

IDAHO DEPARTMENT OF FISH AND GAME

Cal Groen, Director

Project W-160-R-36

Wildlife Research

Supplemental Progress Report



July 1, 2008 to June 30, 2009

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**FEDERAL AID IN WILDLIFE RESTORATION
SUPPLEMENTAL PROJECT PERFORMANCE REPORT**

1. State: Idaho

Grant number: W-160-R

Segment number: 36

Grant name: Wildlife Research

Project title: Graduate Projects

Studies:

- I. Mule Deer and Elk Competition on the Tex Creek Wildlife Management Area in Southeastern Idaho
- II. Influence of Home Range Patterns of Black Bears on CaptureMark-Recapture Estimates Based on Hair Snaring
- III. A Comparative Analysis of the Affects of Road Densities, Hunting Pressure, and Management Strategies on the Movement and Behavior of Black Bears in Northern Idaho
- IV. Effects of Biological Sources of Variation on Mark-Recapture Estimates for Black Bears Based on Non-Invasive Genetic Sampling
- V. Using Demographic Analyses to Develop Monitoring and Management Tools for Wolves in Idaho

2. Report Period: July 1, 2008 to June 30, 2009

Report due date: September 15, 2009

3. Location of work: Panhandle, Clearwater, McCall, Southwest, Magic Valley, Southeast, Upper Snake, and Salmon regions

4. Costs: Preliminary estimated expenditures -

Cost Category	Total Direct	Total Indirect	Total	Federal Share	In-Kind	State Share
Operating	\$88,657	\$15,134	\$103,791	\$77,843	\$0	\$25,948
Personnel	\$0	\$0	\$0	\$0	\$0	\$0
Capital	\$0	\$0	\$0	\$0	\$0	\$0
Total	\$88,657	\$15,134	\$103,791	\$77,843	\$0	\$25,948

5. Objectives:

Conduct research on mule deer, elk, black bears, and wolves; and their habitats to gather biological information to provide valid, scientifically-based information for wildlife managers to make sound wildlife conservation and management decisions.

6. If the work in this grant was part of a larger undertaking with other components and funding, present a brief overview of the larger activity and the role of this project.

Studies under Graduate Projects are part of a larger wildlife research program. Graduate student projects represent contributing subsets of the main project studies. Overall, the Wildlife Research Program is designed to collect information, analyze results, and disseminate information for wildlife managers and policy makers to assist with conservation and management actions.

7. Describe how the objectives were met.

Mule Deer and Elk competition on the Tex Creek Wildlife Management Area in Southeastern Idaho -

ABSTRACT: Populations of mule deer (*odocoileus hemionus*) have declined throughout western North America over the past 40 years, whereas populations of North American elk (*cervus elaphus*) have increased. Consequently, understanding how and to what degree competition with elk affects mule deer populations is important for minimizing future declines. We studied patterns of resource selection and space use by mule deer and elk on winter range at Tex Creek Wildlife Management Area near Idaho Falls, Idaho, USA, during January-March 2007 and January-April 2008 using conditional logistic regression. Modeling results differed significantly between ungulate species and years. During 2007 (a mild winter), mule deer selected low elevations, steep slopes, and areas close to roads, and avoided areas used by elk. In contrast, during 2008 (a severe winter) weather conditions forced populations of both ungulate species onto a restricted wintering area. As a result, spatial overlap of mule deer and elk increased substantially, and patterns of resource selection were similar between species. Mule deer on winter range avoided areas used by elk when possible, but their ability to do so may be limited in some instances by weather conditions or other environmental factors that restrict the distributions of both species.

A detailed progress report for this study is provided in Appendix I.

Influence of Home Range Patterns of Black Bears on Capture-Mark-Recapture Estimates Based on Hair Snaring -

ABSTRACT: We collected more than 600 black bear hair samples during FY09. Of those hair samples, 540 were submitted to Wildlife Genetics International for DNA-based analysis. They provided individual identification for each sample. Analysis of these data will begin in January 2010 and be complete (with a draft manuscript) by the end of FY10.

A detailed progress report for this study is provided in Appendix I.

A Comparative Analysis of the Affects of Road Densities, Hunting Pressure, and Management Strategies on the Movement and Behavior of Black Bears in Northern Idaho

ABSTRACT: The black bear population within the Coeur d'Alene (CDA) River watershed of Northern Idaho is exposed to high forest road densities and moderately high hunting pressure. Hunting regulations allow for the use of bait and dogs in both spring and fall hunting seasons, and dogs are permitted for non-lethal pursuit during the summer pursuit season. To better understand the effects of these pressures, fine scale analysis of habitat selection and movement behavior of these bears is required. Between 1 June 2007, and 25 July 2008, I instrumented 28 adult black bears with Global Positioning Systems (GPS) collars set to acquire positions at 20-minute intervals. I downloaded data from collars retrieved during fall 2007, and winters of 2008 and 2009. I will place these locations into a Geographical Information System (GIS) framework, and combine with topographic, vegetation, hydrological, and human use variables. I will estimate resource selection functions (RSFs) of used versus available habitat covariates at the 2nd and 3rd order of habitat selection using logistic and paired logistic regression. From these RSFs, general habitat selection as well as spatial and temporal relationships between bear locations and roads will be established.

A detailed progress report for this study is provided in Appendix I.

Effects of Biological Sources of Variation on Mark-Recapture Estimates for Black Bears Based on Non-Invasive Genetic Sampling -

ABSTRACT: I conducted research on the demography of a harvested north Idaho black bear (*Ursus americanus*) population to determine the underlying dynamics of changes in population abundance and how much these dynamics were driven by variation in food productivity. I also evaluated how these processes could influence inferences based on mark-recapture analysis. In cooperation with Idaho Department of Fish and Game (Department) and the USDA Forest Service, I used barb-wire corrals to collect black bear DNA during 2003-2006 in the Purcell Mountains of Idaho. We analyzed these DNA samples to determine the number of uniquely identified individuals in each year, N_u . I used a combination of both

genetic and mark-recapture analyses to evaluate the sources of variation in N_u over the 4 years and to what extent this variation was driven by changes in productivity of foods on the landscape. Specifically, I investigated variation in allele frequencies and genetic diversity in relation to changes in abundance and whether variation in vital rates were a function of changing berry productivity in the study area. I found significant variation of allele frequencies over the years and a heterozygote deficiency indicating I sampled ≤ 4 subpopulations within the same area over the 4 years (a Wahlund Effect). My mark-recapture analyses suggest this pattern was probably in response to landscape changes in berry abundance. My results suggest important variation in population dynamics driven by changes in food productivity, which should be considered when using mark-recapture analyses to monitor population trends for black bears.

A detailed progress report for this study is provided in Appendix I.

Using Demographic Analyses to Develop Monitoring and Management Tools for Wolves in Idaho

A Ph.D. student was selected and enrolled in classes and began work on a study plan during FY09. After 1 academic year, the student chose to pursue other research opportunities. A national search was conducted to find a replacement. A replacement student was identified and began working on this project on 1 December 2009.

8. Discuss differences between work anticipated in grant proposal and grant agreement, and that actually carried out with Federal Aid grant funds.

The work accomplished during the reporting period corresponds closely with that described in the grant proposal. The one exception was with finding a replacement graduate student for the “*Using Demographic Analyses to Develop Monitoring and Management Tools for Wolves in Idaho*” project.

9. List any publications or in-house reports resulting from this work.

Atwood, M.P., P. Zager, J.J. Millsbaugh, M.D. Matocq, R.T. Bowyer, and J.G. Kie. 2009. Fecal indices in mule deer wintering in close proximity to elk. Western States Deer and Elk Workshop, Spokane, WA. (presentation).

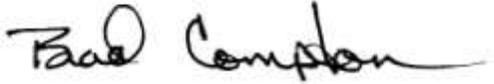
Stenglein, J., C. Mack, D. Ausband, M. Mitchell, P. Zager, S. Nadeau, and L. Waits. 2009. Monitoring an Idaho gray wolf population: a noninvasive approach. Idaho Chapter of The Wildlife Society, Moscow. (presentation).

Stenglein, J.L., L.P. Waits, D.E. Ausband, P. Zager, and C.M. Mack. An efficient noninvasive genetic sampling approach for high-resolution monitoring of a reintroduced wolf population (submitted, revised, and re-submitted).

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APPENDIX I
Detailed Progress Reports for Graduate Projects

STUDY I: Mule Deer and Elk Competition on the Tex Creek Wildlife Management Area in Southeastern Idaho

Resource Selection by Mule Deer on Winter Range: Effects of North American Elk

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ABSTRACT Populations of mule deer (*odocoileus hemionus*) have declined throughout western North America over the past 40 years, whereas populations of North American elk (*cervus elaphus*) have increased. Consequently, understanding how and to what degree competition with elk affects mule deer populations is important for minimizing future declines. We studied patterns of resource selection and space use by mule deer and elk on winter range at Tex Creek Wildlife Management Area near Idaho Falls, Idaho, USA, during January-March 2007 and January-April 2008 using conditional logistic regression. Modeling results differed significantly between ungulate species and years. During 2007 (a mild winter), mule deer selected low elevations, steep slopes, and areas close to roads, and avoided areas used by elk. In contrast, during 2008 (a severe winter) weather conditions forced populations of both ungulate species onto a restricted wintering area. As a result, spatial overlap of mule deer and elk increased substantially, and patterns of resource selection were similar between species. Mule deer on winter range avoided areas used by elk when possible, but their ability to do so may be limited in some instances by weather conditions or other environmental factors that restrict the distributions of both species.

KEY WORDS *Cervus elaphus*, competition, elk, mule deer, *Odocoileus hemionus*, resource selection function.

INTRODUCTION

Populations of mule deer (*Odocoilius hemonius*) have undergone several population fluctuations in the last century (Connolly 1981). Mule deer declined in the 1970s, probably because of a combination of factors ranging from overbrowsing by mule deer, overgrazing by livestock, unfavorable weather patterns, and vegetation succession that

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lead to less productive habitats (Connolly 1981, Mackie et al. 1998). The ultimate reasons for decline, however, are poorly understood. From the 1970s to present, western North America also experienced a marked recovery of North American elk (*Cervus elaphus*). The decline of mule deer along with a corresponding rise in elk numbers has raised the question, are elk negatively affecting populations of mule deer? Interspecific competition with other large herbivores has been hypothesized as a cause of declining mule deer populations (Mower and Smith 1989, Johnson et al. 2000, Stewart et al. 2002). A more complete understanding of the competitive relationship between mule deer and elk is critical for managing declining populations of mule deer.

Morphological and physiological differences between mule deer and elk should have allowed them to exploit habitats differently when critical components of their environment were limited (Schoener 1982). Elk are more generalist feeders than are mule deer, and elk are able to use larger amounts of lower-quality forage than mule deer (Kie et al. 2003). Conversely, mule deer are more selective feeders and require higher-quality forage (Kie et al. 2003). Therefore, elk are well adapted to high-biomass, herbaceous, or mixed-grass communities, whereas mule deer are adapted to low-biomass, high quality, shrubby habitats (Wickstrom et al. 1984).

Changes in habitats or population densities can alter competitive relationships (Mower and Smith 1989). North American ungulates have had to cope with extensive loss of habitat over the past century, and habitats that currently support mule deer and elk have changed so that conditions are now much different from those that existed while mule deer and elk coevolved (Nelson 1982). Elk are a superior competitor to mule deer under many circumstances (Mackie 1981). Advantages in thermodynamic range (Parker and Robbins 1984), energy costs for locomotion in snow (Wickstrom et al. 1984), the ability to use a greater diversity of plant species as food (Collins and Urness 1983, Wickstrom et al. 1984), and less risk to shared predators may give elk the competitive advantage in areas where the 2 species are sympatric (Lindzey et al. 1997).

Large ungulates in temperate and arctic regions must accumulate fat stores during summer and autumn to support them through the winter when forage quality and quantity will not meet maintenance requirements (Mautz 1978). Although summer range is a crucial part of over-winter survival and subsequent fitness for ungulates, mule deer and elk cannot successfully meet their nutritional requirements during winter (Mautz 1978, Pederson and Harper 1978, Cook et al. 2004, Stewart et al. 2005). These life-history traits of mule deer and elk make winter a logical starting point for investigating competition between these large cervids.

Mower and Smith (1989) noted that resource overlap between mule deer and elk was most prominent during winter. Those authors reported that deep snow covered grasses, which forced elk to rely on browse that wintering mule deer populations also used. Stewart et al. (2002) documented that diminishing quality of forage during autumn forced mule deer to move into areas used by elk. Schwartz and Ellis (1981) reported that dietary overlap increased as forage became more limited to sympatric grassland herbivores, and that forage quality may be more important than forage quantity in competition between

selective feeders. Diminishing forage leading to higher dietary overlap has been reported in mule deer and elk (Mower and Smith 1989, Stewart et al. 2002), and mule deer and cattle (Bowyer and Bleich 1984). Stewart et al. (2002) indicated that when cattle were introduced onto range in spring, elk moved to higher elevations, and mule deer shifted to lower elevations. The authors concluded that there was likely a competitive displacement of elk by cattle and a subsequent response to elk movements by mule deer. Stewart et al. (2002) also provided evidence that mule deer were strongly affected by the movements of elk but elk were not affected by movements of mule deer.

Johnson et al. (2000) used resource selection functions (RSF) for mule deer and elk to determine if mule deer and elk were competing during spring. The authors reported resource selection by mule deer was inversely related to resource selection by elk, whereas resource selection by elk was independent of resource selection by mule deer. Elk tended to be on gentler slopes, further from roads and occurred on westerly aspects. Mule deer occurred on steeper slopes, closer to roads, and on easterly aspects. The authors concluded that mule deer were avoiding elk (Johnson et al. 2000).

We hypothesized that mule deer will avoid elk on winter range (i.e., interference competition). We further postulate that varying winter conditions will change the manner in which mule deer and elk select resources. As a corollary, we predict that increasing severity of winter will result in increasing spatial overlap of mule deer and elk. Understanding these relationships between mule deer and elk is critical to developing management strategies to benefit mule deer.

STUDY AREA

We conducted research on the Tex Creek Wildlife Management Area (hereafter Tex Creek) and surrounding lands in southeastern Idaho, USA (Fig 1). Tex Creek is 25 km east of Idaho Falls, Idaho. The area is approximately 34,000 ha, 13,000 ha of which is public land managed by the Department for mule deer and elk winter range. The Tex Creek winter range is a canyon and plateau complex that is bordered by Caribou National Forest to the east. The main canyon is 44 km long with 71 km of tributary canyons (Thomas 1987). Elevations on Tex Creek range from 1550 to 2250 m; temperatures range from -34° C to 37° C annually, and mean annual temperature is 6° C. Mean precipitation ranges 30 to 45 cm moving west to east across Tex Creek and occurs mostly as snow in winter and rain in spring. Plant species important for mule deer and elk winter forage include aspen (*Populus tremuloides*), juniper (*Juniperus* spp.), sagebrush (*Artemisia tridentata*), willow (*Salix* spp.), Douglas-fir (*Pseudotsuga menziesii*), and antelope bitterbrush (*Purshia tridentata*) (Botanical nomenclature from U.S. Department of Agriculture Natural Resources Conservation Service 2009). Common habitat types include Conservation Reserve Program grasslands, agricultural fields, sagebrush steppe, pinion-juniper forest, mixed deciduous-conifer forest, and conifer forest (Thomas 1987).

METHODS

Capture and Radio-collars

We captured adult female mule deer and elk using drive netting and net gunning techniques during January 2007, December 2007, and January 2008 (Barrett 1982, Thomas and Novak 1991). We fitted mule deer and elk with GPS radio-collars (models 2200L, 3300S, and 4400M, Lotek Engineering, Newmarket, ON, Canada). All handling of animals was approved by the Idaho State University Institutional Animal Care and Use Committee and was in compliance with guidelines for the use of mammals in research approved by the American Society of Mammalogists (Gannon et al. 2007).

We programmed GPS collars on mule deer to collect locations 12 times/day and fitted collars with timed drop-off mechanisms (Lotek Engineering, Newmarket, ON, Canada) set to execute 8 months after deployment. We retrieved the dropped radio-collars in summer 2007 and redeployed them with new batteries again in winter 2008. The larger batteries on the elk collars allowed us to program those collars to collect GPS locations 6 times/day, and to fit collars with timed drop-off mechanisms set to execute 18 months after deployment.

The Department conducted population surveys for mule deer during winters 1997, 1999, 2001, 2002, 2003, and 2005, and for elk during winters 1992, 1995, 1997, 2000, and 2005 (Unsworth et al. 1990). We used subunits defined by the Department for these population surveys to determine the distribution of our GPS radio-collars. We calculated a mean population size for the 9 most commonly used (i.e., by deer and elk for winter range) subunits based on these 11 population surveys. We placed GPS radio-collars on mule deer and elk in those 9 subunits in numbers proportional to the population estimates for each subunit to get a representative sample of winter range use by both species. We attached GPS radio-collars to 22 adult female elk and 43 adult female mule deer over the 2 years of our study.

GPS locations and Environmental variables

Not all GPS radio-collars were recovered and not all GPS radiocollared animals returned to the Tex Creek winter range. We obtained 3,504 locations from 16 individual elk and 14,463 locations from 17 individual mule deer during winter 2007. During winter 2008, we obtained 7,764 locations from 16 individual elk and 11,272 locations from 15 individual mule deer. We included GPS locations obtained between January and April and on the Tex Creek winter range in our analysis. Environmental conditions on Tex Creek varied greatly among months and years, and as a result, we divided our analysis into 7 monthly windows (Jan-Mar 2007 and Jan-Apr 2008) to control for confounding effects of environmental conditions on patterns of resource selection. Data from April 2007 were not included in our analyses because animals left the winter range in March and early April of that year.

We used elevation (m), slope (%), aspect (degrees), and distance to the nearest open road (m) as environmental variables to predict elk and mule deer occurrence. We used the

spatial analyst extension in ArcGIS 9.3 (ESRI, Redlands, CA) to import and convert geospatial data (Inside Idaho) into continuous variables for elevation, slope, distance to open road, and aspect. We transformed aspect (a circular variable) into sine and cosine of aspect to measure eastness and northness, respectively (Zar 1996). In addition to other variables we used the predicted values from the final RSF for elk to predict occurrence of mule deer. We used 95% fixed kernel utilization distributions to define the spatial extent of our analyses for elk and mule deer during each of the 7 month and year combinations. As a result, our 7 month and year temporal windows also have varying spatial extents because animals used the winter range differently across months and years.

Resource Selection Functions and Model Averaging

We used ArcGIS to create a set of random locations equal to the number of mule deer and elk locations for each month and year combination; random locations were confined to the 95% utilization distribution of mule deer and elk for each month and year. We used the reference bandwidth to calculate all utilization distributions. We assigned GPS locations to 10×10 m pixels, and spatially joined each location with the corresponding values from the underlying habitat layers.

We employed a matched-case design and used conditional-logistic regression (PROC PHREG; SAS Institute, Cary, North Carolina) to model resource selection by mule deer and elk (Hosmer and Lemeshow 2000, Manly et al. 2002, Boyce 2006). We used a correlation matrix to evaluate collinearity among independent variables (PROC CORR; SAS Institute, Cary, North Carolina), and removed distance to nearest open road from all of our deer models because it was highly correlated with the predicted values from the elk RSF ($|r| \geq 0.81$, $p > 0.01$). We modeled all remaining combinations (4 environmental variables for elk and 3 environmental variables and elk RSF for deer) for each month and year combination. The result was 15 models each for elk and mule deer.

We recorded Akaike's Information Criterion adjusted for small sample size (AIC_c), ΔAIC_c , and the Akaike weight (w_i) for each of our models. Then we selected those models that made up the 95% confidence set based on w_i values (Burnham and Anderson 2002). We calculated weighted model-averaged parameter estimates and unconditional standard errors (SE) for each predictor variable using only models from the 95% confidence set for each month and year combination (Burnham and Anderson 2002). We concluded that model-averaged parameter estimates differed significantly from 0 if a 95% CI around the mean (based on the unconditional SE) did not contain 0. Parameter estimates that were positive indicated that probability of use by mule deer increased with increasing values of that variable (selection), whereas parameter estimates that were negative indicated the opposite relationship (avoidance).

RESULTS

Resource Selection by Mule Deer

Resource selection by mule deer was constant during the 2 contrasting winters. Parameter estimates for percent slope were significant and positive in 6 of 7 month and

year combinations indicating mule deer consistently chose steeper than available slopes during winter (Table 1). Mule deer preference for steeper slopes was exclusive in 2008, although they did choose milder slopes at the end of winter in 2007 (Table 1, Fig 2). Mule deer selection of habitats based on elevation also was consistent across both winters. Mule deer tended to be on lower elevations, parameter estimates for elevation were negative and significant in 6 of 7 month and year combinations (Table 1). Mean elevations of mule deer locations by month were lower in 2007 than in 2008 (Fig 3). Aspect was not selected for or against in our deer models. Mule deer selected habitats differentially with respect to aspect in only 1 of 7 month and year combinations (Jan 2008), in which they chose habitats with westerly aspects (Table 1).

Parameter estimates for the predictive values of the elk RSF were negative and significant in 2 of 3 months in 2007 and positive and significant in all months in 2008 (Table 1). These parameter estimates indicate that mule deer chose habitats further away from elk in 2007 (a mild winter) and consistently chose habitats closer to elk in 2008 (a severe winter). Mule deer and elk were forced onto similar habitats during the more severe winter conditions in 2008.

Resource Selection by Elk

In contrast to mule deer, resource selection by elk with respect to elevation, slope, and aspect was inconsistent. Elk selection of slope was significant for 4 of 7 temporal windows, although mean slope (%) of elk locations was consistently low (< 12%) and tracked mean slope of random locations closely (Table 2, Fig 2). Elk selected gentler slopes in 3 of 7 month and year combinations (Table 2). Elk selected lower elevations once and higher elevations twice during 7 month and year combinations and showed no selection of elevation in 4 of 7 month and year combinations (Table 2). Mean elevations of elk locations closely followed mean elevations of random points early in winter and were higher later in winter (Fig 3).

Aspect was not an important component of resource selection by elk; the only month and year combination that showed significant selection by elk with respect to aspect was February 2008, where elk chose more south and east facing slopes (Table 2). Indeed, distance to open road was the only environmental variable in our models that consistently influenced resource selection by elk. Distance to the nearest open road was both positive and significant in all month and year combinations, indicating that elk consistently chose habitats that were further from open roads than what was available. Mean distance to nearest road was also higher among elk locations than random locations in every month and year combination. Distance to nearest road also was the only environmental variable in any of our models that was unaffected by winter severity (Table 1).

DISCUSSION

Our data provide support for our initial hypothesis that mule deer would avoid elk on winter range (i.e., exhibit interference competition) during the first year of our study, but not the second. Mule deer chose habitats further away from elk during winter 2007 but chose habitats closer to elk during winter 2008. These results provide mixed support for

our initial hypothesis but strong support for our second hypothesis, which was that varying winter conditions on our study area would change the way that mule deer and elk selected resources. Specifically, we predicted that increasing severity of winter would result in greater spatial overlap of mule deer and elk, which occurred (Fig 7).

We recognize that using multiple temporal windows for our resource selection analysis is complex; however, the access to resources by deer and elk changes with varying conditions, which required the design we used. We also recognize that using these multiple temporal windows as a measure of winter severity may be problematic (e.g. mild and severe conditions within the same month). Our objectives, however, were not to determine the effect of winter weather on mule deer and elk but to control for these environmental variables while investigating interactions between these species. We believe that dividing our analysis among month and year combinations to partition varying degrees of winter severity was the best way to accomplish this task.

Large samples of ungulate locations during winter under free-ranging conditions can be difficult to obtain. Limited access (i.e., snowmobile only), severe weather conditions, and monitoring a large geographical area (400 km²) were the most notable obstacles. Our study provides large sample sizes on dozens of individuals over multiple years to investigate patterns of resource selection in mule deer and elk. Severe winter conditions changed the spatial distribution of both species, but the distributions of elk changed more drastically than distribution of mule deer on our study area. The winter range of mule deer contracted and shifted away from areas of deep snow, while elk distribution shifted from ridges and valleys in the eastern part of Tex Creek approximately 10 km to the northwest to the canyon and plateau areas in the western part of Tex Creek, which are core areas of winter range for mule deer (Fig 7).

Johnson et al. (2000) reported selection by mule deer was inversely related to elk resource selection, which led the authors to conclude that mule deer were avoiding elk. Stewart et al. (2002) reported that mule deer avoided elk during the summer but not during autumn when the authors believed that seasonal changes in forage quality force mule deer to move into areas used by elk. These conditions resulted in greater spatial overlap and perhaps some exploitive competition between mule deer and elk (Stewart et al. 2002). Further, Long et al. (In press) reported mule deer avoidance of elk during and immediately after parturition and during mid- to late-summer. The authors hypothesized that periods of avoidance were separated by periods of time when mule deer lacked the ability to avoid elk, likely from limited mobility of neonate mule deer (Long et al. In press).

The life-history characteristics of mule deer and elk are scale sensitive (Bowyer and Kie 2006). Broad spatial scales should provide insight into historically important factors (e.g. annual snow depths, presence of south-facing slopes, etc.) concerning habitat selection by mule deer (Kie et al. 2002). Energy expenditure from locomotion increases with snow depth and energy reserves are depleted faster with colder temperatures (Parker et al. 1984, Wickstrom et al. 1984). As movement becomes more expensive, energy reserves are depleted, and accessible forage diminishes, we hypothesize that avoidance of elk

becomes less beneficial to mule deer. Habitat characteristics such as game trails, windbreaks, solar exposure, thermal cover, and concealment cover that help to decrease energy expenditure would likely increase in importance. These components of habitat are probably not altered by elk. In contrast, habitat components that are possibly affected by elk (e.g. elk consuming browse) are becoming less important.

Competition occurs when mule deer and elk have simultaneous dietary and spatial overlap in habitats where forage is limited (de Boer and Prins 1990, Lindzey 1997). Certainly, forage is limited during winter on most ranges in the northern distribution of mule deer. Dietary overlap among mule deer and elk has been reported on winter range (Mower and Smith 1989, Torstenson et al. 2006), and we have provided evidence of high spatial overlap of mule deer and elk during a severe winter. This evidence, combined with previous documentation of mule deer avoidance of elk (Johnson et al. 2000, Stewart et al. 2002), indicates that avoiding elk may be beneficial to mule deer under a particular set of conditions (e.g. during mild winters with shallow snow).

Mule deer avoided habitats with a high probability of elk use during the milder of 2 winters during our study. In contrast, mule deer showed selection for habitats with a high probability of elk use during the more severe winter. We believe that the selection by mule deer for habitats closer to elk during severe winter conditions are a manifestation of selection against other variables (e.g. deep snow) by mule deer and elk. Our results further corroborate previous evidence that mule deer avoid elk and provide new evidence that the behaviors occur during winter under free-ranging conditions. This avoidance, however, may not be possible during severe winter conditions.

Management Implications

Elk populations are thought to have played a role in mule deer declines on our study area and on ranges across western North America; however, a more pressing question now faces wildlife managers throughout the range of mule deer. Are existing populations of mule deer threatened by elk presence and are potential rebounds in mule deer populations stifled by elk presence? Our results provide further evidence that indicate that mule deer avoid elk under particular circumstances during winter. The consequences of that competition, however, are yet to be determined. We recommend further investigation into the survival and fitness consequences of competition between these unique large herbivores.

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Table 1. Mule deer resource selection (+), avoidance (-), or no significant difference (ns) to predictor variables for resource selection function (RSF) on winter range at Tex Creek, Idaho, USA. We used a model averaged approach and GPS locations collected 24 hrs/day during winters 2007 (mild) and 2008 (severe) to build the RSF.

	Slope	Elevation	Eastness	Northness	Elk RSF
Jan 2007	+	-	ns	ns	-
Feb 2007	+	ns	ns	ns	-
Mar 2007	-	-	ns	ns	ns
Jan 2008	+	-	-	ns	+
Feb 2008	+	-	ns	ns	+
Mar 2008	+	-	ns	ns	+
Apr 2008	+	-	ns	ns	+

Table 2. Elk resource selection (+), avoidance (-), or no significant difference (ns) to predictor variables for resource selection function (RSF) on winter range at Tex Creek, Idaho, USA. We used a model averaged approach and GPS locations collected 24 hours/day during winters 2007 (mild) and 2008 (severe) to build the RSF.

	Slope	Elevation	Eastness	Northness	Distance from open road
Jan 2007	ns	ns	ns	ns	+
Feb 2007	ns	+	ns	ns	+
Mar 2007	-	ns	ns	ns	+
Jan 2008	-	ns	ns	ns	+
Feb 2008	ns	ns	+	-	+
Mar 2008	+	-	ns	ns	+
Apr 2008	-	+	ns	ns	+

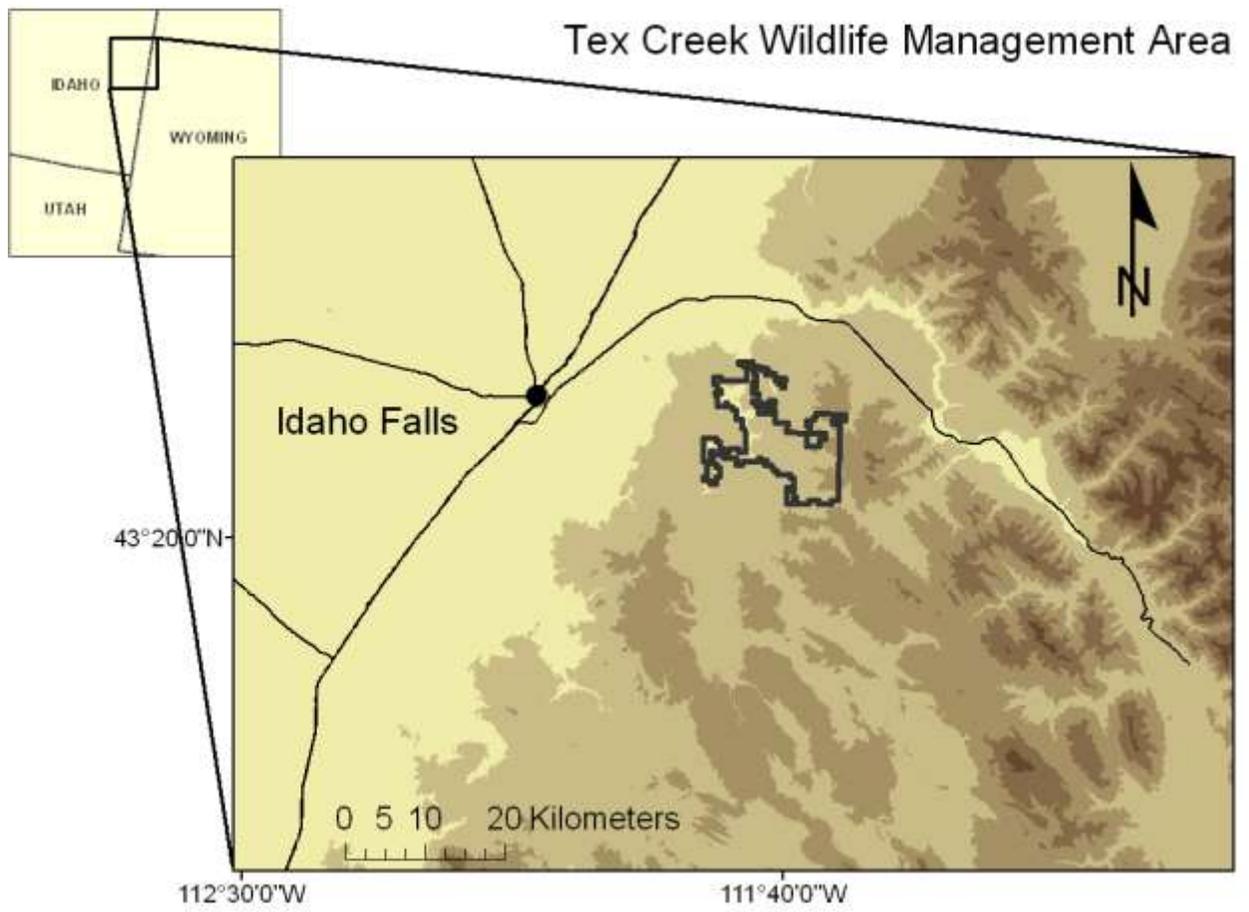


Figure 1. Tex Creek Wildlife Management Area, southeastern Idaho, USA.

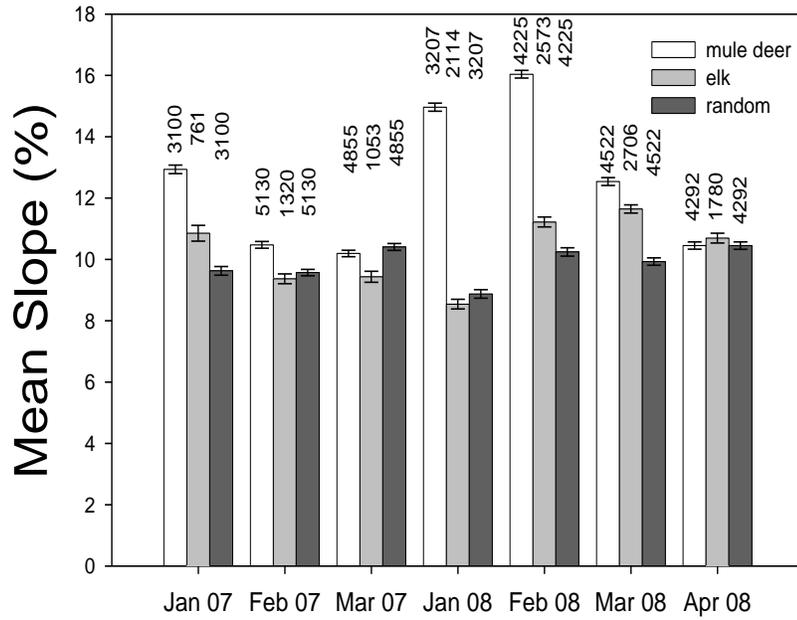


Figure 2. Mean slope (%) where mule deer (open), elk (light grey), and random (dark grey) GPS-collar locations occurred during winters 2007 and 2008 on Tex Creek, Idaho, USA. Sample sizes are above bars and error bars represent SE. Note steeper slopes that mule deer used compared to elk locations and random locations during the more severe winter conditions (Jan-Feb 2008).

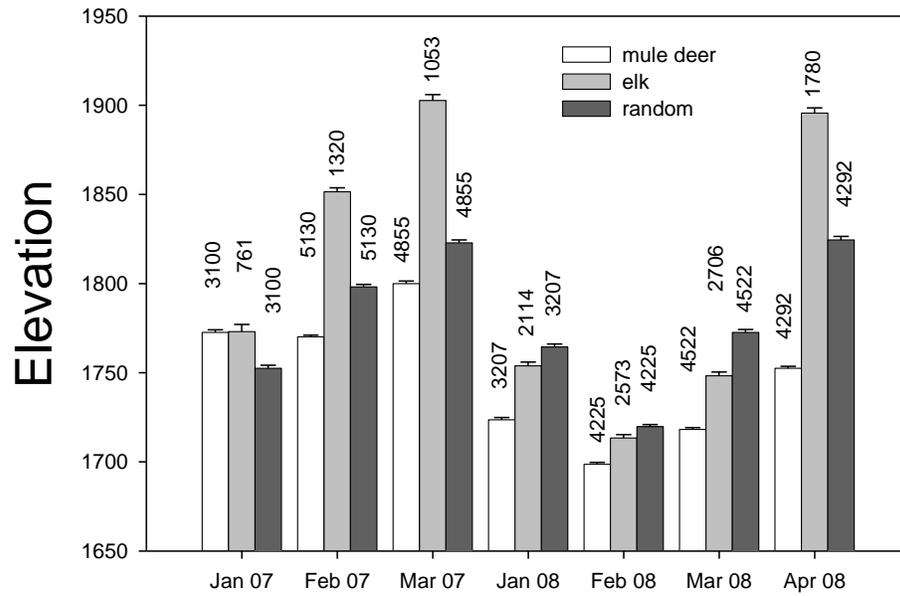


Figure 3. Mean elevation (m) where mule deer (open), elk (light grey), and random (dark grey) GPS-collar locations occurred during winters 2007 and 2008 on Tex Creek, Idaho, USA. Sample sizes are above bars and error bars represent SE.

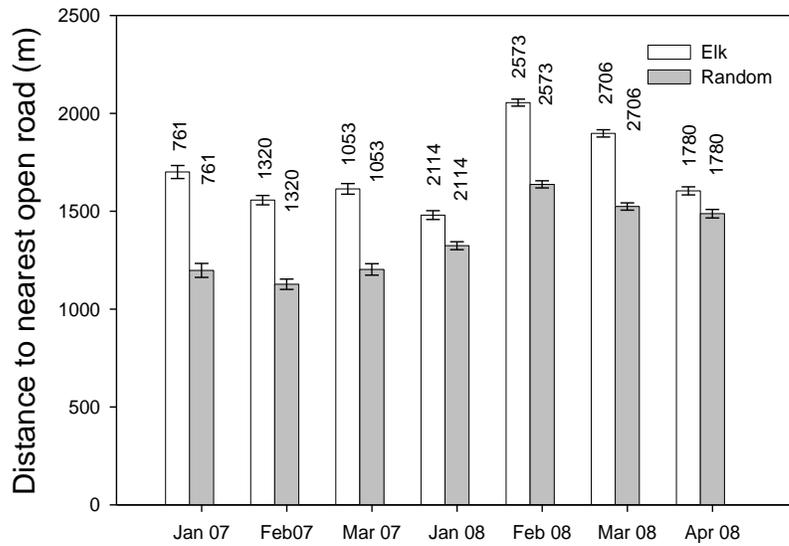


Figure 4. Mean distance from elk (open) and random (light grey) GPS-collar locations to nearest open road (m) during winters 2007 and 2008 on Tex Creek, Idaho, USA. Sample sizes are above bars and error bars represent SE.

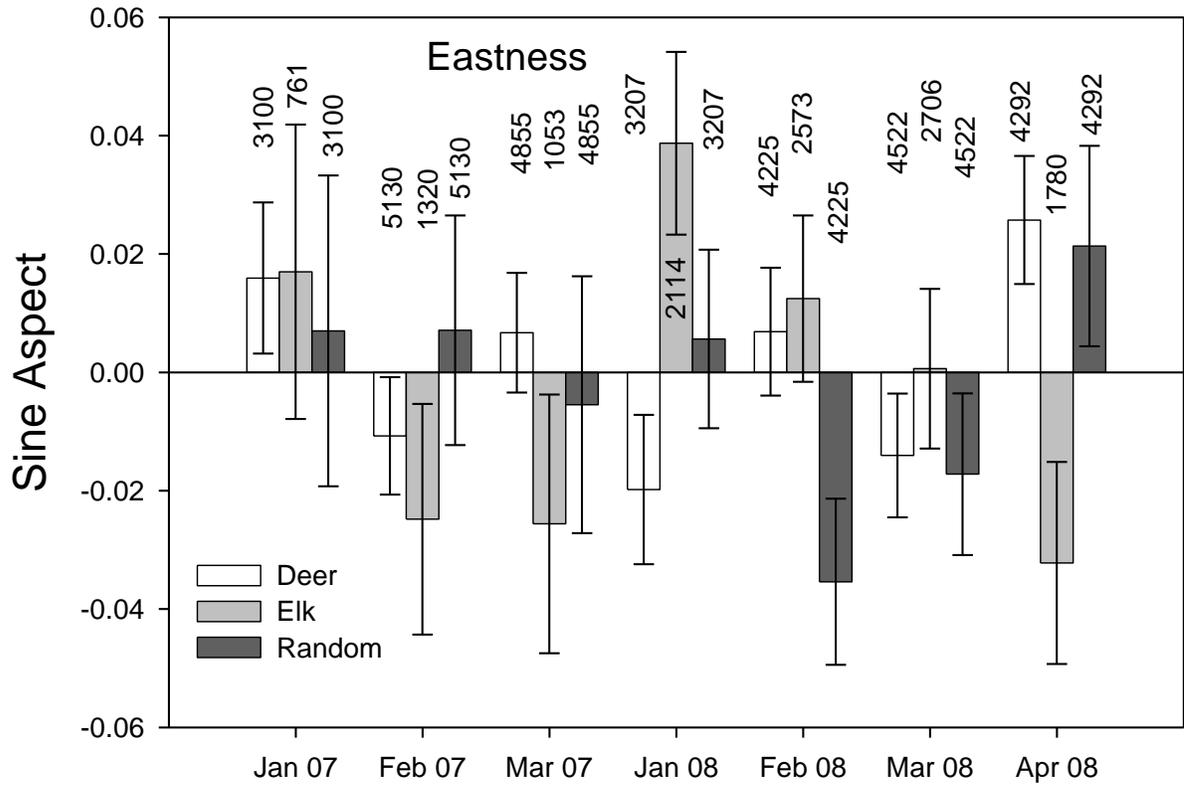


Figure 5. Mean sine of aspect (measure of eastness, east = 1 and west = -1) from mule deer (hollow), elk (light grey), and random (dark grey) locations used in resource selection functions on Tex Creek, Idaho, USA.

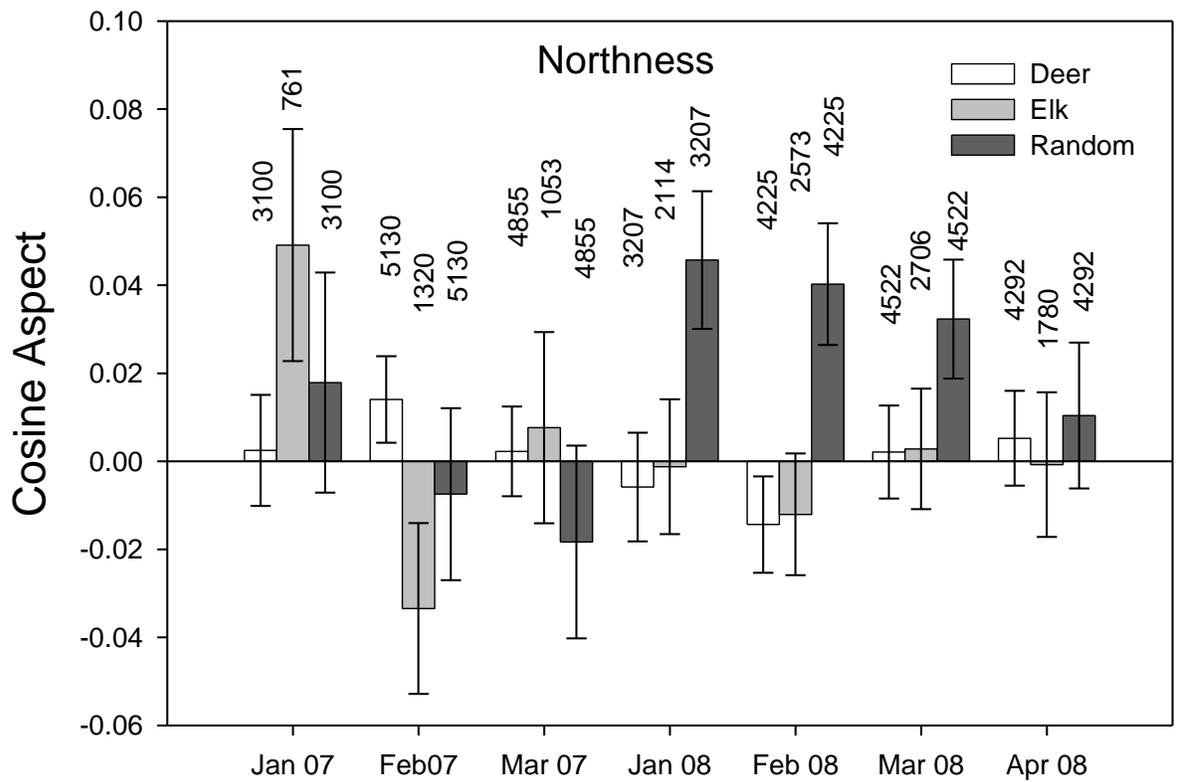


Figure 6. Mean cosine of aspect (measure of northness, north = 1 and south = -1) from mule deer (hollow), elk (light grey), and random (dark grey) locations used in resource selection functions on Tex Creek, Idaho, USA.

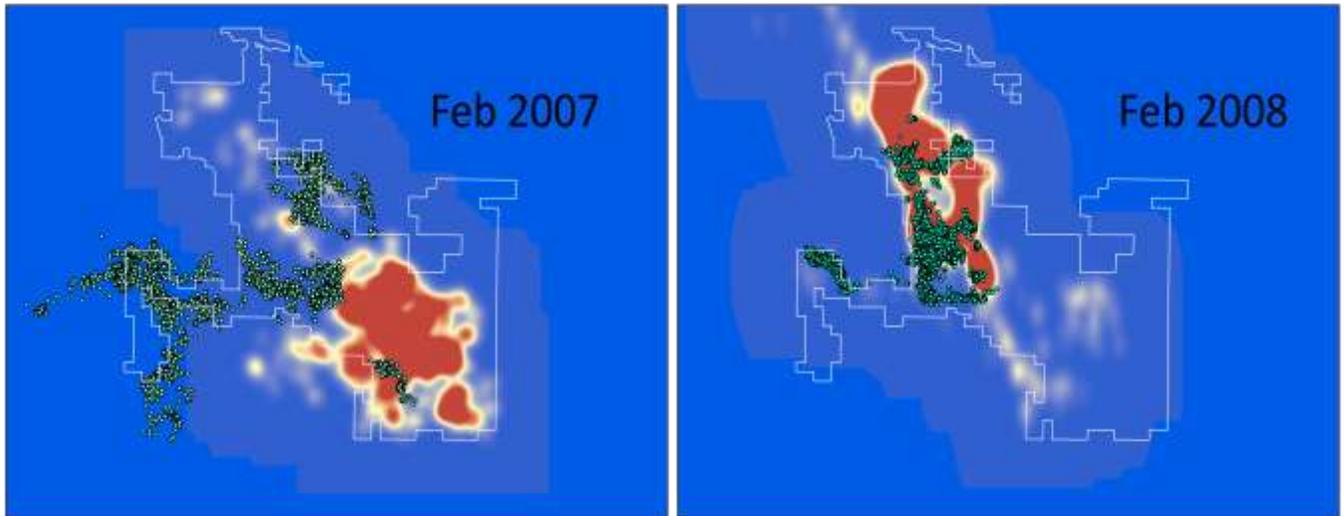


Figure 7. GPS locations of mule deer (green circles) on Tex Creek during February 2007 (mild winter) and February 2008 (severe winter) with elk density. Elk density is represented by a high to low gradient (red to blue, respectively) and was produced using a fixed kernel 95% utilization distribution.

STUDY II: Influence of Home Range Patterns of Black Bears on Capture-Mark-Recapture Estimates Based on Hair Snaring

Influence of home range patterns of black bears on capture-mark-recapture estimates based on hair snaring

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PROBLEM STATEMENT

Capture-mark-recapture (CMR) analyses based on genotyping of hair samples collected from hair trap enclosures are increasingly used for estimating population size and demographic trends for black bears (Woods et al. 1999). The accuracy of such estimates can be strongly dependent on assumptions of equal access to hair stations of all bears within a sampled population and demographic “closure” of the population during sampling. Depending on how sampling grids for hair stations are laid out, the behavior of bears has considerable potential for creating violations of these assumptions. For example, grids with widely spaced hair stations will not accurately sample bears with small home ranges, grids that are too small will not be demographically closed as wide-ranging bears spend little time within the grid, or probabilities of bears visiting a hair station vary and thus not all bears have equal likelihood of being sampled. Though widely acknowledged, this potential has yet to be investigated closely and quantified.

To understand how behavior and activity of bears might influence CMR estimates based on hair trapping, we propose to place a grid of hair trap enclosures in the Coeur d’Alene River drainage where we have 15 GPS collars deployed on black bears from 2007-2009. Conducting simultaneous sampling of behavior and hair snaring offers a unique opportunity to evaluate the extent to which heterogeneity in bear behavior has the potential to violate important assumptions of CMR analysis.

OBJECTIVES

We will use hair collected from the sampling grid to generate estimates of population size through analysis of captures and recaptures of genotyped individuals. We will use GPS location data to:

- 1) Estimate 95% kernel home ranges so we can evaluate the potential influence on estimates of a scaled relationship between:
 - a) Spacing of hair stations and home range size,

- b) Spatial extent of the hair trapping grid and home range size, and
2) Map movements of individual bears with respect to hair stations, to evaluate variability in the likelihood of bears to visit hair stations in their vicinity (capture probability).

STUDY AREA

Game Management Unit 4, Coeur d'Alene River drainage.

TIMEFRAME

FY08 – Field collection.

FY09 – DNA analysis of samples collected during early summer 2008, initial data analysis.

FY10 – complete data analysis and writing.

PROGRESS

We collected more than 600 black bear hair samples during FY09. Of those hair samples, 540 were submitted to Wildlife Genetics International for DNA-based analysis. They provided individual identification for each sample. Analysis of these data will begin in January 2010 and be complete (with a draft manuscript) by the end of FY10.

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STUDY III: A Comparative Analysis of the Affects of Road Densities, Hunting Pressure, and Management Strategies on the Movement and Behavior of Black Bears in Northern Idaho

Multi-scale Effects of Forest Roads on Black Bear Habitat Selection

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ABSTRACT: The black bear population within the Coeur d'Alene (CDA) River watershed of Northern Idaho is exposed to high forest road densities and moderately high hunting pressure. Hunting regulations allow for the use of bait and dogs in both spring and fall hunting seasons, and dogs are permitted for non-lethal pursuit during the summer pursuit season. To better understand the effects of these pressures, fine scale analysis of habitat selection and movement behavior of these bears is required. Between 1 June 2007, and 25 July 2008, I instrumented 28 adult black bears with Global Positioning Systems (GPS) collars set to acquire positions at 20 minute intervals. I downloaded data from collars retrieved during the fall of 2007, and winters of 2008 and 2009. I will place these locations into a Geographical Information System (GIS) framework, and combined with topographic, vegetation, hydrological, and human use variables. I will estimate resource selection functions (RSFs) of used versus available habitat covariates at the 2nd and 3rd order of habitat selection using logistic and paired logistic regression. From these RSFs, general habitat selection as well as spatial and temporal relationships between bear locations and roads will be established.

KEY WORDS behavior, black bear, Coeur d'Alene Mountains, effects, GIS, GPS, habitat selection, Idaho, resource election functions, *Ursus americanus*

INTRODUCTION

As human populations continue to expand numerically and geographically, evaluating the effects of humans on wildlife habitats is of growing interest to both wildlife biologists and managers. One major aspect of human expansion is the immense network of roads that cover the United States, the volume of traffic on these roads, and the areas and natural resources they access. Whether from direct mortalities, access to hunters and poachers, habitat loss, fragmentation, or alteration, roads have the potential to affect large numbers of species and vast amounts of habitat (e.g., Forman and Alexander 1998, Trombulak and Frissell 1999). For wide ranging "landscape species" that rely on access to a spatially heterogeneous matrix of habitat features as opposed to a single habitat type, assessing the effects of these factors can be complex (Schoen 1990, Gaines et al. 2005).

Black bears (*Ursus americanus*), are a highly adaptable landscape species with few natural predators. With high road densities and increasing traffic volumes throughout much of their range however, the potential effects of roads on bears as well as the habitat which they rely upon are of growing concern (IUCN Bear Specialist Group 2007). The Panhandle Region of northern Idaho is just such an area, where a relatively large

population of black bears is potentially affected by varying levels of road densities and traffic volumes. Because of these factors, understanding how road densities and traffic volumes affect black bears and the landscapes they rely upon plays an important role in management of the species. My research will evaluate these effects by assessing whether or not as the relative importance of primary resources and habitat types shift throughout the year, their availability is affected by the presence of roads, and if these effects change with varying traffic volumes. For my research I will use logistic and paired logistic regression to estimate resource selection functions and assess 2nd and 3rd order habitat selection of adult black bears. In addition I will employ a method of assessing functional habitat loss similar to Nielson et al. (2005) to ask the following questions: 1) Do forest roads have an effect on 2nd order habitat selection of black bears, and do these effects vary by sex? 2) Do forest roads have an effect on 3rd order habitat selection of black bears, and do these effects vary by season and by sex? 3) If forest roads affect 2nd or 3rd order habitat selection, do they result in a functional loss of habitat, and to what extent? 4) Do activity patterns of black bears change in response to seasonal variations in traffic volumes?

BACKGROUND

Roads

With over 6.2 million km of public roads in the United States, roads and roadsides cover approximately 1% of the country's land area (Forman and Alexander 1998, Forman 2000). At low densities, closed roads and roads with low traffic volumes are potentially beneficial to wildlife as travel corridors (e.g., Thurber et al. 1994) and because food resources may occur at high densities along them (e.g., Unsworth et al. 1989, Reynolds-Hogland and Mitchell 2007). As traffic volumes and road densities increase however, the threats of both direct (e.g., hunting, poaching, road kill) and indirect (e.g., habitat degradation and fragmentation) hazards also increase. Considerable attention has been given to several effects of such a vast road network, e.g. direct mortalities or habitat fragmentation due to actual road corridors. The ecological effects of roads, however, may reach far beyond the actual roads and roadsides. Effects such as increased human access, sediment loading, the spread of invasive and noxious species, and air, noise, and water pollution, all reach far beyond the actual roadbeds and road corridors (Forman and Alexander 1998, Trombulak and Frissell 1999, Forman and Deblinger 2000, Jackson 2000). When increases in traffic volume and road density coincide, the potential negative effects are exacerbated, and the area adversely affected by roads, or the "road-effect zone", increases (Forman and Alexander 1998, Dyer et al. 2001). When laid out over the 6.2 million km of public roads in this country, this road-effect zone covers an estimated 22% of the contiguous United States (Forman 2000). With such a large area impacted by roads, the ecological effects of these roads on wildlife and their associated habitats are of considerable interest to wildlife managers and biologists alike (Forman and Alexander 1998).

Within this "road-effect zone", habitat may be affected to varying degrees, which may elicit varying behavioral responses from animals. If the alterations to habitat are severe enough, the characteristics that qualify an area as suitable habitat, or even the habitat

itself, may cease to exist. The result of such alterations is actual physical habitat loss. Even if habitat is not physically lost however, the risks associated with an area may increase resulting in alterations to the behavior of a given species within that habitat. These alterations could include: animals using or crossing roads at different times of day or at different times of the year (Kilgo et al. 1998, Glueck et al. 1988), use of roads by different age and sex classes than would otherwise be expected (McLellan and Shackleton 1988), or changes in home range size, shape, and location (Trombulak and Frissell 2000). If the risk associated with a given area becomes too great, the areas adjacent to these roads may see decreased use, or be avoided all together (e.g., Fecske et al. 2002, Wielgus et al. 2002). In these cases, while habitat may not be physically lost, whatever function that habitat may have held for the animal is removed, therefore equating to functional habitat loss. As human populations and the associated road networks continue to expand, increasing amounts of habitat are lost, both physically and functionally. As habitat loss increases, so too does the importance of understanding the mechanisms and extent of these losses.

With an estimated 1 million vertebrates killed on roads each day in the U.S., direct mortality is perhaps the most noticeable of these road effects (Forman and Alexander 1998). Beyond these direct vehicle related mortalities however, overall mortality has been shown to increase with close proximity to roads for many species. In the Coeur d'Alene Mountains of north-central Idaho, Hayes et al. (2002) concluded that elk (*Cervus elaphus*) harvest correlated directly with the density of both open and closed roads. Within the Greater Yellowstone Ecosystem, Craighead (unpublished report) estimated that over 70% of grizzly bear (*U. arctos*) mortalities occurred within 3 kilometers of primary, and 1.5 kilometers of secondary roads, while Johnson et al. (2004) found a direct correlation between grizzly bear mortality and road density. For black bears, Manville (1983) found hunting pressure and harvests rates to be heavier in areas of higher road densities. In addition to, if not in response to, the increased mortality risk associated with roads, many species show avoidance of roads and surrounding areas. While this avoidance may not represent actual physical loss of habitat, if the avoidance is strong enough, the habitat may be effectively lost. If this effective or functional habitat loss includes areas of productive foraging, necessary cover, or other vital resources, the results may be demographically detrimental (Lyon 1979, Dyer et al. 2001, Johnson et al. 2004, Seip et al. 2006, Hebblewhite and Merrill 2008). For a species such as black bears, for which most aspects of reproduction and cub survival are directly linked to protein levels and fat deposition, the loss of productive foraging habitat, physical or functional, can be extremely harmful (Harlow et al. 2002).

Ecology of black bears

Due to the sizeable energy requirements of wintering and birthing in a state of semi-hypothermic fasting, food quality, availability, and body condition are of primary importance to black bears (Harlow et al. 2002). Almost all aspects of black bear reproduction; age at first reproduction, frequency of reproduction, timing of estrous, number of offspring produced, and initial cub survival, are linked to a bear's ability to deposit and store fats and proteins throughout the 6-8 months of the year they are active (e.g., Rogers 1976, Elowe and Dodge 1989, Schoen 1990, Stringham 1990). In addition

to maternal factors, paternity has also been found to be positively correlated to body size and body condition in male black bears (Kovach and Powell 2003).

With a relatively inefficient digestive system, achieving necessary nutrition levels for successful mating, reproduction, and cub rearing requires substantially large volumes of highly productive forage (Schoen 1990). Due to the prominent role of body size and body condition in bear reproduction and survival, much of their attention, and much of their habitat selection is therefore based on finding the most productive foraging habitats available (Schoen 1990). While primary food resources vary throughout their North American range, the majority of black bear research has shown habitat selection to correlate with shifts in food availability (e.g., Unsworth et al. 1989, Schoen 1990, Beecham and Rohlman 1994, Samson and Huot 1998). This focus on productive forage is most apparent during late summer and fall, when bears exhibit hyperphagia. This dramatic increase in feeding, in which caloric intake can go from 5,000-8,000 kilocalories per day during normal summer behavior to 15,000-20,000 kilocalories per day, is therefore vital to reproductive success (Nelson et al. 1983). This peak in foraging has also been shown to include a shift from primarily crepuscular behavior to more diurnal behavior (Garshelis and Pelton 1980, Lariviere et al. 1994). With these temporal and spatial increases in activity comes an increased likelihood of encounters with humans, and therefore increased mortality risk (Schoen 1990). If these encounters with humans occur in areas that contain primary food sources, bears may be forced to choose between increasing their exposure to humans, and losing access to these important resources. With such a high emphasis on food resources, especially during fall months, the loss of habitats containing high quality forage can have pronounced effects. In areas where bears and humans overlap therefore, research and management of the species should include not only assessment of habitat quality, but availability as well. Moreover, availability ought to be defined not simply by what factors make it physically available, but what factors may determine if a given habitat is functionally available as well (Schoen 1990, Lyons et al. 2002).

In addition to the forage requirements of black bears, several other biological factors highlight the need for additional information for species management. Black bears have long life spans (>20 yrs), delayed reproductive age (4-7 years old at first reproduction), and exhibit one of the lowest reproductive rates of any terrestrial animal in North America (2 cubs every 2-4 years per adult female; Eiler et al. 1989, Miller 1990, Beecham and Rohlman 1994, Kasworm and Manley 1994). Monitoring and assessing trends in populations are therefore important aspects of managing such species because populations may decline rapidly, but will recover slowly (Bunnell and Tait 1980, Miller 1990a). Despite an acknowledgement that harvest data alone may not be sufficient for analyzing population trends, management decisions concerning black bears in many states, including Idaho, are often based on these data because of financial and logistic constraints (Bunnell and Tait 1985, Miller 1990a, IDFG 1998) and because there are no “proven” population indices out there. Because of this, information on other aspects of bear biology and ecology is often of interest to wildlife managers. Two such aspects include understanding habitat interactions that are important to black bears, and identifying key threats posed to bears and the resources they rely upon. By assessing the

possibility of functional habitat loss for black bears due to roads, my research will provide valuable information on both of these aspects to regional and local wildlife biologists and managers.

Local black bear management and previous research

The Coeur d'Alene Mountains (CDA; Fig 1) in the central Panhandle of Idaho is an area of high forest road densities, and relatively high traffic volumes. The area receives moderately high hunting pressure throughout spring and fall bear seasons, as well as fall big game seasons (IDFG 1998). In addition, dogs can be used to pursue bears during spring and fall hunting seasons (15 April–31 May, and 30 August–31 October respectively), and the summer “pursuit” season (1 June–31 July, IDFG 1998). Prior research on black bears in the region (Beecham and Rohlman 1994) provided baseline information on habitat use, social organization, breeding habits, and denning behavior. These studies have shown habitat selection to shift from lower elevation forested and riparian areas in the spring, to higher elevation, more open forested and shrubby areas during the summer and fall (Young and Beecham 1986, Unsworth et al. 1989, Beecham and Rohlman 1994). More recently, Lewis (2007) looked at habitat associations of highway crossings by black bears in northern Idaho. In addition to these studies however, the information on multi-scale effects of roads on black bear movement, behavior, and habitat selection provided by this project will contribute to a better understanding of black bears within the region, as well as possible threats to bear populations.

STUDY AREA

My study will take place within the North Fork of the Coeur d'Alene River watershed, in Coeur d'Alene Mountains in north-central Idaho. Elevation within the study area ranges from 750–2000 meters. Dominant tree species include grand fir (*Abies grandis*), Englemann spruce (*Picea engelmannii*), Ponderosa pine (*Pinus ponderosa*), Douglas fir (*Pseudotsuga menziesii*), mountain hemlock (*Tsuga mertensiana*), and lodgepole pine (*P. contorta*). The area is almost entirely of U.S. Forest Service ownership, with the exception of several small private in-holdings along a number of the smaller creek bottoms. No significant burns have occurred within the area since 1910, and the area has not been logged within the past 20 years. Old cut blocks are represented by dense shrub and uniform regeneration growth. Prior to many of the roads being decommissioned or closed over the past 10 years, this had been the most heavily roaded forest in the lower 48 states (J. Hayden, IDFG, pers. com.). During the summer and fall, bears in this area feed primarily on huckleberry (*Vaccinium sp.*), serviceberry (*Amelanchier alnifolia*), blue elderberry (*Sambuca cerulia*), and mountain ash berries (*Sorbus aucuparia*). Average annual rainfall is 66 cm and average annual snowfall of 127 cm.

METHODS

General Methods

Trapping.- I trapped black bears within the study area (Fig 1) from 1 June through August of 2007 and 2008. I used Aldrich foot snares modified for bear safety (Johnson and Pelton 1980). I anesthetized and handled captured bears in accordance with the projects current IACUC animal handling protocol (052-06MMMCWRU). Captured bears were anesthetized using a Telazol/Xylazine combination, administered at 3.15 mg/kg Telazol and 1.83 mg/kg Xylazine. Drugs were administered using a Pneu-Dart 178 pump rifle (Pneu-Dart Inc, Williamsport, PA). Once bears were anesthetized, vital rates including temperature, heart rate, and capillary refill time were monitored continuously. Numbered ear tags were placed in both ears and general morphological measurements were taken. The first premolar (upper or lower) was collected from each bear for cementum aging (Matson's Laboratory, Milltown, MT, USA). Female bears weighing over 35 kg and male bears weighing over 60kg and estimated at over 5 years of age were fit with a Lotek 3300L GPS collar (Lotek, New Market, Ontario, Canada). All GPS collars were fit either with a timed-release mechanism or a cotton spacer which will rot off within two years. During summer 2007, I deployed 15 GPS collars and 3 VHF collars. Of these collared bears, 1 collar was slipped, 1 was returned by a hunter, and 1 was found on a dead bear (cause of mortality unknown). Two additional collars went off the air between October 2007 and March 2008 for undetermined reasons. In February and March of 2008, the dens of the remaining collared bears were located via aerial and ground telemetry, and bears were dug out for collar removal or replacement. I re-instrumented 9 bears during this time. During summer 2008, I deployed 12 additional GPS collars. GPS collars are set to attempt to obtain fixes (GPS point locations) at 20-minute intervals from the date of capture (or 15 April 2008, if a bear was recollared) until 10 November. Those collars were retrieved during winter 2009. Data is stored “on board” each collar until the collar can be retrieved and the data downloaded.

Data acquisition.- Data has been collected from collars retrieved from dened bears as well as dropped collars and hunter returns. As data is collected, I will screen these data for location attempts that did not yield a fix, locations that were obtained either before the bear awoke from anesthesia, or after the collar had been dropped or the bear harvested

Habitat covariates.- To build RSFs, I will compare habitat covariates at used versus available locations (Manly et al. 2002). Habitat covariates will be selected based on previous black bear research within the area (Beecham and Rohlman 1994). The covariates will include slope, aspect, canopy cover, ground cover type (e.g. regrowth, shrub, forbs, grass, bare ground, etc.) and distance to water. I will also build a second set of RSFs that include road density, traffic volume, and distance to roads. To account for the possible combined effects of distance to roads and canopy cover, I will include an interaction term of these 2 covariates. I will obtain elevation data from digital elevation models (DEM; USGS National Elevation Dataset). I will calculate slope and aspect from the DEM using the spatial analysis tool of the Spatial Analyst extension for ArcGIS. I will use the land cover groups specified by the USDA Northern Region Vegetation

Vector Map (VMAP;) for canopy and ground cover (USDA 2006). I will compile road, watershed, and vegetation data from the USDA Forest Service Northern Region Geospatial Library. I will calculate distances to roads and water using the distance tool in the Spatial Analyst extension. I will calculate road density as the number of linear km of roads per square km or area, using the density function in spatial analyst.

I gathered traffic volume data with the use of TrafX road counters (© TRAFx Research Ltd., Canmore, Alberta 2001 – 2007). I placed counters on roads throughout the study area in the fall of 2007, prior to the opening of elk, deer, and bear hunting seasons. They were retrieved at the end of November. I redeployed counters in spring 2008, and left them out throughout the summer and fall of 2008 to calculate average vehicles per day for each road. Because I was not be able to cover every section of every road within the study area, I classified traffic volume as high, medium, and low, based on data retrieved from the counters after the fall of 2008. Using these classes of average daily traffic volume, I created an interaction term between traffic volume and road density. I used the distance tool in spatial analyst to calculate distances to roads of varying traffic volume.

Habitat selection.- To assess the effects of road density and traffic volume on 2nd and 3rd order habitat selection, I will develop habitat selection models based on locations from collars and associated habitat covariates. I will use logistic regression to estimate the exponential approximation of the RSFs developed for the given set of covariates (Boyce et al. 2002, Manly et al. 2002, Johnson et al. 2006). I will include a random intercept to account for individual variation among bears within the population, as well as the unbalanced number of locations among bears (Gillies et al. 2006). In addition, the use of a mixed-effect model will allow for the assessment of functional responses to roads by bears (Gillies et al. 2002).

To assess 2nd order habitat selection (research question #1), I will calculate annual home ranges for each bear using 95% minimum convex polygons (Powell 2000), and will define availability as all habitat within the study area (Fig 1). Because of the large number of locations obtained from 20 minute fix intervals, I will assign random available points throughout the study area equal to the number of actual locations.

To evaluate 3rd order habitat selection (research question #2), I will use matched case-control paired logistic regression (Compton et al. 2002, Whittington et al. 2005). I chose this method because selection will coincide with the temporal scale of my sampling period, every 20 minutes. At this fine scale, not all locations within an animal's home range are equally accessible at a given time. With paired logistic regression, habitat variables at each known location are compared to those at multiple stratified locations, as opposed to randomly selected locations from the entire home range. The locations will be stratified by distance and time, such that each control will be selected within a radius equal to the average distance moved by that bear over a 20-minute interval during that time of day throughout a given season. In this manner, available locations will be more representative of what is actually available to an animal at a given time at this scale.

To assess seasonal variation in habitat selection for research question #2, I will separate the locations of each animal into 3 seasons, based on hunting and pursuit seasons as well

as previously described food associations (Beecham and Rohlman 1984, Unsworth et al. 1989). Season 1 = 1 May–10 July, season 2 = 11 July–31 August, and season 3 = 1 September–31 October. Based on data partitioned into these seasons, I will again use logistic and paired logistic regression to analyze 3rd order habitat selection.

I will compare models at both second and third order scales using Akaike's information criterion (AIC) ranking and AIC weights (Anderson et al. 2000). Because the number of parameters in my models will be large compared to my sample size ($n = 28$), I will use the modified criterion (AICc; Anderson et al. 2000). If any of the parameters are prevalent in all of the top models, I will use model averaging to determine a weighted average model for each subset of data (i.e. by season and by sex; Anderson et al. 2000). I will assess model fit of the averaged model at each scale and from each area using K-fold cross validation (Hosmer and Lemeshow 2000, Boyce et al. 2002). K-fold cross validation has been shown to be a useful form of model validation in use vs. available studies, and those of relatively short duration such as this (Boyce et al. 2000). I will conduct this cross validation at the individual (bear vs. bear within season), sex (male vs. female within season, male vs. female between seasons), and seasonal (same bear between seasons) levels. This multi-scale cross validation will allow me to evaluate overall model fit, as well as further assess individual, sexual, and seasonal variation in habitat selection.

Using the RSF models derived at each scale and for each season, I will test for effects of roads on 3rd habitat selection by season. I will do the same thing for sex, breaking down the seasonal locations by males and females. As before, I will derive RSF models for each sex class of bear, during each season. RSF's for females will be compared against those for females during other seasons, as well as against males within the same season. I will then determine a top RSF model for each grouping (by sex and by season).

For research question #3 (functional habitat loss), I will compare predicted habitat selection without roads, to actual habitat selection with roads. I will create a predictive map of "high quality" bear habitat, based on the top RSF, not including roads. I will then overlay the top RSF including roads, and assess the differences. If there are differences, e.g. areas of predicted high quality habitat are not selected for when roads are included, I will quantify the habitats (prominent covariates and interactions) that make up the difference. From this, I will be able to ascertain which habitat types are being functionally lost, and to what extent, similar to Nielson et al. (2005). This evaluation of functional habitat loss will be conducted at the 2nd and 3rd order scales, by season, and by sex.

For research question #4 (activity patterns), I will partition the day into 3 periods of activity; diurnal (one half hour after sunrise to one half hour before sunset), nocturnal (one half hour after sunset to one half hour before sunrise), or crepuscular (1 hour before and 3 hours after sunrise and sunset). I will calculate average distance between locations for each bear during these time periods as a measure of activity level. I will then partition these activity levels into seasons to assess the primary periods of activity during each

season. In addition to season, these periods of activity will be compared to road density and traffic of volume on an annual basis, as well as by season.

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STUDY IV: Effects of Biological Sources of Variation on Mark-Recapture Estimates for Black Bears Based on Non-Invasive Genetic Sampling

Non-Invasive Genetic Sampling Reveals Black Bear Population Dynamics Driven by Changes in Food Productivity

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ABSTRACT: I conducted research on the demography of a harvested north Idaho black bear (*Ursus americanus*) population to determine the underlying dynamics of changes in population abundance and how much these dynamics were driven by variation in food productivity, and to evaluate how these processes could influence inferences based on mark-recapture analysis. In cooperation with Idaho Department of Fish and Game (Department) and the USDA Forest Service, I used barbwire corrals to collect black bear DNA during 2003-2006 in the Purcell Mountains of Idaho. We analyzed these DNA samples to determine the number of uniquely identified individuals in each year, N_u . I used a combination of both genetic and mark-recapture analyses to evaluate the sources of variation in N_u over the four years and to what extent this variation was driven by changes in productivity of foods on the landscape. Specifically, I investigated variation in allele frequencies and genetic diversity in relation to changes in abundance, and whether variation in vital rates were a function of changing berry productivity in the study area. I found significant variation of allele frequencies over the years and a heterozygote deficiency indicating I sampled ≤ 4 subpopulations within the same area over the four years (a Wahlund Effect). My mark-recapture analyses suggest this pattern was probably in response to landscape changes in berry abundance. My results suggest important variation in population dynamics driven by changes in food productivity, which should be considered when using mark-recapture analyses to monitor population trends for black bears.

INTRODUCTION

Temporal variation in the dynamics and abundance of animal populations is of great interest to conservationists and wildlife managers. Demographics of animal populations vary temporally due to gains or losses through the biological processes responsible for population change: births, immigration, deaths, and emigration. The causes of changes in these vital rates can be due to a variety of abiotic and biotic factors, among them environmental conditions and management actions. Populations are managed under 3 general guidelines: control, conservation, and sustained yield (Caughley 1977). To determine if a population is meeting these management objectives, monitoring is necessary.

Managed populations of harvested and endangered species are monitored over time to determine if a population is meeting management goals, to detect an incipient or undesirable change, and to identify a response to natural perturbations and management

actions (Goldsmith 1991). Inconclusive or ambiguous population monitoring results can have large impacts on effective management of a species, especially if monitoring results fail to detect a change in population abundance or if monitoring results suggest a change in abundance that is not real (Elzinga et al. 2001). Monitoring is often achieved through relative abundance indices and estimates of abundance (Gibbs et al. 1998). A positive linear relationship between an index and actual abundance is often assumed but is generally untested, thus changes in an index may not reflect true changes in abundance (Gibbs 2000). Although indices are not always reliable and lack estimates of precision, they are commonly used because they are relatively easy to obtain and inexpensive to collect (Gibbs 2000). By contrast, obtaining estimates of abundance requires a large expenditure of time and resources; they are generally more reliable than indices, however, because they incorporate a probability of detection to account for unobserved animals (Williams et al. 2002).

Black bears are an important game species of North America (Garshelis 1990) and effective monitoring of black bear populations is necessary to make sound management decisions and ensure persistence (Miller 1990, Garshelis 1993, Pelton 2000, Garshelis and Hristienko 2006). Monitoring bear populations, however, is challenging because bears are difficult to observe due to the dense forest habitat they often occupy, their low densities, and their secretive and solitary behavior (Pelton et al. 1978, Harris 1986, Woods et al. 1999, Rice et al. 2001). Many states, provinces, and territories use abundance indices alone or in conjunction with estimates of abundance (i.e., generated using mark-recapture data) to monitor and manage their populations (Garshelis 1990). Estimating population abundance is one of the most objective methods to monitor bears populations (Garshelis 1990). However, precision of abundance estimates largely depends on the sample size of captured individuals and the total population size, and violation of mark-recapture model assumptions can cause biased estimates (White et al. 1982). Relatively low capture probabilities and violation of critical mark-recapture model assumptions are issues that plague most bear mark-recapture studies (Harris 1986, Miller 1990, Boulanger and McLellan 2001, Boulanger et al. 2004a).

Bias in population estimates caused from violation of mark-recapture model assumptions about equal catchability and population closure (demographic and geographic) can significantly affect inferences on population trend and consequently, effective management. Bears can exhibit variation in capture probabilities, often violating the assumption of equal catchability. Responses of bears to traps can vary depending on the availability of natural food resources, which can change during the period of a mark-recapture study (Harris 1986). Bears that have been trapped may also exhibit a behavioral response due to prior trap history (e.g. a trap happy or trap shy response; Otis et al. 1978). Capture probabilities for bears can also vary depending upon a bear's sex, age, individual behavior and/or other biological attributes (Harris 1986). Demographic closure can be assumed generally for bears if the duration of the study is no longer than 6-10 weeks (Mowat and Strobeck 2000). Because bears have large home ranges that often overlap with study area boundaries, the assumption of geographic closure is difficult to meet (Harris 1986). Specifically, temporary migrations on and off the study area (i.e. violation of the assumption of geographic closure) can affect estimates of

capture probability directly, often causing bias in survival and population estimates (Pollock et al. 1990). The degree of geographic closure violations may vary from year to year for bears depending upon the availability and distribution of food resources, which can make monitoring bears for trend through mark-recapture estimates difficult.

Understanding the dynamics of bear populations is essential to accurately interpret monitoring results; e.g., the composition of a sampled black bear population can be highly transient, particularly if a large proportion of observed bears are dispersers or temporary immigrants. For populations of black bears, vital rates vary strongly with productivity of food. The distribution and abundance of food resources has been found to directly affect growth, survival, reproductive success, and movement rates of black bears (Jonkel and Cowen 1971, Rogers 1976, Rogers 1993). When food resources are scarcely low, survival and reproduction in black bears can be reduced substantially (Jonkel and Cowen 1971, Rogers 1976, Rogers 1993, Beecham and Rohlman 1994); and black bears often respond to wide-spread food scarcities by increasing movement rates and undertaking long-range movements in search of food (Drahos 1951, Garshelis and Pelton 1981, Rogers 1987, Garshelis 1989, Pelton 1989).

Differing behavior of male and female black bears can also influence population dynamics strongly. Females generally exhibit less movement and smaller home ranges than males (Pelton 2000) which can result in increased survival (Bunnell and Tait 1985, Kasworm and Thier 1994). During years of food scarcity, however, the behavior of males and females becomes less disparate, where females increase their movements and are more readily attracted to human food sources (Noyce and Garshelis 1997). Males also have greater natal dispersal distances than females; with subadult males often dispersing long distances to establish their home range and subadult females establishing home ranges within or near their mother's home range (Rogers 1987, Elowe and Dodge 1989, Schwartz and Franzmann 1992, Moyer et al. 2006). These dispersal patterns of black bears can be also influenced and potentially restricted by landscape gradients and anthropogenic factors (Cushman et al. 2006, Schwartz et al. 2006).

I conducted research on the demography of a harvested north Idaho black bear population, to determine the underlying dynamics of changes in population abundance, to determine how much these dynamics were driven by variation in food productivity, and to evaluate how these processes could influence inferences based on mark-recapture analysis. In cooperation with Department and the USDA Forest Service, I used barbwire corrals to collect black bear DNA (Woods et al. 1999) in a mark-recapture framework during 2003-2006 in the Purcell Mountains of Idaho. We analyzed these DNA samples to determine the number of uniquely identified individuals in each year, N_u . I used a combination of both mark-recapture and genetic analyses to evaluate the sources of variation in N_u over the four years and to what extent this variation was driven by changes in productivity of foods on the landscape.

To explore how variation in abundance of important food species for black bears might have contributed to variation in N_u , I used mark-recapture analyses to associate vital rates with measures of productivity of foods important to black bears in northern Idaho. Black

bears in the Pacific Northwest rely heavily on soft mast plant species which have inherently variable abundances and distributions from year to year (Jonkel and Cowan 1971, Lindzey and Meslow 1977, Beecham and Rohlman 1994). Specifically, in northern Idaho, huckleberries are the most important food resource and other berry producing plants, such as buffaloberry, serviceberry, and mountain ash are also important (Beecham and Rohlman 1994). Thus, I evaluated how productivity of each of the four berry species contributed to variation in vital rates. I calculated productivity in different ways: 1) summed productivity over all species, to evaluate contribution of overall berry productivity, 2) productivity of individual species, to account for potentially strong variation in contributions among species, and 3) productivity of species in the summer (mid-July to mid-September; huckleberries, buffaloberry and serviceberry) and fall (mid-September to mid-November; mountain ash) to account for seasonal effects. High values for any of these measures could increase survival and reproduction and reduce emigration and immigration; low values would have the opposite effect.

I used the Pradel model (Pradel 1996) in program MARK (White and Burnham 1999), which offers a technique to estimate and model the vital rates of a population in an ecological and biological context through the use of covariates (Franklin 2001, Boulanger et al. 2004a). The Pradel model estimates apparent survival, which includes both mortality and emigration, and recruitment rate, which include both births and immigration. I hypothesized that apparent survival would decrease during low berry years due to increased mortality and emigration. For recruitment, I hypothesized that a low berry year would decrease recruitment through reduced births or would increase recruitment through increased immigration.

To gain additional insights into the sources variation in N_u , I used genetic analyses to estimate effective population size (N_e ; number of breeding adults; Schwartz et al. 1998), deviations from Hardy-Weinberg (HW) proportions (observed vs. expected heterozygosity; Robertson and Hill 1984), and the genetic structure of the population (i.e., spatial variation in allele frequencies between demes or subpopulations). I estimated N_e and compared estimates to N_u to determine whether or not the variation in abundance was consistent. I calculated deviations of HW proportions, which can provide important insights into the mating system, social behavior, or population genetic structure of the northern Idaho black bear population (Allendorf and Luikart 2006). I evaluated whether spatial genetic structure existed and compared it to N_u to determine if bears from spatially structured subpopulations influenced the patterns of abundance I observed. I then combined both genetic and mark-recapture analyses to fully understand the cause of temporal variation in N_u and how it may influence my estimates of population trend.

STUDY AREA

The study area was located in the Purcell Mountains of the Idaho Panhandle National Forest of Idaho, USA, encompassing approximately 400 km² of forested land and a large river system. Approximately 3 sides of the study area were bordered by state highways. The terrain varied from flat valley bottoms to steep and rugged mountainous slopes, with elevations ranging from approximately 700 m to 2,000 m. Mixed conifer forests of ponderosa pine (*Pinus ponderosa*), lodgepole pine (*P. contorta*), Douglas fir

(*Pseudotsuga menziesii*), western larch (*Larix occidentalis* Nutt.), grand fir (*Abies grandis*), western red cedar (*Thuja plicata*), and western hemlock (*Tsuga heterophylla*) dominated elevations below 1,300 m and Engelmann spruce (*Picea engelmannii*), subalpine fir (*Abies lasiocarpa*) and mountain hemlock (*T. mertensiana*) dominated elevations above 1,300 m. Understory vegetation primarily consisted of thinleaf huckleberry (*Vaccinium membranaceum*), russet buffaloberry (*Shepherdia canadensis*), serviceberry (*Amelanchier alnifolia*), mountain ash (*Sorbus scopulinus*), pacific ninebark (*Physocarpus capitatus*), and oceanspray (*Holodiscus discolor*).

METHODS

Study Design and Mark-Recapture Sampling

In cooperation with the Department and USDA Forest Service, we collected black bear hair samples in a robust design framework for mark-recapture analyses (Pollock 1982). We used a systematic grid design of hair trap stations to minimize capture variation and to evenly distribute efforts across the study area from 2003-2006 (White et al. 1982). We placed hair trap stations systematically within 2.59 km² cells in years 2003 and 2004. From 2005-2006, we placed hair trap stations in areas subjectively determined to maximize capture probabilities (forested habitats and > 100 m from roads) within larger cells (5.83 km²) than previous years to reduce logistical constraints. The duration of primary sampling periods were approximately 6-12 weeks in all 4 years, conducted during the summer when births were nonexistent and bear mortality was low. Within each primary sampling period was ≥ 1 secondary sampling period(s) (i.e. trapping sessions) that were each ≤ 14 days in length. The length of trapping sessions was conducive to assume demographic closure within primary sampling periods and prevent DNA degradation of hair samples from weather exposure.

Hair trap stations consisted of a single strand of 4-pronged barbed wire wrapped around ≥ 3 trees about knee height (45-50 cm; Woods et al. 1999). We placed a pile of decayed wood in the center of the corral where we applied a mixture of liquefied fish, cow blood, and glycerine as a long-distance lure, but that did not give investigating bears a food reward. Every 14 days, we examined each hair trap thoroughly for hairs on each barb and on the ground (when hair fell off barbed wire). We considered hairs collected from each barb a single sample and placed it in a centrifuge tube or coin envelope with approximately 10 ml of silica desiccant. After hair collection, we burned each barb that had snared hair to prevent future DNA contamination.

Individual Genetic Analysis

The Rocky Mountain Research Station (RMRS) Wildlife Genetics Laboratory analyzed hair samples we collected. They analyzed black bear samples at 9 microsatellite markers: *G1A*, *G10D*, *G10B* (Paetkau and Strobeck 1994), *G10H*, *G10J*, *G10M*, *G10X*, *UarMu59* (Paetkau et al. 1998) and *Msut-2* (Kitahara et al. 2000). They identified species, individual identity, and gender in each sample with sufficient DNA. Laboratory methods and error-checking protocol followed the procedures found in Schwartz et al. (2006).

Mark-Recapture Analysis

I modeled the dynamics of the black bear population through a robust design mark-recapture analysis in Program MARK (White and Burnham 1999) to determine the variation in vital rates that were driven by changes in food productivity. I used the Pradel model (Pradel 1996) with Huggins closed capture (Huggins 1989, 1991) to estimate and model apparent survival (ϕ ; probability of survival from time i to $i+1$ and the probability of remaining in the study area between time i to $i+1$), recruitment rate (f ; number of individuals entering the population between time i to $i+1$ per individual present at time i), capture probability (p), and recapture probability (c) using covariates of food production (Franklin 2001, Boulanger et al. 2004b). Estimates of apparent survival include both mortality and emigration and estimates of recruitment rate include both births and immigration. Estimates of capture probability and recapture probability pertain to the exact sampling period and estimates of apparent survival and recruitment rate correspond to the interval before the given sampling period. I also used the Pradel model to derive estimates of population abundance (\hat{N} ; $\hat{N} = \# \text{ unique bears}/p$) each year and realized population growth rate (λ ; $\lambda = \phi_i + f_i$) between years.

I conducted a preliminary analysis to determine if capture and recapture probabilities varied as a function of sex and year. This analysis revealed strong support for variation between sexes, but little support for capture and recapture probabilities varying by year. I was most interested in capture and recapture probabilities varying by year due to varying sample designs and environmental conditions that could have affected capture and recapture probabilities each year. Therefore, I modeled capture probability and recapture probability as nuisance parameters and I used the most biological, parsimonious model of the parameters for analyses of variation in apparent survival and recruitment rates.

The United States Fish and Wildlife Service (USFWS) has been collecting data on the production of huckleberries, buffaloberries, serviceberries, and mountain ash in the Cabinet-Yaak ecosystem since 1989 (W. F. Kasworm, U.S. Fish and Wildlife Service, Unpublished data; Figure 1). The Cabinet-Yaak ecosystem overlaps a portion of the Purcell Mountain study area. Given this proximity; I assume the berry production would be similar. I standardized this berry abundance data and incorporated six different berry covariates into my mark-recapture candidate model set. I included covariates of each of the four different berry species, a summer berry covariate (huckleberry + buffaloberry + serviceberry), and a total berry covariate (huckleberry + buffaloberry + serviceberry + mountain ash). I modeled apparent survival and recruitment rate as a function of the year prior's berry abundance. In addition to berry covariates, I considered both sex and time covariates for apparent survival and recruitment rate model building. I included sex as a covariate due to the likely difference in vital rates between sexes and time was included because of the differences in environmental and unidentified conditions each year that could affect vital rates. To determine the effect of each covariate on apparent survival and recruitment parameters, I evaluated beta estimates and their 95% confidence intervals.

I used Akaike Information Criterion adjusted for small sample sizes (AIC_c) to compare models and to select the most parsimonious model (Burnham and Anderson 1998). I considered models with $\Delta AIC_c < 2$ as being supported by the data; I used these models to generate model-averaged estimates of parameters to account for information contained within all supported models. I tested goodness of fit (GOF) to the Cormack-Jolly-Seber (CJS) live encounter model and estimated overdispersion with Program RELEASE (Burnham et al. 1987) for the recapture portion of the encounter history. I estimated overdispersion using the combined χ^2 values and degrees of freedom (df) from tests 2 and 3 in Program RELEASE by $\hat{c} = \chi^2/df$ (Burnham et al. 1987). I used $QAIC_c$ scores for model selection if over dispersion was detected ($\hat{c} > 1$; Burnham and Anderson 1998).

Population Genetic Analyses

I estimated effective population size (N_e) each year using the linkage disequilibrium method (Program LDN_e; Waples 2006, Waples and Do 2008) to evaluate changes in abundance through changes in allele frequencies and linkage disequilibrium (D), or the non-random association of alleles at different loci. I estimated N_e for each year based on samples sizes unique to each year (N_{e1}) and on equal sample sizes of 50 (N_{e2}). I compared N_{e1} to N_{e2} to evaluate whether variation in estimates of N_e among years were an artifact of varying sample sizes; if $N_{e1} \approx N_{e2}$, I concluded variation in N_{e1} was not a function of sample sizes. I then compared N_e by year to evaluate observable changes in abundance through changes in D.

I estimated genetic variability within the group of individuals sampled each year using Program GENALEX (Peakall and Smouse 2006) to provide information on changes in abundance through changes in genetic diversity. Specifically, I calculated observed heterozygosity (H_o , the proportion of heterozygotes observed in the population), expected heterozygosity (H_e , the proportion of heterozygotes expected under HW equilibrium), and an inbreeding coefficient (F_{is} , a measure of departure from expected HW proportions). I also examined the number of private alleles by year, or alleles only observed in a single year. I then compared each estimate of genetic variability by year and examined differences to determine observable changes in population dynamics.

I used Program STRUCTURE (Pritchard et al. 2000, Falush et al. 2003, 2007) to investigate population substructure by year. This program characterizes populations by differing allele frequencies, while minimizing HW deviations and D (Pritchard et al. 2000). I used the admixture model, where individuals may have mixed ancestry and the correlated allele frequencies option, where allele frequencies in different subpopulations are likely to be similar. This option also allows for improved clustering of closely related populations. I chose to run simulations 50,000 periods before data was collected (burn-in period) and I ran each iteration for 50,000 Markov Chain Monte Carlo (MCMC) repetitions. I ran 10 independent iterations for each population (K) from 1 to 6. The K with the highest log-likelihood was the most supported, with individuals subsequently divided into K populations. When the most supported K was greater than 1, I used an ad hoc method (ΔK) from Evanno et al. (2005) to identify the most likely K due to greater likelihood variances in STRUCTURE as K increases. I examined the estimated proportion of population membership (Q) of each individual and calculated the mean Q

of all individuals for the most supported simulation of K . I calculated the random expectation of Q if membership was equally divided between populations, indicating no population substructure, and then compared the mean Q to the random expectation to determine the presence of population substructure. If mean $Q \approx$ random expectation of Q , then I would conclude no real population substructure exists. I also examined F_{st} values, a measure of allele frequency divergence, among populations identified. Evaluations of Program STRUCTURE have shown it performs well at assigning individuals to populations with low differentiation among populations ($F_{st} = 0.03$), but F_{st} values must be at least 0.05 to attain a population assignment accuracy rate of 97% (Latch et al. 2006). Therefore, population assignment with F_{st} values far below 0.05 will be considered untrustworthy.

RESULTS

Over the 4 years the study area ranged from 367 km² to 453 km² in size; though the general location and relative shape of the study area were largely consistent (Table 1). The location and number of hair trap stations within the study area varied each year, however, only hair trap stations from areas common to all 4 years I analyzed. The total number of capture sessions sampled and total trap days ranged from 1-5 sessions and 1440-5288 days over the 4 years. The number of hair samples used for genetic analyses varied among years due to budgetary constraints (Table 2). A total of 277 (134 females, 140 males, 3 unknown sex) black bears were identified in the Purcell Mountains over the course of 4 years. I did not include bears of unknown sex in the mark-recapture analyses because of sex-specific model considerations. N_u each year ranged from 53-156 bears, with 43-66% more bears identified in 2004 than in other years (Table 2). Specifically, there were 53 bears identified in 2003, 156 bears in 2004, 89 bears in 2005, and 70 bears in 2006 (Fig 1). A large portion (67%) of bears was captured only once in four years, with 54% of those bears captured during 2004. Recapture rates ranged from 0.16-0.43 over the 4 years.

Huckleberry abundance gradually increased from 2003-2006 and the lowest huckleberry abundance in 18 years occurred in 2003 (Fig 2). Buffaloberry abundance generally increased over the 4 years, except for a drop in abundance in 2005. Serviceberry abundance was relatively high in 2003 and relatively low from 2004-2006. Mountain ash abundance gradually increased from 2003-2005 and dropped in 2006. The lowest summer berry abundance (huckleberry + buffaloberry + serviceberry) over the 4 years occurred during 2004 and the highest occurred in 2006. The lowest total berry abundance (huckleberry + buffaloberry + serviceberry + mountain ash) occurred in 2003 and the highest occurred in 2006.

Mark-Recapture Analysis

The goodness-of-fit test did not detect overdispersion of recaptures ($\chi^2 = 59.3$, $df = 67$, $P = 0.74$), so I used AIC_c for model selection. Few berry covariates appeared to influence apparent survival and recruitment rate indicated by the low number of models that were supported by the data ($\Delta AIC_c < 2$; Table 4). AIC_c model selection indicated that apparent survival and recruitment rates were influenced by sex and summer berry abundance.

Models that considered single berry species or total berry abundance were less supported by the data ($\Delta AIC > 2$). The two most supported models showed that variation in summer berry abundance had a large effect on apparent survival ($\beta_1 = 5.84$, 95% C.I. = 1.74-9.94, $\beta_2 = 4.66$, 95% C.I. = 1.08-8.25). Apparent survival increased as summer berry abundance increased. Variation in summer berry abundance appeared to have an effect on recruitment rates (Model 2; $\beta = 1.75$), however, the 95% confidence interval included zero (95% C.I. = -0.509-4.013).

Model-averaged estimates for female apparent survival show that 9% ($\phi = 0.91$, SE = 0.08) of females in 2003 died or emigrated before 2004, 58% ($\phi = 0.42$, SE = 0.06) of females in 2004 died or emigrated before 2005, and 48% ($\phi = 0.52$, SE = 0.06) of females in 2005 died or emigrated before 2006. Model-averaged parameter estimates for male apparent survival show that 17% ($\phi = 0.83$, SE = 0.14) of males in 2003 died or emigrated before 2004, 75% ($\phi = 0.25$, SE = 0.05) of males in 2004 died or emigrated before 2005, and 66% ($\phi = 0.34$, SE = 0.07) of males in 2005 died or emigrated before 2006. Overall, females had higher apparent survival rates than males (Fig 3).

Model-averaged estimates for female recruitment rate show a 17% ($f = 0.17$, SE = 0.08) increase of new females in 2004 per female alive in 2003, a 10% ($f = 0.10$, SE = 0.04) increase of new females in 2005 per female alive in 2004, and a 13% ($f = 0.13$, SE = 0.05) increase of new females in 2006 per female alive in 2005. Model-averaged parameter estimates for male recruitment rate show a 42% ($f = 0.42$, SE = 0.18) increase of new males in 2004 per male alive in 2003, a 25% ($f = 0.25$, SE = 0.06) increase of new males in 2005 per male alive in 2004, and a 32% ($f = 0.32$, SE = 0.08) increase of new males in 2006 per male alive in 2005. Overall, males had higher recruitment rates than females (Fig 4).

Model-averaged estimates of population abundance (\hat{N}) were 149 (SE = 27) females and 132 (SE = 33) males in 2003, 155 (SE = 17) females and 190 (SE = 28) males in 2004, 81 (SE = 9) females and 81 (SE = 13) males in 2005, and 51 (SE = 5) females and 61 (SE = 9) males in 2006 (Fig 1). Model-averaged estimates for realized population growth rate (λ) were 1.08 (SE = 0.098) for females and 1.25 (SE = 0.194) for males from 2003-2004, 0.52 (SE = 0.052) for females and 0.50 (SE = 0.059) for males from 2004-2005, and 0.65 (SE = 0.062) for females and 0.66 (SE = 0.073) for males from 2005-2006.

Population Genetic Analyses

Estimates of N_{e1} for each year with varying sample sizes ranged from 40.7-155.7 bears during 2003-2006 (Table 3; Fig 1). The highest N_{e1} estimate occurred in 2004, with a 61-74% higher estimate than other years. Estimates of N_{e2} (based on equal samples sizes of 50 individuals among years) yielded a range of N_{e2} estimates similar to varying sample sizes (39.2-170.4), with 2004 again exhibiting the highest estimate.

Observed heterozygosity levels were similar in all years except for 2004, with a 20-26% lower observed heterozygosity than other years (Table 3). Expected levels of heterozygosity under HW equilibrium did not vary significantly from year to year (0.761-

0.789). Estimates of F_{is} were comparable in all years except for 2004. Years 2003, 2005, and 2006 did not deviate strongly from HW proportions (F_{is} from -0.004 - 0.009); however, 2004 had stronger deviations from HW proportions (F_{is} 0.212). In 2004 all loci had a significant excess of homozygotes, compared to zero or one loci in other years. The number of private alleles identified each year ranged from 0-3 alleles.

The most likely number of populations (K) sampled each year varied from 1-4 populations (Table 3). In 2003 and 2005 the most likely K was 1 population. The most likely K for 2006 was 2 populations. The mean Q in 2006 ranged from 0.829-0.835 with the random expectation if membership was equally divided between populations at 0.5. F_{st} values among populations in 2006 varied from 0.0011-0.1251 with approximately 55% and 45% of individuals in each population. The most likely K in 2004 was 4 populations. The estimated mean Q in 2004 varied from 0.562-0.729 with the random expectation if membership was equally divided between populations at 0.25. F_{st} values among populations in 2004 varied from 0.0431-0.1159 with approximately 25% of individuals in each of the 4 populations.

DISCUSSION

My mark-recapture and population genetic analyses revealed critical changes in population dynamics of northern Idaho black bears took place during 2003-2006; had these changes in dynamics not been identified, inferring population trends accurately from monitoring data would have been difficult. My analyses showed how variation in abundance of important food species for black bears contributed to variation in the number of uniquely identified individuals, N_u , and the underlying dynamics of changes in N_u . My use of combined mark-recapture and genetic analyses resulted in insights into the demography of the bears I studied that would have been impossible to achieve with either analysis alone.

The 3 different measures of population abundance (N_u , \hat{N} , and N_e) I estimated all exhibited consistent trends. Estimates of \hat{N} followed the same observed relationship as N_u , and patterns of N_e estimates were comparable to the patterns I observed for N_u in all four years, further corroborating a significant change in abundance took place in 2004 compared to other years. Few bears observed in 2004 were captured more than once in that year, and most bears observed in 2004 were captured only once over the 4 years, indicating highly transient bears. Consistent patterns among estimates of N and the lack of variation in capture probabilities among years argue strongly that the patterns I observed were not an artifact of sampling.

Mark-recapture analysis revealed the variation in vital rates of northern Idaho black bears was driven by variation in food productivity during 2003-2006. Specifically, species that fruit primarily during summer (huckleberry, buffaloberry, and serviceberry) best explained variation in vital rates over the 4 years. This is likely because summer berries provide the first abundant food source available to bears after they emerge from their dens, and over the course of a year comprise the majority of nutrition bears in northern Idaho require for self-maintenance, reproduction, and over-winter survival (Jonkel and

Cowan 1971, Beecham and Rohlman 1994). Further, the presence of 3 berry-producing species in the summer likely reduces variability of overall berry productivity; i.e., if 1 species has a poor year, good productivity among the other 2 may compensate, which is not possible in the fall where only 1 berry species fruits. In the event that all 3 summer species have poor production, however, bears are likely to make large movements in search of food resources (Drahos 1951, Garshelis and Pelton 1981, Rogers 1987, Garshelis 1989, Pelton 1989) because they cannot rely on productivity of fall berries alone to ensure over-winter survival.

Apparent survival estimates decreased from 2003-2004 to 2004-2005 and increased from 2004-2005 to 2005-2006; as I hypothesized, summer berry productivity was correspondingly low in 2004, likely causing increased mortality and emigration. Relatively high productivity of summer berries from 2005-2006 likely caused decreased mortality and emigration. The pattern for recruitment was less clear due to uncertainty associated with the estimates (i.e., large standard errors), but the highest estimate occurred from 2003-2004 when productivity of summer berries was lowest. In accordance with my hypothesis, this increase in recruitment was likely due to immigration into the study area, not births. Overall, these patterns and the patterns of \hat{N} and λ correspond to a large increase in bears sampled in 2004 compared to previous and following years. This is further supported by the highest estimate of recruitment occurring in 2003-2004, when the bears observed in 2004 entered the sampled population, and the largest decrease in apparent survival occurring between 2003-2004 and 2004-2005, when the bears observed in 2004 left the sampled population. Given the low reproductive potential and relatively high survival rates of black bears (Jonkel and Cowan 1971, Reynolds et al. 1980, Bunnell and Tait 1985, Kasworm and Thier 1994, Kolenosky 1990), large fluctuations observed in the vital rates were likely driven by variation in immigration and emigration rates, not births and deaths. Predictably for black bears, sex explained variation in the vital rates I observed. Apparent survival for females was higher than for males; I hypothesize this was likely due both to greater survival of females and to stronger fidelity of females to my study area. Bunnell and Tait (1985) showed overall higher survival rates of females than males among 13 black bear populations throughout North America. Numerous black bear studies have observed lower female movements and smaller home ranges compared to males (Jonkel and Cowan 1971, Amstrup and Beecham 1976, Lindzey and Meslow 1977, Reynolds and Beecham 1980, Garshelis and Pelton 1981), indicating stronger site fidelity by females. By contrast, recruitment was higher for males than for females; I hypothesize this was likely due to greater movements and larger home ranges of males compared to females, which increases the probability males will be recruited into the sampled population.

My analyses of HW deviations and genetic substructure revealed population dynamics during 2004 differed from other years. I detected no large deviations from expected HW proportions in 2003, 2005, and 2006. By contrast, observed heterozygosity in 2004 differed strongly from expected HW proportions; the presence of more homozygotes than expected suggested an important change in population dynamics for that year. An excess of homozygotes is typically a function of non-random mating through population

subdivision, (i.e. the presence of multiple subpopulations sampled within a single population, also known as the Wahlund effect; Wahlund 1928, Wright 1931, Cohen 1990, Allendorf and Luikart 2007). Non-random mating through population subdivision produces an excess of homozygotes at all loci in which the subpopulations differ in allele frequency (Cohen 1990, Allendorf and Luikart 2007). I did not detect genetic substructure in 2003 and 2005, suggesting bears from spatially disjunct subpopulations did not influence the patterns of abundance I observed. I detected 2 possible subpopulations in 2006; the low F_{st} value for 2006 (< 0.05), however, indicated evidence for 2 subpopulations was equivocal. I did detect strong evidence for multiple subpopulations sampled in 2004, indicating that the north Idaho black bear population was not panmictic and bears from spatially structured subpopulations influenced the patterns of abundance I observed. The 3 private alleles, (i.e., alleles only observed in a single year) identified in 2004 further supports the likelihood I sampled bears originating from relatively far geographic distances that do not normally mate with bears inhabiting my study area.

My results collectively demonstrate the large variation I observed in N_u and vital rates was attributable to temporary immigration. Evidence for a Wahlund effect (Wahlund 1928) occurring in 2004 that was driven by variation in food productivity is strongly supported by my results. A Wahlund effect occurs when multiple subpopulations, each within HW proportions are inadvertently sampled together. This results in observed deviations of HW proportions in the sampled population, caused by a greater number of homozygotes than expected due to variation in allele frequencies between subpopulations (Sinnock 1975). The presence of a Wahlund effect in 2004 indicates a large number of immigrant bears moved through the study area that year; results of my mark-recapture analysis indicate these movements were in response to low productivity of food. Given the evidence that the northern Idaho black bear population is not panmictic, berry failures that cause bears to make long distance movements and travel through other subpopulations may be important for maintaining genetic diversity of northern Idaho black bears, provided immigrant bears successfully mate and reproduce. If such movements are temporary foraging forays (i.e., an “occasional sally;” Burt 1943) outside of established home ranges, to which bears ultimately return, then genetic exchange among subpopulations due to such movements may be minimal.

My analyses of population genetics indicate the presence of genetically structured subpopulations of black bears in northern Idaho. Ecology of black bears and the habitats they occupy in northern Idaho suggest genetic divergence among geographically proximate subpopulations is likely. Female black bears are natal philopatric and establish home ranges within or near their mother’s home range (Rogers 1987, Elowe and Dodge 1989, Schwartz and Franzmann 1992, Moyer et al. 2006) resulting in low levels of female-mediated gene flow. Male black bears have high dispersal rates resulting in high levels of male-mediated gene flow; however, much cost is associated with dispersal (Rogers 1987, Elowe and Dodge 1989, Schwartz and Franzmann 1992). Cushman et al. (2006) showed gene flow among north Idaho black bears was facilitated by contiguous forest cover at mid-elevations and was inhibited by non-forested land cover; and

Schwartz et al. (2006) determined that a large, agricultural valley in northern Idaho was not a barrier to gene flow, but that it affected the population's genetic structure.

My results have broad implications for inferring population dynamics of black bears, particularly during years of food scarcities. At face value, estimates of abundance for black bears in my study area from 2003-2006 would suggest large fluctuations in population size, uncharacteristic of species like bears with generally slow population growth (Romanovsky 2002). My analyses showed, however, these fluctuations were due to a temporary change in the distribution of bears, not to population growth driven by increased reproduction and survival. Thus, the increase in abundance I observed in my study area in 2004 was real, but its transient nature would make it inappropriate for inferring population trends or making management decisions. Sampling over multiple years allows outliers, like 2004, to be detected and interpreted accordingly. If my sampling was only conducted during 2004, then my estimate of population size would have been misleading but none of the context needed for reaching this conclusion would have been available. In addition, my interpretation of the variation in N_u would have likely been incorrect without understanding the influence of food productivity on movements of black bears and subsequent consequences for estimating their abundance.

My results demonstrate the challenges of monitoring black bears using mark-recapture when critical assumptions are violated. The assumption of geographic closure was severely violated in 2004 due to temporary migrations on and off the study area. Violation of the geographic closure assumption negatively affects estimates of capture probability causing estimates of abundance to be positively biased (Otis et al. 1978); such estimates represent the superpopulation of the sampling grid and surrounding area (Kendall 1999). Violation of closure due to temporary emigration (e.g. resident bears temporarily emigrating in search of food) and one entry, one exit (e.g. transient bears) types of movements, however, cause bias of superpopulation estimates (Kendall 1999). Relatively large differences between my estimates of \hat{N} and N_u for 2004 are likely a result of this bias. My results also confirm that the degree of closure violation when sampling black bears can strongly vary due to food productivity, causing biased estimates of population trend. Methods are available to test and correct for geographic closure (Otis et al. 1978, Stanley and Burnham 1999, Boulanger and McLellan 2001) and to determine the effective sampling area (Wilson and Anderson 1985), however no method is likely robust to the degree of violation I observed due to the long distances bears likely traveled in 2004.

Inferences of population trend based on estimates of \hat{N} , if not clarified by further genetic analysis, would likely conclude a change in the abundance of bears inhabiting my study area from 2003-2006 that was not real, resulting in biased estimates of trend and potentially leading to inappropriate management decisions. Non-invasive genetic sampling not only provides information for traditional mark-recapture analysis, but allows additional information into the demographics of the population to be gained through genetic analyses. This is advantageous for sampling rare or elusive species where detailed information and demographic insights are difficult to acquire using traditional techniques.

Management Implications

This project demonstrates how the distribution of bears can significantly change during years of low food availability. To reduce the risk of sampling largely transient bears, sampling during food scarcities or the year after should be avoided when possible and measures of food productivity should be coupled with abundance data when interpreting monitoring results. It demonstrates the ability of non-invasive genetic sampling, to provide information on both abundance and the underlying dynamics of a population. With the relatively low sample sizes of mark-recapture data achieved with black bears, small differences in the sampling design do not appear to bias results.

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Table 1. Mark-recapture sampling design and effort of black bears in northern Idaho, USA, from 2003-2006.

Year	Area (km ²)	Traps	Capture sessions	Trap days	Trap Density (traps/km ²)
2003	393	80	1	1440	0.20
2004	393	80	3	5288	0.20
2005	453	104	4	4164	0.23
2006	367	46	5	3187	0.12

Table 2. Individual genetic analyses of DNA samples using microsatellite markers and recapture events of black bears in northern Idaho, USA, from 2003-2006.

Year	No. Samples	N _u ^a	Females	Males	Unknown sex	Bears	Recapture events	Recapture rate ^c
						captured > 1 ^b		
2003	352	53	32	20	1			
2004	383	156	80	74	2	21	25	0.16
2005	328	89	50	39	0	27	38	0.43
2006	266	70	36	34	0	18	26	0.37

^a Number of uniquely identified individuals

^b 2003 had only a single capture session (no recaptures)

^c Total number of recapture events divided by N_u

Table 3. Genetic variation parameters, effective population size, number of populations identified of black bears in northern Idaho, USA, from 2003-2006.

Year	# Private alleles ^a	H _o ^b	H _e ^c	F _{is} ^d	N _e ^e	95% N _e CI	K ^f
2003	0	0.754	0.761	0.009	52.5	37-81	1
2004	3	0.622	0.789	0.212	155.7	113-233	4
2005	1	0.785	0.788	0.004	61.1	47-81	1
2006	0	0.780	0.777	-0.004	40.7	32-58	2

^a Number of alleles unique in a single year

^b Observed heterozygosity

^c Expected heterozygosity

^d Inbreeding coefficient [$F_{is} = (H_e - H_o)/H_e$]

^e Effective population size

^f ΔK was used to calculate most likely K (population), when $K > 1$.

Table 4. Model selection results of vital rates influenced by variation in berry productivity based on AIC_c of black bears in northern Idaho, USA, from 2003-2006.

Apparent survival (ϕ)	Recruitment (f)	K	AIC_c	ΔAIC_c	w_i
Sex + Summer Berries	Sex	7	2363.3	0.00	0.315
Sex + Summer Berries	Sex + Summer Berries	8	2363.8	0.51	0.244
Sex + Time	Sex + Time	10	2364.7	1.41	0.156
Sex + Time	Sex	8	2365.1	1.80	0.128
Sex + Serviceberry	Sex	7	2365.4	2.10	0.110
Sex + Serviceberry	Sex + Serviceberry	8	2367.1	3.83	0.046
Sex + Huckleberry	Sex	7	2376.2	12.94	0.000
Sex + Huckleberry	Sex + Huckleberry	8	2378.3	15.00	0.000
Sex + Buffaloberry	Sex + Buffaloberry	8	2380.8	17.56	0.000
Sex + Mountain Ash	Sex	7	2381.3	18.02	0.000
Sex + Buffaloberry	Sex	7	2383.2	19.99	0.000
Sex + Mountain Ash	Sex + Mountain Ash	8	2383.3	20.05	0.000
Sex + Total Berries	Sex	7	2386.6	23.38	0.000
Sex + Total Berries	Sex + Total Berries	8	2388.6	25.34	0.000

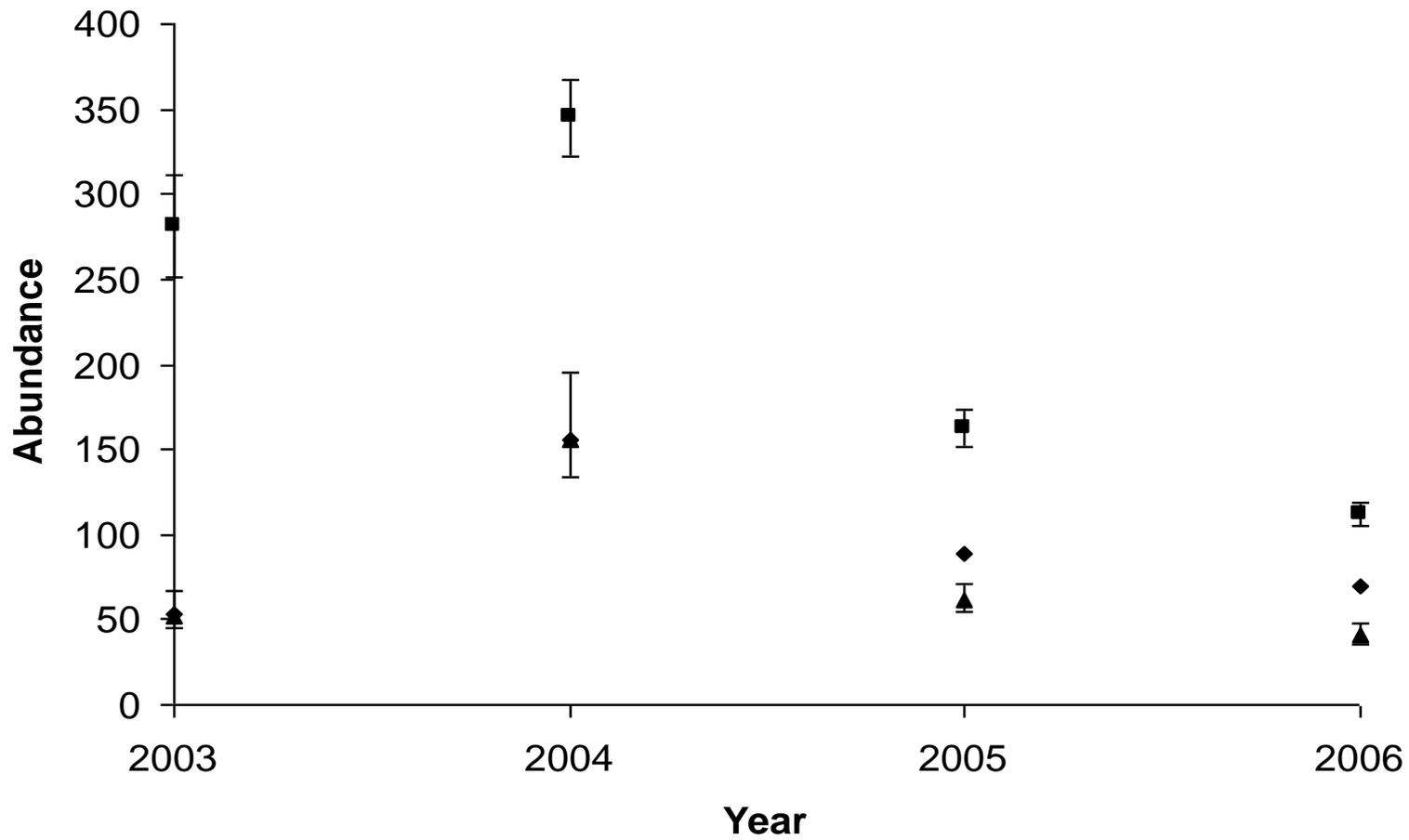


Figure 1. Unique number of individuals (N_u ; \blacklozenge), estimated abundance (\hat{N} ; \blacksquare), and effective population size (N_e ; \blacktriangle) with standard errors for black bears in northern Idaho, USA, from 2003-2006.

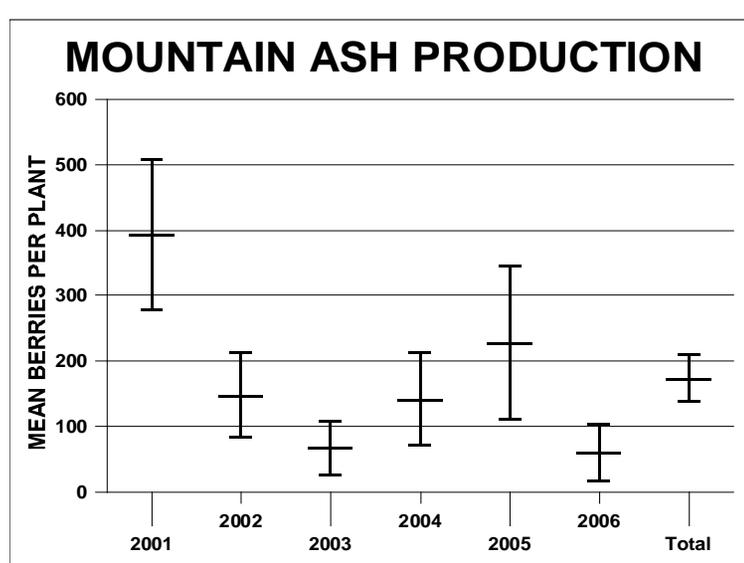
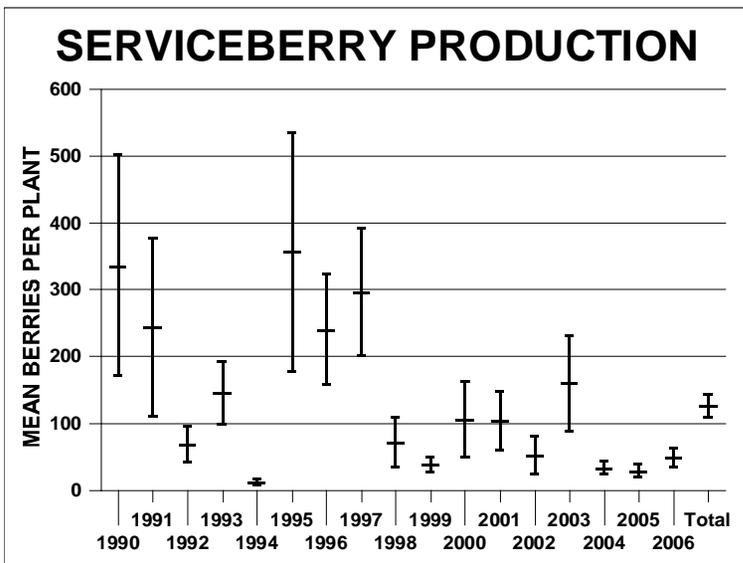
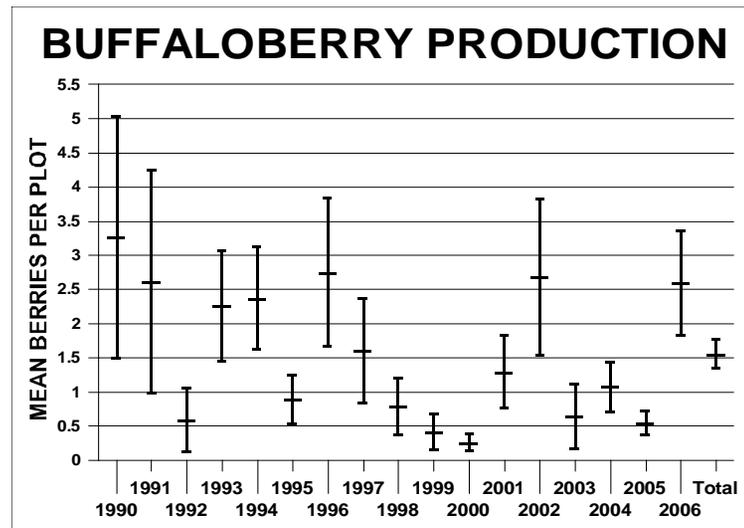
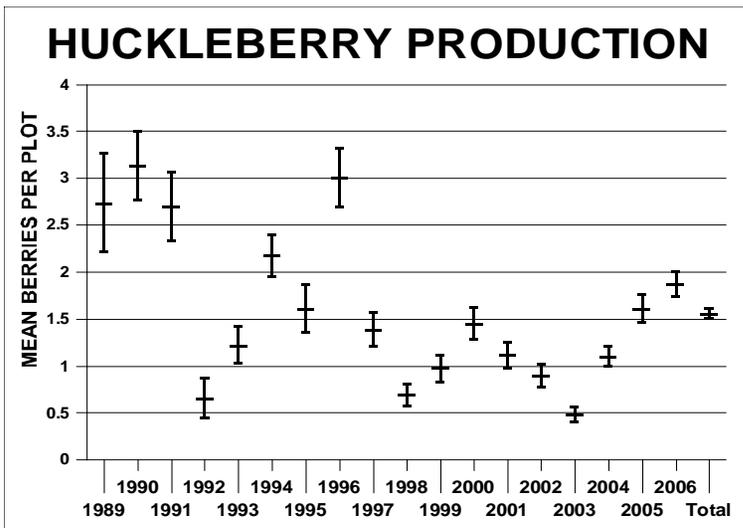


Figure 2. Mean berries per plot and 95% confidence intervals for huckleberries, buffaloberries, serviceberries, and mountain ash in the Cabinet-Yaak Ecosystem, Idaho and Montana, USA, from 1989-2006 (W. F. Kasworm, U.S. Fish and Wildlife Service, Unpublished data).

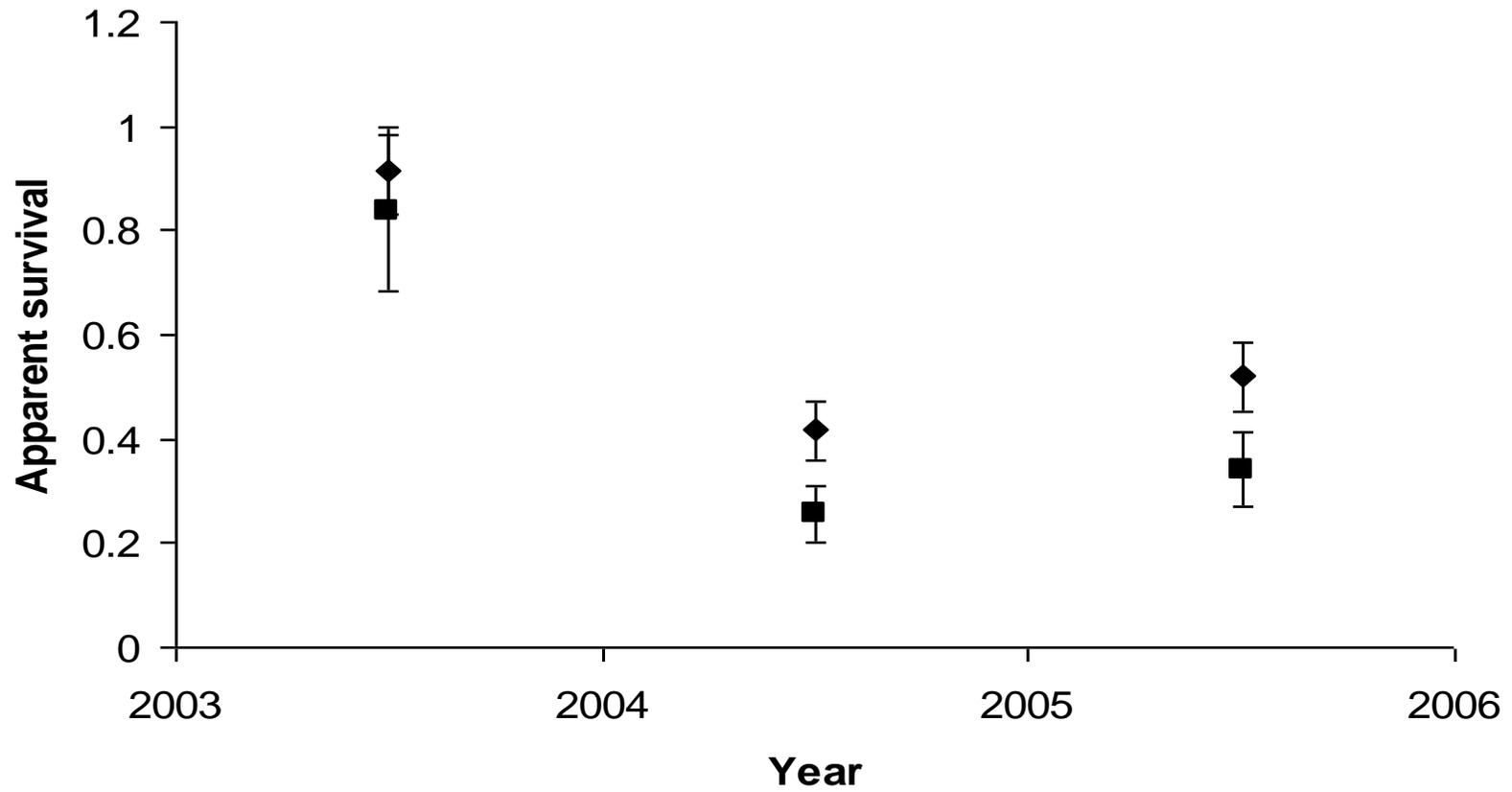


Figure 3. Female (♦) and male (■) model averaged estimates and standard errors for apparent survival of black bears in northern Idaho, USA, from 2003-2006.

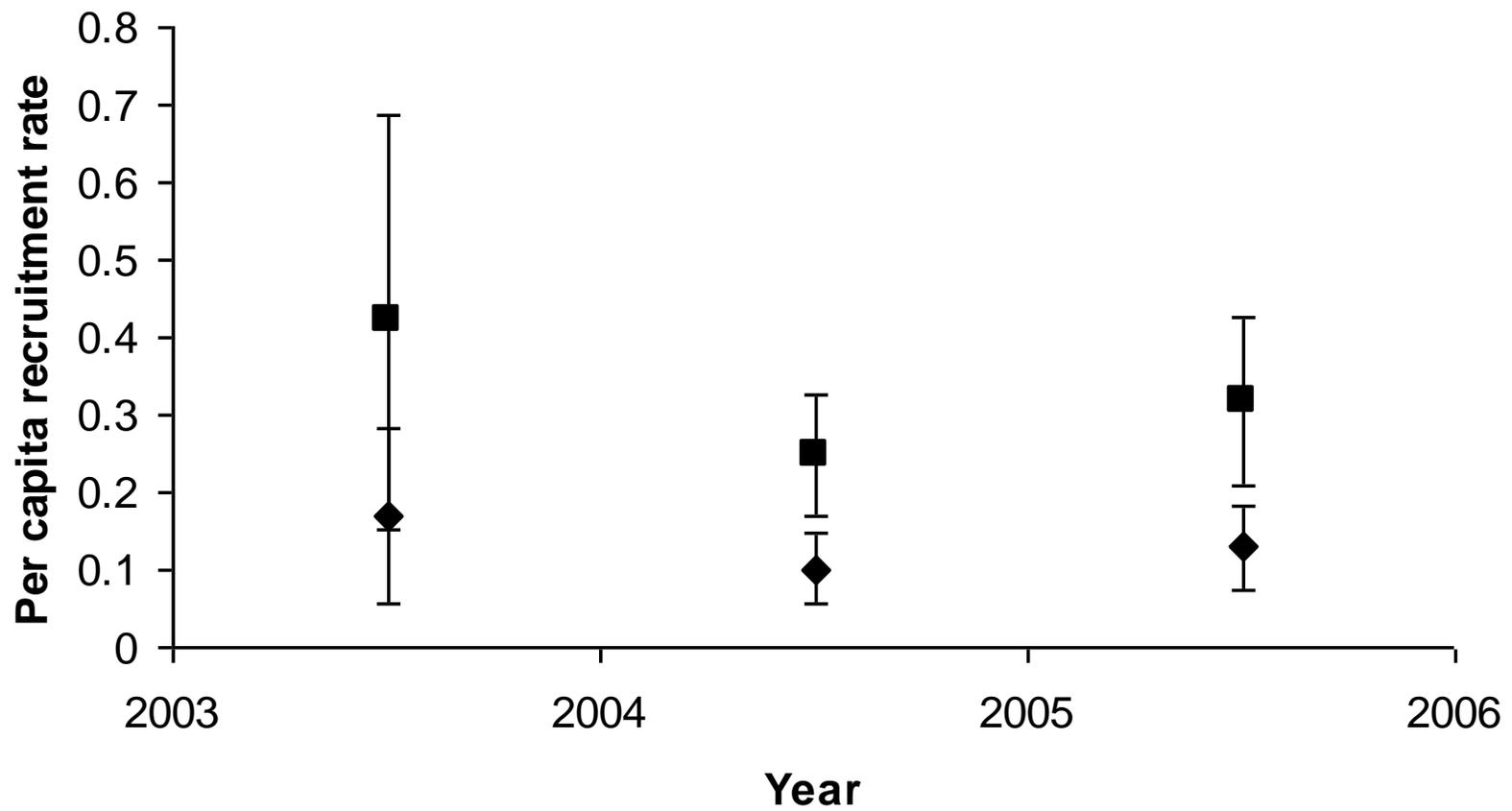


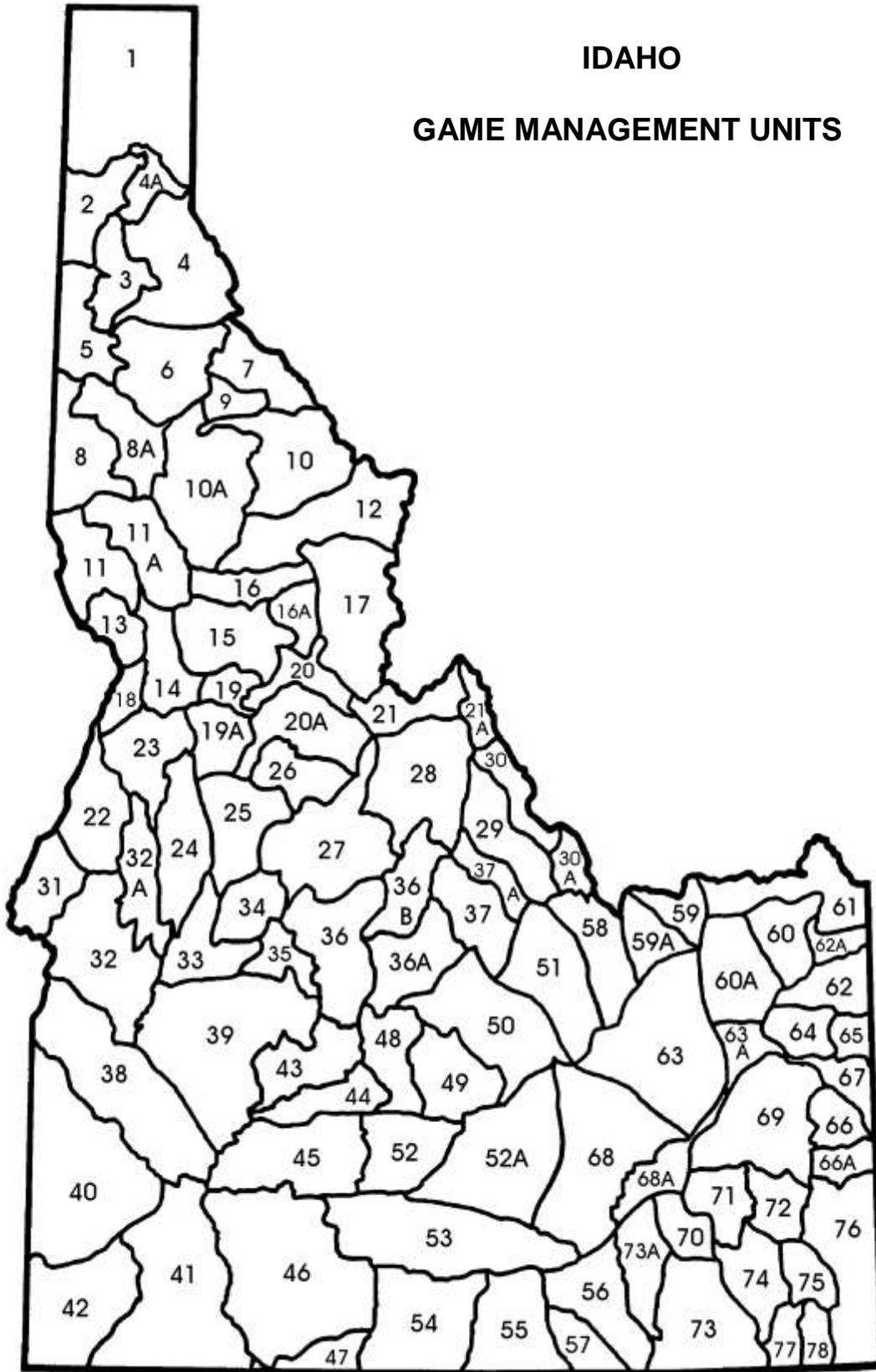
Figure 4. Female (♦) and male (■) model averaged estimates and standard errors for per capita recruitment rate of black bears in northern Idaho, USA, from 2003-2006.

STUDY V: Using Demographic Analyses to Develop Monitoring and Management Tools for Wolves in Idaho

A Ph.D. student was selected and enrolled in classes and began work on a study plan during FY09. After 1 academic year, the student chose to pursue other research opportunities. A national search was conducted to find a replacement. A replacement student was identified and began working on this project on 1 December 2009.

IDAHO

GAME MANAGEMENT UNITS



FEDERAL AID IN WILDLIFE RESTORATION

The Federal Aid in Wildlife Restoration Program consists of funds from a 10% to 11% manufacturer's excise tax collected from the sale of handguns, sporting rifles, shotguns, ammunition, and archery equipment. The Federal Aid program then allots the funds back to states through a formula based on each state's geographic area and the number of paid hunting license holders in the state. The Idaho Department of Fish and Game uses the funds to help restore, conserve, manage, and enhance wild birds and mammals for the public benefit. These funds are also used to educate hunters to develop the skills, knowledge, and attitudes necessary to be responsible, ethical hunters. Seventy-five percent of the funds for this project are from Federal Aid. The other 25% comes from license-generated funds.

