on 03-18-11, but could not be handled to replace non-functional GPS collar. Photographed on Miller Mesa S of U.P. study area on 12-18 to 21-11 with 3 new cubs born about July 2012.
F75	03-26-08 to 12-13-11	Dead. Killed by a puma hunter in North Fork Cottonwood Creek 12-13-11. F75 was about 98 months old at death.
F93	12-05-08 to 07-31-12	Alive.
F94	12-19-08 to 02-01-11	Dead. Shot dead on 02-01-11 by USDA, APHIS, WS agent for predation on domestic elk in Happy Canyon. F94 was about 74 months old at death.
F95	08-01-09 to 07-31-12	Alive.
F96	01-28-09 to 07-31-12	Alive.
F104	05-21-09 to 01-31-12	Dead. Died probably of starvation associated with senescence in lower Roubideau Creek 01-31-12. F104 was about 132 months old at death.

Table 14 continued.

Puma I.D.	Monitoring span	Status: Alive/Lost contact/Dead; Cause of death
F110	09-21-09 to 02-25-10	Dead. Killed by a puma hunter on 02-25-10 in GMU 70 East. F110 was about 41 months old at death.
F111	01-01-10 to 07-31-12	Alive.
F113	01-26-10 to 06-06-10	Dead. F113 died 06-06-10 of injuries consistent with being struck by a vehicle. GPS data indicated that F113 had crossed highway 550 and roads on Loghill Mesa north of Ridgway 24-30 hours before she died in McKenzie Creek. F113 was about 42 months old at death.
F116	01-20-10 to 09-20-11	Dead. Died about 09-20-11 of unknown natural cause associated with pregnancy and birth of new litter of cubs. F116 was about 60 months old at death.
F118	02-25-10 to 07-31-12	Alive.
F119	03-25-10 to 01-28-12	Dead. Died of ruptured uterus and internal bleeding associated with pregnancy in Clay Creek Canyon 01-28-12. F119 was about 95 months old at death.
F135	01-01-11 to 09-20-11	Dead. Died of unknown natural cause in E Fork Dry Creek 09-20-11. Her death orphaned cubs M154 and M155 at 76 days old; both died of starvation or disease when 77 (M154) and 81 (M155) days old.
F136	01-20-11 to 07-31-12	Alive.
F137	01-21-11 to 07-31-12	Alive.
F143	02-15-11 to 07-31-12	Alive.
F152	06-16-12 to 07-31-12	Alive.
F163	07-01-12 to 07-31-12	Alive.
F171	01-20-12 to 07-31-12	Alive.
F172	03-28-12 to 07-31-12	Alive.

Table 15. Preliminary estimated survival rates *(S)* of adult-age pumas during the 4 years in the *reference period* (i.e., the study area is closed to puma hunting) and 2 years in the *treatment period*, Uncompahgre Plateau, Colorado. Survival rates of pumas estimated with the Kaplan-Meier procedure to staggered entry of animals (Pollock et al. 1989). Survival rates are for an annual survival period defined as the biological year (August 1 to July 31). Survival rates were estimated only for periods when $n \geq 5$ individual pumas were monitored in the interval. Puma survival in the *reference period* pertained only to pumas that died of natural causes. Pumas that were killed by people in the *reference period*, a non-natural cause (i.e., two adult pumas: F7 for depredation control 8/3/2008 and M5 killed by a puma hunter off the protected study area and buffer zone 2/20/2009) were right censored. In the *treatment period* all sources of natural and human-caused mortality are considered in the survival estimates.

Biological Year	**Females**			**Males**		
	S	**SE**	***n***	***S***	**SE**	***n***
Reference Annual 2 8/1/2005 to 7/31/2006	1.000	0.0000	10	0.667[a]	0.2222[a]	6[a]
Reference Annual 3 8/1/2006 to 7/31/2007	0.909	0.0867	11	1.000	0.0000	5
Reference Annual 4 8/1/2007 to 7/31/2008	0.831	0.0986	14	1.000	0.0000	7
Reference Annual 5 8/1/2008 to 7/31/2009	0.875	0.1031	13	1.000	0.0000	8
Treatment Annual 1 8/1/2009 to 7/31/2010	0.784	0.1011	19	0.667	0.1924	8
Treatment Annual 1[b] 8/1/2009 to 7/31/2010 With mortalities of all marked adult males				0.333[b]	0.1361[b]	12[b]
Treatment Annual 2 8/1/2010 to 7/31/2011	0.947[c]	0.0568	19	0.250	0.1082	9
Treatment Annual 3 8/1/2011 to 7/31/2012	0.548[d]	0.1063	20	0.167	0.1076	7[d]

[a] Adult male annual *S* 2005 to 2006 is probably underestimated with poor precision because 3 of the 6 pumas were GPS/VHF-monitored for 4 to 5 months at the end of the interval; 1 of 6 adult males died.

[b] This second estimate of adult male puma survival 8/1/2009 to 7/31/2010 includes 5 males that had non-functional (4) or shed (1) radiocollars. All adult males with non-functional or shed radiocollars in this study survived into *treatment year* 1 (TY1), which was expected considering adult male survival in 3 previous years. All 5 of those adult males were detected and killed by hunters in TY1.

[c] Only 1 of 2 adult female puma mortalities is represented in this survival analysis for 8/1/2010 to 7/31/2011, that of F94 killed for depredation control. One other adult female mortality, F25, is not represented because she wore a non-functional GPS collar making it impossible for us to monitor her survival. F25 was shot by a ranch hand on 2/3/2011 when he saw her among cattle.

[d] Sample includes M144, ranges on NW Uncompahgre Plateau N of the study area but not on the U.P. study area, vulnerable to annual hunting.

Table 16. Summary of subadult puma survival and mortality, December 2004 to July 2012, Uncompahgre Plateau, Colorado.

Puma I.D.	Monitoring span	No. days	Status
M5	09-16-05 to 06-30-06	308	Survived to adult stage. M5 was offspring of F3, born August 2004. Independent and dispersed from natal area at 13 months old. Established adult territory on northwest slope of Uncompahgre Plateau at the age of 24 months (protected from hunting mortality in buffer area) and ranged into the eastern edge of Utah (vulnerable to hunting). Killed by a puma hunter on 02-20-09 in Beaver Creek, Utah at about 54 months old.
M11	06-21-06 to 12-02-07	529	Survived to adult stage. M11 was offspring of F2, born May 2005. Independent at 13 months old. Dispersed from natal area at 14 months old. Moved to Dolores River valley, CO, by 12-14-06. Killed by a puma hunter on 12-02-07 when about 30 months old.
F23	01-04-06 to 02-04-06	31	Survived to adult stage. Captured on the study area when about 17 months old. Survived to adult stage; gave birth to first litter at about 21 months old. Killed by a male puma about 06-06-12. F23 was about 94 months old at death.
M31	04-19-06 to 04-26-06	7	Survived to adult stage. M31’s estimated age at capture was 20 months. Dispersed to northern New Mexico and was killed by a puma hunter on 12-11-08 in Middle Ponil Creek, Cimarron Range. He was about 52 months old.
M49	03-26-07 to 10-01-07	189	Survived to adult stage. M49 was offspring of F50, born July 2006. Orphaned at about 9 months old, when F50 died of natural causes. Dispersed from his natal area at about 10 months old and ranged on the northeast slope of the Uncompahgre Plateau. When M49 was about 15 months old, he shed his expandable radiocollar on about 10-01-07 at a yearling cow elk kill on the northeast slope of the Uncompahgre Plateau. He was killed by a puma hunter in Blue Creek in the protected buffer zone north of the study area on 01-24-09; he was about 29 months old, a young adult.
F52	01-10-07 to 05-15-07	125	Survived to adult stage. F52 dispersed from study area as a subadult by 01-16-07. F52’s last VHF aerial location was Crystal Creek, a tributary of the Gunnison River east of the Black Canyon 05-15-07. She was treed by puma hunters on 12-29-08 on east Huntsman Mesa, southeast of Powderhorn, CO. She was about 41-43 months old and could have been in her adult-stage home range. GPS collar nonfunctional. F52 was killed by a puma hunter on 01-09-12 in North Beaver Creek SE of Powederhorn, CO. She was about 79 months old at death.
F66	08-23-07 to 11-05-07 11-25-08 to 06-03-09	74 190	Died in subadult stage. F66 was offspring of F30, born July 2007. Lost contact; her cub collar quit after 11-05-07. Recaptured as an independent subadult on her natal area 11-25-08 when 16 months old. Mother F30 was killed by a puma when F66 was 12 months old, within the age range of normal independence. F66 died of injuries to internal organs that caused massive bleeding attributed to trampling by an elk or mule deer on about 05-28-09 when she was 23 months old. Her range partially overlapped her natal area.
M69	01-11-08 to 04-07-08	87	Survived to adult stage. M69 was captured on the study area when about 14-18 months old. Emigrated from the study area as subadult by 03-19-08. Last VHF aerial location was southwest of Waterdog Peak, east side of Uncompahgre River Valley on 04-07-08. M69 was killed by a puma hunter on 11-06-08 in Pass Creek in the Snowy Range, WY when he was 24 to 28 months old.

Table 16 continued

Puma I.D.	Monitoring span	No. days	Status
F95	12-29-08 to 07-31-12	214	Alive. F95 is the offspring of F93, born about August 2007. She became an independent subadult by about 18 months old (02-11-09 aerial location) and an adult by about 24 month old (Aug. 2009). F95 established an adult home range adjacent to and overlapping the northern portion of her natal area.
M99	02-27-09 to 04-22-09	54	Died in subadult stage. M99 probably killed by another puma (canine punctures in skull including braincase) in Jan. 2010 when he was about 16 months old. His radiocollar quit after 54 days.
M112	02-10-11 to 04-18-11	67	M112 was offspring of F70. Lost contact of M112 after 04-18-11; he may have dispersed or radiocollar quit. M112 associated with F96 and her two radio-collared cubs F129 and M130 during 02-10-11 to 04-18-11.
M115	01-13-10 to 07-21-10	189	Died in subadult stage. M115 was offspring of F28, born in Nov. 2008. He was about 14 months old when first captured on Jan. 13, 2010. When he was recaptured on 03-18-10, he had previously suffered a broken left ulna. M115 was probably independent by 07-15-10 when he was located outside of his natal area on a probably dispersal move. M115 died on about 07-21-10 apparently from complications of his broken left foreleg; probably not allowing him to kill prey sufficiently for survival. M115 was about 20 months old at death.
M120	12-06-12	1	Died in subadult stage. M120 was offspring of F3. M120 was killed by a puma hunter 12-06-12 in his natal area in Spring Creek. He was 17 months old at death.
M134	03-28-11 to 06-10-11	74	Survived to adult stage (barely). M134 was offspring of unmarked female puma in Roubideau Canyon. Independent by about 03-28-11. Shot dead by USDA, APHIS, WS agent while in the act of attacking domestic sheep on 06-10-11 when he was 24 months old at start of adult life stage.
M138	01-26-11 to 06-30-11	155	Survived to adult stage. Entered adult life stage 07-01-11. Killed by a puma hunter 12-23-11 in Horsefly Canyon. M138 was about 29 months old at death.
F140	01-13-12 to 07- 31-12	200	Survived to late subadult stage. Will turn adult in Aug. 2012. Probably offspring of F28. Has established a home range adjacent to natal area where she was initially captured at 5 months old on 01-02-11.
M141	12-23-11	1	Died in subadult stage. M141 was killed by a puma hunter on 12-23-11 in Little Bucktail Creek. He was 16 months old at death.
M144	03-07-11 to 09-08-11	185	Survived to adult stage. Emigrated from U.P. study area. Established adult territory on northwest Uncompahgre Plateau. M144 is sibling of F145 below.
F145	03-08-11 to 09-08-11	184	Survived to adult stage. Emigrated from U.P. study area and to Colorado Mesa. Killed by a puma hunter 01-23-12 in West Bangs Canyon. F145 was 28 months old at death.
F146	03-08-11 to 03-23-11	15	Died in subadult stage. F146 was killed and eaten by a male puma while in competition for an adult bull elk carcass that one of the pumas killed in Coal Canyon on the study area. F146 was about 19 months old at death.
F147	09-16-11 to 04-12-12	209	Lost contact; radiocollar quit after 04-12-12. F147 orphaned at about 12 months old when her mother F24 was killed by a male puma on 09-16-11.

Table 16 continued.

Puma I.D.	Monitoring span	No. days	Status
F149	06-06-12 to 07-31-12	55	F149 (sibling of M161 below) was orphaned at 13.5 months old when her mother F23 was killed by a male puma. F149 dispersed from the natal area by 07-16-12 to E side U.P. study area when she was 14.8 months old.
M150	03-28-11 to 04-11-11	14	Dispersed. M150 was offspring of F111, born on 08-31-09. He was independent by 03-28-11 when he was 19 months old. Lost contact after 04-11-11 when M150 was in Cow Creek southeast of the study area.
F152	05-04-12 to 06-16-12	44	Survived to adult stage. F152 was independent from her mother F93 by 05-04-12 when about 23 months old. She ranged as a subadult and adult on the natal area (07-31-12).
M153	04-12-11 to 09-06-11	147	Survived to adult stage. Consorted with F137 when 23 months old on 09-07-2011. Killed by Wildlife Services agent for depredation on an alpaca in Dallas Creek on 09-13-11. M153 was 23 months old at death.
M161	06-06-12 to 07-31-12	55	M161 (sibling of F149 above) was orphaned at 13.5 months old when his mother F23 was killed by a male puma. M161 dispersed from the natal area by 06-29-12 to E side U.P. study area when he was 14 months old.
F163	01-26-12 to 07-01-12	157	F163 was captured at about 18 months old on the study area. She emigrated from the study area and may have established an adult home range on the N portion of the Uncompahgre Plateau as of July 2012 (07-16-12 most recent location).

Table 17. Records of pumas that dispersed from the Uncompahgre Plateau study area, December 2004 to July 2012.

Puma I.D.	1st capture date on study area	1st capture location→kill or resight location (UTM, NAD27)	Estimated linear dispersal distance (km)*	Puma Information
M5	02-04-05	13S,240577E, 4251037N→ 12S,665853Ex 4277125N	102.2	M5 was offspring of F3, born August 2004. Independent and dispersed from natal area at 13 months old. Established adult territory on northwest slope of Uncompahgre Plateau at the age of 24 months (protected from hunting mortality in buffer area) and ranged into the eastern edge of Utah (vulnerable to hunting). Killed by a puma hunter on 02-20-09 in Beaver Creek, Utah at about 54 months old.
M11	06-27-05	13S,248278E, 4239858N→ 12S,741882Ex 4161575N	84.8	M11 was offspring of F2, born May 2005. Shed expandable radiocollar 10-24 to 11-08-05. Recaptured and re-collared 04-02-06. Independent at 13 months old. Dispersed from natal area at 14 months old. Moved to Dolores River valley, CO, by 12-14-06. Killed by a puma hunter on 12-02-07 when about 30 months old.
M31	04-19-06	12S,746919E, 4225441N→ 13S,500000Ex 4050000N	329.8	M31's estimated age at capture was 20 months. Dispersed to northern New Mexico and was killed by a puma hunter on 12-11-08 in Middle Ponil Creek, Cimarron Range. He was about 52 months old.
M38	09-08-06	13S,249200E, 4239703N→ 12S,703371E, 4316856N	104.1	M38 was offspring of F2, born July 29, 2006. Shed his expandable radiocollar by 03-06-07. Photographs by trail camera in McKenzie Cr. of M38 & Unm. F sibling with F2 on 07-16 to 17-07 at 352-353 days old. M38 was killed by a puma hunter in Ladder Creek southwest of Grand Junction, CO on 01-07-11. He was 53.2 months old at death.
M39	09-11-06	12S,724270E, 4243610N→ 12S,709889E, 4313490N	71.3	M39 was offspring of F8, born August 2006. M39 was killed by a puma hunter in Bangs Canyon, GMU 40 on 03-12-10 when he was 42.8 months old.
M43	09-15-06	12S,760177E, 4242995N→ 12S,739859E, 4308557N	68.6	M43 was offspring of F7, born August 2006. He shed the expandable radiocollar 11-7 to 17-06, after which direct contact was lost. M43 was killed by a puma hunter 01-28-09 in Deer Creek, west slope of Grand Mesa, CO when he was 29.5 months old.
M48	10-18-06	12S,756676E, 4247777N→ 12S,704982E, 4248998N	52.0	M48 was the offspring of F3, born September 2006. M48 was killed by a puma hunter in Tabeguache Creek, GMU 61N on 12-27-09 when he was 38.9 months old.
M49	12-05-06	12S,757241E, 4258259N→ 12S,693350E, 4274559N	66.1	M49 was offspring of F50, born July 2006. Orphaned at about 9 months old, when F50 died of natural causes. Dispersed from his natal area at about 10 months old and ranged on the northeast slope of the Uncompahgre Plateau. When M49 was about 15 months old, he shed his expandable radiocollar on about 10-01-07 at a yearling cow elk kill on the northeast slope of the Uncompahgre Plateau. He was killed by a puma hunter in Blue Creek GMU 61N in the protected buffer zone north of the study area on 01-24-09; he was about 29 months old.
M58	06-27-07	13S,258543E, 4238071N→ 13S,274670E, 4309488N	73.2	M58 was offspring of F16, born May 2007. M58 was killed by a puma hunter on 12-27-09 in the North Fork of the Gunnison River north of Paonia, GMU 521; he was 31 months old.

Table 17 continued.

Puma I.D.	1st capture date on study area	1st capture location→kill or resight location (UTM, NAD27)	Estimated linear dispersal distance (km)*	Puma Information
M63	08-17-07	12S,738144E, 4233628N→ 12S,689111E, 4277908N	66.1	M63 was offspring of F24, born July 14, 2007. He was not radio-collared as a cub. M63 was killed by a puma hunter in Calamity Creek on northwest Uncompahgre Plateau on 01-01-11. M63 was 41.5 months old at death.
M65	08-17-07	12S,738144E, 4233628N→ 12S,684084E, 4314200N	97.0	M65 was offspring of F24, born July 2007. M65 was killed by a USDA, APHIS, WS agent for depredation on llamas in the Little Dolores River on 11-07-09. M65 was 27.8 months old.
M67	08-23-07	13S,257371E, 4235231N→ 12S,725113E, 4242447N	57.7	M67 was offspring of F30, born July 17, 2007 in Fisher Creek on the east slope of the study area. He was not radiocollared as a cub. M67 dispersed from the natal area and was recaptured in Tomcat Creek on the west slope of the study area on 02-24-10 when he was 31 months old. M67 is a resident adult in that area (07-31-11). Killed by puma hunter in GMU61N on 12-18-11 when 52.9 months old.
M68	08-23-07	13S,257371E, 4235231N→ 12S,711262E, 4198681N	80.7	M68 was offspring of F30, born July 2007. He was orphaned at 12 months old when his mother was killed by a puma. He was killed by a puma hunter in the Disappointment Valley in southwest CO on 12-30-08; he was 17 months old.
M69	01-11-08	13S,248191E, 4246810N→ 13T,378900E, 4591990N	369.6	M69 was captured on the study area when about 14-18 months old. Emigrated from the study area as subadult by 03-19-08. Last VHF aerial location was southwest of Waterdog Peak, east side of Uncompahgre River Valley on 04-07-08. M69 was killed by a puma hunter on 11-06-08 in Pass Creek in the Snowy Range, WY when he was 24 to 28 months old.
M82	07-05-08	12S,726901E, 4243463N→ 13S,255316E, 4216768N	60.5	M82 was offspring of F8, born May 29, 2008; sibling of M83 below. He shed his expandable cub radiocollar after 03-20-09. M82 was killed by a puma hunter on 12-10-09 in the Beaver Creek fork of East Dallas Creek, GMU 65. M82 was 19 months old.
M83	07-05-08	12S,726901E, 4243463N→ 12S,670949E, 4314779N	90.7	M83 was offspring of F8, born May 29, 2008; sibling of M82 above. He was not radiocollared as a cub. M82 was killed by a puma hunter on 01-18-11 in Coates Creek west of Glade Park, CO. He was 30 months old at death.
M87	07-31-08	13S,239006E, 4248601N→ 12S,724325E, 4244118N	39.2	M87 was offspring of F3, born July 3, 2008 on the east slope of the study area; sibling of M88 below. He was not radiocollared as a cub. M87 dispersed from the natal area. He was recaptured on the west slope of the study area on 02-09-11 when he was 31 months old. M87 is was resident adult on the west slope of the study area. He was killed by a puma hunter on 12-06-11 at 41 months old north of the study area.
M88	07-31-08	13S,239006E, 4248601N→ 12S,704835E, 4197839N	77.6	M87 was offspring of F3, born July 3, 2008 on the east slope of the study area; sibling of M87 above. He was not radiocollared as a cub. M87 dispersed from the natal area. He was killed by a puma hunter in Dawson Creek, Disappointment Valley on 11-30-10 when he was 29 months old.
M92	09-29-08	13S,246359E, 4226949N→ 12S,750871E, 4222921N	21.9	M92 was offspring of F25, born August 19, 2008. He was radiocollared as a cub; last contact on 12-12-08. M92 dispersed from the natal area and was recaptured in McKenzie Creek, west slope of the study area on 04-22-11 when he was 32 months old. He could not be handled to fit a new radiocollar because of a dangerous tree.

Table 17 continued.

Puma I.D.	1st capture date on study area	1st capture location→kill or resight location (UTM, NAD27)	Estimated linear dispersal distance (km)*	Puma Information
M107	06-28-09	13S,242359E, 4252618N→ 12S,754886E, 4341330N	89.2	M107 was offspring of F94, born May 25, 2009; sibling of F108 below. He was not radiocollared as a cub. M107 dispersed from the nata area. He was killed by a puma hunter in Cottonwood Creek near Molina, CO on 12-09-10 when he was 19 months old.
M114	02-27-10	13S,256933E, 4237862N→ 13S,492615E, 4266192N	237.5	M114 was initially captured at about 30 months old. Emigrated from the U.P. study area. He was killed by a puma hunter on 03-10-12 in Beaver Creek, GMU59. He was about 55 months old at death.
M117	02-05-10	12S,731840E, 4232346N→ 12S,743909E, 4216633N	19.7	M117 was offspring of F119. He wore an expandable cub collar, but shed the collar by 07-15-10 on the natal area when about 11 months old. M117 was killed by a puma hunter in Beaver Creek, San Miguel River at the southern extreme of his natal area on 01-01-11. He was 17 months old at death. It is unknown if M117 was independent from his mother F119 at the time of his death.
M126	09-05-10	12S,734503E, 4224636N→ 12S, 710850E, 4239350N	27.7	M126 was offspring of F118, born Aug. 8, 2010. Lost radio contact after 03-17-11; shed his radiocollar at a mule deer cache. Dispersed from natal area. Killed by a puma hunter on 01-08-12 in Tuttle Draw WNW of Nucla, CO as 17-month-old subadult.
M144	03-07-33	12S,727173E, 4242012N→ 12S,696439E, 4276888N	46.6	M144 was initially captured as an independent subadult in association with subadults F145 and F146 on the study area. Mother is unknown. He moved off the study area on 03-15-11. M144's last aerial radio location was in Blue Creek on northwest Uncompahgre Plateau on 07-13-11; he was about 22 months old. M144 established his adult territory on northwest Uncompahgre Plateau and upper Unaweep Canyon from Sep. 2011 to July 2012.
M161	01-23-12	12S,727932E, 4239430N→ 12S,750473E, 4247250N	23.9	M161 (sibling of F149) was orphaned when his mother F23 was killed by a male puma on 06-06-12; he was 411 days (13.5 mo.) old. M161 dispersed from the natal area by 06-29-12 when he was 14 months old and moved to the east slope of the U.P. study area.
F52	01-10-07	13S,258058E, 4236260N→ 13S,319217E, 4240467N	61.1	F52 was captured on the study area when about 18-20 months old. Dispersed from study area as a subadult by Jan. 16, 2007. F52's last VHF aerial location was Crystal Creek, a tributary of the Gunnison River east of the Black Canyon 05-15-07. She was treed by puma hunters on 12-29-08 on east Huntsman Mesa, southeast of Powderhorn, CO. She was about 41-43 months old . F52 was treed again by puma hunters on about 12-16-09 south of Powderhorn: 13S,319480E,4233219N. F52 was about 53-55 months old. This suggests that F52 has an adult home range in that area. F52 was killed by a puma hunter on 01-09-12 in North Beaver Creek SE of Powederhorn, CO. She was about 79 months old at death.
F97	02-04-09	12S,727529E, 4237648N→ 12S,705930E, 4227299N	24.0	F97 was offspring of F23, born May 23, 2008. She was radio-collared at 8.5 month old in San Miguel Canyon; but, lost contact on 05-12-09 after F97 shed the radiocollar at an elk cache. F97 dispersed from the U.P. study area. She was killed by a puma hunter on 01-22-12 in Dry Creek west of the U.P. study area when she was 43.9 months old.
F106	06-14-09	12S,736451E, 4240278N→ 13S,258089E, 4235866N	46.9	F106 was offspring of F75, born May 7, 2009. She wore an expandable cub collar, but shed it about 03-23-10. F106 dispersed from the natal area and moved to the east slope of the study area where she was photographed at one of our scent station cameras at the mouth of Fisher Creek from 02-27-11 to 03-03-11. She was identified by her eartag. F106 was 21 months old.

Table 17 continued.

Puma I.D.	1st capture date on study area	1st capture location→kill or resight location (UTM, NAD27)	Estimated linear dispersal distance (km)*	Puma Information
F108	06-28-09	13S,242359E, 4252618N→ 12S,752013E, 4263883N	18.2	F108 was offspring of F94, born May 25, 2009; sibling of M107 above. She was fitted with an expandable cub collar; but, shed the collar in the original nursery due to failure of the fastener. F108 dispersed from the natal area. She was killed by a puma hunter on the study area on 11-29-10 when she was 17 months old.
F143	02-15-11	12S,723748E, 4238579N→ 12S,721795, 4264246	25.7	F143 was captured on the study area when about 24 months old. Dispersed N on the Uncompahgre Plateau and established an adult home range on the NW portion of the Uncompahgre Plateau (most recent location 07-16-12).
F145	03-18-11	12S,727181E, 4241468N→ 12S,705833E, 4312909N	74.5	F145 was originally captured in association of M144 and F146; they may be siblings. Mother unknown. She moved off the study area with M144 on 03-15-11. F145 emigrated to Colorado Mesa. She was killed by a puma hunter 01-23-12 in West Bangs Canyon. F145 was 28 months old at death.
F149	06-06-11	12S,729993E, 4242329N→ 12S,715551E, 4285489N	45.5	F149 (sibling of M161) was orphaned when her mother F23 was killed by a male puma on 06-06-12; she was 411 days (13.5 mo.) old. F149 dispersed from the natal area by 07-16-12 when she was 14.8 months old and moved to the NE Uncompahgre Plateau.
F163	01-26-12	12S,732153E, 4232452N→ 12S,695407E, 4280753N	60.7	F163 was initially captured at about 18 months old. She emigrated from the study area and may have established an adult home range on the N portion of the Uncompahgre Plateau as of July 2012 (07-16-12 most recent location).

*Estimated linear dispersal distance (km) from initial capture site on Uncompahgre Plateau study area to hunter kill, or last recapture, radio location, or observation site.

Table 18. Recorded deaths of non-marked and marked pumas struck by vehicles and other unusual causes, in chronological order, on the Uncompahgre Plateau puma study area, Colorado, from 2004 to 2012.

Puma sex & ID if marked	**Estimated age (mo.)**	**Date recorded**	**Cause of death**	**General physical condition**	**Location & UTM NAD27**
M	12	09-24-04	Vehicle collision	Good	Pleasant Valley, County Road 24 13S,252870E,4227520N
F	49	07-28-05	Vehicle collision	Good Not pregnant or lactating	Highway 62 east of Dallas divide 13S,250000E,4222500N
F17[a]	11	08-18-06	Vehicle collision	Good	Highway 550 south of Colona 13S,257602E,4242185N
F	18-24	11-06-06	Vehicle collision	Good	Highway 550 east of Ridgway State Park 13S,259843E,4235985N
F	6	01-30-07	Vehicle collision	Good	Highway 62 west of Dallas divide 12S,762286Ex4218992N
F P1005	36	09-16-08	Asphyxia, lodged in fork of tree	Unknown, decomposed	Davis Point, Roubideau Canyon 12S, 743718E,4255277N
M	12-24	08-13-08	Vehicle collision	Good	Highway 145 west of Placerville 13S,756490E,4212336N
F61[a]	18	11-13-08	Vehicle collision	Good	Highway 550 east of Ridgway State Park 13S,259843E,4235985N
F	12	08-10-09	Vehicle collision	Good	Highway 145 east of Norwood 12S,745739E,4222548N
F16[b]	80	09-11-09	Vehicle collision	Good	Ouray County Road 1 13S,253733E,4240060N
M6[b]	99	05-21-0	Vehicle collision	Good	Highway 550 south of Colona 13S,258610E,4236805N
F113[b]	42	06-06-10	Vehicle collision	Good Not pregnant or lactating	F113 crossed Highway 550 and roads on Loghill Mesa 24-30 hours before she died in McKenzie Creek 13S,257272E,4238435N
M P1018[c]	24	08-25-10	Vehicle collision	Excellent	Highway 62 Leopard Creek 12S,237747E,4220330N
F P1030[c]	6	02-16-11	Vehicle collision	Good	Highway 62 Leopard Creek 12S,760953E,4216683N
M P1034	4	10-07-11	Vehicle collision	Fair	Highway 62 Leopard Creek 12S,762806E,4219531N

[a] Subadult marked (i.e., tattoos, eartags), but not radio-collared.

[b] Adult GPS/VHF-collared pumas.

[c] Non-marked puma with P one-thousand number designation.

Table 19. Pumas monitored with GPS collars on the Uncompahgre Plateau, Colorado, December 2004 to July 2012.

Puma I.D.	Sex	Age stage	Dates monitored
M1	M	adult	12-08-04 to 07-20-06
M4	M	adult	01-28-05 to 01-14-06
M6	M	adult	02-18-05 to 05-14-08
M27	M	adult	03-12-06 to 06-21-06
M29	M	adult	04-14-06 to 01-01-08
M51	M	adult	01-07-07 to 07-15-08
M55	M	adult	01-21-07 to 11-25-10
M100	M	adult	03-27-09 to 01-16-10
M133	M	adult	11-12-10 to 12-01-10
F2	F	adult	01-07-05 to 08-14-08
F3	F	adult	01-21-05 to 12-11-11
F7	F	adult	02-24-05 to 08-03-08
F8	F	adult	03-21-05 to 10-10-06
F16	F	adult	10-12-05 to 09-10-09
F23	F	subadult	01-04-06 to 02-04-06
		adult	02-05-06 to 09-04-09
F24	F	adult	01-17-06 to 07-25-07
F25	F	adult	02-09-06 to 09-09-09
F28	F	adult	03-24-06 to 08-15-07
F30	F	adult	03-30-07 to 02-22-08
F50	F	adult	12-14-06 to 03-26-07
F52	F	subadult	01-10-07 to 05-08-07
F54	F	adult	01-12-07 to 08-18-08
F70	F	adult	01-14-08 to 12-22-11
F72	F	adult	02-12-08 to 07-07-10
F75	F	adult	03-26-08 to 06-03-09
F96	F	adult	01-28-09 to 07-31-12
F104	F	adult	05-29-09 to 01-31-12
F111	F	adult	01-01-10 to 07-31-12
F113	F	adult	01-27-10 to 06-06-10
F135	F	adult	01-01-11 to 09-20-11
F136	F	adult	01-20-11 to 07-31-12
F137	F	adult	04-12-11 to 07-31-12
F152	F	subadult	01-18-12 to 06-15-12
		adult	06-16-12 to 07-31-12
F171	F	adult	01-20-12 to 07-31-12
F172	F	adult	03-28-12 to 07-31-12

Table 20. Number of *Trichinella* larvae recovered from puma tongues, southwest Colorado, 2010-2011.

Puma Seal and/or I.D. Number	**Sex**	**Estimated Age (years)**	**Date collected**	**Location: UTM NAD27 Zone, Easting, Northing**	***Trichinella* Larvae Per Gram (LPG) of Tongue Tissue**
F94	F	5	2/1/2011	13S,246976E,4255108N	1.2
12301	M	1.5	12/12/2010	12S,735100E,4249600N	5.1
6266 (F25)	F	11-12	2/3/2011	13S,252703E,4225101N	0.4
12039	F	4-5	11/22/2010	13S,283349E,4234088N	2.0
12042	M	3-4	11/26/2010	12S,736610E,4230762N	3.2
12045	M	2-3	12/1/2010	13S,283888E,4310965N	8.4
12046	M	3	12/1/2010	12S,729439E,4236264N	5.4
12047 (M32)	M	9-10	12/2/2010	13S,257722E,4239169N	1.0
12048	M	2	12/3/2010	13S,261946E,4241911N	7.6
12302	M	2.5	12/17/2010	13S,316520E,4228320N	5.1
12314	F	5	1/13/2011	13S,305193E,4247057N	1.4
12317	M	1.5	1/17/2011		2.8
12044 (F108)	F	1.5	11/29/2010	12S,752013E,4263883N	0.0
12041 (M55)	M	6-7	11/25/2010	13S,239181E,4248300N	0.0

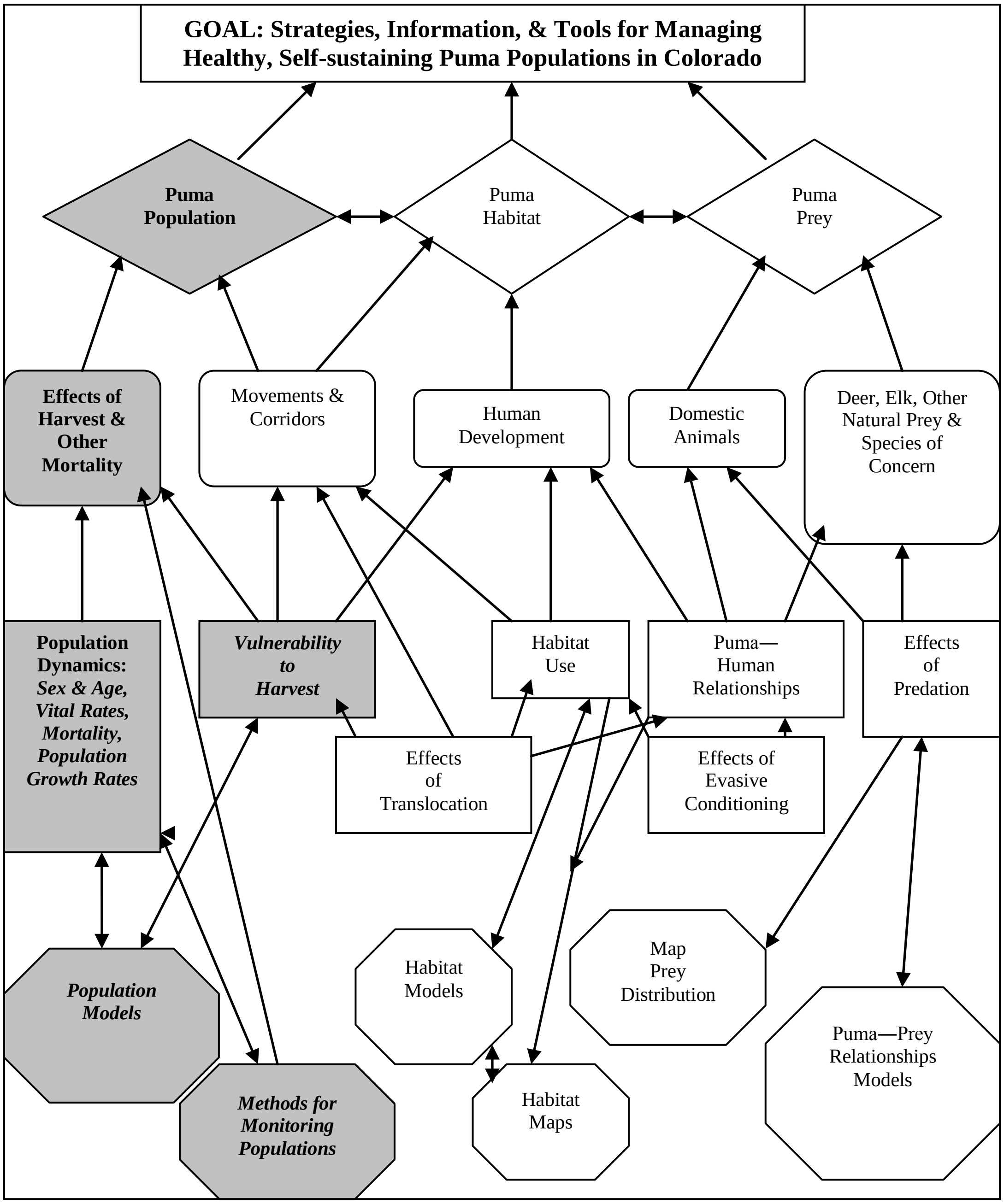


Figure. 1. An ecologically-based conceptual model of the Colorado Puma Research Program that provides the contextual framework for this and proposed puma research in Colorado. Gray-shaded shapes identify areas of research addressed by this puma research on the Uncompahgre Plateau for the puma management goal in Colorado (at top).

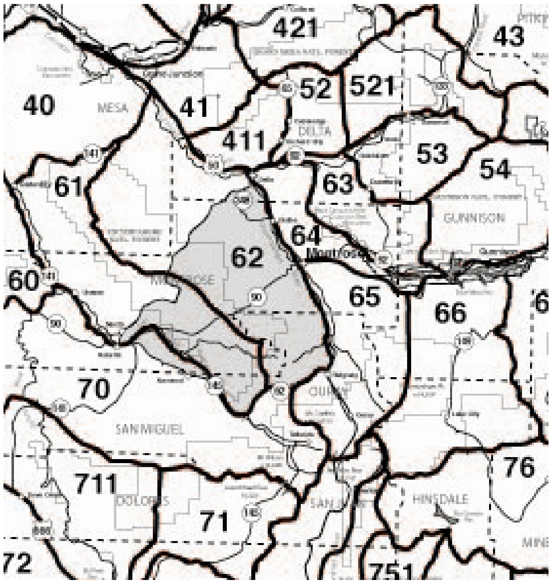


Figure 2. The puma study area on the southern half of the Uncompahgre Plateau, Colorado (shaded in gray) comprising the southern portions of Game Management Units (GMUs) 61 and 62 and a northern portion of GMU 70.

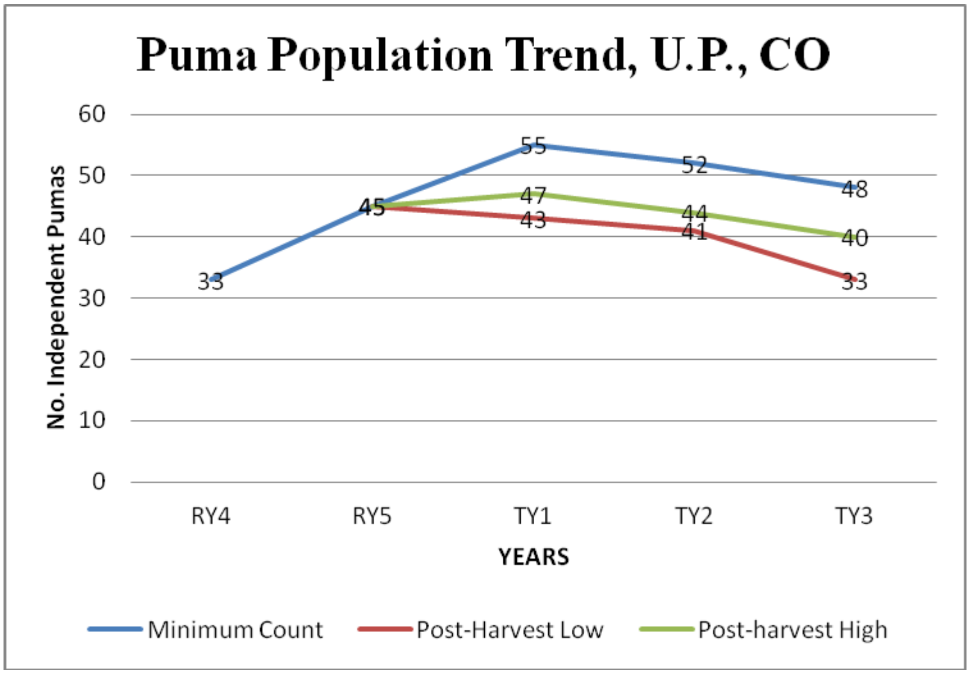


Figure 3. Trends in the population of independent pumas on the Uncompahgre Plateau Puma Study Area, including Reference Years 4 and 5 (RY4, RY5) and Treatment Years 1, 2, and 3 (TY1, TY2, TY3). Numbers represent minimum counts that include all pumas from known radio-collared pumas, visual observations of non-marked pumas, harvested non-marked pumas, and track counts of suspected non-marked pumas on the study area during fall to spring hunting and research capture seasons, except RY5 (45), which had to be modeled from RY4 observation data (33) because the state government hiring freeze that year affected search and capture efforts. The actual minimum count for RY5 was 37 independent pumas. The quota of 8 pumas for TY1 represented a 15% harvest of the model projected 53

independent pumas expected in TY1 and was used to set the quota ahead of the hunting season. Starting in TY1, two capture teams were deployed to count pumas on the study area because the hunting season shortened our fall-winter-spring research period. We deployed a team on each the east and west sides of the study area. The minimum count for TY1 was actually 55 independent pumas, consistent with the model expected 53.

Post-harvest high trend line represents the population of independent pumas after pumas harvested only on the study area by hunters. This trend line represents 14.5% to 16.7% harvest of independent pumas.

Post-harvest low trend line represents the population of independent pumas after pumas harvested on the study area and pumas harvested when they ranged onto adjacent GMUs open to hunting and other mortalities are subtracted from the minimum count. TY1 post-harvest low includes 1 adult female and 3 adult males killed off the study area. The TY2 post- harvest low includes 1 adult male killed off the study area and 2 adult female pumas killed in February 2011 on the study area to protect livestock. The TY3 post-harvest low includes 1 adult female and 4 adult males harvested off the study area and 2 adult females that died of natural causes on the study area. This trend line represents 21.2% to 31.2% harvest of independent pumas.

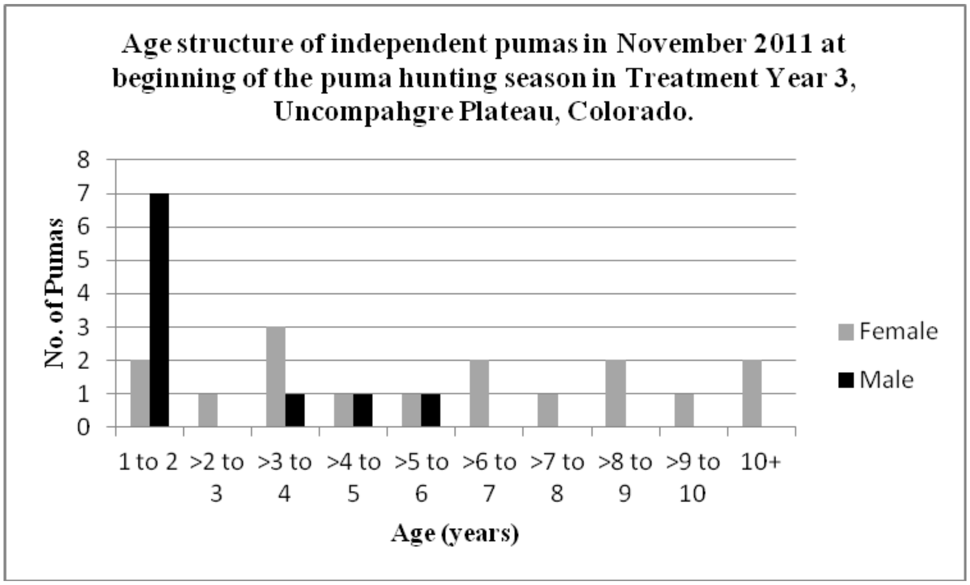


Figure 4. Estimated age structure of independent pumas in November 2011 at the beginning of the puma hunting season in Treatment Year 3 (TY3) on the Uncompahgre Plateau study area, Colorado. All these pumas were captured and sampled by researchers or harvested by hunters and examined by researchers. Mean ± *SD* of female and male ages, respectively: 5.85 ± 3.05 yr. (70.19 ± 36.57 mo.), *n* = 16; 2.25 ± 1.58 yr. (27.00 ± 18.95 mo.), *n* = 10.

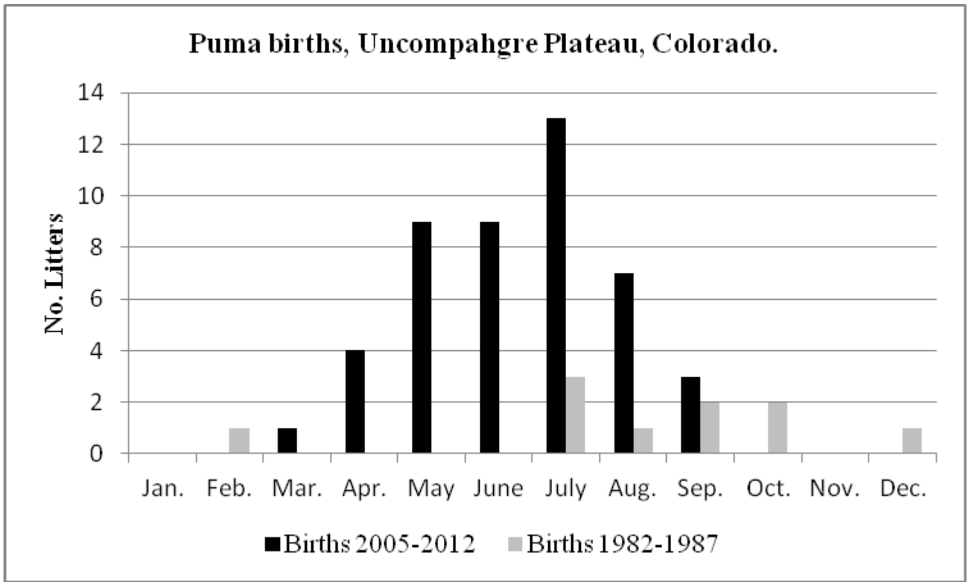


Figure 5. Puma births (black bars) detected by month from May 19, 2005 to July 5, 2012 (*n* = 46 litters of 24 females; 44 of the litters were examined at nurseries when cubs were 26-42 days old and 2 litters confirmed by tracks of ≥1 cubs following GPS-collared mothers F28 and F111 when cubs were ≤42 days old). Also shown (gray bars) are results of the earlier effort by Anderson et al. (1992:48; 1982 to 1987, *n* = 10 litters of 8 females, examined when cubs were <1 to 8 months old), Uncompahgre Plateau, Colorado.

Appendix A. Summary of individual puma cub survival and mortality, 2005 to 2012, Uncompahgre Plateau, Colorado.

Puma I.D.	Estimated Age at capture (days)	Est. Birth date	Est. survival span from 1st capture to fate or last monitor date	Age to last monitor date alive or at death (days, birth to fate)	Status: Alive/Survived to subadult stage/ Lost contact/Disappeared/ Dead; Cause of death	Mother I.D.
M5	183	~8-1-04	02-04-05 to 04-07-08	~1,664	Radio-collared. Survived to subadult stage by 09-16-05; independent at ~13 mo. old. Dispersed from natal area by 09-29-05 at 14 mo. old. Established territory on NW U.P. Killed by hunter in Beaver Creek, UT 02-20-09 at 54.6 months old.	F3
F9	31	5-28-05	06-27-05 to 4-19-06	326-333	Radio-collared. Shed radiocollar 04-19-06 to 04-26-06.	F2
F10	31	5-28-05	06-27-05 to 11-20-05—12-29-05	176-215	Radio-collared. Dhed radiocollar 08-10-05; last tracks of F10 with mother F2 & siblings F9 & M11 observed 11-20-05. F10 disappeared by 12-30-05.	F2
M11	31	5-28-05	06-27-05 to 12-2-07	918	Radio-collared. Shed collar 10-24 to 11-08-05. Recollared on 04-02-06. Survived to subadult stage by 06-21-06, independent at 13 mo. old. Dispersed from natal area by 07-11-06 at 14 mo. old. Killed by a hunter in SW CO 12-2-07 at 918 days (30 mo.) old.	F2
F12	42	5-19-05	07-01-05 to 12-08-05—01-26-06	203-252	Radio-collared. Shed radiocollar 07-28-05—08-01-05. Tracks of F12 found in association with mother F7 on 12-08-05. F12 disappeared by 01-27-06 when she was not visually observed with F7, and her tracks were not seen in association with F7's tracks.	F7
F13	42	5-19-05	07-01-05 to 08-28-05	101	Radio-collared. Killed and eaten by a puma possibly M5 (13 mo. old) about 08-28-05.	F7
F14	26	6-26-05	07-22-05 to 02-07-06—03-10-06	226-257	Radio-collared. Shed radiocollar 01-20-06 to 01-25-06. Tracks of F14 were observed with tracks of mother F8 & sibling M15 on 02-07-06. Disappeared by 03-11-06, only tracks of F8 & M15 were found.	F8
M15	26	6-26-05	07-22-05 to 06-06 to 14-06	345-353	Radio-collared. Shed radiocollar 06-06-06 to 06-14-06.	F8
F17	34	9-22-05	10-26-05 to 08-18-06	330	Radio-collared. Shed radiocollar 06-06-06 to 06-14-06. Killed by a car on highway 550 on 08-18-06. Probably dependent on F16. Died at 10.8 months old	F16
F18	34	9-22-05	10-26-05 to 07-20 to 27-06	301-308	Radio-collared. Probably killed by another puma. Multiple bite wounds to skull. Died at 10 months old.	F16
M19	34	9-22-05	10-26-05 to 07-27 to 08-02-06	308-314	Radio-collared. Shed radiocollar 07-27-06 to 08-02-06.	F16
M20	34	9-22-05	10-26-05 to 05-24-06	244-245	Radio-collared. Shed radiocollar 05-24-06—05-25-06.	F16
F21	37	9-26-05	11-02-05 to 08-16-06	324	Radio-collared. Lost contact; radiocollar quit. Last aerial location 8-16-06, live signal.	F3

Appendix A continued

Puma I.D.	Estimated Age at capture (days)	Est. Birth date	Est. survival span from 1st capture to fate or last monitor date	Age to last monitor date alive or at death (days, birth to fate)	Status: Alive/Survived to subadult stage/ Lost contact/Disappeared/ Dead; Cause of death	Mother I.D.
M22	37	9-26-05	11-02-05 to 12-21-05—12-22-05	86-87	Radio-collared. Killed and eaten by male puma 12-21-05 to 12-22-05.	F3
M26	183	8-1-05	02-08-06 to 03-21 to 24-06	~232-235	Radio-collared. Shed radiocollar 03-21-06 to 03-24-06.	F25
F33	31	5-30-06	06-30-06 to 07-31-06	63-65	Radio-collared. Probably killed and eaten by a male puma 08-01 to 03-06. GPS data on M29 indicate he was *not* involved.	F23
F34	31	5-30-06	06-30-06 to 07-31-06	63-65	Radio-collared. Probably killed and eaten by a male puma 08-01 to 03-06. GPS data on M29 indicate he was *not* involved.	F23
F35	31	5-30-06	06-30-06 to 07-07-06	38	Dead; research-related fatality.[a]	F23
F36	29	6-9-06	07-08-06 to 07-28-06	74	Radio-collared. Killed and eaten by a male puma 08-22-06. GPS data on M29 indicate he was *not* involved.	F28
M37	29	6-9-06	07-08-06 to 07-28-06	74	Radio-collared. Killed and eaten by a male puma 08-22-06. GPS data on M29 indicate he was *not* involved.	F28
M38	41	7-29-06	09-08-06 to 07-16 to 17-07	352-353 1623	Radio-collared. Shed radiocollar found 03-06-07. Photo (trail camera in McKenzie Cr.) of M38 & Unm. F sibling with F2 on 07-16 to 17-07 at 352-353 days old. Killed by puma hunter 01-07-11 in GMU40 Ladder Creek when he was 53.2 months old.	F2
M39	29	8-13-06	09-11-06 to 09-20-06 to 04-25-07	9 255 1307	Radio-collared. Shed radiocollar by 09-20-06, but seen alive on that date. Tracks of 2 cubs following F8 on 04-25-07. Survived to adult stage; dispersed from natal area. Killed by a puma hunter 03-12-10 in GMU 40 when 42.8 months old.	F8
F40	29	8-13-06	09-11-06 to 09-20-06 to 04-25-07	9 255	Radio-collared. Shed radiocollar by 09-20-06, but seen alive on that date. Tracks of 2 cubs following F8 on 04-25-07.	F8
F41	29	8-13-06	09-11-06 to 10-05-06	53-61	Radio-collared. Assumed dead. Shed radiocollar or died (blood on collar) between 10-05-06 (last live signal) & 10-13-06 (collar found).	F8
M42	29	8-13-06	09-11-06 to 11-27-06	106	Dead; research-related fatality.[b]	F8
M43	33	8-13-06	09-15-06 03-01-07	200 899	Radio-collared. Shed radiocollar by 11-7 to 17-06. Killed by a puma hunter 01-28-09 in Deer Creek, west slope of Grand Mesa, CO GMU41 at 29.5 months old.	F7

Appendix A continued

Puma I.D.	Estimated Age at capture (days)	Est. Birth date	Est. survival span from 1st capture to fate or last monitor date	Age to last monitor date alive or at death (days, birth to fate)	Status: Alive/Survived to subadult stage/ Lost contact/Disappeared/ Dead; Cause of death	Mother I.D.
M44	33	8-13-06	09-15-06 to 02-14-07	479	Radio-collared. Shed radiocollar by 10-27-06. Treed, visually observed 02-14-07; sibling (?) M56 also captured, sampled, & marked for 1st time. M44 killed by Wildlife Services for depredation control on 12-05-07, for killing 4 domestic sheep. He was still dependent on F7. He was 15.7 months old.	F7
F45	33	8-13-06	09-15-06 to 5-20 to 23-07	280-283	Radio-collared. Multiple puncture wounds on braincase—parietal & occipital regions; consistent with bites from coyote. F45 switched families, moving from F7 to F2 about 12-19 to 20-06. Last date F45 was with F2 was 04-17-07. Died 05-20 to 23-07 when she was 9.2 months old.	F7
M46	31	9-17-06	10-18-06 to 12-15-06	89 360	Radio-collared. Shed collar by 12-14-06. Tracks of all cubs observed following F3 12-15-06. Tracks & GPS data indicated that F3 apparently with ≥1 of her male cubs (M46, M47, M48) at 360 days old on 09-12-07 in Puma Canyon.	F3
M47	31	9-17-06	10-18-06 to 12-15-06 to 09-12-07	89 360	Radio-collared. Shed collar . Tracks of all cubs observed following F3 12-15-06. Tracks & GPS data indicated that F3 apparently with ≥1 of her male cubs (M46, M47, M48) at 360 days old on 09-12-07 in Puma Canyon.	F3
M48	31	9-17-06	10-18-06 to 12-15-06 to 09-12-07 to 12-27-09	89 360 1187	Radio-collared. Shed radiocollar. Tracks of all cubs observed following F3 12-15-06. Tracks & GPS data indicated that F3 apparently with ≥1 of her male cubs (M46, M47, M48) at 360 days old on 09-12-07 in Puma Canyon. Survived to adult stage; dispersed from natal area. Killed by a puma hunter 12-27-09 in GMU 61N when 38.9 months old.	F3
M49	153	7-1-06	12-05-06 to 07-31-07 to 01-01-07	~456	Radio-collared. M49 was orphaned when his mother died on about 03-26-07; he was ~268 days old. M49 dispersed from natal area and onto NE slope of U.P. Shed radiocollar at a yearling cow elk kill about 10-01-07; he was ~428 days old. Killed by a puma hunter in Blue Creek, northwest Uncompahgre Plateau (GMU 61N) 01-24-09 when ~29 months old.	F50

Appendix A continued

Puma I.D.	Estimated Age at capture (days)	Est. Birth date	Est. survival span from 1st capture to fate or last monitor date	Age to last monitor date alive or at death (days, birth to fate)	Status: Alive/Survived to subadult stage/ Lost contact/Disappeared/ Dead; Cause of death	Mother I.D.
F53	183	7-1-06	01-12-07 to 02-23-07 to 09-02-07	42 ~428 subad.	Radio-collared. Shed radiocollar 02-23-07. F53 visually observed by P. & F. Star (Loghill Mesa), on 09-02-07, when F53 was ~14 months old and an independent subadult.	F54
M56[c]	183	~8-13-06	02-14-07 to 03-01-07	200	Radio-collared. Shed radiocollar 2-27-07. M56 observed 03-01-07.	F7 (?)
F57	35	4-16-07	05-21-07 to 06-06-07	52	Radio-collared. Shed radiocollar 06-07-07. Live mode 06-06-07.	F25
M58	34	5-24-07	06-27-07	324 434	Not radio-collared. Tracks of 3 cubs observed with F16's tracks on 04-12-08, McKenzie Butte-Pinon Ridge Pass. 3 cubs observed with F16 on 08-08-08 by B. & T. Traegde. Survived to adult stage; dispersed from natal area. Killed by a puma hunter 12-27-09 in GMU 521 when 31 months old.	F16
F59	34	5-24-07	06-27-07 to 08-21-07	55 324 434	Radio-collared. Shed collar about 02-14-08. Observed with 11-20-07 with F16, but without siblings M58 and F61. Tracks of 3 cubs observed with F16's tracks on 04-12-08, McKenzie Butte-Pinon Ridge Pass. Three cubs observed with F16 on 08-08-08 by B. & T. Traegde.	F16
M60	34	5-24-07	06-27-07 to 07-11 to 12-07	48-49	Dead; research-related mortality.[d]	F16
F61	34	5-24-07	06-27-07 to 06-29-07	324 434 538	Radio-collared. Radiocollar malfunction. Tracks of 3 cubs observed with F16's tracks on 04-12-08, McKenzie Butte-Pinon Ridge Pass. 3 cubs observed with F16 on 08-08-08 by B. & T. Traegde. Dead. Died probably as independent subadult at 538 days old; struck by car on Hwy 550 mi. marker 111 N. of Ridgway, CO, euthanized by gunshot on 11-13-08.	F16
M62	34	7-14-07	08-17-07		Not radio-collared.	F24
M63	34	7-14-07	08-17-07 to 01-01-11	1267	Not radio-collared. Dispersed from study area. Killed by a puma hunter 01-01-11 in Calamity Creek, GMU61N when he was 41.5 months old.	F24
M64	34	7-14-07	08-17-07	262	Not radio-collared. Two out of potential of 4 of F24's male cubs were visually observed with her on 4/1/08. Assume that 2 male cubs died before the age of 8.5 mo. Eartags were seen on both cubs, but the numbers were not.	F24

Appendix A continued

Puma I.D.	Estimated Age at capture (days)	Est. Birth date	Est. survival span from 1st capture to fate or last monitor date	Age to last monitor date alive or at death (days, birth to fate)	Status: Alive/Survived to subadult stage/ Lost contact/Disappeared/ Dead; Cause of death	Mother I.D.
M65	34	7-14-07	08-17-07 to 11-07-09	262 847	Not radio-collared. Two out of potential of 4 of F24's male cubs were visually observed with her on 4/1/08. Assume that 2 male cubs died before the age of 8.5 mo. Eartags were seen on both cubs, but the numbers were not. Survived to adult stage; dispersed from natal area. Killed by Wildlife Services for depredation control on 11-07-09 when 27.8 months old.	F24
F66	37	7-17-07	08-23-07 to 11-05-07	111 681	Radio-collared. Lost contact; last location 11/5/07. No signals after that date. F66 was photographed with one male sibling, either M67 or M68, & F30 on 5/31-6/1/08. F66 was recaptured and radio-collared as a subadult on 11/25/08. She died from massive trauma & bleeding of internal organs possibly resulting from being trampled by an elk or mule deer on about 05-28-09 as an independent subadult 23 months old.	F30
M67	37	7-17-07	08-23-07 to 12-18-11	1615	Not radio-collared. M67 or M68 was photographed with sibling F66 & mother F30 on 5/31-6/1/08. Dispersed from natal area. Established adult home range on west side of Uncompahgre Plateau. Killed by puma hunter in GMU61N on 12-18-11 when 52.9 months old.	F30
M68	37	7-17-07	08-23-07 to 12-30-08	532	Not radio-collared. M67 or M68 was photographed with sibling F66 & mother F30 on 05-31 to 06-01-08. Survived to subadult stage; dispersed from natal area. Killed by a puma hunter in Disappointment Valley, CO (GMU 71) 12-30-08 at 17.5 months old.	F30
F74	259	6-1-07	03-12-08 to 07-09-08	403	Radio-collared. Shed radiocollar between 7-9-08 and 7-15-08, probably while still dependent on mother F75.	F75
M76	30	5-19-08	06-18-08	~87	Not radio-collared. Probably dead; if not killed when sibling M79 was killed, then probably would starve to death.	F2
M77	30	5-19-08	06-18-08	~87	Not radio-collared. Probably dead; if not killed when sibling M79 was killed, then probably would starve to death.	F2
F78	30	5-19-08	06-18-08	~87	Not radio-collared. Probably dead; if not killed when sibling M79 was killed, then probably would starve to death.	F2

Appendix A continued

Puma I.D.	Estimated Age at capture (days)	Est. Birth date	Est. survival span from 1st capture to fate or last monitor date	Age to last monitor date alive or at death (days, birth to fate)	Status: Alive/Survived to subadult stage/ Lost contact/Disappeared/ Dead; Cause of death	Mother I.D.
M79	30	5-19-08	06-18-08	87	Not radio-collared. Dead. Chewed-off anterior portions of the nasals, maxilla, palate, dentaries, and pieces of the braincase, with 6 or 9 portion of yellow ear-tag and intestines and bits of skin found ~45 m from mother F2's death site on 08/14/08. Cub death probably due to puma-caused infanticide with cannibalism at ~87 days old. Male puma scrapes, about 8, under a rock rim ~50m distance from cub remains, and made ~ time of pumas' deaths.	F2
F80	40	5-23-08	07-02-08		Not radio-collared. Apparently died before 02-04-09; no tracks found in association with F23 & siblings F81 & F97.	F23
F81	40	5-23-08	07-02-08 to 07-29-09	424	Radio-collared. Last live location 7-29-09.	F23
F97	257	5-23-08	02-04-09 to 01-22-12	1339	Radio-collared. Lost contact after 05-12-09; shed collar at elk kill cache on Mailbox Park. Dispersed from study area. Killed by a puma hunter 01-22-12 in Dry Creek when 43.9 months old.	F23
M82	37	5-29-08	07-05-08 to 12-10-09	560	Radio-collared. Survived to subadult stage; dispersed from natal area. Killed by a puma hunter in 12-10-09 GMU 65 when 18.4 months old.	F8
M83	37	5-29-08	07-05-08 to 01-18-11	964	Not radio-collared. Survived; dispersed from study area. Killed by a puma hunter 01-18-11 on Glade Park, GMU40. He was 31.6 months old.	F8
M84	36	6-5-08	07-11-08 to 02-11-09	251	Radio-collared 7-11-08 to 7-22-08; collar removed because of malfunction. Not radio-collared after 7-22-08. Eartag of M84 was found by E. Phillips on 8-25-08 when mother F70's GPS locations located her on either side of the eartag in the East fork Dolores Cyn. M84 recaptured radiocollared again 1-29-09 in Dolores Cyn. in association with F70 & F96's family. Shed radiocollar again about 02-14-09.	F70
F85	36	6-5-08	07-11-08 to 10-01-08	118	Radio-collared. Dead. Probably died of predation or infanticide about 10-1-08 near elk calf kill at age 3.9 months.	F70

Appendix A continued

Puma I.D.	Estimated Age at capture (days)	Est. Birth date	Est. survival span from 1st capture to fate or last monitor date	Age to last monitor date alive or at death (days, birth to fate)	Status: Alive/Survived to subadult stage/ Lost contact/Disappeared/ Dead; Cause of death	Mother I.D.
F86	36	6-5-08	07-11-08 to 07-23 to 08-03-08	~48-59	Radio-collared 7-22-08. Dead. Radio-collar, orange ear-tag #86 with pinna with green tattoo #86 found by J. Timmer 9-1-08. F86 died ~7-23 to 8-3-08 when mother F70's GPS locations located her at F86 remains. Probable predation.	F70
M87	28	7-3-08	07-31-08 to 12-06-11	1251	Not radio-collared. Dispersed from natal area. Recaptured as adult on west slope of study area on 02-09-11. Alive as of 07-31-11. Killed by puma hunter on 12-06-11 at 41 months old north of the study area.	F3
M88	28	7-3-08	07-31-08 to 11-30-10	880	Not radio-collared. Dispersed. Killed by a puma hunter in Disappointment Valley, GMU711 on 11-30-10 when 28.9 months old.	F3
F89	28	7-3-08	07-31-08		Radio-collared.	F3
M90	36	7-9-08	08-14-08	867	Radio-collared. Recaptured as young adult on study area, adjacent to natal area, on 11-16-10. Killed by a puma hunter during TY2 on 11-23-10.	F72
Male 7A	28-35	7-10-08	~08-07-08 to 08-14-08	28 to 35	Not radio-collared. F7's cubs died from starvation after they were orphaned. F7 was shot on 8-3-08 for killing domestic sheep.	F7
Male 7B	28-35	7-10-08	~08-07-08 to 08-14-08	28 to 35	Not radio-collared. F7's cubs died from starvation after they were orphaned. F7 shot on 8-3-08 for killing domestic sheep.	F7
Female 7C	28-35	7-10-08	~08-07-08 to 08-14-08	28 to 35	Not radio-collared. F7's cubs died of starvation after orphaned. F7 shot on 8-3-08 for killing domestic sheep.	F7
M91	35	8-19-08	09-29-08	455	Radio-collared. Killed by a puma hunter on study area during TY1 as dependent cub on 11-17-09 at age 14.9 months.	F25
M92	35	8-19-08	09-29-08	976	Radio-collared. Lost contact after 12-12-08. Dispersed from natal area. Recaptured in McKenzie Creek, west slope of study area on 04-22-11 when 32 months old.	F25
F95	16 mo.	June-07	12-29-08		Radio-collared. Survived to adult stage. Established adult home range overlapping mother F93's home range. To date, July 2012, F95's home range mainly adjacent to N side of natal area.	F93
F98	4-5 mo.	Sep-Oct-08	02-12-09 to 03-08-09	146-176	Radio-collared. Died; probably killed by male puma (infanticide).	Unm.F

Appendix A continued

Puma I.D.	Estimated Age at capture (days)	Est. Birth date	Est. survival span from 1st capture to fate or last monitor date	Age to last monitor date alive or at death (days, birth to fate)	Status: Alive/Survived to subadult stage/ Lost contact/Disappeared/ Dead; Cause of death	Mother I.D.
M99	5 mo.	Sep-Oct-08	2-27-09 to 01-2010	488	Radio-collared. Last location 4-22-09 on Paterson Mt. Died as 16-month old subadult in San Miguel Canyon. Probably killed by another puma.	Unm.F
M101	35	4-15-09	05-20-09 to 09-19-09	157	Radio-collared. Died; killed by puma M55 after he was orphaned due to death of mother F16 by vehicle strike.	F16
M102	35	4-15-09	05-20-09		Radio-collared. Lost contact after 09-04-09. Did not find evidence of M102 associated with deaths of siblings M101 and F103. But M102 probably died.	F16
F103	35	4-15-09	05-20-09 to 09-17-09	159	Radio-collared. Died; killed by puma M55 after she was orphaned due to death of mother F16 by vehicle strike.	F16
M105	38	5-7-09	06-14-09 to 02-09-10	278	Radio-collared. Lost contact after 02-09-10 due to shed collar.	F75
F106	38	5-7-09	06-14-09 to 02-27-11	275 661	Not radio-collared at nursery; F75 returned to nursery during handling. Radio-collared later on 2-10-10. Lost contact due to shed collar 3-16 to 29-10. F106 dispersed from natal area and was photographed at 21 months old at camera and scent-rub station on east slope of Uncompahgre Plateau on 02-27-11.	F75
M107	34	5-25-09	06-28-09 to 02-24-10	241	Not radio-collared; too small. Recaptured 2-24-10; not collared.	F94
F108	34	5-25-09	06-28-09 to 03-05-10	553	Shed radiocollar at nursery; fastener failed. Recaptured and re-collared 2-24-10. Shed collar ~3-5-10. Dispersed from natal area. Killed by a puma hunter on the study area during TY2 on 11-29-11 at 18.1 months old.	F94
M109	34	5-25-09	06-28-09		Not radio-collared; too small.	F94
M112	145	8-31-09	05-04-10	528 595	Radio-collared. Lost contact after 5-4-10 (last live signal) possibly due to failed transmitter. Recaptured and re-radio-collared on 01-24-11. Independent subadult during 02-10-11 to 04-18-11. Lost contact after 04-18-11; he may have dispersed or radiocollar quit.	F70
M115	14 mo.	Nov.-08	07-21-10	610	Radio-collared. M115 died as a subadult (~20 mo. old) due to complications of a broken left foreleg (natural cause).	F28
M117	6 mo.	Aug.-09	02-05-10 to 01-01-11	518	Radio-collared. Lost contact after 5-14-10 (last live signal); shed collar found on 7-15-10 in the natal area. Killed by a puma hunter on the natal area in Beaver Creek, off the U.P. study area on 01-01-11 when he was 17 months old.	F119
P1016(M)	39	6-12-10	06-12-10 to 07-21-10	39	Not radio-collared. Monitored at nursery via mother's GPS/VHF collar. Found dead at nursery due to infanticide by puma M32 on same day as our investigation of nursery.	F72

Appendix A continued

Puma I.D.	Estimated Age at capture (days)	Est. Birth date	Est. survival span from 1st capture to fate or last monitor date	Age to last monitor date alive or at death (days, birth to fate)	Status: Alive/Survived to subadult stage/ Lost contact/Disappeared/ Dead; Cause of death	Mother I.D.
P1017(M)	39	6-12-10	06-12-10 to 07-21-10	39	Not radio-collared. Monitored at nursery via mother's GPS/VHF collar. Found dead at nursery due to infanticide by puma M32 on same day as our investigation of nursery.	F72
M120	30	6-28-10	07-28-10 to 12-02-10	526	Radio-collared. Lost radio contact after 12-02-10. Killed by a puma hunter on his natal area on 12-06-11 when he was 17.2 months old.	F3
M121	30	6-28-10	07-28-10 to 03-28-11	273	Radio-collared. Lost radio contact after 03-28-11.	F3
M122	35	7-8-10	08-12-10 to 04-28-11	274	Radio-collared. Lost radio contact after 04-28-11. Tracks of 2 other siblings of M122 observed on 01-11-11 (neither cub marked).	F104
F123	29	7-15-10	08-13-10 to 02-17-11	217	Radio-collared. Killed on 02-17-11 for depredation control on domestic elk by Wildlife Services agent.	F94
F124	29	7-15-10	08-13-10 to 02-16-11	216	Radio-collared. Killed on 02-16-11 for depredation control on domestic elk by elk farm manager.	F94
M125	29	7-15-10	08-13-10 to 02-01-11	201	Radio-collared. Killed on 02-01-11 for depredation control on domestic elk by Wildlife Services agent.	F94
M126	28	08-08-10	09-05-10 to 01-08-12	221	Radio-collared. Lost radio contact after 03-17-11; shed his radiocollar at a mule deer cache. Dispersed from natal area. Killed by a puma hunter on 01-08-12 in Tuttle Draw WNW of Nucla, CO as 17-month-old subadult.	F118
M127	28	08-08-10	09-05-10 to 09-10-11	398	Radio-collared. Lost radio contact after 07-01-11; shed his radiocollar about 07-01-11. Found dead 09-14-11 on natal area; killed by another puma on about 09-10-11 at age 13 months.	F118
M128	28	08-08-10	09-05-10 to 02-22-11	198	Radio-collared. Lost radio contact after 02-22-11; radiocollar probably quit.	F118
F129	35	08-21-10	09-25-10 to 02-02-12	530	Radio-collared. Fate unknown. Transmitter on mortality mode on 04-28-11. Unable to get to collar until 06-23-11 due to high spring run-off, by then the transmitter had quit. Survived to recapture on 02-02-12 at 17.4 months old, with sibling M131; neither handled due to dangerous trees.	F96
M130	35	08-21-10	09-25-10 to 02-02-12	530	Radio-collared. Died of natural causes associated with injury to right shoulder during first move away from nursery about 10-23-10.	F96
M131	35	08-21-10	09-25-10 to 07-21-11	334	Radio-collared. Lost contact after 07-21-11. Shed his radiocollar about 07-27-11. Survived to recapture on 02-02-12 at 17.4 months old, with sibling F129; neither handled due to dangerous trees.	F96

Appendix A continued

Puma I.D.	Estimated Age at capture (days)	Est. Birth date	Est. survival span from 1st capture to fate or last monitor date	Age to last monitor date alive or at death (days, birth to fate)	Status: Alive/Survived to subadult stage/ Lost contact/Disappeared/ Dead; Cause of death	Mother I.D.
F132	35	08-21-10	09-25-10	35	Not radio-collared. Too small for collar design. Fate unknown. Apparently died; not with F96 and siblings F129 and M130 on 02-02-12.	F96
M134	~18 mo.	~June-09	12-14-10 to 06-10-11	731	Radiocollared as dependent large cub. Independent by about 03-28-11. Dead; killed for depredation control by Wildlife Services agent on 06-10-11.	Unm. F
M139	36	04-18-11	05-24-11 to 07-29-11	102	Radio-collared. Dead of infanticide and cannibalism along with sibling F148; killed and eaten by female or subadult male puma about 07-29-11.	F8
F148	36	04-18-11	05-24-11 to 07-29-11	102	Radio-collared. Dead of infanticide and cannibalism along with sibling M139; killed and eaten by female or subadult male puma about 07-29-11.	F8
F140	~5 mo.	~Aug.-10	01-02-11 to 04-18-11	258	Radio-collared. Lost contact. Shed first collar about 01-24-11. Recaptured and re-collared on 04-01-11. Shed second collar after 04-18-11. Recaptured and re-collared 01-12-12 as 17-month-old subadult on natal range.	Unk./ F28?
M141	~5 mo.	~Aug.-10	01-02-11 to 04-01-11	241	Radio-collared. Lost contact; shed radiocollar about 03-29-11. Recaptured, but could not be handled safely on 04-01-11.	Unk./ F28?
M142	~5 mo.	~Aug.-10	01-02-11 to 04-18-11	258	Radio-collared. Lost contact after 04-18-11 due to shed collar.	Unk./ F28?
P1030	~ 6 mo.	~Aug.-10	02-16-11	183	Struck by vehicle and killed on state highway 62 in Leopard Creek, south boundary of study area on 02-16-11.	Unk.
F147	~7 mo.	~Sep.-10	04-21-11 to 07-31-11	315	Radio-collared. Orphaned at about 12 months old when her mother F24 was killed by a male puma on 09-16-11. She ranged in her natal area until her radiocollar quit after 04-12-12.	F24
F149	45	04-22-11	06-06-11 to 07-16-12	451	Radio-collared. F149 (sibling of M161) was orphaned when her mother F23 was killed by a male puma on 06-06-12; she was 411 days (13.5 mo.) old. F149 dispersed from the natal area by 07-16-12 when she was 14.8 months old.	F23
M150	525	08-31-09	02-07-11 to 04-11-11	588	Radio-collared. M151 was independent by 03-28-11 at 19 mo. old. He dispersed from the natal area by 04-11-11 at 19.5 mo. old. Contact lost after 04-11-11.	F70
M151	253	06-16-10	02-24-11 to 03-07-11	264	Radio-collared. Lost contact after 03-07-11 (GPS location of mother F111 at shed collar of M151).	F111

Appendix A continued

Puma I.D.	Estimated Age at capture (days)	Est. Birth date	Est. survival span from 1st capture to fate or last monitor date	Age to last monitor date alive or at death (days, birth to fate)	Status: Alive/Survived to subadult stage/ Lost contact/Disappeared/ Dead; Cause of death	Mother I.D.
F152	271	06-16-10	03-14-11 to 07-31-12	776	Radio-collared. Lost contact after 03-21-11; shed collar. Recaptured 01-18-12; fit with GPS collar at 19 months old; currently (July 31, 2012) a 25-month-old adult ranging on her natal area (philopatric).	F93
M154	42	07-06-11	08-16-11 to 09-21-11	77	Radio-collared. M154 probably died of starvation following natural death of his mother F135. Sibling M155 also died.	F135
M155	42	07-06-11	08-16-11 to 09-25-11	81	Radio-collared. M155 died of starvation following death of his mother F135. Sibling M154 also died.	F135
M156	43	07-08-11	08-20-11 to 09-05-11	56	Radio-collared. M156 shed the collar about 09-05-11. He was 59 days old.	F137
F157	40	08-18-11	09-27-11 to 01-15-12	150	F157 with sibling F158 died of starvation following death of his mother F70 due to hunter harvest on 12-22-11. Cubs died 24 days after their mother died. The cubs were 150 days old.	F70
F158	40	08-18-11	09-27-11 to 01-15-12	150	F158 with sibling F157 died of starvation following death of his mother F70 due to hunter harvest on 12-22-11. Cubs died 24 days after their mother died. The cubs were 150 days old.	F70
M159	40	08-18-11	09-27-11 to 12-01-11	105	M159 probably died about 12-01-11 when he was located with his family (F70, siblings F157, F158). He was not located with them on 12-12-11 and was not observed with them on 12-13-11. He was 105 days old on 12-01-11.	F70
M161	276	04-22-11	01-23-12 to 07-16-12	451	M161 (sibling of F149) was orphaned when his mother F23 was killed by a male puma on 06-06-12; he was 411 days (13.5 mo.) old. M161 dispersed from the natal area by 06-29-12 when he was 14 months old.	F23
M162	183	07-25-11	01-25-12 to 06-11-12	322	M162 probably was orphaned cub of non-marked adult female puma killed on Pinto Mesa 01-18-12. M162 died of starvation on 06-11-12 when he was 322 days (10.6 mo.) old.	
M170	137	08-29-11	01-13-12 to 03-12-12	199	M170 died about 03-15-12 of unknown natural cause. He was 199 days (6.5 mo.) old.	F171

Appendix A continued

Puma I.D.	Estimated Age at capture (days)	Est. Birth date	Est. survival span from 1st capture to fate or last monitor date	Age to last monitor date alive or at death (days, birth to fate)	Status: Alive/Survived to subadult stage/ Lost contact/Disappeared/ Dead; Cause of death	Mother I.D.
P1033	22	07-10-11	NA	22	Cub P1033 was offspring of F136. It died of predation, probably killed by a puma or black bear in the nursery when about 22 days old, before researchers could examine the entire litter to sample and mark the cubs.	F136

[a] Cub F35 probably starved between 06-30-06 & 07-07-06 after the transmitter on the expandable collar got in its mouth.

[b] Cub M42 died after being captured by dogs, probably from stress of capture associated with severe infection of laceration under right foreleg caused by expandable radiocollar.

[c] Cub M56 was captured in association with F7 and her cubs M43 and M44. He may have been missed at the nursery when M43 and M44 were initially sampled and marked.

[d] Cub M60 died probably of starvation. The expandable radiocollar was around the neck and right shoulder, probably restricted movement.

Colorado Division of Parks and Wildlife
July 2011 - June 2012

WILDLIFE RESEARCH REPORT

State of:	Colorado	:	Division of Parks and Wildlife
Cost Center:	3430	:	Mammals Research
Work Package:	3003	:	Predatory Mammals Conservation
Task No.:	2	:	Cougar Demographics and Human Interactions
		:	Along the Urban-Exurban Front Range of
		:	Colorado
Federal Aid Project No.	W-204-R1		

Period Covered: July 1, 2011 - June 30, 2012

Author: M.W. Alldredge

Personnel: M. Strauser, E. Newkirk, W. Moss, B. Kirby, P. Lundberg, E. Joyce, T. Eyk, J. Halseth, G. Coulombe, R. Platte, K. Blecha, K. Yeager, L. Nold, K. Griffin, D. Kilpatrick, M. Paulek, B. Karabensh, D. Wroe, M. Miller, F. Quartarone, M. Sirochman, L. Wolfe, J. Duetsch, C. Solohub, K. Cannon, J. Koehler, L. Rogstad, R. Dewalt, J. Murphy, D. Swanson, T. Schmidt, T. Howard, D. Freddy CPW; B. Posthumus, Jeffco Open Space; D. Hoerath, K. Grady, D. Morris, A. Hatfield Boulder County Open Space; H. Swanson, R. Hatfield, J. Reale Boulder Open Space and Mountain Parks; S. Oyler-McCance, USGS.

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

ABSTRACT

The use of telomeres as a method to determine the age structure of bear and cougar populations has continued to be examined. Further refinement of the age-to-length relationship for both species is warranted based on preliminary results. We have begun a Ph.D. project with the University of Wisconsin to examine telomeres in detail for bears. This project will also look at stable isotopes to examine foraging ecology and bear use of human food sources. We have also begun an M.S. project with the University of Wisconsin to examine stable isotopes for cougars relative to predation on domestic animals and cougar foraging ecology.

Our principal research objective is to assess cougar population ecology, prey use, movements, and interactions with humans along the urban-exurban Front Range of Colorado. This year capture efforts focused on re-collaring previously collared cougars, and capturing previously unmarked independent age cougars and cubs. In addition to recollaring cougars we collared 6 new independent age cougars. Mortality was very high over the year with 14 mortalities for independent age cougars (predominantly human related, unknown causes and vehicle collisions) (Table 1). Home-range patterns remained consistent to previous years. The effectiveness of aversive conditioning is still showing mixed results, which is likely a factor of the opportunistic nature of cougars using urban environments and a lack of habituation to them. Removing caches does appear to be effective to get cougars to leave urban areas. Cougar/human interactions were minimal this year compared with previous years. Relocation of cougars

as a management tool has had limited assessment, but given some success, still warrants further investigation. Mule deer are the predominant prey in cougar diets, although males will also utilize elk regularly.

WILDLIFE RESEARCH REPORT

COUGAR AND BEAR DEMOGRAPHICS AND HUMAN INTERACTIONS IN COLORADO

MATHEW W. ALLDREDGE

P.N. OBJECTIVE

1. To assess cougar (*Puma concolor*) population demographic rates, movements, habitat use, prey selectivity and human interactions along the urban-exurban Front Range of Colorado.
2. Develop methods for delineating population structure of cougars and black bears (*Ursus americanus*), assessing diet composition and estimating population densities of cougars for the state of Colorado.

SEGMENT OBJECTIVES

Section A: Telomeres and Stable Isotopes

1. Evaluate the potential to develop a model for estimating age of bears and cougars based on telomere length.
2. Determine diet composition of bears and cougars using stable isotopes.

Section B: Front Range cougars

3. Capture and mark independent age cougars and cubs to collect data to examine demographic rates for the urban cougar population.
4. Continued assessment of aversive conditioning techniques on cougars within urban/exurban areas, including use of hounds and shotgun-fired bean bags or rubber bullets.
5. Continue to assess relocation of cougars as a practical management tool.
6. Assess cougar predation rates and diet composition based on GPS cluster data.
7. Model movement data of cougars to understand how cougars are responding to environmental variables.
8. Develop non-invasive mark-recapture techniques to estimate cougar population size.

SECTION A: BEAR AND COUGAR TELOMERES AND STABLE ISOTOPES

BY M. ALLDREDGE

OVERVIEW

Understanding the age structure of a population is very useful to managers, especially for hunted populations. Age structure can provide indications about the appropriateness of current harvest levels, changes that may need to occur in harvest, and the general health of a population. Typical approaches involve estimating age structure based on sampling harvested animals and obtaining ages based on tooth wear and replacement characteristics or from analyzing tooth annuli. Recently a new approach has been developed for some species that estimates the age of animals based on examining the length of telomeres in relation to the age of the animals.

Telomeres are repetitive DNA sequences that cap the ends of eukaryotic chromosomes, whose nucleotide sequence $(T_2AG_3)_n$ is highly conserved across vertebrate species (Meyne et al. 1989). During each cell cycle telomeric repeats are lost because DNA polymerase is unable to completely replicate the 3' end of linear DNA (Watson 1972). Thus, telomeres progressively shorten with each cell division; past research has demonstrated age-related telomere attrition in a variety of laboratory and wild species and has correlated telomere length with individual age (e.g. Hausmann et al. 2003, Hemann and Greider 2000). Using real-time quantitative polymerase chain reaction (Q-PCR; Cawthon 2002), we have demonstrated the potential for quantifying telomere length for black bears of known-age in Colorado (Alldredge 2010).

Understanding diet composition and foraging ecology of bears is also useful to managers, especially in urban areas as bears continually interact with humans and human derived food sources. The dynamics of this interaction and the extent to which bears utilize human food sources is largely unknown. The use of stable isotope analysis is one approach to understanding the amount and timing of utilization of various food sources within a bear's diet. Examining different tissue types from bears can explain patterns of use for various food sources and will provide managers a better understanding of this problem at a population level.

We have initiated a graduate study with the University of Wisconsin and Wisconsin Department of Natural Resources to develop methods of identifying population age structure using telomeres and examining diet composition and foraging ecology using stable isotopes for bears. See attached study plan for a complete project overview and objectives (Appendix I).

During 2011 we collected blood, tissue, hair, and bone samples from 400 bears across the state. These bears were either nuisance bears or hunter harvested bears. Samples from these bears are being utilized for both the telomere and stable isotope components of this project. Preliminary assessments indicate high genetic quality from samples for use in the telomere work. Initial data from stable isotope analyses indicate significant variation in $\delta^{13}C$ and $\delta^{15}N$ (Figure 1) among bears which suggests that differentiation in diets based on stable isotope analysis will be possible.

Additional work has also begun in collaboration with the University of Wisconsin to further examine stable isotope techniques for bears and cougars. This work is specifically designed to look at diet composition of bears within specific temporal windows relevant to current management issues. Similarly, stable isotope analyses for cougars is focused on identifying cougar predation on specific species guilds, identifying the use of small prey items, and determining factors associated with differences in prey utilization. For a complete project description and objectives see the attached study plan (Appendix II).

As an initial step to investigate the utility of using stable isotopes to assess cougar diets we collected hair samples from prey species found at cougar kills. Additionally hair samples were collected from domestic animals (llamas, goats, cats, dogs, etc.) that could potentially be preyed on by cougars. Stable isotope analysis has been done on these prey items and initial findings suggest that examining prey by species guilds does result in significant differences in $\delta^{13}C$ and $\delta^{15}N$ content (Figure 2).

A final component of this project is to develop hair growth curves for black bears, assess how meat is assimilated into tissues (hair) and specifically examine proportional contributions of meat in bear diets in the state of Colorado (See Appendix II for further details). Part of this work will involve working with captive bears at the Wild Animal Sanctuary located in Keensburg, CO. This will involve adding Rhodamine B to their food once per month and pulling hair. The Rhodamine B will appear as a UV mark on the hair and provide a time stamp so that growth curves for hair can be developed. Diets will also be controlled so that assimilation of various diet components into the consumer's tissues can be assessed with known controls. Finally, hair samples will be collected from harvested bears across the state. These hairs will then be segmented into time periods based on the growth curves and examined for time specific diet composition.

SECTION B: FRONT RANGE COUGARS

BY M. ALLDREDGE

INTRODUCTION

We have continued the cougar/human interaction study on the Front Range of Colorado. Given that cougars currently coexist with humans within urban/exurban areas along Colorado's Front Range, varying levels of cougar-human interaction are inevitable. The CPW is charged with the management of cougars, with management options ranging from minimal cougar population management, to dealing only with direct cougar-human incidents, to attempted extermination of cougars along the human/cougar spatial interface. Neither inaction nor extermination represent practical options nor would the majority of the human population agree with these strategies. In the 2005 survey of public opinions and perceptions of cougar issues, 96% of the respondents agreed that it was important to know cougars exist in Colorado, and 93% thought it was important that they exist for future generations (CPW, unpublished data).

There is a growing voice from the public that CPW do more to mitigate potential conflicts, and the leadership of CPW has requested that research efforts be conducted to help minimize future human/cougar conflicts. In order to meet these goals CPW believes it is necessary to directly test management prescriptions in terms of desired cougar population and individual levels of response.

Long-term study objectives for the Front Range Cougar Research project involve directly testing management responses of cougars at various levels of human interaction, as well as collecting basic information about demographics, movement, habitat use, and prey selection. The Cougar Management Guidelines Working Group (CMGWG) (2005) recommended that part of determining the level of interaction or risk between cougars and humans is to evaluate cougar behavior on a spectrum from natural, to habituated, to overly familiar, to nuisance, to dangerous. The CMGWG (2005) clearly stated that there is no scientific evidence to indicate that cougar habituation to humans affects the risk of attack. As a continuation from the pilot study efforts, we have continued to assess the effectiveness of aversive conditioning as a method to alter interaction rates between cougars and humans. We also continue to monitor relocated cougars to determine the effectiveness of relocation as a management tool.

The use of GPS collars obtaining up to 8 locations per day also allows for a detailed examination of demographic rates. We are monitoring cougars that utilize natural habitats and cougars that use a mixture of natural and urban habitats. This allows for an assessment of demographic rates, movement patterns, and habitat use among cougars utilizing these two habitat configurations. We have also begun monitoring cubs (approximately 6 months of age or older), primarily to determine survival but potentially to understand movement patterns and dispersal.

The use of GPS collars also allows us to study predator-prey relationships and diet composition. GPS locations are divided into selection sets based on the likelihood of the set of locations (clusters) representing a kill site. A random sample of these clusters is investigated to determine what a cougar was doing at the site, and whether or not it represents a kill site. Kill sites are thoroughly investigated to determine as much information as possible about what was killed at the site.

STUDY AREA

The original pilot study was conducted in Boulder and Jefferson counties, in an area near Interstate 70 north to approximately Lyons, Colorado, which was also a likely area for addressing long-term research objectives (see Figure 3). The study area for the long term study includes this original area but was expanded south to highway 285. Research efforts in the additional southern portion are generally limited to capturing cougars that are in the urban setting and/or have interacted directly with humans. The study area is comprised of many land ownerships, including private, Boulder city, Boulder County, Jefferson County, and state and federally owned lands. Therefore, we have been directly involved with Boulder city and Boulder and Jefferson county governments to obtain agreements from these entities on conduct of research and protocols for dealing with potential human/cougar interactions prior to conducting any research efforts. We have also acquired permission to access numerous private properties to investigate cougar clusters and to trap cougars.

METHODS

Baiting, using deer and elk carcasses, has been conducted throughout the year, with a focus on areas that do not allow the use of hounds. Bait sites are monitored using digital trail cameras to determine bait site activity. Cage traps were generally used for capture when cougars removed the bait and cached it. Beginning in November, 2011 and continuing through February, 2012, hounds were also used when snow conditions were favorable to capture cougars. Snares were used in situations where hounds could not be used and cougars would not enter cage traps. Captured cougars were anesthetized, monitored for vital signs, aged, measured, and ear-tagged. All independent cougars (> 18 months old) were fitted with GPS collars. All cubs greater than 15 kg (approximately 6 months or older) were ear-tagged with 22 g ear-tag VHF transmitters or 22g ear-tag ptt Argos transmitters.

When cougars interact with humans and elicit a response from CPW District Wildlife Managers (DWMs) they are potential candidates for aversive conditioning. Most incidents prompting response from a DWM occur in neighborhoods, where relocating the cougar is necessary prior to any application of an aversive conditioning treatment. For these situations, all treatments require the relocation of the offending individual to an adjacent open-space property or similar area. In other situations a cougar can be directly conditioned or chased from the area without relocation. Initial data suggested aversive conditioning had mixed results. Here we compare cougar behavior between situations when the cougar is undetected in urban areas versus situations when they are detected and hazed or their kills are removed. Interactions have been limited so we have limited data to assess these activities.

Cougars are only relocated for management purposes, generally in conjunction with human conflict or livestock depredation. Research cougars that have been collared for other purposes of the study may also become part of the relocation group if their levels of human interaction warrant such a management action. Because only a few cougars are relocated each year, we collar and monitor all cougars that are relocated in the northeast region. Cougars are ear-tagged and fitted with a telemetry collar (VHF, or GPS collars may be used depending on the situation).

Release area is critical to the success of any relocation, however, suitable relocation areas may be difficult to find. Such an area must be far enough from the problem area, have suitable prey, and be remote enough so that the individual will not be presented with problem opportunities at or near the release site. Understanding the minimum release distance that has a reasonable chance for relocation success is useful for both logistical reasons and to increase the number of potential release sites.

We evaluated cougar diet composition by using GPS location data to identify likely kill sites. Characteristics of clusters of GPS locations representing cougar-killed ungulate sites (Anderson and Lindzey 2003, Logan 2005) were used to develop a standard algorithm to group GPS points together, to provide a sound sampling frame from which statistical inference could be made about clusters that are not physically investigated. GPS collars collected locations 7 to 8 times/day to reflect time periods when cougars are both active and inactive.

The clustering routine was designed to identify clusters in five unique selection sets (S_1, S_2,…, S_5) in order to identify clusters containing two or more points, those that contained missing GPS locations, and those that were represented by single points. S_1 clusters consist of multiple GPS locations with a 4 day window and within 200 m, while other sets are single points close together in time within varying distance bands. The clustering algorithm was written in Visual Basic and was designed to run within ARCGIS (Alldredge and Schuette, CDOW unpubl. data 2006). The widths of the spatial and temporal sampling windows were user specified, in order to meet multiple applications and research needs. This also enabled adjustment of the sampling frames to improve cluster specifications as needed.

We used the following protocol to investigate cougar GPS clusters in the field. For S_1 clusters, we investigated each cougar GPS location in the cluster by spiraling out a minimum of 20 m from the GPS waypoint while using the GPS unit as a guide, and visually inspecting overlapping view fields in the area for prey remains. Normally, this was sufficient to detect prey remains and other cougar sign (e.g., tracks, beds, toilets) associated with cougar. If prey remains were not detected within 20 m radius of the cluster waypoints, then we expanded our searches to a minimum of 50 m radius around each waypoint. For S_2 through S_5 clusters, we went to each cougar GPS location and spiraled out 50 m around each waypoint, while using the GPS unit as a guide. Depending on the number of locations, topography, and vegetation type and density, we spent a minimum of 1 hour and up to 3 hours per cluster to judge whether the cluster was a kill site.

Kevin Blecha is currently conducting his M.S. research on predator-prey dynamics related to the sampling described above. He is specifically looking at predator-prey relationships relative to various habitat types and levels of human density across the landscape. An assessment of prey availability or reliability is also being made through the use of camera traps within these habitat types and levels of human density. Finally, an assessment of cougar use on domestic animals (livestock and pets) is being made (see Appendix III for more details).

Joe Halseth has also initiated a study to examine prey selection and kill site dynamics with regard to conspecifics and scavenging. Kill sites are being investigated within 24 hours of the kill to determine prey species, to place cameras and to sample ungulates for age and to test for CWD. Some work has indicated that cougars may select for CWD positive animals but sample sizes have been limited. We intend to sample a large number of ungulates and address this topic further. Additionally we have documented significant amounts of prey sharing among cougars and significant amounts of scavenging from cougar kills. Understanding these kill site dynamics will provide information on kill rates, consumption rates and intra/interspecific interactions (see Appendix IV for more details).

We have also initiated two additional graduate projects at CSU to focus on other aspects of the Front Range Cougar Study. First we have begun a Ph.D. project with Mevin Hooten at CSU through the statistics department to develop movement models and examine cougar GPS data for various movement patterns relative to roads, human density/activity, and other landscape/environmental features (Appendix V). The other project that we have begun is a M.S. project with Bill Kendall at CSU through the Fish, Wildlife, and Conservation Biology Department to examine techniques to develop non-invasive population estimation methodology for cougars (Appendix VI).

RESULTS AND DISCUSSION

Collared cougars from the previous year were captured and re-collared to replace exhausted batteries throughout the year. An additional 6 independent age cougars were also captured and collared during the year (Table 1). Currently there are 18 independent age cougars in the study with functioning GPS collars.

Home ranges for collared cougars have been determined using minimum convex polygons (MCP) to depict the general pattern of use and potential overlap, but likely over-represent the actual area used by an individual. Home ranges exhibit similar patterns to previous years (Figures 4 and 5), being fairly linear in a north-south direction. Adult male home ranges (Figure 6) were much larger than adult female home ranges (Figure 7). Subadult male home ranges were smaller than adult male home ranges, but were also characterized by large movements and significant overlap with adults. Female home ranges were smaller with sizes between 80 and 120 km^2. Female home ranges also had significant overlap, especially among related individuals. We have also seen significant long-range movements and dispersals (Figure 8). Long-range movements are significant movements outside of a cougar's typical home range with the individual returning to the original area. Dispersals are similar movements but the individual does not return to its original area.

There were a total of 14 mortalities for adult collared cougars during the 2011-12 year (Table 1). Causes of death included vehicle collision (3), unknown sources (5), hunting (1), intraspecific (1) and management or landowner related death (4).

Cougar-human interaction was comparable to the previous year, which appears to be less interaction than in the first years of the study. This gives us little opportunity to test aversive conditioning techniques. Given the minimal response to aversive conditioning, we are altering our methods of examining it as a management tool. We will now have managers aversively condition any cougar that they encounter interacting with humans and that warrants such action. We will then compare the cougar's responses to this aversive conditioning to events where the cougar was in the same situation but was undetected by humans and therefore not aversively conditioned.

Relocation of cougars is also a management technique that we have evaluated in the past and has shown mixed results relative to age, sex and relocation distance. The NE region has expressed renewed interest in this and we will begin pilot work to investigate this in more detail. We will evaluate relocation distance relative to Directive W2 and the distance recommendations made for management as well as some more long-distance relocations. As this proceeds we will develop a more detailed study to thoroughly investigate cougar relocation parameters.

From Aug 1, 2008 through September 1, 2012 we have visited ~3700 clusters (S1-S5 types). However, not all of these clusters were considered to be random samples, and thus preliminary inferences have only been drawn from this subset. Starting in January, 130 cameras were deployed in random locations representing the range of habitat types and human densities. Cameras have been checked as needed and results appear to be promising with regard to the number of species that have been detected and the perfromance of the cameras. For a detailed summary of the predator-prey component of the project, prelimary results and prey composition in cougar diets see Appendix III.

The prey selection and kill site dynamics study was initiated in January (see Appendix IV for study objectives and methods). To date, we have collected 50 individual samples from deer killed by cougars and tested these for CWD. A small proportion of these have been positive for CWD. We have investigated numerous potential kill sites and placed cameras on 68 fresh kill sites to document the activity. We have documented 5 occasions when multiple cougars shared a kill and several scavenging events. Many scavenging events occur after the cougar has consumed the prey and has left. Other scavenging events have occurred while the cougar was still consuming the prey item, including cases where bears have usurped the prey item killed by the cougar.

Starting in November we began investigating snow tracking and lures as potential techniques to estimate cougar abundance. Snow tracking proved to be very difficult because there was limited snow throughout the winter and snow conditions were poor. When snow tracking was feasible tracks of collared cougars were followed and samples (primarily hair) were collected. This approach is highly dependent on environmental conditions and therefore may not be broadly applicable.

Efforts documented in the literature to lure cougars to specific locations and capture and individual with either a photograph or genetic sample have been limited and relatively unsuccessful. We have begun to rigorously test various options to lure cougars to specific locations and extract genetic samples. One option that has not been tested in other studies is the use of game calls to attract cougars. We placed 4 different types of sites at random locations to determine which types of lures or combinations of lures (bait, bait and scent, bait and call, bait, scent and call) would be the most reliable method of attracting cougars. We found that calls were significantly more effective (21 detections at sites with calls compared to 2 detections at sites without calls) at attracting cougars to a site (see Appendix VI for a detailed summary).

Although we were relatively effective at luring cougars to a specific location with calls, we were not successful in extracting genetic samples at these locations. Cougars appeared to ignore scratch pads and were hesitant to take any meat reward left at the site. Cougars did seem interested in the calls and on several occasions investigated the call or stole the call from the site. In the coming year we will investigate methods of extracting genetic samples from cougars approaching the call, likely this will involve barbed wire as a hair snag (see Appendix VI for a summary and study plan for continued research).

Throughout the year we have also been analyzing cougar GPS collar data to examine habitat use and movement patterns. Much of this has been geared towards the development of new statistical methods or refinement of statistical methods. A resource selection function (RSF) was run and we found that cougars are selecting for forest and shrub cover types and selecting against agriculture, city, and bare cover types. Additionally, we found little evidence to suggest cougars are avoiding roads. Finally, we have been working on a continuous-time discrete-space model of cougar movement. For more details of this approach see Appendix V.

SUMMARY

The use of telomeres as a method to determine the age structure of bear and cougar populations is promising and will be investigated further in the coming year. Further refinement of the age-to-length relationship for both species is warranted. In addition to this, length relationships relative to genetic relatedness and individual stressors will give further insight into interpreting results from future data. We will also be investigating the effects of hibernation on telomere length using both captive and wild bears.

The use of stable isotopes from bears and cougars is beginning to show some very interesting results. Examining stable isotopes from various bear tissue types will help elucidate temporal patterns in diet composition, including the use of human foods by bears. It has also become clear that stable isotopes will be a useful tool in examining cougar diets, especially in the use of small prey items that are likely overlooked with other traditional techniques.

In addition to re-collaring previously collared cougars, an additional 6 independent age cougars were collared during the year. Mortality remained high over the year with 14 cougars dying during the year. Home-range patterns remained consistent to previous years. The effectiveness of aversive conditioning is still showing mixed results, which is likely a factor of the opportunistic nature of cougars using urban environments and a lack of habituation to them. Relocation of cougars as a management tool has had limited assessment, but given some success, still warrants further investigation. Mule deer are the predominant prey in cougar diets, although males also utilize elk regularly. We will continue to assess predator-prey dynamics, population estimation techniques, and movement patters during the coming year.

LITERATURE CITED

Alldredge, M.W. 2007. Cougar demographics and human interactions along the urban-exurban front range of Colorado. Wildlife Research Report July: 153-202. Colorado Division of Wildlife, Fort Collins, USA.

Anderson, C.R., and F.G. Lindzey. 2003. Estimating cougar predation rates from GPS location clusters. Journal of Wildlife Management 67:307-316.

Boutin-Ganache, I., M. Raposo, M. Raymond, and C. F. Deschepper. 2001. M13-tailed primers improve the readability and usability of microsatellite analyses performed with two different allele-sizing methods. Biotechniques, 31:25-28.

Cawthon, R. M. 2002. Telomere measurement by quantitative PCR. Nucleic Acids Research 30:e47.

Cougar Management Guidelines Working Group. 2005. *Cougar Management Guidelines*, 1st ed. WildFutures, Bainbridge Island, Washington, USA.

Ernest, H. B., M. C. T. Penedo, B. P. May, M. Syvanen, and W. M. Boyce. 2000. Molecular tracking of mountain lions in the Yosemite Valey region in California: genetic analysis using microsatellites and faecal DNA. Molecular Ecology 9:433-441.

Harrison, R. L., P. B. S. Clarke, and C. M. Clarke. 2004. Indexing swift fox populations in New Mexico using scats. American Midland Naturalist 151:42-49.

Haussmann, M.F., D.W. Winkler, K.M. O'Reilly, C.E. Huntington, I.C.T. Nisbet, and C.M. Vleck. 2003. Telomeres shorten more slowly in long-lived birds and mammals than in short-lived ones. Proceedings of the Royal Society of London Series B 270:1387-1392.

Hemann, M. T., and C. W. Greider. 2000. Wild-derived inbred mouse strains have short telomeres. Nuclei Acids Research 28: 4474-4478.

Hoss, M., M. Kohn, S. Paabo, F. Knauer, and W. Schroder. 1992. Excrement analysis by PCR. Nature 359:199.

Logan, K.A. 2006. Cougar population structure and vital rates on the Uncompahgre Plateau, Colorado. Wildlife Research Report July:95-122. Colorado Division of Wildlife, Fort Collins, USA.

Menotti-Raymond, M. and S. J. O'Brien. 1995. Evolutionary conservation of ten microsatellite loci in four species of Felidae. Journal of Heredity 86:319-322.

Menotti-Raymond, M., V. A. David, L. A. Lyons, A. A. Shcaffer, J. F. Tomlin, M. K. Hutton, and S. J. O'Brien. 1999. A genetic linkage map of microsatellites in the domestic cat (Felis catus). Genomics 57:9-23.

Meyne, J, R. L. Ratliff, and R. K. Moyzis. 1989. Conservation of the human telomere sequence $(TTAGGG)_n$ among vertebrates. Proceedings of the National Academy of Sciences 86: 7049-7053.

Nakagawa, S., N.J. Gemmell, and T. Burke. 2004. Measuring vertebrate telomeres: applications and limitations. Molecular Ecology 13:2523-2533.

Sinclair, E. A., E. L. Swenson, M. L. Wolfe, D. C. Choate, B. Bates, and K. A. Crandall. 2001. Gene flow estimates in Utah's cougars imply management beyond Utah. Animal Conservation 4:257-264.

Smith, D. A., K. Ralls, B. L. Cypher, and J. E. Maldonado. 2005. Assessment of scat-detection dog surveys to determine kit fox distribution. Wildlife Society Bulletin 33:897-904.

Taberlet, P., and J. Bouvet. 1992. Bear conservation genetics. Nature 358:197.

Watson, J.D. 1972. Origin of concatameric T4 DNA. Nature-New Biology 239:197-201.

Prepared by ____
Mathew W. Alldredge, Wildlife Researcher

Table 1: Capture history, aversive conditioning treatments and current status of all independent age cougars captured as part of the Front Range cougar study.

Cougar ID	Sex	Age	Date	Location	Occurrence	Capture	Release Loc	Conditioning	Status
AM02	M	1	6/14/07	Lacey Prop.	Baiting	Cage	On-site	NA	Alive
		1.5	1/10/08	White Ranch	Capture effort	Hounds	On-site	NA	Alive
		1.5	2/9/08	Coal Creek	Intraspecific mortality				Dead
AM04	M	7	7/14/07	White Ranch	Baiting	Cage	On-site	NA	Alive
		7	10/17/07	Eldorado Springs	Livestock depredation	Cage	White Ranch	Beanbag	Alive
		8	4/29/08	Magnolia/Flagstaff	Replace Collar	Hounds	On-site	NA	Alive
		8	5/5/08	South Boulder	Seen in town	Free-dart	Lindsey	Beanbag	Alive
		8	8/4/08	North Boulder	Killed deer in town	Cage	Centennial Cone	Beanbag	Alive
		9	2/24/09	Boulder Canyon	Punctured intestine				Dead
AM06	M	5	11/21/07	Heil Valley Ranch	Capture effort	Hounds	On-site	NA	Alive
		6	12/30/08	Heil Valley Ranch	Replace Collar	Hounds	On-site	NA	Alive
		7	2/2/10	Reynolds Ranch	Replace Collar	Hounds	On-site	NA	Alive
		7	2/15/10	White Ranch	Hunter				Dead
AF03	F	4	11/29/07	Flagstaff	Deer kill	Cage	On-site	NA	Alive
AF01	F	2	12/17/07	Table Mesa	Deer kill	Cage	On-site	NA	Alive
		4.5	12/15/10	White Ranch	Replace Collar	Hounds	On-site	NA	Alive
			3/12/12	BCOS Lindsey	Deer kill	Free-dart	On-site	NA	Alive
AM05	M	2	12/19/07	White Ranch	Capture effort	Hounds	On-site	NA	Alive
		4	12/4/09	White Ranch	Replace collar	Hounds	On-site	NA	Alive
		5	4/4/10	Golden	Roadkill				Dead
AM07	M	1.5	12/26/07	Heil Valley Ranch	Capture effort	Hounds	On-site	NA	Alive
			4/19/08	Highway 7	Roadkill				Dead
AF08	F	1.5	12/26/07	Heil Valley Ranch	Capture effort	Hounds	On-site	NA	Alive
		3	6/18/09	West Horsetooth	Deer kill-remove collar	Cage	On-site	NA	Alive
AM09	M	1.5	12/28/07	Heil Valley Ranch	Capture effort	Hounds	On-site	NA	Alive
		2.5	12/27/08	Hwy 34 (mile 70)	Roadkill				Dead
AF10	F	7	1/15/08	Apex Open Space	Deer Kill	Cage	On-site	NA	Alive

			2/13/08	I-70	Roadkill				Dead
AF19	F	8+	3/4/08	Heil Valley Ranch	Capture effort	Hounds	On-site	NA	Alive
		8+	3/18/09	North Boulder	Deer Kill	Cage	Heil Valley Ranch	Beanbag	Alive
			4/13/09	Left Hand Canyon	Deer Kill	Cage	Heil Valley Ranch	NA	Alive
		8+	1/20/09	Dowe Flats	Deer Kill	Cage	On-site	NA	Alive
			11/5/10	Foothills Hwy, N. Boulder	Roadkill				Dead
AF11	F	1.5	3/5/08	South Table Mesa	Deer Kill	Cage	On-site	NA	Alive
			8/20/08	US-40/Empire	Roadkill				Dead
AM20	M	4	3/6/08	White Ranch	Capture effort	Hounds	On-site	NA	Alive
			5/18/08	West of White Ranch	Livestock Depredation	Shot			Dead
AF15	F	6	3/18/08	Coffin Top	Capture effort	Hounds	On-site	NA	Alive
		7	4/2/09	Hall Ranch	Replace Collar	Hounds	On-site	NA	Alive
			3/25/10	Coffin Tip	Replace Collar	Hounds	On-site	NA	Alive
		8-9	2/4/11	Hall Ranch	Deer Kill	Snare	On-site	NA	Alive
		9+	2/2/12	Longmont Dam Rd	Deer Kill	Snare	On-site	NA	Alive
AF17	F	9+	3/29/08	Sugarloaf	Pet depredation	Cage	Within 1 mile	Beanbag	Alive
			5/20/08	Four-mile Canyon	Unknown mortality				Dead
AF12	F	2	5/8/08	N. Boulder	Deer Kill	Cage	US Forest Boulder Canyon	Beanbag	Alive
			5/29/08	N. Boulder	Livestock depredation	Cage	Near Ward	Beanbag	Alive
			2/13/09	N. Boulder	Deer Kill	Snare	None		Dead
AM13	M	2	5/8/08	Sugarloaf	Livestock depredation	Cage	On-site	Beanbag	Alive
			12/17/08	Heil Valley Ranch	Replace Collar	Hounds	On-site	NA	Alive
		3	12/17/09	Heil Valley Ranch	Replace Collar	Hounds	On-site	NA	Alive
			3/27/12	Hall Ranch	Detected by camera				Alive
AM14	M	2	5/15/08	South Boulder	Seen under deck	Free-dart	Lindsey	None	Alive
			5/20/08	South Boulder	Deer kill	Free-dart	West of Rollinsville	Beanbag	Alive
			4/14/09	Rollins Pass	Replace Collar	Hounds	On-site	NA	Alive
		3	2/16/10	Left Hand Canyon	Replace Collar	Hounds	On-site	NA	Alive
		4.5	6/22/11	Allenspark	Elk Kill	Cage	On-site	NA	Alive
		4-5	11/9/11	Hwy 72	Raccoon Kill	Free-dart	On-site	NA	Alive

		4-5	12/4/11	Allenspark	Shot/depredation				Dead
AF34	F	1.5	12/5/08	Heil Valley Ranch	Capture effort	Hounds	On-site	NA	Alive
			3/18/09	N. Boulder	Deer kill	Cage	Heil Valley Ranch	Beanbag	Alive
		2.5	1/4/10	Heil Valley Ranch	Replace Collar	Hounds	On-site	NA	Alive
		3.5	12/31/10	Hall Ranch	Replace Collar	Hounds	On-site	NA	Alive
		4.5	12/28/11	Hall Ranch	Replace Collar	Hounds	On-site	NA	Alive
		5.5	2/13/12	W of Hall Ranch	Unknown mortality				Dead
AM18	M	1.5	12/24/08	Evergreen	Deer kill	Cage	Mt. Evans SWA	None	Alive
			3/14/09	Evergreen	Livestock depredation	Cage	None		Dead
AF16	F	3	12/29/08	Evergreen	Deer Kill	Snare	Flying J Open Space	None	Alive
			3/20/09	Evergreen	Livestock depredation	Cage	Mt. Evans SWA	Beanbag	Alive
AF45	F	5	1/2/09	Gold Hill	Deer kill	Cage	On-site	NA	Alive
			11/24/10	N.Boulder	Euthanized/Lisa Wolfe			NA	Dead
AF40	F	1.5	1/27/09	White Ranch	Capture effort	Hounds	On-site	NA	Alive
		1.5	1/28/09	White Ranch	Replace Collar	Hounds	On-site	NA	Alive
		2.5	2/22/10	White Ranch	Replace Collar	Snare	On-site	NA	Alive
		4-5	3/4/12	Idaho Springs	Fawn Kill	Snare	On-site	NA	Alive
AF24	F	10+	2/12/09	North Boulder	Deer Kill	Cage	Hall Ranch	None	Alive
			2/25/09	Hwy 7	Replace Collar	Hounds	On-site	NA	Alive
			4/4/09	North Boulder	Raccoon Kill	Free-dart	Heil Valley Ranch	None	Alive
			5/31/09	North Boulder	Encounter	Shot			Dead
AM31	M	1.5	12/31/08	Evergreen	Chicken coop	Hounds	On-site	None	Alive
			3/29-09	Conifer	Livestock depredation	Cage	Mt. Evans SWA	None	Alive
		2.5	2/16/10	Douglas, WY	Hunter				Dead
AF37	F	1.5	12/31/08	Evergreen	Chicken coop	Free-dart	On-site	None	Alive
			8/11/09	I-70	Roadkill				Dead
AM21*	M	1.5	8/29/09	N. Boulder	Encounter	Free-dart	Ward	None	Alive
		2	3/???/10	Loveland	Livestock depredation				Dead
AF32	F	1.5	9/28/09	Indian Hills	Livestock depredation	Cage	Within 1 mile	None	Alive
		3.5	11/28/10	Golden	In neighborhood	Free-dart	White Ranch	None	Alive
		3.5	12/1/10	Golden	In neighborhood	Cage	Radium	None	Alive
			9/23/11	Green Mtn. Res.	Found dead				Dead

AM46	M	2	11/13/09	Evergreen	Elk kill	Cage	On-site	None	Alive
				Genesee	Livestock depredation	Shot			Dead
AF50	F	3	11/24/09	West of Boulder	Deer kill	Cage	On-site	NA	Alive
AM44	M	6	12/15/09	White Ranch	Capture effort	Hounds	On-site	NA	Alive
			3/18/10	White Ranch	Replace collar	Hounds	On-site	NA	Alive
		7-8	3/20/11	White Ranch	Elk kill	Snare	On-site	NA	Alive
		9	5/30/12	SW of White Ranch	Shot/depredation				Dead
AM606	M	2	1/6/10	Boulder	Seen in town	Free-dart	MacGregor Ranch	None	Alive
			9/23/11	Laporte	Shot killing goat				Dead
AF54	F	4	1/14/10	White Ranch	Capture effort	Hounds	On-site	NA	Alive
			5/16/11	White Ranch	Deer Kill/Replace Collar	Cage	On-site	NA	Alive
AF52	F	4	1/28/10	Hall Ranch	Capture effort	Hounds	On-site	NA	Alive
		5-6	3/24/11	Hall Ranch	Deer Kill	Cage	On-site	NA	Alive
AM51	M	1.5	1/28/10	Hall Ranch	Capture effort	Hounds	On-site	NA	Alive
AF56	F	1.5	2/22/10	Conifer	Livestock depredation	Cage	Mt. Evans SWA	Beanbag	Alive
AF55	F	4	2/23/10	Conifer	Livestock depredation	Cage	Mt. Evans SWA	Beanbag	Alive
			3/13/10	Conifer	Pet Depredation	Cage		Euthanized	Dead
AM53	M	4	3/13/10	Genesee	Elk Kill	Cage	On-site	NA	Alive
			3/3/11	Medved property	Shot/hunter				Dead
AM60	M	2	3/29/10	Walker Ranch	Baiting	Cage	On-site	NA	Alive
AF58	F	1.5	4/4/10	Table Mesa	Baiting	Cage	On-site	NA	Alive
			6/3/10		Roadkill				Dead
AF62	F	5	4/13/10	Walker Ranch	Elk Kill	Cage	On-site	NA	Alive
		6	4/13/11	Walker Ranch	Baiting	Cage	On-site	NA	Alive
			12/10/11	Gross Dam	Non-target/released	Cage	On-site	NA	Alive
AF59	F	5	4/22/10	Blue Jay/Jamestown	Deer Kill	Cage	On-site	NA	Alive
		5	1/6/11	N. Boulder	Deer Kill	Cage	On-site	NA	Alive
		5-6	12/29/11	Sunshine Canyon	Deer Kill	Free-dart	On-site	NA	Alive
		6	3/6/12	NW of Boulder	Unknown mortality				Dead
AM63	M	1	9/22/10	Paradise Park	Deer Kill	Cage	White Ranch	None	Alive
			9/30/10		Road Kill				Dead

AF57	F	3	11/3/10	Lacy Property	Baiting	Snare	On-site	NA	Alive
		4-5	2/4/12	JCOS Ralston Buttes	Replace Collar	Hounds	On-site	NA	Alive
AF61	F	4-5	11/18/10	Flagstaff	Deer Kill	Free-dart	On-site	NA	Alive
		4-5	3/2/11	Coal Creek Canyon	Raccoon Kill	Cage	Walker Ranch	None	Alive
		5	12/10/11	Gross Dam Rd	Baiting	Snare	On-site	NA	Alive
AF64	F	1.5	1/20/11	Heil Valley Ranch	Baiting	Cage	On-site	NA	Alive
		3-4	7/19/12	N of Nugget Hill	Kill	Snare	On-site	NA	Alive
AM67	M	1.25	12/16/10	White Ranch	Baiting	Cage	On-site	NA	Alive
			3/4/12	Big Thompson	Shot/Depredation	Snare			Dead
AF69	F	1.5	12/1/10	N. Boulder	Deer Kill	Free-dart	On-site	NA	Alive
		2	4/6/11	N.Boulder/Town	Deer Kill	Free-dart	Reynolds Ranch	None	Alive
		4	3/31/12	Wonderland	Deer Kill	Cage	On-site	NA	Alive
AM70	M	2	1/23/11	Gold Hill	Deer Kill	Cage	On-site	NA	Alive
			3/2/11	Boulder Heights	Dog Kill	Cage	Reynolds Ranch	None	Alive
		3	2/26/12	Buckhorn Rd	Unknown mortality				Dead
AM71	M	2	1/27/11	Heil Valley Ranch	Baiting	Cage	On-site	NA	Alive
		3	12/23/11	Casper, WY	Shot/hunter	Hounds			Dead
AM72	M	4	2/6/11	Heil Valley Ranch	Baiting	Snare	On-site	NA	Alive
		5	5/2/12	Heil Valley Ranch	Unknown mortality				Dead
AF73	F	4	3/6/11	Sunshine Canyon	Baiting	Cage	On-site	NA	Alive
		3-4	10/28/11	Four Mile Canyon	Deer Kill	Cage	On-site	NA	Alive
AM74	M	4	2/23/11	White Ranch	Baiting	Cage	On-site	NA	Alive
		5	3/7/12	Golden Gate Canyon	Deer Kill	Snare	On-site	NA	Alive
AM76	M	2-3	3/6/11	Heil Valley Ranch	Baiting	Cage	On-site	NA	Alive
		3	12/27/11	Heil Ranch	Replace collar	Hounds	On-site	NA	Alive
AF77	F	5	3/9/11	Morrison Mountain	Baiting	Cage	On-site	NA	Alive
AM78	M	2	3/18/11	W. Evergreen	Deer Kill	Cage	On-site	NA	Alive
			5/12/11	Soda Creel/I-70	Road Kill				Dead
AF79	F	4	3/18/11	Mt. Evans	Dumpsite	Cage	On-site	NA	Alive

		4-5	2/17/12	Mt. Evans	Replace Collar	Hounds	On-site	NA	Alive
AM80	M	1.75	3/18/11	Mt. Evans	Dumpsite	Cage	On-site	NA	Alive
AM84	M	2	4/9/11	Shield Park HOA	Sheep depredation	Cage	Deer Creek Canyon	None	Alive
		3	5/4/12	S. Deer Creek	Shot/depredation				Dead
AF91	F	1.5	2/4/12	Cotter Mine	Capture effort	Hounds	On-site	NA	Alive
		2	7/20/12	I-70	Road Kill				Dead
AF86*	F	1.5	3/13/12	Gross Dam Rd.	Collared	Snare	On-site	NA	Alive
AF22	F	1.5	2/29/12	Golden	Baiting	Cage	On-site	NA	Alive
AF87	F	4-5	11/18/11	Heil Ranch	Baiting	Snare	On-site	NA	Alive
		4	12/7/11	Hall Ranch	Deer Kill	Cage	On-site	NA	Alive
AF88	F	1.5	10/14/11	N. Boulder	Deer Kill	Cage	On-site	NA	Alive
		2	1/11/12	White Ranch	Possible Intraspecific				Dead

SW023	F	1	4/9/09		Rehab	Release	Pike forest	None	Alive
			11/14/09	Lost Valley Ranch	Found dead				Dead
SW026	M	1	10/20/09		Rehab	Release	Hermit Park	NA	Alive
		3	8/19/11	New Mexico	Shot/hunter				Dead
SW107	M	1	5/7/10		Rehab	Release	Radium	NA	Unkn
			3/22/11		Shot/hunter				Dead
AF995	F	1	8/25/11		Rehab	Release	Wonderland	NA	Alive
		2	6/23/12	Sunshine Canyon	Road Kill				Dead

Table 2: Capture history, aversive conditioning treatments and current status of all cougar cubs captured as part of the Front Range cougar study.

Cougar ID	Sex	Age	Mother	Date	Location	Occurrence	Capture	Release Loc	Conditioning	Status
AF35	F	3	AF16	12/29/08	Evergreen	Deer Kill	Cage	Flying J Open Space		Alive
				12/31/08	Evergreen	Roadkill				Dead
AM36	M	3	AF16	12/29/08	Evergreen	Deer Kill	Cage	Flying J Open Space		Alive
				1/8/09	Evergreen	Starvation				Dead
AM30	M	8	AF01	1/30/09	S. Boulder	Deer Kill	Cage	On-site		Alive
										Dead
AM38	M	8	AF01	1/30/09	S. Boulder	Deer Kill	Cage	On-site		Alive
				3/27/09	S. Boulder	Encounter	Free-dart	Lindsey	Beanbag	Alive
				3/30/09	S. Boulder	Pet Depredation	Free-dart	Centennial Cone	None	Alive
				4/9/09	Morrison	Encounter	Free-dart	None	Euthanized	Dead
AM29	M	6	Euth.	2/11/09	N. Boulder	Deer Kill	Free-dart	Hall Ranch	None	Alive
		12		6/15/09	N. Boulder	Encounter	Free-dart	Masonville	Beanbag	Alive
				10/23/09	Big Thompson	Goat Depredation	Shot			Dead
AM21* collared	M	12	Unkn	3/25/09	Table Mesa	Baiting	Cage	On-site	NA	Alive
										Dead
AM25	M	12	Unkn	5/22/09	Indian Hills	Deer Kill	Cage	On-site	None	Alive
				9/13/09		Raccoon	Free-dart	Perforated intestine		Dead
AM41	M	12	Unkn	5/22/09	Indian Hills	Deer Kill	Free-dart	On-site	None	Alive
					Indian Hills	Encounter	Shot			Dead
AM65	M	4-5	AF32	11/28/10	Golden	In Neighborhood	Free-	White Ranch	None	Alive

							dart			
AM66	M	4-5	AF32	11/28/10	Golden	In Neighborhood	Free-dart	White Ranch	None	Alive
				12/1/10	White Ranch	Recapture	Hounds	Radium	None	Alive
AF68	F	10	AF50	2/9/11	Flagstaff	Deer Kill	Cage	On-site	NA	Alive
AM83	M	9	AF52	3/24/11	Hall Ranch	Deer Kill	Cage	On-site	NA	Alive
AM85	M	9	AF62	4/13/11	Walker Ranch	Baiting	Cage	On-site	NA	Alive
AF86*	F	9	AF62	4/13/11	Walker Ranch	Baiting	Snare	On-site	NA	Alive
collared										Alive

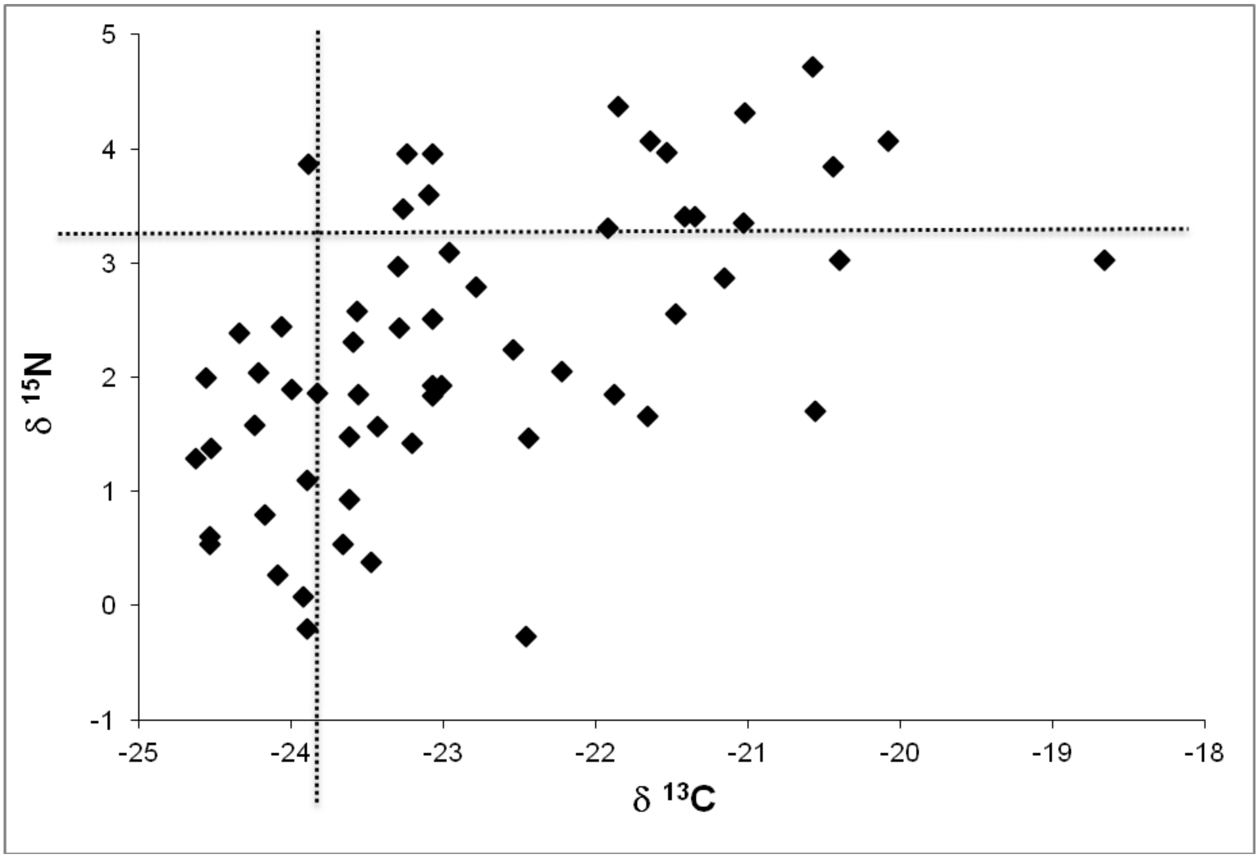


Figure 1: Carbon and nitrogen in hair from 60 bears harvested in Colorado during the 2011 hunting season showing the variability in concentrations reflecting dietary differences.

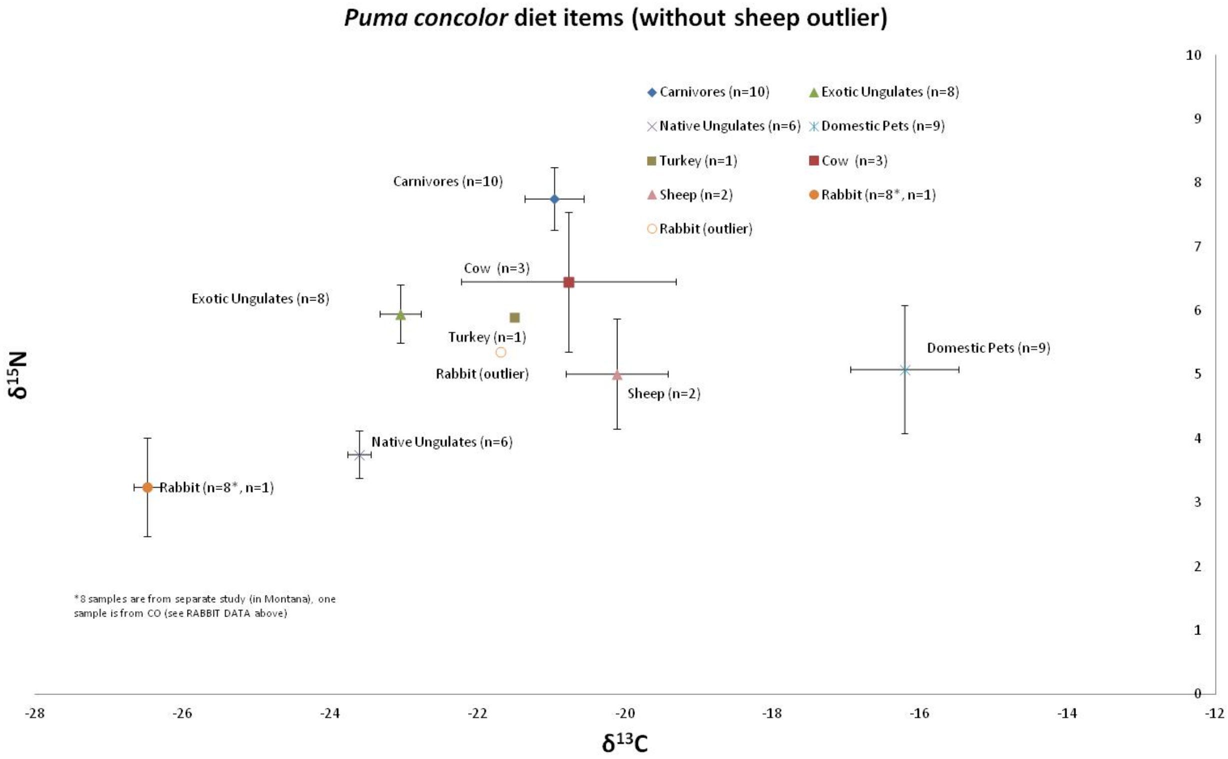
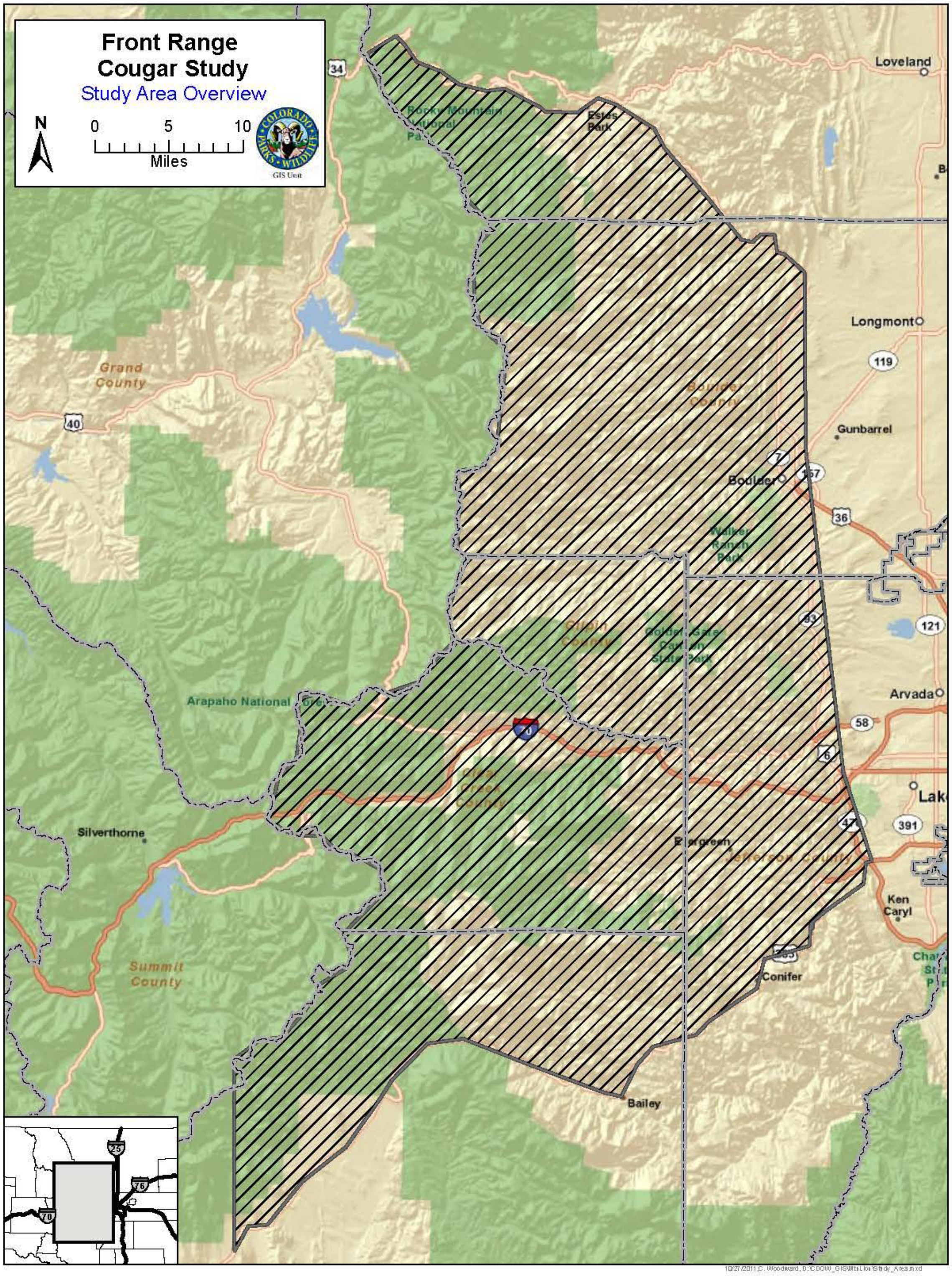


Figure 2: Carbon and nitrogen content in hair samples from cougar prey items found in the Front Range of Colorado. Prey items grouped into guilds demonstrates differences in carbon and nitrogen content based on similarities in prey species diet.



10/27/2011, C. Woodward, D:\CDOW_GIS\MtLion\Study_Area.mxd

Figure 3: Study area for the main Front Range cougar study where most capture effort and field work is conducted.

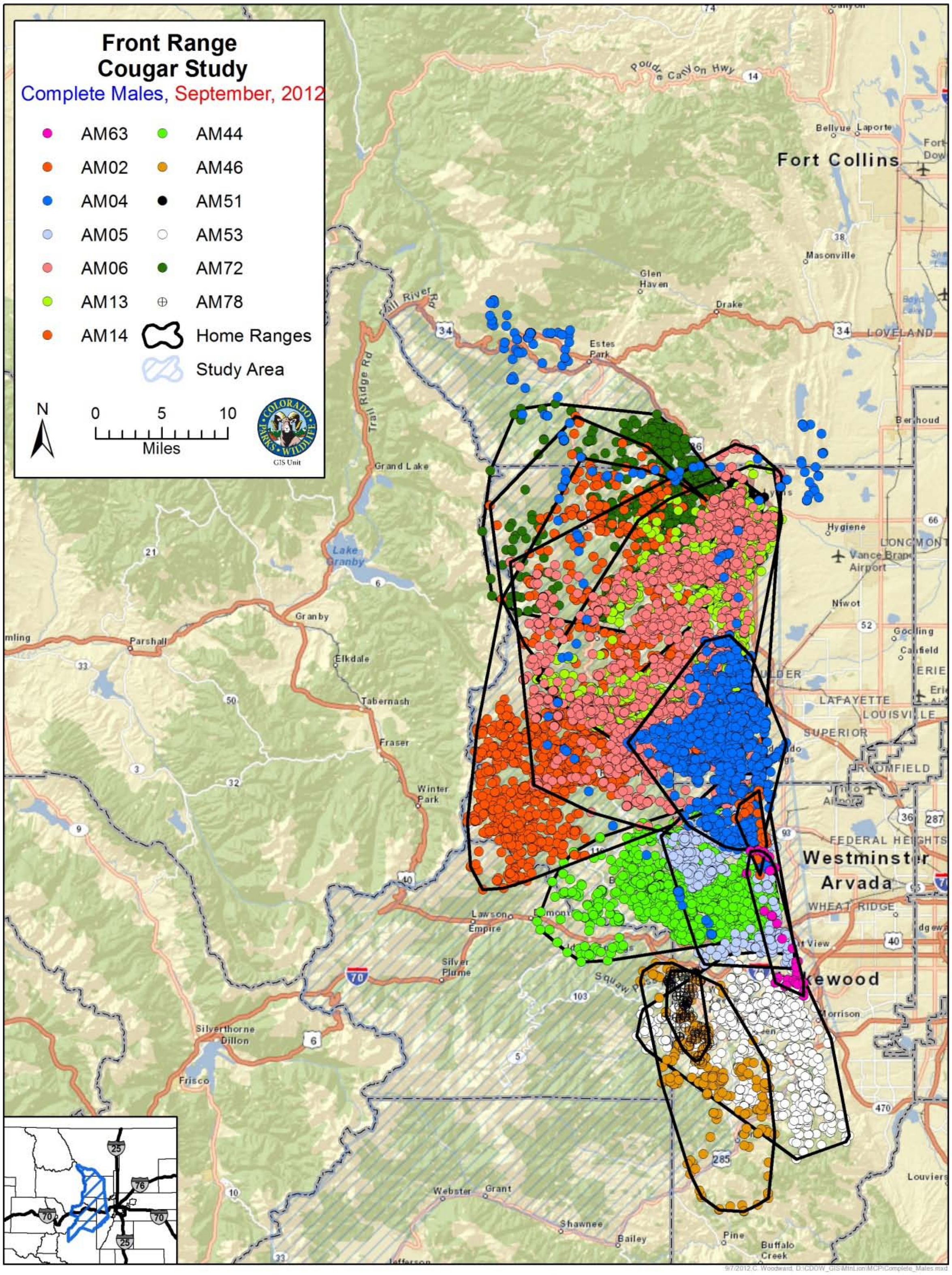
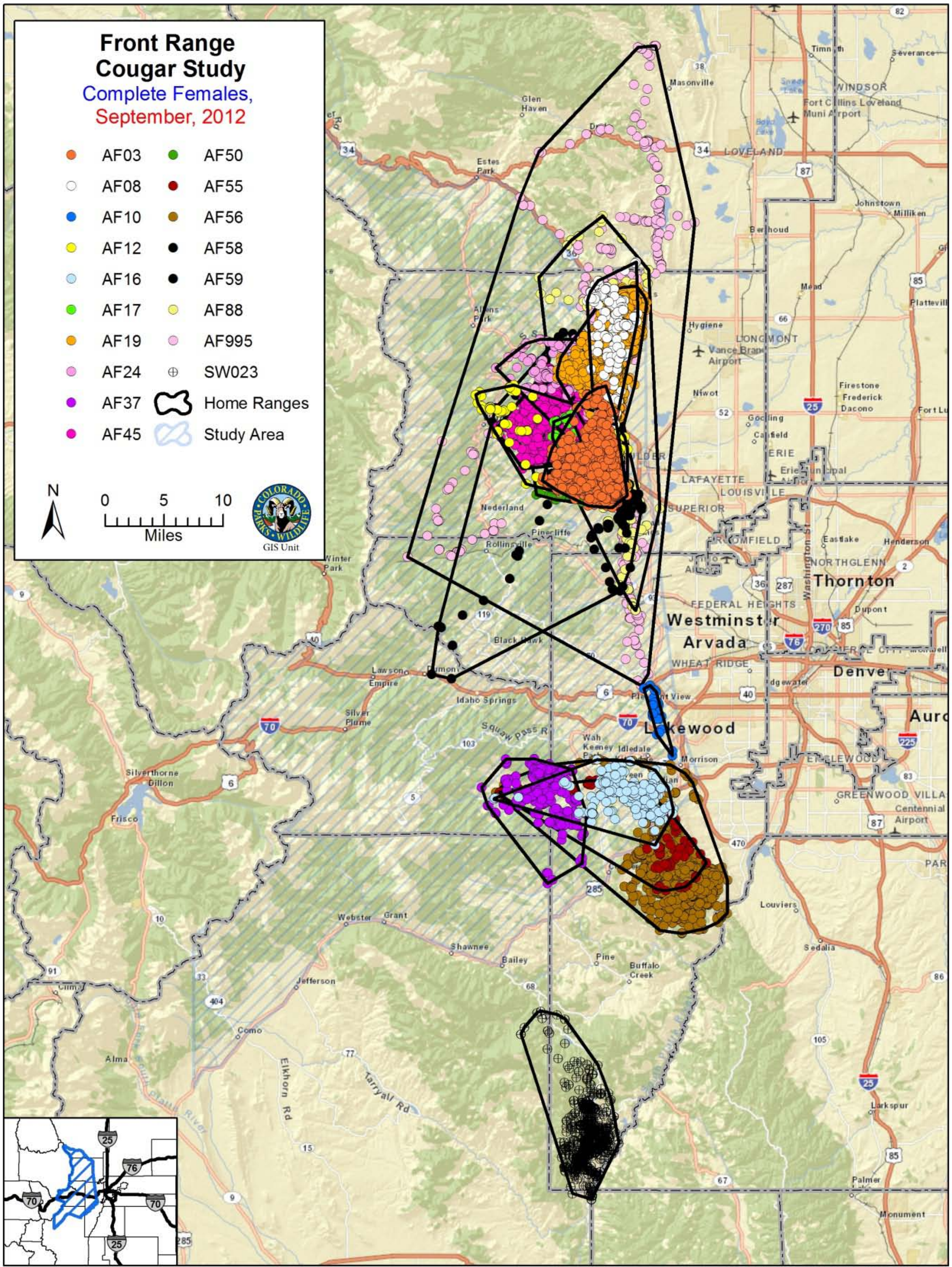


Figure 4: MCP home-ranges for male cougars that have previously been collared but are no longer in the study because of mortality or dispersal.



9/7/2012 C. Woodward D:\CDOW_GIS\MtnLion\MCP\Complete_Females.mxd

Figure 5: MCP home-ranges for female cougars that have previously been collared but are no longer in the study because of mortality or dispersal.

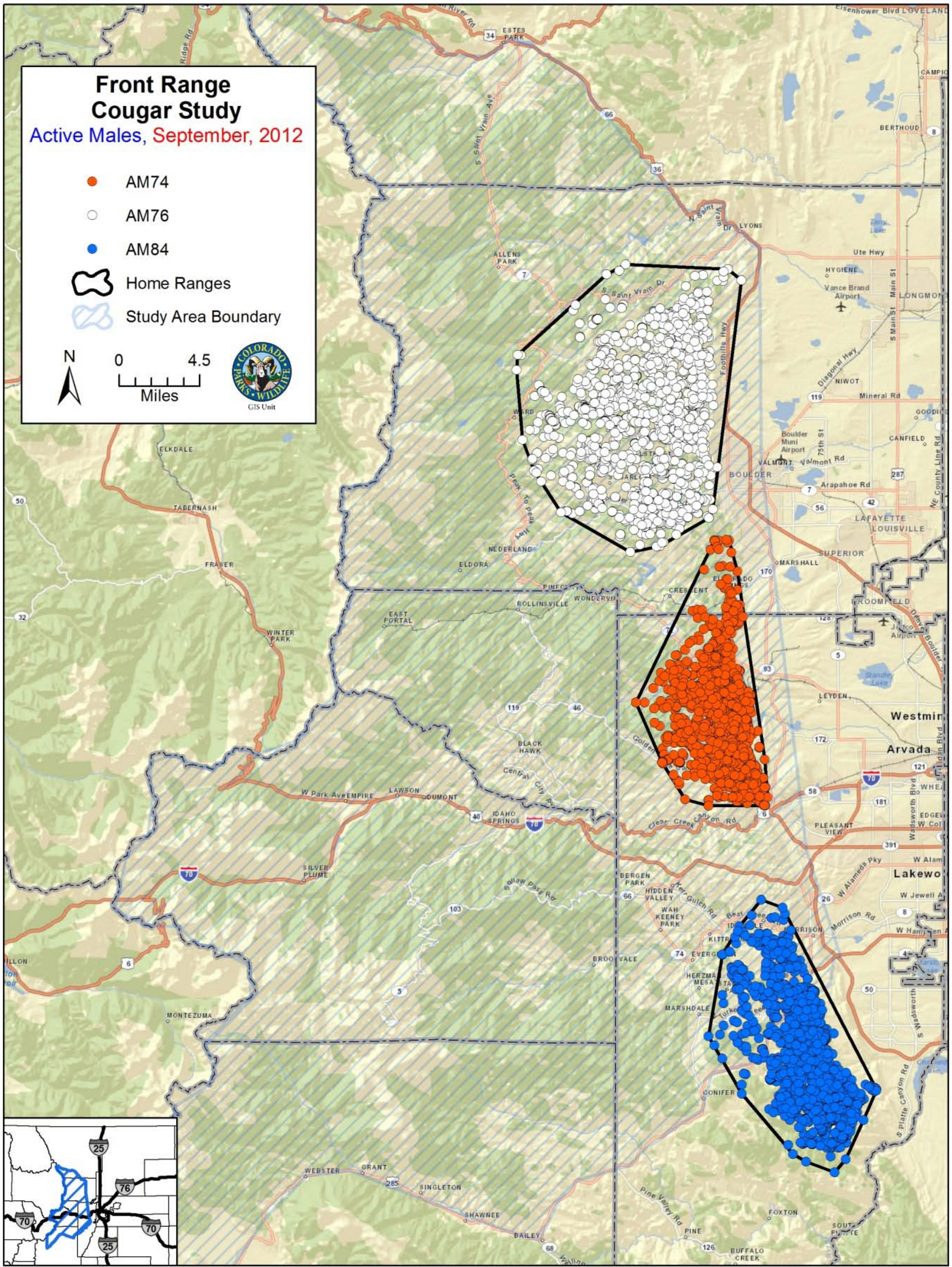


Figure 6: MCP home-ranges for male cougars that are currently in the study and being monitored.

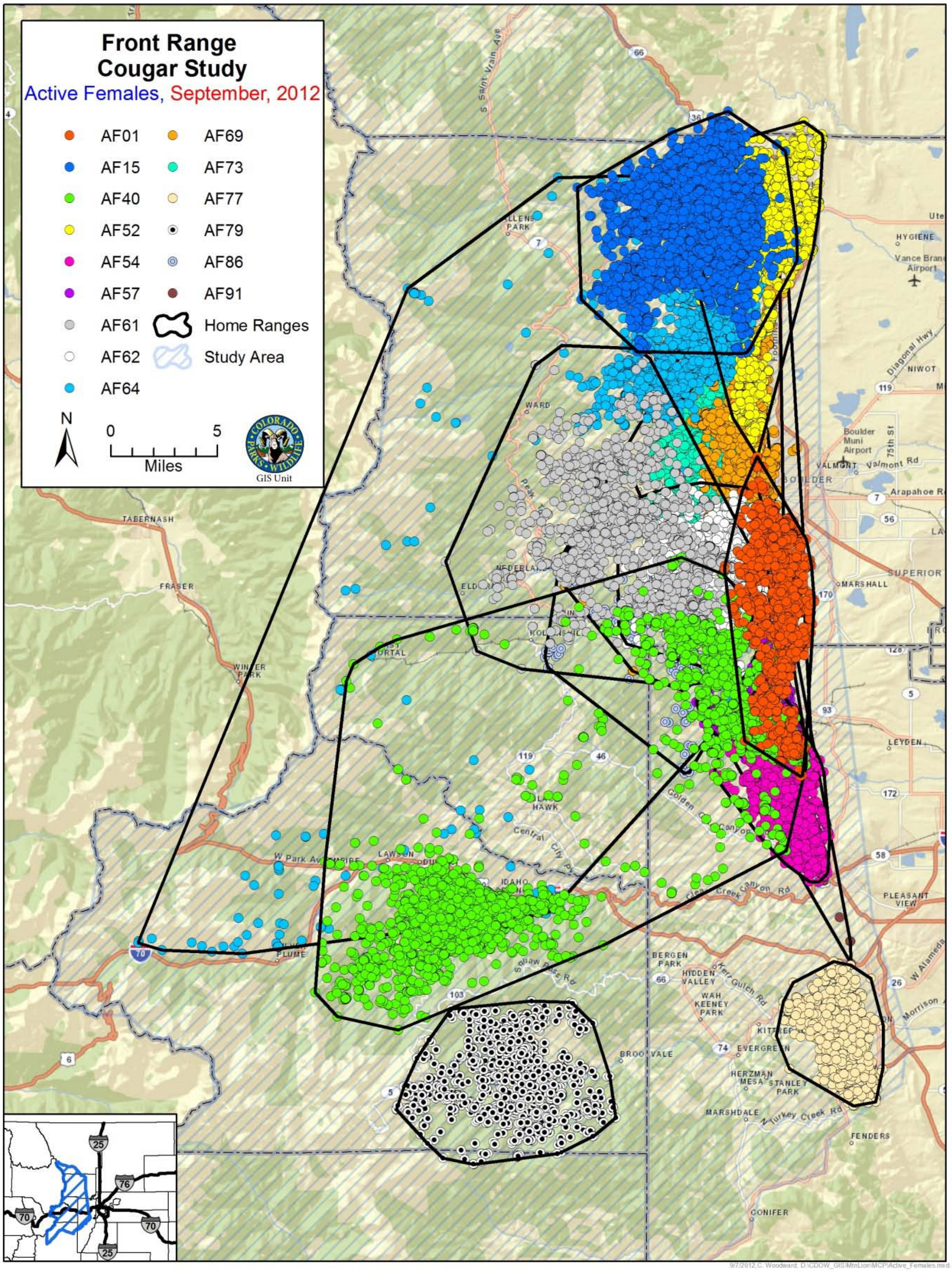


Figure 7: MCP home-ranges for male cougars that are currently in the study and being monitored.

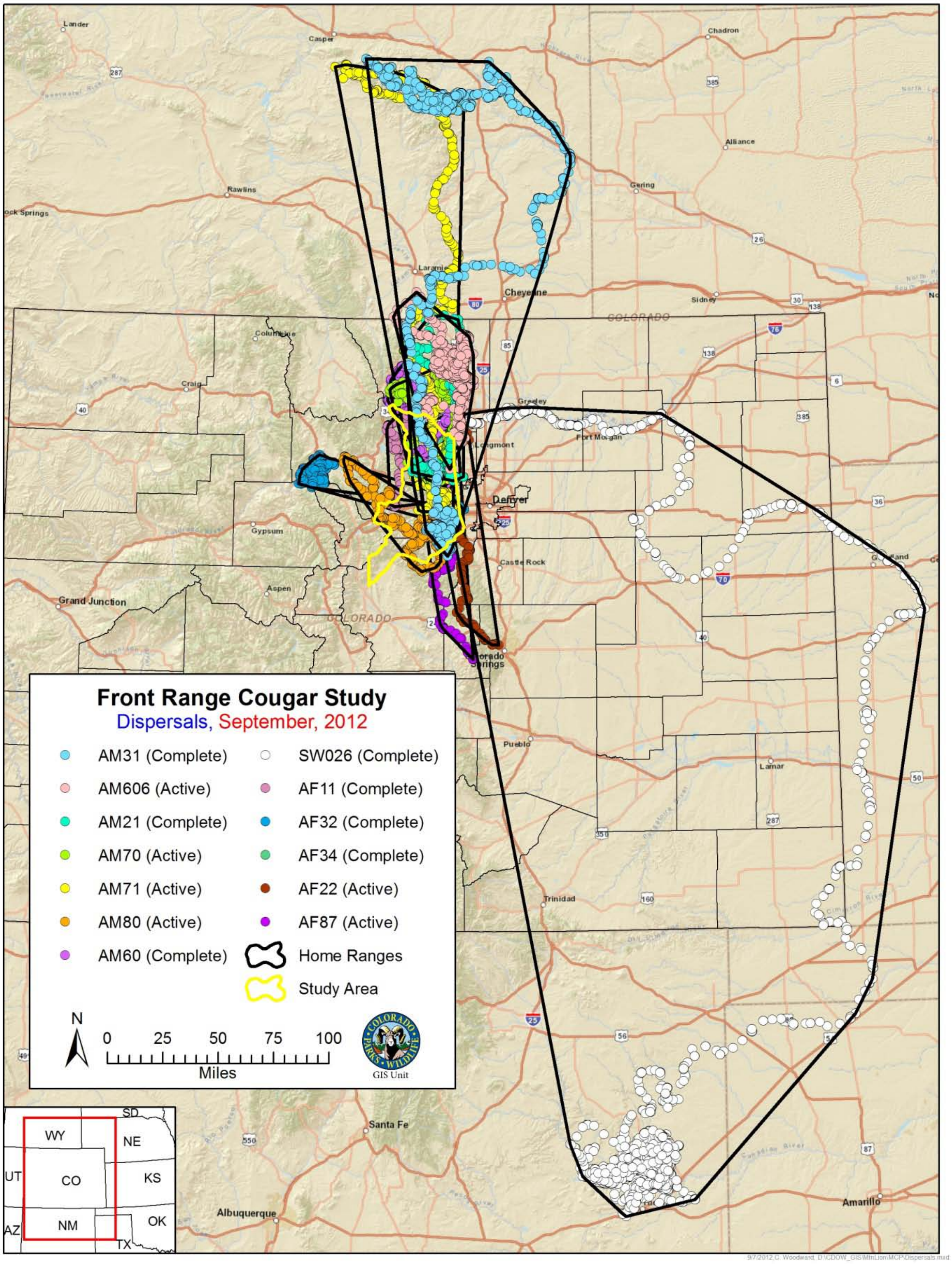


Figure 8: Dispersal/movement paths for cougars collared within the study area but traveled large distances outside of the study area.

APPENDIX I

Front Range Cougar Research
2011-2012 & 2012-2013

SPATIO-TEMPORAL PATTERNS OF DIET AND TELOMERE LENGTH IN COLORADO BLACK BEARS

UW-Wisconsin & Colorado Parks and Wildlife

Becky Kirby
Jonathan Pauli
Mat Alldredge

Research Proposal

SPATIO-TEMPORAL PATTERNS OF DIET AND TELOMERE LENGTH IN COLORADO BLACK BEARS

(Study Plan Draft for submission to CPW August 2012)
Becky Kirby, Ph.D. student, UW-Madison

Background and Need

Diet and foraging ecology

The effect of human-derived food on free-ranging wildlife populations is recognized as a growing problem across North America. This has been particularly evident among carnivore populations and especially related to human-wildlife conflict. In the past twenty years, American black bear (*Ursus americanus*) conflicts have expanded along the wildland-urban interface, and are generally attributed to access to human foods (Beckmann et al. 2008; Greenleaf et al. 2009). In Colorado, by the early 2000s, a third of bear mortalities resulted from conflicts with humans (Baruch-Mordo et al. 2008). Being opportunistic omnivores, black bears vary their food intake widely throughout seasons (Robbins et al. 2004) and can become habituated to human resources they encounter (McCarthy and Seavoy 1994; Beckman and Berger 2003), but reliance on such resources and subsequent effects on individual and population dynamics remains largely unknown.

Bear-human conflicts in Colorado and bear population dynamics exhibit high geographical and temporal variation (Baruch-Mordo et al. 2008; Beston 2011). Whether increased conflicts are due to growing populations, or alternatively environmental-mediated behavioral changes, remains unknown; and without a thorough understanding of individual, environmental, and population characteristics that contribute to nuisance bears, effective management has proven difficult. As conflicts are predicted to continue to rise, multi-pronged approaches that quantify the influence of anthropogenic foods are needed, as well as those that can assess regional population trends.

Stable isotope analysis has yielded significant contributions to wildlife ecology in the last several decades (Kelly 1999; Crawford et al. 2008); of particular interest to managers has been quantifying diet components of free-ranging vertebrates using carbon and nitrogen isotopes. Because corn and sugar utilize a distinct photosynthetic pathway from native plants in temperate North America, corn-dominated human food (waste and agriculture) exhibit distinct carbon ($\delta^{13}C$) values, which can be measured in consumer tissues (Jahren et al. 2006) (Figure 1). In addition, measuring nitrogen ($\delta^{15}N$) values can indicate trophic position and animal content in the diet; higher nitrogen values reflect higher trophic positions (Hobson and Welch 1992). Traditional diet reconstruction methods (such as scat or stomach content analyses) tend to underestimate highly digestible resources. Because diet analysis with stable isotopes uses the abundance of two elements (^{13}C and ^{15}N) it avoids this bias. Further, sampling tissues with different metabolic rates allows for higher resolution of temporal patterns of resource use (Hilderbrand et al. 1996). Using isotopic mixing models, we can calculate the percent of diet obtained from native plants, heterotrophs and human-derived food items (Phillips et al. 2005).

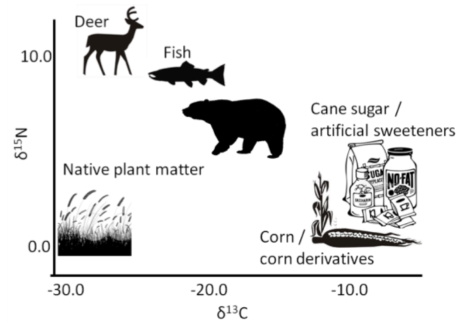


Figure 1. Illustration of carbon and nitrogen isotopic signatures of potential black bear diet sources. Bear tissues will exhibit signatures based upon their diet, which can be reconstructed using mixing models.

Recently, ^{13}C and ^{15}N have been employed attempting to discern bear reliance on human foods in Missoula, Montana and Yosemite, California. Although the study in Montana was unable to distinguish "wildland" from "urban" bears (Merkle et al. 2011), researchers in Yosemite were successful in using stable isotopes to differentiate management bears (conditioned to human food) from non-management bears (Hopkins et al. 2012). CPW (lead by Heather Johnson) is currently investigating bear behavior and habitat use as relates to nuisance bears. Our project will complement the work being conducted around Durango by assessing diet contributions across a broad spatial scale (the entire state), using stable isotope analysis and hunter-harvested animals.

Aging and telomeres

Quantifying the age structure of a population is central to understand its population growth rate and to forecast changes in population size. The age of bears, as well as other mammals, is typically determined by pulling a vestigial premolar and counting cementum annuli (Schroeder and Robb 2005). The estimated age from counts of cementum annuli is highly accurate, but requires the animal to be captured or harvested. With rising numbers of studies using noninvasive sampling for DNA analyses of hair, feather, and scat samples, an aging technique that could be applied to these samples would be desirable. Currently in Colorado a noninvasive hair-snare project is underway to estimate regional black bear densities. Consequently, a noninvasive aging technique would provide managers with the age structure of bears within Colorado and more power in modeling population trends and forecasting population growth. Recently developed in a few other species (e.g. Pauli et al. 2011), telomere length shows promise as such a molecular aging technique.

Previous research has demonstrated age-related telomere attrition in a variety of species and has correlated telomere length with individual age (e.g. Hemann and Greider 2000, Haussmann et al. 2003, Pauli et al. 2011). Telomeres are repetitive DNA sequences that cap the ends of eukaryotic chromosomes, whose nucleotide sequence $(T_2AG_3)_n$ is highly conserved across vertebrate species (Meyne et al. 1989). During each cell cycle telomeric repeats are lost because DNA polymerase is unable to completely replicate the 3’ end of linear DNA (Watson 1972). Thus, telomeres progressively shorten with each cell division. Telomerase, a reverse transcriptase, counteracts this loss in the germline, but tends to be far less active in somatic cells; this activity seems to vary with body mass, with larger animals having less telomerase activity (Seluanov et al. 2007). Additionally, lifestyle-related activities, in particular oxidative stress, can affect telomere length negatively (von Zglinicki 2002).

Using hair samples, Pauli et al. (2011) quantified telomere length via real-time quantitative polymerase chain reaction (Q-PCR) to age martens (*Martes* spp.). When accounting for a few covariates thought to influence telomere length (sex of the animal, size of the population, and geographic location), Pauli et al. (2011) found they could obtain accurate estimates of age class, and that age estimation via their model in fact exceeded those typically obtained from counts of cementum annuli. Thus, they concluded that quantification of telomere length could be a promising tool to age carnivores and estimate demographic structure for noninvasively collected hair samples (Pauli et al. 2011).

Subsequently, Jon Pauli and Mat Alldredge explored telomere length as a possible age marker for black bears using bears of known-age in Colorado and Wyoming. They found a weak linear relationship with telomere length declining with increasing animal age (Pauli and Alldredge, personal communication). Although much variation existed, this relationship was deemed to be potentially useful for further exploration. Taking advantage of the availability of hunter-harvested and nuisance bear samples with known ages, both diet via stable isotopes and aging via telomere length can be explored, with the goal of assigning biologically meaningful age classes.

Creating a reliable aging model would be aided by a deeper understanding of telomere dynamics and those biological covariates responsible for variation in rates of telomeric attrition. Telomere lengths and rates of attrition vary strongly between individuals and sexes of the same species, and inconsistently across vertebrate groups (Monaghan and Haussmann 2006). As they seem to reflect biological, rather than chronological age, understanding the factors that relate to telomere dynamics would enhance the development of an accurate model for aging and indicate which covariates are most influential. The vast majority of studies have focused on a single sampling in time (such as the hunter-harvested samples we are analyzing), and only a few studies have been conducted on free-ranging populations. Dunshea et al. (2011) recently called for more longitudinal studies to elucidate the factors that affect telomere dynamics. Thus, in order to rigorously investigate telomeres as an aging model for bears, further understanding of telomere attrition rates within individuals, not simply at a single time point, is required.

Multiple factors could play a role in aging, and hence telomere shortening (Monaghan and Haussmann 2006). Some species exhibit sex differences in telomere attrition (though not consistently across species) (Barret and Richardson 2011), and body size is known to be correlated with telomerase activity (Seluanov et al. 2007). Additional lifestyle factors such as nutritional condition and diet have also been linked to aging in some species (e.g. Shi et al. 2007, Cassidy et al. 2010), which are frequently related to oxidative stress, a known agent of telomeric shortening (von Zglinicki 2002). Further, considerable effort has been spent to understand bear hibernation physiology (e.g. Hellgren 1998), which may also be an important factor in aging. During hibernation, bears experience a slower metabolic rate, which results in less cell turnover (Koizumi et al. 1992). However, oxidative stress also increases as part of this process of metabolic depression (Chauhan et al. 2002), and aging is known to be strongly negatively affected by oxidative stress (von Zglinicki 2002, Cattan et al. 2008). This presents a potentially interesting question regarding whether telomeric attrition rates would be accelerated or attenuated during hibernation. Given that studies have shown torpor and lowered body temperatures to be associated with increased longevity (Lyman et al. 1981; Turbill et al. 2011), one would expect hibernation to slow the rate of telomere shortening. Only one study to our knowledge has examined telomeric attrition rates within individuals utilizing a lowered metabolic rate, a laboratory experiment on Djungarian hamsters (Turbill et al. 2012). In this case, they manipulated the environmental conditions of the individuals, and found that telomeres in fact increased in length more in those hamsters that used torpor more frequently (due to telomerase activity), in particular at lower body temperatures (Turbill et al. 2012).

However, bears have substantially different hibernation characteristics than smaller mammals, in that undergo less severe hypothermia, avoid bone loss, (McGee-Lawrence et al. 2008) and have minimal loss of muscle strength as well (Harlow et al. 2001). Understanding how hibernation and other life history tradeoffs affect telomere dynamics in black bears (Monaghan and Haussmann 2006) would augment their value as an age marker.

To this end, we will work with collared wild and captive black bears on the Colorado Front Range, in addition to the hunter-harvested samples. We will investigate black bear diet and telomere length to assess characteristics of bears reliant on human foods, and investigate telomere dynamics with the intent to develop an aging model.

Objectives

1. Quantify diet via stable isotopes in hunter-harvested and nuisance bears
    a. Examine relevant covariates (including age, sex, body condition, size, location, land use, distance to human development/agriculture, etc.) as related to stable isotope signatures of individuals
    b. Examine different tissue types to compare individual bear diet recently and averaged over a lifetime
2. Quantify telomere length in hunter-harvested bears
    a. Assess how biologically relevant covariates (as in Objective 1) are related to telomere length
    b. Model biologically relevant age classes as a function of telomere length and explore the potential of this model in aging noninvasively collected hair samples
3. Investigate individual telomere attrition rate longitudinally in wild and captive bears

a. Determine telomeric attrition rates for individual free-ranging and captive bears in Colorado and examine longitudinal changes in telomere length in relation to attributes of hibernation and other relevant covariates including body condition, sex, and habitat use
b. Use those most influential covariates to inform model building for Objective 2

Anticipated Benefits

This project will yield increased understanding of factors involving black bear reliance on human food and complement CPW's ongoing behavioral study of black bears around Durango. This study will also examine regional patterns of black bear diet on a broad spatial scale. Further, we will explore the potential of aging black bears via telomeres with the hope that it could be applied to noninvasively collected hair snares ongoing in the state. The ability to estimate age structure for the black bears within the state of Colorado would increase understanding of population trends. Finally, examining hibernation and other effects on telomeric attrition rates will be the first such study in a large mammal in the wild, and has potential applications to our understanding of aging generally, as well as informing an aging model.

Approach

Objectives 1 and 2: Conduct a broad spatial scale analysis of diet and telomere length in CO black bears

Hunter-harvested and nuisance bears

In 2011, we worked closely with District Managers to opportunistically collect samples from hunter-harvested and nuisance bears. When possible, District Managers collected four tissue samples from each bear: hair, whole blood, muscle tissue, and teeth. Hair, blood, and teeth are being analyzed with stable isotopes for diet; and hair and blood are also being utilized for DNA extraction for determination of telomere length. Muscle tissue will be preserved for future DNA extraction if necessary. Analyzing different tissue types with stable isotopes will allow us to ascertain diet in different snapshot windows of time. For black bears, blood represents recent diet (last 2-4 weeks), hair represents diet during the period of growth (May-Oct), and teeth (or bone) represent lifetime-accumulated diet (Hilderbrand et al. 1996). In August 2011, we presented our project design to District Managers and distributed one thousand collection kits to stations throughout the state with sampling instructions. Managers were asked to collect a few mls of whole blood (from body cavity, mouth, or vein), pull clumps of guard hair (with follicles intact), cut a small muscle sample (size of a dime), and pull a vestigial premolar (additional to the one pulled for aging). Samples were stored in the freezer until brought to Madison in December 2011. We received samples from about 400 individual bears with hair and teeth and muscle, and about half of those also had whole blood. Further, when available, nuisance bears that were tranquilized for transport, or euthanized, were sampled in a similar manner. Nuisance bears will also be sampled in 2012 for comparison. Data collected included customer ID, sex, and zygomatic width (on a paper ruler). We will also have access to age and reproductive history (for females) obtained from the teeth sent by CPW to Matson's laboratory, and GPS coordinates (about 300 of the harvests appear to have reasonable coordinates) (Figure 2).

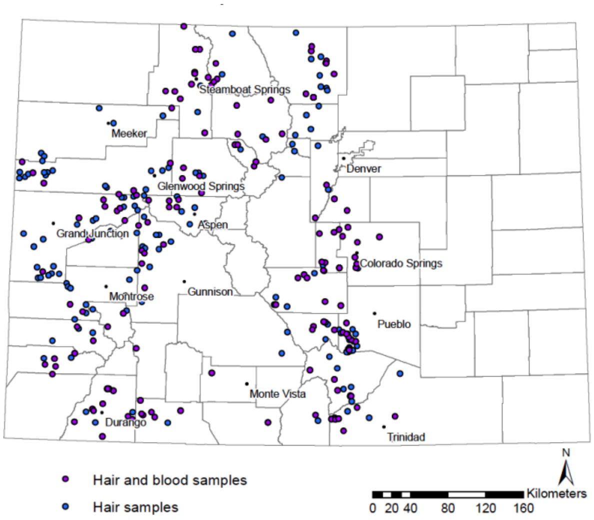


Figure 2. Location of hunter-harvested samples collected in fall 2012

Diet samples and location-specific covariates

We will collect potential diet samples (Table 1) from CO in summer/fall 2012.

Table 1. Potential native Colorado bear diet items obtained from (Irwin and Hammond 1985; Raine and Kansas 1990; Baldwin and Bender 2009)

Herbaceous	Spring beauty (*Claytonia lanceolata*)
	Fireweed (*Epilobium angustifolium*)
	Glacier lily (*Erythronium grandiflorum*)
	Dandelions (*Taraxacum* spp.)
	Cow parsnip (*Heracleum maximum*)
	Grasses, sedges, rushes
Hard mast	Oak acorns (*Quercus* spp.)
Berries	Chokecherries (*Prunus* spp.)
	Blueberry (*Vaccinium* spp.)
	Currants (*Ribes* spp.)
	Buffaloberry (*Sheperdia* spp.)
Mammals	Elk (*Cervus elaphus*)
	Mule deer (*Odecoileus hemionus*)
	Small mammals (Rodentia, Lagomorpha)
Insects	Ants (*Camponotus* spp., *Formica* spp.)
	Yellowjackets/wasps (*Vespidae* spp.)

We will obtain samples from four major regions (Northern Front Range, Southern Front Range, Southwest San Juan Mountains, and Piceance Basin in the Northwest). We will also opportunistically sample human food waste.

In order to elucidate environmental covariates that might influence diet choices, we will obtain other relevant covariates from available CPW spatial layers and datasets, as well as National Landcover Datasets (NLCD) and other publically available datasets. Using location data provided by hunters, for those bears with GPS coordinates, we will extract covariates that measure human influence, such as land use, distance to human development and agriculture. Ultimately, we will examine patterns of isotopic signature and telomere length in relation to individuals and environmental covariates.

<u>Objective 3: Examine factors contributing to individual telomeric attrition rate</u>

Free-ranging bears

Rates of telomere loss are often not constant with age (Hall et al. 2004), thus understanding the factors that contribute to these rates necessitates multiple samples from the same individuals. In order to examine individual telomeric attrition rate through time, we will work with bears on the Front Range that are being captured and tracked via GPS collars as a part of Mark Vieira's population study in summer 2012. We will attempt to collar at least 8 adult black bears (preferentially focusing on females, but an even split between sexes is appropriate) on the Northern Front Range in Colorado. Gender and organ specific differences in telomeres have been found in other species (e.g. Cherif et al. 2003), and may exist in bears. Further, because of the potential cost of reproduction (Heidinger et al. 2012), and sex differences in hibernation and thus amount of oxidative damage (Beaulieu et al. 2011), females are of primary interest.

Few telomere studies have followed individuals longitudinally, and none have been conducted on large mammals to our knowledge. As discussed earlier, telomere length can vary widely across individuals and species, and can be counteracted in high metabolic creatures (birds, bats) with high levels of telomerase (Munshi-South and Wilkinson 2010). In most mammals, telomerase activity decreases with age in somatic tissues, but animals that undergo physiological adaptations such as torpor, may counteract telomere shortening in other ways (Turbill et al. 2011). Thus, we will examine telomere length before, during, and after hibernation to estimate rate of change. We will obtain hair samples seasonally, 4 times a year for two years, in order to have replicate datasets and explore variation in weather covariates. Blood samples will be collected at first capture and subsequent den checks for stable isotope and body condition analyses (such as leptin, Spady et al. 2009), stable isotope, and telomeres. Capture procedures will principally follow the protocol approved for Heather Johnson's Durango study (see Appendix A).

To avoid disturbing bears within their dens multiple times, we will obtain pre- and post-hibernation hair samples using a hair snag, requiring only a single visit to a den each winter. We will use an individual's GPS data to place a hair snag and trail camera within its known range to obtain a hair sample prior to hibernation. If the identify of the individual is uncertain, we can confirm it by genotyping DNA extracted from the hair follicles (genotypes will already be known from Mark Vieira's study). Hair snags will be repeated after emergence from the dens in the spring. Measuring hibernation physiology throughout the winter has proven difficult, and frequently is rather invasive, most methods requiring surgery. The majority of metabolic suppression in black bears appears to be independent of lowered core body temperature (Tøien et al. 2011). Also, core and surface body temperatures can differ greatly, possibly due to shivering thermogenesis for muscle maintenance (Harlow et al. 2004). Further, a recent study allowed heart rate monitoring through the year, and found individual differences (Laske et al. 2011). Heart rate then, rather than core body temperature may be a better proxy for metabolic rate. Since we do not currently have the capacity to measure these throughout the winter, the first winter we will use hibernation length as obtained from GPS collar data and ambient temperature as proxies for hibernation physiology. In subsequent winters, pending adequate funding, we will obtain collars with heart rate monitors (or retrofit the current collars) to use as an index of hibernation.

Captive bears

Most studies quantifying telomere attrition rates have been conducted in captivity. Here we will compare rates of telomere loss between captive and wild bears. If telomere dynamics prove to be similar, then captive studies may be representative of wild populations. However, it is well established that captive animals may undergo behavioral and physiological changes (e.g. Morato et al. 2001). We are fortunate to have access to a captive facility that houses black bears on the Front Range, The Wild Animal Sanctuary in Keenesburg, CO. We will work with 12 female bears, 2 each across a range of age classes if available. The facility houses about 70 black bears with relatively well-known histories, so selecting this subset should be possible. Using 2 bears each of similar age will allow for contrasts of telomere length differences within age class. Similar to wild caught bears, we will pull hair 4 times a year, and conduct a full workup while denning. Handling procedures will be the same as those for free-ranging black bears (see Appendix A), but we will not collar these bears, and likely trapping will be unnecessary as they are fairly tame. We will not have movement data on these bears, but being in captivity, their habitat and activity can be well estimated. For the captive bears, we will only include females in the study because males are sterilized, which may have unknown effects on aging.

Analyses (all objectives)

Stable isotope analysis

Bear hair will have follicles clipped off and placed aside for DNA extraction. The remaining hair shaft will be rinsed three times with 2:1 chloroform: methanol solution to remove surface oils (Cryan et al. 2004), dried for 72 hours at 60°C, and homogenized with surgical scissors. Whole blood samples will be dried for 72 hours at 60°C, and homogenized with a spatula. Teeth will undergo collagen extraction by soaking in 32% HCl for 24 hours to remove biogenic carbonates, followed by drying at 60°C for one week, then freeze drying for three days, and homogenized in a ball mill (Mixer Mill MM200, Restch Inc. Newton, PA, USA) (Owen et al. 2011). Diet samples will also be dried for 72 hours at 60°C and homogenized in a ball mill. For ^{13}C and ^{15}N analysis, samples will be weighed, placed in tin capsules and submitted to the Stable Isotope Facility at the University of Wyoming to be analyzed with a Costech 4010 elemental analyzer attached to a Thermo Finnigan DeltaPLUS XP Continuous Flow Isotope Ratio Mass Spectrometer. Results will be provided as per mil (parts per thousand [‰]) ratios relative to the international standards of Peedee Belemnite (PDB; $\delta^{13}C$) and atmospheric nitrogen (AIR; $\delta^{15}N$) with calibrated internal laboratory standards.

Telomere length analysis

Blood and hair samples will be extracted with standard procedures using Qiagen Dneasy tissue extraction kit. We will quantify the relative length of telomeres using real-time quantitative polymerase chain reaction (Q-PCR) (Cawthon 2002). This approach has been found to be highly accurate (Cawthon 2002), in particular for within species comparison (Nakagawa et al. 2004). The method determines relative telomere length by comparing a DNA sample's ratio of telomere repeat copy number (T) to single copy gene number (S) to that of an arbitrary reference DNA. Relative differences in telomere length between individuals then, is exhibited by contrasting the T/S ratio of one individual to that of another. Any single copy gene sequence can be employed for standardization, and we chose to use the single copy gene, 36B4, which was originally employed to develop this method for quantifying telomere length in humans (Cawthon 2002) and subsequently found to be highly suitable for a wide range of species, including black bears (Pauli et al. 2011).

Data analysis

By quantifying the isotopic signature in tissues of bear and that of diet sources, we can quantify the contribution of isotopically distinct items to the diet of the bear. To do this, we will employ isotope mixing models that use Bayesian statistics to quantify the proportional importance of each diet type to bears in the last month (blood), summer (May-Oct) (hair), and lifetime (tooth). Specifically, we will utilize the MixSIR models (Parnell et al. 2010; Layman et al. 2011), which incorporate prior information on variability in isotopic signatures and proportional contributions of sources, resulting in more precise estimates of consumption. We will also explore IsotopeR models (Hopkins and Ferguson 2012), recently developed and validated in Yosemite black bears, which incorporate additional sources of uncertainty and variation. For the hunter-harvested bears, we will compare differences in diet between black bear age-sex groups, and other human and land use covariates with ANOVA-type analyses, and explore dietary shifts within individuals with a GLM approach. For the wild and captive bears, we will also explore dietary changes throughout the year as related to habitat use.

We will then compare differences in telomere length between age groups and individual hunter-harvested and nuisance bears, and explore relationships to other covariates with simple correlations and t-tests. If as expected, more complex modeling is necessary, we will use the Bayesian Network modeling shell Netica (Norsys Software Corp. Vancouver, Canada) to develop and test models with various covariates that all include telomere length (Marcot et al. 2006, Pauli et al. 2011). Telomere length of blood and hair samples will be compared within individuals as well to determine tissue type differences. Stable isotopic signature will be included as a covariate. For free-ranging and captive bears, rates of telomere shortening will be calculated and examined between individuals and seasons, compared with hibernation length and other covariates.

Ultimately, this study is using multiple techniques at two scales: a broad scale single time point sample, and a smaller regional scale with multiple samples. Together, these approaches will illuminate the relative importance of factors influencing diet and aging in Colorado black bears.

Schedule

Activity	Timeline
Sample collection - harvested bears	Fall 2011
Sample collection - nuisance bears	Fall 2011-2012
Trap and collar bears	Summer 2012 (if necessary, 2013)
Hair snags for collared bears	Fall/Spring 2012-2014
Diet sample collection	Summer/Fall 2012-2013
Den visits	Winter 2012-2014
Stable isotope analysis	Winter 2011-ongoing, completed by fall 2013
Telomere analysis	Winter 2011-ongoing, completed by Spring 2015
Project completion	Summer 2015

Literature Cited

Baldwin, R.A., and L.C. Bender. 2009. Foods and nutritional components of diets of black bear in Rocky Mountain National Park, Colorado. Canadian Journal of Zoology 87:1000–1008.

Barrett, E.L.B. and D.S. Richardson. 2011. Sex differences in telomeres and lifespan. Aging Cell 10:913-921.

Baruch-Mordo, S., S.W. Breck, K.R. Wilson, and D.M. Theobald. 2008. Spatiotemporal distribution of black bear-human conflicts in Colorado, USA. Journal of Wildlife Management 72:1853-1862.

Beaulieu, M., Reichert, S., Maho, Y.L., Ancel, A., and R. Criscuolo. 2011. Oxidative status and telomere length in a long-lived bird facing a costly reproductive event. Functional Ecology 25:577–585.

Beckmann, J.P., and J. Berger. 2003. Using black bears to test ideal-free distribution models experimentally. Journal of Mammalogy 84:594-606.

Beckmann, J.P., L. Karasin, C. Costello, S. Matthews, and Z. Smith. 2008. Coexisting with black bears: perspectives from four case studies across North America, WCS Working Paper No. 33, Wildlife Conservation Society, New York, New York.

Beston, J. 2011. Variation in life history and demography of the American black bear. Journal of Wildlife Management 75(7):1588-1596.

Cassidy, A., De Vivo, I., Liu, Y., Han, J., Prescott, J., Hunter, D.J., and E.B. Rimm. 2010. Associations between diet, lifestyle factors, and telomere length in women. The American Journal of Clinical Nutrition 91:1273-80.

Cattan, V., Mercier, N., Gardner, J.P., Regnault, V., Labat, C., Mäki-Jouppila, J. Nzietchueng, R. Benetos, A., Kimura, M., Aviv, A., and P. Lacolley. 2008. Chronic oxidative stress induces a tissue-specific reduction in telomere length in CAST/Ei mice. Free Radical Biology and Medicine 44, 1592–1598.

Cawthon, R. M. 2002. Telomere measurement by quantitative PCR. Nucleic Acids Research 30:e47.

Chauhan, V.P.S., Tsiouris, J.A., Chauhan, A., Sheikh, A.M., Brown, W.T., and M. Vaughan. 2002. Increased oxidative stress and decreased activities of Ca2+/Mg2+-ATPase and Na+/K+-ATPase in the red blood cells of the hibernating black bear. Life Sciences 71:153-161.

Cherif, H., Tarry, J.L., Ozanne, S.E. and C.N. Hales. 2003. Ageing and telomeres: a study into organ- and gender-specific telomere shortening. Nucleic Acids Research 31:1576–1583.

Crawford, K., McDonald, R.A. and S. Bearhop. 2008. Applications of stable isotope techniques to the ecology of mammals. Mammal Review 38:87-107.

Cryan, P., Bogan, M., Rye, R., Landis, G., and C. Kester. 2004. Stable hydrogen isotope analysis of bat hair as evidence for seasonal molt and long-distance migration. Journal of Mammalogy 85:995–1001.

Dunshea, G., Duffield, D., Gales, N., Hindell, M., Wells, R.S., and S.N. Jarman. 2011. Telomeres as age markers in vertebrate molecular ecology. Molecular Ecology Resources 11:225–235.

Gillies, C.S., M. Hebblewhite, S.E. Nielsen, M.A. Krawchuk, C.L. Aldridge, J.L. Frair, D.J. Saher, C.E. Stevens, and C.L. Jerde. 2006. Application of random effects to the study of resource selection by animals. Journal of Animal Ecology 75:887-898.

Greenleaf, S.S., S.M. Matthews, R.G. Wright, J.J. Beecham, and H.M. Leithead. 2009. Food habitat of American black bears as a metric for direct management of human-bear conflict in Yosemite Valley, Yosemite National Park, California. Ursus 20:94-101.

Hall, M.E., Nasir, L., Daunt, F., Gault, E.A., Croxall, J.P., Wanless, S., and P. Monaghan. 2004. Telomere loss in relation to age and early environment in long-lived birds. Proceedings of the Royal Society B 271:1571–1576.

Harlow, H.J., Lohuis, T., Anderson-Sprecher, R.C., and T. D. I. Beck. 2004. Body surface temperature of hibernating black bears may be related to periodic muscle activity. Journal of Mammalogy 85(3):414-419.

Harlow, H.J., Lohuis, T., Beck, T.D.I., and P.A. Iaizzo. 2001. Muscle strength in overwintering bears. Nature 409:997.

Haussmann, M.F., D.W. Winkler, K.M. O’Reilly, C.E. Huntington, I.C.T. Nisbet, and C.M. Vleck. 2003. Telomeres shorten more slowly in long-lived birds and mammals than in short-lived ones. Proceedings of the Royal Society of London Series B 270:1387-1392.

Heidinger, B.J., Blount, J.D., Boner, W., Griffiths, K., Metcalfe, N.B., and P. Monaghan. 2012. Telomere length in early life predicts lifespan. PNAS 109(5):1743-1748.

Hellgren, E.C. 1998. Physiology of hibernation in bears. Ursus 10:467-477.

Hemann, M. T., and C. W. Greider. 2000. Wild-derived inbred mouse strains have short telomeres. Nuclei Acids Research 28: 4474-4478.

Hildebrand, G.V., Farley, S.D., Robbins, C.T., Hanley, T.A., Titus, K. and C. Servheen. 1996. Use of stable isotopes to determine diets of living and extinct bears. Canadian Journal of Zoology 74:2080-2088.

Hobson, K.A. and H.E. Welch. 1992. Determination of trophic relationships within a high Arctic marine food web using stable-isotope analysis. Marine Ecology Progress Series 84:9-18.

Hopkins J.B. III, and J.M. Ferguson. 2012. Estimating the diets of animals using stable isotopes and a comprehensive Bayesian mixing model. PLoS ONE 7(1):e28478.

Hopkins, J.B. III, Koch, P.L., Schwartz, C.C., Ferguson, J.M., Greenleaf, S.S., and S.T. Kalinowski. 2012. Stable isotopes to detect food-conditioned bears and to evaluate human-bear management. Journal of Wildlife Management 76(4):703–713.

Irwin, L.L., and F.M. Hammond. 1985. Managing black bear habits for food items in Wyoming. Wildlife Society Bulletin 13:477–483.

Jahren, A.H., Saudek, C. Yeung, E.H., Kao, W.H.L., Kraft, R.A., and B. Caballero. 2006. An isotopic method for quantifying sweeteners derived from corn and sugar cane. American Journal of Clinical Nutrition 84:1380-1384.

Johnson, M.L. Bait use by black bears (*Ursus americanus*) in Wisconsin: applications of GPS telemetry, GIS, and remote photography. 2007. MS Thesis, University of Wisconsin, Green Bay, Wisconsin, USA.

Kelly, J.F. 1999. Stable isotopes of carbon and nitrogen in the study of avian and mammalian trophic ecology. Canadian Journal of Zoology 78:1-27.

Koizumi, A., Tsukada, M., Wada, Y., Masuda, H. and R. Weindruch. 1992. Mitotic activity in mice is suppressed by energy restriction-induced torpor. Journal of Nutrition 122:1446–1453.

Layman, C.A., Araujo, M.S., Boucek, R., Hammerschlag-Peyer, C.M., Harrison, E., Jud, Z.R., Matich, P., Rosenblatt, A.E., Vaudo, J.J., Yeager, L.A., et al. 2012. Applying stable isotopes to examine food-web structure: an overview of analytical tools. Biological Reviews 87(3):545-562.

Laske, T.G., Garshelis, D.L., and P.A. Iaizzo. 2011. Monitoring the wild black bear's reaction to human and environmental stressors. BMC Physiology 11:13.

Lyman, C. P., O'Brien, R. C., Greene, G. C. and E.D. Papafrango. 1981. Hibernation and longevity in the Turkish hamster *Mesocricetus brandi*. Science 212:668–670.

Marcot, B.G., Steventon, J.D., Sutherland, G.D., and R.K. McCann. 2006. Guidelines for developing and updating Bayesian belief networks applied to ecological modeling and conservation. Canadian Journal of Forest Research 36:3063–3074.

McCarthy, T., and R. Seavoy. 1994. Reducing non-sport losses attributable to food conditioning: human and bear behavior modification in an urban environment. International Conference on Bear Research and Management 9:75-84.

Merkle, J.A., Derbridge, J.J., and P.R. Krausman. 2011. Using stable isotope analysis to quantify anthropogenic foraging in black bears. Human-Wildlife Interactions 5:159–167.

Meyne, J, R. L. Ratliff, and R. K. Moyzis. 1989. Conservation of the human telomere sequence (TTAGGG)n among vertebrates. Proceedings of the National Academy of Sciences 86: 7049-7053.

McGee-Lawrence M.E., Carey, H.V., and S.W. Donahue. 2008. Mammalian hibernation as a model of disuse osteoporosis: the effects of physical inactivity on bone metabolism, structure, and strength. American Journal of Physiology – Regulatory, Integrative and Comparative Physiology. 295(6):R1999-R2014.

Monaghan, P. and M.F. Haussmann. 2006. Do telomere dynamics link lifestyle and lifespan? Trends in Ecology and Evolution 21:47–53.

Morato R.G., Conforti V.A., Azevedo F.C., Jacomo A.T., Silveira L., Sana D., Nunes A.L., Guimarães M.A., and R.C. Barnabe. 2001. Comparative analyses of semen and endocrine characteristics of free-living versus captive jaguars (*Panthera onca*). Reproduction 122(5):745-751.

Munshi-South, J., and G. S. Wilkinson. 2010. Bats and birds: Exceptional longevity despite high metabolic rates. Ageing Research Reviews 9:12-19.

Nakagawa, S., N.J. Gemmell, and T. Burke. 2004. Measuring vertebrate telomeres: applications and limitations. Molecular Ecology 13:2523-2533.

Owen, K., Charlton-Robb, K., and R. Thompson. 2011. Resolving the trophic relations of cryptic species: an example using stable isotope analysis of dolphin teeth. PLoS ONE 6(2):e16457.

Parnell, A.C., Inger, R., Bearhop, S., and A.L. Jackson. 2010. Source partitioning using stable isotopes: coping with too much variation. PLoS ONE 5:e9672.

Pauli, J.N., Whiteman, J.P., Marcot, B.G., McClean, T.M. and M. Ben-David. DNA-based approach to aging martens (*Martes americana* and *M. caurina*). 2011. Journal of Mammalogy 92(3):500-510.

Phillips, D.L., Newsome, S.D., and J.W. Gregg. 2005. Combining sources in stable isotope mixing models: alternative methods. Oecologia 144:520-527.

Raine, R.M., and J.L. Kansas. 1990. Black bear seasonal food habits and distribution by elevation in Banff National Park, Alberta (Victoria, British Columbia). In: Bears: Their Biology and Management, Vol. 8, A Selection of Papers from the Eighth International Conference on Bear Research and Management, 297-304.
Robbins, C.T., C.C. Schwartz, and L.A. Felicetti. 2004. Nutritional ecology of ursids: a review of newer methods and management implications. Ursus 15(2):161-171.

Schroeder, M.A. and L.A. Robb. 2005. Criteria for gender and age. Pp. 303–338 in Techniques for wildlife investigations and management (C. E. Braun, ed.). Wildlife Society, Bethesda, Maryland.

Seluanov, A., Zhuoxun C., Hine, C., Sasahara, T.H.C., Ribeiro, A.A.C.M., Catania, K.C., Presgraves, D.C., and V. Gorbunova. 2007. Telomerase activity coevolves with body mass, not lifespan. Aging Cell 6:45–52.

Shi, Q., Hubbard, G.B., Kushwaha, R.S., Rainwater, D., Thomas III, C.A., Leland, M.M., VandeBerg, J.L., and X.L. Wang. 2007. Endothelial senescence after high-cholesterol, high-fat diet challenge in baboons. American Journal of Physiology- Heart and Circulatory Physiology 292:H2913-H2920.

Spady, T.J., Harlow, H.J. Butterstein, G., and B. Durrant. 2009. Leptin as a surrogate indicator of body fat in the American black bear. Ursus 20(2):120-130.

Tøien, Ø, Blake, J. Edgar, D.M., Grahn, D.A., Heller, C., and B.M. Barnes. 2011. Hibernation in black bears: independence of metabolic suppression from body temperature. 331:906-909.

Turbill, C., Smith, S., Deimel, C., and T. Ruf. 2012. Daily torpor is associated with telomere length change over winter in Djungarian hamsters. Biology Letters 8:304-307.

Turbill, C., Bieber, C. and T. Ruf. 2011. Hibernation is associated with increased survival and the evolution of slow life histories among mammals. Proceedings of the Royal Society B 278.

von Zglinicki, T. 2002. Oxidative stress shortens telomeres. Trends in Biochemical Sciences 27:339–344.

Watson, J.D. 1972. Origin of concatameric T4 DNA. Nature-New Biology 239:197-201.

Appendix A

CAPTURE AND HANDLING PROCEDURES FOR FREE-RANGING BLACK BEARS

Black bears will be initially captured and collared during the summer months and annually re-captured in their dens during winter months to obtain samples for telomere analysis, and reproductive and body condition information.

Summer

We will capture and collar adult black bears during summer months (May-Sept) using cage traps and foot snares. We will use cage traps in areas close to Fort Collins with high human activity, and where there is good road access. Snares will be used for more remote trapping locations, away from human activity and where vehicle access is limited. Once a bear has been captured using either method, field crews will use an identical protocol to process animals.

Cage Traps

We will capture bears with two different trap designs, as specified in Heather Johnson's protocol, a cage trap designed and used extensively by Beck (1993), and a newly designed trap to specifically target female bears. The trap developed by Beck is 1.8 m long and 1.0 m in height and width. The frame is constructed of angle iron, all side and top panels are wire mesh of 1.9 x 1.9 in size, and the trap has a floor that is 16-gauge steel. A spring-powered, solid aluminum door is mounted on a full-length hinge at one end and a latching mechanism holds the door closed. The door is triggered via a treadle pedal on the floor, and a standard garage door coil spring provides closing power. A hinged panel along the back of the trap allows access for administering immobilizing drugs via jabpole. In total, the trap weighs approximately 236 kg. In the first study in which these traps were used, only 1 bear in 134 captures was injured, as the individual broke a canine on the wire mesh.

We will also use a smaller, lighter trap designed by Mat Alldredge, in conjunction with Tom Davies, Lyel Willmarth, Heather Johnson, and others, which discourages the capture of large males and increases portability in the field. These traps were built to be slightly larger than those that have been successfully used for cougars (Alldredge et al. personal communication) and are 34in high, 60in long, and 25in wide. The frame is built with 1x1in heavy gauge steel, covered with 1x1in heavy gauge, high tinsel, steel mesh. The smaller dimensions of the mesh will reduce the possibility that animals will break their teeth on the cage. The sides of the trap have additional braces to increase overall strength and support. The door of the trap comprises one end of the structure and is designed drop and latch to the bottom of the frame. Bait is hung from a cable attached to an archery trigger, and the door falls shut when the trigger is released. Due to the smaller size of the trap, it only weighs approximately 60 kg.

Cage traps will be positioned so they are in the shade, and exposure to sun and precipitation is minimized. All cage traps will be clearly marked with warning signs. Cages will be baited with rotting fish, fruit, or road kill. They will be set in the late afternoon or evening and checked the following morning to minimize the time an animal spends in a trap. If the bear is a cub or yearling (too small for a GPS collar), it will be released without being immobilized. If the bear is an adult, it will be immobilized following procedures described below. Bears will be immobilized with a jabpole, syringe pole, or syringe (hand injection), with the injection targeted into muscle tissue along the shoulder or thigh.

Aldrich Foot Snares

Aldrich foot snares were specifically developed to capture bears and have proven to be safe and effective (Jonkel 1993). The spring activated snare secures a ¼ inch steel cable around the foot of the bear, closing tight with the action of a small piece of angle iron fashioned into a sliding lock mechanism. The inside of the snare loop is wrapped with duct tape to minimize surface abrasion on the skin of the foot. We will modify snares with additional duct tape and/or surgical tubing over the cable to serve as a "cub stopper" such that small bears (cubs and yearlings) have a low probability of being captured (Jonkel 1993). An in-line swivel is placed in the cable to avoid torsion of the foot and a potential bone fracture. A short lead is attached to the snare to further minimize stress to the leg.

The lead is then secured to an anchor tree at least 10 inches in diameter with a ¼ in steel cable clamped and stapled to the base of the tree so the bear cannot climb it. Branches of the tree are lopped off with a saw or axe about 8 ft up, so the bear cannot hang itself from a branch by the snare cable. An area of ≥ 5 meters is cleared around the snare site to eliminate potential that the bear is able to twist the snare loop around any obstacles (saplings, brush, etc). Large branches will be angled over the snare to force ungulates to step over or go around it, minimizing the possibility of catching non-target animals. Additional details of setting snares can be found in Jonkel (1993). A disadvantage of using foot snares is that all bears that are caught (even if they are a male bear or too small to collar) must be immobilized to be released. Other non-target animals that are caught (i.e. mountain lions, coyotes, etc) will be immobilized with Telazol and released. Snares will be set in the evening and checked in the morning, operated when ambient temperatures are between 32 and 90°F. Snared bears will be immobilized using a jabpole or CO_2 dart gun with the injection targeted into muscle tissue along the shoulder or thigh.

Animal Processing

During summer months bears will be anesthetized with butorphanol, azaperone, and medetomidine (BAM), a drug combination that has been successful immobilizing black bears and is reversible with atipamezole (a medetomidine antagonist), allowing a faster and safer release of animals around urban environments (Wolfe et al. 2008). BAM will be administered at a volume of 0.4ml/23kg (50 lbs) with a dosage of 0.26mg/kg for butorphaneol, 0.22mg/kg for azaperone and 0.09mg/kg for medetomidine. We will initially give the recommended dose based on estimated animal weight and boost as necessary by ½ and ¼ of the original dose for the first and second boosters, respectively. To reverse immobilization we will intravenously administer atipamezole. We will dispense a volume of 1ml/1ml at a dosage of 5mg/1mg of medetomidine or 0.45mg/kg. One dose should be sufficient to reverse BAM. Bears immobilized with BAM should not be consumed for 45 days afterward, information which will be printed on collars and ear-tags (see below).

Following the injection of BAM, field personnel will approach and gently prod the bear to ensure that the animal is fully anesthetized, administering additional doses as needed. Once anesthetized, the bear will be removed from the trap or snare and placed in a sternally recumbent position with front and rear legs extended. If the bear will not be collared it will be subcutaneously injected with a passive integrated transponder (PIT) tag and marked with a single black or brown ear-tag that is labeled with the appropriate consumption date information. Afterwards, the bear will be administered atipamezole and released. Adult female bears will be discriminated from subadults based on weight, and nipple size and coloration (Beck 1991).

Adult bears will be fully processed. They will immediately be treated with eye ointment and blindfolded to reduce visual stimuli and protect the eyes from debris and bright light. Throughout the time a bear is anesthetized, its vital signs (heart rate, respiration and temperature) will be monitored. Normal ranges for vital rates of adult bears: heart rate = 60-90 beats/minute, respiration = 15-20 breaths/minute, and temperature = 99.6 - 101.0°F (Jonkel 1993). If a bear's body temperature exceeds the normal range, field staff will cool the underside of the bear with water, particularly the armpits, groin and stomach. If heart rate and respiration values fall outside normal expectations we will reverse the anesthesia and release the bear.

In processing bears, we will check each animal for any lacerations that occurred in the capture process and treat them with topical antibiotics. Additionally, bears will be given an injection of

Oxytetracycline (9mg/lb) or Baytril (7.5 mg/kg) to reduce chances of infection from darting and tooth extraction (described below). Adult bears will be subcutaneously injected with a PIT tag. If the individual has been identified by CDOW Area staff as a "conflict" bear it will be marked in accordance with CDOW Administrative Directive W-2. Individuals will be weighed using a portable spring scale and pulley system and their breeding status will be recorded (lactating, cubs present, evidence of suckling, etc). We will take multiple body size measurements including total length, chest girth and neck girth. During winter months we will also use bioelectrical impedance analysis to measure bear body fat (Farley and Robbins 1994, Hilderbrand et al. 1998). Additionally we will draw blood and collect a hair sample. These samples will be used for telomere and stable isotope analysis. To age captured bears using tooth cementum annuli counts (Stoneberg and Jonkel 1966, Willey 1974), we will remove the first vestigial premolar (or if unavailable the lower first premolar) using a dental elevator. For tooth extraction, we will topically apply Lidocaine and subcutaneously administer Ketofen for analgesia (1cc/100lb). A piece of foam gel will then be placed on the removal site and left for adhesion and filling of the wound.

We will attach a GPS collar (~700 g) with a ~2 year life expectancy. Collars will be programmed to collect ≥4 locations/day, and will be labeled with the appropriate consumption date based on immobilization. The GPS collar will include a VHF transmitter that allows tracking via standard telemetry equipment and the retrieval of collars (we will use both North Star and Vectronic collars). We will recapture each collared bear each winter to assess fecundity and body condition, and take samples for telomere analysis during hibernation. GPS collars will upload the location of each individual every day via a satellite system and the location will be available to researchers in real-time.

When animal processing procedures are completed, the blindfold will be removed and the immobilization reversal will be administered. Field staff will observe the bear from a safe distance to ensure that the animal recovers to a standing position (Wolfe et al. 2008).

Winter

Den Checks

To assess fecundity, body condition, and obtain winter hair and blood samples for telomere analysis, we will recapture collared bears each winter. Bears will be tracked to their dens using GPS collar locations, and researchers will dig through the snow as needed to access the den. Adult bears (and accompanying yearlings) will be anesthetized with Telazol using a jabpole or CO_2 dart gun. Telazol will be administered intramuscularly with a dose of 1.5 – 2.5mg/lb at a lower concentration (5cc at 100mg/ml). Bears will be immobilized at a higher concentration (3cc at 166 mg/ml) if they are particularly agitated or large. We will initially give the recommended dose based on estimated animal weight and boost as necessary by ½ and ¼ of the original dose for the first and second boosters, respectively. Unlike BAM, there is no reversal drug for Telazol. That said, an immobilized bear can be returned to its den for recovery, reducing animal stress and increasing researcher safety.

Once immobilized, bears will be removed from the den, placed on a blanket, and processed in a similar manner to that described above. Field staff will check the fit of the GPS collar and make any necessary modifications, and clean up any neck wounds with saline solution. Newborn cubs in the den will be tucked inside the jacket of a field crew member, next to their body, so that the cub says warm and quiet. After processing, bears will be returned to the den; adults and yearlings will be positioned on their side and newborn cubs will be placed on their mother's back. The den entrance will be covered with sticks and boughs and a layer of snow to discourage the bear from leaving the den. We will retain a small opening in the snow to ensure that the bear has a fresh supply of air (Jonkel 1993).

Injuries and Euthanasia

If an animal is seriously injured (e.g. fractured or broken appendage, vertebrae, pelvis, or jaw, severe dislocation, laceration or any other injury that severely compromises its ability to survive and/or causes severe pain or distress) during capture, it will be quickly and humanely euthanized. Bears will be deeply anesthetized with BAM or Telazol and euthanized via an intravenous potassium chloride (KCl; 400-800 mEq) injection or gunshot to the head or neck. Carcasses that are euthanized will be disposed of in a landfill or left in an area appropriate for scavengers.

Literature Cited

Beck, T.D.I. 1991. Black bears of west-central Colorado. Technical Publication 39, Colorado Division of Wildlife, Fort Collins, Colorado.

Beck, T.D.I. 1993. Development of black bear inventory techniques; job progress report. Colorado Division of Wildlife, Project Number W-153-R-6.

Farley, S.D., and C.T. Robbins. 1994. Development of two methods to estimate body composition of bears. Canadian Journal of Zoology 72:220-226.

Hilderbrand, G.V., S.D. Farley, and C.T. Robbins. 1998. Predicting body condition of bears via two field methods. Journal of Wildlife Management 62:406-409.

Jonkel, J.J. 1993. A manual for handling bears for managers and researchers. Office of Grizzly Bear Recovery, US Fish and Wildlife Service, University of Montana, Missoula, Montana.

Stoneberg, R.P., and C.J. Jonkel. 1966. Age determination of black bears by cementum layers. Journal of Wildlife Management 30:411-414.

Willey, C.H. 1974. Aging black bears from first premolar tooth sections. Journal of Wildlife Management 38:97-100.

Wolfe, L.L., C.T. Goshorn, and S. Baruch-Mordo. 2008. Immobilization of black bears (*Ursus americanus*) with a combination of butorphanol, azaperone, and medetomidine. Journal of Wildlife Diseases 44:748-752.

APPENDIX II

Front-range Cougar Research
2011-2012 & 2012-2013

Stable isotope analyses for reconstructing the diets of large predators in Colorado's Front Range

UW-Wisconsin & Colorado Parks and Wildlife

Wynne Moss
Jonathan Pauli
Mat Alldredge

Research Proposal

Stable isotope analyses for reconstructing the diets of large predators in Colorado's Front Range

(Study Plan for submission to 2011-2012 Mammals Research Report)

Wynne Moss, M.S. student, UW-Madison

Introduction

Understanding the diet composition of large carnivores is of fundamental importance in wildlife ecology and management, as apex predators can exert strong effects on prey populations and ecosystem processes. In multi-prey systems, quantifying prey selection and predation rates can be a major challenge, in part due to the difficulty of accurately reconstructing the diets of cryptic predators (Knopff et al. 2010). Because black bears (*Ursus americanus*) and cougars (*Puma concolor*) are two of the major predators of neonate elk and mule deer, management of Colorado's native ungulate populations would benefit from a greater understanding of the diets of these predators. In addition, the diet composition of bears and cougars is likely to be affected by urbanization and changes in prey density, such as are currently being experienced along Colorado's Front Range. Bears and cougars may shift foraging in response to increasing human density (Kertson et al. 2011, Merkle et al. 2011). As human-derived diet items become available, it is unknown how these sources will affect and be affected by large predators, and how reliance on alternate prey items will affect predator and native ungulate survival. As such, a rigorous examination of cougar and bear diets, especially in relation to human density, temporal and spatial scale, and age-sex classes is integral to minimizing human-wildlife interactions, managing Colorado's carnivores, and predicting how predator-prey relationships may change in the future.

Approaches to diet reconstruction

Diet composition studies on large, wide-ranging predators have historically relied upon scat analyses and kill site investigations to identify the relative importance of different diet items. An important advantage of these approaches is the ability to study temporal resource use, which can vary highly by season. However, both fecal and kill site analyses can create significant biases in estimating diet composition. For instance, kill site investigations may fail to detect the presence of small prey items (Knopff et al. 2009), and are limited to meat, which only represents a small proportion of the total diet for omnivores like bears. More importantly, studies using kill site analyses often require the capture and handling of individuals to assign GPS collars, and therefore can be limited in the number of individuals monitored. Alternatively, though they can be collected non-invasively and in large sample sizes, scat samples cannot be traced back to an individual or even to age-sex classes without the use of molecular techniques. Therefore, studying resource use for the same individual over a temporal or spatial scale can only be accomplished if paired with genotyping. Finally, fecal analyses underestimate the importance of highly digestible food items, like meat, and overestimate indigestible items, like invertebrate chitin (Dickman and Huang 1988, Hewitt and Robbins 1996) and thus can create inaccurate estimates of dietary contributions.

The analysis of naturally occurring stable isotopes has become an increasingly useful tool for ecologists and managers in understanding a myriad of animal behaviors, including dispersal, prey selection, and resource use (Kelly 2000). Dietary analysis using stable isotopes evaluates the ratios of heavy and light isotopes of carbon, nitrogen, and oxygen in tissue samples, and thus avoids the pitfalls of more traditional methods, which fail to detect highly digestible materials and provide only a snapshot of resource use. Additionally, stable isotopes can provide information on foraging at different time periods. For instance, blood reflects the isotopic signature of food consumed over the previous two months (Hilderbrand et al. 1996) and collagen extracted from bone can be used to estimate diet over an individual's lifetime. Hair, which can be collected non-invasively, reflects items consumed during active phases of hair growth. Specifically, underfur represents autumn diet (Jones et al. 2006), while guard hair can be cut into smaller segments to provide a finer temporal scale within the molt (Pauli et al. 2009). By comparing the isotopic values of hair or other tissue to the isotopic signatures of potential diet items, it is possible to quantify the relative importance of each food source.

We plan to develop techniques for stable isotope analysis of Colorado’s two largest predators: black bears and cougars. Such analyses will allow management agencies to non-invasively reconstruct the diets of these two species, and examine covariates that influence foraging behavior and predation along the urban-wildland interface.

Black Bear (*Ursus americanus*)

Black bears are the primary predators of neonate ungulates in many regions of the Rocky Mountains and may significantly impact ungulate population dynamics (Zager and Beecham 2006). Neonatal ungulates represent a pulsed, highly accessible resource to opportunistic carnivores like bears. So much so, that ungulate meat often becomes the primary dietary source of bears during the calving season (Hilderbrand et al. 1999). Although the spring and summer months correspond to a peak in ungulate consumption by black bears (Zager and Beecham 2006), the extent to which bears rely on ungulate meat can strongly vary both geographically and by age-sex group. This variation has not been particularly well characterized, and diet interpretation has been complicated by important methodological differences between studies. Furthermore, few studies have rigorously measured resource use by individual black bears on a fine spatiotemporal scale, although this information would greatly improve our ability to predict ungulate depredation rates by black bears. In fact, only one study (Baldwin and Bender 2009) has investigated the seasonal diets of black bears in Colorado. This study used fecal analysis, and therefore may have underestimated the importance of meat, which is highly digestible, and overestimated the contribution of less digestible materials like plants and insects.

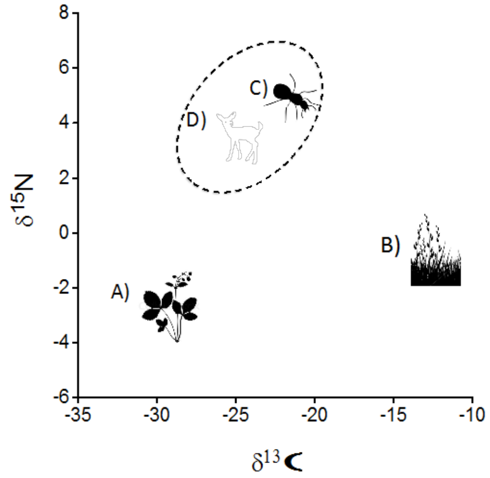


Figure 1.: Illustration of carbon and nitrogen isotopic signatures (‰) of potential diet items: A) C3 plants and berries; B) C4 plants; C) insects (Hymenoptera); D) native ungulates (mule deer and elk). Isotopic values of diet items obtained from preliminary data (C;D) or literature (A: Hobson et al., 2000; B: Merkle et al., 2011)

Table 1. Summary of results for previous studies used to estimate black bear diet in the Intermountain West. Data is presented as average proportion of diet for each food group; ranges are shown parenthetically.

Method	% meat	% plant	% insect
Stable isotope analysis^1	26* (5-54)	58 (25-95)	-*
Fecal analysis^2	8 (0-12)	63 (51-74)	24 (5-37)

1: Stable isotope analyses (*n*=4): Jacoby et al., 1999; Hobson et al., 2000; Fortin, 2011; Merkle et al., 2011
2: Fecal analyses (*n*=4): Irwin and Hammond, 1985; Raine and Kansas, 1990; Bull et al., 2001; Baldwin and Bender, 2009
*Meat cannot be differentiated from insects in stable isotope analyses; meat estimate may include contribution of insects

The diets of black bears in the Western US have also been investigated with stable isotopes; these studies generally detected higher rates of meat consumption than scat analysis (**Table 1**). However, previous isotopic studies relied upon analysis of ^{15}N and ^{13}C in bulk hair samples and, thus, integrated ungulate consumption over an entire period of hair growth (May-October). In reality, black bear diet is strongly seasonal, with predation of ungulates peaking the month after calving (Barber-Meyer et al. 2008). Averaging bear diet across the entire period of hair growth obscures important seasonal patterns in their foraging ecology. Instead, by segmenting guard hair into biologically relevant sections, one could explore seasonal differences in diet for a large number of individual bears, and avoid the need to serially sample bears through an entire year. Such a methodological advancement would be especially relevant to management agencies that are increasingly employing non-invasive sampling programs. However, for this approach to be successful in free-ranging populations, species- and site-specific molt chronologies and rates need to be quantified for black bears in Colorado.

Finally, stable isotope analyses depend upon differences between the isotopic signatures of diet sources; isotopic differences enable the construction of a “mixing space”, which is used to estimate the proportional importance of each diet source (**Fig. 1**). If diet sources do not differ isotopically, their relative importance cannot be distinguished with mixing models. The $\delta^{13}C$ and $\delta^{15}N$ values of insects and ungulates do not significantly differ; consequently, previous studies were unable to differentiate the importance of these items in bear diet (Hobson et al., 2000; Bull et al., 2001; Mattson, 2001; Baldwin and Bender, 2009). This limitation is particularly problematic because insects and ungulates represent the two most important summer diet sources for black bears in the Intermountain West.

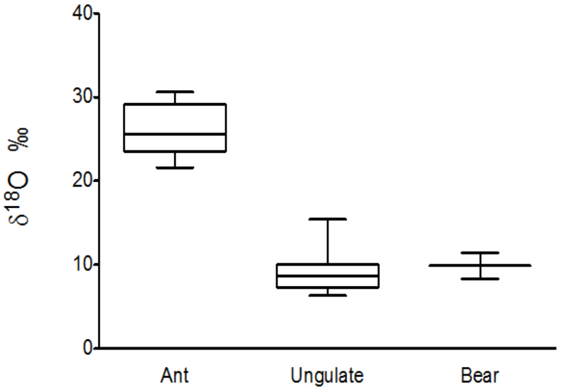


Figure 2.: $\delta^{18}O$ values for ant (*Formica* spp. and *Camponotus* spp.; *n* = 6), ungulate (mule deer and elk; *n* = 8) and black bear (*n* = 2) samples collected in Colorado. Values given in ‰ VSMOW

Though not previously used in black bear diet reconstruction, the analysis of a third isotope, ^{18}O, could distinguish between these two diet sources and allow us to more reliably calculate the seasonal importance of ungulates. Insect chitin and haemolymph are enriched in ^{18}O over atmospheric water due to the loss of the lighter isotope, ^{16}O, during molting (Schimmelmann and DeNiro 1986, Ellwood et al. 2011). Conversely, other potential diet sources, including berries, herbaceous matter, and ungulate meat, do not show an enrichment in oxygen. In a preliminary analysis of ant and ungulate samples collected across the state of Colorado, we found that ants are significantly enriched over ungulates in ^{18}O (**Fig. 2)**. Therefore, the analysis of oxygen in concert with carbon and nitrogen may allow the differentiation of diet sources that were heretofore isotopically indistinguishable, and enable a rigorous quantification of seasonal black bear diet in Colorado. Before applying this to free-ranging black bears, a controlled diet study is needed to quantify the processes affecting ^{18}O assimilation.

Cougar (*Puma concolor*)

Cougars, though capable of preying upon a wide variety of species, generally select for ungulates (Anderson and Lindzey 2003, Knopff et al. 2010). Yet, cougars may also utilize alternate prey, such as smaller mammals and domesticated pets and livestock.

Virtually all data on cougar diet have been derived from scat samples or kill sites and to date, there have been no estimations of diet using stable isotope analysis. As such, it is quite possible that our estimates of cougar prey use are biased towards larger, less digestible items. In regions of increasing human density, where cougars more often prey upon non-ungulates (Kertson et al. 2011), these biases may be more severe. Therefore, along the Front Range in particular, stable isotope analyses could reveal novel, cryptic diet items that have been missed by previous approaches.

Our preliminary analysis of diet items collected from cougar kill sites indicates that prey species possess distinct isotopic signatures (**Fig. 3**). Consequently, we can quantify the proportional importance of these prey items.

Finally, data from kill site investigations and GPS-collars currently being collected by CPW provide a unique opportunity to refine and verify our stable isotope analyses, as well as examine the effects of numerous covariates. By analyzing the stable isotope signature of GPS-collared cougars, we can relate cougar diet composition to individual age, sex, and habitat use. This may provide useful insights into cougar foraging in an urban landscape and may identify age and/or sex classes that are prone to depredation of pets and livestock.

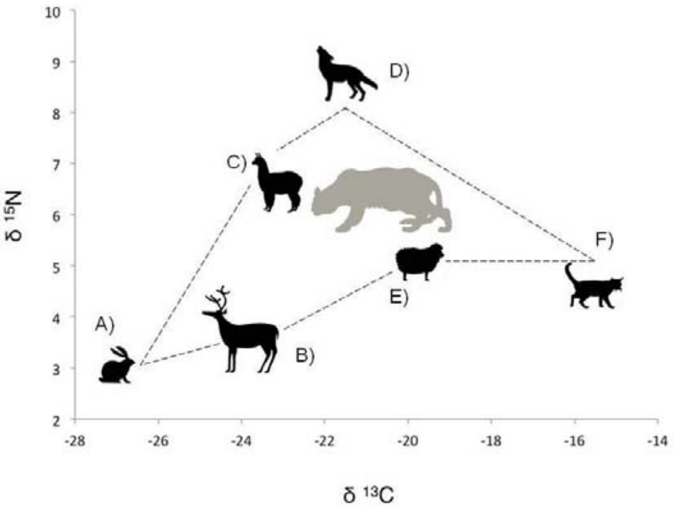


Figure. 3. Illustration of carbon and nitrogen isotopic signatures (‰) of potential diet times: A) rabbits; B) native ungulates (elk and mule deer); C) domesticated ungulates (llamas, alpacas, cows); D) native mesocarnivores (skunks, coyotes, raccoons); E) sheep and goats; F) domestic pets (dogs and cats). Isotopic values of diet items obtained from preliminary data conducted by Pauli (UW) and Alldredge (CPW).

Objectives

1. Quantify the molt chronology and guard hair growth rate of captive black bears. Baseline information on black bear hair growth will enable us to reliably sub-section and analyze guard hairs of bears to evaluate seasonal shifts in their diet.
2. Using captive bears and feeding trials, evaluate the efficacy of carbon, nitrogen, and oxygen isotopes in diet reconstruction. We will also develop correction factors and mixing models that can be used by managers in isotope analyses of free-ranging bears.
3. Develop mixing models to reconstruct diets for collared cougars in CPW's front-range cougar study. Prey composition from kill site investigations will be used to establish prior probabilities for mixing models. In addition, we will compare diet across habitats and age-sex classes to determine what factors influence reliance on domestics or smaller-bodied prey.

Methods

Black bear

Black bears incorporate dietary C, N, and O into hair keratin during the period of molt from early May to October (Jacoby et al. 1999, Hobson et al. 2000); therefore hair isotopic signature reflects diet during this time. However, the stable isotope signature of hair differs from diet due to chemical equilibrium reactions (fractionation) as well as preferential allocation of certain molecules to different tissue types (routing). A controlled feeding study will allow us to quantify the effects of routing and fractionation in bears, since any difference in $\delta^{15}N$, $\delta^{13}C$, and $\delta^{18}O$ values between hair and diet can be attributed to these two processes. By measuring the isotopic signature of hair in relation to a known diet, we can ascertain a correction constant (trophic discrimination factor) to be applied to samples from free-ranging bears; this correction factor has not been previously calculated for black bears. Additionally, there is evidence that isotopic routing and fractionation are influenced by diet and composition of weight gain (Hilderbrand et al., 1999; Voigt et al., 2008; Caut et al., 2009; Newsome et al., 2010), therefore these rates might change temporally with bear nutritional status and diet. We will simulate temporal changes in diet by varying dietary meat concentration during the month when highest meat consumption is expected in the wild. The routing and fractionation rates obtained from this process will be more specific to black bears in late spring to fall and improve the accuracy of our isotopic analyses.

Our controlled feeding program for black bears will run from April-October 2013 at The Wild Animal Sanctuary in Keenesburg, CO. From April to May, all individuals (*n* = 8; **Fig. 4**) will be fed a pelleted high protein omnivore diet. During this time, the isotopic signature of growing hair will equilibrate, ensuring that the isotopic signature of diet is the same for all bears and providing a baseline value to which changes in diet can be compared. In June, we will increase the meat component of the diet to mimic what we expect for the range of free-ranging black bears during ungulate calving. Bears will be divided into four treatment groups: high ungulate diet (*n* = 2; 90% ungulate meat, 10% omnivore diet), medium ungulate diet (*n* = 2; 25% ungulate meat, 75% omnivore diet) or no ungulate diet (*n = 2;* 100% omnivore diet). Ungulate meat will be obtained from road-killed mule deer or elk collected by CPW. The fourth treatment group (*n* = 2) will be fed 100% omnivore diet spiked with 99% atom ^{18}O enriched glycine (Sigma Aldrich; Pauli et al. 2009). Using an ^{18}O-enriched diet will simulate the seasonal pulse in insectivory (**Fig. 2**), and will allow us to measure isotopic discrimination of ^{18}O from diet. Finally, for the remainder of the study, from July to October, all bears will be fed non-enriched 100% omnivore diet. Once per month, from April-October, we will collect hair and blood from each individual for stable isotope analysis. In addition, because drinking water may also be a major source of ^{18}O, we will control access to water sources with a constant ^{18}O signature and monitor δ^{18}O values of water throughout the summer to control for fluctuations in water isotopic signature.

During our controlled feeding study, we will also establish hair growth curves for black bears. Once per month, we will feed all individuals 22 mg/kg of the non-toxic dye rhodamine B. Rhodamine B has proven to be an effective and safe biomarker in a variety of wildlife species wildlife (Fisher 1999, Fry et al. 2010, Palphramand et al. 2011). The dye is incorporated into hair and other keratinous tissue growing at the time of ingestion, forming a distinct band visible under a UV light microscope (Fry et al. 2010). We will collect hair from all individuals and measure the distance between bands to calculate hair growth (mm/day) from April-October. In addition, by measuring from tip to root, we will establish a growth curve that can be applied to non-marked bears, allowing us to sub-sample hair to analyze a segment corresponding to a particular month. Hair growth rates will be modeled with a Gompertz function (Pauli et al. 2009). For free-ranging bears, the parameter estimates can be used to section hair samples into biologically meaningful seasons (e.g., timing of ungulate calving).

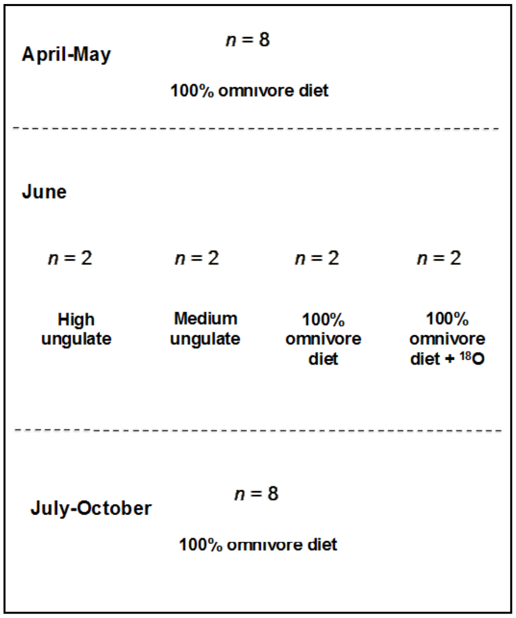


Figure 4. Controlled feeding program for black bears. For all months except June, bears (*n* = 8) will be fed an omnivore diet. In June we will divide bears into four treatment groups to represent varying degrees of ungulate use; in addition, one group will be fed an ^{18}O enriched omnivore diet to determine the efficacy of ^{18}O as a tracer in bear studies.

After the feeding trial is complete, hair in the anagen phase will be plucked from captive bears and segmented according to the growth model developed, with each segment representing the diet during the month in which it was grown. We will separately analyze hair from each month from May-October. After segmenting, all hair samples will be rinsed three times with 2:1 chloroform: methanol solution to remove surface oils (Cryan et al. 2004), dried for 72 hours at 60°C, and homogenized with surgical scissors. For ^{13}C and ^{15}N analysis, samples will be weighed, placed in tin capsules and submitted to the Stable Isotope Facility at the University of Wyoming. Analysis of ^{13}C and ^{15}N levels will be conducted with a Costech 4010 elemental analyzer attached to a Thermo Finnigan DeltaPLUS XP Continuous Flow Isotope Ratio Mass Spectrometer. For ^{18}O analysis, samples will be submitted to the Stable Isotope Facility at the University of Wyoming and analyzed with a temperature conversion elemental analysis attached to a continuous flow Thermo Scientific Delta V mass spectrometer. Diet (omnivore pellets and ungulate meat) and water samples will also be analyzed. Results will be provided as per mil (parts per thousand [‰]) ratios relative to the international standards of Peedee Belemnite (PDB; $\delta^{13}C$), atmospheric nitrogen (AIR; $\delta^{15}N$) and Vienna Standard Mean Ocean Water (VSMOW; $\delta^{18}O$) with calibrated internal laboratory standards. We will compare the isotopic values (in $\delta^{13}C$, $\delta^{15}N$, and $\delta^{18}O$) of all treatment diets to those of sectioned bear hair for each month of our study to calculate the rates of discrimination and timing of nutrient incorporation in black bears. Using the same diet throughout hair growth will allow the quantification of intra- and inter-individual variation and the construction of accurate mixing models. Finally, enriching the diet in ^{18}O will determine whether oxygen is a useful tracer to quantify the diet of black bears

Cougar

From 2011-2012 CPW has collected hair samples from cougars captured along the front-range of Colorado as part of the parent project: *Cougar Demographics and Human Interactions Along the Urban-Exurban Front-range of Colorado* (Alldredge 2008). All required sampling will be done as part of this project and has been approved by CPW's Animal Care and Use Committee. In total, hair samples from approximately 30 cougars are available in conjunction with hair samples from known prey items killed by these individual cougars. These samples will be the foundation of the proposed project.

Fieldwork will concentrate on continuing hair collection of potential cougar diet items from kill sites in Colorado. Species targeted will include mule deer (*Odocoileus hemionus*), elk (*Cervus elaphus*), fox (*Vulpes spp.*), coyote (*Canis latrans*), raccoon (*Procyon lotor*), skunk (*Mephitis mephitis*), mountain cottontail (*Sylvilagus nuttallii*) wild turkey (*Meleagris gallopavo*) and a variety of domestic animals. Specifically, we will collect hair samples from carcasses found while investigating kill sites. We will also collect the following domestic species: alpaca (*Vicugna pacos*), goat (*Capra aegagrus hircus*), cow (*Bos taurus*), donkey (*Eqqus africanus asinus*), llama (*Lama glama*), sheep (*Ovis aries*), horse (*Equus ferus caballus*), cat (*Felis catus*), dog (*Canis lupus familiaris*), and chicken (*Gallus gallus domesticus*).

Cougars exhibit less seasonal variability in diet than bears; therefore, hair will be analyzed in bulk rather than by sub-sections. In addition, potential prey items for cougars can be distinguished by carbon and nitrogen signatures alone (**Fig. 3)** and as such, we do not need to analyze oxygen signatures for either cougars or their prey. For these reasons, a captive study of cougars to examine hair growth rate and oxygen incorporation is not necessary.

In the laboratory, all samples will be dried for 72 hr at 60°C and homogenized in a ball mill (Mixer Mill MM200, Retsch Inc., Newtown, PA, USA). Cougar and prey hair samples will be rinsed three times with 2:1 chloroform : methanol solution to remove surface oils (Cryan et al. 2004), dried for 72 hr at 60°C, and homogenized with surgical scissors. Samples will be weighed, placed in tin (^{13}C and ^{15}N) capsules and submitted to the Stable Isotope Facility at the University of Wisconsin. Analysis of ^{13}C and ^{15}N levels will be conducted with a Costech 4010 elemental analyzer attached to a Thermo Finnigan DeltaPLUS XP Continuous Flow Isotope Ratio Mass Spectrometer. Results will be provided as per mil (parts per thousand [‰]) ratios relative to the international standards of Peedee Belemnite (PDB; $\delta^{13}C$) and atmospheric nitrogen (AIR; $\delta^{15}N$) with calibrated internal laboratory standards.

By quantifying the isotopic signature of prey items (i.e., dietary sources), and consumers (i.e., cougars) one can quantify the contribution of each isotopically distinct diet item to the consumer. To that end, we will use a Bayesian-based approach to quantify the proportional importance of each prey type to individual cougars. Specifically, we will employ the MixSIR models (Parnell et al. 2010, Layman et al. 2011), which are based on a series of linear equations that utilize Bayesian statistics to identify proportional contribution of source pools to a diet. Most importantly, MixSIR models incorporate prior information on the variability in isotopic signatures and the proportional contributions of sources. As a consequence, such Bayesian-based models can substantially narrow the reported ranges of source pool contributions to consumers and result in more precise estimates of prey consumption. We will explore how cougar diet differs between age-sex groups and across habitat types. Specifically, we will examine differences in cougar diets associated with distance from urban areas and in relation to human density.

Anticipated Benefits

This project will be the first to assess the utility of oxygen isotopes in diet reconstruction of free-ranging wildlife. We will use captive feeding trials to develop novel methods for analyzing the stable isotopes of carbon, nitrogen, and oxygen in black bears. Predation of ungulates is an increasing concern for sportsmen and stakeholders in Colorado, and is likely to become a focus of CPW studies. Through the use of stable isotopes and segmented guard hairs, this study could provide managers with a new method for quantifying the importance of ungulate prey items at a fine temporal scale for a large sample size. If successful, the use of oxygen isotopes in mammalian diet reconstruction could be applicable to other carnivores and management settings.

This study will also be the first to use isotope analysis to reconstruct cougar diet. Such an approach can identify potentially important prey items that may have been previously overlooked in kill site and scat analyses. In addition, the use of GPS collar data will enable us to test assumptions about the importance of different prey items across an increasingly urbanized landscape. This increased understanding could have important implications for the mitigation of human-wildlife interactions and the conservation of cougars in North America.

Finally, the novel approaches developed herein can be applied to other populations across the state, and may additionally be applicable to other carnivores and cryptic predators. Ultimately, these models can provide managers with additional tools to non-invasively investigate the foraging ecology of large carnivores.

Literature Cited

Anderson, C.R., and F.G. Lindzey. 2003. Estimating cougar predation rates from GPS location clusters. Journal of Wildlife Management 67:307–316.

Baldwin, R.A., and Bender, L.C. 2009. Foods and nutritional components of diets of black bear in Rocky Mountain National Park, Colorado. Canadian Journal of Zoology 87:1000–1008.

Barber-Meyer, S.M., L.D. Mech, and P.J. White. 2008. Elk Calf Survival and Mortality Following Wolf Restoration to Yellowstone National Park. Wildlife Monographs 169:1–30.

Bull, E., Torgersen, T., and Wertz, T. 2001. The importance of vegetation, insects, and neonate ungulates in black bear diet in northeastern Oregon. Northwest Science 75:244–253.

Caut, S., E. Angulo, and F. Courchamp. 2009. Variation in discrimination factors ($\delta^{15}N$ and $\delta^{13}C$): the effect of diet isotopic values and applications for diet reconstruction. Journal of Applied Ecology 46:443–453.

Cryan, P., M. Bogan, R. Rye, G. Landis, and C. Kester. 2004. Stable hydrogen isotope analysis of bat hair as evidence for seasonal molt and long-distance migration. Journal of Mammalogy 85:995–1001.

Dickman, C.R., and C. Huang. 1988. The Reliability of Fecal Analysis as a Method for Determining the Diet of Insectivorous Mammals. Journal of Mammalogy 69:108–113.

Ellwood, M.D.F., R.G.W. Northfield, M. Mejia-Chang, and H. Griffiths. 2011. On the vapour trail of an atmospheric imprint in insects. Biology Letters 7:601–604.

Fisher, P. 1999. Review of using Rhodamine B as a marker for wildlife studies. Wildlife Society Bulletin 27:318–329.

Fortin, J.K. 2011. Niche separation of grizzly (*Ursus arctos*) and American black bears (*Ursus americanus*) in Yellowstone National Park. Dissertation, Washington State University.

Fry, T.L., T. Atwood, and M.R. Dunbar. 2010. Evaluation of rhodamine B as a biomarker for raccoons. Human-Wildlife Interactions 4:275–282.

Hewitt, D.G., and C.T. Robbins. 1996. Estimating grizzly bear food habits from fecal analysis. Wildlife Society Bulletin 24:547–550.

Hilderbrand, G., S. Jenkins, C. Schwartz, T. Hanley, and C. Robbins. 1999. Effect of seasonal differences in dietary meat intake on changes in body mass and composition in wild and captive brown bears. Canadian Journal of Zoology 77:1623–1630.

Hilderbrand, G.V., S.D. Farley, C.T. Robbins, T.A. Hanley, K. Titus, and C. Servheen. 1996. Use of stable isotopes to determine diets of living and extinct bears. Canadian Journal of Zoology 74:2080–2088.

Hobson, K.A., B.N. McLellan, and J.G. Woods. 2000. Using stable carbon (δ^{13}C) and nitrogen (δ^{15}N) isotopes to infer trophic relationships among black and grizzly bears in the upper Columbia River basin, British Columbia. Canadian Journal of Zoology 78:1332–1339.

Irwin, L.L., and Hammond, F.M. 1985. Managing black Bear Habits for Food Items in Wyoming. Wildlife Society Bulletin 13:477–483.

Jacoby, M., G. Hilderbrand, C. Servheen, C. Schwartz, S. Arthur, T. Hanley, C. Robbins, and R. Michener. 1999. Trophic relations of brown and black bears in several western North American ecosystems. Journal of Wildlife Management 63:921–929.

Jones, E.S., D.C. Heard, and M.P. Gillingham. 2006. Temporal variation in stable carbon and nitrogen isotopes of grizzly bear guardhair and underfur. Wildlife Society Bulletin 34:1320–1325.

Kelly, J.F. 2000. Stable isotopes of carbon and nitrogen in the study of avian and mammalian trophic ecology. Canadian Journal of Zoology 78:1–27.

Kertson, B.N., R.D. Spencer, and C.E. Grue. 2011. Cougar Prey Use In A Wildland–Urban Environment In Western Washington. Northwestern Naturalist 92:175–185.

Knopff, K.H., A.A. Knopff, A. Kortello, and M.S. Boyce. 2010. Cougar Kill Rate and Prey Composition in a Multiprey System. Journal of Wildlife Management 74:1435–1447.

Knopff, K.H., A.A. Knopff, M. B. Warren, and M. S. Boyce. 2009. Evaluating Global Positioning System Telemetry Techniques for Estimating Cougar Predation Parameters. Journal of Wildlife Management 73:586–597.

Layman, C.A., M.S. Araujo, R. Boucek, C.M. Hammerschlag-Peyer, E. Harrison, Z.R. Jud, P. Matich, et al. 2011. Applying Stable Isotopes to Examine Food-web Structure: An Overview of Analytical Tools. Biological Reviews 87:545-562.

Mattson, D.J. 2001. Myrmecophagy by Yellowstone grizzly bears. Canadian Journal of Zoology 79:779–793.

Merkle, J.A., J.J. Derbridge, and P.R. Krausman. 2011. Using stable isotope analysis to quantify anthropogenic foraging in black bears. Human-Wildlife Interactions 5:159–167.

Newsome, S.D., G.B. Bentall, M.T. Tinker, O.T. Oftedal, K. Ralls, J.A. Estes, and M.L. Fogel. 2010. Variation in δ^{13}C) and δ^{15}N) diet-vibrissae trophic discrimination factors in a wild population of California sea otters. Ecological Applications 20:1744–1752.

Palphramand, K.L., N. Walker, R.A. McDonald, and R.J. Delahay. 2011. Evaluating seasonal bait delivery to badgers using rhodamine B. European Journal of Wildlife Research 57:35–43.

Pauli, J.N., M. Ben-David, S.W. Buskirk, J.E. DePue, and W.P. Smith. 2009. An isotopic technique to mark mid-sized vertebrates non-invasively. Journal of Zoology 278:141–148.

Parnell, A.C., R. Inger, S. Bearhop, and A.L. Jackson. 2010. Source Partitioning Using Stable Isotopes: Coping with Too Much Variation. Plos One 5.

Raine, R.M., and Kansas, J.L. 1990. Black Bear Seasonal Food Habits and Distribution by Elevation in Banff National Park, Alberta (Victoria, British Columbia). In: Bears: Their Biology and Management, Vol. 8, A Selection of Papers from the Eighth International Conference on Bear Research and Management:297-304.

Schimmelmann, A., and M.J. DeNiro. 1986. Stable isotopic studies on chitin. III. The D/H and $^{18}O/^{16}O$ ratios in arthropod chitin. Geochimica Et Cosmochimica Acta 50:1485–1496.

Voigt, C.C., K. Rex, R.H. Michener, and J.R. Speakman. 2008. Nutrient routing in omnivorous animals tracked by stable carbon isotopes in tissue and exhaled breath. Oecologia 157:31–40.

Zager, P., and J. Beecham. 2006. The role of American black bears and brown bears as predators on ungulates in North America. Ursus 17:95–108.

APPENDIX III

Front Range Cougar Research
Winters, 2011–2012 & 2012–2013

PUMA FORAGING IN AN URBAN TO RURAL LANDSCAPE

CSU & Colorado Parks and Wildlife

Kevin Blecha
Randy Boone
Mat Alldredge

Research Summary

June 30, 2012

PUMA FORAGING IN AN URBAN TO RURAL LANDSCAPE

Kevin Blecha, M.S. student, CSU

Introduction

The Rocky Mountain Front Range of Colorado has experienced drastic increases in human population, and a surge of suburban and exurban landscapes are sprawling into areas occupied by cougar (*Puma concolor*). Some evidence suggests that cougar show avoidance of these areas of high human density. However, cougar use of human developed landscapes does occur at some level and thus conflicts arise between cougars and humans. This study examines cougar predation characteristics and prey selection in reference to landscape features such as prey availability, anthropogenic development, and hobby livestock.

Long term objectives:

1) *Examine cougar selection of feeding sites in relation to these main variables:*
    a. *Human density/activity*
    b. *Prey availability*
    c. *Hobby livestock availability*
2) *Examine cougar dietary compositions and kill rates in relation to:*
    a. *Individual cougar characteristics (i.e. sex/age)*
    b. *Landscape characteristics*

A current paradigm in cougar management revolves around the idea that cougar populations may not be sustained without ungulate prey (CMGWG 2005). Exurban and suburban landscapes of the Front Range are relatively free of human hunting pressure, which is possibly linked to elevated levels of cougar's primary ungulate prey (mule deer [*Odocoileus hemionus*]). Cougars may be drawn to these areas because they are more likely to increase their encounter with deer, as landscape features used by a primary prey species may be the primary driver for selection of feeding locations of cougar (Pierce et al. 1999, Pierce et al. 2000, Atwood et al. 2007). However, contrary to the idea that increased cougar use of a landscape is a function of increasing prey availability, other recent studies have found that cougar exhibit avoidance to/select against areas of high human activity (Mattson 2007, Burdett et al. 2010, Kertson 2011). Therefore, it is unclear which primary factor may drive landscape use by cougar in the Colorado Front Range. Many studies on other vertebrate species point out that an animal forages optimally (MacArthur and Pianka 1966, Emlen 1966), in which it may sacrifice hunting in areas with high foraging availability for the security provided by areas further away from risks (Willems and Hill 2009). However, whether or not cougar forage optimally in reference to prey availability and human disturbance factors is untested. Testing whether the likelihood of cougar feeding events on the landscape changes in various combinations of low/high prey encounter probability and low/high human disturbance levels, may shed light on: 1.) Whether or not cougar are feeding in exurban areas based on high availability of prey. 2.) The degree of optimal foraging behavior in cougar.

Cougar have the ability to prey on all species of livestock, but the highest losses in Colorado occur in commercial sheep ranching. In the Front Range region however, hobby livestock depredations represent a majority of the owner losses. Hobby livestock owners inhabiting the sprawling exurban and developing rural areas of the Front Range that live in or adjacent to highest suitable cougar habitat are at the highest risk of experiencing a hobby livestock depredation (Torres et al. 1996, Michalski et al. 2006). When a cougar is observed or found on property containing livestock, that cougar may be wrongly accused or suspected of hunting livestock as prey. Protection of livestock, including hobby livestock, is enough justification for wildlife managers/livestock owners to destroy the cougar. It is unknown whether or not cougar, while hunting, select for areas with hobby livestock or whether cougar hunt on ranched landscapes selectively or opportunistically. Detailed information on whether or not certain classes (sex/age) of cougar are more likely to seek prey near hobby livestock is important for predicting which type of cougar may be more likely to commit a depredation offense. Knowing whether cougar that have committed a livestock depredation in the past are more likely to hunt near properties containing hobby livestock will shed light on whether or not individual cougar may behave as specialist toward livestock prey items.

Understanding what biological and environmental factors influence cougar predation is important to the management of cougar and the subsequent prey species. It has been hypothesized that stimuli from human disturbances may increase energetic costs (Frid and Dill 2002), thus a decrease in fitness may occur through decreased mating opportunities (Schoener 1971, Pyke et al. 1977) or through lowered survival of offspring. If human activities increase an animal's search time for acquiring food, through direct disturbances or alterations in landscape configuration, the energetic demands are increased, and thus changes in foraging characteristics may reflect the disturbance/alteration (Gill and Sutherland 2000, Blumstein et al. 2005). Kertson (2010) did find a shift in prey composition in residential areas toward higher proportions of smaller and/or domestic prey. In addition, cougars are known to show individual differences in predation characteristics based on sex, age, and reproductive status (Ackerman et al. 1986, Murphy 1998, Laundre 2005, Laundre 2008, Cooley et al. 2008, Knopff et al. 2010). To assess how different landscapes, seasons, and individual cougar differences influence prey consumption, I will examine characteristics of cougar dietary composition/overlap and feeding rates.

Segment Objectives:

1. Advance model-based methods for identifying feeding events/locations from GPS cluster data.
    a. Assess proportion of sites representing feeding events for various cluster types
    b. Assess prevalence of scavenging in feeding events
    c. Assess prey composition by various cluster types

2. Develop a fine scale dataset for depicting the distribution of primary cougar prey in relation to general habitat and human density/activity
    a. Deploy camera traps across gradients of:
        i. General habitat conditions
        ii. Human density
        iii. Distance to house
3. Develop a thematic map of hobby livestock presence/absence
4. Compare species composition (frequency of occurrence) of cougar diets and compare predation rates on mule-deer and alternative prey items
    a. Assess species composition of cougar diets
    b. Assess use of ungulate prey by prey sex/age
    c. Assess seasonal differences in prey usage

Methods

This study is an extension of a parent project: *Cougar Demographics and Human Interactions Along the Urban-Exurban Front-range of Colorado* (see elsewhere in this annual report) project initiated by the Colorado Division of Wildlife (now Colorado Parks and Wildlife [CPW]), which is charged with managing Colorado's cougar population. Conflicts between cougar and humans have increased dramatically in the past two decades, thus the FRCP was initiated to address questions regarding cougar natural history, population estimation, response to aversive conditioning, response to relocation, livestock depredation opportunity, and predator/prey relationships.

The 2862 km^2 extent of the study area, shown in Figure 1, encompasses the foothill/mountainous regions of Boulder County, north Jefferson County, and portions of Clear Creek, Gilpin and Larimer Counties. This area is characterized by a patchwork of private and publicly owned land held by federal, state, and municipal governing agencies. However, if a subject leaves the study area, standard GPS tracking and field data will be collected on the subject until it establishes what appears to be a maintained home range. All objectives listed below require using cougar fitted with GPS radio collars, and thus only subjects captured in the parent project (See elsewhere in this annual report) are utilized in this project.

Objective 1(Advance model-based methods for extracting feeding events/locations from GPS cluster data)

Many future analyses conducted in this project require determining the exact location and timing (±30 meters) of feeding/predation events from a sample of cougar subjects. Deploying GPS radio collars on the sample of cougars allows us to study predator-prey relationships. The collars collect GPS locations (a.k.a: fixes or points) 7 to 8 times/day, at 3 or 4 hour intervals. GPS locations are classified into clusters (groups of GPS locations), based on the spatial and temporal relationships of a GPS location to other GPS locations (Anderson and Lindzey 2003). These GPS clusters are then classified into selection sets based on the likelihood of the set of locations (clusters) representing a kill site (Alldredge et al. 2008), thus providing a sound sampling frame from which statistical inferences can be made about GPS clusters that are not physically investigated.

To identify unique clusters of GPS locations, a rule-based clustering algorithm was written in Visual Basic and was designed to run within ArcGIS (Alldredge and Schuette, CDOW unpubl. Data 2006). The algorithm was designed to identify clusters in five selection sets (S1, S2, S3, S4, and S5), in order to stratify cluster investigation efforts over a range of different GPS location characteristics for each collared cougar and specified time period (1 month intervals). S1 clusters consists of ≥2 GPS locations within 200 meters and within a 4 day window. To help account for missing GPS fixes; S2 clusters consist of any two consecutively collected GPS points, separated by a range of 200-500 meters, but are missing the scheduled GPS fix in between the two points. To account for the potential that a cougar may feed quickly or to have a short handling time, S3, S4, and S5 clusters were created to sample locations collected along presumed travelling paths. S3 clusters are any two locations within a range of 200 – 500 meters, while S4 clusters are any two locations separated with a range of 500-1000 meters. S5 cluster types are any single GPS location separated ≥1000 meters from any other GPS location. In addition to the spatial and temporal criteria, ≥1 of the points comprising the cluster must have been collected during the night-time.

The study period is divided into monthly sub-periods, and sampling is conducted throughout each month. This continuous monitoring was implemented so that a large temporal continuum of conditions (i.e. changes in season, weather, human activities) can be accounted for. For each monthly sub-period, and by each individual cougar subject, we ran the clustering algorithm script on the GPS locations retrieved from the collar. For each cluster of GPS points a random number was assigned so that a sampling frame could be created for each of the five cluster selection sets (S1, S2, S3, S4, S5). For each month and each cougar, the top two random S1 clusters, top S2 cluster, top S3 cluster, top S4 cluster, and top S5 cluster are visited by an investigator. In some months, S2, S3, or S4 cluster types are not created, and thus were not represented in every combination of month and cougar.

S1 cluster types have spatial and temporal attributes very similar to the way GPS clusters were defined in previous published research on cougar predation. These studies identified clusters as ≥2 GPS points (fix frequency of 3-4 hour intervals) within a 1-2 day period (Anderson and Lindzey 2003) or ≥2 GPS points (3 hour intervals) within a 6 day period (Knopff et al. 2009) within 200 m. White et al. (2011) identified clusters as ≥2 GPS points within 100 m recorded within a 1-2 day period, and then (Kertson et al. 2011) identified clusters as ≥3 locations within 100 m during a 24 hour period. Unique to our study is the effort placed on non-S1 cluster types (S2, S3, S4, S5 types).

Randomly sampled clusters are investigated by trained biologists to determine the probable action carried out by the cougar (feeding/resting) while at the site. If evidence of feeding activities is detected (presence of carcass remains), then we determine:

1. Species, sex, and age of prey remains
2. Whether the focal cougar actually killed the animal

We used the following protocol to investigate cougar GPS clusters in the field: For S1 clusters, we investigate each cougar GPS location in the cluster by spiraling out a minimum of 20 m from the GPS waypoint while using the GPS unit as a guide, and visually inspecting overlapping field of view in the area for prey remains. Normally this is sufficient to detect prey remains and other cougar sign, (e.g., tracks, beds, latrines) associated with cougar. If prey remains are not detected within 20 m radius of cluster waypoints, then we expand our search to a minimum of 50 m radius around each waypoint. For S2, S3, S4 and S5 clusters, we visit each cougar GPS location and spiral out to a maximum of 50 m around each waypoint, while using the GPS unit as a guide. Depending on the number of locations, topography, vegetation type and density, we spend a minimum of 1 hour and up to 3 hours per cluster to judge whether the cluster was a feeding site.

For future analysis, we will follow methods of Anderson and Lindzey (2003) and then improved upon by Knopff et al. (2009). First, clusters investigated in the field are classified by the presence (1) or absence (0) of feeding evidence. Predictive logistic regression models or mixed effects regression models will be created using the presence/absence data as dependent variables, while a suite of attributes will be used as independent variables (Table 1). If sufficient data is available, regression models will be created on an individual cougar basis in order for the model to be reapplied to the un-sampled clusters of that particular cougar. If little variation exists between subjects, then a single model or fewer models (potentially by cougar sex/age/gender) may be implemented.

For this report, three preliminary analyses were conducted to explore characteristics of GPS cluster locations and their abilities to identify feeding events. For all analyses, a sample unit is defined as an individual cougar in order for the variance to represent inter-subject variability. Simple descriptive statistics (mean, standard deviation, confidence intervals) were then calculated across a sample of cougars with sufficient amount of data.

These analyses were:

1. Assess the proportion of randomly sampled clusters sites that represent feeding sites, for each cluster type (S1 – S5).
2. Assess the proportion of feeding sites that are known scavenging events. These known scavenging events are identified from clusters visited by field investigators who determined that the prey cause of death was from a hunter, vehicle collision, deposition of carcass, bait site, or non-focal lion (feeding site was visited by another GPS collared lion prior to arrival of subject cougar). Because the cause of death is not always obtainable, this scavenging frequency is a minimum estimate.
3. Assess the composition of prey resulting from S1 cluster types versus Non-S1 cluster types (S2 – S5's). It is anticipated that the non-S1 cluster types may be useful for identifying feeding events on smaller prey items. An assumption is that smaller prey items have shorter handling times, and thus a cougar can easily "dine and dash" within the time period between two consecutive GPS locations recorded by the collar. Prey species were grouped into three prey classes: Deer, Elk, and Non-Cervid.

Objective 2 (Development of prey distribution in relation to general habitat and human density/activity)

In order to answer questions related to cougar selection of prey, a measure of fine-scale prey availability must be derived. Detailed spatial and temporal prey availability data is not attainable for the large spatial and temporal extent of the study area/period, as obtaining abundance estimates for even conspicuous animals is difficult in the exurban areas of the Front Range [i.e., deer (CDOW 2006)]. Therefore, an array of 131 camera trap units (Reconyx HyperFire, Holmen, Wisconsin) were distributed throughout the study area to sample encounter rates of prey across the various landscape types. Estimated photographic rates will be interpreted as the probability of encountering a particular prey species, rather than a density or abundance metric. Royle and Nichols (2003) show that heterogeneity in the detection probability parameter of a typical occupancy modeling framework (MacKenzie et al. 2002) is usually most dependent on underlying localized abundance of a surveyed site, especially if all other variables influencing detectability are accounted for. Using camera traps to derive repeated presence-absence data is a novel approach at deriving detection probability estimates that are less influenced by variables other than the localized abundance of a targeted species at a site. Camera traps are less likely to be influenced by observers or sight-ability as the detection of a subject is automated (O’Brien 2010). Encounter rates derived from camera traps may be subject to heterogeneity across ambient temperatures, seasons, species, and body mass of a targeted animal (Rowcliff et al. 2011), but changes in encounter rates between camera traps/sites reflect relative changes in abundance if we assume that detection probabilities are constant among these camera traps/sites (O’Brien 2010). In addition, previous work has shown correlations between camera trapping rates and abundance measures in various ungulate studies (O’Brien et al. 2003, Rowcliffe et al. 2008, Rovero and Marshal 2009). Measures taken to limit inter-site heterogeneity in detection probability include blocking study periods into shorter discrete seasons, in order to account for differences in ambient temperatures, movement behaviors, and animal congregation behaviors (e.g.: seasonal grouping of deer) (Rowcliffe et al. 2011). This study will not attempt to make cross-species comparisons, as the ability to account for inter-species detection heterogeneity is more difficult as movement characteristics from species to species are unique (Efford and Dawson 2012).

Camera-trap photograph encounter rates (number of independent photographs per unit time), for each particular prey species of interest, will be measured on a localized scale (25x25 m grid resolution). This high resolution scale was chosen as it fits the fine scale upon which cougar may make decisions regarding hunting and feeding locations, especially considering cougar are shown to select for edge habitats when killing deer (Laundre and Hernandez 2003). Sunquist & Sunquist (1989) suggest that most large stalking felid species must approach within 30 m of a prey item before attacking. Past work characterizing cougar hunting habits in relation to habitat edge, characterize “edge habitats” as a distance band 15-20 m from the interface of two habitat types (Altendorf et al. 2001, Holmes and Laundre 2006). This high resolution was also chosen based on the resolution of the readily available major land-cover data. A ground-truthed land-cover dataset from the Colorado Vegetation Classification BASINWIDE project (CDOW 2003) was chosen for representing major vegetation types. The temporal extent of this study is approximately 1 year (December 2011 – December 2012). The spatial extent of this study consists of Boulder County, Gilpin County, northern Jefferson County, and Clear Creek Counties of the Front Range region of Colorado (Figure 1). The study area extent was chosen to reflect a majority of the home ranges inhabited by cougars fitted with GPS collars.

To gather sighting data used to calculate encounter rates, camera traps were placed on a stratified random sample of 25 m grid cell sites (n = 131). Sites are defined by single 25x25 m cells, delineated with the boundaries of the 25 m grid cells used in the BASINWIDE Colorado Vegetation Classification Project (CDOW 2003). Because there is potential to model a variety of species potentially preyed upon by cougar, each with differing movement and habitat selection patterns, sites chosen for surveys were randomly placed (Kays et al. 2010, Harmsen et al. 2010, O’Brien et al. 2010). This is particularly important in multi-species assessments, as placing cameras in habitats targeting certain species with low detection probabilities (as commonly done) may violate assumptions, thus causing biased results (Tobler et al. 2008). A stratified random design was utilized in which six major land-cover types, three housing density levels, and three levels characterizing the proximity to houses are represented (Table 2). Not all combinations of strata are present within the study area. Some of these categories may eventually be measured as continuous variables when included in final analysis, and thus these levels and strata are only used to guide the placement of cameras to ensure broad and even sampling across a range of possible habitat conditions.

Placement of the camera unit within the 25 x 25 m site was chosen by randomly generating a point location and azimuth (0-359º). In forested habitats, or habitats providing a stable structure for mounting a trail camera, the unit was placed on the tree/structure closest to the randomly generated point. In sites not providing a suitable mounting location, cameras were placed on a stake (Figure 2). Some pruning of shrubbery/branches was permitted if maximum visibility was limited and if no more than 10% of the cameras detection zone was obstructed. If maximum visibility range of the camera sensor was limited, and pruning was not an option, the cameras direction was adjusted to a new randomly chosen azimuth. If no alternative azimuth was available because of complete 360 º obstruction, then the camera was moved to an alternative random location within the 25x25 m cell. Trail cameras were elevated ~50 cm from the ground to standardize the angle and viewing range of the infrared sensor and/or camera lens. However, camera heights were slightly modified to accommodate snow accumulations and growth of low lying vegetation. Cameras were positioned so that the unit is parallel with the contour of the ground while the planar detection zone is perpendicular to the ground. Camera units were programmed to record a burst of 5 pictures (1 picture/second) upon triggering, with a quiet period of 30 seconds between triggers. Care was taken to have cameras placed so that vegetation movements in the wind will not give false triggers, as false triggers will consume memory and battery life.

A General Linearized Modeling technique will be used to model the encounter rates of each particular prey species across un-sampled sites of the study area, given *a-priori* selected landscape covariate data such as major land-cover (BASINWIDE vegetation data set), elevation, aspect, hydrology, NDVI, edge proximity, etc. A distribution map of predicted encounter rates for each of the prey species, for each month, will be used to infer spatial relative encounter rate estimates. Relative encounter rate estimates across species may not be readily compared using this technique unless efforts are made to assess the probability of detection among targeted species. Particular focus, sampling effort, and analysis time may be placed on the late winter period and late summer periods. The late winter period (i.e. March – May) is of special interest as this is a period of relative stability in ungulate behaviors, as well as the presumed lowest period of prey availability for cougar. The late summer period (August-Sep), which will initiate after the ungulate birthing pulse, will represent a period of relatively stable ungulate behavior and highest presumed prey availability. Significant covariates with high predictive capabilities will be used to interpolate encounter rates at other non-sampled 25 m cells across the study area, for each monthly time period of interest, for each of the six most common prey species [Mule deer, elk (*Cervus elaphus)*, raccoon (*Procyon lotor*), housecat (*Felis catus*), red fox (*Vulpes vulpes*), coyote (*Canis latrans*)] of cougar on the front range. Study period lengths and encounter rate definitions (i.e.: change photographs/day to photographs/week) may be manipulated to simplify calculations and modeling. Ultimately, whichever statistical modeling technique is used, the metric shall be interpreted as the rate of encountering a prey item at that given cell on the landscape within the monthly time period of interest.

Objective 3 (Cougar selection of hunting areas near hobby livestock)

Formal knowledge on the distribution of hobby livestock of the Front Range does not exist. This will be countered by creating a thematic presence/absence map of all parcels of land containing hobby livestock items. Any parcel of land with the confirmed presence of hobby livestock items will be verified through roadside observations of all private land containing evidence of hobby livestock enclosures. Information regarding hobby livestock presence/absence in the individual parcels may be also gathered from:

- Knowledge from CPW staff working in the study area.
- Knowledge from collaborating agency staff in study area.
- Communications with local residence and livestock owners.

- Specific CPW wildlife/livestock conflict reports.
- Kill-site investigators' knowledge of vicinity of any visited cougar GPS location cluster.

Road-side observations and personal landowner visitations may be conducted to verify any presence/absence data collected above. Larger pastures inhabited by commercial stock (cattle/sheep/horses) will be denoted separately, as the amount of area utilized by livestock at any one time may be relatively small compared to the overall aerial coverage of the pasture at hand.

Objective 4 (Compare dietary composition and predation rates)

This component of the study will utilize the larger long term data set of GPS cluster confirmed feeding events spanning November 2008 –November 2012. For each cougar, the number of feeding events for each particular prey species is divided by the total number of confirmed feeding events. A sample unit is defined as an individual cougar in order for the variance to represent inter-subject variability. Simple descriptive statistics (mean, standard deviation, confidence intervals) were then calculated across a sample of cougars with sufficient amount of data. For this exploratory analysis, frequencies of occurrence measures were calculated by:

1.) Ungulate sex
2.) Ungulate age (fawn, sub-adult, adult)
3.) Prey class (deer, elk, non-cervid)

The dataset was divided into three seasons based on time periods relevant to behavior of mule-deer (cougar's primary prey):

Summer (Jun. 1 – Sep. 30): Fawn rearing, male social aggregations
Fall (Oct. 1 – Dec. 31): Mating
Winter/Spring (Jan. 1 – May 31): Winter social aggregations

A preliminary estimate of feeding rates (i.e. feeding events/year) was calculated using estimated proportions of clusters representing kill types, total numbers of GPS clusters produced by the clustering algorithm, and frequency of occurrence by prey class. First a sample of subjects, each having at least one year of continuous data collected anytime between January 2011 through June 2012, were used to estimate the mean number of GPS location clusters produced by the algorithm for each of the five selection sets. For each selection set, the mean number of clusters was multiplied by the proportion of clusters representing feeding sites (see objective 1) to produce the annual number of feeding events. Next, the proportion representing each prey class was multiplied by this annual number of feeding events to estimate the number of deer, elk, and non-cervid prey feeding events per year.

Preliminary Results and Discussion

Objective 1

Investigations on the sampled GPS location clusters were conducted on a dataset spanning November 2008 – June 2012. Investigations were collected on 53 different GPS collared cougar subjects. Each analysis conducted in this annual report may utilize a slightly different subset of cougars, as subjects were only included if they produced a sufficient number of clusters/feeding events and relatively high quality GPS data for a period >*x* days, where *x* depends on the particular analysis. Cluster investigations were conducted on a total of 3723 clusters, with 2383 of these being from the first and second randomly selected clusters from each month for each subject. It is projected by the end date of the project (December 2012), ~4500 clusters will have been visited with approximately 2900 of these being from the randomly selected set. As mentioned prior, this study is unique in the effort placed on non-S1 cluster types (S2, S3, S4, S5 types), which made up 950 of the 2383 randomly selected clusters visited by investigators.

Feeding events were present at a mean proportion of 0.43 ±0.03(1/2 95% confidence interval) of the S1 clusters, while a much smaller proportion was found at the S2, S3, S4, and S5 sites (range of 0.04 – 0.11) (Figure 3).

Using a sample of 48 cougars with at least two feeding events, the minimum known scavenging proportion of total feeding events was a mean 0.11 (±0.05). Out of the 18 cougars with at least 15 feeding events being documented at their respective top randomly selected clusters, all but one had confirmed evidence of scavenging (Table 3).

The prey composition for S1 cluster types was compared against all non-S1 cluster types pooled. Results are shown in Figure 4. Deer made up a majority of both S1 (0.70 ±0.07) and non-S1 cluster types (0.55 ±0.15). The Non-cervid class of the non-S1 clusters (0.34 ±0.15) made up a proportion nearly twice as high as the S1 cluster types (0.17± 0.07). Elk surprisingly made up approximately equal proportions (~0.11) between S1 and non-S1 clusters. It is likely that many of the feeding events on deer and elk found at non-S1 clusters are scavenging events, or potentially cases where the subject was displaced (i.e. by another cougar, human, other predator species) from the site soon (<3 hours) after submission of the prey.

Future analysis will use logistic regression models, using a host of variables, to model predation events from GPS cluster data. Several factors influence the success of how a logistic regression model will perform. The first factor is the detection probability of an observer finding evidence of feeding remains likely declines as a function of the amount of time elapsed between the date the cougar was present at a site and the date that the observer visited the site. The second factor is that some subject's collars had low GPS fix rate success. Lower fix rates logically translate to a negatively biased number of clusters created by the clustering algorithm, which in turn can bias kill rate estimates (see objective 4). A third factor is that GPS location clusters that form on natal dens of female cougar may have similar characteristics as clusters formed on feeding events. All three of these factors will be addressed in subsequent analysis.

Objective 2

For the assessment of prey distribution, a single camera was deployed at each of the 131 sites between November 6, 2011 and March 1, 2012. Cameras will remain in the field until November 30, 2012. Cooperation with private landowners was required at 74 of the sites. Remaining sites are located on various tracts of public land owned by Jefferson County Open Space, USFS, Clear Creek County, Denver Mountain Parks, Colorado Parks and Wildlife, City of Boulder Open Space and Mountain Parks, and Boulder County Parks and Open Space. All camera sites were revisited at least once between March and July to assess battery life, memory card space, and over-all functionality. In some cases, new vegetation growth occurred directly in front of the camera that would either render pictures useless (blocked field of view) or trigger the camera when winds moved the vegetation, thus using up memory card space. Cursory inspections of the incoming data indicate that every site has been triggered at least once by a common cougar prey species.

Simultaneously answering questions relating cougar use of the landscape relative to prey distribution and human disturbance will give valuable insights to how a large top tier carnivore fits predictions of optimal foraging theory. Specifically, insight to how a top tier predator perceives its landscape and whether or not tradeoffs are being made between maximizing food intake and reducing risks posed by humans is important to advancing knowledge of how animals use resources and perceive their environment. Applications of optimal foraging theory to large carnivorous species are rare, and thus would add knowledge to whether or not predictions drawn from model species are scalable to the highest trophic levels. In addition, results from this study are important to conservation and management of the landscapes occupied by cougar. A study that simultaneously examines the influences of human development and prey distributions on cougar is important to predicting how well foraging behaviors of cougar may adapt to future urban sprawl. Finally, this study will provide knowledge on hypotheses regarding whether or not elevated prey resource levels are a driver of cougar use of exurban and suburban landscapes.

Currently, analysis in the camera trap portion of this study allows for the assessment of cougar use for a particular prey species on an individual species basis. Much focus will be placed on species most commonly preyed upon by cougar, such as deer, elk and raccoon. Incorporating a wider range of species, in addition to accounting for detectability differences between species, would potentially allow future analysis to assess the selection of one particular species over other available species. In addition, fine scale species distribution data are rare, and thus these data may be useful to other wildlife/land managers and researchers.

Objective 3

Mapping of parcels of land containing hobby livestock was initiated in April 2012 through digitizing USGS color high resolution (0.6 m) ortho-imagery that was fortunately available for the extent of the study area. Specifically, point features were created on any tract of land that showed obvious evidence of hobby livestock husbandry practices (corrals/ barns/ fences/ bareground). Roadside visual mapping efforts will be conducted Fall 2012 after deciduous trees have shed leaves.

Knowing if cougar seek hobby livestock in certain seasons is important to predicting cougar/human conflicts. It is suspected that the spring periods are when livestock depredations are most reported. Speculations exist that cougar are seeking alternative prey sources during the spring months when primary prey sources (ungulates) are at their lowest availability.

Increasing harvest rates of species involved in human/wildlife conflicts are a common practice for managers of wildlife populations. However, increasing the harvest quota may not be a suitable management method to decrease human/cougar conflicts in a localized area for various reasons. First, increases in the quota for maximum harvest have not resulted in a substantial increase of harvested cougar (CDOW 2004). Second is that other research has found that small areas with high harvest may only exhibit increased immigration rates especially from younger age classes (Cooley et al. 2009), with no significant overall decrease in density. Thus, a population skewed toward a younger age structure may occur (Cooley et al. 2009). If speculations are true that younger cougar, relative to older cougar, are more likely to prey on hobby livestock, then hobby livestock owners may suffer an increased level of losses in the future.

Objective 4

A sample of 25 cougars (7 male and 18 female cougar) was used to assess cougar usage of ungulate prey based on ungulate sex. At deer feeding sites, cougars and/or other scavengers often crush the skulls, which contain antler pedicels, the primary characteristic for assessing gender in the field if antlers are not yet present. Investigators were unable to determine sex on a high proportion (Summer: 82%, Fall: 64%, Winter: 62%) of the deer carcasses found. Sex was rarely determinable on fawn deer, as skulls of fawns appear to be more fragile and thus a large majority of the deer with undetermined gender seemed to be of the fawn age class. Thus it is assumed that proportions of males and female carcasses in the unidentifiable sex group are similar to the proportions found in identifiable carcasses, and thus proportions of male and females are corrected to reflect this.

Using only cougars having ungulate kills with determinable sex, we used a sample of 13 cougars in the summer, 17 cougars in the fall, and 21 cougars in the winter to calculate the proportion of ungulate feeding events comprising each sex. In the summer season, the male to female prey ratio was 1:2.4 (Figure 5), but this ratio was nearly reversed in the fall season with a 1.8:1 ratio. During the winter season, the ratio was near unity at 1:1.02. Confidence intervals for all estimates are very wide, even for normal approximations.

The proportion of deer carcasses identified as fawn (<1 YOA [years of age]), sub-adult (1-2 YOA), adult (>2 YOA), and unknown age, were calculated from a sample of 26 individual cougars (7 males and 19 female cougar), with sufficient data, that fed upon $\geq$ 1 deer carcass in each of the three seasons. Age class of deer was identified by mandibular tooth eruption patterns, and thus age was assigned to be unknown if the lower mandibles could not be found by cluster investigators.

Deer carcasses falling in the unknown age class made up a moderate amount of the carcasses found in each of the three seasons (summer: 0.13, fall: 0.40, winter: 0.25). A comparison of the proportions of each age class, for each season, is shown in Figure 6. Most notably was the difference in fawn and adult usage in the summer versus the winter season. During the summer, 0.68 ± 0.12 of the carcasses were found to be fawns (usually new born fawns), while only 0.25 ± .12 were adults. However in the winter season adults made up 0.65 ± 0.10 while fawns made up only 0.18 ± 0.08. Sub-adults appeared to be preyed on the least of the three age classes.

For prey composition, we calculated the frequency of occurrence of food items, averaged over a sample of collared cougars with sufficient data (11 males and 22 females) where investigations on random sub-sample of clusters from each cougar yielded a total of 620 different prey items. Across all seasons, deer were preyed on most frequently (0.68 ±0.07), non-cervids second most frequently (0.20 ±0.07), and elk least frequently (0.12 ±0.04). While, the proportion of mule deer prey dominated over the alternative prey classes (elk and non-cervids) in all seasons, a slightly lower proportion of mule deer was observed in the winter season (0.61 +0.1) compared to the following summer season (0.73 ±0.08) (Figure 7).

Testing for seasonal differences in prey-species composition indices and frequency of occurrence of individual species may have relevance to prey-switching abilities of cougar. Following these assumptions:

- Winter season (January - May) represents the time period with lowest primary prey (deer) availability, while the summer season (June - September) represents the highest availability of primary prey.
- Alternative prey follows similar trends in availability as primary prey.
- Energetic demands are equal throughout the year.

One may utilize the seasonal differences as a proxy to test whether or not cougar switch from using predominately deer or other natural prey items, to other prey species when faced with lower levels of primary prey availability. Additional background work will be conducted to assess the validity of these assumptions in the applicability to the study at hand. Also, future work will incorporate additional investigations, increase the sample size of subjects, and use more refined methods of variance estimation to assess whether the seasonal differences found in this analysis are significantly different.

Showing differences in prey-species composition indices and frequency of occurrences of individual species between differing sex and age classes is important to management/conservation of the prey species. Management techniques that change the sex or age structure of the cougar population may impact populations of certain prey species. For instance, if younger cougar are more likely to feed on alternative prey species, then using techniques that shift the cougar population to a younger age structure may have a larger impact on populations of alternative prey.

Preliminary estimates of feeding rates were calculated from a sample of 22 subjects (7 males, 15 females), with sufficient data. The mean numbers of GPS location clusters produced by the algorithm for each of the five cluster types are displayed in Figure 8. Using feeding proportions (Figure 3), a mean of

93.3 (95% C.I.: 85-101) annual feeding events may be expected, with 70.6 of these occurring from S1 clusters and 22.7 occurring from non-S1 cluster types. If our study had only conducted field investigations on S1 cluster types (only visit clusters consisting of $\geq$2 GPS locations within 100 m), feeding/kill rates may be biased low by 25%. Multiplying this preliminary kill rate by the proportions of each prey class utilized by cluster type (Figure 4), while ignoring the additional variance induced by this measure, a preliminary mean estimate of 62.5 deer, 10.8 elk, and 20.12 non-cervid prey items are consumed per cougar annually.

These feeding/kill rates are only preliminary, and additional work must be done to disentangle several factors that produce positive and negative biases in these estimates. Factors such as poor GPS fix success rate, and failure of field investigators to find prey remains at a clusters can negatively bias feeding/kill rates. Our clustering algorithm can assign more than one unique cluster identification record to a group of GPS points if the cougar spends more than 8 days at the cluster (i.e. in the case of elk predations), and thus this can place a positive bias on feeding kill/rates. While this was a simple deterministic model, future analysis will be conducted on a per-subject basis, where the results from logistic regression models (as developed in Objective 1 above) will be used to identify probability of any un-visited cluster, based on a suite of variables, to represent a feeding site.

Cougars were once thought to rarely scavenge, but recent work has indicated that scavenging events are more common than once thought (Nowak et al. 2000, Bauer et al. 2005, Bacon and Boyce 2010, Knopff et al. 2010). Not separating predation events from scavenging events will positively bias the number of prey items killed per capita. Cursory examinations of the GPS cluster location data collected in vicinities inhabited by multiple collared cougars indicate that sharing of food items does exist to some degree. Sharing of food items is essentially a form of scavenging, and even the most prompt visits to cluster sites are sometimes unable to disentangle whether the focal cougar (GPS collared cougar) indeed killed the prey item. This is especially important in predation studies that do not have all cougars in an area collared. Additional work must be done to either 1) estimate which clusters represent scavenging events, or 2) conduct a post-hoc estimation of the proportion of feeding events that are actually scavenging events.

To assess the degree of scavenging and the effect on predation rate calculations, known scavenging events on feeding sites derived from carcass dumps, road-kill, hunter-kill, and non-cougar usurped kills will be identified. Cougar shared/usurped kills will be identified by examining direct spatial and temporal overlap of GPS cluster data collected from pairs of GPS collared cougars. Not all cougars in an area are monitored with GPS collars, and thus a separate model will be created that predicts the amount of kill sharing as a function of pair-wise home range overlap.

Assessing whether differences exist in cougar dietary composition and feeding rates of deer, between levels of high and low human density may be relevant toward discussions of whether or not suburban/exurban landscapes have an impact on cougar fitness, or on the contrary, how cougar may adapt to these potential human disturbances. Describing feeding rates on certain species such as deer and elk are important to Front Range wildlife managers. Predator-prey models incorporating species specific predation rates as parameters will also benefit from these kill rate estimates. Knowing the impact of cougar on populations of prey items, that are also harvestable by humans, is important to the management of these particular game species.

Literature Cited

Ackerman, B.B., F.G. Lindzey, and T.P. Hemker. 1986. Predictive energetic models for cougars. In S.D. Miller and D.D. Everett, (eds.): Cats o the world: biology, conservation, and management. National Wildlife Federation, Washington D.C., U.S.A.

Alldredge, M.A., E.J. Bergman, C.J. Bishop, K.A. Logan, and D.J. Freddy. 2008. Pilot evaluation of predator-prey dynamics on the Uncompahgre Plateau. Colorado Division of Wildlife: Research Report, Federal Aid no: W-185-R.

Altendorf, K.B., J.W. Laundre, C.A. Lopez Gonzalez, J.S. and J.S. Brown. 2001. Assessing effects of predation risk on foraging behavior of mule deer. Journal of Mammalogy 82:430-439.

Anderson, C.R. and F.G. Lindzey. 2003. Estimating cougar predation rates from GPS location clusters. The Journal of Wildlife Management 2:307-316.

Atwood, T.C., E.M. Gese, and K.E. Kunkel. 2007. Comparative patterns of predation by cougars and recolonizing wolves in Montana's Madison Range. Journal of Wildlife Management 71:1098-1106.

Bacon, M.M. and M.S. Boyce. 2010. Scavenging of an elk carcass by multiople cougars in SE Alberta. Canadian Field Naturalist 124:242-245.

Bauer, J.W., K.A. Logan, L.L. Sweanor, and W.M. Boyce. 2005. Scavenging behavior in puma. The Southwestern Naturalist 50:466-471.

Blumstein, D.T., E. Fernandez-Juricic, P.A. Zollner, and S.C. Garity. 2005. Inter-specific variation in avian responses to human disturbance. Journal of Applied Ecology 42:943-953.

Burdett, C.L., K.R. Crooks, D.M. Theobald, K.R. Wilson, E.E. Boydston, L.M. Lyren, R.N. Fisher, T.W. Vicers, S.A. Morrison, and W.M. Boyce. 2010. Interfacing models of wildlife habitat and human development to predict the future distribution of puma habitat. Ecosphere 1(1):art4. doi:1890/ES10-0005.1.

CDOW. 2003. Colorado Vegetation Classification project. http://ndis.nrel.colostate.edu/coveg/

CDOWa. 2004. Mountain lion data analysis unit management plan, Northern Front Range: DAU L-4. Prepared by Viera M. accessed online 4-12-2011 http://wildlife.state.co.us/NR/rdonlyres/0A9B2D73-CA94-47AA-88AE-281DD7186498/0/L4DAUPlan112004.pdf

CDOW. 2006. Baily deer herd management plan: Data Analysis Unit D-17. Prepared by Linstrom, A. accessed online 4-12-2011 http://wildlife.state.co.us/NR/rdonlyres/0D501E5C-2ADD-4606-8F53-C8A5BC151FD0/0/D17DAUPlan_Bailey.pdf

Cougar Management Guidelines Working Group (CMGWG). 2005. Cougar management guidelines. WildFutures, Bainbridge Island, WA.

Cooley, H.S. H.S. Robinson, R.B. Wielgus, and C.S. Lambert. 2008. Cougar prey selection in a white-taild deer and mule deer community. Journal of Wildlife Management 72:99-106.

Cooley, H.S. H.S. Robinson, R.B. Wielgus, G.M. Koehler, H.S. Robinson and B.T. Maletzke. 2009. Does hunting regulate cougar populations? A test of the compnesatory mortality hypothesis. Ecology 90:2913-2921.

Efford, M.G. and D.K. Dawson. 2012. Occupancy in continuous habitat. Ecosphere 3(4):1-15 (Article 32).

Emlen, J.M. 1968. Optimal choice in animals. The American Naturalist 102:385-389.

Frid, A., and L. Dill. 2002. Human-caused distrubance stimuli as a form of predation risk. Conservation Ecology 6(1): 11. [online] URL: http://www.consecol.org/vol6/iss1/art11

Gill, J.A. and W.J. Sutherland. 2000. The role of behavioural decision making in predicting the consequences of human disturbance. In: Gosling, L.M. and W.J. Sutherland (Eds.), Behaviour and Conservation. Cambridge University Press, Cambridge.

Harmsen, B.J., R.J. Foster, S. Silver, L. Ostro, and C.P. Doncaster. 2010. Differential use of trails by forest mammals and the implications for camera-trap studies: a case study from Belize. Biotropica 42:126-133.

Holmes, B.R. and J.W. Laundre. 2006. Use of open, edge and forest areas by pumas in winter: are pumas foraging optimally? Wildlfie Biology 12:201-209.

Kays, R., S. Tilak, B. Kranstauber, P.A. Jansen, C. Carbone, M.J. Rowcliffe, T. Fountain, J. Eggert, and Z. He. 2010. Monitoring wild animal communities with arrays of motion sensitive camera traps.

IEEE International Workshop on Practical Issues in Building Sensor Network Applications. Accessed online 4-12-2011 http://arxiv.org/abs/1009.5718

Kertson, B.N. 2010. Cougar ecology, behavior, and interactions with people in a wildland-urban environment in western Washington. Ph.D. Dissertation. University of Washington, Pullman. pp. 149.

Kertson, B.N., R.D. Spencer, J.M. Marzluff, J. Hepinstall-Cymerman, and C.E. Grue. 2011. Cougar space use and movements in the wildland-urban landscape of western Washington. Ecological Applications in-press doi:10.1890/11-0947.1.

Knopff, K.H., A.A. Knopff, M.B. Warren, and M.S. Boyce. 2009. Evaluating Global Positioning System telemetry techniques for estimating cougar predation parameters. Journal of Wildlife Mangement 73:586-597.

Knopff, K.H., A.A. Knopff, and M.S. Boyce. 2010. Scavenging makes cougars susceptible to snaring at wolf bait stations. Journal of Wildlife Mangement 74:644-653.

Knopff, K.H., A.A. Knopff, A. Kortello, M.S. Boyce. 2010. Cougar kill rate and prey composition in a multiprey system. Journal of Wildlife Management 74:1435-1447.

Laundre, J.W. 2005. Puma energetics: A recalculation. The Journal of Wildlife Mangement 69:723-732.

Laundre, J.W. 2008. Summer predation rates on ungulate prey by a large keystone predator: how many ungulates does a large predator kill? Journal of Zoology 275:341-348.

Laundre, J.W. and L. Hernandez. 2003. Winter hunting habitat of pumas Puma concolor in northwestern Utah and southern Idaho, USA. Wildlife Biology 9:123-129.

MacKenzie, D.I., J.D. Nichols, G.B. Lachman, J.A. Royle, C.A. Langtimm. 2002. Estimating site occupancy rates when detection probabilities are less than one. Ecology 8:2248-2255.

MacArthur, R.H. and E.R. Pianka. 1966. On optimal use of a patchy environment. The American Naturalist 100:603-609.

Mattson, D.J. 2007. Mountain lions of the Flagstaff Uplands. Southwest Biological Science Center: 2003-2006 Progress Report: U.S. Geological Survey Open-File Report 2007-1050.

Michalsiki, F., R.L.P. Boulhosa, A. Faria, and C.A. Peres. 2006. Human-wildlife conflicts in a fragmented Amazonian forest landscape: determinants of large felid depredation on livestock. Animal Conservation 9:179-188.

Murphy, K.M. 1998. Ecology of the cougar (Puma concolor) in the northern Yellowstone ecosystem: Interactions with prey, bears, and humans. Ph.D. Dissertation, University of Idaho, Moscow.

Nowak, M.C., T.E. Taylor, and G.W. Witmer. 2000. Prolonged scavenging by a female mountain lion in northeastern Oregon. Northwest Naturalist 81:63-65.

O'Brien, T.G. 2010. Abundance, density, and relative abundnace: a conceptual framework. In O'Connell et al. (eds.), Camera Traps in Animal Ecology: Methods and Analysis., Newyork, Springer.

O'Brien, T.G., M.F. Kinnaird and H.T. Wibisino. 2003. Crouching tigers, hidden prey: Sumatran tiger and prey populations in a tropical forest landscape. Animal Conservation 6:131-139.

O'Brien, T.G., M.F. Kinnaird, and H.T. Wibisono. 2010. Estimation of species richness of large vertebrates using camera traps: An example from an Indonesian rainforest. In O'Connell et al. (eds.), Camera Traps in Animal Ecology: Methods and Analysis., Newyork, Springer.

Pierce B.M., V.C. Bleich, J.D. Wehausen, R.T. Bowyer. 1999. Migratory patterns of mountain lions: Implications for social regulation and conservation. Journal of Mammalogy 80:986-992.

Pierce B.M., V.C. Bleich, and , R.T. Bowyer. 2000. Social organization of mountain lions: Does a land-tenure system regulate population size? Ecology 81:1533-1543.

Pyke, G.H., H.R. Pulliam, and E.L. Charnov. 1977. Optimal foraging: A selective review of theory and tests. The Quarterly Review of Biology. 52:137-154.

Rovero, F. and A.R. Marshall. 2009. Camera trapping photographic rate as an index of density in forest ungulates. Journal of Applied Ecology 46:1011-1017.

Rowcliffe, J.M., J. Field, S.T. Turvey, and C. Carbone. 2008. Estimating animal density using camera traps without the need for individual recognition. Journal of Applied Ecology 45: 1228-1236.

Rowcliffe, J.M., C. Carbone, P.A. Jansen, R. Kays, and B. Kranstauber. 2011. Quantifying the sensitivity of camera traps: an adapted distance sampling approach. Methods in Ecology and Evolution. doi: 10.1111/j.2041-210X.2011.00094.x

Royle, J.A. and J.D. Nichols. 2003. Estimating abundance from repeated presence-absence data or point counts. Ecology 84:777-790.

Schoener, T.W. 1971. Theory of feeding strategies. Annual Review of Ecology and Systematics 2:369-404.

Sunquist, M.E. , and F.C. Sunquist. 1989. Ecological constraints on predation by large felids. Page 283-301 in J.L. Gittleman, editor. Carnivore behavior, ecology, and evolution. Cornell University Presss, Ithaca.

Torres, S.G., T.M. Mansfield, J.E. Foley, T. Lupo, and A. Brinkhaus. 1996. Mountain lion and human activity in California: Testing Speculations. Wildlife Society Bulletin 24:451-460.

White, K.R., G.M. Koehler, B.T. Maletzke, and R.B. Wielgus. 2011. Differential prey use by male and female cougars in Washington. Journal of Wildlife Management 75:1115-1120.

Willems, E.P. and R.A. Hill. 2009. Predator-specific landscapes of fear and resource distribution: effects on spatial range use. Ecology 90:546-555.

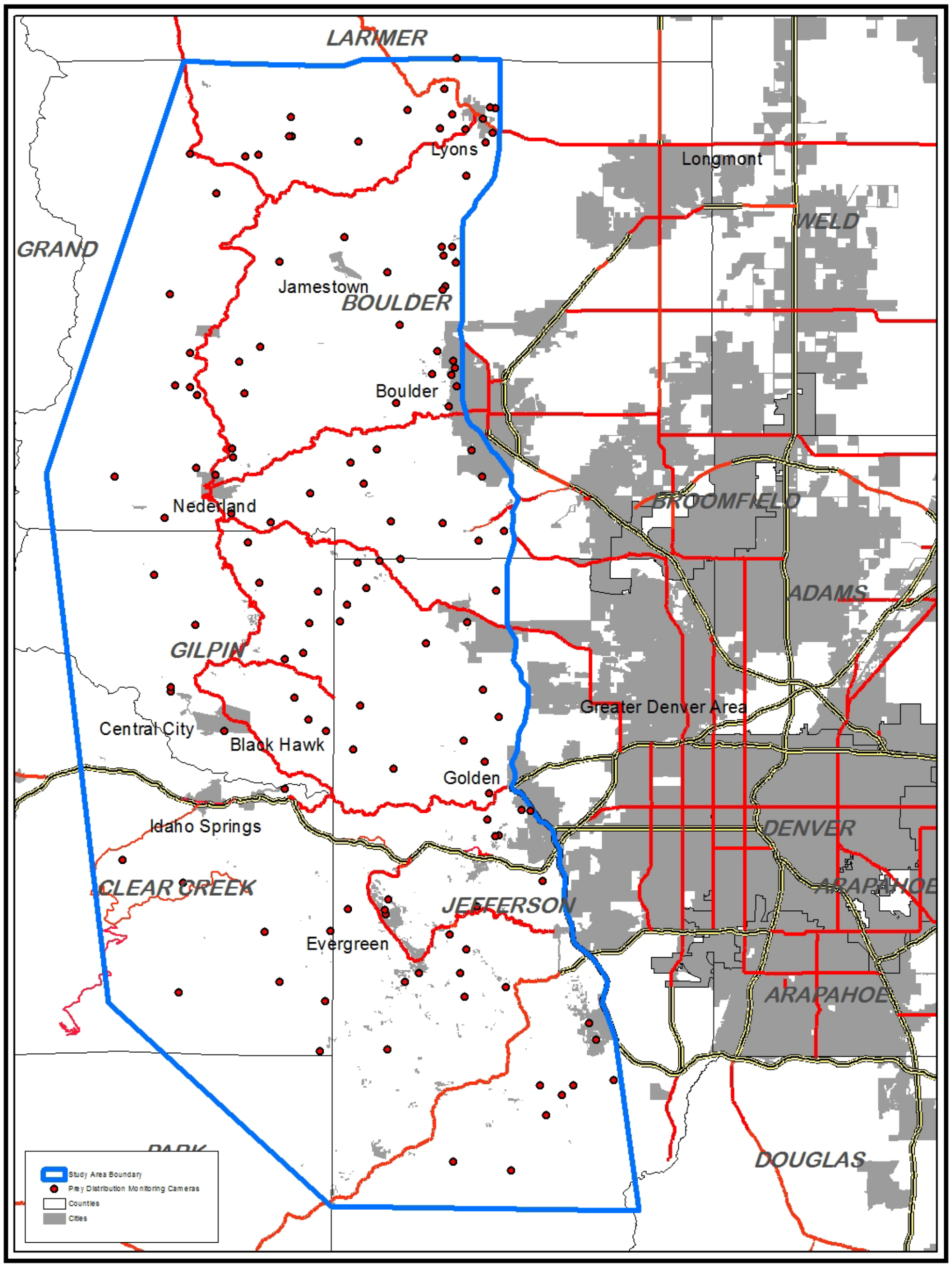


Figure 1: The 2862 km^2 study area, delineated with the blue polygon, lies on a portion of the eastern slope of the Colorado Front Range. The study area encompasses the approximate home ranges for a sample of GPS collared cougars.

Figure 2: For randomly selected sites with no available trees, camera units are installed on a T-post with a bracket custom made for the unit.

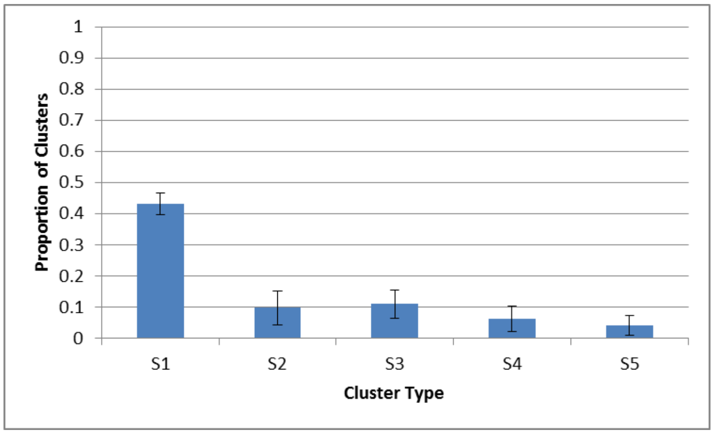


Figure 3: Proportion of clusters representing feeding events for each cluster type (selection set). Error bars represent 95% confidence intervals using a normal approximation.

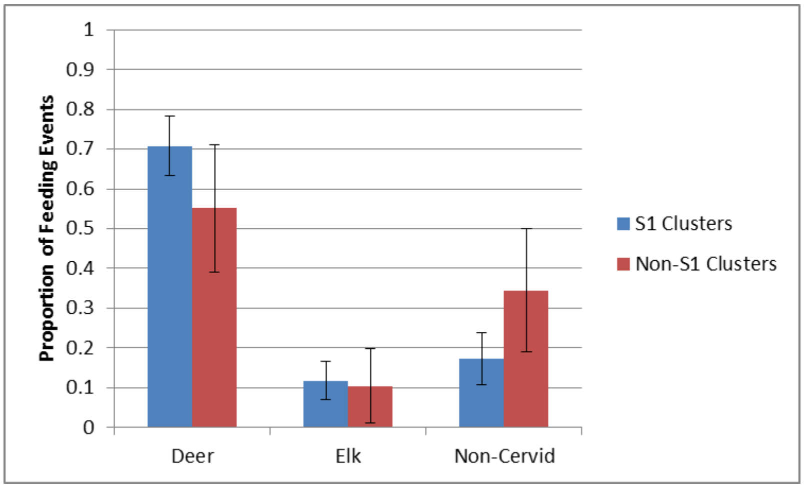


Figure 4: Comparison of S1 and Non-S1 cluster types in the proportion of feeding events represented by deer, elk, and non-cervid prey class. Error bars represent 95% confidence intervals using a normal approximation.

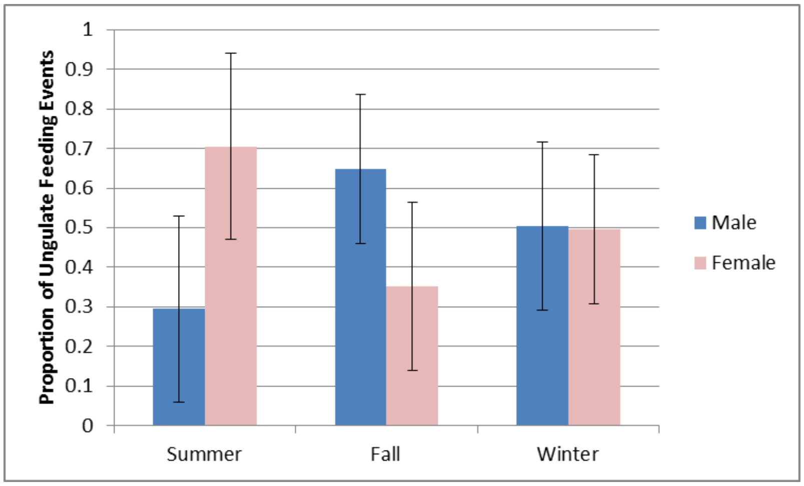


Figure 5: Comparison of male and female ungulate proportion comprising feeding events in Summer, Fall, and Winter seasons. Error bars represent 95% confidence intervals using a normal approximation.

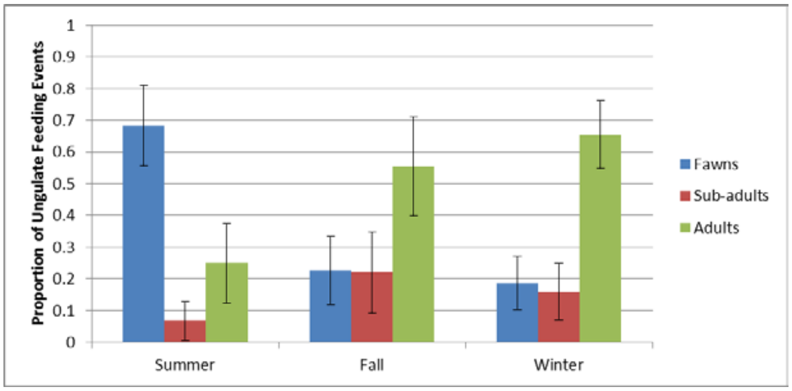


Figure 6: Comparison of proportion of ungulate feeding events representing fawns, sub-adults, and adult age classes for the Summer, Fall, and Winter seasons. Error bars represent 95% confidence intervals using a normal approximation.

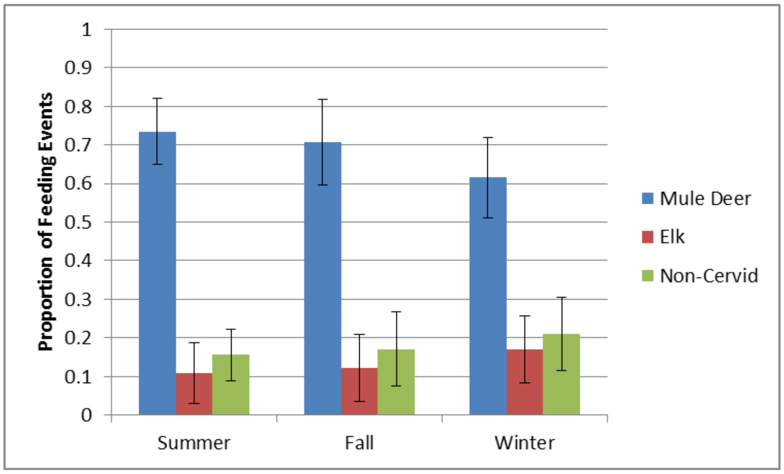


Figure 7: Comparison of the proportion of feeding events for each prey class, by Summer, Fall, and Winter seasons. Error bars represent 95% confidence intervals using a normal approximation.

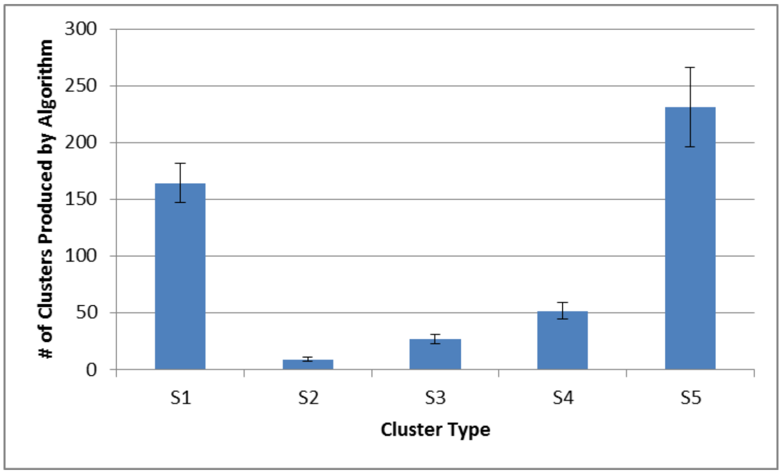


Figure 8: Comparison of the number of clusters produced by clustering algorithm of each cluster type. Total number of non-S1 clusters (S2, S3, S4, and S5 types) outnumber S1 cluster types by almost 2:1.

Explanatory Variable Major Type	Variable	Description
GPS Based	Total duration of cluster (hours)	Cumulative number of hours spent at cluster site
	Fidelity to cluster site	Proportion of points collected during the time window of the cluster that actually occur at the cluster
	GPS fix success	Proportion of successful GPS locations recorded by the GPS collar during the time window of the cluster
	GPS location dispersion	Standard deviation of GPS locations (UTM units) of all points at the cluster site
	Proportion of night-time locations	Proportion of GPS locations at cluster occurring at night
Accelerometer Based	Average X axis measure	Higher X axis values collected ±30 minutes of GPS fix time potentially indicate more straightline movements (i.e. travelling)
	Average Y axis measure	Higher mean Y axis values collected ±30 minutes of GPS fix time potentially indicate more head twisting movements (i.e. prey handling)
	Standard deviation X or Y axis	High stdev in X axis values potentially indicate less consistent behavior (i.e. mix of travelling, feeding, resting)
Observer/Cluster Investigator Based	Terrain ruggedness	Rugged terrain more difficult to search
	Visitation lag*	Time between dates of cougar arrival and investigator visitation
	Observer experience*	Number of cluster investigations previously participated in
Environmental Based	Season	
	Season adjusted temperature	Ambient temp from on-board GPS collar themometer, rescaled by seasonal high and low temp
	Snow cover*	
	Landcover	i.e. Grassland, Shrubland, Forested, Urban
	Human activity	where the potential of feeding behavior to be interrupted by humans may be higher (i.e. distance to house, housing density)
	Canopy Cover*	
Cougar Subject Specific	Age	Independent Sub-adult or Adult
	Sex	Male or Female
	Maternal Status	Presence/absence of kittens, age and size of kittens, etc

** Variable is unable to be derived for unvisited clusters*

Table 1: Variables to be used in logistic regression models for predicting feeding events from GPS location cluster data.

Strata	Factor	Sub-Type	Description	# of Sites
Major habitat	DEC		Deciduous trees present	19
	GRS		Site dominated by grassland	20
	HEC		Site dominated by coniferous forest >8000 ft elevation	20
	LEC		Site dominated by coniferous forest <8000 feet in elevation	27
	SHR		Site dominated by scrub/shrub	20
	URB	Stream1	Site located in urban/suburban housing density levels and within 100 meters of a perennial stream	6
	URB	Stream2	Site located in urban/suburban housing density levels and 100-750 meters of a perennial stream	6
	URB	Stream3	Site located in urban/suburban housing densities and >750 m from a perrenial stream	6
	MIX		Site located in a mix of one of the major habitat classes	8
Housing Density	Rural		Housing density >16.18 ha/unit	67
	Exurban		Housing density 0.68-16.18 ha/unit	47
	Suburban/Urban		Housing density <0.68 ha/unit	18
Proximity to Dwelling	House 1		Site located < 200 m of house	61
	House 2		Site located within 200-700 m of house	50
	House 3		Site located >700 m from house	21

Table 2: Placement of cameras followed a stratified random sampling design across three major stratifications. These strata were chosen in order to spread out the cameras across a range of conditions, as well as to ensure adequate characterization of prey availability in relation to human activity.

Cougar Subject ID	Sub-sample size (total feeding events)	Proportion confirmed scavenging
AF61	22	0.000
AF01	25	0.040
AM13	21	0.048
AF19	20	0.050
AF69	17	0.059
AF54	27	0.074
AF15	49	0.082
AF62	22	0.091
AF40	32	0.094
AF79	18	0.111
AF34	27	0.111
AM14	26	0.115
AF59	17	0.118
AF77	16	0.125
AF73	15	0.133
AM44	22	0.136
AF52	20	0.200
AM76	20	0.200
AM74	17	0.235

Table 3: Proportion of feeding events determined to be known scavenging cases for individual cougar subjects with at least 15 feeding events documented at randomly selected cluster sites.

APPENDIX IV

Front Range Cougar Research
Winters, 2011–2012 & 2012–2013

Predator-Prey Dynamics in Relation to Chronic Wasting Disease and Scavenging Interactions at Cougar Kill Sites

Colorado Parks and Wildlife

Joe Halseth
Mat Alldredge

Research Proposal

February, 2012

Predator-Prey Dynamics in Relation to Chronic Wasting Disease and Scavenging Interactions at Cougar Kill Sites

Joe Halseth and Mat Alldredge, CPW

Need

The current Colorado Parks and Wildlife (CPW) cougar (*Puma concolor*) research on the Front-range is utilizing advanced GPS radio collar technology. This technology allows researchers to track cougar movements on an almost real time basis, with the roughly 25 current active project collars uploading seven times per day. These dependable GPS uploads give researchers the ability to identify possible kill sites quickly, sometimes as soon as 6 to 12 hours after a kill is made. This provides the opportunity to explore previously un-researched facets of cougar behavior during the relatively short time interval from the point a cougar makes a kill, to the point at which it abandons the carcass. Feeding behavior, intraspecific kill site interaction, and scavenger competition can now be investigated.

Similar data to that collected in Krumm et al.'s (2005) and Miller et al.'s (2008) cougar studies, which examined cougar selection of Chronic Wasting Disease (CWD) positive mule deer (Odocoileus hemionus), can now be collected with a greater degree of efficiency. The study areas of each of the two prior CWD cougar projects lie within the more broad boundaries of the current Front Range cougar project, and a larger number of known cougars will increase sample sizes of CWD tissues from cougar killed mule deer. Additionally, much of the field work from the two previous studies is nearly a decade old which justifies another project to compare to past results. The ability to collect a potentially larger sample size will yield more accurate findings, identify gaps in need of further study, and/or detect developing trends in regards to possible temporal patterns.

The ongoing cougar project's available technology and resources, and the relatively minor additional project costs, provide the opportunity to initiate a camera study to explore cougar feeding behavior and scavenger interaction in the period immediately following a cougar kill. Site visitation of fresh cougar kills also allows for the collection of adequate tissue samples to test for CWD, in order to further explore if cougars are selecting for CWD positive mule deer or other ungulates.

Background

Cougar behavior and scavenger interaction:

Although there have been significant cougar research projects in the U.S. and Canada, only recent GIS advancements have allowed researchers the ability to monitor cougar movements and locations with dependable accuracy on a real-time basis. With GPS collar technology, researchers can collect data on kill sites, prey items, home ranges, den locations, preferred habitats, and a variety of other previously under-explored areas of cougar ecology and behavior.

This new technology initiated many projects that examined cougar feeding behavior. These projects collected extraordinary data documenting duration of kill site occupation, prey analysis, biomass consumption, and feeding patterns (Anderson and Lindzey 2003, Bauer et al 2005, Knopff et al 2010, Blecha and Alldredge unpublished data). However, actual behavior, feeding activity, consumption rates, and scavenger interactions has yet to be thoroughly documented. Placing cameras on fresh kill sites will document feeding behavior, such as the length of actual feeding sessions, and will identify any patterns of behavior that exist during the progression of feeding on a prey item. Additionally, placing cameras will document interaction with competing scavengers and conspecifics. A goal of this proposed project is to document how often scavengers challenge cougars on fresh kills and how successful these competing scavenging species are at stealing the food item. Using the time stamped photos from cameras, we will be able to determine the average time it takes for competing scavengers to arrive on site after a kill and the rate in which the scavenger species successfully displaces the cougar. Seasonal variation in scavenging rates of fresh carcasses will be analyzed, especially with regard to bear activity and changes in diet competition.

Basic cougar ecology suggests that with the exception of family groups and mating interaction, cougars are largely solitary animals (Seidensticker et al. 1973). On numerous occasions throughout the course of the ongoing lion project, researchers have documented two cougars on the same kill site. One can only speculate on their interaction. This proposed project also seeks to document behavior in such situations to observe if cougars are sharing kills or challenging one another for feeding opportunities.

CWD component:

Ongoing cougar research on the northern Front-range (Alldredge, unpublished data) as well as other significant cougar research (Logan and Sweanor 2001, Anderson and Lindzey 2003, Hornocker 1970) has shown that cougars prey on a wide diversity of prey species, but select for deer and elk in higher proportions. Additionally, the northern Front Range has been identified as the epicenter of the Chronic Wasting Disease (CWD) epidemic, possessing the highest infection rates in the state (Miller et al. 2000). CWD is a naturally occurring prion disease effecting deer, elk and moose. Early stages of infection are difficult to recognize but advanced signs of CWD infected deer are more readily identified by humans, with symptoms including poor body condition, reduced coordination, excessive salivation, and increased isolation from other deer (Williams and Young 1980). Basic predation theories suggest that predators prey upon young, sick, and older individuals in greater proportion than fit, mature individuals (Errington 1946, Slobodkin 1968). Optimal foraging theory predicts that predators ought to choose the most "profitable" prey (MacArthur and Pianka 1966, Schoener 1971, Pulliam 1974), which should be the largest prey available that can safely be killed. Thus, we might assume cougars can identify a deer in the later stages of CWD infection. Miller et al. (2008) speculated that cougars could have the ability to identify the most subtle changes in behavior or body condition in early stage CWD positive deer, causing them to be more vulnerable to predation.

While it is known that cougars prey on deer or other ungulates as a primary food source, only two studies have explored whether cougars are selecting for CWD positive deer (Krumm et al. 2005, Miller et al. 2008.) Krumm et al. (2005) found the percentage of CWD infected mule deer killed by cougars was significantly higher than hunter harvested deer in the same area. Miller et al. (2008) found infected deer were much more likely to be killed by cougars than uninfected ones. There is little information on cougar selection of CWD infected elk but this proposed study will document any CWD occurrence in cougar killed elk.

Over the past three years, the ongoing cougar project has established many positive relationships with a variety of stakeholders within the study area. Prominent land owners, land managers, various municipal organizations, and many members of the general public are aware and supportive of the project. Additionally, District Wildlife Managers, Boulder County, and Boulder City open space rangers often respond to reports of cougar kills on open space parks and protected areas and within the urban fringe areas. These established relationships increase the frequency of lion activity alerts, yielding more opportunities for researchers to capture cougars. These relationships will continue to provide increased future opportunities to place cameras on fresh cougar kills and collect CWD testable tissues. Advanced GPS technology, coupled with in-field awareness and participation, will allow researchers to capitalize on opportunities and bolster sample size.

It is the responsibility of CPW to utilize the best science when managing Colorado's wildlife resources. Exploring cougar kill site behavior will determine loss rates from scavenging/competition of fresh carcasses. This could provide insight on actual prey consumption and clarify an important variable in estimating the frequency of cougar deer and elk kills. Documenting feeding behavior has not previously been done in this proposed fashion and will provide invaluable information on basic cougar ecology and behavior. Collecting samples for CWD testing will provide a welcome opportunity to compare new data to the two previous studies and to existing (and evolving) CPW CWD data. Furthering our understanding of the relationships between predator/prey and disease dynamics will afford biologists better information in managing Front Range wildlife populations.

Objectives

1. Document sharing and/or abandonment rates of cougars occupying kill sites in response to presence of other cougars and/or scavengers
2. Document time from kill until presence of competing scavengers
3. Document feeding patterns and length of individual feeding sessions.
4. Compare CWD infection rates from cougar killed deer and elk to existing CPW CWD infection rates to determine if cougars are selecting for CWD positive deer and elk.

Methods

Researchers will monitor cougar movements using GPS data on a GIS to detect possible kill sites as early as possible. This is already successfully being done by viewing collar locations to address the feasibility to initiate capture operations in order to re-collar specific cougars. After a location is deemed permissible and realistic to access, researchers will travel to the kill site area and navigate to the potential kill site location. Personnel will use a VHF signal to monitor cougar location during the approach to avoid contact. In the event the cougar is onsite, at the kill, researchers will reassess approach and come back at another time.

In the event a kill is found, a maximum of two cameras will be placed to document feeding activity and scavenger interaction. Multiple cameras will be used in the event the cached prey item is slightly moved and to monitor activity within a larger area. Cameras will be affixed to adequate stationary objects and camouflaged with vegetation to minimize sight manipulation and detection. The reconyx cameras currently used in the parent cougar project are 4x6 inches and emit a low glow instead of a flash during nighttime photographs. Cameras will be left in place up to two weeks after the cougar has left the kill site.

If the prey item is a mule deer or other ungulate, retropharyngeal lymph nodes and/or the medulla oblongata at the obex will be collected for CWD testing. Additionally a lower incisor will be obtained for accurate age analysis. Krumm et al. (2005) collected 54 testable samples from cougar killed mule deer in 42 months. Miller et al. (2008) observed 11 CWD positive collared deer succumbed to cougar predation at a rate nearly four times that of uninfected collared deer. With the large number of collared cougars in the current Front Range cougar project (n≈25), we predict the ability to collect a target sample size of 4-5 tissue samples per month. A large sample is necessary to determine if cougars are selecting for CWD positive deer, as the power to detect a 10% difference using binomial proportions is only 0.75 (n=200).

While some disturbance to cougars may be unavoidable if the animal is alerted upon researcher approach, precautions will be taken to not force cougars off a kill. Past experiences, especially those associated with capture activities, on the Front Range cougar project have shown that a cougar is not likely to be affected if briefly disturbed at their kill. Ideally, the potential kill site will be approached between feeding sessions when the cougar is day bedded offsite. Initial kill site investigations are currently being conducted in the parent cougar project to establish the probability the kill site is detected by technicians at a later date. There have been no instances of abandonment. Additionally, many bait sites occupied by cougars are visited daily by technicians to switch memory cards in cameras, adjust location of placed bait carcasses, and/or refresh bait as needed to keep a cougar in the immediate area. Often times this is done for a series of days until researchers can attempt to conduct a capture. Even with these daily visits, patterns of bait site abandonment have not been observed. However, if these kill site visits and camera placements prove to disturb the cougar, and a pattern of kill site abandonment is observed, site visits and camera placement will cease.

Literature Cited

Anderson, C.R. and Lindzey, F. G.2003. Estimating cougar predation rates from GPS clusters. Journal of Wildlife Management 67, 307-316.

Bauer, J.W., Logan, K.A., Sweanor, L.L and Boyce, W.M. 2005. Scavenging Behavior in Puma. The Southwestern Naturalist. 50(4): 466-471.

Errington, P.L. 1946. Predation and vertebrate populations. Quarterly Review of Biology 21:145-245.

Hornocker, M. G. 1970. An analysis of mountain lion predation upon mule deer and elk in the Idaho primitive area. Wildlife Monographs 16, 3-39.

Knopff, K.H., Knopff A.A., Kortello, A., and Boyce M.S. 2010. Cougar kill rate and prey consumption in a multiprey system. Journal of Wildlife Management 74(7):1435-1447.

Krumm, C.E., Conner, M.M., Hobbs T.N., Hunter D.O., and Miller M.W. 2005. Mountain lions prey selectively on prion-infected mule deer. Biology letters 6, 209-211.

Logan, K.A and Sweanor, L.L. 2001. Desert Puma: Evolutionary ecology and conservation of an enduring carnivore. Washington D.C., USA: Island Press.

MacArthur, R.H., and E.R. Pianka. 1966. On the optimal use of a patchy environment. American Naturalist 100:603-609.

Miller M.W., Swanson H.M., Wolfe L.L., Quartarone F.G., Huwer S.L., et al. (2008) Lions and Prions and Deer Demise. PLoS ONE 3(12): e4019

Miller M.W., Williams E.S., McCarthy C.W., Sparaker T.R., Kreeger, T.J., Larson, C.T., and Thorne, E.T. 2000. Epizootiology of chronic wasting disease in free-ranging cervids in Colorado and Wyoming. Journal of Wildlife Diseases 36:4, 676-690.

Pulliam, H.R. 1974. On the theory of optimal diets. American Naturalist 108:59-75.

Schoener, T.W. 1971. Theory of feeding strategies. Ann. Rev. Ecol. Syst. 2:369-404.

Seidensticker IV, J.C., Hornocker, M.G., Wiles, W.V. and Messick, J.P. 1973. Mountain lion social organization in the Idaho primitive area. Wildlife Monographs (35).

Slobondkin, L. 1968. How to be a predator. American Zoologist 8:43-51.

Sweanor, L.L., Logan, K.A. and Hornocker, M.G. Puma responses to close approaches by researchers. Wildlife Society Bulletin 33:3, 905-913.

Williams, E.S and Young, S. 1980. Chronic wasting disease of captive mule deer: a spongiform encephalopathy. Journal of Wildlife Diseases 16:1, 89-98.

APPENDIX V

Continuous-Time Discrete-Space Models for Animal Movement Data

Ephraim M. Hanks
Department of Statistics, Colorado State University, Fort Collins, CO, U.S.A.
Email: hanks@stat.colostate.edu

Mevin B. Hooten
U.S. Geological Survey, Colorado Cooperative Fish and Wildlife Research Unit,
Colorado State University, Fort Collins, CO U.S.A.

Mat W. Alldredge
Colorado Division of Parks and Wildlife
Fort Collins, CO, U.S.A.

October 22, 2012

Abstract

The processes influencing animal movement and resource selection are complex and varied. Past efforts to model changing behavior over time used Bayesian statistical models with variable parameter space, such as reversible-jump Markov chain Monte Carlo approaches, which are computationally demanding and inaccessible to many practicioners. We present a continuous-time discrete-space (CTDS) model of animal movement that can be fit using standard generalized linear modeling (GLM) methods. This CTDS approach allows for the joint modeling of location-based as well as directional drivers of movement. Changing behavior over time is modeled using a varying-coefficient framework which maintains the computational simplicity of a GLM approach, and variable selection is accomplished using a group lasso penalty. We apply our approach to a study of two mountain lions (*Puma concolor*) in Colorado, USA.

KEYWORDS: Agent-Based Model; Animal Movement; Mountain Lion; Multiple Imputation; Varying-Coefficient Model.

1. Introduction

Animal telemetry data have been used extensively in recent years to study animal movement, space use, and resource selection (e.g., Johnson et al., 2011; Hanks et al., 2011; Fieberg et al., 2010). The ease with which telemetry data are being collected is increasing, leading to vast increases in the number of animals being monitored, as well as the temporal resolution at which telemetry locations are obtained (Cagnacci et al., 2010). This combination can result in huge amounts of telemetry data on a single animal population under study. Additionally, the processes driving animal movement are complex, varied, and changing over time. For example, animal behavior could be driven by the local environment (e.g., Hooten et al., 2010), by conspecifics or predator/prey interactions (e.g., Merrill et al., 2010), by internal states and needs (e.g., Nathan et al., 2008), or by memory (e.g., Van Moorter et al., 2009). The animal's response to each of these drivers of movement is also likely to change drastically over time (e.g., Nathan et al., 2008; Hanks et al., 2011; McClintock et al., 2012).

The large amount of telemetry data available, even for one animal, and the complex behavior displayed in animal movement results in a challenging situation for statistical modeling. There is no shortage of existing statistical models of animal movement; however, most of these models are computationally demanding, and most are inaccessible to the practicioner. For example, consider the agent-based model of animal movement of Hooten et al. (2010). The agent-based framework is highly flexible, allowing for location-based (static) and directional (dynamic) drivers of movement, but is computationally expensive. Analyzing the movement path of one animal using the approach of Hooten et al. (2010) can require computational time on the order of days using standard computing resources. The velocity-based framework for modeling animal movement of Hanks et al. (2011) allows for time-varying behavior through a changepoint model of response to drivers of movement, and is more computationally efficient than the approach of Hooten et al. (2010), requiring computational time on the order of hours. Similarly, the mechanistic state-switching approach of McClintock et al. (2012) allows for time-varying behavior through a state-switching approach. These three approaches use Bayesian statistical models, and both Hanks et al. (2011) and McClintock et al. (2012) allow for time-varying behavior by letting the model parameter space vary, either through a reversible-jump Markov chain Monte Carlo approach (Green, 1995) or the related birth-death Markov chain Monte Carlo approach (Stephens, 2000). These methods can be quite computationally demanding, require the user to tune the algorithm to ensure convergence, and can be inaccessible to many practitioners.

The agent-based model of Hooten et al. (2010) assumes a representation of the animal's movement path that is discrete in both space (grid cells) and time (fixed time intervals). The velocity-based movement model of Hanks et al. (2011) assumes a representation of the movement path that is continuous in space and discrete in time. The state-switching model of McClintock et al. (2012) assumes a representation of the movement path that is discrete in time and continuous in space.

In this paper, we present a continuous-time, discrete-space (CTDS) model for animal movement which allows for flexible modeling of an animal's response to drivers of movement in a computationally efficient framework. Instead of a Bayesian approach, we adopt a likelihood-based approach for inference, and instead of a state-switching or change-point model for changing behavior over time, we adopt a time-varying coefficient model. We also allow for variable selection using a lasso penalty. This CTDS approach is highly computationally efficient, requiring only minutes or seconds to analyze movement paths that would require hours using the approach of Hanks et al. (2011) or days using the approach of Hooten et al. (2010), allowing the analysis of longer movement paths and more complex behavior than has been previously possible. To make this CTDS approach for modeling animal movement and resource selection accessible to practitioners, code to implement this approach is available online (www.stat.colostate.edu/~hanks) in the form of a package for the R statistical computing environment (R Development Core Team, 2012) with worked examples.

In Section 2 *Preliminaries*, we describe the continuous-time continuous-space model of Johnson et al. (2008) which is used to make inference on the posterior predictive distribution of an animal's continuous movement path, conditioned on observed telemetry locations. We then describe the method of multiple imputation (Rubin, 1987) which we use to integrate over the uncertainty in the animal's continuous movement path. In Section 3 *Continuous-Time Discrete-Space Movement Model*, we describe the CTDS model for animal movement, and show how inference can be made on parameters in this model using standard software for generalized linear models (GLMs). In Section 4 *Time-Varying Behavior and Variable Selection* we use a varying-coefficient approach to model changing behavior over time, and use a lasso penalty for variable selection. In Section 5 *Drivers of Animal Movement* we discuss modeling potential covariates in the CTDS framework. In Section 6 *Example: Mountain Lions in Colorado* we illustrate our approach through an analysis of mountain lion (*Puma concolor*) movement in Colorado, USA. Finally, in Section 7 *Discussion* we discuss possible extensions to the CTDS approach.

2. Preliminaries

2.1 Continuous-Time Continuous-Space Movement Model

To model animal movement, we make use of the continuous time correlated random walk (CTCRW) model of Johnson et al. (2008) to characterize a distribution for the continuous path conditioned on observed telemetry data. Let $\mathbf{S} = \{\mathbf{s}(t), t = t_0, t_1, \ldots, t_T\}$ be a collection of time-referenced telemetry locations for an animal. If the animal's location and velocity at an arbitrary time t are $\mathbf{s}(t)$ and $\mathbf{v}(t)$, respectively, then the CTCRW model can be specified as follows, ignoring the multivariate notation for simplicity:

$$v(t) = \psi_1 + \frac{\psi_2 e^{-\psi_3 t}}{\sqrt{2\psi_3}} \omega \left(e^{2\psi_3 t}\right) \ , \tag{1}$$

$$s(t) = s(0) + \int_0^t v(u)du \ ,$$

where $\boldsymbol{\psi} = [\psi_1, \psi_2, \psi_3]$ control the movement and $\omega(t)$ is standard Brownian motion. This model can be discretized and formulated as a state-space model, which allows for efficient computation of discretized paths $\tilde{\mathbf{S}}$ at arbitrarily fine time intervals via the Kalman filter (Johnson et al., 2008). If a Bayesian framework is used for inference on $\boldsymbol{\psi}$, Johnson et al. (2008) shows how the posterior predictive distribution of the animal's continuous path $\tilde{\mathbf{S}}$ can be approximated using importance sampling. We will refer to the posterior predictive path distribution as $[\tilde{\mathbf{S}}|\mathbf{S}]$, where the bracket notation '$[\cdot]$' denotes a probability distribution.

2.2 Multiple Imputation

Our general strategy is to construct a model conditioned on the continuous path $\tilde{\mathbf{S}}$, and then integrate over the uncertainty in the posterior predictive distribution $[\tilde{\mathbf{S}}|\mathbf{S}]$ (e.g., Hooten et al., 2010; Hanks et al., 2011). If we treat the unobserved continuous path $\tilde{\mathbf{S}}$ as missing data, then we can make inference on model parameters using multiple imputation (Rubin, 1987). We motivate multiple imputation as posterior predictive inference on the imputation distribution within a Bayesian framework. Our treatment is similar to that of Rubin (1987) and Rubin (1996).

If we desire posterior predictive inference $[\boldsymbol{\theta}|\mathbf{S}]$ concerning environmentally relevant movement parameters $\boldsymbol{\theta}$, conditioned on the telemetry data $\mathbf{S}$ and the posterior predictive path distribution $[\tilde{\mathbf{S}}|\mathbf{S}]$, then we can write:

$$[\boldsymbol{\theta}|\mathbf{S}] = \int_{\tilde{\mathcal{S}}} [\boldsymbol{\theta}|\tilde{\mathbf{S}}][\tilde{\mathbf{S}}|\mathbf{S}]d\tilde{\mathbf{S}}. \tag{2}$$

In the multiple imputation literature, the posterior predictive path distribution $[\tilde{\mathbf{S}}|\mathbf{S}]$ is called the imputation distribution and specifies a statistical model for the missing data $\tilde{\mathbf{S}}$ conditioned on the observed data $\mathbf{S}$. We will use the CTCRW model of Johnson et al. (2008) as the imputation distribution $[\tilde{\mathbf{S}}|\mathbf{S}]$ in our CTDS approach to modeling animal movement. The CTCRW model is a mechanistic model of animal movment that has been successfully applied to studies of aquatic (Johnson et al., 2008) and terrestrial (Hooten et al., 2010) animals, and can represent a wide range of behavior.

Hooten et al. (2010) and Hanks et al. (2011) use composition sampling to obtain samples from the posterior predictive distribution $[\boldsymbol{\theta}|\mathbf{S}]$ in (2) by sampling iteratively from $[\boldsymbol{\theta}|\tilde{\mathbf{S}}]$ and $[\tilde{\mathbf{S}}|\mathbf{S}]$. Alternately, under the multiple imputation framework the posterior distribution $[\boldsymbol{\theta}|\mathbf{S}]$ is assumed to be asymptotically Gaussian. The posterior can then be approximated using only the posterior predictive mean and variance, which can be obtained using conditional mean and variance formulae:

$$
\begin{aligned}
E(\boldsymbol{\theta}|\mathbf{S}) &\approx \int_{\Theta} \boldsymbol{\theta} \int_{\tilde{\mathcal{S}}} [\boldsymbol{\theta}|\tilde{\mathbf{S}}][\tilde{\mathbf{S}}|\mathbf{S}] d\tilde{\mathbf{S}} d\boldsymbol{\theta} \\
&= \int_{\tilde{\mathcal{S}}} \left(\int_{\Theta} \boldsymbol{\theta}[\boldsymbol{\theta}|\tilde{\mathbf{S}}] d\boldsymbol{\theta} \right) [\tilde{\mathbf{S}}|\mathbf{S}] d\tilde{\mathbf{S}} \\
&= E_{\tilde{\mathbf{S}}|\mathbf{S}} \left(E(\boldsymbol{\theta}|\tilde{\mathbf{S}}) \right) \qquad (3)
\end{aligned}
$$

and likewise:

$$
\mathrm{Var}(\boldsymbol{\theta}|\mathbf{S}) \approx E_{\tilde{\mathbf{S}}|\mathbf{S}} \left(\mathrm{Var}(\boldsymbol{\theta}|\tilde{\mathbf{S}}) \right) + \mathrm{Var}_{\tilde{\mathbf{S}}|\mathbf{S}} \left(E(\boldsymbol{\theta}|\tilde{\mathbf{S}}) \right). \qquad (4)
$$

As the posterior distribution $[\boldsymbol{\theta}|\tilde{\mathbf{S}}]$ converges asymptotically to the sampling distribution of the maximum likelihood estimate (MLE) of $\boldsymbol{\theta}$, we can approximate $[\boldsymbol{\theta}|\tilde{\mathbf{S}}]$ by obtaining the asymptotic sampling distribution of the MLE. This allows us to use standard maximum likelihood approaches for inference, which can be much more computationally efficient than their Bayesian counterparts for this class of models.

The multiple imputation estimate $\hat{\boldsymbol{\theta}}_{MI}$ is typically obtained by approximating the integrals in (3) and (4) using a finite sample from the imputation distribution. The procedure can be summarized as follows:

1. Draw K different realizations (imputations) $\tilde{\mathbf{S}}^{(k)} \sim [\tilde{\mathbf{S}}|\mathbf{S}]$ from the imputation distribution.
2. For each realization, find the MLE $\hat{\boldsymbol{\theta}}^{(k)}$ and asymptotic variance $Var(\hat{\boldsymbol{\theta}}^{(k)})$ of the estimate based on the the full data: $(\tilde{\mathbf{S}})$.
3. Combine results from different imputations using finite sample versions of the conditional expectation (3) and variance (4) results:

$$
\hat{\boldsymbol{\theta}}_{MI} = \frac{1}{K} \sum_{k=1}^{K} \hat{\boldsymbol{\theta}}^{(k)} \qquad (5)
$$

$$
Var(\hat{\boldsymbol{\theta}}_{MI}) = \frac{1}{K} \sum_{k=1}^{K} var(\hat{\boldsymbol{\theta}}^{(k)}) + \frac{1}{K} \sum_{k=1}^{K} \left(\hat{\boldsymbol{\theta}}^{(k)} - \bar{\hat{\boldsymbol{\theta}}} \right)^2. \qquad (6)
$$

Equations (5) and (6) are the commonly used combining rules (Rubin, 1987) for multiple imputation estimators in the scalar case. When $\boldsymbol{\theta}$ is vector-valued, similar approaches can be used (Rubin, 1987).

3. Continuous-Time Discrete-Space Movement Model

Having described the multiple imputation framework, we now focus on specifying a model of animal response to drivers of movement $[(\tilde{\mathbf{S}}, \mathbf{S})|\boldsymbol{\theta}]$ that is flexible and computationally efficient. In doing so, we will assume a discrete (e.g., gridded) model for space (e.g., Hooten et al., 2010), and model animal movement as a continuous-time random walk through the discrete, gridded space.

Let the the study area be defined as a graph $(\mathbf{G}, \mathbf{A})$ of M locations $\mathbf{G} = (G_1, G_2, \ldots, G_M)$ connected by "edges" $\boldsymbol{\Lambda} = \{\lambda_{ij} : i \sim j, i = 1, \ldots, M\}$ where $i \sim j$ means that the nodes G_i and G_j are directly connected. For example, in a gridded space each grid cell is a node and the edges connect each grid cell to its first-order neighbors (e.g., a rook's neighborhood). In typical studies, the spatial resolution of the grid cells in $\mathbf{G}$ will be determined by the resolution at which environmental covariates that may drive animal movement and selection are available.

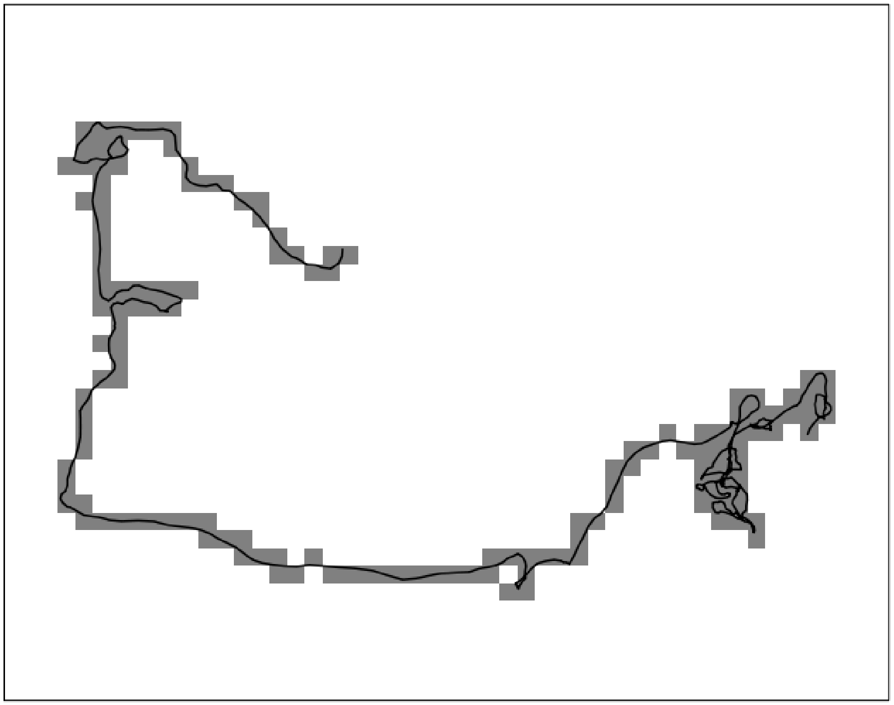
A path realization $\tilde{\mathbf{S}}$ from the CTCRW model is continuous in time and space (Figure 1). If we consider a discrete, gridded space $\mathbf{G}$, then the continuous-time, continuous-space path $\tilde{\mathbf{S}}$ is represented by a continuous-time, discrete-space path $(\mathbf{g}, \boldsymbol{\tau})$ consisting of a sequence of grid cells $\mathbf{g} = (G_{i_1}, G_{i_2}, \ldots, G_{i_T})$ transversed by the animal's continuous-space path and the residence times $\boldsymbol{\tau} = (\tau_1, \tau_2, \ldots, \tau_T)$ in each grid cell.

In practice, this transformation from continuous space to discrete space results in a compression of the data to a temporal scale that is relavent to the resolution of the environmental covariates that may be driving movement and selection. For example, if an animal is moving slow relative to the time it takes to traverse a grid cell in $\mathbf{G}$, then the quasi-continuous path $\tilde{\mathbf{S}}$ may contain a long sequence of locations within one grid cell. The discrete-space representation of the path represents this long sequence of locations as one observation (a grid cell G_{it} and residence time τ_t). This data compression is especially relevant for telemetry data, in which observation windows can span years or even decades for some animals.

3.1 Random Walk Model

The discrete-space representation $(\mathbf{g}, \boldsymbol{\tau})$ of the movement path allows us to use standard discrete-space random walk models to make inference about possible drivers of movement. While we will relax this assumption later to account for temporal autocorrelation in movement behavior, we initially assume that the the t-th observation (G_{i_t}, τ_t) in the sequence is independent of all other observations in the sequence. Under this assumption, the likelihood of the sequence of transitions $\{(G_{i_t} \rightarrow G_{i_{t+1}}, \tau_t), t = 1, 2, \ldots, T\}$ is just the product of the likelihoods of each individual observation. We will focus on modeling each transition $(G_{i_t} \rightarrow G_{i_{t+1}}, \tau_t)$, and will drop the t subscript in this section to simplify notation.

Figure 1: Continuous-time continuous-space and continuous-time discrete-space representations of an animal's movement path.



If an animal is in cell G_i at time t, then define the Poisson rate of transition from cell G_i to a neighboring cell G_j as

$$\lambda_{ij}(\boldsymbol{\beta}) = \exp\{\mathbf{x}'_{ij}\boldsymbol{\beta}\} \tag{7}$$

where $\mathbf{x}_{ij}$ is a vector containing covariates related to drivers of movement specific to cells i and j, and $\boldsymbol{\beta}$ is a vector of parameters that define how each of the covariates in $\mathbf{x}_{ij}$ drive animal movement. The total transistion rate λ_i from cell i is the sum of the transition rates to all neighboring cells: $\lambda_i(\boldsymbol{\beta}) = \sum_{j\sim i} \lambda_{ij}(\boldsymbol{\beta})$ and the time τ that the animal resides in cell G_i is exponentially-distributed with rate parameter equal to the total transition rate $\lambda_i(\boldsymbol{\beta})$:

$$[\tau|\boldsymbol{\beta}] = \lambda_i(\boldsymbol{\beta}) \exp\left\{-\tau\lambda_i(\boldsymbol{\beta})\right\}. \tag{8}$$

When the animal transitions from cell G_i to one of its neighbors, the probability of transitioning to cell G_k, an event we denote as $G_i \to G_k$, follows a multinomial distribution with probability proportional to the transition rate λ_{ik} to cell G_k:

$$[G_i \to G_k|\boldsymbol{\beta}] = \frac{\lambda_{ik}(\boldsymbol{\beta})}{\sum_{j\sim i}\lambda_{ij}(\boldsymbol{\beta})} = \frac{\lambda_{ik}(\boldsymbol{\beta})}{\lambda_i(\boldsymbol{\beta})}. \tag{9}$$

The residence time and eventual destination are independent events, and so the likelihood of the observation $(G_i \to G_k, \tau)$ is the product of the likelihoods of its parts:

$$\begin{aligned}[G_i \to G_k, \tau|\boldsymbol{\beta}] &= \frac{\lambda_{ik}(\boldsymbol{\beta})}{\lambda_i(\boldsymbol{\beta})} \cdot \lambda_i(\boldsymbol{\beta}) \exp\left\{-\tau\lambda_i(\boldsymbol{\beta})\right\} \\ &= \lambda_{ik}(\boldsymbol{\beta}) \exp\left\{-\tau\lambda_i(\boldsymbol{\beta})\right\}. \qquad (10)\end{aligned}$$

3.2 Latent Variable Representation

We now introduce a latent variable representation of the transition process that is equivalent to (10), but allows for inference within a standard generalized linear modeling framework. For each j such that $i \sim j$, define z_j as

$$z_j = \begin{cases} 1 & , \ G_i \to G_j \\ 0 & , \ \text{o.w.} \end{cases}$$

and let

$$[z_j, \tau|\boldsymbol{\beta}] \propto \lambda_{ij}^{z_j} \exp\left\{-\tau\lambda_{ij}(\boldsymbol{\beta})\right\} \tag{11}$$

where n_i is the number of neighbors of the i-th grid cell. Then the product of $[z_j, \tau|\boldsymbol{\beta}]$ over all j such that $i \sim j$ is equivalent to the likelihood (10) of the observed transition:

$$\begin{aligned}\prod_{j:i\sim j}[z_j,\tau|\boldsymbol{\beta}] &= \prod_{j:i\sim j}\lambda_{ij}^{z_j}\exp\left\{-\tau\lambda_{ij}(\boldsymbol{\beta})\right\}\\ &= \lambda_{ik}(\boldsymbol{\beta})\exp\left\{-\tau\lambda_i(\boldsymbol{\beta})\right\}\ \text{, where } G_i\to G_k\\ &= [G_i\to G_k,\tau|\boldsymbol{\beta}]\end{aligned}$$

The benefit of this latent variable representation is that the likelihood of $z_j, \tau|\boldsymbol{\beta}$ in (11) is equivalent to the kernel of the likelihood in a Poisson regression with the canonical log link, where z_j are the observations and $\log(\tau)$ is an offset or exposure term. The likelihood of the entire continuous-time, discrete-space path $(\mathbf{g},\boldsymbol{\tau})$ can be written (returning t to the notation) as:

$$[\mathbf{g},\boldsymbol{\tau}|\boldsymbol{\beta}] = [\mathbf{Z},\boldsymbol{\tau}|\boldsymbol{\beta}] \propto \prod_{t=1}^{T}\prod_{i_t\sim j_t}\left[\lambda_{i_tj_t}^{z_{j_t}}(\boldsymbol{\beta})\exp\{-\tau_t\lambda_{i_tj_t}(\boldsymbol{\beta})\}\right] \tag{12}$$

where $\mathbf{Z} = (\mathbf{z}_1,\ldots,\mathbf{z}_T)'$ is a vector containing the latent variables $\mathbf{z}_i = (z_{i_1}, z_{i_2},\ldots,z_{i_K})'$ for each grid cell in the discrete-space path. Inference can be made on $\boldsymbol{\beta}$ in (12) using standard Poisson GLM approaches (e.g., maximum likelihood). This provides a computationally efficient framework for the statistical analysis of potential drivers of movement within the multiple imputation framework of Section 2.2. Multiple path realizations (imputations) can be drawn from $[\tilde{\mathbf{S}}|\mathbf{S}]$. Each continuous path $\tilde{\mathbf{S}}$ can then be transformed into a CTDS path $(\mathbf{g},\boldsymbol{\tau})$, which can then be used to make inference on $\boldsymbol{\beta}$ using (12). The results from the multiple imputed paths can then be combined using (5) and (6), resulting in a multiple imputation estimate $\hat{\boldsymbol{\beta}}_{MI}$ and estimated variance $Var(\hat{\boldsymbol{\beta}}_{MI})$.

4. Time-Varying Behavior and Variable Selection

In this section we describe how covariate effects can be allowed to vary over time using a varying-coefficient model, and how variable selection can be accomplished through shrinkage estimation.

4.1 Changing Behavior Over Time

Animal behavior and response to drivers of movement can change significantly over time. These changes can be driven by external factors such as changing seasons (e.g., Grovenburg et al., 2009) or predator/prey interactions (e.g., Lima, 2002), or by internal factors such as internal energy levels (e.g., Nathan et al., 2008). The most common approach to modeling time-varying behavior in animal movement is state-space modeling, typically within a Bayesian framework (e.g., Morales et al., 2004; Jonsen et al., 2005; Getz and Saltz, 2008; Nathan et al., 2008; Forester et al., 2009; Gurarie et al., 2009; Merrill et al., 2010). In the state-space framework, the animal is assumed to exhibit a number of behavioral states, each characterized by a distinct pattern of movement or response to drivers of movement. The number of states can be either known and specified in advance by the researcher (e.g., Morales et al., 2004; Jonsen et al., 2005) or allowed to be random (e.g., Hanks et al., 2011; McClintock et al., 2012).

State-space models are an intuitive approach to modeling changing behavior over time, but there are limits to the complexity that can be modeled using this approach. Allowing the number of states to be unknown and random requires a Bayesian approach with a changing parameter space. This is typically implemented using reversible-jump MCMC methods (e.g., Green, 1995; McClintock et al., 2012; Hanks et al., 2011), which are computationally expensive and can be difficult to tune. Our approach is to use a computationally efficient GLM (12) to analyze parameters related to drivers of animal movement. Instead of using the common state-space approach, we employ varying-coefficient models (e.g., Hastie and Tibshirani, 1993) to model time-varying behavior in animal movement.

The model for the Poisson transition rate (7) will typically contain an intercept term and multiple covariates $\{x\}$. For simplicity in notation, consider the case where there is only one covariate x in the model (7) and no intercept term. The varying-coefficient approach generalizes easily to the case where there are multiple covariates and an intercept term in (7). In a time-varying-coefficient model, we allow the parameter $\beta(t)$ to vary over time in a functional (continuous) fashion. The transition rate (7) then becomes:

$$\lambda_{ij}(\beta(t)) = \exp\{x_{ij}\beta(t)\}$$

where t is the time of the observation and x_{ij} is the value of the covariate related to the Poisson rate of moving from cell i to cell j. The functional regressor $\beta(t)$ is typically modeled as a linear combination of n_{spl} spline basis functions $\{\boldsymbol{\phi}_k(t), k = 1, \ldots, n_{spl}\}$:

$$\beta(t) = \sum_{k=1}^{n_{spl}} \alpha_k \boldsymbol{\phi}_k(t).$$

B-spline basis functions are among the most-widely used choices for $\{\phi_k(t)\}$, and are appropriate in most cases. Fourier basis functions are commonly used for $\{\phi_k(t)\}$ when the behavior is thought to be periodic (e.g., diurnal variation in behavior).

Under this varying-coefficient specification, (7) can be rewritten as

$$\begin{aligned}\lambda_{ij}(\beta(t)) &= \exp\{x_{ij}\beta(t)\} \\ &= \exp\left\{x_{ij}\sum_{k=1}^{n_{spl}} \alpha_k \boldsymbol{\phi}_k((t))\right\} \\ &= \exp\left\{\boldsymbol{\phi}_{ij}' \boldsymbol{\alpha}\right\} \qquad (13)\end{aligned}$$

where $\boldsymbol{\alpha} = (\alpha_1, \ldots, \alpha_{n_{spl}})'$ and $\boldsymbol{\phi}_{ij} = x_{ij} \cdot (\phi_1(t), \ldots, \phi_{n_{spl}}(t))'$. The result is that the varying-coefficient model can be written as a GLM with a modified design matrix. This specification provides a flexible framework for allowing the effect of a driver of movement (x) to vary over time that is computationally efficient and simple to implement using standard GLM software.

4.2 Shrinkage Estimation

The model we have specified in (12) is likely to be overparameterized, especially if we utilize a varying-coefficient model (13). Animal movement behavior is complex, and a typical study could envision a large number of potential drivers of movement, but an animal's response to each of those drivers of movement is likely to change over time, with only a few drivers being relevant at any one time. Under these assumptions, many of the parameters α_k in (13) are likely to be very small or zero. Multicollinearity is also a potential problem, as many potential drivers of movement could be correlated with each other.

We propose a shrinkage estimator of $\boldsymbol{\alpha}$ using a lasso penalty (Tibshirani, 1996). The typical maximum likelihood estimate of $\boldsymbol{\alpha}$ is obtained by maximizing the likelihood $[\mathbf{Z}, \boldsymbol{\tau}|\boldsymbol{\alpha}]$ from (12), or equivalently by maximizing the log-likelihood $\log[\mathbf{Z}, \boldsymbol{\tau}|\boldsymbol{\alpha}]$. The lasso estimate is obtained by maximizing the penalized log-likelihood, where the penalty is the sum of the absolute values of the regression parameters $\{\alpha_k\}$:

$$\hat{\boldsymbol{\alpha}}_{\text{lasso}} = \max_{\boldsymbol{\alpha}} \left\{ \log[\mathbf{Z}, \boldsymbol{\tau}|\boldsymbol{\alpha}] - \gamma \sum_{k=1}^{K} |\alpha_k| \right\}. \tag{14}$$

As the tuning parameter γ increases, the absolute values of the regression parameters $\{\alpha_k\}$ are "shrunk" to zero, with the parameters that best describe the variation in the data being shrunk more slowly than parameters that do not. Cross-validation is typically used to set the tuning parameter γ at a level that optimizes the model's predictive power.

Shrinkage approaches such as the lasso are well-developed for GLMs, and computationally-efficient methods are available for fitting GLMs to data (e.g., Friedman et al., 2010). Recent work has also applied the lasso to multiple imputation estimators (e.g., Chen and Wang, 2011). The main challenge in applying the lasso to multiple imputation is that a parameter may be shrunk to zero in the analysis of one imputation but not in the analysis of another. The solution is to use a so-called group lasso (Yuan and Lin, 2006), in which a group of parameters is constrained to either all equal zero or all be non-zero together.

The lasso can be seen as a constrained optimization problem, with $\boldsymbol{\alpha}_{\text{lasso}}$ minimizing the mean squared error subject to the constraint that $||\boldsymbol{\alpha}_{\text{lasso}}||_1 \leq \nu$, where $||\cdot||$ is the L-1 norm and the value of ν is determined by the tuning parameter γ. As the estimate $\boldsymbol{\alpha}_{\text{lasso}}$ typically falls on the boundary of the constrained space, conventional approaches for estimating the standard error of $\boldsymbol{\alpha}_{\text{lasso}}$ are unavailable. Bayesian versions of the lasso (Park and Casella,

2008) and group lasso (Raman et al., 2009) provides alternatives that allow for estimating the uncertainty about the parameters $\boldsymbol{\alpha}_{\text{lasso}}$ through posterior analysis.

The Bayesian approach entails significantly more computational complexity, and may not be as accessible to practitioners. We instead focus on the likelihood-based stacked lasso estimate of Chen and Wang (2011). In this estimate, instead of computing the lasso estimate $\boldsymbol{\alpha}_{\text{lasso}}$ for each imputation individually, and then combining the results using (5) and (6), the imputed data from all estimates are "stacked" together and a group lasso estimate is obtained for the combined data. We note that this likilihood-based stacked lasso approach does not allow for the estimation of the variance of $\boldsymbol{\alpha}_{\text{lasso}}$. If uncertainty estimates are a priority, we recommend choosing a parsimonious selection of potential drivers of movement *a priori* that exhibit little multicollinearity and computing the traditional multiple imputation estimates $\hat{\boldsymbol{\alpha}}_{MI}$.

5. Drivers of Animal Movement

We now provide some examples showing how a range of hypthesized drivers of movement could be modeled within the CTDS framework. Following Hooten et al. (2010), we consider two distinct categories for drivers of movement from cell G_i to cell G_j: static drivers ($\{p_{ki}, k = 1, 2, \ldots, K\}$) which are determined only by the characteristics of cell G_i, and dynamic drivers ($\{q_{lij}, l = 1, 2, \ldots, L\}$) which vary with direction of movement. Under a time-varying coefficient model for each driver, the transition rate (7) from cell G_i to cell G_j is

$$\lambda_{ij}\left(\beta(t)\right) = \exp\left\{\beta_0(t) + \sum_{k=1}^{K} p_{ki}\beta_k(t) + \sum_{l=1}^{L} q_{lij}\beta_l(t)\right\} \tag{15}$$

where $\beta_0(t)$ is a time-varying intercept term, $\{\beta_k(t)\}$ are time-varying effects related to static drivers of movement, and $\{\beta_l(t)\}$ are time-varying effects related to dynamic drivers of movement. We consider both static and dynamic drivers in what follows.

5.1 Static (Location-Based) Drivers of Movement

Hooten et al. (2010) denote location-based, non-directional drivers of movement as static drivers of movement. Static drivers of movement can be used to examine differences in animal movement rates that can be explained by the environment an animal resides in. In the CTDS context, static drivers would be covariates dependent only on the characteristics of the cell where the animal is currently located. Large positive (negative) values of the corresponding $\beta_k(t)$ would indicate that the animal tends to transition quickly (slowly) from a cell containing the cover type in question.

5.2 Dynamic (Directional) Drivers of Movement

In contrast to static drivers, which describe the effect that the local environment in which the animal resides has on movement rates, dynamic drivers of movement (Brillinger et al., 2001; Hooten et al., 2010; Hanks et al., 2011) capture directional selection by the individual.

A dynamic driver of movement is defined by a vector which points toward (or away) from something that is hypothesized to attract (or repel) the animal in question. Let $\mathbf{v}_l$ be the vector corresponding to the l-th dynamic driver of movement. In the CTDS model for animal movement, the animal can only transition from cell G_i to one of its neighbors $G_j : j \sim i$. Let $\mathbf{w}_{ij}$ be a unit vector pointing from the center of cell G_i in the direction of the center of cell G_j. Then the covariate q_{lij} relating the l-th dynamic driver of movement to the transition rate from cell G_i to cell G_j is the dot (or inner) product of $\mathbf{v}_l$ and $\mathbf{w}_{ij}$:

$$q_{lij} = \mathbf{v}_l' \mathbf{w}_{ij}.$$

Then p_{lij} will be positive when $\mathbf{v}_l$ points nearly in the direction of cell G_j, negative when when $\mathbf{v}_l$ points directly away from cell G_j, and zero if $\mathbf{v}_l$ is perpendicular to the direction from cell G_i to cell G_j.

5.3 Examples

We now provide multiple examples of drivers of movement to illustrate the range of effects that can be modeled using this framework.

5.3.1. Overall Movement Rate

The intercept term $\beta_0(t)$ in (15) can be seen as a static driver of movement in which $p_{0i} = 1$ for every cell $G_i \in \mathbf{G}$. This intercept term controls the animal's overall rate of transition from any cell, and thus models the animal's overall movement rate. Allowing the intercept parameter $\beta_0(t)$ to vary over time could reveal changes in activity levels over time. For example, we might expect $\beta_0(t)$ to be larger at night for nocturnal species and smaller during the day.

5.3.2. Movement Response to Land Cover Types

Indicator variables could be used to examine how animal movement differs between different landscape cover types (e.g., forest vs. plains) by setting $p_{ki} = 1$ for each cell G_i that is classified as containing the k-th cover type. As in the case of the static intercept, allowing the parameter $\beta_k(t)$ related to the k-th cover type to vary over time can reveal variation in an animal's movement pattern through the cover type. For example, an animal may move quickly through open terrain during the day, but may move more slowly through the same terrain at night.

5.3.3. Environmental Gradients

Animals may use environmental gradients for navigation. For example, a mule deer might move predominantly in the direction of increasing elevation during a spring migration (e.g., Hooten et al., 2010), or a seal might follow gradients in sea surface temperature to navigate toward land (e.g., Hanks et al., 2011). Such effects can be modeled by including a dynamic driver of movement in (15) defined by a gradient vector $\mathbf{v}_l$ which points from the center of cell G_i in the direction of steepest increase in the covariate x_l. Positive values of β_l indicate that the animal moves generally towards cells with higher values of x_l, while negative values of β_l indicate that the animal moves generally towards cells with lower values of x_l.

5.3.4. Activity Centers

Many animals exhibit movement patterns that are centered on a location in space. This central location may be temporary, such as a kill site for a predator (e.g., Knopff et al., 2009), or more permanent, such as a den for a central place forager (e.g., Hanks et al., 2011; McClintock et al., 2012). This concept is the basis for the relatively new class of spatial capture-recapture models (e.g., Royle and Young, 2008). Movement around an activity center can be modeled in the CTDS framework by including a dynamic driver of movement in (15) defined by a vector $\mathbf{v}_l$ which points from the center of cell G_i to the location of the activity center. Then a positive value for β_l would indicate that the animal is generally drawn toward this activity center. If the activity center is considered to be temporary (such as a kill site for a predator), than a time-varying-coefficient model should be used. The variable selection obtained through the lasso estimate can indicate the range of time in which the animal's movement is centered around the activity center. If the activity center is considered to be permanent through the duration of the study, a varying-coefficient model may not be needed.

Under the likelihood-based specification of the CTDS model for animal movement, it is necessary to specify the locations of all hypothesized activity centers beforehand. In Section 6, we show an example of the specification of hypothesized activity centers (potential kill sites for mountain lions) using the original telemetry data. If a Bayesian formulation of the CTDS model were used, then the location of hypothesized activity centers could be random, and inference could be made on their locations jointly with inference on the movement parameters, as in spatial capture-recapture models (e.g., Royle and Young, 2008).

5.3.5. Conspecific Interaction

An animal's movement patterns can be greatly affected by interaction with conspecifics. For example, one animal could follow the trail left by another animal, two animals could avoid one another by changing course when they become close enough to sense the other animal, or a pair of animals could maintain proximity as they move together across the landscape. While there are many possible approaches to modeling such dependence in behavior, we choose to model each of these interactions through the inclusion of dynamic effects in the CTDS modeling framework. For example, a dynamic driver could be included in the movement model for one animal that is defined by a vector pointing to the current location of another animal to examine whether the animal being modeled is attracted to ($\beta_l > 0$) or avoids ($\beta_l < 0$) the conspecific.

5.3.6. Directional Persistence

The CTCRW model of Johnson et al. (2008) is based on a correlated random walk model for velocity that allows for directional persistence in animal movement. So far, we have assumed that each discrete movement step in our CTDS model is independent, but this assumption is not met if the animal exhibits any directional persistence. To account for directional persistence in the CTDS approach, we use an autoregressive approach by including a dynamic driver of movement at each discrete movement step that is defined by a vector pointing in the direction of the previous move. If the animal moved west in the previous discrete movement step, then the autoregressive vector for the next step points west as well. Positive values of the β related to this dynamic driver of movement indicate that the animal is likely to maintain its direction of movement over time.

5.4 Spatial and Temporal Scale

The choice of scale for a study can greatly influence results (e.g., Boyce, 2006). When speaking of the scale of a study, one could look at the grain, or resolution, at which the process is modeled, or the extent (coverage) over which the process is modeled. The spatial and temporal extent of a study of animal movement are determined by the telemetry data and the posterior predictive path distribution $[\tilde{\mathbf{S}}|\mathbf{S}]$. However, when implementing the CTDS approach, the researcher must make three choices pertaining to the grain or resolution: (1) the temporal scale at which the CTCRW movement path of the animal is sampled, (2) the spatial scale of the grid over which the discrete-space movement will be modeled, and (3) the temporal scale of the varying coefficient model, which is determined by the number and resolution of spline knots in the spline basis expansion.

As the CTCRW model of Johnson et al. (2008) is a continuous-time model, we recommend sampling from the movement path at as fine an interval as is feasible. In practice, this will be limited by computational resources and the size of the study. The temporal resolution needs to be fine enough that realizations from the posterior predictive path distribution $[\tilde{\mathbf{S}}|\mathbf{S}]$ are quasi-continuous and adequately capture the residence time τ in each grid cell in the CTDS representation of the movement path.

The choice of spatial resolution of the raster grid on which the CTDS process occurs implicitly specifies a time scale at which the movement process is modeled. Coarser spatial resolution (larger grid cells) will correspond to longer residence times $\boldsymbol{\tau}$ in the CTDS mode. The spatial resolution should be chosen so that the time scale at which an animal transitions from one grid cell to another is a time scale at which the animal in question can make meaningful choices about movement and resource selection. The time scale implicit in the choice of spatial resolution can be examined by plotting a histogram of the residence times in the CTDS representation of the movement path.

If the lasso penalty is used, then it is common to choose a saturated spline basis expansion in the varying-coefficient model, where one spline knot is specified at each data point in time. We recommend specifying a spline basis expansion with knots at a similar temporal resolution to the temporal resolution of the telemetry data. The lasso penalization will shrink the overparameterized expansion to a more-parsimonious model that best fits the data. While a finer temporal resolution could be used, the posterior predictive path distribution is unlikely to show changes in behavior at time scales smaller than the time scale of the original data. Using a coarser temporal resolution will force $\beta(t)$ to be smooth. This would imply that changes in animal behavior are gradual and occur at time scales larger than the time scale of the data.

6. Example: Mountain Lions in Colorado

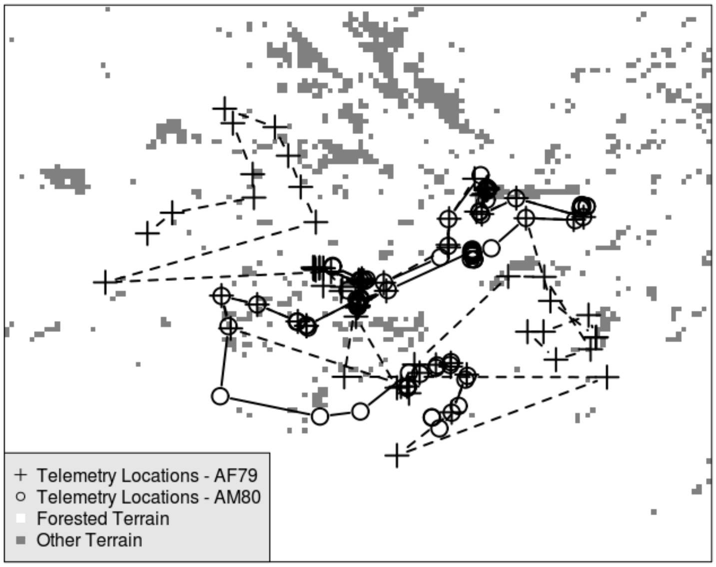
We illustrate our CTDS random walk approach to modeling animal movement through a study of mountain lions (*Puma concolor*) in Colorado, USA. As part of a larger study, a female mountain lion, designated AF79, and her subadult cub, designated AM80, were fitted with global positioning system (GPS) collars set to transmit location data every 3 hours. We analyze the location data $\mathbf{S}$ from the first 18 days of location information for these two animals (Figure 2).

We fit the CTCRW model of Johnson et al. (2008) to both animals' location data using the 'crawl' package (Johnson, 2011) in the R statistical computing environment (R Development Core Team, 2012). Ten imputations from the posterior distribution of the quasi-continuous path distribution $[\tilde{\mathbf{S}}|\mathbf{S}]$ were obtained at one-minute intervals. The result is a quasi-continuous path at extremely fine temporal resolution for each imputation.

For covariate data, we used a landcover map of the state of Colorado created by the Colorado Vegetation Classification Project (http://ndis.nrel.colostate.edu/coveg/), which is a joint project of the Bureau of Land Management and the Colorado Division of Wildlife. The map contained gridded landcover information at 100m square resolution. Figure 3

Figure 2: Telemetry data for a female mountain lion (AF79) and her male cub (AM80). A static covariate was defined by landcover that was not predominanty forested (a). Potential kill sites were identified, and a dynamic covariate defined by a vector pointing toward the closest kill site (b) was also used in the CTDS model.

(a) Static Cover - Not Forest



(b) Distance to Nearest Potential Kill Site For AM80

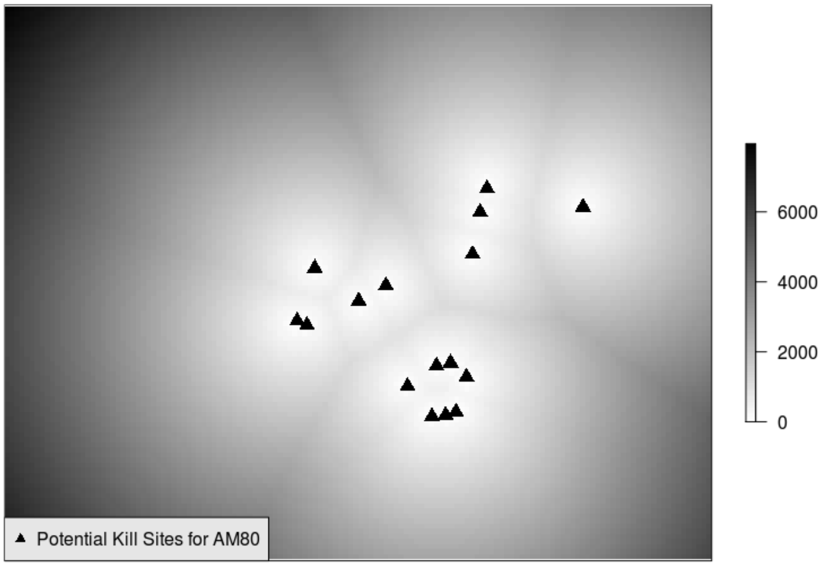
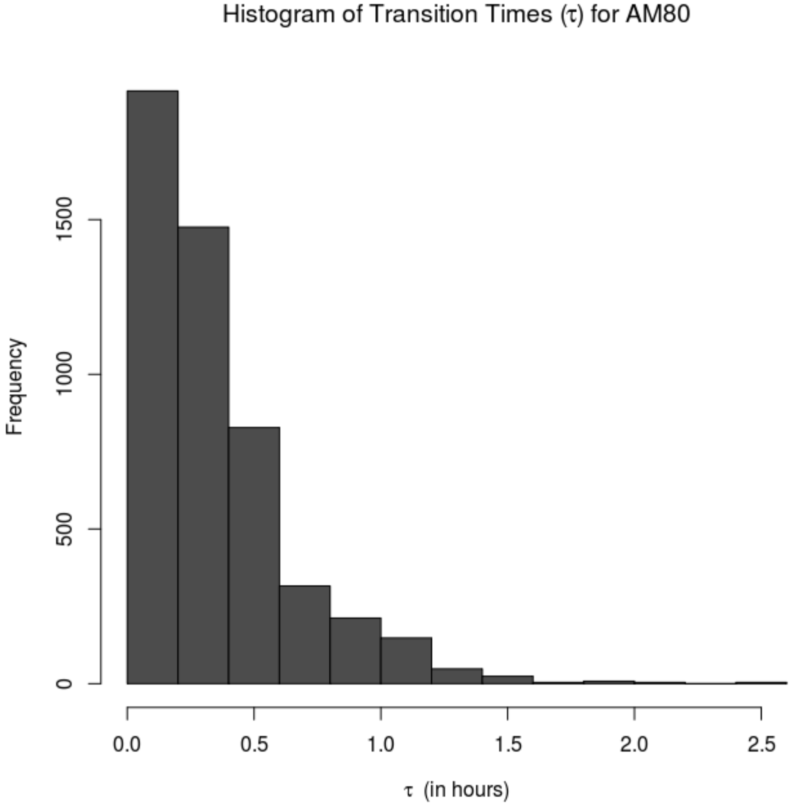


Figure 3: Residence times in each 100m-square grid cell in the continuous-time discrete-space representation of the movement path of a male mountain lion (AM80).



shows a histogram of the residence times $\boldsymbol{\tau}$ in each grid cell in the CTDS representation of the movement path of AM80. The area traveled by the two animals in our study was predominantly forested. To assess how the animals' movement differed when in terrain other than forest, we created an indicator covariate where all forested grid cells were assigned a value of zero, and all cells containing other cover types, including developed land, bare ground, grassland, and shrubby terrain, were assigned a value of one (Figure 2a). This covariate and an intercept covariate were used as static (locational) covariates in the CTDS model for both animals.

For each animal, we created a set of potential kill sites (PKS) by examining the original GPS location data (Figure 2). A location was classified as a PKS if two or more GPS locations were found within 200m of the site within a six-day period (Knopff et al., 2009). We then created a covariate raster layer containing the distance to the nearest PKS for each grid cell (Figure 2). A dynamic (directional) covariate defined by a vector pointing towards the nearest PKS was included in the CTDS model for both animals.

To examine how the movement path of the cub AM80 affected the movement path of the mother AF79, we included a directional covariate in the CTDS model for AF79 defined by a vector pointing from the mother's location to the cub's location at each time point. Similarly, we included a directional covariate in the CTDS model for AM80 defined by a vector pointing from the cub's location to the mother's location at each time point.

For each animal, we also included a directional covariate pointing in the direction of the most recent movement at each time point. This covariate measures the strength of correlation between moves, and thus the strength of the directional persistence shown by the animal's discrete-space movement path. As we are assuming an underlying correlated movement model (the CTCRW model of Johnson et al. (2008)), we expect the CTDS movement to be correlated in time as well.

To allow for varying behavior over time, we used a varying-coefficient model for each covariate in the model. For all covariates except the directional covariate related to directional persistence, we used a B-spline basis expansion, with regularly spaced spline knots at hourly intervals. For the covariate related to directional persistence, we used a B-spline basis expansion with regularly spaced knots at 3-hour intervals. By choosing a coarser temporal resolution in our varying-coefficient model for directional persistence than for our other covariates we imply that the directional persistence should vary at larger time scales than the animal's response to other covariates, which drive animal movement at finer time scales.

We fit the CTDS model for each path using the 'glmnet' R package (Friedman et al., 2010), using a lasso penalty, with tuning parameter chosen to minimize the average squared error of the fit in a 10-fold cross-validation.

6.1 Results

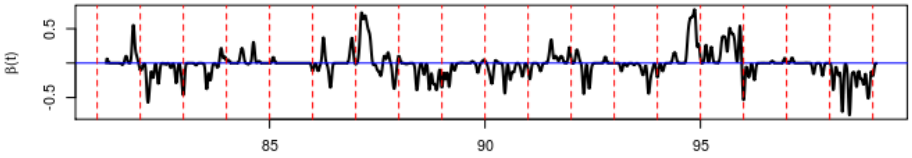
The time-varying results for the static and dynamic drivers of movement for AF79 are shown in Figure 4, with the corresponding results for AM80 shown in Figure 5. As we used a lasso penalty, we can only obtain point estimates (confidence intervals are unavailable) of the time-varying effects $\{\beta(t)\}$. A comparison of the differences between the results for AF79 and AM80 yields some insight into how the movement patterns of these two animals differ.

The static intercept effect measures the animal's general movement rate over time. Figure 4(a) and Figure 5(a) show the time-varying deviation from the grand mean for each animal. The static intercept for both lions tends to be higher during nighttime hours than during the day, indicating higher overall rates of movement during nocturnal hours.

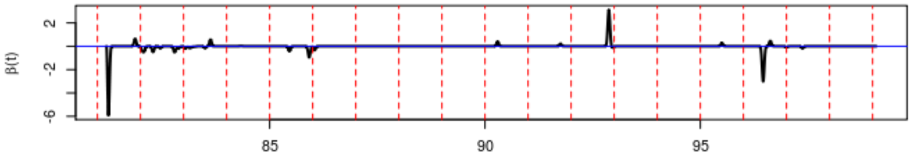
The static (location-based) response to non-forrested terrain (Figure 4b and Figure 5b) is zero for much of the study window for both AF79 and AM80. Forested terrain makes up over 90% of the study area, thus the mountain lions encounter different terrain infrequently.

Figure 4: Time-varying results for the static and dynamic covariates in the continuous-time discrete-space model for a female mountain lion (AF79).

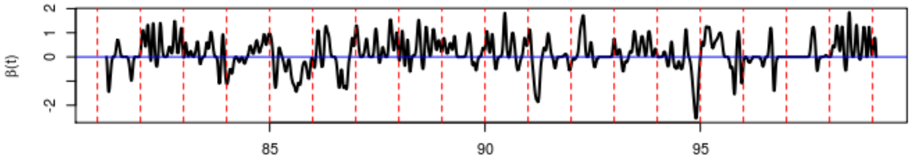
(a) Static Intercept (Deviation from Overall Intercept = 2.78)



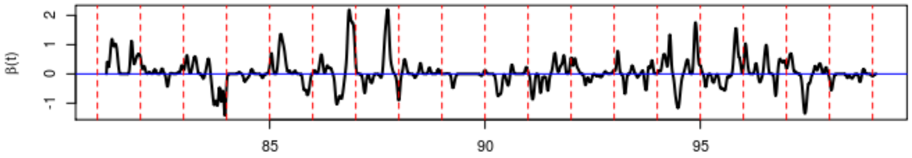
(b) Static Cover - Not Forest



(c) Dynamic Covariate - Direction to Nearest Potential Kill Site



(d) Dynamic Covariate - Direction to AM80



(e) CRW Autoregressive Parameter

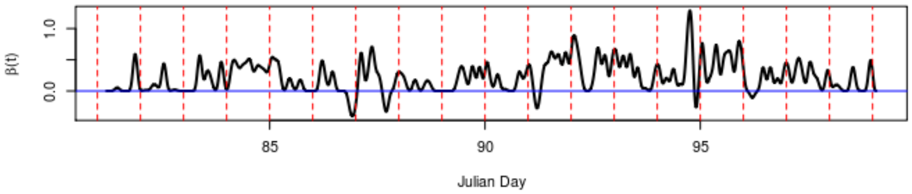
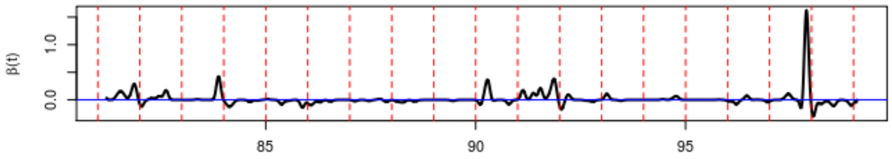
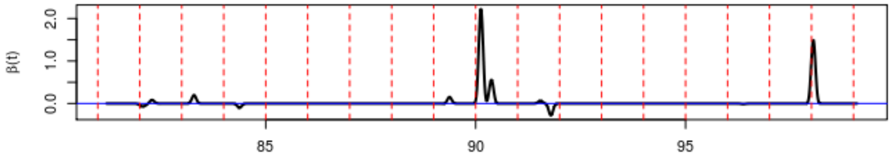


Figure 5: Time-varying results for the static and dynamic covariates in the continuous-time discrete-space model for a male mountain lion (AM80).

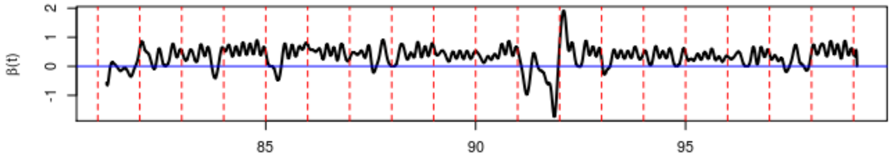
(a) Static Intercept (Deviation from Overall Intercept = 3.04)



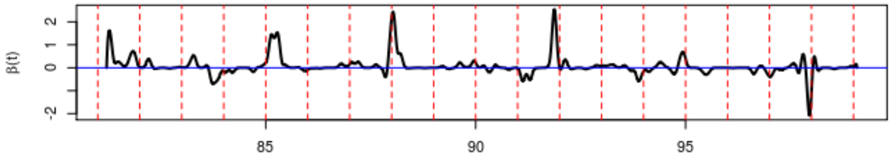
(b) Static Cover - Not Forest



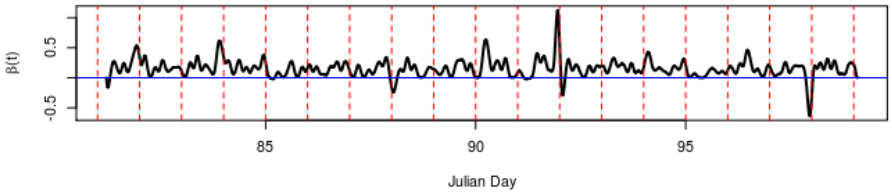
(c) Dynamic Covariate - Direction to Nearest Potential Kill Site



(d) Dynamic Covariate - Direction to AF79



(e) CRW Autoregressive Parameter



The male cub (AM80) moves through non-forested terrain at a relatively high rate of speed, while the female's response is mixed.

The male cub (AM80) has a very consistent positive response to the dynamic covariate pointing toward the nearest kill site (Figure 5c), indicating that much of the male's movement resembles a random walk centered on an attractive central location (a PKS). In contrast, the female's response to this covariate (Figure 4c) is less consistent, indicating that the female is less tied to the PKS than her cub.

The results for the dynamic covariate pointing towards the female AF79 in the analysis of the cub AM80 (Figure 5d) show that the cub is drawn strongly towards the female during a number of brief periods during the observation window. The female's response to her cub (Figure 4d) indicates more heterogeneity as the female alternately returns to her cub and moves away from him.

The results for the autoregressive parameter (Figure 4e and Figure 5e) indicate that correlated movement is the norm for both animals. The magnitude of the autoregressive parameter is greater for the female (Figure 4e) than for the male (Figure 5e), indicating that the female may have a greater tendency for directed movements than the male. The CTDS movement paths are based on the CTCRW model of animal movement, which results in correlated movement paths. The inclusion of this autoregressive parameter is meant to account for the correlation in the CTDS path representation of the underlying correlated CTCRW movement path.

7. Discussion

While we have couched our CTDS approach in terms of modeling animal movement, we can also view this approach in terms of resource selection (e.g., Manly, 2002). Johnson et al. (2008) describe a general framework for the analysis of resouce selection from telemetry data using a weighted distribution approach where an observed distribution of resource use is seen as a re-weighted version of a distribution of available resources, and the resource selection function (RSF) defines the preferential use of resources by the animal. Warton and Shepherd (2010) and Aarts et al. (2012) describe a point-process approach to resource selection that can be fit using a Poisson GLM, similar to the CTDS model we describe here. In the context of Warton and Shepherd (2010), the CTDS approach can be viewed as a resource-selection analysis with the available resources at any time defined as the neighboring grid cells. The transition rate (15) of the CTDS process to each neighboring cell contains information about the availability of each cell, as well as the RSF that defines preferential use of the resources in each cell.

It is notable that the entire analysis in Section 6 takes less than five minutes from start to finish using a modest computer with 4 GB of memory and a 1.67 GHz quad-core processor. This increase in computational efficiency relative to the approaches of Johnson et al. (2008), Hooten et al. (2010), Hanks et al. (2011), and McClintock et al. (2012) allows for inference on complex behavior at finer temporal resolution than has been possible previously. To make our CTDS approach accessible to practitioners, we have created an R-package ('ctds') that contains R-code to fit the CTDS model using multiple imputation as described in Sections 2-4. A script file contained in the 'ctds' package allows for the re-analysis of the telemetry data of the two mountain lions analyzed in Section 6. This R-package can be downloaded from the first author's website (http://www.stat.colostate.edu/˜hanks).

The CTDS approach to modeling animal movement is flexible, and can be extended using standard approaches for GLMs. For example, if population-level inference is desired, the movement paths from multiple animals could be analyzed jointly, with population-level parameters in the GLM being shared by all animals. Similarly, interaction terms could be included in the model by including multiplied covariates in the design matrix. Fitting movement models in a GLM framework allows for many natural extensions with little additional effort.

The use of dynamic (directional) drivers of movement has a long history. Brillinger et al. (2001) model animal movement as a continuous-time, continuous-space random walk where the drift term is the gradient of a "potential function" that defines an animal's external drivers of movement. Tracey et al. (2005) use circular distributions to model how an animal moves in response to a vector pointing towards an object that may attract or repel the animal. Hanks et al. (2011) and McClintock et al. (2012) make extensive use of gradients to model directed movements, and movements about a central location. In our study of mountain lion movement data, we used directional drivers of movement to model conspecific interaction between a mother (AF79) and her cub (AM80). Interactions between predators and prey could also be modeled using dynamic covariates defined by vectors pointing between animals. Some movements based on memory could also be modeled using dynamic covariates. For example, a dynamic covariate defined by a vector pointing to the animal's location one year prior could be used to model seasonal migratory behavior. The ability to model both static and dynamic drivers of movement make the CTDS framework a flexible and extensible framework for modeling complex behavior in animal movement.

References

Aarts, G., Fieberg, J., and Matthiopoulos, J. (2012). Comparative interpretation of count, presence–absence and point methods for species distribution models. *Methods in Ecology and Evolution* **3,** 177–187.

Boyce, M. (2006). Scale for resource selection functions. *Diversity and Distributions* **12,** 269–276.

Brillinger, D., Preisler, H., Ager, A., and Kie, J. (2001). *The use of potential functions in modeling animal movement*, chapter 24. Nova Publishers.

Cagnacci, F., Boitani, L., Powell, R. a., and Boyce, M. S. (2010). Animal ecology meets GPS-based radiotelemetry: a perfect storm of opportunities and challenges. *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences* **365,** 2157–62.

Chen, Q. and Wang, S. (2011). Variable selection for multiply-imputed data with application to dioxin exposure study. Technical Report 217, University of Wisconsin-Madison Department of Biostatistics and Medical Informatics.

Fieberg, J., Matthiopoulos, J., Hebblewhite, M., Boyce, M. S., and Frair, J. L. (2010). Correlation and studies of habitat selection: problem, red herring or opportunity? *Philosophical transactions of the Royal Society of London. Series B, Biological sciences* **365,** 2233–44.

Forester, J. D., Im, H. K., and Rathouz, P. J. (2009). Accounting for animal movement in estimation of resource selection functions: sampling and data analysis. *Ecology* **90,** 3554–65.

Friedman, J., Hastie, T., and Tibshirani, R. (2010). Regularization paths for generalized linear models via coordinate descent. *Journal of statistical software* **33,** 1.

Getz, W. and Saltz, D. (2008). A framework for generating and analyzing movement paths on ecological landscapes. *Proceedings of the National Academy of Sciences* **105,** 19066.

Green, P. J. (1995). Reversible jump markov chain monte carlo computation and bayesian model determination. *Biometrika* **82,** 711.

Grovenburg, T., Jenks, J., Klaver, R., Swanson, C., Jacques, C., and Todey, D. (2009). Seasonal movements and home ranges of white-tailed deer in north-central south dakota. *Canadian Journal of Zoology* **87,** 876–885.

Gurarie, E., Andrews, R. D., and Laidre, K. L. (2009). A novel method for identifying behavioural changes in animal movement data. *Ecology Letters* **12,** 395–408.

Hanks, E., Hooten, M., Johnson, D., and Sterling, J. (2011). Velocity-based movement modeling for individual and population level inference. *PLoS ONE* **6,** e22795.

Hastie, T. and Tibshirani, R. (1993). Varying-Coefficient Models. *Journal of the Royal Statistical Society: Series B (Statistical Methodology)* **55,** 757–796.

Hooten, M. B., Johnson, D. S., Hanks, E. M., and Lowry, J. H. (2010). Agent-based inference for animal movement and selection. *Journal of Agricultural, Biological, and Environmental Statistics* **15,** 523–538.

Johnson, D., London, J., and Kuhn, C. (2011). Bayesian Inference for Animal Space Use and Other Movement Metrics. *Journal of Agricultural, Biological, and Environmental Statistics* **16,** 357–370.

Johnson, D., London, J., Lea, M., and Durban, J. (2008). Continuous-time correlated random walk model for animal telemetry data. *Ecology* **89,** 1208–1215.

Johnson, D., Thomas, D., Ver Hoef, J., and Christ, A. (2008). A general framework for the analysis of animal resource selection from telemetry data. *Biometrics* **64,** 968–976.

Johnson, D. S. (2011). *crawl: Fit continuous-time correlated random walk models for animal movement data.* R package version 1.3-2.

Johnson, D. S., Thomas, D. L., Ver Hoef, J. M., and Christ, A. (2008). A general framework for the analysis of animal resource selection from telemetry data. *Biometrics* **64,** 968–76.

Jonsen, I., Flemming, J., and Myers, R. (2005). Robust state-space modeling of animal movement data. *Ecology* **86,** 2874–2880.

Knopff, K., Knopff, A., Warren, M., and Boyce, M. (2009). Evaluating global positioning system telemetry techniques for estimating cougar predation parameters. *The Journal of wildlife management* **73,** 586–597.

Lima, S. (2002). Putting predators back into behavioral predator–prey interactions. *Trends* *in Ecology & Evolution* **17,** 70–75.

Manly, B. (2002). *Resource selection by animals: statistical design and analysis for field studies.* Springer.

McClintock, B. T., King, R., Thomas, L., Matthiopoulos, J., McConnell, B., and Morales, J. (2012). A general discrete-time modeling framework for animal movement using multistate random walks. *Ecological Monographs* page In Press.

Merrill, E., Sand, H., Zimmermann, B., McPhee, H., Webb, N., Hebblewhite, M., Wabakken, P., and Frair, J. L. (2010). Building a mechanistic understanding of predation with GPS-based movement data. *Philosophical Transactions of the Royal Society B: Biological Sciences* **365,** 2279–2288.

Morales, J. M., Haydon, D. T., Frair, J., Holsinger, K. E., and Fryxell, J. M. (2004). Extracting more out of relocation data: building movement models as mixtures of random walks. *Ecology* **85,** 2436–2445.

Nathan, R., Getz, W., Revilla, E., Holyoak, M., Kadmon, R., Saltz, D., and Smouse, P. (2008). A movement ecology paradigm for unifying organismal movement research. *Proceedings of the National Academy of Sciences* **105,** 19052.

Park, T. and Casella, G. (2008). The Bayesian lasso. *Journal of the American Statistical Association* **103,** 681–686.

R Development Core Team (2012). *R: A Language and Environment for Statistical Computing.* R Foundation for Statistical Computing, Vienna, Austria.

Raman, S., Fuchs, T., Wild, P., Dahl, E., and Roth, V. (2009). The bayesian group-lasso for analyzing contingency tables. In *Proceedings of the 26th Annual International Conference on Machine Learning*, pages 881–888. ACM.

Royle, J. and Young, K. (2008). A hierarchical model for spatial capture-recapture data. *Ecology* **89,** 2281–2289.

Rubin, D. B. (1987). *Multiple Imputation for Nonresponse in Surveys.* John Wiley and Sons, New York, New York, USA.

Rubin, D. B. (1996). Multiple imputation after 18+ years. *Journal of the American Statistical Association* **91,** 473.

Stephens, M. (2000). Bayesian analysis of mixture models with an unknown number of components - an alternative to reversible jump methods. *The Annals of Statistics* **28,** 40–74.

Tibshirani, R. (1996). Regression shrinkage and selection via the lasso. *Journal of the Royal Statistical Society. Series B (Methodological)* pages 267–288.

Tracey, J. a., Zhu, J., and Crooks, K. (2005). A set of nonlinear regression models for animal movement in response to a single landscape feature. *Journal of Agricultural, Biological, and Environmental Statistics* **10,** 1–18.

Van Moorter, B., Visscher, D., Benhamou, S., Börger, L., Boyce, M. S., and Gaillard, J.-M. (2009). Memory keeps you at home: a mechanistic model for home range emergence. *Oikos* **118,** 641–652.

Warton, D. and Shepherd, L. (2010). Poisson point process models solve the pseudo-absence problem for presence-only data in ecology. *The Annals of Applied Statistics* **4,** 1383–1402.

Yuan, M. and Lin, Y. (2006). Model selection and estimation in regression with grouped variables. *Journal of the Royal Statistical Society: Series B (Statistical Methodology)* **68,** 49–67.

APPENDIX VI

Front-Range Cougar Research
Winters, 2011–2012 & 2012–2013

The Use of Lures, Hair Snares, and Snow Tracking as Non-Invasive Sampling Techniques to Detect and Identify Cougars

CSU - Colorado Cooperative Fish and Wildlife Research Unit & Colorado Parks and Wildlife

Kirstie Yeager
Bill Kendall
Mat Alldredge

Research Proposal

July 31, 2012

The Use of Lures, Hair Snares, and Snow Tracking as Non-Invasive Sampling Techniques to Detect and Identify Cougars

Kirstie Yeager, M.S. student, CSU

Introduction

To set harvest quotas, evaluate management practices, and understand the dynamics of predator-prey systems, it is desirable to have reliable estimates of population size. However, answering occupancy and abundance questions regarding carnivores has long been a challenging task (Ke̒ry et al. 2011). In general, carnivores are elusive and occupy large home ranges that often vary in size across the population (Anderson et al. 2004). As a result, it can be very difficult and expensive to obtain a representative sample that is large enough to produce a reliable estimate (Ruell et al. 2009). Despite the cost, it is essential that managers have accurate population estimates that can support management decisions (Dreher et al. 2007, Immell and Anthony 2008). Here, we focus on cougars (*Puma Concolor*). In the state of Colorado, cougars are a game species; and it is imperative that their population be responsibly managed. Wildlife personnel are also tasked with managing increasing cougar-human conflict in residential and recreational areas. Developers are pushing west into previously undisturbed habitat; pet loss complaints and depredation claims continue; and each year, municipalities acquire more land to be made available to the public.

Abundance estimation is a primary objective of the sponsoring agency, Colorado Parks and Wildlife (CPW). A common way to estimate populations within an ecological context is to sample using capture-recapture, or mark-resight, methods (Pollock et al. 1990). Cougars have been sampled using a variety of strategies, some in need of further testing. Abundance estimates are then generated using models like the 2-occasion Lincoln-Peterson estimator (Williams et al. 2002) or the Huggins model (Huggins 1989). Generally, mark-resight models are bound by assumptions like closure and equal probability of capture/detection (Otis et al. 1978). However, in wildlife studies, detection is often less than certain (<1) and variable across the population (Link 2003). Before attempts can be made at estimating the population, the probability of detection must first be assessed and factors influencing detection should be explored. A thorough understanding of detection attributes within the population of interest will provide CPW with foundational information they need to estimate the population.

Mark-resight analysis requires that animals first be detected and identified through sampling procedures requiring their capture or through noninvasive means where direct human contact is not needed (Pauli et al. 2010). Due to carnivore ecology and behavior, trapping and handling practices are generally costly and difficult making noninvasive genetic sampling methods (NGS) an attractive alternative (Long et al. 2008). In addition, NGS has other benefits in that it minimizes stress and disturbance to the study animals; and when successful, it allows a larger sample size at a lower cost (Pauli et al. 2010). Herein, we will consider noninvasive methods.

Researchers have tested several noninvasive techniques, some quite creative, on a variety of carnivores to detect and count individuals. Track surveys have been used with success in occupancy studies but fall short in their ability to produce accurate abundance estimates (Diefenbach et al. 1994, Sargeant et al. 1998, Wilson and Delahay, 2001, Hayward et al. 2002, Choate et al. 2006, Gompper et al. 2006). However, when track surveys are combined with the collection of genetic material, species identification can be confirmed (McKelvey et al. 2006) and/or individuals identified, allowing for abundance estimates using mark-recapture analysis (Ulizio et al. 2006). Cameras, lures, and/or hair snares have also been used to survey cougars (Long et al. 2003, Choate et al. 2006, Sawaya et al. 2011), lynx (McDaniel et al. 2000, Schmidt and Kowalczyk 2006), bobcats (Harrison 2006), ocelots (Weaver et al. 2005), multiple felids (Harrison 1997, Downey et al. 2007), and carnivore communities (Sargeant et al. 1998, Long et al. 2007, Ruell and Crooks 2007, Castro-Arellano et al. 2008, Crooks et al. 2008). Though dozens of lures have been tested along with several novel hair-snaring devices, results have been erratic, suggesting no single method superior above all others.

With regard to cougars, the potential of NGS has not been realized. Inconsistent results have left the techniques needing further testing and refinement. In past studies involving attractants, almost all have primarily used scents. Few surveys have incorporated auditory calls despite the fact that felids may exhibit a greater response to auditory and visual lures than to olfactory stimulus (Chamberlain et al. 1999). Further testing of this component is needed to assess whether calls will increase detection at the sites. Furthermore, McDaniel et al. (2000) described a hair-snaring device that consists of a board with a scent-lure-covered carpet pad and nails protruding through it nailed to a tree. Harrison (2006), McKelvey et al. (2006), Schmidt and Kowalczyk (2006), Long et al. (2007), and Sawaya et al. (2011) tested similar mechanisms on a variety of felids. These designs snagged hair part of the time though the quality of the hair and whether or not the hair was from the target species was inconsistent. Modifications in snare designs are needed to improve the reliability of the hair snagged, thus increasing the likelihood of obtaining a usable sample.

Barbed wire is an alternative hair-snaring mechanism to traditional scratch-pad designs. Barbed wire has long been used to collect hair samples from grizzly and black bears (Woods et al. 1999, Mowat and Strobeck 2000, Poole et al. 2001, Boersen et al. 2003, Belant et al. 2005, Boulanger et al. 2006, Dreher et al. 2007, Kendall et al. 2008, Settlage et al. 2008, Proctor et al. 2010). Ebert and Schulz (2009) used barbed wire to snag hair from wild boar; and Belant et al. (2007) obtained hair from white-tailed deer. We could not find a study that used barbed wire in an attempt to snag hair from a felid species. However, we collected hair suspected to be cougar from a barbed-wire fence during a pilot snow-tracking survey.

Snow tracking is another NGS method that has been implemented in a variety of cougar studies. Seidensticker et al. (1973) applied winter tracking to evaluate movement patterns relative to kill sites, reproductive status, and topography and vegetation. Hemker et al. (1984) used snow tracking to locate cougar sign needed in population estimation. Snow tracking can also be used to facilitate DNA sample collection where hair or scat found along a track can be genotyped to yield an individual identification (McKelvey et al. 2006, Ulizio et al. 2006). Sawaya et al. (2011) reported winter tracking cougars under favorable conditions returned hair samples 80% of the time after tracking on average 1.09 km. However, because success is largely dependent upon optimal snow conditions and timing after snow fall (Squires et al. 2004), this method may only be effective in specific geographic regions. Its utility has not been tested on the Front Range.

Winter tracking may prove useful as a secondary method of detection in capture-recapture surveys. Utilizing multiple detection methods can reduce problems with bias due to the capture variation that arises when a single survey method is used (Noyce et al. 2001). For example, individuals that develop trap shyness to established sites can be detected alternatively via track surveys. Wildlife managers have used additional resources, such as animals collected during hunter harvest, as another ‘occasion’ in a detection history of a capture-recapture analysis (Garshelis and Visser 1997, Diefenbach et al. 2004, Nicolai et al. 2005, Dreher et al. 2007). Applying a secondary collection method, alternative to capture, can also reduce costs (Pauli et al. 2010) as the capture and handling of carnivores is often of great expense to federal and state agencies (Long et al. 2003, Immell and Anthony 2008).

Eliminating or accounting for genotyping errors is essential in satisfying the assumption of known identity in mark-recapture models. Failure to do so can result in an over or under estimation of abundance depending upon the type of error (Lukacs and Burnham 2005). Hair and scat collected using NGS methods typically have a low quality and quantity of DNA (Broquet et al. 2007). Inherently small quantities of DNA are susceptible to sample contamination and degradation in the field and in the laboratory (Taberlet and Luikart 1999). The resulting poor DNA samples may fail to amplify or display genotyping errors by allelic dropout or false alleles exhibiting false homozygotes and heterozygotes respectively (Buchan et al. 2005). Ernest et al. (2000) report an 8% allelic dropout rate during fecal amplification compared to a < 1% error rate in blood and muscle assays. Strict data collection and laboratory protocols can minimize genotyping errors (Taberlet et al. 1999). When possible, errors can be observed by comparing NGS results to more reliable profiles generated through blood and tissue analyses (Ernest et al. 2000, Mills et al. 2000, Mondol et al. 2009). If errors are found, it may be necessary to run multiple tests on a single sample (Taberlet et al. 1999).

DNA can be used to both confirm species and identify individuals (Woods et al. 1999, Kėry et al. 2010). For our purposes, cameras should confirm species identification. Individuals are typically identified using nuclear DNA as it has a high level of variability needed to differentiate individuals (Menotti-Raymond and O'Brien 1995). Menotti-Raymond et al. (1999) developed a genetic linkage map for the domestic cat containing 253 microsatellite loci. These loci can be used in the analysis of other felids. How many and which microsatellites are used depend upon the degree of genetic diversity between individuals in the population sampled (Woods et al. 1999). Menotti-Raymond et al. (1995), Culver et al. (2000), Ernest et al. (2000), Anderson et al. (2004), and Mondol et al. (2009) reported between 7 and 12 loci with a high degree of variability was adequate to express enough heterozygosity to differentiate individuals in their respective studies.

In summary, many attempts have been made to realize NGS methods capable of producing reliable responses. Up until now, results have been mixed. It is our goal to develop noninvasive field methods and apply laboratory techniques that will reliably detect and identify cougars (Chapter 1). If we succeed, we will apply this technique to a population of known individuals and estimate a detection probability (Chapter 2). We will also address closed mark-recapture model assumptions and investigate potential sources of capture variation (Chapter 3). Finally, we will evaluate whether snow tracking as a means to locate genetic material is a useful tool given the snow conditions on the Front Range (Chapter 4).

Chapter 1. Evaluation of attractants and hair-snaring mechanisms

Our objective is to further evaluate NGS methods to find a technique that is adequate for detecting a cougar and obtaining a genetic sample. We conducted pilot research last winter (February – April 2012) testing scent lures and predator calls in conjunction with two hair-snaring devices, a scratch pad similar to that of McDaniel et al. (2000) and a novel hair-snaring mechanism that used bait. Our preliminary results indicate that calls are successful attractants; but we were not able to obtain a genetic sample from either hair-snaring mechanism. This winter (November 2012 – April 2013), we will repeat our lure assessment in an attempt to solidify our findings regarding the utility of calls. We will also test an alternative hair-snaring technique that uses barbed wire.

Methods - Pilot

We first considered home range information for collared cougars monitored by Colorado Parks and Wildlife (CPW) and selected an area 1200 km² west of Boulder from the northern Boulder County boundary to I-70 and west of Highways 36 and 73. Several individuals with size-varying and over-lapping home ranges resided in this area. Because the average female home range is 100 km² (Alldredge unpublished data), we subdivided the 1200 km² area into twelve 100 km² quadrats and selected 4 to survey due to these areas having the highest densities of collared individuals. We then partitioned each primary quadrat into twenty-five 4 km² parcels. After considering private property limitations, we randomly chose 20 parcels (4 or 6 in each primary quadrat). Finally, we selected exact site locations based on areas of low human activity, specific landscape features (ridgelines, drainages, game trails, canopy cover), and restrictions imposed by city and county officials. It should be noted that most of the area contained within 1 randomly selected quadrat was closed due to raptor nesting activity. Therefore, we placed our site to the south in the neighboring quadrat. At the conclusion of the season, it was discovered that miscalculations were made in the placement of 2 other sites. One site was placed in error 427 m to the east and the other 139 m to the north of the intended quadrats.

We established 4 different types of sites. All sites consisted of a baited hair snare, a scratch pad with catnip, a visual lure (aluminum pie pan), and a camera. The sites varied by whether or not they included a scent lure, a call, neither, or both (Table 1). Components were assigned randomly except when permit restrictions prohibited us from putting calls at certain sites resulting in a slightly uneven distribution (6, 4, 5, and 5). Based on recommendations by lynx researchers (G. Merrill personal communication), we chose Pikauba® as our scent lure in our preliminary study design. However, Pikauba® was not successful in eliciting a rubbing response during the first observed cougar visits. Therefore, we added the scent, Canine Call®, to sites chosen to have scent lures. We secured calls (Wasatch Wildlife Product® FurFindR®) to a tree at approximately head height. The manufacturer programmed calls to transmit a 5 second distressed fawn or fox recording on 30 second intervals. These calls were also equipped with light sensors rendering them dormant during daylight hours. At each site, we placed an infrared motion-sensor camera (Reconyx® PC85 Rapidfire® or PC800 Hyperfire®) programmed to take 5 photos in rapid succession when triggered. To further evaluate the utility of the attractants, we positioned a second camera (Cuddeback® Capture®) within 18 – 102 m of 7 sites (2, 1, 2, and 2). The camera placed at a distance of 102 m was located across a drainage where the visual flasher was still visible. We chose locations in areas where animals were likely to pass (saddles, game trails, old roads, ridgelines) and at a distance where, in our opinion, the lure(s) should be effective attractant(s). All sites were checked for activity on average every 5 – 6 days. As necessary, baits and scents were replenished and new batteries were placed in cameras and calls.

	1	**2**	**3**	**4**
Visual flasher	X	X	X	X
Scratch pad w/catnip	X	X	X	X
Baited hair snare	X	X	X	X
Scratch pad w/scent	X			X
Animal distress call		X		X
Camera	X	X	X	X

Table 1. All site components for the 4 site types (scent, call, neither, and both) established in random locations.

In conjunction with lures, we tested 2 hair-snare designs. We constructed a hair-snaring mechanism described as a metal mesh (1”) cube (25 cm edge) open on one end. We wired a 0.1 – 0.2 kg piece of deer meat in the back of the snare. To snag hair, we attached a 20 cm spring and a 13 cm barrel cleaning brush (Figure 1). To minimize access to the bait by non-target species, we secured the snares to trees at an initial height of 70 – 95 cm using cable and U clamps. After observing the first few cougar responses, we lowered the snares to 40 – 70 cm. After further observations, we lowered the snares all the way to the ground. The other snare was a scratch pad design. This mechanism consisted of a board (14 X 14 cm), cotton batting or a carpet swatch, and a piece of metal altered to snag hair (Figure 2). We placed one scratch pad with catnip oil and dried catnip at each site. We applied the scent lure, Pikauba® then Canine Call®, to another and placed it at selected sites. Scratch pads were nailed to a tree at an average height of 55 cm.

To supplement the data collected at the 20 random sites, we targeted individual cougars by placing attractants and a camera in the vicinity of a collared animal. We established 4 sites with scent lures, a call, and a flasher. These sites were active for 3 – 6 days. Late in the season, we established 4 additional sites to test a modified site design. At these locations, we used calls and bait as the attractants. These sites were active for a longer period (5 – 16 days).

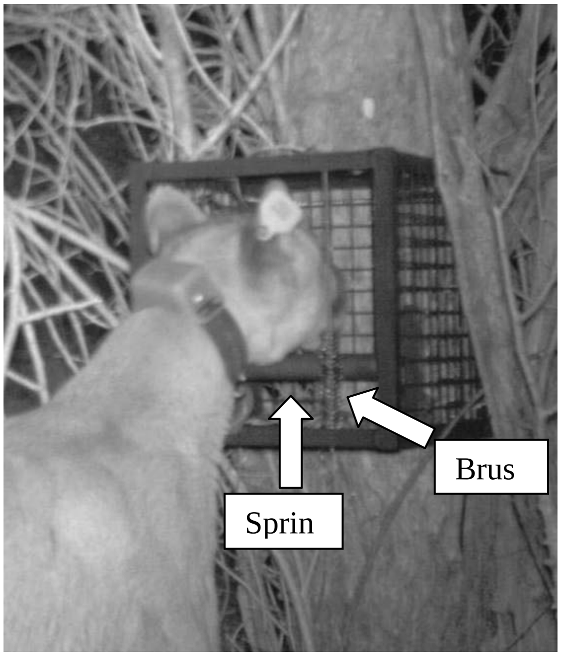


Figure 1. Bait was wired in the back corner. In attempting to obtain the bait, the cougar would flex the spring catching hair and contact the barrel brush.

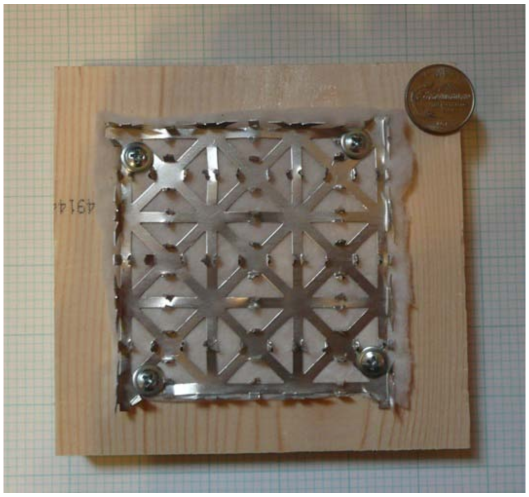


Figure 2. Catnip, Pikauba®, and Canine Call® were applied to scratch pads.

Results – Pilot

Our preliminary findings clearly indicate that calls are effective attractants. At the random sites, we observed 23 detections by at least 13 individuals. All but 2 occurred at sites with calls (Table 2). In addition, we observed 12 detections at the 8 selectively placed sites. It should be noted that 11 of the 12 visits occurred several hours or days after the site was established indicating that the cougars were not immediately drawn in. Furthermore, some sites were visited by a different cougar other than the one it was intended to attract. Photographic evidence also supports cougar interest in calls. Photos documented cougars looking towards calls and attempting to climb trees in pursuit of the devices. On 3 occasions, we observed the same cougar carrying off a call thought to be securely cabled to a tree. Contrary to calls, we have no data to suggest the scent lures we employed are successful in attracting cougars. We did not document a single rubbing response to any of the 20 scratch pads containing catnip nor the 11 scratch pads with Pikauba® or Canine Call®. In addition, we compared photos recorded on the 7 ancillary cameras with cougar activity observed at the sites. These cameras detected 3 cougars as they moved through the area. Two were not observed at the corresponding lure site (~18 m away) providing evidentiary support for the futility of the attractants (Canine Call®, catnip, and bait). Due to a small sample size of 7, results are far from conclusive but suggest, at the very least, placing a supplementary camera in a high traffic area near a site may be a useful tool in assessing the value of a lure. Finally, our hair-snaring mechanism containing the bait was ineffective. Cougars showed little interest in the meat wired in the back of the snare. In conclusion, we determined that calls were effective in attracting a cougar to a site but our attempts to entice a response necessary to obtain a sample failed.

	#	Days Active	Ave. Interval Checked	Detections	2^{nd} Cameras	2^{nd} Camera Detections
Bait only	**5**	**36**	**5.03**	**1**	**2**	**0**
Bait & scent	**6**	**41**	**5.08**	**1**	**2**	**2**
Bait & call	**4**	**37**	**5.87**	**9**	**1**	**0**
Bait, scent, & call	**5**	**49**	**5.10**	**12**	**2**	**1**
Totals and Averages	**20**	**41**	**5.23**	**23**	**7**	**3**

Table 2. This table summarizes the 20 randomly placed locations and includes the number of each site established, the average period each site was active, the average interval that each was checked, the number of detections per site type, the number of secondary cameras placed near each type, and the number of secondary camera detections.

Methods – Winter 2012/2013

To further evaluate attractants, our site selection criterion remains unchanged. We will use the same four 100 km² quadrats subdivided into twenty-five 4 km² parcels. But, we chose to undergo a new random selection process and selected a fresh set of 4 parcels in each quadrat. We repeated this process 3 times as we will move the lure sites to a new location within the quadrat every 4 – 5 weeks. The criteria for choosing exact site locations will also stay the same (based on human use, landscape features, and imposed restrictions).

Consistent with our pilot season, we will have 4 different types of sites (Table 3). However, we will focus on specific attractants by eliminating components we consider to be futile (visual lures and catnip). We will further modify the site configuration to accommodate the application of a new hair-snaring technique. All sites will have a natural, animal-scent component (bait) consisting of deer meat, blood, or hide. The sites will vary by whether or not they contain a scent lure (Pikauba®), a call (fawn distress), neither, or both. At sites designated to have scents, we will place Pikauba® near the base of a tree and 150 - 180 cm up in the same tree to aid in scent dispersal. Where applicable, we will cable the calls 30 – 60 cm up from the base of a tree. We will place the bait near the base next to the scent and/or call. We will then brush in the area surrounding the tree making most of the perimeter impenetrable. An obvious entry way will remain (Figure 3). Based on the terrain and configuration of the vegetation, we will stretch lines of barbed wire across the open entry camouflaging it with sticks, leaves, and other natural materials. The height of the wire will depend upon whether we desire a cougar to step over, under, or through 2 strands. Wire placement may be modified as the season progresses. In view of the entrance, we will secure an infrared motion-sensor camera (Reconyx® PC85 Rapidfire® or PC800 Hyperfire®) programmed to rapidly take 5 photos when triggered. In addition, we will further test the utility of a secondary camera (Cuddeback® Capture®) by deploying 8 additional cameras (2 at each of the 4 site types) in high traffic areas and within 20 – 50 m of lure sites.

	1	2	3	4
Bait	X	X	X	X
Scent, Pikauba®		X		X
Fawn Distress Call			X	X

Table 3. All sites will have bait. In addition, one site will have Pikauba®, one site will have a call, and one site will have both.

Figure 3. Last winter, we were able to test the modified site design, minus barbed wire. We placed a predator call and deer meat near the base of a tree and outlined the perimeter with thick brush leaving a single entry point. We positioned a camera towards the entry way. A cougar was observed entering and exiting as intended. The horizontal stick above its back represents a possible placement for a strand of barbed wire.

To limit the possibility of sample contamination and degradation, sites will be checked every 5 – 7 days. We will consider all hair on a single barb as one sample (Poole et al. 2001, Dreher et al. 2007). Hair will be removed using sterile tweezers and placed in a small paper envelope. Paper envelopes will then be put in a plastic bag with a desiccant and stored at room temperature (Taberlet and Luikart 1999, Roon et al. 2003). Barbs will be re-sterilized by passing a flame under them to burn any remaining material (Kendall et al. 2008, Settlage et al. 2008). In addition, we will use photographic evidence to confirm individual identification and the presence of multiple animals. When sites are checked, we will also replenish the scent and bait and change the batteries in cameras and calls.

At the conclusion of the survey, we will consider data from 4 sampling periods, each 4 – 5 weeks in length. The first occurred during the pilot season in 2012 (March 1 – 31). The other 3 will transpire this winter through the following time intervals: November 17 – December 20, December 21 – January 23, and January 24 - February 26. We will evaluate a total of 68 sites (approximately 17 of each type).

The utility of the attractants will be judged by the quantity of detections and by comparing detection probabilities. If a cougar is documented by photo at the site, the attractant(s) will be counted once (per day per cougar). Detections will be totaled relative to each of the 4 site types. In addition, our analysis will state the total count at sites with Pikauba® and the total count at sites where calls were used.

Detection probabilities (p) will also be used to confirm the value of an attractant. p can be estimated as a proportion of the number of detections (n) to the number of available animals (A) using the a simple binomial model;

$$\hat{p} = \frac{n}{A}$$

(Williams et al. 2002). We will apply the detection counts described above and determine an individual's availability status by using secondary camera information and GPS location data. First, animals detected by the secondary camera will be counted as available. The remaining study animals found to be within a specified proximity to the site (500 – 1000 m) will also be counted. Using detection and availability counts, we will estimate the probability that a cougar is detected given that it is available and compare the resulting values.

We will evaluate the results of the hair-snaring component by using site-specific photographic evidence as well as assessing the quantity and quality of the hair we collect. Photos will indicate if the animal moved through the site in the manner we intended. To assess quantity, we will conduct a visual evaluation at the time of sample collection. Samples will be scored 1 – 3 (1 indicating < 5 hairs collected on a single barb, 2 indicating 6 – 15 hairs collected, and 3 indicating > 15 hairs collected). Laboratory analysis will judge sample quality through amplification and genotyping error rates.

We will process the hair samples at USGS Fort Collins Science Center, FORT Molecular Ecology Lab. When possible, we will extract DNA from 10 hairs (Goossens et al. 1998, Boersen et al. 2003) using Qiagen DNeasy® Tissue Kits (Qiagen Inc., Valencia, CA). We will then genotype each sample using 9 – 12 microsatellite primers shown to have high variability in cougars (Ernest et al. 2000, Sinclair et al. 2001, Anderson et al. 2004). The DNA will be amplified by polymerase chain reaction (PCR) using a M13-tailed forward primer as described by Boutin-Ganache et al. (2001). For known individuals, we will compare the resulting genotypes with blood and tissue samples collected by CPW during capture. DNA from blood and tissue is first extracted by personnel at the CPW Foothills Wildlife Research Facility. It is then genotyped using the same microsatellite loci at the USGS lab. We will address genotyping errors and if possible, rerun the samples.

Chapter 2. Estimating the detection probability (p) using a noninvasive-sampling technique applied to a known population of cougars

Our objective is to estimate a detection probability (p) needed for future population assessments. We will apply a noninvasive-survey technique to a population of cougars on the Front Range and estimate p using a simple binomial model,

$$\hat{p} = \frac{n}{A}$$

A is the total number of available animals; n is the number counted per occasion; and $\hat{p}$ is the estimated probability of detecting an individual given that it is available (Williams et al. 2002). To estimate p, we will use cougars previously captured and collared by Colorado Parks and Wildlife (CPW). The collars applied to these animals are equipped with Global Positioning System (GPS) features. We will monitor cougar location data relative to study sites and consider their availability (A) to be detected. Those detected will be counted (n); and p will be estimated.

The simple binomial model is bound by 2 assumptions (Williams et al. 2002). The first assumption is that the fate of all cougars is known. This assumption will be met since we are using a population of marked individuals whose location and mortality status will be known for the duration of the study. The second is that each detection is an independent event. We will place sites far enough apart so that one site should not influence detection at another.

We are investigating detection for the purpose of gaining foundational knowledge that will be necessary in future population assessments. Concurrently, we hope to develop a mark-resight technique capable of producing reliable abundance estimates. Therefore, we also address closed mark-recapture model assumptions. Due to sample contamination and degradation, noninvasive-sampling techniques, similar to those that we will employ, may lead to the assumption violation of misidentification through genotyping errors (Taberlet and Luikart 1999, Lukacs and Burnham, 2005). Because our survey will use a population of known individuals, we can assess this source of error in the lab by comparing the samples we collect noninvasively with archived blood and tissue collected from the animal during the initial capture (Dreher et al. 2007). Another assumption, equal capture/detection, is especially pertinent to our work. It assumes that p is equal throughout all members of the population (White et al. 1982). This is often not the case in wild populations (Pollock et al. 1990). We will further investigate potential sources of capture variation in chapter 3.

As suggested by Otis et al. (1978), we will attempt to minimize factors that may induce capture variation through study design. We anticipate trap response (a positive or negative behavioral response to the trapping event) to be one potential source of variation (Pollock et al. 1990). For example after the first few visits, it is possible that a cougar will lose interest in a site upon discovering that the distressed fawn call and deer scent is not a real animal. This ‘trap shy’ response can create positive bias in a population estimate (Williams et al. 2002). Mowat and Strobeck (2000) recommend reducing capture disparity by moving sites throughout the field season. It is our belief that choosing new locations will aid in keeping the sites novel as the efficacy of the lure site relies on provoking curiosity so that a cougar will alter its behavior and approach the site. The success of the hair snare is also dependent upon enticing interest. If in a fresh location, the cougar may once again be tempted to investigate the same fawn call and deer scent. In addition, selecting new locations is likely to accommodate the activity patterns of more animals. This will increase the likelihood that all individuals will come in contact with the sites thus reducing animal-specific heterogeneity (C. Anderson personal communication, Boulanger et al. 2006).

February – April 2012, we conducted pilot research testing lures and hair-snaring mechanisms. After evaluating cougar responses, we concluded that applying predator calls to a site designed to use a barbed-wire hair snare could attract an individual and obtain a genetic sample. In short, we will establish lure sites within the range of known cougars to gain insight into the detection process.

Methods – Winter 2012/2013

To estimate p, we refined the site selection process employed last year by expanding our sampling effort over a wider geographic range. Our study area remains an approximately 1200 km² area west of Boulder; but for the purpose of estimating p, we will sample the entire area (unlike our pilot season, where we chose four 100 km² areas to sample). The northern boundary is 4 km north of the town of Lyons and follows an arbitrary straight line due west. The southern boundary will follow Highway 6 then Interstate 70. Highway 36 and Highway 93 will define the approximate eastern boundary; and Highway 72 will delineate the west. Colorado Parks and Wildlife (CPW) monitors a higher density of collared animals in the eastern half of this area. Collared animals are an integral component in estimating p. Therefore, we divided the 1200 km² zone into 2 strata, east and west. In choosing individual site locations, we first created a map of the study area using ArcGIS® 10 and over-layed a 2 X 2 km grid. Partial cells around the perimeter were combined with neighboring cells. Next, we applied the Reversed Randomized Quadrant-Recursive Raster (RRQRR) algorithm to randomly select 15 cells in the eastern strata and 6 cells in the western strata for a total of 21 sites having an even spatial distribution (Theobald et al. 2007). Then, we considered access issues in each cell chosen. Cells we considered inaccessible or unusable, possibly due to private property restrictions or high human use, were discarded; and another cell was randomly chosen. To keep the sites novel, we repeated the selection process without replacement 3 more times as it is our intention to move the sites 4 times throughout the season. Finally, we will select exact site locations to correspond to areas known to be favorable cougar habitat: ridgelines, drainages, game trails, treeline edges, and saddles (K. Logan personal communication).

All 84 sites will contain the same elements. Our primary attractant will be a predator call (Wasatch Wildlife Products® Custom FurFindR®) programmed to play a 5 second distressed fawn recording on 30 second intervals. These calls are also furnished with light sensors rendering them inactive during daylight hours. We will cable the calls 30 – 60 cm up from the base of a tree and further secure them using a 30 X 30 cm piece of chicken wire. To incorporate a natural prey scent, we will place bait (deer meat, hide, or blood) near the tree base. We will then build an impenetrable perimeter around the tree with thick brush leaving an obvious entry way to the call and bait. We will configure lines of barbed wire within the entrance. Terrain and vegetation features will determine the height of the wire and consequently whether we desire a cougar to step over, under, or through 2 strands. In addition, we will attempt to conceal the wire with sticks and other natural materials (Chapter 1, Figure 3). At each site, we will position an infrared motion sensor camera (Reconyx® PC85 Rapidfire® or PC800 Hyperfire®) set to rapidly take 5 photos when triggered.

To minimize the possibility of sample contamination and degradation, we will conform to the same protocols for sample collection outlined in Chapter 1. We will check the sites for activity every 5 – 7 days collecting all samples, replenishing bait, and changing batteries. We will consider hair on a single barb as one sample and denote quantity with a score of 1 – 3 (1 equals < 5 hairs, 2 equals 6 – 15 hairs, and 3 equals > 15 hairs). We will remove hair using sterile tweezers and re-sterilize the barb by passing a flame under it (Kendall et al. 2008, Settlage et al. 2008). We will place the hair in a small paper envelope. Paper envelopes will then be put in a plastic bag with a desiccant and stored at room temperature (Taberlet and Luikart 1999).

We will evaluate 21 sites during each of the four 4 – 5 week sampling periods: November 17 – December 20, December 21 – January 23, January 24 – February 26, and February 27 – April 1 for a total of 84 sites. We will tally detections as one per night per cougar based on photographic confirmation. Dependent kittens will not be counted. Though we expect all animals visiting the sites to be detected by camera, hair samples may also provide proof of cougar presence as well as identifying unmarked animals.

Hair samples will be processed at the USGS Fort Collins Science Center, FORT Molecular Ecology Lab. Taberlet et al. (1996) suggested that to achieve a correct genotype at a 99% confidence level, 8 U template DNA is needed (1 U is equivalent to the DNA content of 1 diploid cell). Therefore when possible, we will extract DNA from 10 hairs (Goossens et al. 1998, Boersen et al. 2003) using Qiagen DNeasy® Tissue Kits (Qiagen Inc., Valencia, CA). Samples will be genotyped using 9 – 12 microsatellite primers shown to have high variability in cougars (Ernest et al. 2000, Sinclair et al. 2001, Anderson et al. 2004). We will amplify the DNA by polymerase chain reaction (PCR) using a M13-tailed forward primer as described by Boutin-Ganache et al. (2001). To assess error, the results from hair genotyping will be compared with archived blood and tissue samples collected by CPW during capture. If possible, we will re-process hair samples shown to contain error at one or more alleles.

A total count of available animals (A) is needed to estimate p. An individual's availability must first be appropriately defined for the estimate to be unbiased. We will base availability on GPS locations included within a circular buffer zone around a site. The radial distance will be determined by collar error due to fix rates and missed fixes. We are limited by the technology we will employ. Collars will be programmed to download a fix every hour from 7 pm to 7 am and every 4 hours during the day. It will not be possible to know an animal's exact location at all times resulting in potential error in determining availability. Availability missed due to collar error will result in positively biased p estimates. For example, if an individual was not detected but passed within the buffered region without recording a fix, the availability value would be low resulting in a high p estimate. Positive bias would also result if the same individual was detected but never considered available. This can be troublesome as a positively biased estimate for p applied in population estimation will result in a negatively-biased abundance estimation. We will determine the distance the collar error becomes negligible by increasing the buffer zone until all animals detected can also be considered available. For animals not detected, we will connect successive locations with a straight line. If this line passes through any part of the buffer zone, we will count the animal as available.

We will apply the count of available animals (A) and the number detected (n) to estimate the detection probability (p). First, we will use a simple binomial model to estimate p for each of the *i* sites:

$$\hat{p}_i = \frac{n_i}{A_i}$$

(Williams et al. 2002). Next, we will estimate the average p for all 84 sites, where *x* is the number of sites,

$$\hat{\hat{p}} = \frac{\sum_{i=1}^{x} \hat{p}_i}{x}$$

(Thompson 2002). We will then estimate the sampling variance per site,

$$\widehat{\text{var}}\,(\hat{p}_i) = \frac{\hat{p}_i(1 - \hat{p}_i)}{A_i}$$

(Williams et al. 2002).

Next, we will estimate the process variation, or the actual variation, in the detection probability across sites (Link and Nichols 1994). Total variance [$\widehat{\text{var}}\,(\hat{p})$] is comprised of two components: process variation ($\text{var}\,(p_i) = \sigma^2$) and sampling variation [$\text{var}\,(\hat{p}_i|p_i)$] (Burnham and White 2002). The site locations are chosen relative to the sampling method. Because the sites are inherently variable, no two sites will have the exact same vegetation composition, aspect, elevation, etc. To eliminate the variation associated with the sampling and estimation procedure and focus on the precision of the detection probability across all sites, we will consider the variance using a variance components approach. Our variance estimate will consider process variation only and be derived by subtracting the estimated average of the sampling variances across all sites from the total variance.

$$\sigma^2 = \widehat{\text{var}}\,(\hat{p}) - E[\text{var}\,(\hat{p}_i|p_i)]$$

(Gould and Nichols 1998),

where

$$\widehat{\text{var}}\,(\hat{p}) = \frac{\sum_{i=1}^{x} (\hat{p}_i - \hat{\hat{p}})^2}{(x - 1)}$$

(Link and Nichols 1994),

and

$$E[\text{var}\,(\hat{p}_i \mid p_i)] = \frac{\sum_{i=1}^{x} \left(\frac{\hat{p}_i(1 - \hat{p}_i)}{A_i} \right)}{x}$$

(Williams et al. 2002).

Finally, we will construct confidence intervals around $\hat{\bar{p}}$ at an alpha level of 0.05 using a test-statistic obtained from the student's *t* distribution with degrees of freedom 60>120 (1.99), (Ott and Longnecker 2010),

$$\hat{\bar{p}} \pm (1.99) \times \hat{SD}(p)$$

where

$$\widehat{SD}(\hat{p}) = \sigma \qquad \text{(Williams et al. 2002).}$$

Confidence intervals are typically computed using the standard error (SE) of the estimate. However, because we are subtracting the sampling variation to determine the actual variation (σ^2), we will use the standard deviation (SD = $\sqrt{\sigma^2}$) to construct a confidence interval.

In addition, we will explore the variance components between survey periods. We acknowledge that a sample size of 4 is most likely inadequate to make temporal inferences. Multiple years of data may be needed to assess the process variation in p over survey periods that are 4 – 5 weeks in length.

Chapter 3. Investigating the assumptions of the Lincoln-Peterson Estimator and other closed mark-recapture models

Mark-recapture methods are widely used in wildlife population monitoring. For the results to produce unbiased and precise estimates, model assumptions must be addressed (White et al. 1982). Closed mark-recapture models maintain 4 primary assumptions. These assumptions include: demographic and geographic closure; no tag loss; accurate individual identification; and equal probability of capture (Otis et al. 1978). The Lincoln-Peterson estimator is a simple, two-period, mark-recapture model, based on the relationship

$$\frac{m_2}{n_2} = \frac{n_1}{\hat{N}}$$

where m_1 individuals are marked at occasion 1 and released; n_2 individuals are counted on occasion 2 with m_2 marked animals; and N is abundance and is often the parameter of interest (Link 2003). The Lincoln-Peterson estimator is bound by closed-model assumptions (Williams et al. 2002). This model does not accommodate unequal detection (Link 2003). If capture variation is found to exist in the target population, then the simple application of the Lincoln-Peterson estimator may prove inadequate requiring the use of more complex models (Otis et al. 1978). Therefore if during initial planning, it is suspected that study design or idiosyncrasies unique to the species or population of interest may violate one or more model assumption, then, an a priori model set considering covariate and group attributes should be constructed to investigate possible sources of these violations (Willson et al. 2011). Model selection criteria like Akaike's Information Criterion (AIC, Stanley and Burnham 1998) can then be applied to compare model fit using program MARK (White and Burnham 1999).

The assumption of geographic and demographic closure (no immigration, emigration, births or deaths) is of fundamental importance in the analysis of mark-recapture data under closed models (Stanley and Burnham 1998). This assumption is likely to be violated in most field applications (Kendall 1999). Consequently, complex models have been developed to test for closure violations (Stanley and Burnham 1998) and to estimate the parameters responsible (Kendall 1999). However, because we are estimating detection (p) with a population of cougars wearing Global Positioning System (GPS) collars, their location in or out of the study area will be known with certainty. We will also know their survival state. Newly captured animals will be added to the study; and individuals that die will be censored.

Next, we will consider the assumptions of no tag loss and accurate individual identification within the context of our study. At the time of capture, Colorado Parks and Wildlife (CPW) marks all cougars with a unique set of tags (inscribed with a 3 – 4 character identification code) one in each ear. The direction of the tag is specific to sex (i.e. left facing forward and right facing back for females). CPW also applies a GPS collar around the neck that is clearly visible in photos. It is highly unlikely that a cougar would lose all 3 marks. In addition, GPS location information will confirm an individual was in the vicinity of a site. Tag loss is doubtful; but misidentification through genotyping error is possible, potentially biasing population estimates. False genotypes result in population over-estimation and multiple individuals assigned the same genotype produce under-estimates (Lukacs and Burnham, 2005). We will assess genotyping error rates by comparing samples obtained via our noninvasive method with blood and tissue samples collected during initial capture (Dreher et al. 2007).

The final assumption of equal capture or detection is often violated in wildlife studies (Link 2003). Otis et al. (1978) presented, in detail, 3 models (and all combinations thereof) in which the assumption of equal capture can be relaxed. Model Mt considers variation between sampling occasions. The second primary model, Mb, accommodates a behavioral change due to trap response, ‘trap happy’ or ‘trap shy’. Failure to consider capture variation due to a behavioral response can lead to biased population estimates. A ‘trap happy’ response can negatively bias estimates; whereas, a ‘trap shy’ response will create positive bias (Williams et al. 2002). Finally, model Mh assumes individual heterogeneity amongst individuals. Unmodeled heterogeneity may overstate precision and include bias (Link 2003).

Accounting for individual heterogeneity in abundance estimation has long plagued researchers (Link 2003). Burnham and Overton (1978) presented a jackknife estimator that can provide some robustness in population estimation where heterogeneity in capture probabilities is present. However, because the jackknife estimator is not a maximum likelihood estimator, the sources of variation cannot be evaluated by comparing models using likelihood ratio tests or through model selection criteria (Akaike’s Information Criterion, AIC) (Williams et al. 2002). Pledger (2000) fit finite mixture models in addressing capture heterogeneity by maximum likelihood, thus allowing for model comparisons. Finite mixture theory groups individuals into two or more mixtures and considers the probability that an individual is in one mixture (π) and the probability that an individual is in the other mixture ($1-\pi$), though it is not possible to discern to which group an individual belongs. Huggins (1989) presents another maximum likelihood approach that will accommodate individual capture covariates when deriving the abundance parameter. Sources of heterogeneity modeled as covariates can then be evaluated through likelihood ratio tests and model selection criteria. Huggins (1989) suggests that the asymptotic properties of his estimators are normal in large sample sizes. However, in the case of small sample sizes, which are common in most wildlife studies, his simulations showed a skewed distribution (Huggins 1989). In response, he describes a conditional bootstrap method that can be applied to overcome problems of nonnormality thus producing reasonable confidence interval estimations.

The potential for model assumption violations is largely dependent upon survey design and nuances of the target population (Otis et al. 1978). Last winter, we evaluated attractants and a hair-snaring mechanism applied to a population of cougars. Based on the limited success described in past studies, we anticipated few, if any, detections. However, in our short survey, we observed dozens of responses including multiple detections of the same individuals. We have not found any other study with the promise of a reliable noninvasive-survey method to both attract a cougar and obtain a genetic sample. If a high detection rate persists throughout the ensuing season, we will be presented with an ideal opportunity to explore possible sources of capture variation.

Much is already known about cougar behavior to suggest that capture variation is possible and likely (C. Anderson personal communication). Resident adult home-range sizes vary between season

(Seidensticker et al. 1973), sex (Dickson and Beier 2002, Anderson et al. 2004), and female reproductive status (Hornocker 1969). Males generally occupy larger areas than females (Anderson et al. 2004); but female home-range sizes are more variable (Hornocker 1969). Movements differ among behaviors such as hunting, feeding, and mating (Beier et al. 1995). Scrapes suggest a pattern in how cougars travel (Hornocker 1969) indicating that cougars do not occupy all parts of their home ranges equally. In addition, females with small kittens will be less mobile and confined to a smaller area (Seidensticker et al. 1973). Activity also varies relative to time of day, peaking in the evening hours (Sweanor et al. 2008). Finally, transients create another dynamic in the population that must be considered (Lindzey et al. 1994).

The population of cougars on the Front Range to which we will apply our sampling protocol is subject, but not limited, to all of the potential sources of variation described above. We expect to have a sample size of approximately 20 – 25 individuals based on past capture success. Due to a small sample size relative to the large quantity of factors that may contribute to capture variation, we propose the Huggins model, which considers relationships as covariate and applies a bootstrap procedure, may be an appropriate estimator for use in future population assessments.

To fit this model, we will formulate an a priori list of potential categorical and continuous variables (Figure 4) and describe any additional field protocols needed to collect the data. At the time of capture, we will note age (adult or subadult), sex (male or female), and if possible, reproductive status (with kittens or without). Photographic evidence at the sites may also aid in determining the reproductive condition of females. Using photos, we will log the time of the detection (day, dusk, night, or dawn) and the presence of multiple adult cougars. We will also consider general site specific aspects (ridgeline, drainage, or saddle). Finally, we will assign a behavior to each animal found to be available (feeding, traveling, mating, or denning) via GPS location cluster data.

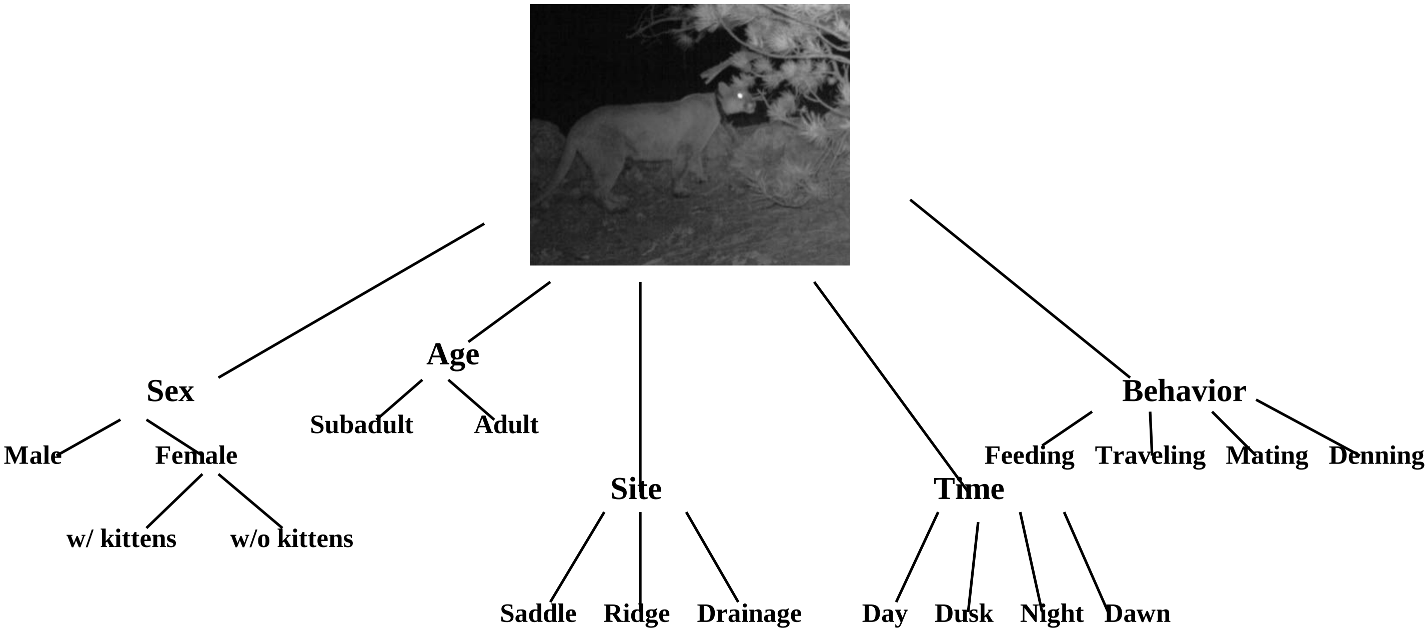


Figure 4. Variables we will consider in the detection process.

At this time, it is not our intention to estimate the population size. Our efforts are simply exploratory in nature. Therefore, we reserve the possibility of investigating additional variables post hoc. If possible, we will also apply information we gain from our exploratory investigations to testing models with simulated data.

Chapter 4: The utility of snow tracking on the Front Range

Our objective is to evaluate the utility of snow tracking as a means to obtain genetic material given the terrain and snow conditions on the Front Range and determine if it can be useful as a secondary source of detection in population estimation. Last February, we began our assessment by locating and following tracks from known cougars collecting hair and scat samples found en route. The samples that we collected will be processed and evaluated based on the successful genotyping of individuals. We found that many variables contributed to the condition of the track and to the quality and quantity of genetic material available forcing us to reconsider our initial objectives. This winter (November 2012 – April 2013), we will test a modified sampling plan that we feel will have a more practical application given the conditions on the Front Range.

Methods - Pilot

Our initial objective was to investigate the ability to find a genetic sample relative to the time after a snow fall event and the age of the track. On the first day after it snowed, we would randomly choose 5 cougars whose track would be surveyed on day 1, 2, 3, 4, or 5. On the second day, we would choose 4 more animals to be surveyed on day 2, 3, 4, or 5. We would continue this trend for 5 days (Table 4). For example, on day 3, we would survey 3 tracks: a one-day old track, a two-day old track, and a three-day old track.

Days after snow **Track age**	**1**	**2**	**3**	**4**	**5**
1	1	1	1	1	1
2		1	1	1	1
3			1	1	1
4				1	1
5					1

Table 4. Schematic of a 5-day sampling period following a snow-fall event considering tracks that are 1 – 5 days old.

Our interest also lay in determining the distance we needed to travel to obtain an adequate sample. We would use GPS collared cougars to locate tracks and to evaluate genotyping success by comparing DNA from hair and scat samples with archived DNA from blood and tissue collected during capture. We would follow the track 1 – 2 km, collect all hair and scat found (differentiating between hair found on the surface of the snow, snagged on brush, or in day beds), assess which samples were genotyped successfully, and determine the average minimum distance we needed to survey before an individual could be identified.

Results – Pilot

We were able to locate multiple hair samples and 2 scats by following tracks on average 1.25 km but only within 1 – 2 days after a snow fall. Snow conditions were not conducive to tracking over the full 5 day survey period we had described. Many factors, such as temperature, wind conditions, aspect, snow depth, canopy cover, and vegetation type, contributed to our ability to find and follow a track and to the quality and quantity of genetic material we found. In addition, our initial study plan dictated we would follow a track for 1 – 2 km. However, we often lost the track on dry ground or encountered private property boundaries and could not continue.

We also struggled with determining which cougars to survey and where to start. We used the GPS collared animals monitored by CPW to efficiently locate the tracks but needed to randomly choose a starting point and a direction of travel (forward or backward). Our study area is a mosaic of private parcels and public open space. Cougars available to be surveyed needed to have moved a considerable distance across public land shortly after it snowed. We found that, on any given day, only ~¼ of the collared cougars were traveling on public land. Consequently the morning after a snowfall, we were limited to 4 – 6 possible tracks to pursue.

We encountered other logistical challenges while tracking. Snow conditions on sunny, exposed, south-facing slopes, deteriorated rapidly thus ending our progress by mid-morning the first day. Conversely, tracks remained in good condition for several days in north-facing, wooded areas. We had difficulty tracking where more than one cougar was present, especially females with multiple kittens. We also encountered males following in the tracks of the females we were surveying. Not knowing for certain which cougar left the sample was problematic to our initial objectives. In addition, other species like deer would travel over cougar tracks making tracking difficult and obscuring genetic material.

We determined that we could find hair and scat by following tracks but only within a day after it snowed, preferably first thing in the morning. Snow conditions on the Front Range did not allow for a 5-day sampling period. We have not yet processed the samples to determine if the hair and scat we collected contain enough DNA to genotype without error.

Methods – Winter 2012/2013

We recognize that inconsistent snow conditions may deem winter tracking to be an inefficient field method when applied as the sole surveying procedure used in population estimation; but snow tracking does have some utility in that genetic samples are easily located along a fresh track. Our modified objectives will consider the function of snow tracking as a secondary means to detect cougars and be based on resources that we believe wildlife managers might have available to them.

We will work within the 1200 km² area west of Boulder detailed in Chapter 2 (page 13). We divided the region into 3 equal sections (~400 km²), north to south. To maximize efficiency, we will use roads and established trails. We chose 10 stretches of road/trail 2 km in length per section. The 10 segments will be ordered so that they can be most efficiently checked. However, the starting point will be randomly selected per snow event. Based on observed weather patterns, we expect to have 1 – 2 snow fall events per week requiring enough resources to employ 3 persons at most twice a week. The morning after a snow fall, 1 person per section will begin checking each stretch, one after the other, until a track is found.

When possible, tracks will be followed in the backward direction and mapped via a Garmin® handheld GPS unit until an adequate sample is found. We consider an adequate sample to be a tuft of >15 hairs collected from a branch or fence, hair collected from a daybed, or a scat. Hair will be collected using sterile tweezers and placed in a small, paper envelope. The envelope will then be put in a plastic bag with desiccant and stored at room temperature (Taberlet and Luikart 1999). We will re-sterilize tweezers with a bleach water rinse. Scats will be placed in a plastic bag and frozen (Ernest et al. 2000). If time allows, we will continue checking stretches for additional tracks.

Hair and scat samples will be processed at the USGS Fort Collins Science Center. When possible, we will extract DNA from 10 hairs (Goossens et al. 1998, Boersen et al. 2003) using Qiagen DNeasy® Tissue Kits or from scat using Qiagen Stool DNA Kits (Qiagen Inc., Valencia, CA). Once the DNA has been extracted, samples will be genotyped using 9 – 12 microsatellite primers shown by Ernest et al. (2000), Sinclair et al. (2001), and Anderson et al. (2004) to have high variability in cougars. The DNA will be amplified by polymerase chain reaction (PCR) using a M13-tailed forward primer as described by Boutin-Ganache et al. (2001). When possible, we will compare the results from hair and scat genotyping with archived blood and tissue samples from the same individuals.

At the end of the season, we will consider how many samples we were able to locate and genotype correctly relative to the effort we applied. We will determine if snow tracking could be useful as a complement to our lure site component. We will also consider winter tracking with regard to cougars previously captured and collared by CPW by quantifying the proportion of marked to unmarked animals detected during snow tracking.

Timeline

February – April 2012	Pilot Season
November 17 – December 20, 2012	1st Sampling period
December 21 – January 23, 2013	2nd Sampling period
January 24 - February 26, 2013	3rd Sampling period
February 27 – April 1, 2013	4th Sampling period
November – April, 2013	Snow tracking efforts

Lures will be tested (Chapter 1) during the pilot season and the 1st – 3rd sampling periods. The detection probability (Chapter 2) and capture variation (Chapter 3) will be considered during the 1st – 4th sampling periods. Snow tracking efforts (Chapter 4) will be dictated by snow conditions throughout the winter.

Literature Cited

Anderson, C. R., F. G. Lindzey, and D. B. McDonald. 2004. Genetic structure of cougar populations across the Wyoming Basin: Metapopulation or megapopulation. Journal of Mammalogy **85**:1207-1214.

Beier, P., D. Choate, and R. H. Barrett. 1995. MOVEMENT PATTERNS OF MOUNTAIN LIONS DURING DIFFERENT BEHAVIORS. Journal of Mammalogy **76**:1056-1070.

Belant, J. L., J. F. Van Stappen, and D. Paetkau. 2005. American black bear population size and genetic diversity at Apostle Islands National Lakeshore. Ursus **16**:85-92.

Belant, J. L., Seamans, T. W., Paetkau, D. 2007. Genetic Tagging Free-Ranging White-Tailed Deer Using Hair Snares. Ohio Journal of Science **107**:50-56.

Boersen, M. R., J. D. Clark, and T. L. King. 2003. Estimating black bear population density and genetic diversity at Tensas River, Louisiana using microsatellite DNA markers. Wildlife Society Bulletin **31**:197-207.

Boulanger, J., M. Proctor, S. Himmer, G. Stenhouse, D. Paetkau, and J. Cranston. 2006. An empirical test of DNA mark-recapture sampling strategies for grizzly bears. Ursus **17**:149-158.

Boutin-Ganache, I., M. Raposo, M. Raymond, and C. F. Deschepper. 2001. M13-tailed primers improve the readability and usability of microsatellite analyses performed with two different allele-sizing methods. Biotechniques **31**:24-+.

Broquet, T., N. Menard, and E. Petit. 2007. Noninvasive population genetics: a review of sample source, diet, fragment length and microsatellite motif effects on amplification success and genotyping error rates. Conservation Genetics **8**:249-260.

Buchan, J. C., E. A. Archie, R. C. Van Horn, C. J. Moss, and S. C. Alberts. 2005. Locus effects and sources of error in noninvasive genotyping. Molecular Ecology Notes **5**:680-683.

Burnham, K. P. and W. S. Overton. 1978. ESTIMATION OF SIZE OF A CLOSED POPULATION WHEN CAPTURE PROBABILITIES VARY AMONG ANIMALS. Biometrika **65**:625-633.

Burnham, K. P. and G. C. White. 2002. Evaluation of some random effects methodology applicable to bird ringing data. Journal of Applied Statistics **29**:245-264.

Castro-Arellano, I., C. Madrid-Luna, T. E. Lacher, and L. León-Paniagua. 2008. Hair-Trap Efficacy for Detecting Mammalian Carnivores in the Tropics. Journal of Wildlife Management **72**:1405-1412.

Chamberlain, M. J., J. W. Mangrum, B. D. Leopold, and E. P. Hill. 1999. A comparison of attractants used for carnivore track surveys. Southeastern Association Fish & Wildlife Agencies (Seafwa), Tallahassee.

Choate, D. M., M. L. Wolfe, and D. C. Stoner. 2006. Evaluation of Cougar Population Estimators in Utah. Wildlife Society Bulletin **34**:782-799.

Crooks, K. R., M. Grigione, A. Scoville, and G. Scoville. 2008. Exploratory Use of Track and Camera Surveys of Mammalian Carnivores in the Peloncillo and Chiricahua Mountains of Southeastern Arizona. The Southwestern Naturalist **53**:510-517.

Culver, M., W. E. Johnson, J. Pecon-Slattery, and S. J. O'Brien. 2000. Genomic ancestry of the American puma (Puma concolor). Journal of Heredity **91**:186-197.

Dickson, B. G. and P. Beier. 2002. Home-Range and Habitat selection by adult cougars in Southern California. Journal of Wildlife Management **66**:1235-1245.

Diefenbach, D. R., M. J. Conroy, R. J. Warren, W. E. James, L. A. Baker, and T. Hon. 1994. A TEST OF THE SCENT-STATION SURVEY TECHNIQUE FOR BOBCATS. Journal of Wildlife Management **58**:10-17.

Diefenbach, D. R., J. L. Laake, and G. L. Alt. 2004. Spatio-temporal and demographic variation in the harvest of black bears: Implications for population estimation. Journal of Wildlife Management **68**:947-959.

Downey, P. J., E. C. Hellgren, A. Caso, S. Carvajal, and K. Frangioso. 2007. Hair Snares for Noninvasive Sampling of Felids in North America: Do Gray Foxes Affect Success? Journal of Wildlife Management **71**:2090-2094.

Dreher, B. P., S. R. Winterstein, K. T. Scribner, P. M. Lukacs, D. R. Etter, G. J. M. Rosa, V. A. Lopez, S. Libants, and K. B. Filcek. 2007. Noninvasive Estimation of Black Bear Abundance Incorporating Genotyping Errors and Harvested Bear. Journal of Wildlife Management **71**:2684-2693.

Ebert, C., D. Huckschlag, H. K. Schulz, and U. Hohmann. 2009. Can hair traps sample wild boar (Sus scrofa) randomly for the purpose of non-invasive population estimation? European Journal of Wildlife Research **56**:583-590.

Ernest, H. B., M. C. T. Penedo, B. P. May, M. Syvanen, and W. M. Boyce. 2000. Molecular tracking of mountain lions in the Yosemite Valley region in California: genetic analysis using microsatellites and faecal DNA. Molecular Ecology **9**:433-441.

Garshelis, D. L. and L. G. Visser. 1997. Enumerating megapopulations of wild bears with an ingested biomarker. Journal of Wildlife Management **61**:466-480.

Gompper, M. E., R. W. Kays, J. C. Ray, S. D. Lapoint, D. A. Bogan, and J. R. Cryan. 2006. A comparison of noninvasive techniques to survey carnivore communities in northeastern North America. Wildlife Society Bulletin **34**:1142-1151.

Goossens, B., L. P. Waits, and P. Taberlet. 1998. Plucked hair samples as a source of DNA: reliability of dinucleotide microsatellite genotyping. Molecular Ecology **7**:1237-1241.

Gould, W. R. and J. D. Nichols. 1998. Estimation of temporal variability of survival in animal populations. Ecology **79**:2531-2538.

Harrison, R. L. 1997. Chemical attractants for central American felids. Wildlife Society Bulletin **25**:93-97.

Harrison, R. L. 2006. A Comparison of Survey Methods for Detecting Bobcats. Wildlife Society Bulletin **34**:548-552.

Hayward, G. D., D. G. Miquelle, E. V. Smirnov, and C. Nations. 2002. Monitoring Amur tiger populations: characteristics of track surveys in snow. Wildlife Society Bulletin **30**:1150-1159.

Hemker, T. P., F. G. Lindzey, and B. B. Ackerman. 1984. POPULATION CHARACTERISTICS AND MOVEMENT PATTERNS OF COUGARS IN SOUTHERN UTAH. Journal of Wildlife Management **48**:1275-1284.

Hornocker. M. G. 1969. WINTER TERRITORIALITY IN MOUNTAIN LIONS. Journal of Wildlife Management **33**:457-464.

Huggins, R. M. 1989. ON THE STATISTICAL-ANALYSIS OF CAPTURE EXPERIMENTS. Biometrika **76**:133-140.

Immell, D. and R. G. Anthony. 2008. Estimation of Black Bear Abundance Using a Discrete DNA Sampling Device. Journal of Wildlife Management **72**:324-330.

Kendall, K. C., J. B. Stetz, D. A. Roon, L. P. Waits, J. B. Boulanger, and D. Paetkau. 2008. Grizzly Bear Density in Glacier National Park, Montana. Journal of Wildlife Management **72**:1693-1705.

Kendall, W. L. 1999. Robustness of closed capture-recapture methods to violations of the closure assumption. Ecology **80**:2517-2525.

Kery, M., B. Gardner, T. Stoeckle, D. Weber, and J. A. Royle. 2011. Use of Spatial Capture-Recapture Modeling and DNA Data to Estimate Densities of Elusive Animals. Conservation Biology **25**:356-364.

Lindzey, F. G., W. D. Vansickle, B. B. Ackerman, D. Barnhurst, T. P. Hemker, and S. P. Laing. 1994. COUGAR POPULATION-DYNAMICS IN SOUTHERN UTAH. Journal of Wildlife Management **58**:619-624.

Link, W. A. and J. D. Nichols. 1994. ON THE IMPORTANCE OF SAMPLING VARIANCE TO INVESTIGATIONS OF TEMPORAL VARIATION IN ANIMAL POPULATION-SIZE. Oikos **69**:539-544.

Link, W. A. 2003. Nonidentifiability of population size from capture-recapture data with heterogeneous detection probabilities. Biometrics **59**:1123-1130.

Long, E. S., D. M. Fecske, R. A. Sweitzer, J. A. Jenks, B. M. Pierce, and V. C. Bleich. 2003. Efficacy of photographic scent stations to detect mountain lions. Western North American Naturalist **63**:529-532.

Long, R. A., T. M. Donovan, P. Mackay, W. J. Zielinski, and J. S. Buzas. 2007. Comparing Scat Detection Dogs, Cameras, and Hair Snares for Surveying Carnivores. Journal of Wildlife Management **71**:2018-2025.

Long, R. A., P. MacKay, W. J. Zielinski, and J. C. Ray. 2008. Noninvasive Survey Methods for Carnivores. Island Press, Washington, DC.

Lukacs, P. M. and K. P. Burnham. 2005. Estimating population size from DNA-based closed capture-recapture data incorporating genotyping error. Journal of Wildlife Management **69**:396-403.

McDaniel, G. W., K. S. McKelvey, J. R. Squires, and L. F. Ruggiero. 2000. Efficacy of lures and hair snares to detect lynx. Wildlife Society Bulletin **28**:119-123.

McKelvey, K. S., J. Von Kienast, K. B. Aubry, G. M. Koehler, B. T. Maletzke, J. R. Squires, E. L. Lindquist, S. Loch, and M. K. Schwartz. 2006. DNA analysis of hair and scat collected along snow tracks to document the presence of Canada lynx. Wildlife Society Bulletin **34**:451-455.

Menotti-Raymond, M. A. and S. J. O'Brien. 1995. Evolutionary Conservation of Ten Microsatellite Loci in Four Species of Felidae. Journal of Heredity **86**:319-322.

Menotti-Raymond, M., V. A. David, L. A. Lyons, A. A. Schaffer, J. F. Tomlin, M. K. Hutton, and S. J. O'Brien. 1999. A genetic linkage map of microsatellites in the domestic cat (Felis catus). Genomics **57**:9-23.

Mills, L. S., K. L. Pilgrim, M. K. Schwartz, and K. McKelvey. 2000. Identifying lynx and other North American felids based on MtDNA analysis. Conservation Genetics **1**:285-288.

Mondol, S., R. Navya, V. Athreya, K. Sunagar, V. M. Selvaraj, and U. Ramakrishnan. 2009. A panel of microsatellites to individually identify leopards and its application to leopard monitoring in human dominated landscapes. Bmc Genetics **10**.

Mowat, G. and C. Strobeck. 2000. Estimating population size of grizzly bears using hair capture, DNA profiling, and mark-recapture analysis. Journal of Wildlife Management **64**:183-193.

Nicolai, C. A., P. L. Flint, and M. L. Wege. 2005. Annual survival and site fidelity of northern pintails banded on the Yukon-Kuskokwim Delta, Alaska. Journal of Wildlife Management **69**:1202-1210.

Noyce, K. V., D. L. Garshelis, and P. L. Coy. 2001. Differential vulnerability of black bears to trap and camera sampling and resulting biases in mark-recapture estimates.

Otis, D. L., K. P. Burnham, G. C. White, and D. P. Andersen. 1978. Statistical inference from capture data on closed animal populations. Wildlife Monographs 62.

Ott, R. L., M. Longnecker. 2010. An Introduction to Statistical Methods and Data Analysis. Brooks/Cole. Belmont, CA.

Pauli, J. N., J. P. Whiteman, M. D. Riley, and A. D. Middleton. 2010. Defining Noninvasive Approaches for Sampling of Vertebrates. Conservation Biology **24**:349-352.

Pledger, S. 2000. Unified maximum likelihood estimates for closed capture-recapture models using mixtures. Biometrics **56**:434-442.

Pollock, K. H., J. D. Nichols, C. Brownie, and J. E. Hines. 1990. STATISTICAL-INFERENCE FOR CAPTURE-RECAPTURE EXPERIMENTS. Wildlife Monographs:1-97.

Poole, K. G., G. Mowat, and D. A. Fear. 2001. DNA-based population estimate for grizzly bears Ursus arctos in northeastern British Columbia, Canada. Wildlife Biology **7**:105-115.

Proctor, M., B. McLellan, J. Boulanger, C. Apps, G. Stenhouse, D. Paetkau, and G. Mowat. 2010. Ecological investigations of grizzly bears in Canada using DNA from hair, 1995–2005: a review of methods and progress. Ursus **21**:169-188.

Roon, D. A., L. P. Waits, and K. C. Kendall. 2003. A quantitative evaluation of two methods for preserving hair samples. Molecular Ecology Notes **3**:163-166.

Ruell, E. W. and K. R. Crooks. 2007. Evaluation of Noninvasive Genetic Sampling Methods for Felid and Canid Populations. Journal of Wildlife Management **71**:1690-1694.

Ruell, E. W., S. P. D. Riley, M. R. Douglas, J. P. Pollinger, and K. R. Crooks. 2009. Estimating Bobcat Population Sizes and Densities in a Fragmented Urban Landscape Using Noninvasive Capture–Recapture Sampling. Journal of Mammalogy **90**:129-135.

Sargeant, G. A., D. H. Johnson, and W. E. Berg. 1998. Interpreting carnivore scent-station surveys. Journal of Wildlife Management **62**:1235-1245.

Sawaya, M. A., T. K. Ruth, S. Creel, J. J. Rotella, J. B. Stetz, H. B. Quigley, and S. T. Kalinowski. 2011. Evaluation of noninvasive genetic sampling methods for cougars in Yellowstone National Park. The Journal of Wildlife Management **75**:612-622.

Schmidt, K. and R. Kowalczyk. 2006. Using scent marking stations to collect hair samples to monitor eurasian lynx populations. Wildlife Society Bulletin **32**:462-466.

Seidensticker, J. C., M. G. Hornocker, W. V. Wiles, and J. P. Messick. 1973. Mountain lion social organization in the Idaho primitive area. Wildlife Monographs **35**:3-60.

Settlage, K. E., F. T. Van Manen, J. D. Clark, and T. L. King. 2008. Challenges of DNA-Based Mark–Recapture Studies of American Black Bears. Journal of Wildlife Management **72**:1035-1042.

Sinclair, E. A., E. L. Swenson, M. L. Wolfe, D. C. Choate, B. Bates, and K. A. Crandall. 2001. Gene flow estimates in Utah's cougars imply management beyond Utah. Animal Conservation **4**:257-264.

Squires, J. R., K. S. McKelvey, and L. F. Ruggiero. 2004. A snow-tracking protocol used to delineate local lynx, Lynx canadensis, distributions. Canadian Field-Naturalist **118**:583-589.

Stanley, T. R. and K. P. Burnham. 1998. Information-theoretic model selection and model averaging for closed-population capture-recapture studies. Biometrical Journal **40**:475-494.

Sweanor, L. L., K. A. Logan, J. W. Bauer, B. Millsap, and W. M. Boyce. 2008. Puma and Human Spatial and Temporal Use of a Popular California State Park. Journal of Wildlife Management **72**:1076-1084.

Taberlet, P., S. Griffin, B. Goossens, S. Questiau, V. Manceau, N. Escaravage, L. P. Waits, and J. Bouvet. 1996. Reliable genotyping of samples with very low DNA quantities using PCR. Nucleic Acids Research **24**:3189-3194.

Taberlet, P. and G. Luikart. 1999. Non-invasive genetic sampling and individual identification. Biological Journal of the Linnean Society **68**:41-55.

Theobald, D. M., D. L. Stevens, D. White, N. S. Urquhart, A. R. Olsen, and J. B. Norman. 2007. Using GIS to generate spatially balanced random survey designs for natural resource applications. Environmental Management **40**:134-146.

Thompson, S. K. 2002. Sampling. John Wiley and Sons, Inc. New York, NY.

Ulizio, T. J., J. R. Squires, D. H. Pletscher, M. K. Schwartz, J. J. Claar, and L. F. Ruggiero. 2006. The efficacy of obtaining genetic-based identifications from putative wolverine snow tracks. Wildlife Society Bulletin **34**:1326-1332.

Weaver, J. L., P. Wood, D. Paetkau, and L. L. Laack. 2005. Use of scented hair snares to detect ocelots. Wildlife Society Bulletin **33**:1384-1391.

White, G. C., B. Leffler, L. Los Alamos National, and P. Los Alamos National Environmental Research. 1982. Capture-recapture and removal methods for sampling closed populations. Los Alamos National Laboratory ; Available from NTIS, Los Alamos, N.M.; Springfield, Va.

White, G. C. and K. P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study **46**:120-139.

Williams, B. K., J. D. Nichols, and M. J. Conroy. 2002. Analysis and Management of Animal Populations. Academic Press. San Diego, CA.

Willson, J. D., C. T. Winne, and B. D. Todd. 2011. Ecological and Methodological Factors Affecting Detectability and Population Estimation in Elusive Species. Journal of Wildlife Management **75**:36-45.

Wilson, G. J. and R. J. Delahay. 2001. A review of methods to estimate the abundance of terrestrial carnivores using field signs and observation. Wildlife Research **28**:151-164.

Woods, J. G., D. Paetkau, D. Lewis, B. N. McLellan, M. Proctor, and C. Strobeck. 1999. Genetic tagging of free-ranging black and brown bears. Wildlife Society Bulletin **27**:616-627.

Colorado Parks and Wildlife
July 2011 – June 2012

WILDLIFE RESEARCH REPORT

State of:	Colorado	:	Parks and Wildlife
Cost Center:	3430	:	Mammals Research
Work Package:	7210	:	Customer Services/Research Support
Task No.:	1	:	Library Services
Federal Aid Project No.	N/A		

Period Covered: July 1, 2011 – June 30, 2012

Author: Kay Horton Knudsen

Personnel: Kay Horton Knudsen, Chad Bishop, Mat Alldredge

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

ABSTRACT

The Colorado Parks and Wildlife (formerly Colorado Division of Wildlife) Research Center Library has existed for several decades in the Ft. Collins office. A library housed in the Denver office was moved to Ft. Collins many years ago. Early librarians, Marian Hershcopf and Jackie Boss, can be credited with the physical organization of the Library including seven decades of Federal Aid reports, almost 50 years of Wildlife Commission reports and a unique book and journal collection.

Jackie Boss retired in April 2007 and the Library was temporarily closed to all services. Kay Horton Knudsen was hired as the new Research Center Librarian and began employment on August 30, 2008. The goal, as stated by a former supervisor, was to reopen the Library and expand the electronic and digital capabilities of library services to the entire Agency.

Chad Bishop became the Mammals Research Leader in July 2009. His duties include supervision of the Research Center Library. Mat Alldredge was Acting Mammals Leader from April to July 2012.

A progress report and current status of the Library are detailed below.

WILDLIFE RESEARCH REPORT

COLORADO PARKS AND WILDLIFE RESEARCH LIBRARY SERVICES

KAY HORTON KNUDSEN

P.N. OBJECTIVE

Provide an effective support program of library services at minimal cost through centralization and enhancement of accountability for Colorado Parks and Wildlife employees, cooperators and wildlife educators.

SEGMENT OBJECTIVES

1. Continue to improve and modernize library services.
2. Continue to develop, improve, and implement the CPW Research Center Library web-site.

SUMMARY OF LIBRARY SERVICES

The Research Center Library celebrates its fourth full year of operation since reopening in 2008. Work continues on upgrading website features, filling literature research requests and taking a more long-term view on improving Library services. Following the merger with State Parks, the librarian reached out to the Parks biology staff to assist with their information retrieval needs.

During the first year of operation, in additional to cleaning and physical organization, a priority task was choosing and implementing a web-based Integrated Library System (ILS) and purchasing statewide access for CPW staff to online research databases. The second year emphasis was on meeting Agency staff and promoting the Library in a series of training demonstrations. Moving into the third and now fourth year of operation, major projects were purchase of a new federated search feature for the Library website, digitization of CPW publications and continued contact with staff statewide to meet their bibliographic research needs. Since the Library serves as a historic archive for CPW publications, each meeting with staff also includes a request to be included in the dissemination of white papers, journal articles and internal reports. Day-to-day duties continue to be responding to research and document retrieval requests, cataloging newly acquired material and digitizing internal CPW reports.

EOS International is the vendor for the ILS. It was decided to initially purchase the basic modules (a hosted system with library catalog, circulation, cataloging and serials control). The Library website was released to CPW staff in March 2009. The next module purchased from EOS was Indexer – this feature allows for full-text searching of PDFs linked to bibliographic records and was implemented in December 2009. The latest modules are Knowledge Builder and Classification Management. They are used to archive and index historic research documents.

In addition to the catalog of books and reports housed in the Ft. Collins Library, the Library website also gives CPW staff access to research databases. Current subscriptions include BioOne, four of EBSCO’s specialty databases (Environment Complete, Fish and Fisheries Worldwide, Wildlife and Ecology Studies Worldwide and Criminal Justice with Full Text), SORA (Avian journals) and the JSTOR Life Sciences collection. The decision was made in late 2011 to discontinue the print subscriptions to many of the major journals. Online access to the journals was retained and continues as the primary usage point for staff. This online subscription often includes the publisher’s full-text online archives; backfiles of major wildlife and aquatic journals were purchased when necessary to expand the full-text capability. CPW staff statewide are authenticated through WildNet and WildPoint (intranet) eliminating the need for individual usernames and passwords.

A federated, or integrated, search feature for the Library website was on the wish-list from day one. Federated searching combines access to the Research Library catalog, all of the third-party databases listed above, as well as most of the online journals into one all-in-one search. EBSCOHost's Integrated Search (EHIS) was chosen in the fall of 2010 and the link was made available on the Library website in the spring of 2011. Library handouts were updated and a new handout created to explain the features and offer tips on the use of the all-in-one search. The entire federated search industry is evolving and the librarian will continue to work with EBSCO staff to resolve problems and maintain links to all resources.

The next major project envisioned at the reopening of the Library was the digitization of CPW publications. Research on various digitization options took place in 2009/2010. An HP printer/scanner with optical character recognition software was purchased, installed and tested by summer 2010. The first document series to be digitized was *Outdoor Facts*. The second series digitized was the much larger *Special Reports* collection. The resulting PDFs are attached to bibliographic records for each title within the series and are available via the Library catalog for download by CPW staff throughout the state. In late 2011, Federal Aid staff in Denver donated a large collection of Terrestrial Federal Aid reports to the Library. It was decided to use these for the next digitization project. With the help of a work-study student from Colorado State University, several decades of early reports have been scanned and uploaded to the Library catalog.

Other projects in the Library this year included: 1) processing journal subscription renewals and updates to include full-text online access, 2) finishing a project to catalog the backlog of theses/dissertations, 3) continuing to add links to PDF formats into the catalog's bibliographic file, 4) printing and cataloging the Data Analysis Unit (DAU) reports, 5) gathering the transition and merger documents as produced by the Division of Wildlife and State Parks Transition teams to maintain a historic record in the Library collection, 6) sorting and organizing the Government Documents collection, and 7) distribution and cataloging of the Mammals, Avian and Aquatics Research annual reports.

The librarian attended the following conferences and workshops: 1) Colorado Association of Libraries annual conference in Loveland, October 2011, 2) Google User's one day workshop at CSU, January 2012, 3) InterLibrary Loan conference, CSU, April 2012, and 4) "Bridging the data divide", 2-day data curation workshop sponsored by NCAR/UCAR/NOAA in Boulder, April 2012. There was also the opportunity throughout the year to participate in several online "webinars" sponsored by various vendors and library agencies to expand knowledge on trends in the library field.

With expanded library services, the number of requests for documents or research assistance has grown. Most questions received in the Library are from CPW staff or from outside researchers (generally consultants and out-of-state natural resources employees). The Library is not open on a walk-in basis to the general public but the librarian does assist the Help Desk at the Denver office with questions they receive. CPW employees generally request journal articles or items from the Library collection; outside researchers most often want a copy of a CPW publication. The chart below shows the number of reference questions and document requests handled by the librarian during the past 4 years. Please note that one request from a CPW staff member may be for multiple journal or book titles. It is also interesting that the current record for number of requests per month was set in January 2012.

	2008/09		2009/10		2010/11		2011/12
July			20		45		28
August	15		25		34		52
September	21		30		37		53
October	33		38		41		42
November	14		28		46		52
December	28		32		34		52
January	33		62		48		**64***
February	30		43		43		43
March	35		36		46		36
April	24		23		30		42
May	13		17		51		53
June	20		26		27		36
TOTAL	**266**		**380**		**482**		**553**

STATISTICS: As of June 30, 2012, the Research Center Library holds 18,512 titles and 24,371 items (these are the multiple copies of a title) and has 142 registered patrons (CPW staff). Usage statistics for the research databases are given in the chart below.   For BioOne and EBSCO the numbers are for the total searches run; for JSTOR the statistics are for the number of successful full-text article requests.

	BioOne	EBSCO searches	JSTOR
July 2011	217	1370	348
August 2011	194	3356	306
September 2011	188	1662	262
October 2011	222	2047	238
November 2011	192	1226	209
December 2011	190	1193	289
January 2012	144	1167	259
February 2012	131	2097	214
March 2012	191	2729	411
April 2012	117	1221	198
May 2012	122	1125	189
June 2012	88	1210	134
TOTAL	**1996**	**20,403**	**3057**

Prepared by ____________________________

Kay Horton Knudsen