IDAHO DEPARTMENT OF FISH AND GAME

Cal Groen, Director

Project W-160-R-37

Wildlife Research

Progress Report

July 1, 2009 to June 30, 2010

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FEDERAL AID IN WILDLIFE RESTORATION
ANNUAL PROJECT PERFORMANCE REPORT

1. **State:** Idaho

Grant number: W-160-R

Segment number: 37

Grant name: Wildlife Research

Project titles: Statewide Ungulate Ecology
Statewide Bird Ecology

Studies: Statewide Ungulate Ecology

- **I** Survival and population modeling of mule deer
- **II** Effects of wolf predation on elk populations
- **III** Effects of Habitat and Nutrition on Mule Deer and Elk Populations
- **IV** Hells Canyon Bighorn Sheep

Statewide Bird Ecology

- **I** Greater-Sage Grouse (*Centrocercus urophasianus*) habitat and population trends in southern Idaho
- **II** Population characteristics and habitat use of exploited forest grouse populations
- **III** Translocation of mountain quail into historic habitats
- **IV** Pheasant ecology and management

2. **Report Period:** July 1, 2009 to June 30, 2010

Report due date: September 15, 2010

3. Location of work**: Clearwater, McCall, Southwest, Magic Valley, Southeast, Upper Snake, and Salmon regions (statewide)**

4. **Objectives:** Conduct research on mule deer, elk, bighorn sheep, sage grouse, forest grouse, mountain quail, and pheasants; and their habitats to gather biological information to provide valid, scientifically-based information for wildlife managers to make sound wildlife conservation and management decisions.

5. **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 Statewide Ungulate Ecology and Statewide Bird Ecology 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.

6. Describe how the objectives were met.

Studies: Statewide Ungulate Ecology
Study I Survival and population modeling of mule deer

ABSTRACT Survival of fawns during winter is an important component of mule deer population dynamics. Unfortunately, this vital rate is highly variable and difficult to estimate with traditional survey methods. Our objectives were to model winter survival of mule deer fawns across large gradients in weather and predation regimes across southern and central Idaho. We monitored survival of 20 to 34 mule deer fawns in 6 to 10 study areas for a total of 185 to 253 annually during 1998 to 2006. During 2007 to 2009, we monitored survival in 10 Population Management Units for a total sample 268 to 286. Relocation points for all individuals and years in a single study site were combined to estimate seasonal 95% adaptive kernel home ranges (summer and winter) for the study area. Monthly temperature (mean) and precipitation (total) values extracted from spatial PRISM layers were combined to produce climate covariates that represent seasonal changes that affect plant phenology and deer energy expenditure within the study area home range. Vegetation variables were developed from the SAGEMAP and REGAP vegetation layers. To evaluate the influence of vegetation type on predicted survival, we stratified study areas (GMUs) into similar vegetation communities with cluster analysis.

Statewide survival of fawns ranged from 0.32 to 0.71 during 1999 to 2006, with relatively low survival in 2002 and 2006 in most GMUs. Survival for individual GMUs ranged from 0.08 in GMU 72, 2002 to 0.92 in GMU 32, 2005. The best supported model for all study areas from 1999 to 2006 explained 56% of the variance in survival and included growing season precipitation in late summer/fall, early and late winter precipitation, and minimum temperature for late fall, early winter, and late winter. Our ability to predict fawn survival varied among vegetative communities; models explained 73% of the variance in aspen, 57% in conifer study areas, and 91% in shrub-steppe study areas. August-September precipitation increased survival in Aspen and Shrub-Steppe, but had no impact in Conifer, and its influence on survival varied across time periods. October precipitation decreased survival in Aspen, increased in Conifer, and had no effect in Shrub-Steppe. Precipitation in November-December and the January-March/November-December ratio had consistent effects across vegetation communities, decreasing survival primarily during 16 February- 15 April, but perhaps having a weaker impact in Shrub-Steppe. Colder October temperatures generally decreased survival consistently in all vegetation types, with the strongest effect during the 16 December – 15 February survival time period, decreasing thereafter. Colder November-December temps consistently decreased survival during 16 December – 15 February, and continued to do so during 16 February- 15 April in conifer, but not in Aspen or Shrub- steppe. Colder January-March temperatures reduced survival only in Aspen over the entire winter. Data

collected from 2007 to 2009 will be used to validate models for the population management units.

Study II Effects of wolf predation on elk populations

ABSTRACT During FY10, we captured 80 elk including 40 cows, 7 bulls, and 33 calves. Elk were fitted with GPS radio collars and we obtained various measurements and biological samples. We measured rump fat on captured elk either by palpation or a combination of palpation and ultrasonography. Cow elk serum was evaluated for levels of PSPB to determine pregnancy. We extracted a canine tooth from all elk >3.5 years age and teeth were sectioned to determine age. Elk fecal samples were examined to determine the presence of giant liver fluke ova and all were negative. We captured 8 wolves that were fitted with GPS-Argos radio collars. We conducted monitoring flights on a biweekly basis to determine survival status of elk and wolves. We conducted necropsies on 30 elk and 9 wolves. We downloaded location data from GPS collars obtained from dead animals and collars recovered following release. We currently have 21,328 wolf GPS locations, and 103,153 elk GPS locations. We conducted aerial surveys on both study areas to measure the spatial distribution and abundance of elk and alternate ungulate prey. We measured snow depth and sinking depth at 5 sites. We developed a data base in Access to store data from captures, monitoring flights, necropsies, and to store location data from GPS collars.

Study III Effects of Habitat and Nutrition on Mule Deer and Elk Populations

ABSTRACT Data from the AVHRR satellite sensor was used to model the phenology of ecoregions within the state of Idaho. The U. S. Geological Survey Environmental Research Observation and Science Center used these data to generate a Normalized Difference Vegetation Index time series from 1989 through 2007. Metrics were derived from this temporal series that correspond to the start of the growing season, peak of the growing season, end of the growing season, duration of the growing season, peak NDVI value for the growing season, and the time integrated NDVI. These metrics were analyzed for trends within the 10 level 3 ecoregions that fall within Idaho, as described by the Environmental Protection Agency. The non-parametric Mann-Kendall trend test was used to ascertain any trends. Overall, the temporal variables were relatively unchanged with a few exceptions. The end of the growing season extended in the Middle Rockies ecoregion by over one week ($p = 0.093$), the end of the season extended by over 12 days in the Wasatch and Uinta Mountain ecoregion ($p = 0.08$), and the peak of the growing season occurred more than two weeks earlier in the Blue Mountain ecoregion ($p = 0.048$). More widespread trends took place in the peak NDVI and time integrated NDVI metrics. Peak NDVI values increased in the Northern Rocky ecoregion by over 4% ($p < 0.001$), in the Columbia Plateau ecoregion by almost 3% ($p = 0.012$), by over 2% in the Blue Mountain ecoregion ($p = 0.09$), and by over 2% in the Wasatch and Uinta Mountain ecoregion ($p = 0.09$). The time integrated NDVI increased in the Blue Mountain ecoregion by 15% ($p = 0.05$), by 33% in the Northern Rocky ecoregion ($p = 0.02$), by 29% in the Idaho Batholith ecoregion ($p = 0.05$), and in the Wasatch and Uinta Mountain ecoregion by 22% ($p = 0.05$). The peak NDVI value and time integrated NDVI

metrics have been correlated to primary production and above ground biomass which indicates that while the timing of the growing season may not be shifting within the state, but areas are accruing more yearly biomass during the course of the growing season.

Study IV Hells Canyon Bighorn Sheep

ABSTRACT The Hells Canyon Initiative is a state, federal, and private partnership to restore Rocky Mountain bighorn sheep *Ovis canadensis canadensis* in the Hells Canyon area of Oregon, Idaho, and Washington. During 2009-2010, 118 radio-collared bighorn sheep were monitored regularly in 12 populations. Average annual survival of radio-collared ewes was 0.90 and annual ram survival was 0.78. Lamb survival was highly variable among populations. Pneumonia-caused mortality of lambs was confirmed in 5 populations and suspected in 3 others. The overall metapopulation was estimated at 840 bighorn sheep, essentially unchanged from an estimate of 830 sheep in 2008 - 2009.

Multi-year collaborative projects were continued with the University of Idaho Caine Veterinary Teaching Center and the Department of Veterinary Microbiology and Pathology at Washington State University to better understand the clinical causes of disease, in particular *Mycoplasma ovipneumoniae,* in bighorn sheep. An experiment copasturing bighorn and domestic sheep was completed during this period and two manuscripts are in preparation. A new collaboration was also started with Penn State University to analyze the spatial and temporal ecology of disease in the Hells Canyon bighorn sheep metapopulation.

Studies: Statewide Bird Ecology

Study I Greater Sage-grouse (*Centrocercus urophasianus*) habitat and population trends in southern Idaho.

Part I Greater-Sage Grouse (*Centrocercus urophasianus*) lek attendance rates in southern Idaho

ABSTRACT: Lek counts are used to assess population trends of greater sage-grouse (*Centrocercus urophasianus*). Our objective is to estimate the probability of birds attending leks and the probability of detecting attending birds during a lek count in order to relate counts of birds at leks to population abundance. We also attempted to estimate survival of nests and survival of yearling and adult birds in our study population. We used mark-resighting techniques to model the probability of male greater sage-grouse attending leks and counts of grouse from blinds located within 20 m of leks to estimate detectability. Birds were captured in the winters of 2006 – 2009 and fitted with 16.5 g necklace-style radio transmitters. Triangulation from 2 locations off each lek was used to "re-sight" marked birds. We fit a Cormack-Jolly-Seber model to these data using program MARK. We restricted our predictor variables to time (Julian date), year, age of birds (adult or yearling), and their interactions. The top model chosen by AIC model selection procedures included the variables of year, age, and a quadratic time trend. For our final analysis of the complete data set, we will include additional variables such as lek-specific covariates and weather in our candidate set of models, which should result in more precise estimates of attendance probability. Preliminary analysis of our

detectability study indicate that between 79% and 91% of the birds actually attending a lek are included in the counts performed during a lek route. Nest success for our study area ranged from 13.3% and 27.6% during the period from 2007-2009. Further analysis of these data should produce estimates of yearling and adult survival for this population, in addition to identifying variables that affect the probability of individual birds being included in lek route counts, adult and yearling survival, and nests surviving to hatch.

Part II Impacts of elevated infrastructure on Greater Sage-grouse in Idaho: collision, mitigation, and spatial ecology

ABSTRACT We used probability-sampling methods to estimate collision rates of greater sage-grouse (hereafter sage-grouse) with fences in breeding areas across southern Idaho, and conducted a field experiment to test the effectiveness of fence marking as a mitigation method to reduce fence collision rates during the spring of 2010. We randomly selected 1x1 km sampling units within approximately 2.5 km of active leks monitored by Idaho Department of Fish and Game, used one-stage cluster sampling to estimate fence collision rates within each lek route, and used stratified cluster sampling to estimate global collision rates over all study areas. We sampled 63.3 km of fence from March – May 2010, of which approximately 50.3 km was sampled on 2 occasions. During fence collision surveys we located 50 avian collision sites, 36 of which were known sage-grouse. Twelve of these collision sites (8 sage-grouse) were found opportunistically outside of randomly selected sampling areas, and therefore were not used to generate collision rate estimates. Estimated collision rates varied spatially between lek routes (0-1.904 all species; 0-1.428 known sage-grouse), and global collision rates were lower during the second round of sampling (0.162-0.486 all species; 0.099-0.356 known sage-grouse). Experimental fence marking studies were conducted using a complete block repeated measures design, with 8 different sites as the blocking variable and 5 sampling periods from March – May 2010. Pooling data over all sites and sampling rounds resulted in uncorrected collision rates approximately 5.74 times higher in unmarked control than marked treatment fence segments. Repeated sampling showed a peak of number of collisions found in late March and early April, with a decline for the remainder of the sampling periods. Future work will evaluate spatial and temporal variation in fence marking treatment effects using repeated measures mixed effects modeling, and collision risk models will model the influence of site and landscape features on collision rates at multiple spatial scales across all study areas. These analyses will be completed as part of a M.S. thesis in wildlife resources, and will be completed by May 2011.

Part III Factors influencing survival and detectability bias of avian fence collision surveys in sagebrush steppe

ABSTRACT We used hen ring-necked pheasant (*Phasianus colchicus*) carcasses as surrogates for greater sage-grouse (*Centrocercus urophasianus*) to study factors influencing survival and detection bias associated with avian fence collision surveys in southern Idaho, USA, during spring 2009. We randomly placed 50 pheasant carcasses on each of 2 study areas, estimated detection probability during fence-line surveys, and

monitored survival and retention of carcasses and their associated sign over a 31-day period. Survival modeling suggested site and habitat features had little impact on carcass survival, and the constant survival models were most supported by the data. Model averaged carcass daily survival probability was low on both study areas, and ranged from 0.776-0.812. Survival of all carcass sign varied strongly by location, and the top sign survival model included a site effect parameter. Model averaged daily survival probability for collision sign on the 2 study sites ranged from 0.863-0.988, and varied between sites. Logistic regression modeling indicated detection probability of carcasses during fence-line surveys for avian collision victims was influenced by habitat type and microsite shrub height at the carcass location. Carcasses located in big sagebrush (*Artemisia tridentata*) habitats were detected at a lower rate (0.36) than carcasses in little and black sagebrush (*A. arbuscula*, *A. nova*) habitats (0.71). Increasing shrub height at the carcass location from the little sagebrush mean of 16.5 cm to the big sagebrush mean of 36.0 cm reduced the detection probability by approximately 30 percent. Avian fence collision surveys in sagebrush-steppe habitats should be conducted at a maximum of 2-week sampling intervals to reduce the impact of survival bias on collision rate estimates. Two-week sampling intervals may be too long in areas with low carcass and sign survival, therefore survival rates should be estimated on all study areas to determine the appropriate sampling interval duration. Researchers should be aware of the effects of local vegetation on detection probabilities, and methods to correct detection probabilities based on collision site attributes should be applied to ensure more accurate collision rate estimates. Additionally, caution should be used when aggregating or comparing uncorrected collision data from areas with differing vegetation, as detection probabilities are likely different between the sites.

Part IV Use of dwarf sagebrush by nesting greater sage-grouse

ABSTRACT I investigated habitat characteristics for 156 Greater Sage-Grouse (*Centrocercus urophasianus*) nests and 138 random plots among eight populations in southern Idaho, US, during 2003-2005. In addition to traditional habitat measurement methods, I included two visual obstruction variables observed from the nesting female's perspective - effective height and horizontal cover. Independent, uncorrelated variables were obtained from principal components analysis for two sub-categories of habitat; habitat throughout southern Idaho and sites dominated by dwarf species of sagebrush (little sagebrush *Artemisia arbuscula* and black sagebrush *A. nova*). Height of shrubs and shrub canopy cover accounted for the single most variability among southern Idaho habitat and sites dominated by dwarf sagebrush, respectively. Shrub density and horizontal cover were the second and third factors contributing to the variability for both sub-categories. Grouse used sites with greater shrub height and canopy cover than available at random. Successful nests tended to have greater shrub density and somewhat greater horizontal cover than random points. Successful yearlings had the greatest horizontal cover among the age and fate categories of nests. A gradient of shrub heights was observed with dwarf sagebrush being the shortest while mesic sites with mixed shrubs and sites dominated by three-tip sagebrush (*A. tripartita*) being the tallest. Despite lacking height, dwarf sites maintained similar density and horizontal cover as other shrub sites. Logistic regression models were developed using variables obtained

from principal components analysis. Horizontal cover 3 m from the center of the plot, alone, was the best model for separating nests from random plots in southern Idaho habitat and had the best classification rate among dwarf sagebrush habitat. Habitat measurements of dwarf sagebrush dominated nest sites were within the habitat guidelines used for managing Greater Sage-Grouse and is the first time this habitat type has been described. Nests (N = 46) in habitat dominated by dwarf sagebrush had 43% success and nests with dwarf sagebrush species concealing the nest bowl (N = 22) had 50% nest success. Certain grass and shrub species have similar overall heights but considerably different effective concealment cover. Grouse researchers should consider using effective height and horizontal cover habitat measurements in future research of shrub-steppe grouse.

Study II Population characteristics and habitat use of exploited forest grouse populations

ABSTRACT Forest grouse (dusky grouse [*Dendragapus obscurus*], ruffed grouse [*Bonasa umbellus*], and spruce grouse [*Falcipennis canadensis*]), are popular among gamebird hunters in Idaho. Unfortunately, abundance, population trends, and harvest rates are largely unknown in the state. We established 12 random line transects and 2 management roadside surveys to estimate ruffed grouse density during the last 3 years in the Squaw Creek drainage of game management unit 32A. Densities have averaged 5.05 ± 0.41 males/km^2 (mean ± 95% CI) for transects and 4.15 ± 2.60 males/km^2 for roadsides. Twenty-six ruffed grouse males have been captured, radio-collared, and monitored for survival and movements. Adult male home ranges (24 ± 13 ha, n=14) were similar (P = 0.156) to yearling males (62 ± 77 ha, n=5) as were dispersal from drumming logs (AD 123 ± 37 m, JV 208 ± 132 m, P=0.096). Survival of 19 radio-marked males averaged 56.7 ± 0.1%/year with only 1 male being harvested. A hunter check station was operated for 18 days during the 2009 hunting season with 498 hunters checked and 96 forest grouse (69 ruffed grouse, 27 dusky grouse) harvested. Hunters were categorized by primary species hunted; 70% big game, 19% grouse, 5% predator, 2% turkey, and 4% by recreationists scouting for big game or woodcutting. Grouse only hunters accounted for 69% of the forest grouse harvest, 22% by big game, 9% recreationists (big game scouting and firewood cutting), 0% predator, and 0% by turkey hunters. During the first 2 weeks of the hunting season, 69% of the harvest was juvenile birds and 51% of the entire juvenile harvest occurred during this time. A hatch curve was developed with 37% of the hatch occurring during the second week of June. Less than 10% of the radio-marked males have been harvested, possibly due to road closures to motorized vehicles in the study area.

Study III Translocation of mountain quail into historic habitats.

ABSTRACT Our objectives were to augment a founding population of mountain quail released in the Bennett Hills in 2006-07 through subsequent releases of mountain quail in 2008-09 and, assess radio-marked quail survival and reproduction. I attempted to detect the establishment of a self-sustaining population. High mortality rates immediately

following release influenced survival functions. Mountain quail survival rates were generally high during autumn (>85%). Winter survival rates represented the lowest of the year (≤ 40%). Winter stochastic events were the ultimate factor influencing mountain quail survival. The majority of translocated mountain quail (63%) did not move from breeding ranges. We observed average mountain quail productivity despite a seeming shift in mountain quail reproductive strategy. A preponderance of female parental care coupled with seemingly promiscuous behavior by multiple males and females is evidence that care-giving males may be plastic in their parental and mating effort.

Study IV Pheasant ecology and management

Part I Territorial male pheasant density response to habitat changes

ABSTRACT Surveys of pheasants (*Phasianus colchicus*) were conducted during the springs of 1994-1999 on 13, one-mile sections of agricultural land in Gooding County, Idaho. Crop reports and aerial photos are being used to determine cover types within the sections throughout the sampling years. The data has been edited and is now being analyzed.

Part II A web based information brochure for upland game birds of Idaho

ABSTRACT Ring-necked pheasants (*Phasianus colchicus*) as well as other upland game birds are a popular species among Idaho hunters. Research conducted in Idaho has not been adequately provided to the general public. Compilation of the work presented on the Department's webpage will allow interested publics to learn ecology and management. This will be expanded to include all 12 upland game birds in Idaho.

7. 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.

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

Stenglein, J.L., L.P. Waits, D.E. Ausband, **P. Zager**, and C.M. Mack. 2010. An efficient noninvasive genetic sampling approach for high-resolution monitoring of a reintroduced wolf population. Journal of Wildlife Management 74:1050-1058.

Ausband, D.E., M.S. Mitchell, K. Doherty, **P. Zager**, C. Mack, and J. Holyan. 2010. Predicting gray wolf rendezvous site locations to reduce survey effort and enable population monitoring. Journal of Wildlife Management 74:1043-1049.

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Long, R.A., J.G. Kie, R.T. Bowyer, and **M.A. Hurley**. 2009. Resource selection and movements by female mule deer: Effects of reproductive stage. Wildlife Biology 15:288-298.

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Connelly, J. W., S. T. Knick, C. E. Braun, W. L. Baker, E. A. Beever, T. J. Christiansen, K. E. Doherty, E. O. Garton, S. E. Hanser, D. H. Johnson, M. Leu, R. F. Miller, D. E. Naugle, S. J. Oyler-McCance, D. A. Pyke, K. P. Reese, M. A. Schroeder, S. J. Stiver, B. L. Walker, and M. J. Wisdom. *In press.* Conservation of greater sage grouse: a synthesis of current trends and future management. Studies in Avian Biology.

Connelly, J. W., E. T. Rinkes, and C. E. Braun. *In press.* Characteristics of Greater Sage-Grouse habitats: a landscape species at micro and macro-scales. Studies in Avian Biology.

Garton, E. O., **J. W. Connelly**, C. A. Hagen, **A. Moser**, and M. A. Schroeder. *In press.* Greater Sage-Grouse Populations: an icon at risk? Studies in Avian Biology.

Knick, S. T. and **J. W. Connelly**. *In press*. Greater Sage-Grouse and sagebrush: an introduction to the landscape. Studies in Avian Biology.

Lowe, B. S., D. J. Delehanty, and **J. W. Connelly**. 2009. Greater sage-grouse *Centrocercus urophasianus* use of threetip sagebrush relative to big sagebrush in south-central Idaho. Wildlife Biology 15:229-236.

Musil, D. D. *In press*. Use of dwarf sagebrush by nesting greater sage-grouse. Studies in Avian Biology.

Musil, D. D., and **J. W. Connelly**. 2009. Survival and reproduction of pen-reared vs. translocated wild pheasants *Phasianus colchicus*. Wildlife Biology 15:80-88.

Reese, K. P., and **J. W. Connelly**. *In press.* Harvest management for Greater Sage-Grouse: a changing paradigm for game bird management. Studies in Avian Biology.

Shepherd, J. F., **J. W. Connelly,** and K. P. Reese. *In press.* Modeling landscape-scale greater sage-grouse nesting and brood-rearing habitat. Studies in Avian Biology.

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APPENDIX I
Progress Reports for Studies in Statewide Ungulate Ecology

STUDY I: SURVIVAL AND POPULATION MODELING OF MULE DEER

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ABSTRACT Survival of fawns during winter is an important component of mule deer population dynamics. Unfortunately, this vital rate is highly variable and difficult to estimate with traditional survey methods. Our objectives were to model winter survival of mule deer fawns across large gradients in weather and predation regimes across southern and central Idaho. We monitored survival of 20 to 34 mule deer fawns in 6 to 10 study areas for a total of 185 to 253 annually during 1998 to 2006. During 2007 to 2009, we monitored survival in 10 Population Management Units for a total sample 268 to 286. Relocation points for all individuals and years in a single study site were combined to estimate seasonal 95% adaptive kernel home ranges (summer and winter) for the study area. Monthly temperature (mean) and precipitation (total) values extracted from spatial PRISM layers were combined to produce climate covariates that represent seasonal changes that affect plant phenology and deer energy expenditure within the study area home range. Vegetation variables were developed from the SAGEMAP and REGAP vegetation layers. To evaluate the influence of vegetation type on predicted survival, we stratified study areas (GMUs) into similar vegetation communities with cluster analysis.

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KEY WORDS mule deer, *Odocoileus hemionus*, survival, weather

INTRODUCTION

Mule Deer in the intermountain region of western North America exhibit wide population fluctuations creating challenges for wildlife managers. As with most temperate and mountain ungulates, adult females exhibit minimal annual variation in survival, while variation is high in juveniles (Gaillard et al. 2000). This variation in juvenile survival is of sufficient magnitude to drive the annual rate of increase of mule deer populations (Unsworth et al. 1999) and many other temperate ungulates (Festa-Bianchet and Smith 1994, Loison and Langvatn 1998, Gaillard et al. 2000, Raithel et al. 2007). Thus, juvenile survival and recruitment has often become the monitoring priority of wildlife managers attempting to predict changes in ungulate populations (Montana Adaptive Harvest Management, Idaho Mule Deer Management Plan, Lukacs et al 2009).

Mule deer fawns in the intermountain west experience 2 distinct mortality periods before they are recruited as 1-year-olds, generally the first 2 months of life (June-July) and late winter, February-April (Mackie et al. 1998, Bishop et al. 2009, Hurley et al. 2010). Neonate survival (birth to 6 months of age) may be adequately measured via age ratio surveys(during December) when coupled with estimates of adult female age structure and age specific fecundity (Harris et al. 2008). Fawn survival from 6 to 12 months of age (winter) is variable and logistically difficult life stage to measure. Currently, the best method is estimating survival of individually marked fawns. More efficient methods to accurately estimate winter fawn survival are needed for mule deer population monitoring.

Annual variation in precipitation and temperature are common within the intermountain environments and has been linked to population dynamics of mule deer (Peek et al. 2002). Fawn survival differences between environments are likely related to overall habitat condition, predator complex, competing ungulates and density, whereas in year-to-year variation at each site, is related to annual weather conditions. Winter survival of mule deer fawns is a function of

energy assimilation and expenditure, and the balance between them determines mortality (Hobbs 1984, 1987). Fawns accumulate less fat reserves than adults during the summer and fall, making them more susceptible to weather severity, poor quality habitat, predators, harassment, or disease. Thus, fawns experience higher winter mortality and variation in survival than adult deer (Bartmann 1984, White et al. 1987, Pac et al. 1991, White and Bartmann 1998, Unsworth et al. 1999). For example, winter fawn survival ranged between 0 and 0.857 in one study area due to changes in malnutrition mortality in southwestern Idaho (Bishop et al. 2005). Like most other temperate ungulates (Gaillard et al. 2000), mule deer population growth is highly sensitive to adult female survival, although adult female survival tends not to vary in response to the typical range of climatic variation in nature as a result of environmental canalization (Bartman et al. 1992, Unsworth et al. 1999, Hurley et al., Gaillard and Yoccoz 2003). In general, sensitivity of population growth is highest to prime age ungulate survival, followed by senescent female, and finally juvenile survival is lowest (Gaillard et al. 2000). Variance in survival follows the opposite trend, where juveniles experience the widest variance in survival, often in response to climate (Portier et al. 1998, Gaillard et al. 2000, Coulson et al. 2001)

Primary causes of mule deer fawn mortality during winter in western North America are predation and malnutrition (Bartman et al. 1992, Mackie et al. 1998, Unsworth et al. 1999, Bishop et al. 2005). However, understanding the relative contributions of predation mortality and malnutrition during winter to variation in mule deer fawn survival is unknown. Although predation and malnutrition interact with climate (Portier et al. 1998, Colman et al. 2001, Mech et al. 2001) to mediate the predation-malnutrition risk trade-off in ungulates, the effects on mule deer fawn survival are difficult to partition and likely influenced by habitat quality (Kjellander et al. 2004, Pierce et al. 2004, Kauffman et al. 2007, Bishop 2009). The degree to which predation is compensatory or additive to other forms of mortality is also related to environmental factors or population density (Bartman et al. 1992). Weather conditions may influence prey vulnerability through nutrition (Mech 2007, JWM), distribution (Kauffman et al. 2007), or mobility and alternate prey (Patterson and Messier 2000). Predator caused mortality may vary annually, even without measurable changes in predator populations, as a result of different weather conditions (cite). Alternately, predation may decrease variance in a system masking the effects of weather. In Banff National Park, coefficient of variation in elk population size was one third after wolf recolonization compared to pre-wolf (Hebblewhite et al. 2005, Wilmers et al. 2005). These interactions complicate the understanding of climate effects on mule deer survival.

Another difficulty for wildlife managers in predicting mule deer (and other ungulate) population dynamics is integrating monitoring with population modeling for harvest management. Unfortunately, early winter age ratios have been previously shown to be poor predictors of recruitment or winter fawn survival (citations). Therefore, wildlife managers must still rely on known-fate style overwinter survival to make harvest decisions, and use life stage matrix models to estimate population size and rates of increase of ungulates such as mule deer (Idaho Mule deer Management Plan 2007, Montana Adaptive Harvest Plan, Colorado). White and Bartmann (1998) developed a model based on overwinter fawn survival, annual doe survival, and December fawn/doe ratios to calculate a population's annual rate of increase, λ. Accurate parameter estimates, especially winter fawn survival, are obviously critical but often lacking. A second problem, however, is that such overwinter survival data is often collected after harvest regulations have been determined for the following hunting season and it is those yearlings that

often comprise a large percentage of the harvest. Managers submit harvest recommendations for the upcoming year by early January, and final harvest regulations are adopted by the Commission in March. Thus, managers do not have the information (overwinter survival) needed at the time of season setting to make the best management decisions. It would be ideal if managers had some reliable way of predicting overwinter survival based on other drivers of survival, such as climate drivers or habitat (e.g., citations Roe deer, Ibex paper).

Prediction has not been a common use of statistical models of survival in the wildlife or medical literature. Only a few wildlife studies, especially, have examined predictive capacity of survival models. Frair et al. (2007) validated the predictive capacity of survival models by comparing the number of elk mortalities occurring in each ranked decile of predicted hazard for resident and migrant elk based on the logic of k-folds cross validation from habitat models (Boyce et al. 2002). Unfortunately, no studies have attempted to evaluate predictive capacity of climate-based models for mule deer survival.

Therefore, our objectives were to test the predictive capacity of models for winter survival of mule deer fawns across large gradients in climate and predation regimes across southern and central Idaho. Specifically, we hypothesized that if climate-alone drives overwinter mule deer fawn survival, then models of survival based on climate covariates should predict observed survival well. Alternately, if predation or other survival factors differ or interact across habitats, we hypothesized that the habitat-climate interaction will weaken the predictive capacity of climate-only survival models. Because of the difficulty in characterizing predation regimes across large regional scales, if broad habitat types are indicative of differing predation regimes, we would expect the predictive capacity of models to decrease along a predation risk gradient.

STUDY AREA

Our study areas covered a wide climatic, predation and habitat gradient of mule deer range in Idaho. Eight winter ranges were selected as permanent study areas across central and southern Idaho (Table 1, Fig. 1). The McCall study area (44°45', 116°53') is located in Game Management Units (GMUs) 32 and 32A near Weiser, Idaho. The terrain is comprised of steep canyon breaks with several benches and elevations range from 1700-5000 ft. Vegetation is dominated by sagebrush (*Artemisia* spp.) and grassland habitat types with occasional mountain brush sites. Cottonwood (*Populus trichocarpa*), aspen (*Populus tremuloides*) and conifers occur in some drainage bottoms. The Boise study area (43°35', 116°02') is located in GMU 39 on the Boise River Wildlife Management Area to the northwest of Lucky Peak Reservoir. Terrain consists of relatively steep canyon breaks and elevations range from 3100 ft at Lucky Peak Reservoir to 5200 ft near Lucky Peak. The vegetation is comprised of sagebrush-grass habitat types and a portion of the area burned in 1992 and 2000. The Twin Falls study area (42°21', 114°21') is located in GMU 54 in the South Hills. The terrain is comprised of rolling hills dissected by several major drainages with elevations ranging from 4150-5900 ft. Vegetation is composed of sagebrush-grass habitat types with scattered pockets of bitterbrush (*Purshia tridentata*). The Pocatello study area (42°43', 111°43') is located in GMU 72 in the Soda Hills. Terrain consists of rolling hills and open valleys; elevations range from 5800-6800 ft. Vegetation consists of sagebrush-grass habitat types at lower elevations with juniper (*Juniperus* spp.) and mountain mahogany (*Cercocarpus* spp.) habitats above; aspen and douglas-fir

(*Pseudotsuga menziesii*) sites occur on north slopes and high ridges. The Idaho Falls study area (43°40', 111°25') is located in GMU 67 near the South Fork of the Snake River. Terrain is moderate to steep with elevations from 5000-6700 ft. Vegetation is dominated by sagebrush and mountain shrub with some juniper habitat types. The Challis study area (44°26', 114°15') is located in GMU 36B around Centennial Flat. Terrain comprises rolling foothills cut by small drainages with steep mountain slopes above; elevations range from 5200-8300 ft. Vegetation consists of sagebrush-grass habitat types and occasional mountain mahogany with conifer patches at higher elevations. GMUs 56 and 73A are the 2 study areas in the southeast Idaho mule deer research project; refer to Hurley and Unsworth (1999) for study area descriptions. These 2 areas are referred to as the SE Idaho research areas. In addition to the permanent study areas, other sites in various areas have been investigated for one or more winters (Fig. 1).

METHODS

Capture and Survival Monitoring

We monitored 20 to 34 mule deer fawns in each study area for a total of 185 to 253 annually (Table 2). We used helicopters to herd deer into drive nets as our primary capture method (Beasom et al. 1980), but occasionally fawns were also captured with a netgun fired from a helicopter (Barrett et al. 1982, Van Reenen 1982) or clover traps (Clover 1958?). Fawns were physically restrained and blindfolded during processing with an average handling time of < 6 minutes. Two types of radio-collars were used: expandable collars (Telonics model 500 with CB-5 Telonics Inc. 932 E. Impala Avenue, Mesa, AZ) and belt collars (Lotek model LMRT-3, Lotek Wireless Inc., 115 Pony Drive Newmarket, Ontario Canada). Collars weighed 320 - 400 grams, were equipped with mortality sensors and fastened with temporary attachment plates or surgical tubing, allowing the collars to fall off the animals after approximately 8-10 months. We measured fawn mass to the nearest 0.4 kilogram with a calibrated spring scale, chest girth (directly behind shoulders on the exhale), hind foot length (tip of hoof to calcaneous) (Robinette et al. 1973) to assess early winter body size and condition. An Idaho Department of Fish and Game (IDFG) veterinarian or veterinary technician was on site during most captures to assist with sampling and assure animal welfare. Animal capture protocol was approved by the Animal Care and Use Committee, IDFG Wildlife Health Laboratory, Caldwell, Idaho, USA. Fawns were monitored with telemetry for mortality from the ground every 2 days between capture and 15 May. If radio signals could not be detected from the ground within 1 week, animals were located via aircraft. When a mortality signal was detected, the fawn was located and cause of death was determined using a standard protocol (Wade and Bowns).

Defining Seasons and Herd Unit Home Ranges

Weather conditions were summarized by month, with 1 April to 30 September classified as summer, and 1 October to 31 March classified as winter. Time varying covariates for survival models varied monthly from 1 December to 30 April. Relocation points for all individuals and years in a single study site were combined to estimate a 95% adaptive kernel home range (Worton 1989) for the study area. Migratory periods were excluded from the home range estimates, locations between 1 June and 31 October were used for summer, 1 December to 15 April for winter. Climate and habitat information was then summarized by home range of radio collared deer for winter and summer within each study area.

Variable Development

Monthly temperature (mean) and precipitation (total) values were combined to produce climate covariates that represent seasonal changes that affect plant phenology and deer energy expenditure (Table 1). Vegetation type variables were developed from the SAGEMAP vegetation layer created in 2005 by the USGS, Forest and Rangeland Ecosystem Science Center, Snake River Field Station, Boise, ID (Citation). The layer contained 126 land cover classes with a minimum mapping unit (MMU) of approximately 1 acre (0.4 ha). We reclassified the image into18 vegetation classes plus an interspersion index with FRAGSTATS with importance to mule deer ecology. These vegetation classes included; aspen woodland, riparian, other deciduous woodland, juniper woodland, mahogany woodland, coniferous forest, deciduous shrub land, mesic sagebrush shrub land, xeric sagebrush shrub land, wet meadow, mesic grassland, xeric grassland (includes Conservation Reserve Program lands), invasive grassland, agriculture, recent burn, open water, developed, and unavailable. The weighted mean accuracy assessment of the reclassification was 84.4%.

Survival Model Development

We used discrete-time known fates modeling to determine covariate effects on fawn survival and to develop models that can be used to successfully predict fawn survival 16 December to 15 May. The analysis was based on radio telemetry data, assuming a binomial error structure and a log-log link for the proportional hazards model, using Program SURPH (Lady and Skalski 2002, Ver. 2.1). We divided winter into 3 periods depending on observed changes in the hazard function over winter (16 Dec. – 15 Feb, 16 Feb. – 15 Apr., 16 Apr. – 15 May). Covariate effects were parameterized independently for each period and permitted to be annually constant (ie. same values for each period) or time-varying (period specific values). Time –varying variables included minimum temperature and cumulative precipitation for each of the 3 winter periods, all other variables varied annually but were constant between periods. Vegetation type variables were constant across all years of the study.

A candidate set of models was developed considering 3 possible effects of climate on mule deer survival; energy assimilation, energy expenditure, and combinations of both for a total of 20 models. Known fate survival models were ranked using the Akaike Information Criteria (AIC) (Burnham and Anderson 1998). There was essentially no model selection uncertainty, so we did not model average. All interaction terms were added to the top models one at a time and retained in the top model set if AIC increased (Hosmer and Lemeshow 1999).

Methods for combining similar study areas

To evaluate the influence of vegetation type on predicted survival, we pooled study areas (GMUs) into similar vegetation communities. We used summer range as the basis for clusters as these season ranges show the greatest potential variance is nutritional quality and vegetation types. We used a hierarchical weighted means cluster analysis based on summer range vegetation types within summer home ranges used by the wintering herd unit (Fig 3) to identify vegetation community clusters. Defining differences identified in the 3 primary clusters were % conifer (>40%), %aspen (6 to 13%), and % mesic sagebrush (>60%) on summer range. The candidate set of models identified for the entire data set was then parameterized for each of the new groups.

Validation of models

Survival models were constructed with the entire data set and then validated with a procedure similar to K-fold cross validation used in the habitat modeling literature ((Hastie et al. 2001, Boyce et al. 2002, Frair et al. 2007). Year by study area units (n= 64) were randomly assigned to 5 equal parts (folds). SURPH data files were divided by folds, with the remaining 80% of data file used for model training and the 20% fold reserved for validation. This was repeated until each of the folds was paired with the corresponding training data sets. Coefficients of the variables identified by the original model were estimated for each training set and tested for differences from the entire data set coefficient with Z-tests. Survival by period was then estimated for the corresponding 20% of the fold data using the training set data coefficients. We tested for the correlation between observed and predicted survival using Pearson's correlation coefficient.

RESULTS

Observed survival

Statewide survival of fawns ranged from 0.32 to 0.71 during 1999-2006 (Table 2, Fig. 2). Survival was consistently more variable in the eastern portion of the state, GMUs 67 and 72 than in west, GMUs 32 and 39 (Fig. 2). Relatively low survival was observed in 2002 and 2006 in most GMUs, extremes were 0.08 in GMU 72, 2002 and 0.92 in GMU 32, 2005 (Table 2). No difference was observed in male vs. female survival (log-rank $\chi^2 = 0.01$, $P = 0.99$). Male mass was significantly larger than female (mean = 37.1kg vs. 34.2 Kg, $t = -13.1$, $P = 0.00$) and mass was significantly related to survival ($z = -2.5$, P = 0.013). This suggests that increased mass of males improves survival counteracting the typically lower survival of males during winter (Hurley et al. *In Press*). Survival rate varied by period, with most mortality occurring from 16 February to 15 April (P1 = 0.90, P2 = 0.69, P3 = 0.92).

Survival rate appeared synchronized within vegetation communities, potentially by the influence of weather (Fig. 2). We tested for synchrony between survival rates across GMU's using Pearsons correlations within major vegetation community grouping. Survival rates were synchronous within aspen (overall r= 0.87, P-values all < 0.10, n = 6-pairwise GMU correlations), sage-steppe (GMU 54, r= 0.68, p<0.10), but not within coniferous habitats (overall r=0.401. P –values > 0.28, 3 pairwise GMU correlations).

Overall survival model

The best supported model for all study areas included growing season precipitation in late summer/fall, early and late winter precipitation, and minimum temperature for late fall, early winter, and late winter (Table 3). The model explained 56% of the variance (ANODEV) in overwinter fawn survival. The 2nd ranked model had a delta AIC = 4.39, suggesting little model uncertainty in the top model.

Vegetation type survival models

Model selection results differed among vegetative communities. The top model for ASPEN study areas was identical to the best model for all study areas, however, this model explained

73% of the variance in fawn survival in these areas (Table 4). The best supported model for CONIFER study areas included late fall through early winter precipitation and temperature (Table 4). Summer growing season precipitation and late winter precipitation and temperature were not important in these habitats. This Conifer model explained 57% of the variance in overwinter fawn survival. The best supported model for SHRUB-STEPPE areas included only growing season precipitation (April-July and Aug – Sept) and explained 52% of the variance (Table 4), The 2nd ranked model had a delta AIC = 0.22 and similar support from AIC weights (0.367 vs. 0.331), but explained 91% of the variance in fawn survival. This model included late summer precipitation, late fall/early winter precipitation and temperature. Even though this competing model explained more of the variance, it was not selected as the top model because some of the first period survival rates were not estimable, causing inconsistent coefficients for this first period.

Covariate effects

Across all vegetation communities, winter survival increased with higher August-October precipitation, and the effect was apparently strongest from 15 December through 15 February of each year. Higher November-December precipitation and a higher ratio of January-March/November-December decreased fawn survival. As minimum temperature increased in October, it decreased winter survival (Table 5), but in November-March, increasing minimum temperatures (warmer winters) increased winter survival.

The effect of season precipitation and temperature varied between vegetation communities. August-September precipitation increased survival in Aspen and Shrub-Steppe, but had no impact in Conifer (Table 5), and its influence on survival varied across time periods (Table 5). October precipitation decreased survival in Aspen, increased in Conifer, and had no effect in Shrub-Steppe. Precipitation in November-December and the January-March/November-December ratio had consistent effects across vegetation communities, decreasing survival primarily during 16 February- 15 April, but perhaps having a weaker impact in Shrub-Steppe.

Colder October temperatures decreased survival consistently in all vegetation types, but again, with no effect on shrub steppe mule deer, and the effect was strongest on 15 December – 15 February survival, tapering thereafter. Colder November-December temps consistently decreased 15 December – 15 February survival, and continued to do so in 15 February – 15 April in conifer, but not in Aspen or Shrub steppe. Colder January - March temperatures reduced survival only in Aspen over the entire winter. The only regional effect of previous April - July precipitation was to decrease survival of shrub-steppe mule deer.

Model validation and prediction

Overall, our ability to predict overwinter survival based on climate covariates across Idaho was marginal (R^2= 0.52, fig 4). Partitioning by habitat, however, improved our ability to predict survival in some vegetation communities, but not others (Fig. 5a, b, c). Shrub steppe predicted the best, and had the narrowest and highest range of observed survival rates (R^2 = 0.91), followed by Aspen (R^2 = 0.73), and then conifer (R^2 = 0.57).

Discussion

Predicting population demographics in mammalian ecology is difficult. Survival of life stages may vary in direct response to environmental stochasticity and predation. Juvenile ungulates in temperate environments are particularly vulnerable to annual variations in climate (Gaillard et al. 2000) and variable survival in this life stage is often the primary driver of population rates of increase. Unfortunately, predicting survival of juvenile ungulates requires extensive expenditures of limited resources.

The issue is exacerbated when attempting to predict survival of juvenile mule deer in the intermountain region of the Western United States. Winter fawn survival is the most variable life stage in this region, both temporally and spatially. Bishop et al. (2005) observed winter survival of fawns was not synchronous even in adjacent winter ranges and cautioned that extrapolating survival to other populations can lead to erroneous population estimates. This may be explained by site-specific effects of weather that influenced forage quality, energy expenditure, and predation risk.

Forage quality for ungulates varies by habitat and is often altered by precipitation and temperature during the growing season (Marshal et al. 2005, Stewart et al. 2005). Our results of increased survival with late summer-fall precipitation, the period of lowest rainfall in the arid intermountain west, highlights the importance of quality forage available to adults and fawns during and post weaning. Increased rainfall will promote growth of forbs (Marshal et al. 2005), a highly selected forage for mule deer (Wallmo et al. 1977,(Hobbs et al. 1983). Increased summer-fall nutrition improved calf and adult female survival, fecundity rates, and age of first reproduction in elk (Cook et al. 2004). Growing season rainfall also increases quality and quantity of winter forage (cite) which has been shown to increase survival of fawns and adult female mule deer (Bishop et al. 2009). The effect of October precipitation on survival varied between the vegetation communities. This is likely related to forage value and distribution of cheat grass (Bromus tectorum), an invasive grass species. The fall germination and growth of this grass provides valuable late fall and winter forage in snow free areas (Bishop 2001). This is often observed in the conifer communities and associated winter ranges of western Idaho, but not in the eastern aspen communities. The forage production and quality of shrub-steppe communities is highly influenced by growing season precipitation, more so than communities with over story vegetative cover. This supports our observation that mule deer fawn survival can be predicted from growing season precipitation alone.

The interaction of weather conditions and habitat selection may influence fawn survival through predation risk. During severe winters, mule deer will concentrate in habitats that minimize energy expenditure. This behavior may move deer into areas of increased predator density or effectiveness. Hebblewhite (2005) observed increased predation rates of elk by wolves with increasing snow depth. Predation risk has been shown to vary by habitat in a wolf-elk system (Hebblewhite et al. 2005), a wolf-caribou system (McLoughlin et al. 2005) and a Red-backed vole-martin system (Andruskiw et al. 2008). African lions hunted in habitats which afforded high probability of capture success rather than prey density (Hopcraft et al. 2005). Alternately, movement closer to human habitation may decrease predation risk (Hebblewhite et al. 2005).

MANAGEMENT IMPLICATIONS

We have developed models to predict mule deer fawn survival from December 16 to May 15 in Idaho's semi-arid intermountain habitats. The models exclude information related to cause specific mortality of deer because this information is not commonly available to all managers; instead we consider vegetation community groupings to determine the predator-caused mortality rate. Estimates of winter fawn survival are critical for successful population modeling and harvest management of mule deer. Once these models are validated or modified for different vegetations communities, managers will realize extensive cost savings if winter fawn survival is now included in their monitoring program (radiocollared fawns) or increased accuracy if fawn survival is not. We invite mule deer managers and researchers in similar environments to evaluate performance of our models against their current monitoring program as further validation.

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Table 1. Study area climate variation, all in one table.

Climate Variable	Description	Value Range Min.	Value Range Max.	Expected effect
Total precipitation (mm)				
POM	October - March (inclusive) preceding birth	98.8	505.5	Doe Condition - Neonate Health
PJM	January - March preceding birth	34.5	255	Doe Condition - Neonate Health
monthly	Monthly, December through April time varying			Fawn Energy Expenditure
aprilsept	April - September of birth year	106.9	418.5	Fawn and Doe Nutrition
apriljuly	April - July of birth year	81.4	340.8	Doe Nutrition
augsept	August - September following birth	12.3	111.7	Fawn Nutrition
octmar	October - March following birth	106.1	745.7	Fawn Energy Expenditure
Oct	October following birth	3.3	91.8	Fawn Energy Expend - Nutrition
novdec	November to December following birth	21	354.1	Fawn Energy Expenditure
janmar	January to March following birth	34.5	362.3	Fawn Energy Expenditure
Mean minimum temperature (C)				
TPOM	Previous October-March	-9.4	-1.9	Doe Condition - Neonate Health
TPJM	previous January-march	-11.8	-0.69	Doe Condition - Neonate Health
tmonthly	Monthly, December - April time varying			Fawn Energy Expenditure
toctmar	October-March	-9.4	-1.9	Fawn Energy Expenditure
Toct	October	-3.7	4.72	Fawn Energy Expend - Nutrition
tnovdec	November-December	-11.83	-2.45	Fawn Energy Expenditure
tjanmar	January-March	-12.1	-0.69	Fawn Energy Expenditure

Table 2. Winter survival rates and standard errors (SE) of radio-collared mule deer fawns in central and south Idaho.

Group	1999	2000	2001	2002	2003	2004	2005	2006
Sex								
Females	0.66 (0.056)	0.50 (0.124)	0.70 (0.048)	0.44 (0.047)	0.70 (0.057)	0.53 (0.052)	0.65 (0.077)	0.30 (0.043)
Males	0.67 (0.053)	0.39 (0.145)	0.73 (0.048)	0.42 (0.052)	0.68 (0.051)	0.54 (0.049)	0.67 (0.098)	0.34 (0.048)
Overall	0.67 (0.038)	0.45 (0.095)	0.71 (0.034)	0.43 (0.035)	0.68 (0.039)	0.53 (0.036)	0.65 (0.061)	0.32 (0.032)
GMU Specific								
32	--	--	0.53 (0.114)	0.68 (0.093)	0.64 (0.096)	0.32 (0.093)	0.92 (0.056)	0.44 (0.104)
39	0.88 (0.083)	0.85 (0.080)	0.90 (0.067)	0.48 (0.100)	0.57 (0.098)	0.38 (0.099)	0.79 (0.083)	0.55 (0.101)
33	--	--	--	--	--	--	0.92 (0.054)	0.04 (0.042)
54	0.86 (0.074)	0.59 (0.129)	0.63 (0.099)	0.59 (0.105)	0.80 (0.080)	0.85 (0.071)	0.73 (0.087)	0.49 (0.105)
56	0.64 (0.096)	0.47 (0.094)	0.72 (0.084)	0.30 (0.085)	--	--	--	--
73A	0.73 (0.078)	0.83 (0.070)	0.75 (0.088)	0.28 (0.083)	0.89(0.105)	0.48 (0.100)	--	--
72	0.65 (0.099)	0.74 (0.099)	0.75 (0.088)	0.08 (0.056)	0.76 (0.085)	0.53 (0.106)	0.60 (0.098)	0.10 (0.064)
76	--	--	--	--	--	--	--	0.62 (0.098)
67	0.62 (0.115)	0.62 (0.107)	0.74 (0.101)	0.36 (0.096)	0.92 (0.054)	0.54 (0.098)	0.68 (0.093)	0.16 (0.073)
60A	--	--	--	--	--	0.84 (0.073)	--	--
69	--	--	--	--	0.73 (0.087)	--	--	0.24 (0.085)
58	--	--	0.81 (0.098)	--	--	--	0.80 (0.089)	--
36B	0.36 (0.105)	0.36 (0.106)	0.77 (0.083)	0.77 (0.083)	0.39 (0.102)	0.33 (0.096)	0.62 (0.106)	0.23 (0.089)

- [a] Survival rates and SEs were calculated following Pollock et al. (1989).

Table 3. Candidate set of models to estimate overwinter survival of mule deer fawns for all study areas combined. The set of models displayed here was used for all modeling.

Model	Parameters	Δ AIC$_c$	Weight	Deviance
Full	1 6			2443.42
Annual: au sept +oct+toct+ratio jm to nd+ novdec + tnovdec +tjanmar	22	0	0.86	2801.42
Annual: augsept +oct+toct+rationd to jm+ tnovdec +janmar+tjanmar	22	4.39	0.10	2805.82
Annual: augsept +oct+t ct+novdec+tnovdec	16	6.13	0.04	2819.56
Autumn/Winter: oct+toct+ratio jm to nd + novdec+tnovdec+ tjanmar	19	11.33	0.00	2818.76
Autumn/Winter: oct+toct+ratio nd to jm+ tnovdec +janmar+tjanmar	19	11.94	0.00	2819.36
Annual: aprilsept +oct+toct+rationd to jm+ tnovdec +janmar+tjanmar	22	14.72	0.00	2816.14
Annual: aprilsept +oct+toct+novdec+tnovdec	16	21.76	0.00	2835.2
Autumn/Winter: oct+toct+ novdec+tnovdec	13	22.56	0.00	2841.98
Annual: apriljuly+oct+toct+novdec+tnovdec	16	24.85	0.00	2838.28
Annual: augsept +oct+toct+rationd to jm+janmar+tjanmar	19	25.91	0.00	2833.34
Winter: novdec+tnovdec	7	44.43	0.00	2875.86
Winter: octmar+tjanmar	7	55.34	0.00	2886.76
Winter: octmar+toctmar	7	58.97	0.00	2890.4
Annual: aprilsept + octmar + toctmar	10	60.06	0.00	2885.48
Winter: janmar+tjanmar	7	70.05	0.00	2901.48
Summer: augsept + apriljuly	7	108.2	0.00	2939.62
Summer: augsept +oct	7	127.57	0.00	2959
Summer: augsept +oct + toct	10	129.99	0.00	2955.42
Summer: augsept	4	144	0.00	2981.42
Summer: aprilsept	4	158.77	0.00	2996.2
Null	1	416.76	0.00	3260.18

Table 4. Top model set for each of the habitat groupings

Model	K	Δ AIC$_c$	Weight	Deviance	R^2
Aspen Model					
Full	84			1172.7	
augsept +oct+toct+ratio jm to nd+ novdec + tnovdec +tjanmar	22	0	0.484	1289.5	0.73
aprilsept +oct+toct+ratio nd to jm+ tnovdec +janmar+tjanmar	22	0.78	0.328	1290.0	0.73
apriljuly+oct+toct+novdec+tnovdec	16	2.67	0.127	1304.8	0.69
oct+toct+ratio jm to nd +novdec+tnovdec+tjm	19	5.11	0.038	1301.0	0.70
Null	1	269.98	0	1602.9	
Conifer Model					
Full	75			954.9	
oct+toct+novdec+tnovdec	13	0	0.523	1092.9	0.57
aprilsept +oct+toct+novdec+tnovdec	16	2.41	0.157	1089.0	0.58
augsept +oct+toct+novdec+tnovdec	16	2.43	0.155	1089.1	0.58
apriljuly+oct+toct+novdec+tnovdec	16	3.22	0.104	1089.8	0.58
Null	1	159.63	0	1277.2	
Shrub-Steppe Model					
Full	27			315.9	
augsept + apriljuly	7	0	0.367	340.3	0.52
augsept +oct+toct+novdec+tnovdec	16	0.22	0.331	320.2	0.91
augsept	4	0.91	0.234	347.6	0.38
augsept +oct	7	6.0	0.018	346.3	0.40
Null	1	14.0	0	366.9	

Table 5. Coefficients for top models determined by AIC. Coefficients significant at P < 0.05 are bolded.

Variable	Period	Model			
		Overall	Aspen	Conifer	Shrub Steppe
Baseline		**0.9540**	**0.9337**	**0.9462**	**0.8784**
Aug-sept precipitation	1	**-0.0186**	**-0.0249**		**-0.0417**
	2	-0.0019	-0.0025		-0.0049
	3	-0.0006	-0.0066		**-0.0356**
Oct precipitation	1	**-0.0271**	-0.0093	**-0.0241**	
	2	-0.0039	**0.0089**	**-0.0094**	
	3	-0.0077	0.0062	-0.0191	
Nov-dec Precipitation	1	0.0014	0.0048	0.0007	
	2	**0.0080**	**0.0174**	**0.0076**	
	3	0.0031	0.0064	0.0038	
Ratio of jan-mar to nov-dec Precipitation	1	0.0126	0.1363		
	2	**0.1185**	**0.1566**		
	3	0.0616	0.1765		
Minimum temperature Oct	1	**0.1716**	**0.3329**	0.0870	
	2	0.0371	-0.0349	0.0414	
	3	-0.0309	-0.0468	-0.0272	
Minimum temperature nov-dec	1	**-0.3358**	-0.0469	**-0.2177**	
	2	**-0.0819**	**0.1346**	**-0.1839**	
	3	0.1736	0.3632	-0.0443	
Minimum temperature jan-mar	1	0.0230	-0.0805		

	2	**-0.1097**	**-0.1416**	
	3	**-0.2139**	**-0.3039**	
April-July precipitation	1			**0.0062**
	2			0.0032
	3			0.0045

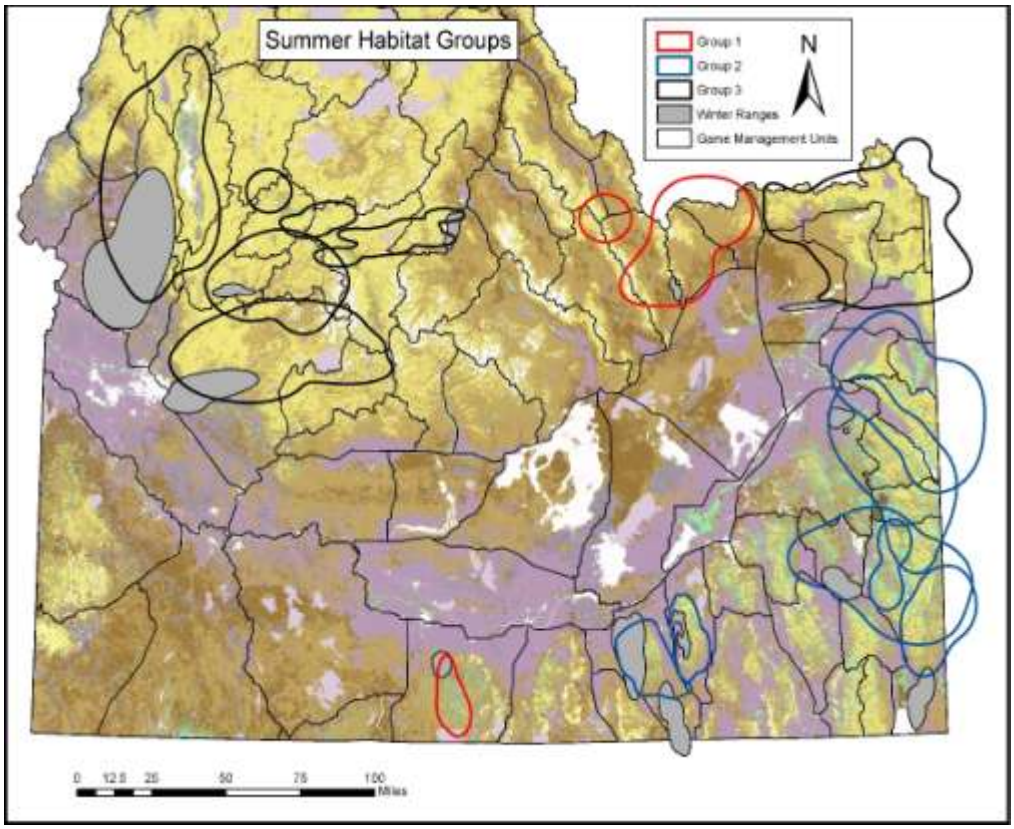

Figure 1 Summer (open outline) and winter (grey) 95% kernal density home ranges of mule deer subpopulation. Red outlines indicate shrub-steppe community types, blue outlines aspen community types and black outlines conifer.

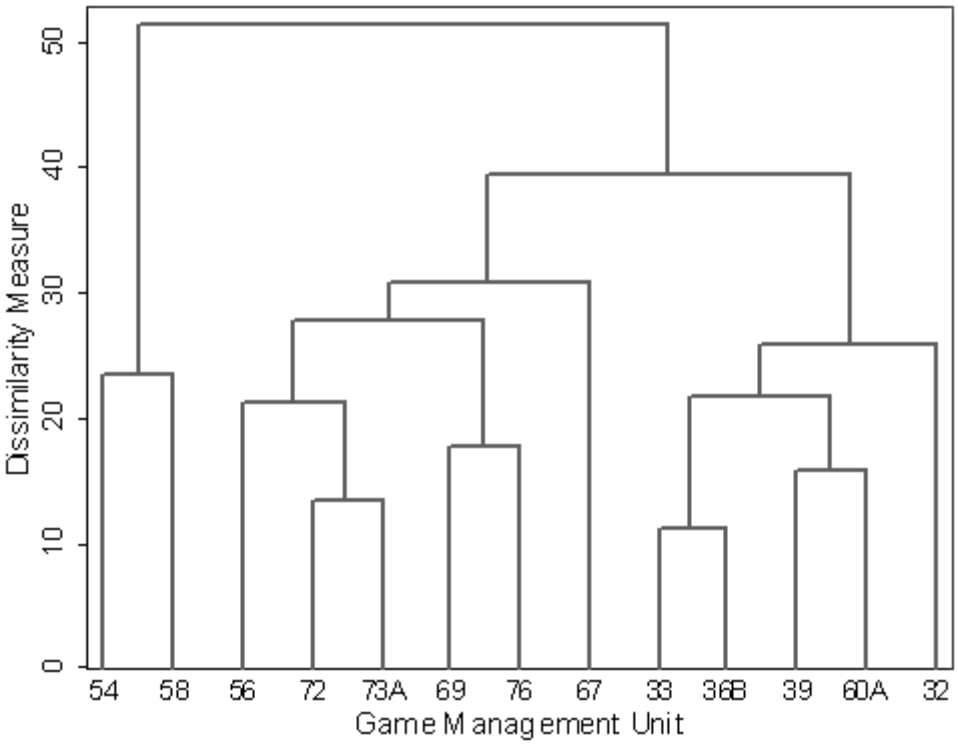

Figure 2. Hierarchical weighted means Grouping of GMUs based on summer vegetation vegetative types. The GMUs are grouped according to similarity of vegetation type percentages within the study area (GMU) summer home range.

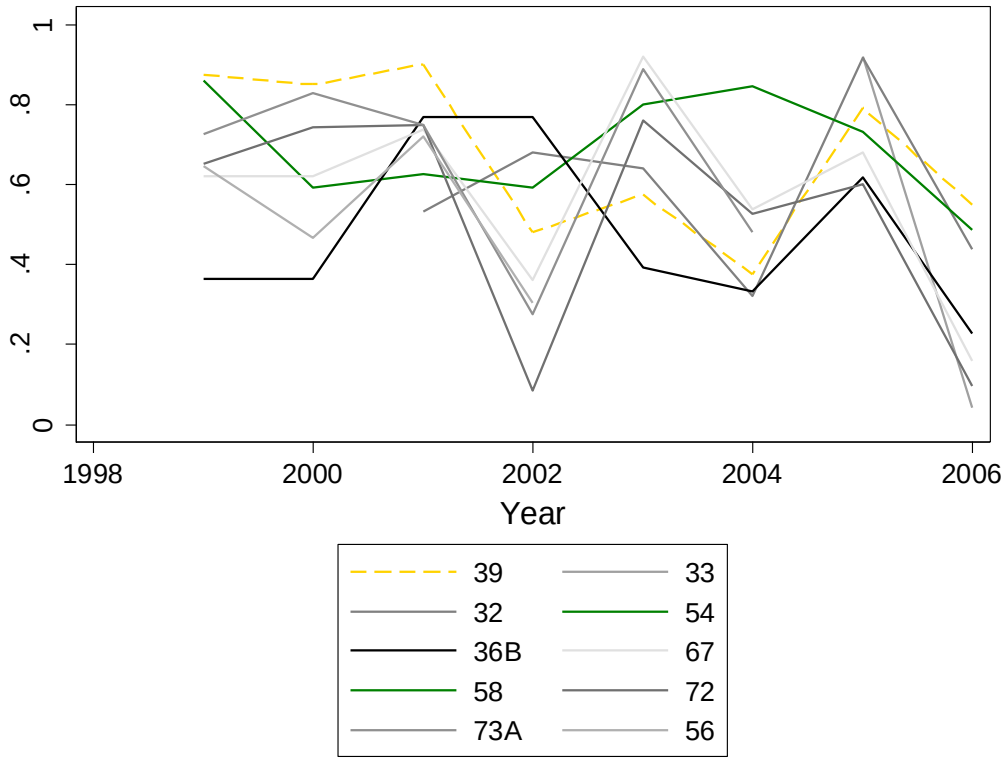

Figure 3. Winter survival estimates of mule deer 16 December to 15 May.

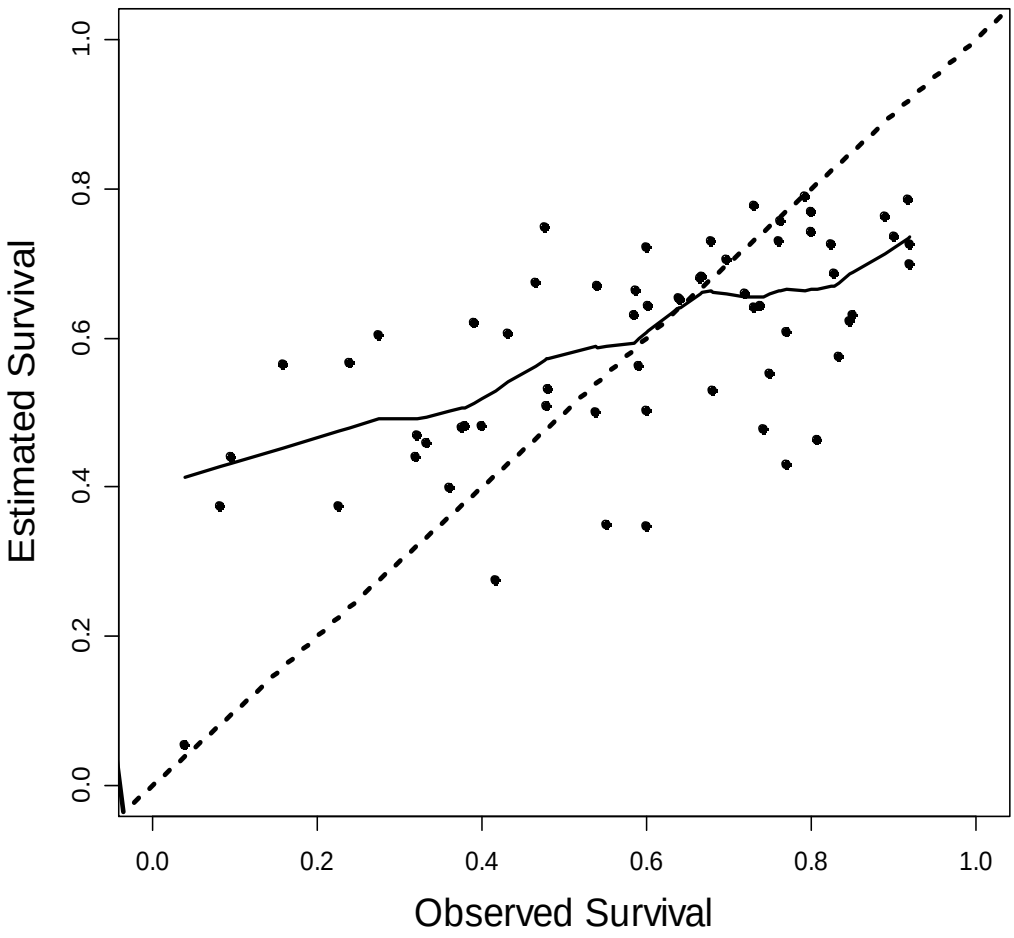

Figure 4. Predicted versus observed winter survival of 6-month old mule deer fawns in southern Idaho during 1999-2006. Survival was estimated using known fates analysis in SURPH from n= 866 radiocollared mule deer fawns. A spline fit is added to show where model is over or under fitting data.

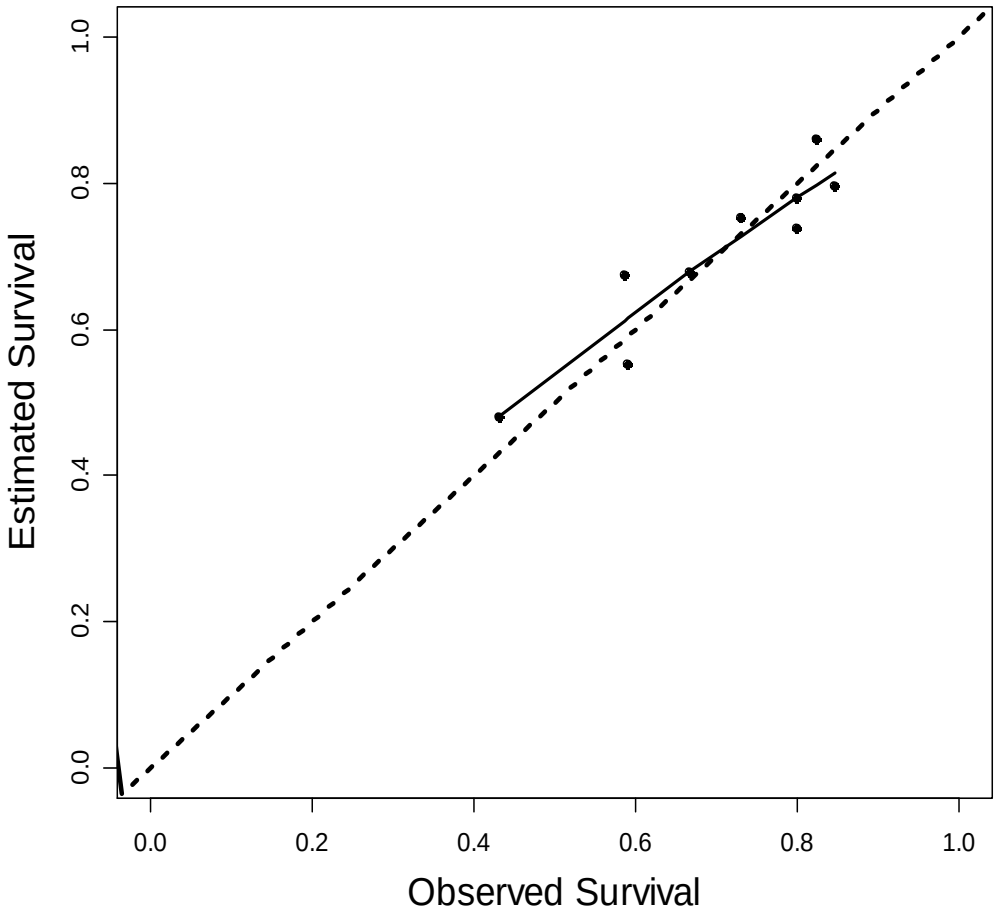

Figure 5a. Predicted versus observed winter survival of 6-month old mule deer fawns in southern Idaho within a) shrub –steppe, b) XXX and c) EEEEE game management units (GMU) during 1999-2006. Survival was estimated using known fates analysis in SURPH from n= XXX radiocollared mule deer fawns. A spline fit is added to show where model is over or under fitting data.

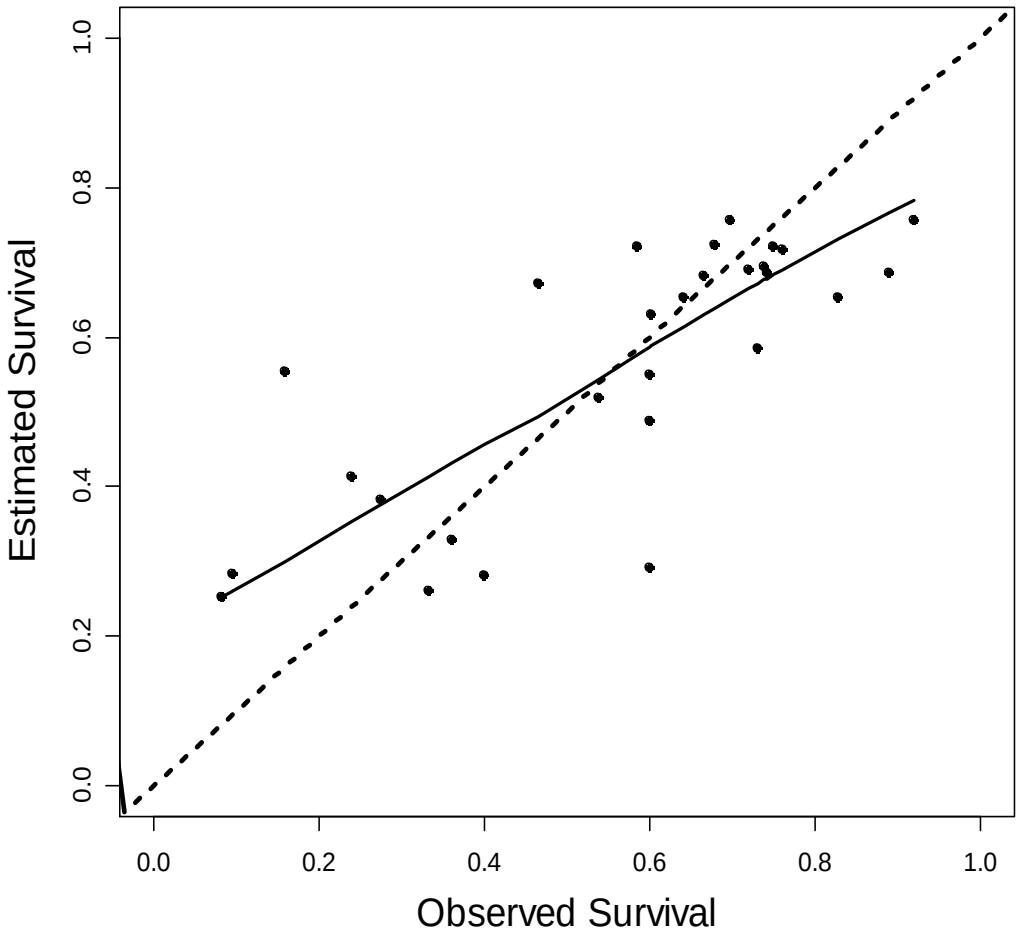

Figure 5b Aspen

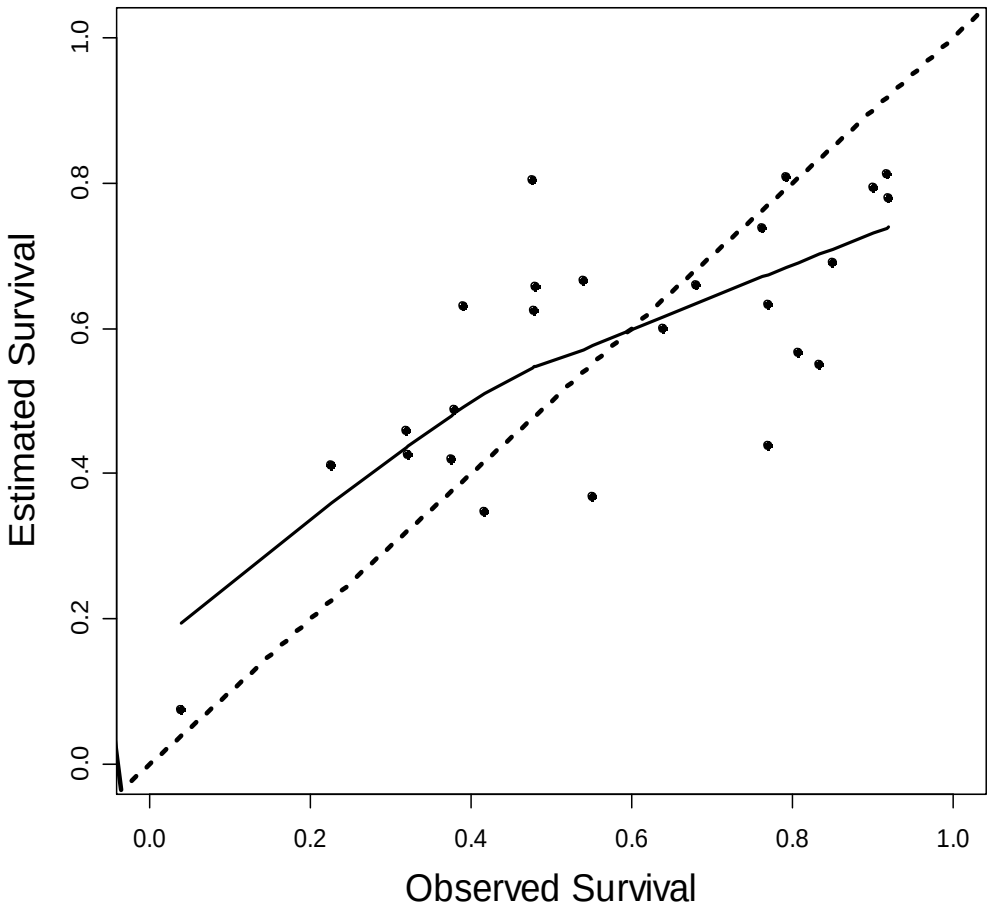

Figure 5c Conifer

STUDY II: EFFECTS OF WOLF PREDATION ON ELK POPULATIONS II

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ABSTRACT During FY10, we captured 80 elk including 40 cows, 7 bulls, and 33 calves. Elk were fitted with GPS radio collars and we obtained various measurements and biological samples. We measured rump fat on captured elk either by palpation or a combination of palpation and ultrasonography. Cow elk serum was evaluated for levels of PSPB to determine pregnancy. We extracted a canine tooth from all elk >3.5 years age and teeth were sectioned to determine age. Elk fecal samples were examined to determine the presence of giant liver fluke ova and all were negative. We captured 8 wolves that were fitted with GPS-Argos radio collars. We conducted monitoring flights on a biweekly basis to determine survival status of elk and wolves. We conducted necropsies on 30 elk and 9 wolves. We downloaded location data from GPS collars obtained from dead animals and collars recovered following release. We currently have 21,328 wolf GPS locations, and 103,153 elk GPS locations. We conducted aerial surveys on both study areas to measure the spatial distribution and abundance of elk and alternate ungulate prey. We measured snow depth and sinking depth at 5 sites. We developed a data base in Access to store data from captures, monitoring flights, necropsies, and to store location data from GPS collars.

There is considerable debate over the influences of wolf predation on elk population dynamics. Much of the currently available information comes from work done in various studies on the northern Yellowstone herd. The effect of wolves on northern Yellowstone elk is intensely debated among scientists and resource managers with some (Eberhardt et al. 2003, White and Garrot 2005) describing more substantial effects than others (Smith et al. 2003, 2004; Wright et al. 2006). Because wolves are often selective for calves, senescent elk, or elk with poorer body condition, researchers have speculated that wolf mortality might be largely compensatory (Smith et al. 2003, 2004). However, wolf predation on prime age elk and elk in "good" condition can be substantial (Kunkel and Pletscher 1999, White and Garrott 2005). Calf:cow ratios typically decline appreciably in the presence of wolf caused mortality, and low recruitment is a driving force of population decline (Raithel et al. 2007). With the reestablishment of wolves in the northern Rocky Mountains, the addition of wolf predation caused reduced elk survival rates (Kunkel and Pletscher 1999) lower population growth rates (Hebblewhite et al. 2002, Garrott et al. 2005), lower calf recruitment (White and Garrott 2005), and elk abundance declines (Kunkel and Pletscher 1999, Garrott et al. 2005), in some herds. Additionally, the effect of wolf predation on ungulate demographics can be exacerbated in the presence of additional large carnivores (Messier 1994, Van Ballenberghe and Ballard 1994).

Wolf kill rates (number of prey or mass of prey killed/wolf/unit time) tend to be somewhat constant over wide ranges of prey densities (Eberhardt 1997, 2000; Eberhardt and Petersen 1999; Boutin 1992; Smith et al. 2004; Garrott et al. 2005; although see Hebblewhite et al. 2003), declining only at very low prey densities (Eberhardt 1997, 2000). This suggests wolf kill rates

can be largely independent of prey density. Alternatively, wolves have responded to changes in prey density functionally or numerically in a density dependent fashion (type II functional response) (Dale et al. 1994, Messier 1994). At present, the regulatory (density dependent) nature of wolf predation is unclear (Boutin 1992, Messier 1994, Van Ballenberghe and Ballard 1994).

Although wolf kill rates tend to be somewhat constant, rates of wolf caused mortality often vary widely on spatial and temporal scales, even within individual study areas (Garrott et al. 2005). Consequently, the impacts of wolf caused elk mortality cannot be generalized. The variation largely may be a function of differences in relative densities of elk and wolves. When present, elk are often the preferred prey of wolves (Carbyn 1983, Huggard 1993, Hebblewhite et al. 2003, Garrott et al. 2005), and kill rates are generally a direct function of the prey biomass required by wolves (Thurber and Peterson 1993, Dale et al. 1994, Ballard et al. 1997, Carbyn 1983). Additionally, wolf predation on elk is potentially influenced by other covariates including the relative abundance and vulnerability of alternate prey (Carbyn 1983, Kunkel et al. 1999), the presence of other large predators (Messier 1994, Van Ballenberghe and Ballard 1994, Atwood et al. 2007), habitat characteristics (Kunkel et al. 2004, Creel and Winnie 2005, Mao et al. 2005, Bergman et al. 2006, Atwood et al. 2007), season (Smith et al. 2004), and winter weather severity, especially related to snow accumulation (Huggard 1993, Jaffe 2001, Mech et al. 2001, Hebblewhite et al. 2002, Smith et al. 2004). Wolves tend to selectively prey on calves (Carbyn 1983, Husseman et al. 2003, Smith et al. 2004, Creel and Winnie 2005, Wright et al. 2006), older and senescent elk (Carbyn 1983, Kunkel et al. 1999, Atwood et al. 2007, Wright et al. 2006, Eberhardt et al. 2003), and may select adult bulls over adult cows (Hamlin 2006, Atwood et al. 2007). Thus, the relative abundance of various sex, age, and condition classes of elk might influence predation rates on other classes, in much the same manner as alternate prey.

We will model wolf-caused mortality of elk (wolf caused mortality rates) as a function of wolf abundance, relative to elk abundance (wolf-days/elk). To an extent, this approach assumes density independent predation impacts (Gasaway et al. 1983), whereas the impact of wolf predation on survival might vary with elk density (Messier 1994), because the ratio relationship ignores functional responses of wolves (Messier and Crete 1985). Beyond the primary relationship between wolf-days/elk and survival, we will attempt to explain additional variability with other covariates such as snow and alternate prey densities. This work will provide a basis for measuring the additivity of wolf predation and will provide a tool for predicting wolf caused elk mortality. This will mirror similar work previously conducted on moose-wolf systems (Messier 1994, Eberhart et al. 2003) and it will further preliminary work on elk-wolf systems (Eberhardt et al. 2003). Our findings will have immediate application for wolf and elk managers in Idaho and elsewhere.

We have the following predictions: 1) Wolf caused mortality of elk will increase with increasing wolf-days/elk; 2) The abundance of alternate prey biomass, relative to wolf-days, will dilute the effect of wolf predation; 3) Wolf caused mortality will increase with increasing snow accumulation; 4) The risk of detection, and conditional risk of elk mortality will vary with topography and vegetation composition (*sensu* Hebblewhite and Merrill 2007), and 5) Wolf caused mortality will vary seasonally.

STUDY AREA

We conducted this work on two study areas in Idaho. The North Fork of the Clearwater River study area (North Fork) in north central Idaho included the upper reaches of the North Fork of the Clearwater River drainage and the Kelly Creek drainage in Game Management Unit (GMU) 10. The area was characterized by steep, rugged, mountainous terrain ranging in elevation from 650m to 2,400m. Forest vegetation types ranged from drier ponderosa pine (*Pinus ponderosa*) and Douglas fir (*Pseudotsuga menziesii*) types on xeric sites to western red cedar (*Thuja plicata*) types on mesic sites. Higher elevations included Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) forest stands. Upland meadows were common throughout the area. Much of the southerly exposures at low elevations were dominated by extensive shrub-fields that resulted from large wildfires in the early 1900s. The Lowman study area in southwest Idaho included the upper reaches of the South Fork of the Payette River drainage in GMUs 33, 34, and 35. The study area was mountainous and ranged in elevation from 900m to 2700m. Vegetation associations were drier than those on the North Fork study area. Most of the lower elevation habitats along the South Fork of the Payette River included seral shrub-fields and meadows resulting from an extensive wildfire in 1989.

METHODS

We captured and radio-collared adult male and female elk, and calf elk during winter via helicopter darting, ground darting and in a corral trap. Darted elk were immobilized with Carfentanil and reversed with Naltrexone. We fitted elk with VHF and GPS telemetry collars (Telonics, Mesa, Arizona; Lotek, Newmarket, Ontario, Canada; Vectronic Aerospace, Berlin, Germany; Telemetry Solutions, Concord, California). Elk were monitored with fixed-wing flights biweekly to detect mortality signals. If a mortality signal was detected, the carcass was investigated to determine cause of death as described by Hamlin et al. (1984). Blood samples were drawn from each adult female for pregnancy testing, with analysis for Pregnancy Specific Protein-B (Biotracking, Moscow, Idaho – Noyes et al. 1997). Chest girth was measured to estimate body weight (Millspaugh and Brundige 1996, Cook et al. 2003). Elk <3.5 years of age were aged via tooth wear and replacement (Quimby and Gaab 1957). A canine tooth was extracted from anesthetized adults >3.5 years age to determine age via cementum annuli (Matson's Laboratory, Milltown, Montana – Hamlin et al. 2000). Annual sex and age specific survival rates were calculated following Pollock et al. (1989). We used 2 methods to measure body condition that related directly to fat composition. When practical, we measured maximum rump fat thickness via ultrasonography (Stephenson et al. 1998, Cook et al. 2001a, 2001b). However, this method requires use of an ultrasound device, which was not practical in many field situations. Alternatively, body fat was evaluated with a body condition score (Gerhart et al. 1996, Cook et al. 2001a, 2001b).

Wolves were captured via leg-hold trapping and helicopter darting. Wolves were immobilized with either Telazol or a combination of Medetomidine and Ketamine. The effects of Medetomidine were reversed with Atipamezol. Wolves were fitted with VHF and GPS

telemetry collars, that were capable of data download through the Argos satellite system (Telonics, Mesa, Arizona; Lotek, Newmarket, Ontario, Canada).

We will model elk survival and wolf-caused mortality as a function of wolf- days/elk (Garrott et al. 2005). Wolf-days/elk will serve as the primary explanatory variable to predict survival rate. Data analysis will be performed with a model-based approach using Program MARK. The data include 1) the number of elk, 2) the number of wolf days, and 3) age- and sex-specific wolf-caused elk mortality and survival. All 3 parameters vary spatially and temporally.

The abundance of elk and ungulate alternate prey (deer, moose) were determined with sightability surveys of study areas (Unsworth et al. 1994). Survey estimates were obtained twice annually corresponding to summer and winter range use. Summer surveys were conducted in late June so that composition ratios could be used to measure early, neonate mortality, and winter surveys were conducted pre-capture during December and January, to estimate additional calf mortality, using composition ratios and estimated cow survival. Those elk abundance estimates were adjusted continuously, post-survey, using age/sex-specific elk survival rates and movements data from radio-marked elk. As elk died or move across the landscape, age and sex-specific densities were adjusted proportionately. The initial resolution was at the level of the survey search units which encompassed 20-60km^2.

The level and spatial distribution of wolf-days were determined with GPS-Argos marked wolves. Study area boundaries were defined by wolf pack territories. We generalized that territories typically contained the associated pack with the possible addition of some transients, and that packs were confined to territories. There was also the added logistic benefit of capturing only wolves from individual packs. Territory boundaries were defined by 95% kernel home range contours. GPS collars were programmed to obtain a fix every 7.5 hours and Argos downloads occurred every 15 days.

The presence of elk among varying levels of wolf-days was determined with GPS locations of radio-collared elk. Elk collars were programmed to obtain locations every 2 hours, during the entire year.

Alternate ungulate prey densities were measured during elk surveys. Moose and mule deer are the predominate alternate ungulate prey and their abundance was estimated using the sightability procedure (Unsworth et al. 1994), during summer and winter study area surveys conducted for elk. Other less abundant ungulate species included mountain goats and white-tailed deer. We indexed white-tailed deer and mountain goat abundance with survey observations.

Winter weather severity, especially related to snow characteristics, can substantially affect wolf caused mortality. Snow effects are complex and relate to depth, density, and crusting that all vary with aspect, slope, temperature, etc. Snow water equivalent (SWE), which is a function of depth and density (SWE/density = snow depth), is an often used metric in ungulate research. SWE and depth are typically highly correlated. Sinking depth is potentially more informative that depth alone, and is a function of depth, density, and can be influenced by crusting. Depth

alone has been used to predict the influence of snow on wolf predation on elk (Hebblewhite et al. 2002, Huggard 1993). We measured snow accumulation in 2 ways. First, data on depth and SWE are available from SNOTEL sites. The SNOTEL data is available at no cost and is reliable, although the sites are typically at much higher elevations than wintering ungulates. Additionally, we established permanent snow measuring sites on the Lowman (3 sites) and North Fork (2 sites) study areas. The sites were at low elevation, on southerly slopes and included areas occupied by elk during winter. We measured snow depth and indexed sinking depth with a drop pentrometer, biweekly at each site.

RESULTS

Adult elk survival was substantially lower at Lowman during 2008, than during 2009 (Table 1). This was largely a function of higher winter mortality that likely was related to deeper snow accumulation during 2008. Adult cow survival and calf survival on the North Fork study area was lower than survival in Lowman during 2009 (Table 1). Bull survival rates were similar on the two study areas (Table 1).

On the Lowman study area wolf predation was the primary source of mortality (Table 2). Across all sex and age classes, wolf predation was responsible for 28% of elk deaths, and wolf predation with potential malnutrition involvement caused an additional 15% of deaths. Malnutrition alone was also a significant source of mortality (13%) (Table 2). Most deaths associated with malnutrition (83%) occurred during the winter of 2008, when snow accumulation was substantial. Predation accounted for 51% of all deaths. The primary cause of bull elk mortality was hunter harvest. On the North Fork study area, wolf predation was the primary source (60%) of mortality (Table 3). Overall, predation by all predator species accounted for 76% of elk deaths.

Pregnancy rates were at “normal” levels for adult elk (Table 4), although sample sizes were insufficient to draw any solid conclusions.

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Table 1. Annual survival rates of adult elk, and 6-month (capture (approx. Jan) – 31 May) survival rates of calf elk, radio-collared on the North Fork and Lowman study areas in Idaho.

Study Area	Bulls		Cows		Calves	
Year	S[a]	(SE)	S	(SE)	S	(SE)
Lowman						
2008	0.42	(0.08)	0.74	(0.05)	0.35	(0.06)
2009	0.65	(0.09)	0.92	(0.04)	0.30	(0.06)
North Fork						
2009	0.71	(0.10)	0.79	(0.06)	0.09	(0.03)

[a] Survival rates and standard errors calculated following Pollock et al. (1989).

Table 2. Causes of death determined from necropsies of dead radio-collared elk on the Lowman study area in Idaho, 2008-2009.

Cause of Death	Bulls	Cows	Calves
Wolf	4	4	11
Wolf/malnutrition[a]	3	2	5
Cougar	0	1	1
Cougar/malnutrition	0	1	0
Predation, species unknown	0	1	1
Predation, spp. unk./malnut.	0	0	0
Malnutrition	2	2	5
Natural accident	1	1	0
Hunter harvest	11	1	0
Auto accident	0	0	1
Cause unknown	0	6	3

[a] Necropsy revealed that the death was caused by predation, but malnutrition was indicated by mostly or entirely depleted bone marrow fat.

Table 3. Causes of death determined from necropsies of dead radio-collared elk on the North Fork study area in Idaho, 2009.

Cause of Death	Bulls	Cows	Calves
Wolf	1	7	7
Wolf/malnutrition[a]	0	0	1
Cougar	1	0	0
Cougar/malnutrition	0	0	0
Predation, species unknown	0	0	2
Predation, spp. unk./malnut.	0	0	0
Malnutrition	0	0	0
Natural accident	0	0	0
Hunter harvest	1	0	0
Auto accident	0	0	0
Cause unknown	2	3	0

[a] Necropsy revealed that the death was caused by predation, but malnutrition was indicated by mostly or entirely depleted bone marrow fat.

Table 4. Pregnancy rates of adult (2 year +) elk captured on the Lowman and North Fork study areas in Idaho.

Study Area	Lowman	North Fork
Year	% Pregnant[a] (n)	% Pregnant (n)
2008	77 (24)	
2009	80 (4)	100 (9)
2010	67 (18)	75 (12)

[a] Pregnancy determined from levels of Pregnancy Specific Protein-B following Noyes et al. (1997).

STUDY III: EFFECTS OF HABITAT AND NUTRITION ON MULE DEER AND ELK POPULATIONS

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ABSTRACT Data from the AVHRR satellite sensor was used to model the phenology of ecoregions within the state of Idaho. The U. S. Geological Survey Environmental Research Observation and Science Center used these data to generate a Normalized Difference Vegetation Index time series from 1989 through 2007. Metrics were derived from this temporal series that correspond to the start of the growing season, peak of the growing season, end of the growing season, duration of the growing season, peak NDVI value for the growing season, and the time integrated NDVI. These metrics were analyzed for trends within the 10 level 3 ecoregions that fall within Idaho, as described by the Environmental Protection Agency. The non-parametric Mann-Kendall trend test was used to ascertain any trends. Overall, the temporal variables were relatively unchanged with a few exceptions. The end of the growing season extended in the Middle Rockies ecoregion by over one week ($p = 0.093$), the end of the season extended by over 12 days in the Wasatch and Uinta Mountain ecoregion ($p = 0.08$), and the peak of the growing season occurred more than two weeks earlier in the Blue Mountain ecoregion ($p = 0.048$). More widespread trends took place in the peak NDVI and time integrated NDVI metrics. Peak NDVI values increased in the Northern Rocky ecoregion by over 4% ($p < 0.001$), in the Columbia Plateau ecoregion by almost 3% ($p = 0.012$), by over 2% in the Blue Mountain ecoregion ($p = 0.09$), and by over 2% in the Wasatch and Uinta Mountain ecoregion ($p = 0.09$). The time integrated NDVI increased in the Blue Mountain ecoregion by 15% ($p = 0.05$), by 33% in the Northern Rocky ecoregion ($p = 0.02$), by 29% in the Idaho Batholith ecoregion ($p = 0.05$), and in the Wasatch and Uinta Mountain ecoregion by 22% ($p = 0.05$). The peak NDVI value and time integrated NDVI metrics have been correlated to primary production and above ground biomass which indicates that while the timing of the growing season may not be shifting within the state, but areas are accruing more yearly biomass during the course of the growing season.

INTRODUCTION

Earlier springs, longer autumns, shorter winters and more variability may be expected in a warming climate. But, has this been happening? The global surface temperature has increased roughly 0.1C - 0.2C every decade for the last 30 years (Albritton et al. 2001, Houghton et al. 2001, Hansen et al. 2006). The date of the year on which flowers bloom, elk calves are born and migratory birds take flight all shift slightly from year to year based on climatological and ecological triggers. If we are experiencing a warming event globally, we would typically expect these events to take place earlier in the year (Penuelas and Filella 2001). For wildlife species this can disrupt the synchrony between animal, vegetation and their dependencies upon each other. Quantifying and characterizing these potential shifts is critical to understanding the potential impacts across landscapes in the western United States.

Landscape phenology refers to the timing of life-cycle events in vegetation across a landscape. In temperate climates such as Idaho, vegetation experiences discernable seasonal cycles including spring greening, a seasonal growth peak, and a fall/winter senescence. This differs

from other landscapes where greening coincides with seasonal rain events, such as monsoons in southwest deserts of the U.S. White et al (2009) have explored different factors, influences, and definitions of phenological metrics, and emphasize that the start of a growing season may be used as a reference point for life-cycle events, such as budbreak or leaf out, or climatological processes (e.g., snow melt or rain events). While phenology is variable and changes slightly from year to year, it would be expected that a changing climate would shift some phenological events permanently. There is a certain amount of elasticity in the timing of egg laying in birds and calving in ungulates, but long term shifts will start to push the evolutionary limits of these species (Lyon et al. 2008). Many wildlife species have developed a synchronous relationship that is tied to the beginning of the growing season. For instance, migratory birds require adequate forage along their migration route at layover spots so that when they arrive at their breeding grounds they have sufficient body stores for egg laying (Van Der Jeugd et al. 2009). If areas start growing earlier the birds run the risk of getting to an area after it has produced its optimal forage. Not only are phenological processes important for cueing life events in wildlife, they are also important for ensuring an adequate amount of resources when and where they are needed.

Satellite based sensors such as the Advanced Very High Resolution Radiometer (AVHRR) have been used to discern phenological events and trends within the growing season from global (Moulin et al. 1997), regional (Li et al. 2002, Balzter et al. 2007, Philippon et al. 2007), and even local scales (Pettorelli et al. 2007). The AVHRR series of satellites was first launched in October of 1978. Since then, this program has continued to provide a consistent database of global multispectral data. The multispectral data from the satellite can then be converted into a ratio called the Normalized Difference Vegetation Indices (NDVI) using the near infrared (NIR) and visible red ———) wavelengths of light (Tucker 1979, Reed et al. 1994). The NDVI has been shown to be highly correlated with leaf area (Pocewicz et al. 2004) and above ground biomass (Rouse Jr et al. 1974, Roy and Ravan 1996). This correlation is explained by interaction of electromagnetic energy from the sun and the physical structure and moisture content of vegetation. Healthy vegetation absorbs the red wavelength while the cell walls and moisture within plant scatter and reflect the NIR. The more healthy vegetation there is the more red is absorbed and NIR reflected, which leads to a higher NDVI value. As plants grow from spring through to autumn, NDVI values will increase as the plants grow and leaf area expands then decrease as the plants senesce.

The AVHRR sensor was initially designed to monitor weather patterns and both oceanic and terrestrial thermal emissions, but the AVHRR archive has been used in a wide variety of earth resource studies. Early research using AVHRR data included land cover classification (Norwine and Greegor 1983), estimates of net primary production (Goward and Dye 1987, Running and Nemani 1988, Box et al. 1989, Running et al. 1989), and African food crop vulnerability (Justice and Hiernaux 1986, Van Dijk et al. 1989). As the temporal length of the dataset grew research started to include trend analysis and change detection (Fuller 1998, Tucker and Nicholson 1999). The length of these records allows researchers to identify true trends rather than simple interannual variability within vegetation. Early work in using these satellite derived NDVI metrics involved simply developing appropriate statistical methods and applying these to short

time series. Reed et al (1994) effectively demonstrated that AVHRR phenological data can be used in conjunction with ancillary data to predict land use change within similar land cover types. By doing this they were able to show that not only can multispectral satellite data be used to define land cover types, but by adding in the phenology of areas these land cover predictions can be improved. Projects such as the Gap Analysis, Landfire Project, Multi-Resolution Land Characteristics Consortium have used these techniques to generate land cover datasets with higher levels of overall accuracy. The use of satellite phenological data has grown to include analyzing NDVI values over time with sinusoidal techniques. Treating NDVI time series as a sine wave, it is possible to discern amplitudes and phases that correspond to important phenological events (Jakubauskas et al. 2001). Building off these advancements, research has begun to include more trend analysis within phenological metrics. Subsequent work has used simple linear regression to determine the directionality and significance of any trends (Tateishi and Ebata 2004, Reed 2006, Bradley and Mustard 2008). However, it has been recognized that due to the autocorrelated nature of the time series data certain assumptions required by simple linear regression may not be met and require a more robust statistical analysis (de Beurs et al. 2009). Realizing the nature and potential of these time series data, unique analysis techniques are becoming more widely used. The nonparametric Mann-Kendall trend test (Mann 1945, Kendall and Gibbons 1962) has demonstrated its robustness in discerning trends within ecological processes such as hydrologic monitoring (Esteban-Parra et al. 1998, Lins and Slack 1999, Burn and Elnur 2002, Yue et al. 2002). However, it has more recently been used to assess trends at a large scale using NDVI derived metrics (Julien and Sobrino 2009, Alcaraz-Segura et al. 2010). This test detects trends over time but does not provide information regarding the magnitude of change. While many researchers use a simple linear regression alone to test for a trend, the autocorrelated nature of NDVI data and it susceptibility to extreme starting or ending values in a time series require a nonparametric test such as the Mann-Kendall test (de Beurs et al. 2009). But if an area shows a significant trend using the Mann-Kendall test it is then appropriate to fit a simple least squares regression and use the calculated slope to determine the magnitude of any trend.

While the data acquired by the AVHRR sensor is powerful for detecting and mapping phenological dynamics, it is imperative that what exactly is being measured is explicitly explained and understood (Bradley and Mustard 2008). The satellite measurements are simply a value assigned to the amount of electromagnetic radiation that is reflected from the earth's surface and must be interpreted as such. Areas as varied as agricultural fields, riparian areas, alpine meadows and tropical rainforests can potentially return the exact same values (Huete et al. 2002). When we look at an area on a landscape level, the boundaries that are established to define different analysis units can be either relative to ecological or political criteria. An example of ecological criteria includes areas of similar temperature, precipitation, soil composition, vegetative components, physiography, and hydrology. The United States Environmental Protection Agency (EPA) has used these criteria to delineate areas within the state of Idaho that fall into 10 categories (U.S. Environmental Protection Agency 2000). These areas represent boundaries within which there are areas of similar ecological composition.

For this research we ask two questions, one regarding the temporal aspects of the growing season and another regarding the accrual of seasonal biomass. First, are any ecoregions within the state experiencing earlier springs, earlier peak in the growing season, or earlier senescence? Additionally, are any ecoregions experiencing changes in the amount of biomass that develops over the course of the growing season? We use a robust non-parametric trend test to investigate whether these phenological changes that have taken place within the 10 ecoregions in Idaho.

METHODS

Study Area

The overall study area for this research is the state of Idaho. A thorough discussion of the physiography and vegetation of Idaho can be found in Caicco et al (1995). Two sets of boundaries were chosen, one ecologically based and one politically based. First, ecoregions as defined by the EPA were chosen at a level 3 detail. This level breaks Idaho into 10 areas; Blue Mountains, Central Basin and Range, Columbia Plateau, Idaho Batholith, Middle Rockies, Northern Basin and Range, Northern Rockies, Snake River Plain, Wasatch and Uinta Mountains, and the Wyoming Basin (Fig 1.2). These areas are defined in detail by McGrath et al (2002).

The Rocky Mountains are separated into 3 different ecoregions within Idaho. The Northern Rockies start at the Lochsa River and continue well into British Columbia, Canada. Although this area is well inland from the Pacific Ocean, it is still considered to be a modified marine climate. Vegetation in this area consists of Douglas and subalpine firs, ponderosa pine, and species that are more suited for coastal environments such as western hemlock, western red cedar, and grand fir. This area is differentiated from the Idaho Batholith to the south by the lesser amount of granitic soil parent materials. To the north, the Central Canadian Rocky Mountains reach to higher elevations and contain much more area that is covered in perennial snow and ice.

A small portion of the Columbia Plateau extends from the eastern breaks of the Cascade Mountains in Washington State into the western portion of Idaho. The Columbia Plateau ecoregion in Idaho has been extensively cultivated for wheat due to a basalt layer that is covered by fertile loess soils. Historically this area consisted of dry grasslands and sagebrush steppe.

The Blue Mountains are just to the south of the Columbia Plateau. This area is dissected by Hells Canyon creating an environment that involves high peaks in the Seven Devils and Wallowa Mountains to low canyons along the Snake River. Hiking and river sports are important to this area, but also a wide variety of wildlife and fish species including Rocky Mountain bighorn sheep (*Ovis canadensis canadensis*), mountain goat (*Oremnos americanus*), and white sturgeon (*Acipenser transmontanus*) are found here.

The center of the state is dominated by the Idaho Batholith. The Salmon River winds through the heart of this heavily dissected landscape. The Idaho Batholith is higher than the Northern Rocky and Blue Mountain ecoregions and logging is the dominant land use.

To the east of the Idaho Batholith is the Middle Rockies ecoregion. Dry sagebrush valleys extend up to the highest peak in the state, Borah Peak at 12,662 feet. A drier environment, species such as Douglas-fir and subalpine fir compose much of the open canopy forest cover which leads into shrub and grassland dominated valleys.

The Snake River Plain cuts across the southern portion of the state from the eastern border to the west. The Snake River is the dominant influence in this ecoregion providing irrigation important to sugar beet, potato, and alfalfa.

The southwestern corner of the state is covered by the Northern Basin and Range ecoregion. Sagebrush grasslands and saltbush greasewood dominate this region that is higher and cooler than the Snake River Plain.

The final three comprise the southeastern corner of the state. These areas represent the extreme edges of the ecoregion, but nonetheless are still indicative of the ecosystems they represent. First, the Central Basin and Range extends north from the Great Salt Lake into southern Idaho. This ecoregion is similar to that of the Northern Basin and Range, but has more northern facing slopes, has more widespread grazing, and is not well suited for irrigated cropland.

The second small section is the Wasatch and Uinta Mountain ranges. Engelmann spruce and subalpine fir are found here, which separate it from the Central Basin and Range ecoregion. Higher and more mountainous this ecoregion is also heavily grazed in the summer months.

Last, a small section of the Wyoming Basin constitutes the most southeastern corner of the state. Grazing is important here, but portions of this ecoregion are also used for irrigated hay production. Rolling plains descend into flat, poorly drained valleys that support sedges and other marshy plant species.

Derivation and explanation of phenological metrics

For this research weekly NDVI values from the AVHRR sensor were used to create a time series from 1989 through 2007. These maps are generated at a 1km by 1km spatial resolution. For each year a weighted least-squares algorithm was applied to the 52 weekly points to generate a smooth curve (Swets et al. 1999) (Fig 1.1). NDVI time series have been shown to have erroneous spikes that can be attributed to cloud cover or other atmospheric perturbations, but the algorithm is weighted to account and adjust for these typically lower values (Holben 1986, Swets et al. 1999). Daily NDVI values are then interpolated along this smooth curve. To determine the starting date of the growing season, a backward looking algorithm tests an NDVI value against a mean generated from the 18 previous values to see if it has departed significantly which would signal the starting date of the growing season (Jesslyn Brown, personal communication, June 5, 2009). The algorithm used to calculate starting date is reversed to generate the ending date of the growing season. This date is usually marked by one of two events, either vegetational senescence or snow fall. The peak of the growing season is simply the Julian date upon which the largest NDVI value falls. The peak NDVI value reached during a season correlates to the maximum amount of vegetative capacity of a site for that year. The time integrated NDVI (TIN)

value is calculated using a combination of the starting and ending dates, starting NDVI value and ending NDVI value, where a non-growing season baseline is calculated using the NDVI values associated with the start and stop dates, and the growing season is determined by using the start and end dates. The TIN is a unitless metric that represents net accumulation of NDVI values above the non-growing season baseline during the growing season. The duration of the growing season is simply the number of days between the start and the end of the growing season.

Datasets

All of the images containing the phenological data for the years from 1989 through 2007 were reprojected into NAD 83 zone 11 using ERDAS IMAGINE 9.3 (ERDAS Inc. 2008) using nearest neighbor interpolation with rigorous transformation. The GMU polygon layer was downloaded from the IDFG webpage (Idaho Department of Fish and Game 2007) and reprojected into NAD 83 zone 11 within ArcGIS (ESRI 2008). Ecoregion shapefiles were downloaded from the EPA website and reprojected exactly as the GMUs were (U. S. Environmental Protection Agency 2010). We based our trend analysis on averaged values of each NDVI-based metric within individual ecoregions, and included only 1km x 1km pixels that were ≥75% within that region. Pixels that did not contain enough data within the smoothed NDVI dataset to create accurate starting, ending, or maximum date were censored. We averaged NDVI metrics across a study area using the "isectpolyrst" script within the Geospatial Modeling Environment (Beyer 2004) and exported the data into database format. The output included mean, standard deviation, minimum, maximum, and median among all of the pixel values within an ecoregion.

Analysis

We applied two types of analysis to the datasets, one to test for the significance of any trends at the ecoregions level and a second to test for the directionality and magnitude of trends. First, to test if the trend was significant; the R statistical language (R Development Core Team 2009) was used to perform the non-parametric Mann-Kendall trend test (Mann 1945, Kendall and Gibbons 1962, McLeod 2009, Alcaraz-Segura et al. 2010). The trend tests were conducted on the mean values calculated from the zonal (or ecoregion) statistics. To determine the directionality and magnitude of any trends within these ecoregions metrics over time, a simple least squares regression line was fitted for each metric and ecoregion that were significant at a 95% confidence interval resulting from the Mann-Kendall trend test. Corresponding magnitude and directionality were calculated by using the slope generated from the least squares regression.

RESULTS

Ecoregion Level Results

Start of the growing season

The start of the growing season did not trend significantly from 1989 through 2007 in any of the ecoregions according to the Mann Kendall test results. Overall starting dates among the ecoregions ranged widely from early March in the Columbia Plateau to late April in the Idaho Batholith (Table 1.2).

Table 1.2 - Mann Kendall trend test, linear regression, and simple statistics for start of season phenology in ecoregions across Idaho.

Trend Results for SOST	Mann Kendall Results		Linear Regression Results		Basic Statistics (Julian day)		
	Tau	*p* - value	intercept	slope	Mean	Minimum	Maximum
Columbia Plateau	0.19	0.26	72.14	-0.32	73	60	88
Blue Mountains	0.03	0.89	84.68	-0.01	86	76	94
Snake River Plain	0.11	0.53	77.12	-0.22	78	65	88
Central Basin and Range	0.05	0.78	80.91	0.12	82	70	96
Northern Rockies	0.16	0.36	105.40	-0.31	104	80	120
Idaho Batholith	0.05	0.78	114.22	-0.06	114	103	130
Middle Rockies	0.13	0.44	101.97	0.13	101	96	115
Wyoming Basin	0.12	0.48	89.45	0.18	87	80	103
Wasatch and Uinta Mountains	0.25	0.14	96.14	0.52	95	90	112
Northern Basin and Range	0.01	1.00	75.20	-0.11	75	60	88

End of the growing season

The end of the growing season is extending later in the fall within two ecoregions. The growing season in the Middle Rockies, encompassing much of the center of the state, is extends more than one week ($p = 0.093$) into the autumn. The growing season is nearly 12 days longer ($p = 0.08$) in the Wasatch and Uinta mountains. By early October the Columbia Plateau has reached

the end of its growing season when the vegetation reaches senescence. In contrast, vegetation in the Idaho Batholith grows continuously up until the New Year when the growing season is ended by the first snow fall (Table 1.3).

Table 1.3 - Mann Kendall trend test, linear regression, and simple statistics for end of season date in ecoregions across Idaho. Dates that extend into the next year may exceed 365.

Trend Results for EOST	Mann Kendall Results		Linear Regression Results		Basic Statistics (Julian day)		
	Tau	*p* - value	intercept	slope	Mean	Minimum	Maximum
Columbia Plateau	0.09	0.62	274.94	-0.13	274	249	302
Blue Mountains	0.18	0.29	319.89	0.35	327	310	333
Snake River Plain	0.01	1.00	304.94	0.13	314	292	323
Central Basin and Range	0.26	0.12	315.59	0.95	326	311	348
Northern Rockies	0.02	0.94	342.78	0.01	348	319	362
Idaho Batholith	0.03	0.89	351.42	-0.05	352	337	369
Middle Rockies	0.29	0.09	340.45	0.41	343	335	357
Wyoming Basin	0.17	0.33	328.40	0.52	340	318	357
Wasatch and Uinta Mountains	0.30	0.08	336.75	0.63	345	323	356
Northern Basin and Range	0.20	0.23	301.57	0.71	323	291	332

Peak of the growing season

The peak of the growing season in the Blue Mountains occurred more than two weeks earlier (-15 days, $p = 0.048$) in 2007 versus 1989. There was no significant change in the peak of the growing season in the other ecoregions (Table 1.4).

Table 1.4 - Mann Kendall trend test, linear regression, and simple statistics for peak of the growing season phenology in ecoregions across Idaho.

Trend Results for MAXT	Mann Kendall Results		Linear Regression Results		Basic Statistics (Julian day)		
	Tau	*p* -	intercep	slop	Mea	Minimu	Maximu

		value	t	e	n	m	m
Columbia Plateau	0.03	0.89	160.36	-0.02	157	147	178
Blue Mountains	0.30	0.08	172.17	-0.80	170	142	184
Snake River Plain	0.16	0.36	160.70	-0.42	161	142	172
Central Basin and Range	0.10	0.58	155.12	-0.30	154	129	173
Northern Rockies	0.03	0.89	197.04	-0.06	197	176	217
Idaho Batholith	0.16	0.36	204.26	-0.30	206	176	219
Middle Rockies	0.13	0.44	185.73	-0.34	181	170	202
Wyoming Basin	0.01	1.00	168.05	0.03	168	153	189
Wasatch and Uinta Mountains	0.03	0.89	173.90	-0.19	172	154	197
Northern Basin and Range	0.12	0.48	153.20	-0.23	149	133	174

Duration of the growing season

There was no significant change in the duration of the growing season between 1989 and 2007 in any of the ecoregions (Table 1.5).

Table 1.5 - Mann Kendall trend test, linear regression, and simple statistics for the growing season duration in ecoregions across Idaho.

Trend Results for DUR	Mann Kendall Results		Linear Regression Results		Basic Statistics (number of days)		
	Tau	*p* - value	intercept	slope	Mean	Minimum	Maximum
Columbia Plateau	0.04	0.83	40.13	0.09	201	184	232
Blue Mountains	0.13	0.44	38.56	0.33	241	224	254
Snake River Plain	0.12	0.48	32.48	0.02	236	213	251
Central Basin and Range	0.20	0.23	38.98	-0.05	245	221	270
Northern Rockies	0.12	0.48	36.98	0.75	244	219	271

Idaho Batholith	0.02	0.94	39.30	0.67	238	224	251
Middle Rockies	0.09	0.62	40.28	0.33	241	226	261
Wyoming Basin	0.03	0.89	46.41	-0.26	254	221	272
Wasatch and Uinta Mountains	0.06	0.73	52.53	0.64	250	224	265
Northern Basin and Range	0.17	0.33	29.08	0.19	248	212	267

Peak NDVI values

Three ecoregions saw increases in the peak NDVI value reached during the course of the growing season. First, the Columbia Plateau has seen an increase of almost 3% ($p = 0.012$). The Northern Rockies have seen an increase of over 4% ($p < 0.001$). The Blue Mountains increased by just over 2% ($p = 0.093$). Lastly, the Wasatch and Uinta Mountains increased by over 2% ($p = 0.093$) (Table 1.6).

Table 1.6 - Mann Kendall trend test, linear regression, and simple statistics for peak NDVI values in ecoregions across Idaho.

Trend Results for MAXN	Mann Kendall Results		Linear Regression Results		Basic Statistics (NDVI)		
	Tau	*p* - value	intercept	slope	Mean	Minimum	Maximum
Columbia Plateau	0.43	0.01	0.43	0.27	0.43	0.41	0.44
Blue Mountains	0.29	0.09	0.41	0.20	0.41	0.39	0.42
Snake River Plain	0.18	0.29	0.38	0.16	0.38	0.37	0.40
Central Basin and Range	0.01	1.00	0.40	0.07	0.39	0.38	0.41
Northern Rockies	0.70	< 0.01	0.44	0.37	0.44	0.43	0.45
Idaho Batholith	0.22	0.21	0.41	0.13	0.41	0.40	0.42
Middle Rockies	0.04	0.83	0.39	0.05	0.39	0.38	0.40
Wyoming Basin	0.10	0.58	0.39	-0.16	0.39	0.38	0.41
Wasatch and Uinta Mountains	0.29	0.09	0.42	0.21	0.42	0.41	0.44

Northern Basin and Range	0.11	0.53	0.37	0.10	0.37	0.35	0.39

Time Integrated NDVI

Four areas saw significant trends within the time integrated NDVI values over the 19 year time span. It is important to reiterate that this metric likely relates to the amount of biomass that accrues over the entire length of the growing season. This is influenced both by the length of the growing season and the daily NDVI values. The resulting trends represent an increased total amount of primary productivity of an area for the course of the entire growing season. The Blue Mountains are accumulating over 15% (p = 0.05) more biomass during the course of the growing season. The Northern Rockies are accumulating the most with 33% (p = 0.02) more biomass during the growing the season. In the center portion of the state the Idaho Batholith is accumulating 29% (p = 0.05) more per year. The final ecoregion is the Wasatch and Uinta Mountains which have increased 22% (p = 0.05) per year (Table 1.7).

Table 1.7 - Mann Kendall trend test, linear regression, and simple statistics for time integrated NDVI in ecoregions across Idaho.

Trend Results for TOTN	Mann Kendall Results		Linear Regression Results		Basic Statistics (unitless)		
	Tau	*p* - value	intercept	slope	Mean	Minimum	Maximum
Columbia Plateau	0.12	0.48	40.13	0.09	38	34	56
Blue Mountains	0.33	0.05	38.56	0.33	40	33	49
Snake River Plain	0.11	0.53	32.48	0.02	33	28	45
Central Basin and Range	0.01	1.00	38.98	-0.05	41	29	54
Northern Rockies	0.40	0.02	36.98	0.75	44	33	64
Idaho Batholith	0.49	< 0.01	39.30	0.67	44	33	57
Middle Rockies	0.25	0.14	40.28	0.33	44	32	53
Wyoming Basin	0.16	0.36	46.41	-0.26	45	32	57
Wasatch and Uinta Mountains	0.33	0.05	52.53	0.64	56	42	74
Northern Basin and Range	0.13	0.44	29.08	0.19	31	21	42

DISCUSSION

Ecoregion level changes

At the ecoregion level, trends were most pronounced in the metrics relating to primary production metrics rather than the temporal metrics. Start of season, end of season and peak of season were relatively unchanged. The length of the satellite record used here was from 1989 through 2007 representing 19 years of observations. Previous studies have found trends with shorter time series, but these studies used a simple linear regression slope applied to the data to determine any trends (Tateishi and Ebata 2004, Reed 2006, Bradley and Mustard 2008) . As discussed earlier this method makes some detrimental assumptions (de Beurs et al. 2009). When a more robust test is applied trends are much less common as detected through this research. In heterogeneous landscapes such as Idaho, the interannual variation in the temporal metrics is high due to many different factors. Satellite sensors such as the AVHRR are tools that simply measure electromagnetic radiation. This leaves the onus on the user of the data to account for what is actually being measured on the ground. Some areas trended in both the time integrated NDVI and maximum NDVI values while some areas only trended in one metric or the other.

These differences depict three different scenarios that are playing out across these landscapes. Time integrated NDVI is a function of the length of the growing season, daily NDVI values, and the maximum NDVI value. To increase this metric at least one of these would have to change. If an area has trended in just the time integrated NDVI this would suggest that the area is not seeing higher overall values, but values that stay consistently high through the growing season. The one area that falls into this category is the Idaho Batholith which has seen a highly significant positive trend which means that each year there is more and more biomass across the landscape. This area has a variety of vegetation from western red cedar (*Thuja plicata*) in the northern portions to dry ponderosa pine (*Pinus ponderosa*) in the elevationally lower areas. But what this metric indicates is that this area, even though it has seen a few large fires, has been steadily accruing biomass over the last 19 years. The Idaho Batholith is comprised of soils that are very prone to erosion when vegetation is removed (Meyer et al. 2001). While increased attention has been placed on vegetation retention to abate erosion, it is unlikely that these influences are enough to account for the overall increase across the entire area. The Idaho Batholith contains high alpine areas that have a growing season that is typically limited by the timing and persistence of snowfall. Callaghan et al (1995) suggest that areas such as these will experience longer growing seasons as a result of a warming climate. A second important observation in the changing metrics is areas that have had positive trends in the maximum NDVI value, but have not increased in the time integrated NDVI. The Columbia Plateau and Northern Rockies are the two ecoregions that fall into this category. These two areas constitute the two most northerly ecoregions in this study. The changing dimensionality of the growing season in these areas is characterized by having more biomass during the peak of the growing season, but not necessarily having more biomass that accrues over the course of the growing season. The last possible combination of these two metrics is an ecoregion that has experienced trends in both the time integrated NDVI and the maximum NDVI metric. Just one area falls into this category, the Wasatch and Uinta Mountain ecoregion.

This type of analysis provides a methodology for continued monitoring of changes in landscape phenology that can arise for a number of reasons.

CONCLUSION

We used the data generated by the USGS EROS center to look at landscape scale phenological events across the state of Idaho to see if any areas are showing the influences of a warming climate between the years of 1989 through 2007. This type of analysis is venturing into new and novel approaches as datasets become easier to use and better correlated to ground based events. Historically, phenological events were documented via point-based sampling then interpolated across a landscape. Now we have the ability to use remotely sensed data to ascertain phenological events at even the most remote field sites. This ability has applications across most ecology-based disciplines. Wildlife investigations are only one application of these phenological datasets. Snowmelt monitoring, mapping successional stages of vegetation, post-wildfire vegetation regeneration are just a few other applications. With newer and higher resolution sensors such as the Moderate-Resolution Imaging Spectroradiometer (MODIS) still actively collecting multispectral data, we can expect newer phenological datasets that can build on these methods and continue into the future.

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Table 1.1 Available data products from the USGS EROS center.

Description	**Acronym**	**Brief Description**	**Phenological Significance**
Start of Season	SOST	Date on which the NDVI values significantly depart from the winter average.	When the first green forage becomes available.
End of Season	EOST	Date on which the NDVI values return to a winter average.	When vegetation senesces.
Date of Maximum NDVI Value	MAXT	Date on which the maximum NDVI value falls within the growing season.	Represents the length of time animals have until forage reaches maximum growth for the season.
Maximum NDVI Value	MAXN	NDVI value on the date representing the maximum value for the growing season.	When the growing season peaks, how intense was it at that particular point in time.
Time Integrated NDVI	TIN	Value representing accumulated biomass over entire growing season	Over the course of the growing season, this is how much overall vegetational production there was.
Duration of Growing Season	DUR	Length of time between the starting day and ending day of the growing season	How long is green forage available?

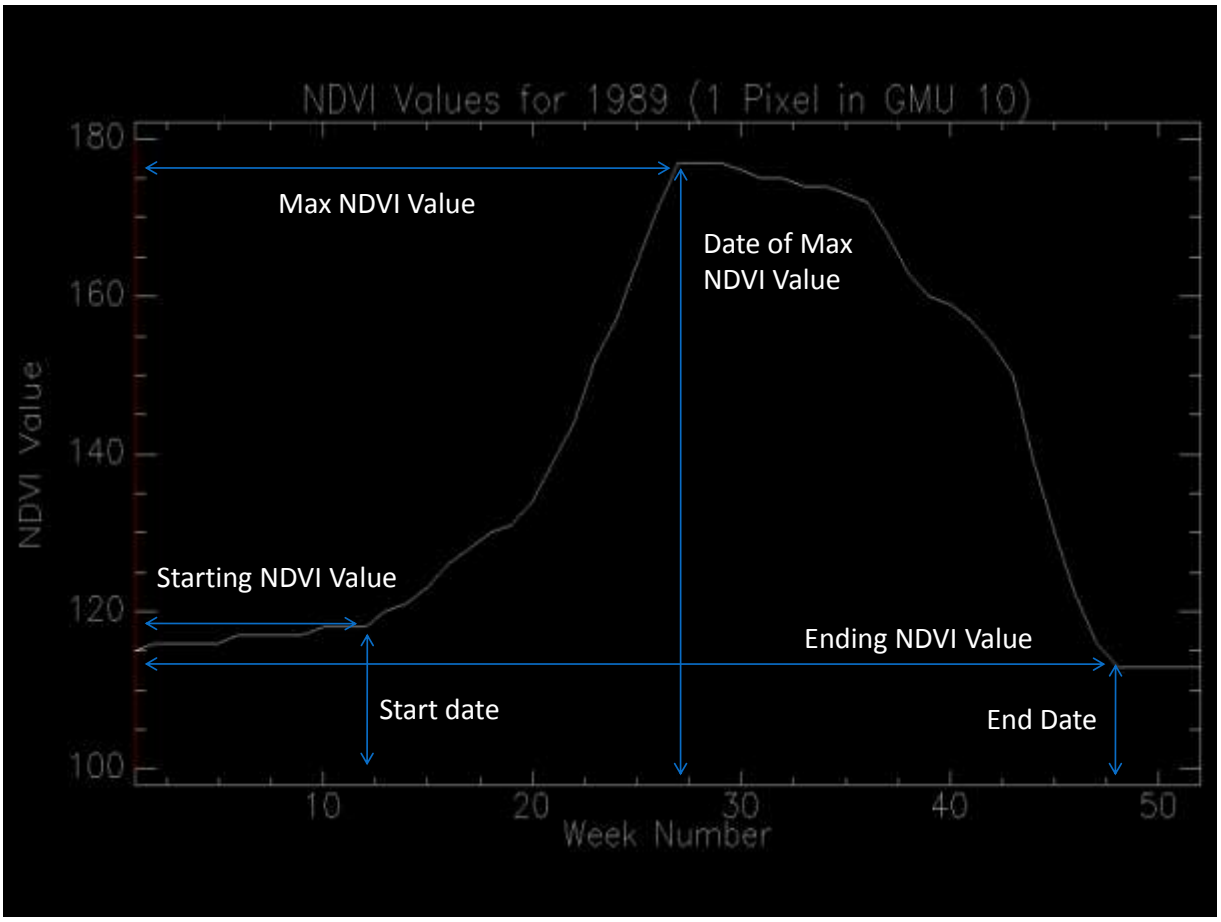

Figure 1.1 Grapical depiction of a growing season as measured by the normalized difference vegetation index (NDVI). Values on the y-axis have been multiplied by 400 and represent the amount of absorbed red and reflected near-infrared energy within the electromagnetic spectrum. This relationship has been shown to be highly correlated with above ground biomass (Rouse Jr et al. 1974, Roy and Ravan 1996). Phenologic moments are shown that coincide with the start, peak, and end of the growing season.

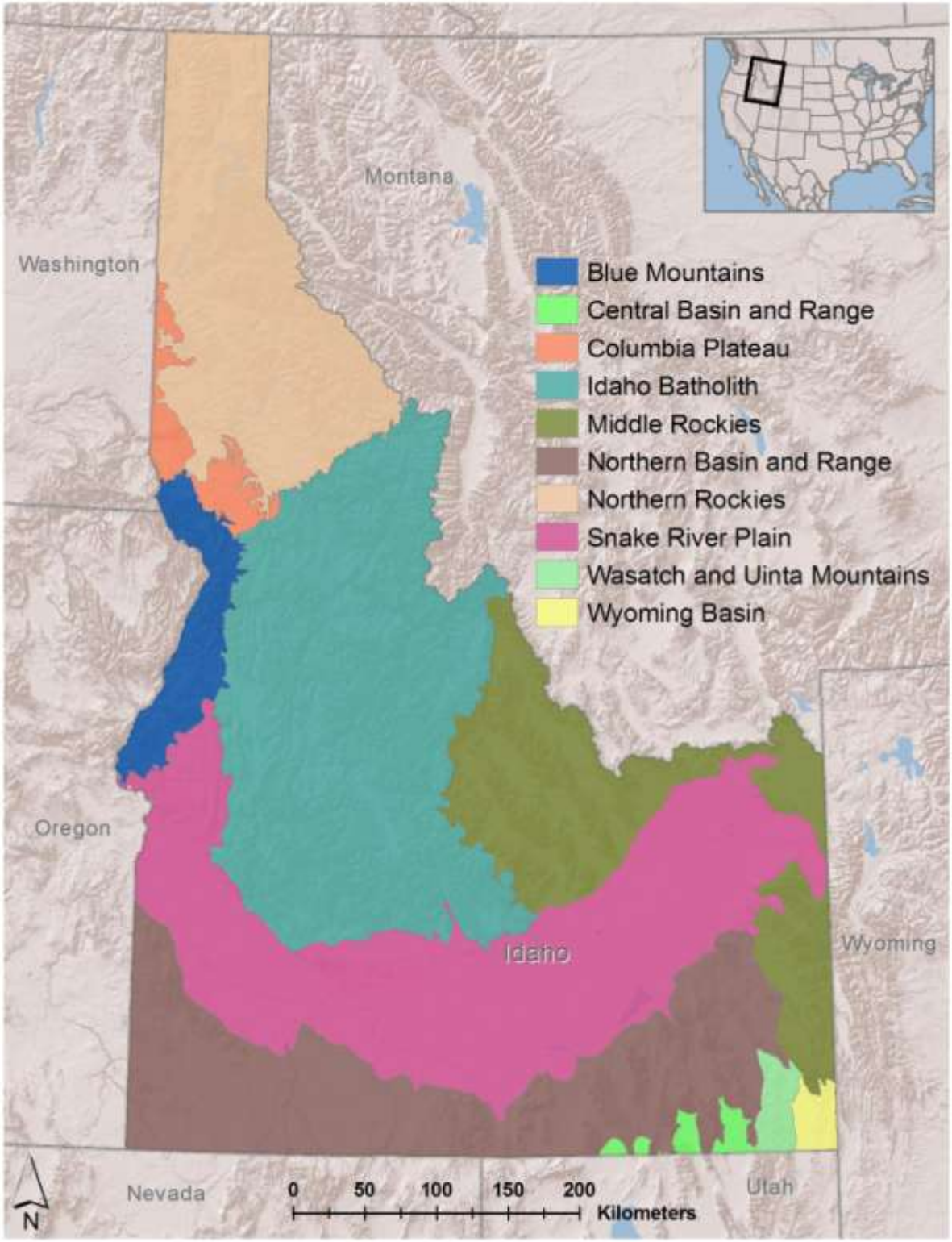

Figure 1.2 Level 3 ecoregions of Idaho as defined by the Environmental Protection Agency.

STUDY IV: HELLS CANYON BIGHORN SHEEP

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ABSTRACT The Hells Canyon Initiative is a state, federal, and private partnership to restore Rocky Mountain bighorn sheep *Ovis canadensis canadensis* in the Hells Canyon area of Oregon, Idaho, and Washington. During 2009-2010, 118 radio-collared bighorn sheep were monitored regularly in 12 populations. Average annual survival of radio-collared ewes was 0.90 and annual ram survival was 0.78. Lamb survival was highly variable among populations. Pneumonia-caused mortality of lambs was confirmed in 5 populations and suspected in 3 others. The overall metapopulation was estimated at 840 bighorn sheep, essentially unchanged from an estimate of 830 sheep in 2008 - 2009.

Multi-year collaborative projects were continued with the University of Idaho Caine Veterinary Teaching Center and the Department of Veterinary Microbiology and Pathology at Washington State University to better understand the clinical causes of disease, in particular *Mycoplasma ovipneumoniae*, in bighorn sheep. An experiment copasturing bighorn and domestic sheep was completed during this period and two manuscripts are in preparation. A new collaboration was also started with Penn State University to analyze the spatial and temporal ecology of disease in the Hells Canyon bighorn sheep metapopulation.

KEY WORDS Bighorn sheep, disease, *Mycoplasma ovipneumoniae, Ovis canadensis*

The Hells Canyon Initiative was started in 1995 as a program to accelerate restoration of bighorn sheep in Hells Canyon and the surrounding areas of Idaho, Oregon, and Washington and to focus research applicable to bighorn sheep restoration and management throughout the western United States and Canada. The concept was formalized in 1997 with the completion of an interagency memorandum of agreement and restoration plan (Hells Canyon Bighorn Sheep Restoration Committee 1997). The restoration plan was updated in 2004 (Hells Canyon Bighorn Sheep Restoration Committee 2004).

STUDY AREA

The Hells Canyon Initiative project area encompasses 2,273,194 ha (5,617,062 ac) in the Snake River drainage in Oregon, Idaho, and Washington from the mouth of Clearwater River, Idaho, south to Brownlee Reservoir. It is bounded on the east by the hydrologic divide between the Salmon and Snake rivers near Riggins, Idaho, south to Brownlee Creek on the Payette National Forest, Idaho, and extends just west of the Eagle Cap Wilderness, Wallowa-Whitman National Forest, Oregon. Major drainages include the Snake, Grande Ronde, Imnaha, and lower Salmon rivers. There are currently 16 bighorn sheep populations, or herds, established in the project area (Fig 1). Over 1.3 million acres (24%) of the project area is potential bighorn sheep habitat, 68% of which is publicly owned, primarily managed by the U.S. Forest Service (USFS). Other public land managers are the states of Oregon, Idaho, and Washington and the Bureau of Land Management (BLM).

METHODS

Population Monitoring

One hundred eighteen radio-collared bighorn sheep (82 ewes, 36 rams) were monitored in 12 populations from 1 June 2009 to 31 May 2010. We monitored radio-collared ewes weekly from mid April through September to determine productivity and as needed to document summer lamb survival and we monitored movements and survival of all sheep a minimum of biweekly year round. We also captured and radio-collared additional bighorn sheep via helicopter netgun and at a corral trap during this reporting period. The state wildlife agencies of Idaho, Oregon, and Washington conducted surveys from a helicopter (Robinson 44 and Hiller 12E) and on the ground in 2009-2010 to estimate total population size.

RESULTS AND DISCUSSION

Adult survival

Sixteen radio-collared adult bighorn sheep (8 ewes, 8 rams) died during this period in the Asotin (4), Black Butte, Imnaha, and Lostine (3 ea) Mountain View (2), and Muir Creek, Redbird, and Upper Hells Canyon, Oregon (1 ea) populations. Cause of death could be determined for 11 animals. Known causes of mortality were trauma (4), predation or probable predation (4), and harvest (3 rams; Mountain View, Asotin, and Imnaha) (Fig 2). One of the sheep that died from trauma (Black Butte) also had chronic bronchopneumonia. Averaged over the metapopulation, annual survival of radio-collared ewes was 0.90 (72/82) and rams was 0.78 (28/36). By population, annual survival of ewes in populations (7) with 5 to 15 radio-collared females ranged from a low in Black Butte and Asotin of 83%, 89% at Imnaha, and 100% in Lostine, Redbird, Sheep Mountain, and Wenaha (Tables 2 and 3).

Lamb Survival

Lamb survival was highly variable among populations. In 2009, with the exception of the Imnaha population, summer lamb survival in the 5 populations where lambs were diagnosed with pneumonia was 50% or less (Tables 3, 5), and recruitment in these populations ranged from 0 to 33 lambs/100 ewes (Table 4). Pneumonia-caused lamb mortality was suspected in another 4 populations where summer survival was 0 – 33% and recruitment was 0 to 16 lambs/100 ewes (Table 4), although no dead lambs were recovered.

Capture

Sixty-four bighorn sheep were captured and radio-collared in February and March 2010. Fifty-seven (36F, 21M) bighorn sheep were captured by helicopter netgun (Quicksilver Air, Peyton, CO), sampled, radio-collared, and released February 8 – 13. The capture was conducted collaboratively by Idaho Department of Fish and Game, Oregon Department of Fish and Wildlife, Washington Department of Fish and Wildlife, Wallowa-Whitman and Umatilla

National Forests, State of Idaho BLM, and the Wild Sheep Foundation. Additional funding was provided by Shikar-Safari Club International. Another 7 bighorn sheep (6F, 1M) were captured in a corral trap in the Wallowa Mountains Lostine population by the Oregon Department of Fish and Wildlife. This brought the total number of collared sheep in the project area to 111 females and 50 males.

Population Monitoring

Approximately 840 bighorn sheep are estimated to occur in 15 herds or populations within the project area (Table 6). There was little change in numbers from last year. We are currently seeking to incorporate a sightability model we have developed for bighorn sheep in Hells Canyon into a software program to better estimate population size and to obtain confidence limits on the estimate. We were unsuccessful at working with existing University of Idaho software "Aerial Survey" http://www.cnrhome.uidaho.edu/fishwild/Garton/tools and are looking for other options.

Disease Research and Management

Collaborative research projects continued between the Hells Canyon Initiative bighorn sheep restoration project and the University of Idaho Caine Veterinary Teaching Center and WSU College of Veterinary Medicine to incorporate field data from Hells Canyon bighorn sheep and laboratory analysis and experiments to better understand causes of pneumonia in bighorn sheep.

In 2009 - 2010, a primary project was a captive experiment to test the role of *Mycoplasma ovipneumoniae* in bighorn sheep pneumonia associated with domestic sheep contact. Previous experiments in which bighorn and domestic sheep were commingled cumulatively resulted in pneumonia in 88 of 90 (97.8%) of the bighorn sheep within 100 days, while bighorn sheep similarly commingled with other ungulates experienced only sporadic fatality (9 experiments, 4 of 56 animals, 7.1%). *Mycoplasma ovipneumoniae* is host-specific for Caprinae, is widely carried by domestic sheep, and has been associated with pneumonia of wild bighorn sheep. We hypothesized that the presence of this organism may be a primary initiating cause of pneumonia in bighorn sheep.

When we commingled *M. ovipneumoniae*-free domestic and bighorn sheep (N=4 each) for 100+ days, one bighorn sheep died with acute pneumonia but the others remained healthy, a survival rate markedly different ($P = 0.002$) from previous bighorn-domestic sheep contact studies, and similar to bighorn sheep survival in previous contact studies with other ungulates. After 104 days, we experimentally infected one of the commingled domestic sheep with *M. ovipneumoniae*, inducing epizootic pneumonia in the three remaining bighorn sheep. The disease was severe but non-fatal. A single *M. ovipneumoniae* strain type was detected in the nose of all domestic and bighorn sheep, and the lungs of pneumonic bighorn sheep at necropsy. Some strains of Pasteurellaceae isolated from the lungs of pneumonic bighorn sheep were indistinguishable from strains detected pre-commingling in only the domestic sheep, and others were indistinguishable from strains detected pre-commingling in only the bighorn sheep. Finally, we found retrospective evidence of *M. ovipneumoniae* involvement in previous domestic

sheep – bighorn sheep commingling experiments in the form of *M. ovipneumoniae* specific 16S rDNA sequences and seroconversion to *M. ovipneumoniae* in sera from a bighorn sheep that survived commingling. Overall, these results suggest *M. ovipneumoniae* is a primary pathogen of epizootic pneumonia in bighorn sheep commingled with domestic sheep, and Pasteurellaceae are capable of acting sporadically as primary agents, or consistently as secondary opportunistic infections in *M. ovipneumoniae*-induced pneumonia.

A second disease-related project that was initiated in 2009-2010 involves a retrospective analysis of population and health data in the bighorn sheep metapopulation. This project is being undertaken in collaboration with Penn State University. We hope to examine the relative role of outside introduction of pathogens vs. transmission among bighorn sheep populations within the metapopulation in perpetuating pneumonia in bighorn sheep in Hells Canyon.

Hells Canyon Bighorn Sheep Restoration Committee

The Hells Canyon Bighorn Sheep Initiative is conducted by the Hells Canyon Bighorn Sheep Restoration Committee. The committee is comprised of state, federal, tribal, and private organizations, of which each provide an administrative (A) and technical (T) committee member. Organizations and committee members in 2009 were:

Idaho Department of Fish and Game
- Dale Toweill, Wildlife Program Coordinator (A)
- Frances Cassirer, Tri-State Coordinator/Wildlife Research Biologist (T)

Oregon Department of Fish and Wildlife
- Don Whitaker, Program Coordinator (A)
- Vic Coggins, District Wildlife Biologist (T)

Washington Department of Fish and Wildlife
- Donny Martorello, Big Game Program Manager (A)
- Paul Wik, Wildlife Biologist (T)

USDA Forest Service
- Tim Schommer, Forest Biologist, Wallowa-Whitman National Forest (A)
- Mark Penninger, District Wildlife Biologist, Wallowa-Whitman National Forest (T)

USDI Bureau of Land Management
- Tom Rinkes, Wildlife Biologist, Idaho State Office(A)
- Craig Johnson, Wildlife Biologist, Salmon-Clearwater Resource Area (T)

Foundation for North American Wild Sheep
- Gray Thornton, President/CEO (A)
- Lloyd Oldenburg and Rick Brigham, Members (T)

Nez Perce Tribe (Ad hoc member)

- Keith Lawrence, Wildlife Biologist (A)
- Curt Mack, Wildlife Biologist (T)

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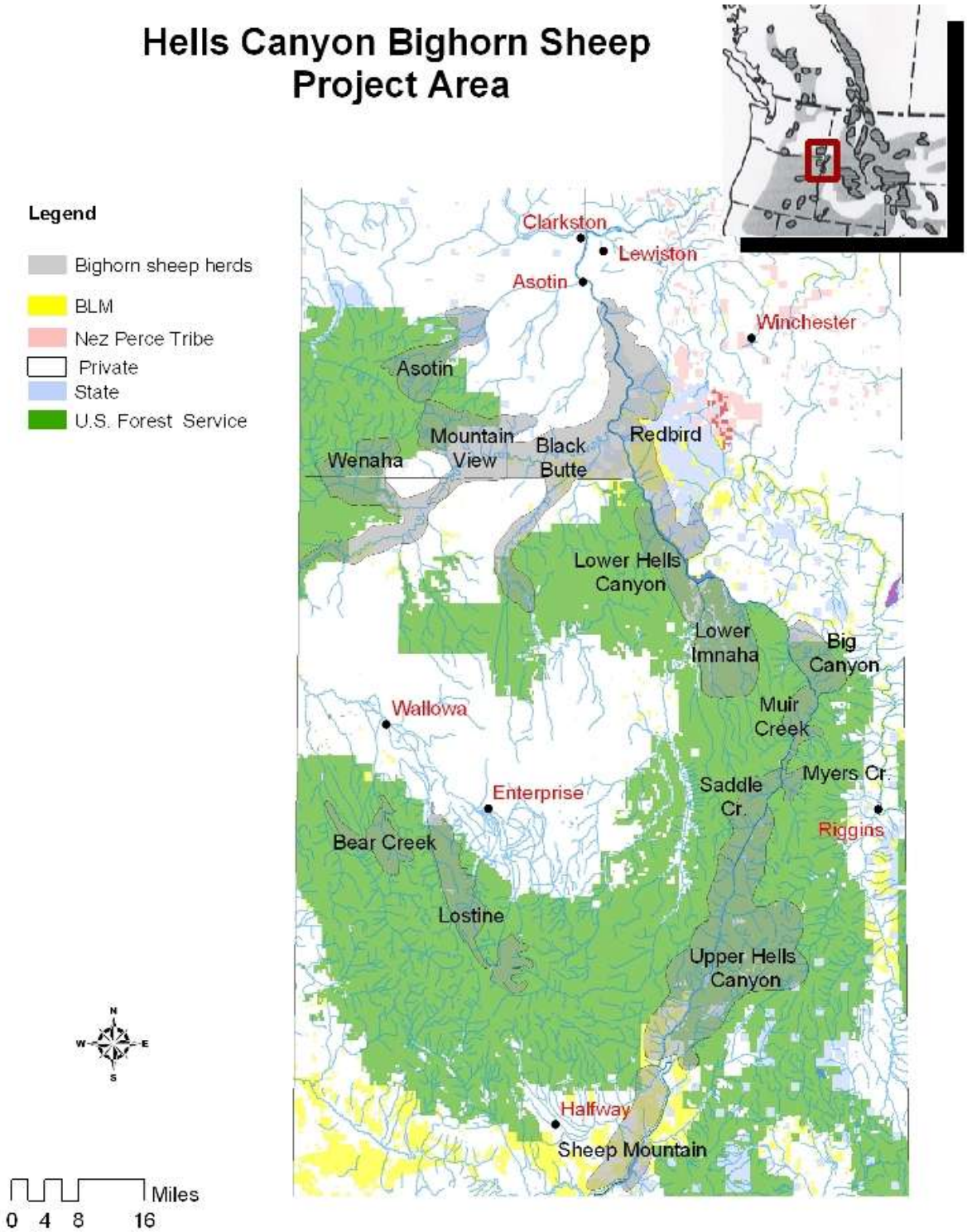

Figure 1. Hells Canyon Initiative Project Area.

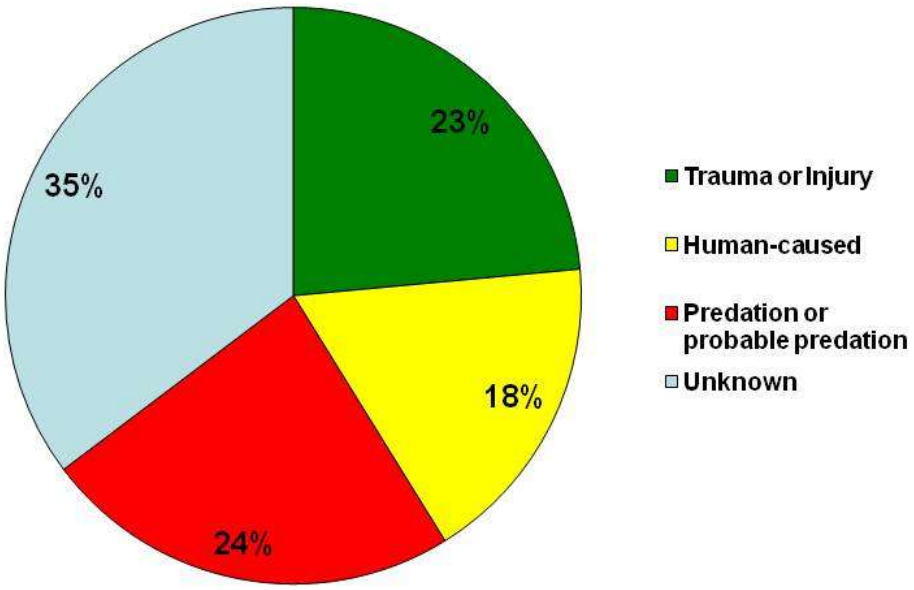

Figure 2. Causes of mortality of 17 adult radio-collared bighorn sheep in Hells Canyon, 1 June 2009 to 31 May 2010.

Table 1. Annual ewe survival in 10 Hells Canyon bighorn sheep populations, 1 June 1997 to 31 May 2010.

Year	*Black Butte*	*Redbird*	*Wenaha*	*Asotin*	*Lostine*	*Imnaha*	*Big Canyon*	*Muir Creek*	*Mountain View*	*Sheep Mountain*
1997-1998	0.92	1	0.91							
1998-1999	1	1	1	0.88			1	1		
1999-2000	0.58	1	1	0.86			0.93	0.93		
2000-2001	0.71	1	0.73	1	1	0.86	0.57	0.67		
2001-2002	0.80	0.92	1	1	1	1	1	1		
2002-2003	1	0.91	1	0.80	0.94	1	0.88	0.50		
2003-2004	0.84	0.73	0.75	0.83	0.94	0.80	0.29	0.88		
2004-2005	0.75	1	1	0.80	0.81	0.92	1	1		
2005-2006	1	0.91	0.89	0.88	0.80	0.91	1	0.78		
2006-2007	1	0.86	0.93	0.73	0.88	1	1	0.86	1.00	1
2007-2008	0.93	0.91	0.62	1	0.92	0.80	1	1	0.60	0.88
2008-2009	0.75	1	1	1	1	0.56	0.70	1	1.00	0.71
2009-2010	0.83	1	1	0.83	1	0.89	1	1	0.67	1
Average	0.85	0.94	0.91	0.88	0.93	0.87	0.86	0.89	0.82	0.90

Table 2. Annual ram survival in 10 Hells Canyon bighorn sheep populations, 1 June 1997 to 31 May 2010.

Year	*Black Butte*	*Redbird*	*Wenaha*	*Asotin*	*Lostine*	*Imnaha*	*Big Canyon*	*Muir Creek*	*Mountain View*	*Sheep Mountain*
1998-1999							1	1		
1999-2000							1	1		
2000-2001	1	1	0.67		0.80	0.67	0.80	0.50		
2001-2002	0.8	0.80	1		1	1	0.75	0.50		
2002-2003	0.3	0.75	1		0.80	1	0.33			
2003-2004	0.5	0.83	1	0.8	0.80	0.67	1	1		
2004-2005	1	0.60	0.88	0.75	0.83	0.67	1	1		
2005-2006		1	1	0.33	1	0.50		1		
2006-2007	1	0.83	0.83	1	0.78	0.83			1	
2007-2008	1	0.75	0.71	0.60	1	0.63	1		0.67	
2008-2009	1	0.83	0.86	0.92	0.88	0.83			1	
2009-2010	0.75	1	1	0.82	0.86	0.60				
Average	0.83	0.83	0.88	0.78	0.82	0.81	0.79	0.87	0.78	

Table 3. Observed productivity and summer lamb survival in 9 herds in Hells Canyon, 2009.

Herd	No. radio-collared ewes observed with lambs (%)		Percent summer survival[a]
Asotin Creek, Washington	9/12	(75)	78%
Big Canyon, Idaho	1/1	(100)	0%
Black Butte, OR/WA	10/11	(91)	**10**
Imnaha, Oregon	8/9	(89)	**63**
Lostine, Oregon	8/12	(75)	0
Mtn View/Wenaha,WA OR	8/10	(80)	**20**
Muir Creek, Oregon	2/3	(75)	33
Redbird, Idaho	9/15	(60)	**44**
Sheep Mountain, Oregon	3/5	(60)	**33**

[a] Survival from birth to 1 October. Herds in bold are those where lambs were recovered with pneumonia.

Table 4. Hells Canyon bighorn sheep population counts, 2009-2010.

Herd	Survey date(s)	Total	Ewes	Lambs	Rams	**Estimated population**
Asotin, WA[a]	2/19/10	100	46	17	37	**105**
Bear Creek, OR		44	22	12	10	**50**
Big Canyon, ID[a]	3/14/10	22	12	3	7	**25**
Black Butte, WA/OR[a]	2/26/10	44	29	3	12	**65**
Lostine, OR[a]		54	34	3	17	**60**
Lower Hells Canyon, OR		30	19	3	8	**35**
Lower Imnaha, OR[a]		118	73	18	27	**135**
Mtn View/Wenaha, WA/OR[a]	2/16 – 2/19/10	91	50	10	31	**120**
Muir Creek, OR[a]	3/14/10	24	13	5	6	**35**
Myers Creek, ID		9	6	0	3	**10**
Redbird, ID[a]	3/10/10	109	59	17	39	**115**
Saddle Creek, OR		30	18	7	5	**35**
Sheep Mountain, OR[a]		8	7	1	0	**8**
Upper Hells Canyon, ID		13	8	0	5	**15**
Upper Hells Canyon, OR		21	13	1	7	**30**
Total		**717**				**843**
Average						**56**

[a] Populations monitored intensively under the Hells Canyon Initiative.

Table 5. Dead pneumonic bighorn lambs submitted to the Washington Animal Diagnostic Laboratory June 1, 2009 – May 31, 2010

Animal ID	*WADDL ID*	*Mortality Date*	*Herd*	*Sex*	*Age (approx days)*	*Weight (kg)*	*Mycoplasma ovipneumoniae PCR*
09OR04	2009-5924	5/25/2009	Sheep Mt	F	21	7.38	Neg
09OR05	2009-6303	6/3/2009	Mt View	F	28	7.8	No test
09OR08	2009-6844	6/15/2009	Mt View	F	28	7.3	Pos
09OR07	2009-6844	6/17/2009	Sheep Mt	F	9	5	Neg
09ID01	2009-7171	6/24/2009	Redbird	M	35	5.7	Neg
09WA01	2009-7615	7/8/2009	Black Butte	F	56	8	Pos
09WA02	2009-7927	7/14/2009	Black Butte	M	56	11	Pos
09WA03	2009-7927	7/14/09	Black Butte	M	56	9.8	Pos
09OR10	2009-7995	7/16/2009	Imnaha	M	56	9.7	No test

APPENDIX 2
Progress Reports for Studies in Statewide Bird Ecology

STUDY I; PART 1: GREATER-SAGE GROUSE (*CENTROCERCUS UROPHASIANUS*) LEK ATTENDANCE RATES IN SOUTHERN IDAHO

Greater Sage-Grouse lek attendance rates in southern Idaho

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ABSTRACT: Lek counts are used to assess population trends of greater sage-grouse (*Centrocercus urophasianus*). Our objective is to estimate the probability of birds attending leks and the probability of detecting attending birds during a lek count in order to relate counts of birds at leks to population abundance. We also attempted to estimate survival of nests and survival of yearling and adult birds in our study population. We used mark-resighting techniques to model the probability of male greater sage-grouse attending leks and counts of grouse from blinds located within 20 m of leks to estimate detectability. Birds were captured in the winters of 2006 – 2009 and fitted with 16.5 g necklace-style radio transmitters. Triangulation from 2 locations off each lek was used to "re-sight" marked birds. We fit a Cormack-Jolly-Seber model to these data using program MARK. We restricted our predictor variables to time (Julian date), year, age of birds (adult or yearling), and their interactions. The top model chosen by AIC model selection procedures included the variables of year, age, and a quadratic time trend. For our final analysis of the complete data set, we will include additional variables such as lek-specific covariates and weather in our candidate set of models, which should result in more precise estimates of attendance probability. Preliminary analysis of our detectability study indicate that between 79% and 91% of the birds actually attending a lek are included in the counts performed during a lek route. Nest success for our study area ranged from 13.3% and 27.6% during the period from 2007-2009. Further analysis of these data should produce estimates of yearling and adult survival for this population, in addition to identifying variables that affect the probability of

individual birds being included in lek route counts, adult and yearling survival, and nests surviving to hatch.

KEY WORDS Greater Sage-Grouse (*Centrocercus urophasianus*), lek attendance, lek route, nest survival, sightability, survival.

Recent trends based on population monitoring indicate that populations of Greater Sage-Grouse (*Centrocercus urophasianus*) are generally declining throughout their range (Connelly and Braun 1997, Connelly et al. 2004). Schroeder et al. (2004) estimated that the range of sage-grouse has shrunk to approximately 56% of the pre-settlement distribution. Braun (1998) stated that according to available data, the number of male sage-grouse counted on breeding grounds each spring decreased from the 1950s through the 1990s throughout their range. While these indices are somewhat crude due to the nature of historical data, they are cause for alarm and justify more intense investigation.

The mating strategy of sage-grouse offers a convenient opportunity to observe and count individuals that congregate on breeding grounds (Patterson 1952, Jenni and Hartzler 1978, Connelly et al. 2003). Each spring, males assemble and display on leks, and females visit the leks to select a male for breeding (Höglund and Alatalo 1995). Due to the conspicuousness of displaying males and the lack of cover that is typical of leks, these congregations are relatively easy to locate (Schroeder et al. 1999). Moreover, lek sites are usually traditional and persist for long periods of time (Dalke et al. 1963).

Lek routes currently provide the best index to breeding population levels throughout much of the species' range (Connelly et al. 2000). The current method for conducting a lek route includes locating all or some portion of the leks of a breeding population visually from low-flying aircraft or audibly from the ground, identifying groups of leks for developing lek routes, then revisiting each lek within a route at least 4 times throughout the spring to count the number of males present (Connelly et al. 2003). Trends are estimated from these data by calculating the greatest number of males counted on a single visit across all leks within a route, for multiple years.

Because lek counts and lek routes may not be representative of the entire population of interest, alone they simply provide an index to breeding population levels and fall under what Anderson (2001) calls "convenience sampling". Although these congregations of breeding sage-grouse offer easy counting of individuals, leks may not be random subsets of the population. Yearling males and adult males may not be attracted to the breeding grounds in proportion to their actual ratio, and females only spend a fraction of the time on leks that males do (Dalke et al. 1963, Jenni and Hartzler 1978, Emmons and Braun 1984, Walsh et al. 2004). Furthermore, not all birds attending a lek during a lek route census are necessarily going to be observed and included in the number reported. Size, behavior, and location within the lek may all affect the sightability of the attending birds by an observer. Any population estimates resulting from a lek route would likely be biased for a particular sex or age class, depending on the time of the counts. Moreover, using counts from a lek route to estimate numbers of male sage-grouse is of little use as no valid technique exists to assess precision of such estimates (Anderson 2001).

There are concerns with using a lek route even as an index. Using uncorrected counts as an index may be unreliable because counts are contingent upon the following assumptions: 1) the sample is proportional to the population; 2) the proportion remains constant among years when trends are estimated; 3) the proportion remains constant among sites where relative abundance is to be compared; and 4) the detection probability is the same for all observers (Anderson 2001, White 2005). Despite stringent guidelines for conducting lek routes (Connelly et al. 2003), these assumptions may not be realistic. Nichols (1992) stated that detection probabilities vary over time and space due to factors that are beyond our control. Further, if these assumptions are not verified, there is a risk of reporting highly biased results (White 2005).
The objectives of this project include determining: 1) how the probability of attending a lek differs among adult male, yearling male, and female sage-grouse; 2) how and what biological factors affect these probabilities; and 3) what variables affect the sightability of birds attending a lek; in order to 4) develop a method to obtain unbiased estimates of abundance for each segment of a population from lek count data. Secondary objectives of this study are to estimate survival of nests, juveniles, and adult birds, determine the sex ratio of wintering populations, and estimate the harvest rate of the study population.

STUDY AREA

Our research is being conducted on Browns Bench, in Twin Falls County in south central Idaho and this area extends into Elko County in north central Nevada. Brown's Bench is bordered to the east by Salmon Falls Creek Reservoir and to the west by an area of rolling hills locally known as Monument Springs. This area receives approximately 24 cm of precipitation annually and ranges in elevation from 1524 m to 2300 m. The major cover types include low sagebrush (*Artemisia arbuscula*) /black sagebrush (*A. nova*) /grass, Wyoming sagebrush (*A. tridentate ssp. wyomingensis*)/grass, mountain sagebrush (*A. tridentate ssp.vaseyana*)/grass, mountain shrub, and crested wheatgrass seedings (*Agropyron cristatum*). Other, less dominant cover types include aspen woodland (*Populus tremulodies*), mountain mahogany woodland (*Cercocarpus montanus*), and wet meadow/riparian (Hironaka et al. 1983, Klott et al. 1993). Livestock grazing is the most common land use for the entire study area. The Bureau of Land Management manages most of the land with privately owned ranches comprising the rest.

We used 9 leks in 2007 located on Browns Bench south of the Three Creek Road and north of the Idaho/Nevada border (Figure 1). In 2008 and 2009, we included an additional 5 leks located north of Three Creek Road, east of Cedar Creek and west of Salmon Falls Creek. We also included an additional 2 leks within the Brown's Bench area that were detected to be active in 2009. A preliminary study of sage-grouse in this area conducted in 2005-2006 indicated that the population was non-migratory. Nevertheless, data collected in 2007-2009 indicate that some of the birds comprising the winter population of our study area move large distances throughout the year (over 30 km) and are likely be migratory (Connelly et al. 2000).

Methods

Pilot Study

Trapping and marking. − In 2005-2006 male sage-grouse were captured using spotlighting (Wakkinen et al. 1992, Connelly et al. 2003) in the vicinity of the leks during early to mid-March. These males were resident in the area and had been displaying irregularly on sunny days since February. Twenty-three females (22 adults and 1 yearling) had been captured in earlier studies during previous years.

Attendance Probability and Sightability Estimation. − Lek counts by 3 observers with 1 observer counting displaying males and attending females according to the protocol by Connelly et al. (2003) commenced on 25 March and were repeated approximately weekly with the seventh count occurring on 3 May 2005. Simultaneously, 2 observers located radio marked grouse using dual-element, null-peak Yagi antenna systems mounted on pickup trucks. This allowed each radio-marked grouse within radio range of the lek to be identified as to whether or not it was recorded by the first observer during the lek count.

Survival. − We will use the known-fate model in Program MARK with the following covariates to estimate survival: year, sex, age, and season. Assumptions for survival estimates from radio telemetry include: the marked sample is randomly selected from the population; marked animals are independent; marking does not influence survival; and when fate is unknown (censored) known survival time is assumed to be independent of the animal's actual fate.

We will use Akaike's Information Criterion (AIC) for model selection and all parameter estimates will be generated using model averaging based on AIC weights (Burnham and Anderson 1998, Murray 2006).

Field Study

Trapping and marking. − Sage-Grouse were captured using spotlighting (Wakkinen et al. 1992, Connelly et al. 2003) between late January and early March in 2007, 2008, and 2009. We attempted to capture and fit birds with radio transmitters relatively early in the winter to minimize their potential association with a particular lek as the mark-resight method we used to estimate detection probabilities relies on the assumption that samples (marked individuals) are randomly selected. We expected that random samples would help facilitate the search for new leks using radio telemetry, where selecting a sample of individuals that may have strong ties to a known lek would not likely lead to the discovery of new leks. All captured birds were classified by gender and age using wing characteristics (Dalke et al. 1963, Beck et al. 1975), and females were weighed.

In 2007, we began trapping on 18 January and continued through 15 March, at which time 32 male and 28 female sage grouse had been captured. In 2008, trapping occurred between 30 January and 15 March, during which 44 male and 17 female sage grouse were captured. In 2009, trapping began on 21 January and continued through 13 March, resulting in 47 males and 39 females captured. All captured birds were fitted with 16.5 g necklace style radio transmitters

(Advanced Telemetry Systems, Isanti, Minnesota) and received a numbered aluminum leg band. In 2007, birds also received three 14 mm, colored flat-band, wrap-around style leg bands (A. C. Hughes, Middlesex, United Kingdom) in a unique combination.

Attendance Probability and Sightability Estimation. − Approximately 5 days just prior to beginning lek routes (approximately 15-20 March), we spent the early morning hours searching for previously unrecorded lek locations. We visited all suspected breeding habitat within the study area and listened for sounds of strutting males as per Connelly et al. (2003), and located radioed males. We attempted to maximize the number of leks in our routes in an effort to increase our resightings (sample size), which should improve the precision of parameter estimates (Pollock et al. 1990).

Lek routes were conducted from 20 March to 16 May in 2007, from 17 March to 21 May in 2008, and from 16 March to 27 May in 2009. Leks were counted from 0.5 hours before to 1 hour after sunrise when weather conditions were clear to partly cloudy and there was little to no wind (Connelly et al. 2003). In this manner, it was possible to visit 2 to 3 leks per day, allowing 15 minutes at each lek, and approximately 15 minutes to travel between leks. We included all leks in the vicinity currently included in the Idaho Department of Fish and Game (Department) standard lek route, as well as any additional ones we discovered. The number of leks in our routes ranged from 9 in 2007 to 17 in 2009. We grouped the leks in our route into twos or threes based on their proximity to each other and all leks within a group were visited on the same day. We defined a resighting occasion as the time it took for all leks within our study area to be visited exactly once and we completed a total of 8 resighting occasions in 2007, 9 in 2008, and 11 in 2009.

Lek routes were conducted by 2 researchers, each equipped with a telemetry receiver and spotting scope. Both individuals approached each lek at approximately a 90-degree arc from each other, being extremely cautious not to flush birds. Upon arriving at a predetermined position that allowed good visibility of the entire lek, the primary observer counted total number of male and female sage-grouse attending according to the protocol by Connelly et al. (2003). Both researchers then scanned through the list of frequencies of radioed birds using hand-held three-element, null-peak YAGI antenna systems, noting signals strong enough and in the general direction that would indicate positive lek attendance. We used predefined compass bearings to delineate the "edge" of the lek relative to each observation point for all leks. Items recorded included date, weather conditions, starting time, observer location, and observer's name for each count, and number and frequency of radioed birds determined to be on the lek during a lek count. If direction and signal strength from both positions indicated the bird was likely on the lek, the bird was assumed to be attending. Researchers then moved to the next lek, repeating the previous steps.

After 2 years of pilot study, we did not feel confident with the assumption that all birds attending a lek were seen by the lek route observer, and more importantly, that radio-marked birds with signals detected on the lek were absolutely included in the count. As a result, we felt it necessary to add a detection probability or "sightability" component to our overall study.

For each day we conducted lek routes, additional personnel made observations at 1 or 2 leks from a stationary blind. We position blinds on 3 leks in 2007, on 4 in 2008, and on 6 in 2009. These leks represented the range of sizes of leks and cover types for this study area. We positioned blinds within 20 m of the edge of the leks within sagebrush to minimize potential effects the introduced structure may have on the birds' behavior. We set up blinds no less than 1 week from the day of the first lek route to allow birds adequate time to adjust to the structure. On the day of the lek count, a single observer entered the blind 2 hours before sunrise to minimize the possibility of flushing birds. When visibility was sufficient to see birds on the lek, but no later than 0.5 hours before sunrise, each observer conducted a count of adult males, yearling males, and females visiting the lek. After each count, the observer then scanned the frequencies with a receiver and a small hand held two-element antenna for radioed birds on the lek. We recorded the following data: observer's name, date, general weather conditions, time of each count, number of each segment of the population visually counted, number of displaying males, and number and frequency of radios heard. The observer repeated counts and frequency scans every 15 minutes. The observer continued collecting data from the blind until all of the birds left the lek.

Using mark-resight techniques, we used the Recaptures Only model in Program MARK to estimate the probability of attending a lek for sage-grouse from a relatively discrete breeding population. Program MARK allows the modeling of detection probabilities with group-specific, time-specific, and individual-specific covariates, which can greatly improve the precision of the estimates (White 2005). The data requirements for this model include the detection histories for each marked (radioed) bird, which is simply a record of detected or not detected for each bird during each lek route (resighting occasion). The covariates we plan to use in the model include: sex, age, date, time of day, size of lek, moon phase, weather conditions at the time of observations and during the previous winter, and year.

We used Akaike's Information Criterion (AIC) for model selection and all parameter estimates were generated using model averaging based on AIC weights (Burnham and Anderson 1998). We used the generalized mark-resight population size estimation method described by Bowden and Kuffeld (1995) to estimate abundance and the corresponding variance for all segments of the population.

We will determine the sightability bias of sage-grouse that attend leks and use multivariate regression to evaluate effects of biologically relevant variables and observer bias on the sightability of male and female sage-grouse from a lek route census. We will also develop models of visibility bias similar to those used for helicopter surveys of ungulates (Samuel et al. 1987, Bodie et al. 1995). We will use counts conducted from blinds located at the edge of leks to compare counts conducted during lek routes under the assumption that counts from the blind will include 100% of attending grouse. The variables we plan to use in the sightability model are: total number of males attending, number of yearling males attending, density and height of cover, number of females attending, time of day relative to sunrise, weather, distance to center of lek, and size (area) of lek.

Nest Survival. − For precocial species, nest survival is the probability that ≥1 egg hatches from a given nest, and daily nest survival rate is defined as the probability that a nest will survive a 24 hour period (Dinsmore et al. 2002). These metrics are key components of avian demographics and are used to drive and evaluate management strategies (Jehle et al. 2004, Stanley 2004).

Females with radio transmitters were located daily from the beginning of April to the end of June, or until nesting efforts ceased. We continued to monitor hens with failed nests in an effort to detect secondary nest attempts. Once hens were noted in the same location on 3 consecutive days, we attempted to get a visual confirmation of nesting without flushing the bird. Nests were then monitored daily from a single location approximately 30 m from the site to minimize disturbance (Schroeder 1997). If the female was not located on the nest, we approached the nest to determine fate. We attempted to determine fate, number of chicks hatched, or cause of failure from shell/egg remnants and other cues at the nest (Wallestad and Pyrah 1974, Martin and Guepel 1993).

We plan to use the nest survival model in Program MARK (White and Burnham 1999) to estimate daily survival and determine which biologically relevant factors affect variability of nest survival. The nest survival model in Program MARK expands on the daily nest survival model described by Bart and Robson (1982), allowing the use of individual, group-specific, and time-specific covariates (Dinsmore et al. 2002). This method does not require the restrictive assumptions such as constant survival or that failure occurred at the midpoint of an interval, as do models such as Mayfield's estimator (Jehle et al. 2004). The assumptions of this model are: (1) nests are correctly aged when first discovered; (2) nest fates are determined accurately; (3) nest monitoring does not affect survival; (4) nest fates are independent; and (5) daily nest survival rates are homogeneous.

The minimum data required for the nest survival model include: the day the nest was found; the last day the nest was checked alive; the last day the nest was checked; and the fate of the nest. We will also estimate the day incubation began from monitoring of daily hen activity so that we can include nest age as a covariate. Other covariates that will be included are: year, date of nest initiation, hen age, condition of hen at time of capture (hen weight), average temperature, and daily precipitation.

We will use Akaike's Information Criterion (AIC) for model selection and all parameter estimates will be generated using model averaging based on AIC weights (Burnham and Anderson 1998).

Juvenile and Adult survival. − We attempted to locate all radioed birds with radio telemetry at a minimum of once per week throughout the year. UTM coordinates of each bird was recorded within 50-100 m using a hand-held GPS unit to minimize disturbance. The transmitters we used were equipped with a mortality sensor and all deceased birds were examined in the field in an attempt to determine source of mortality from the remains.

We will use the known-fate model in Program MARK with the following covariates to estimate survival: year, sex, age, and season. Assumptions for survival estimates from radio telemetry include: the marked sample is randomly selected from the population; marked animals are independent; marking does not influence survival; and when fate is unknown (censored) known survival time is assumed to be independent of the animal's actual fate.

We will use Akaike's Information Criterion (AIC) for model selection and all parameter estimates will be generated using model averaging based on AIC weights (Burnham and Anderson 1998, Murray 2006).

Sex Ratio. − Because hens only visit leks once or twice to choose a mate and copulate, it was not possible to detect enough radioed hens on leks to estimate their attendance probability or their population size. We will, however, be able to estimate hen population size from the estimated male population size if we can determine the sex ratio of the population.

During the trapping phase of the 2009 field season, we opportunistically collected fecal samples from each bird that defecated during handling to determine if sex ratios can be accurately estimated from fecal DNA. Each sample was labeled with bird ID and sex and stored in 2 ml vials in 90% ethanol solution until DNA extraction. DNA was extracted in a room dedicated to low quantity DNA samples using the Qiagen tissue kit. All samples were amplified 3 times using 3 sets of sex ID primers to determine which primer set or groups of primers would give the highest success and most accurate results. We used one multiplex PCR that contained fluorescently-labeled sex ID primers 2550F/2718R (Fridolfsson and Ellegren 1999) and P8/P2 (Griffiths et al. 1998) and a singleplex PCR using primers 1237 L/1272 H (Kahn et al. 1998). Primers 2550F/2718R amplify a portion of the CHD gene flanking an intron where, for most species, the sex-dependent fragment (on the W chromosome) found only in females is shorter than the sex-independent fragment (on the Z chromosome) found in both males and females (450bp vs. 600bp). Primers P8/P2 and 1237L/1272H amplify a different intron of the CHD gene than 2550F/2718R, and the sex-dependent fragment is longer than the sex-independent fragment for most birds (290 – Z, 345 - W and 240 – Z, 270 – W, respectively). PCR products were separated with an ABI3130xl Genetic Sequencer (Applied Biosystems, Foster City, CA, USA) and visualized with the accompanying Genemapper software.

We estimated PCR success rates as the number of successful PCRs for each primer set out of the total number conducted for each primer set (51 for females and 106 for males). We estimated error rates for the failure to amplify the W chromosome for females as number of times only the Z chromosome amplified divided by number of successful amplifications. We estimated the error rates for males as the number of times the W chromosome amplified for males out of the total number of successful PCRS. Based on the success rates and error rates, we estimate the number of PCRs that will be needed to obtain error rates less than 2%.

We also estimated the sex ratio of the original samples, the sex ratio estimated from the number samples that worked with the original 3 replicates of PCR and the predicted sex ratio if unresolved male samples or female samples had been amplified up to six times.

PRELIMINARY RESULTS

Pilot Study

Attendance Probability and Sightability Estimation. – In 2005, adult males attended leks most frequently (96%) and were counted on the leks most frequently (84% of time present within the vicinity of a lek; Table 1), yearling males less frequently (74% attendance and 69% detection when attending), and females rarely (15% attendance and 8% detection). Combined adult and yearling males attended during 88% of counts and were counted (observed on the lek) 79% of the time that they were present within the vicinity of a lek. Because our ultimate goal is to estimate the actual number of sage-grouse present in the entire vicinity sampled by the lek route, the most meaningful measure is the probability of detecting birds within the population. These rates are somewhat lower than attendance rates for males (68%, SE = 4.16%; Table 1) and extremely low for females (1.3%, SE = 0.90%). The same radio-marked males were seen repeatedly at leks over the course of the 7 weekly counts, but radio-marked females were only seen a single time at any lek.

Maximum number greater sage-grouse counted during 3 scans of birds at 5 leks during 7 weekly surveys from 25 March to 3 May 2005 ranged from 53-156 males (combined adults and yearlings) and 1-8 females (Table 2). Based on the 68% (SE = 4.2%) probability of detecting radio-marked males, this outcome implies that there were a total of 233 males attending the leks on Brown's Bench during the breeding season in 2005.

Lek attendance data for 2006 have not been completely analyzed. However, it appears that peak male counts occurred much earlier in 2006 compared to 2005 (Figure 2). Moreover, the highest count of yearling males in 2006 was only 4 birds and in 2006 yearlings had a higher probability of attending leks than adults

Survival. − Survival data from 2005-2006 have not yet been analyzed.

Field Study

Attendance Probability and Sightability Estimation. − Of the 33 male sage-grouse trapped in 2007 or surviving from previous years, 5 left the study area and 5 died during or before the breeding season.

Of the 44 males captured in 2008, 29 were used for the attendance probability analysis. Of the birds censored, 3 slipped their collars, 9 died, 2 left the study area, and 1 either died or slipped its collar. Five birds surviving from 2007 with live transmitters were also included for a total sample size of 34 (30 adults and 4 yearlings).

Twenty-five of the 49 males captured in 2009 were used for the attendance analysis. Of those censored, 9 died, 8 left the study area, 1 had a faulty transmitter, and 6 either left or had faulty equipment. Thirteen birds from 2008 remained in the study, for a total sample size of 38 (36 adults and 2 yearlings).

In 2007 we conducted a total of 9 complete lek routes on the 9 leks in our study area. Additionally, we collected data from blinds on 3 of the leks for a total of 21 blind-mornings during our lek routes. We detected a total of 91 resightings of 16 adult males, 14 resightings of 3 yearling males, and a total of 8 resightings of 7 females on our study leks. The probability of detecting a marked bird on a lek in our study during the 9-week duration in 2007 was 0.63 (SD=0.23, n=17) for adult males, 0.52 (SD=0.46, n=3) for yearling males, and 0.09 (SD=0.08, n=14) for females.

In 2008 we conducted a total of 9 complete lek routes on 14 leks and collected attendance data from blinds on 4 of the leks for a total of 31 blind-mornings. We detected a total of 23 resightings of 4 yearling males and 206 resightings of 30 adult males. The probability of detecting a marked bird on our study area during the 9-week study in 2008 was 0.76 (SD=0.23, n=30) for adult males and 0.64 (SD=0.29, n=4) for yearling males.

We conducted a total of 11 complete lek routes on 17 leks and collected data from blinds on 6 distinct leks for a total of 43 blind mornings in 2009. We detected a total of 7 resightings and 271 resightings from 2 yearling and 36 adult males respectively. The probability of detecting a marked male during the 11-week study in 2009 was 0.68 (SD=0.18, n=36) for adults and 0.32 (SD=0.06, n=2) for yearlings. For the same 9-week duration used in 2007 and 2008, the detection probability in 2009 was 0.82 (SD=0.21) for adults and 0.39 (SD=0.08) for yearlings.

Peak male counts for 2008 followed a trend similar to that of 2006 and 2007 (Figure 3) with male sage-grouse peak counts occurring relatively early in the season. Counts in 2005 followed a pattern considered normal for sage-grouse with numbers of males peaking in late April.

To date, we have modeled the 2006, 2007, and 2008 data for attendance probability. The bootstrap goodness-of-fit test in program Mark showed our global model was a good fit to the data with only slight overdispersion (c-hat = 1.23). The top model chosen by AIC model selection procedures was an additive model with the variables of age, year, and a quadratic time trend (Table 3). The next most parsimonious model resulted in Δ AIC value of 0.274 was similar to the previous model with an interaction term between year and the quadratic time trend.

A graphical representation of the predicted attendance probabilities from the models averaged based on AIC weights shows a general increase in attendance probability up to the 4th or 5th occasion (second to third week of April), followed by a decrease until the final occasion (Fig. 3). The yearling pattern of attendance appears to mimic that of the adults, but with approximately 10% lower probability of attending. An annual trend is also apparent from this figure, with a gradual increase in attendance from 2006 through 2008. Figure 4 shows the 95% confidence limits for these estimates.

Modeling of the 2007-2009 data is nearly complete. These data were separated from the 2005 and 2006 data because the number of resighting occasions was extended to 10. Preliminary results from this analysis indicate a 4th order polynomial better fits the within-season variation of lek attendance than a quadratic term. The result is a pattern where “peak” attendance is reached

near the end of March, slowly decreases through early May, and then precipitously drops to zero over the following 2-3 weeks.

Analyses of these data also indicates that individual males that did not survive the breeding season (March 15 – June 31), generally attended leks at a higher rate than those that did survive. These results support the theory that lekking activity for males is costly.

During lek routes, we detected on average 90.88% (SD=22.99) of the total birds counted from the adjacent blind in 2007, 87.69% (SD=24.38) in 2008, and 78.8% (SD=28.99) in 2009. The detection rates in 2009 were driven lower primarily due to two leks that were located on hill tops with birds extending down the back side of the hills relative to the observation point.

Nest Success. − We monitored the daily movements of 17 females throughout the nesting season in 2007. We detected a total of 15 nests initiated by these hens, which included a second nest attempt from a single adult. Only 2 of the 15 nests (13.3%) successfully hatched. In 2008 we monitored 14 hens through the nesting season. We detected a total of 11 nests initiated, which again included a second nest attempt from a single adult hen. Only 3 of the 11 nests (27.3%) successfully hatched in 2008. In 2009, we detected 22 first nest attempts from 23 hens monitored. Seven of these hens also attempted a second nest, for a total of 29 nests initiated. Eight of these 29 nests produced a successful hatch, resulting in a nest success of 27.6%.

Sex ratio – Fecal samples were obtained from 17 female and 36 male sage-grouse. These data are being analyzed to determine PCR success rates, error rates, and sex ID success rates. In addition, we have collected fecal samples from 11 females and 7 males with varying exposure time in the field ranging from 1-124 days to determine if success and error rates for sex ID from fecal droppings change in response to exposure time.

FUTURE WORK

Final analysis of the lek attendance modeling and sex ID from fecal droppings is complete and these results are currently being interpreted for inclusion in separate manuscripts to be submitted for publication in peer-reviewed, scientific journals. Analysis of survival data, nest success, and sightability of sage-grouse on leks is planned for this fall.

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Table 1. Radio-marked male and female greater sage-grouse captured and relocated in vicinity of leks on Brown's Bench, Idaho, which were observed during 7 standard lek counts conducted from 25 March to 3 May 2005.

Demographic information	Adult Males	Yearling Males	Total Males	Hens
Radio-tagged Individuals	12	8	20	23
Radio-weeks available during lek counts	79	47	126	157
Radio-weeks available on/near leks	76	35	111	24
No. seen during lek counts	62	24	86	2
Overall attendance rate	96.2%	74.5%	88.1%	15.3%
Probability of detecting attending birds	83.8%	68.6%	78.9%	8.3%
Probability of detecting birds in population (SE)	78.5%	51.1%	68.2% (4.16%)	1.27% (0.90%)
Counts of all birds (w/ and w/o radios) at leks:				
Maximum	152	7	156	8
Mean	148.4	3.8	152.2	3.43
Projected number of birds in Population:				
From maximum count			233	628
From average count			196	1646

Table 2. Maximum number of male and female greater sage-grouse counted during 3 scans of birds present at 5 leks during 7 weekly surveys of Brown's Bench Lek Route, Idaho, from 25 March to 3 May 2005.

	Maximum Males Counted in 3 Scans						
Leks	3/25	3/30	4/1	4/5	4/15	4/26	5/3
TWOSEC	1	32	52	52	47	56	34
WALTS	7	3	4	23	24	29	18
SADDLE	40	18	31	42	46	50	41
LUCUS	4	0	5	9	5	8	8
TROUGH	16	0	3	8	13	13	14
Total Males	68	53	95	134	135	156	115

	Maximum Females Counted in 3 Scans						
Leks	3/25	3/30	4/1	4/5	4/15	4/26	5/3
TWOSEC	0	0	0	1	0	0	0
WALTS	2	0	0	1	0	0	1
SADDLE	3	1	1	0	1	4	0
LUCUS	1	0	0	0	0	0	0
TROUGH	2	0	0	0	2	1	3
Total Females	8	1	1	2	3	5	4

Table 3. Top 7 models selected by AIC for predicting attendance probabilities of male sage-grouse on leks in 2006, 2007, and 2008.

Attendance Probability Model	AIC	Delta AIC	AIC Weights	Number of Parameters
Age + Year + t^2	574.872	0	0.322	6
Year * t^2 + Age	575.146	0.274	0.281	10
Age * t^2 + Year	577.071	2.199	0.107	8
Year + t^2	577.424	2.552	0.090	5
Year * t^2	577.748	2.876	0.077	9
Age * year + t^2	578.921	4.049	0.043	8
Age + year + t	579.563	4.691	0.031	12

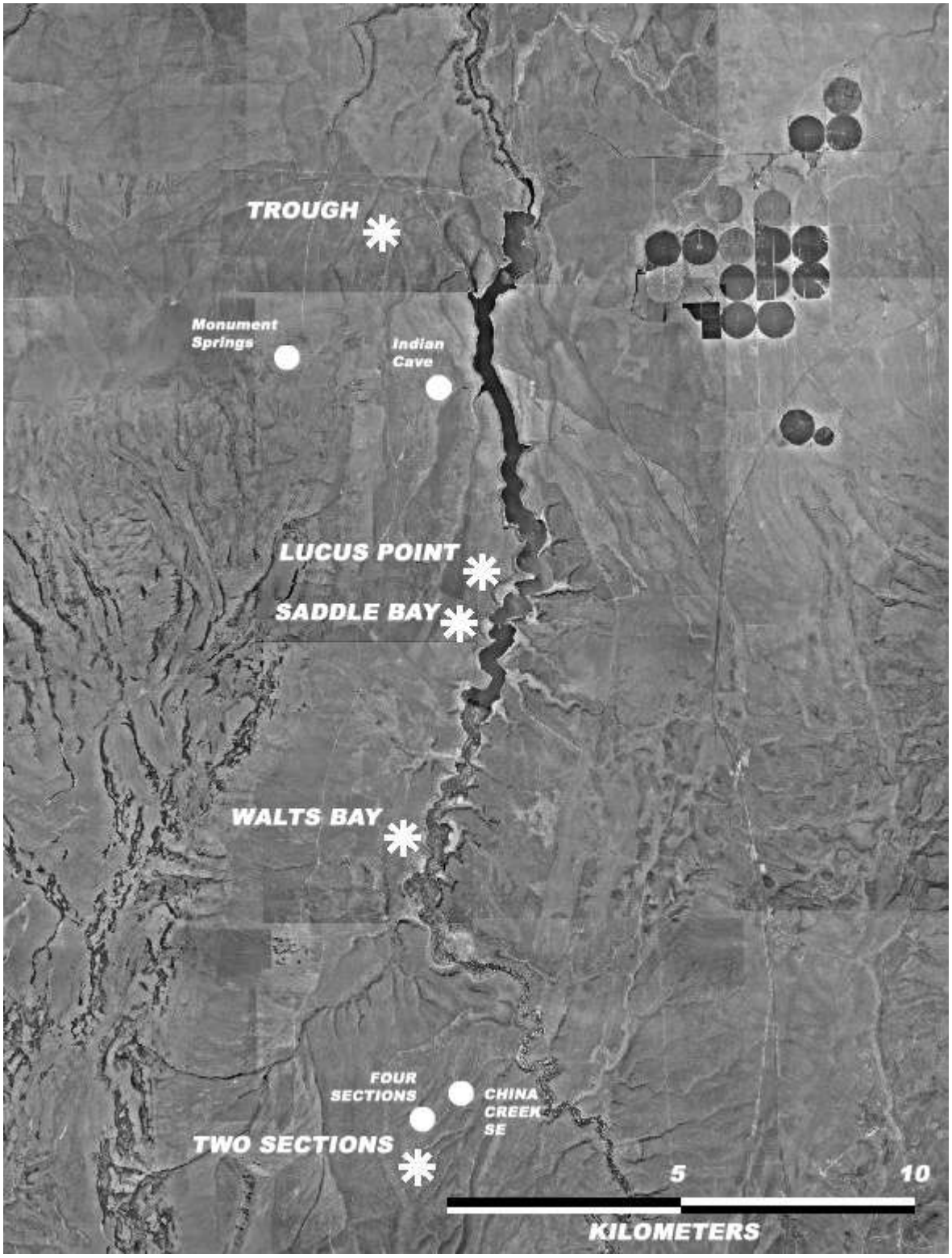

Figure 1. Brown's Bench study area with leks. Asterisks indicate leks used in lek count.

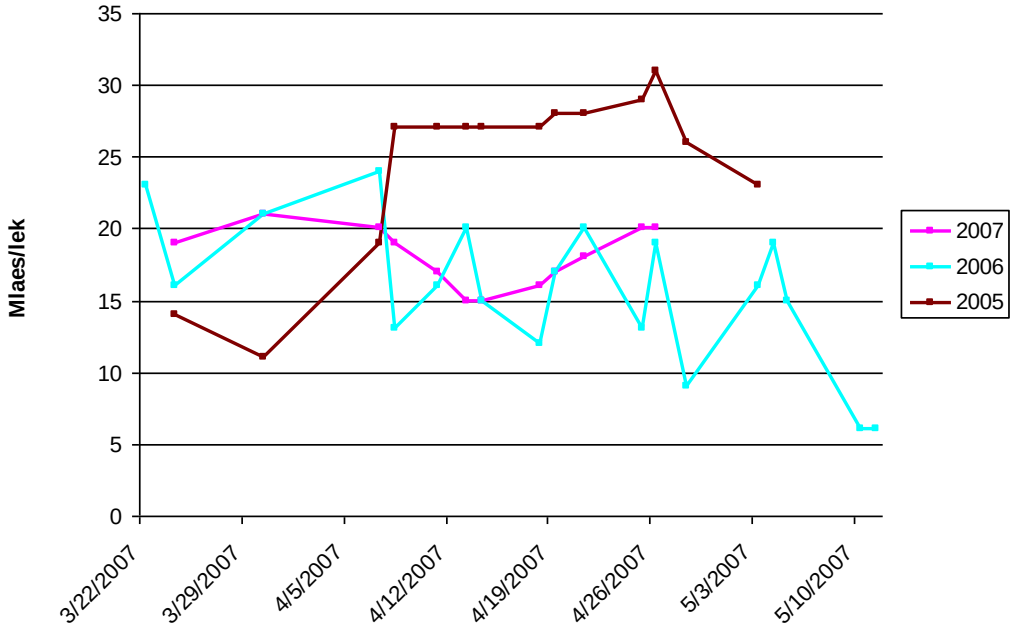

Figure 2. Lek attendance patterns on Browns Bench for 2005 through 2007.

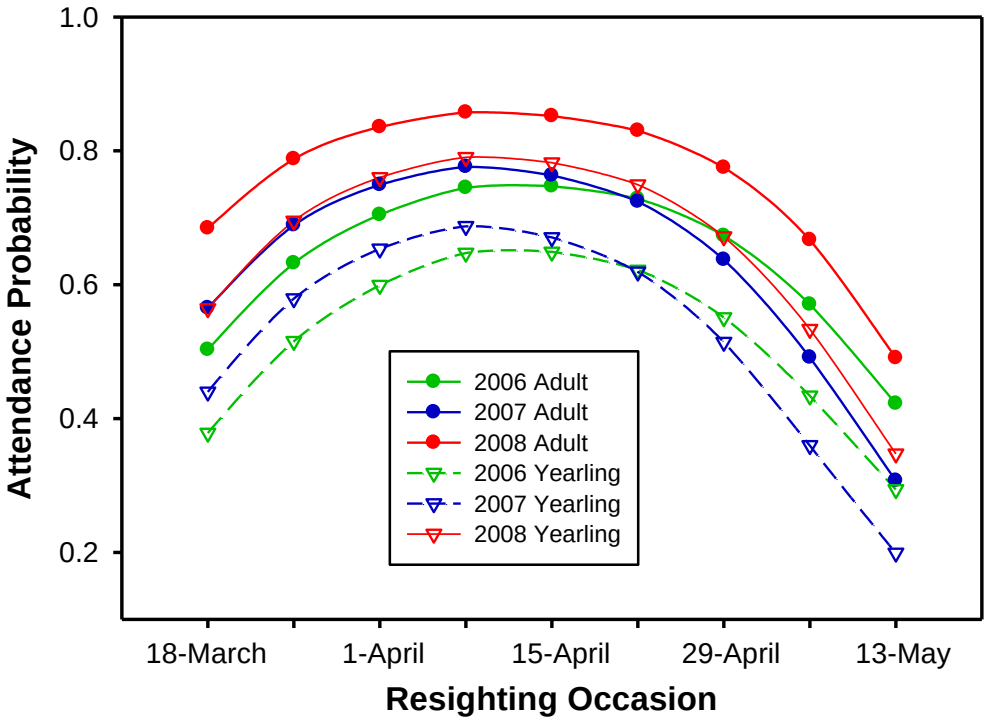

Figure 3. Predicted attendance probabilities from the models averaged based on AIC weights for adult and yearling sage-grouse in 2006, 2007, and 2008.

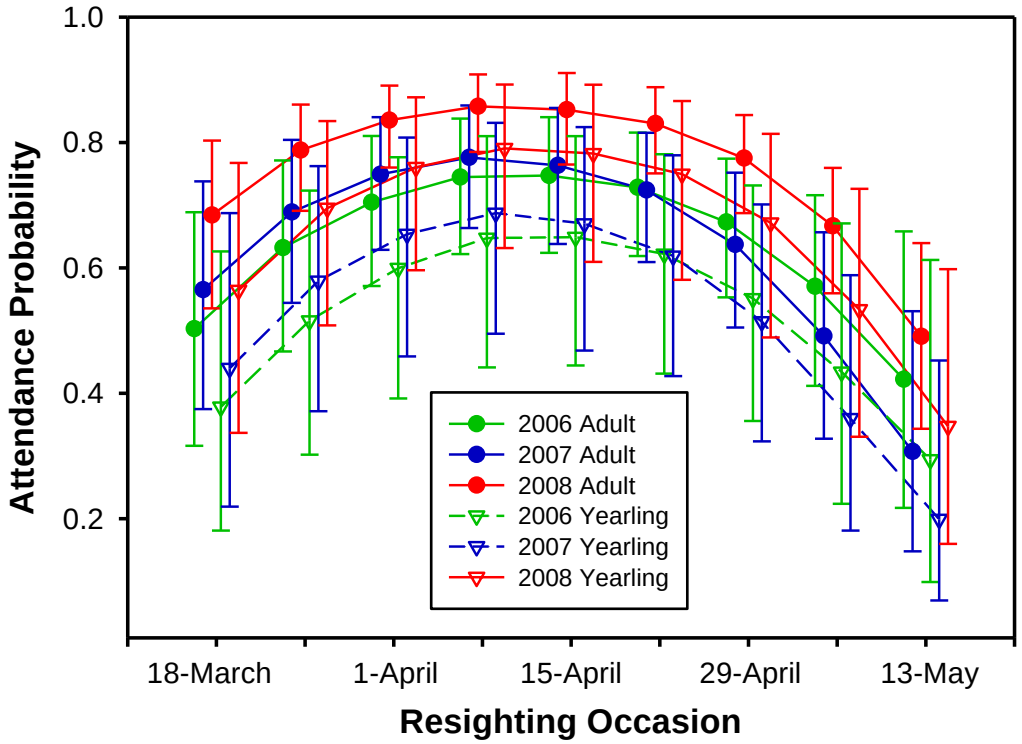

Figure 4. Predicted attendance probabilities with 95% confidence bands from the models averaged based on AIC weights for adult and yearling sage-grouse in 2006, 2007, and 2008.

STUDY I; PART II: IMPACTS OF ELEVATED INFRASTRUCTURE ON GREATER SAGE-GROUSE IN IDAHO: COLLISION, MITIGATION, AND SPATIAL ECOLOGY

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ABSTRACT We used probability-sampling methods to estimate collision rates of greater sage-grouse (hereafter sage-grouse) with fences in breeding areas across southern Idaho, and conducted a field experiment to test the effectiveness of fence marking as a mitigation method to reduce fence collision rates during the spring of 2010. We randomly selected 1x1 km sampling units within approximately 2.5 km of active leks monitored by Idaho Department of Fish and Game, used one-stage cluster sampling to estimate fence collision rates within each lek route, and used stratified cluster sampling to estimate global collision rates over all study areas. We sampled 63.3 km of fence from March – May 2010, of which approximately 50.3 km was sampled on 2 occasions. During fence collision surveys we located 50 avian collision sites, 36 of which were known sage-grouse. Twelve of these collision sites (8 sage-grouse) were found opportunistically outside of randomly selected sampling areas, and therefore were not used to generate collision rate estimates. Estimated collision rates varied spatially between lek routes (0-1.904 all species; 0-1.428 known sage-grouse), and global collision rates were lower during the second round of sampling (0.162-0.486 all species; 0.099-0.356 known sage-grouse). Experimental fence marking studies were conducted using a complete block repeated measures design, with 8 different sites as the blocking variable and 5 sampling periods from March – May 2010. Pooling data over all sites and sampling rounds resulted in uncorrected collision rates approximately 5.74 times higher in unmarked control than marked treatment fence segments. Repeated sampling showed a peak of number of collisions found in late March and early April, with a decline for the remainder of the sampling periods. Future work will evaluate spatial and temporal variation in fence marking treatment effects using repeated measures mixed effects modeling, and collision risk models will model the influence of site and landscape features on collision rates at multiple spatial scales across all study areas. These analyses will be completed as part of a M.S. thesis in wildlife resources, and will be completed by May 2011.

KEY WORDS *Centrocercus urophasianus*, fence collisions, fence marking, Idaho, sage-grouse

Collision mortality is a widespread and relatively common phenomenon among European grouse species (Bevanger 1990, Catt et al. 1994, Baines and Summers 1997, Moss et al. 2000). Collision with deer fences is a major source of mortality for capercaillie (*Tetrao urogallus*) in Scotland (Catt et al. 1994, Baines and Summers 1997, Moss et al. 2000), and may be contributing to population declines of that species (Moss 2001). In Scotland red grouse (*Lagopus lagopus scoticus*) and black grouse (*Tetrao tetrix*) appear to be more common collision

victims than capercaillie, however, population consequences for these species are not believed to be as severe (Baines and Summers 1997, Baines and Andrew 2003). Similarly, Norwegian studies have found collision is a common source of mortality for capercaillie, black grouse, and ptarmigan (*Lagopus* spp.) in that country (Bevanger 1990, Bevanger and Brøseth 2000, Bevanger and Brøseth 2004), and collision mortality may approach harvest mortality in some areas (Bevanger 1995b). Moreover grouse may be morphologically predisposed to collision mortality due to their high wing loading and heavy body weight (Janss 2000).

Research concerning the relative extent and impacts of collision mortality on North American grouse are limited. Wolfe et al. (2007) studied mortality patterns of lesser prairie-chickens (*Tympanuchus pallidicinctus*) in Oklahoma and New Mexico, and found 39.8% of mortality in Oklahoma was caused by fence collision. Patten et al. (2005) concluded fragmentation caused by fences, power lines, and roads in Oklahoma have resulted in higher mortality rates for female lesser prairie-chickens in Oklahoma than New Mexico. The increased female mortality has resulted in more variable nesting strategies, leading to increased vulnerability to stochastic population fluctuations (Patten et al. 2005). Additionally, Beck et al. (2006) found 33% of the juvenile mortality of greater sage-grouse (*Centrocercus urophasianus*) (hereafter sage-grouse) on an Idaho study area was caused by collision with power lines.

Existing research into factors affecting grouse collision mortality suggests collision may be influenced by biological, landscape, and habitat features (Bevanger 1994), however, results often vary by species or region (Baines and Summers 1997). For example, Bevanger (1995a) suggested male capercaillie and black grouse have a higher probability of collision than females due to their increased size, however, Wolfe et al. (2007) found female lesser prairie chickens were more susceptible to collision mortality due to their increased movement patterns during the breeding season. In Norway ptarmigan collision mortality appears to peak in winter (Bevanger 1995a, Bevanger 1995b, Bevanger and Brøseth 2004) and black grouse in autumn (Bevanger 1995b), but varied results exist for capercaillie, with studies documenting peaks in both winter (Bevanger 1995a) and spring (Bevanger 1995b). In addition to possible influences of season and sex, research has suggested grouse collision mortality may be influenced by the structure, type, and height of surrounding vegetation (Bevanger 1990, Catt et al. 1994, Baines and Summers 1997, Bevanger and Brøseth 2004), topography (Bevanger 1990), and local bird densities (Baines and Andrew 2003, Bevanger and Brøseth 2004). Furthermore, some authors have found evidence for collision "hot spots" where mortality is concentrated (Bevanger and Brøseth 2000, Baines and Andrew 2003), while others have not found evidence for spatially aggregated collision sites (Baines and Summers 1997).

In a review of the interactions of birds with utility infrastructure, Bevanger (1994) suggested that factors influencing collision can be classified as biological, topographical, meteorological, or technical. Biological factors influencing collision risk include the previously mentioned high wing loading and heavy body weight (Bevanger 1998, Janss 2000), as well as factors such as vision (Bevanger 1994), crepuscular or nocturnal activity patterns (Avery et al. 1978), local or migratory movement patterns (Meyer 1978, Malcom 1982), bird densities (Baines and Andrew 2003, Bevanger and Brøseth 2004), and vegetation characteristics (Bevanger 1990, Catt et al.

1994, Baines and Summers 1997). Topographical features influencing collision rates may include mountain valleys and coastlines (Cooper and Day 1998), as well as sloping terrain, ridges (Bevanger 1990), and other features that influence local or migratory movement corridors (Bevanger 1994). Meteorological conditions include fog, precipitation, strong winds, and other weather conditions affecting bird flight intensity and behavior (Bevanger 1994). There is ample evidence for changes in flight behavior due to weather conditions (Meyer 1978, James and Haak 1978, Brown and Drewien 1995, Savereno et al. 1996); however, evidence for increased collision risk due to weather conditions is mostly anecdotal. Brown and Drewien (1995) did find increased collision rates during periods of strong gusty winds; however, they found no relationship between collision rates and precipitation. Technical designs related to infrastructure construction may also influence collision risk (Bevanger 1994), but several authors have suggested there is not strong empirical evidence that infrastructure design affects collision risk (Bevanger and Brøseth 2000, Janss 2000, Bevanger and Brøseth 2004).

Recent concerns regarding impacts of elevated infrastructure on greater sage-grouse in Idaho (Idaho Sage-Grouse Advisory Committee 2006) have highlighted the lack of empirical data concerning collision frequency and grouse in North America. The spatial extent of fences and other elevated structures has increased dramatically in sagebrush habitats during the last 50 years (Connelly et al. 2000, Connelly et al. 2004), and their potential impact on sage-grouse has not gone unnoticed (Braun 1998, Connelly et al. 2000, Connelly et al. 2004). Furthermore, fence marking methods to reduce avian collision have been developed and applied for lesser prairie chickens in Oklahoma (Wolfe et al. 2009), but the effectiveness of this mitigation method has not been experimentally tested. For these reasons, this research was pursued with the following objectives: 1) estimate collision rates of greater sage-grouse with fences on study areas in greater sage-grouse breeding habitats, 2) Evaluate the effectiveness of fence marking as a mitigation method to reduce avian fence collision rates in sage-grouse breeding areas.

STUDY AREA

Fence Sampling and Collision Estimation

We conducted fence collision surveys in 14 greater sage-grouse breeding habitats across 4 large geographic regions of southern Idaho in 2010. We sampled fences within 2 lek routes in the East Jarbidge region, the Antelope Pocket and Browns Bench routes, respectively. In the northern Magic Valley region we sampled fence in 3 breeding habitats, the North Shoshone, Picabo Hills, and Paddelford Flats lek routes. In the Big Desert region south of Arco we sampled fence in 3 lek routes, Big Desert routes 1 and 3, and the Fingers Butte route. Lastly we sampled fence in 6 lek routes in the Upper Snake region of southeast Idaho, the Crooked Creek, Lidy, Table Butte, Medicine Lodge, Plano, and Red Road routes. Elevations on sampling areas ranged from approximately 1450 m on the Browns Bench lek route to approximately 2000 m on the northern portions of the Red Road and Medicine Lodge lek routes. Habitat types on the breeding season sampling areas varied considerably, and included large stands of big (*Artemisia tridentata*), little (*A. arbuscula*) and black (*A. nova*), 3-tip (*A. tripartita*), and mixed sagebrush types, as well as large grasslands and large, bare pasture and burned areas. Non-native grasses such as cheatgrass (*Bromus tectorum*) and crested wheatgrass (*Agropyron christatum*) were common on many study

sites. Since there was considerable spatial variation in habitat, these sites were representative of the variety of habitat conditions on southern Idaho rangelands.

Experimental Fence Marking

We replicated our 2010 experimental fence marking study on 8 sage-grouse lek routes across south-central and southeast Idaho. The Browns Bench site is located west of Salmon Falls Creek reservoir in southern Twin Falls county, and is dominated by little sagebrush habitats. Paddelford Flat is south of Idaho Highway 22, bordered on the east by Craters of the Moon National Monument, and dominated by mixtures of 3-tip and big sagebrush. Big Desert lek routes 1 and 3 are south of Arco and southwest of Big Southern Butte, and dominated primarily by mixtures of big and 3-tip sagebrush, with small previously burned grasslands interspersed. The Fingers Butte site is southwest of Quaking Aspen Butte in the Big Desert south of Arco, and dominated by large stands of little and big sagebrush, with large areas of bare ground interspersed. The Crooked Creek drainage is dominated by little and mixed sagebrush types, and bound to the south by Idaho highway 22, and to the north east and west by the Beaverhead Mountains and Caribou Targhee National Forest. The Table Butte site is located just northwest of Table Butte in Jefferson county, and is primarily dominated by big sagebrush. The Lidy lek route is approximately 8 km west of Dubois, Idaho, is bound to the north by the Beaverhead Mountains, and dominated primarily by large stands of little sagebrush. Elevations on experimental marking areas ranged from approximately 1580 m on the Browns Bench site to approximately 1900 m on the Crooked Creek site.

METHODS

Fence Surveys and Collision Estimation

We used a cluster sampling approach to estimate avian fence collision rates in sage-grouse breeding habitat. Specifically, 14 lek routes monitored by Idaho Department of Fish and Game (hereafter IDFG) were selected for inclusion in the study based on accessibility and known breeding bird use. We buffered each lek in the route with ≥ 1 displaying male documented the previous year (2009) by 1.5 km using ArcGIS software. Once each lek route was buffered a 1x1 km spatial grid was superimposed over the buffered leks within each route using ArcGIS software, and the grid cells that intersected with (i.e. contained) BLM pasture boundaries (our surrogate for fence) were used to define the sampling frame. Once the sampling frame was defined for each lek route a stratified cluster sample of 1x1 km grid cells was randomly selected using the Hawthe's Tools (Beyer 2004) extension in ArcGIS. We allocated the sample of 80 grid cells to each stratum in proportion to the number of cells in each stratum. For example, if one strata (lek route) contained 10% of the total number of cells in the sampling frame, then that strata was allocated 8 sample cells (10% of 80). Eighty cells were selected as the overall sample size because we estimated this as the maximum number of cells that could be sampled in a one month period given time and logistical constraints, which was needed to facilitate repeat sampling necessary in this study.

Within each sampling unit (i.e. grid cell) all fence sections (sampling elements) were searched for sage-grouse carcasses or sign, and feather tufts on the barbed wire, using 1-2 searchers (1 on each side of the fence, or 1 searcher sampling each side in turn). During these surveys fence searchers walked approximately 1-3 m on either side of the fence, and monitored the area up to approximately 15 m from each side of the fence for carcasses or collision evidence. Fence segments not previously identified with the BLM pasture boundary layer, as well as all fence types inside our spatial clusters were digitized using handheld GPS units and ArcGIS software. We sampled fence from 15 March – 18 May 2010 on sage-grouse breeding areas.

Statistical estimation of the average number of collisions per linear kilometer of fence (i.e. collision rate) and total number of collisions located on each lek route follows from elementary sampling theory (i.e. Scheaffer et al. 2006). Specifically, estimation of the collision rates ($\bar{x} = \hat{\mu}$) and variance of the rates ($\hat{V}(\bar{x})$) on each study area were calculated as follows:

$$\bar{x} = \frac{\sum_{i=1}^{n} x_i}{\sum_{i=1}^{n} m_i}$$

$$\hat{V}(\bar{x}) = \left(\frac{N-n}{Nn\bar{m}^2}\right) s_r^2$$

Where:

$$s_r^2 = \frac{\sum_{i=1}^{n}(x_i - \bar{x}m_i)^2}{n-1}$$

Notation in these formulas represents the following values: N = the total number of clusters in the population; n = the number of clusters sampled; m_i = the length of fence in kilometers in cluster$i$; $\bar{m}$ = the average length of fence in kilometers per cluster in the sample; and x_i = the number of collision sites located in the ith cluster. It should also be noted that $\bar{m}$ in the variance formula is used instead of the true average cluster size for the population ($\bar{M}$) which is unknown. Furthermore, estimation of the total number of collision sites on each study area ($\hat{\tau}$) and variance of this estimate ($\hat{V}(\hat{\tau})$) were calculated as follows:

$$\hat{\tau} = \frac{N}{n}\sum_{i=1}^{n} x_i$$

$$\hat{V}(\hat{\tau}) = N^2\left(\frac{N-n}{Nn}\right) s_t^2$$

Where:

$$s_t^2 = \frac{\sum_{i=1}^{n}(x_i - \bar{x}_t)^2}{n-1}$$

We emphasize that $\bar{x}_t$ (the average number of collision sites located in the n clusters sampled) is not the same as $\bar{x}$ (the average number of collision sites located per linear kilometer of fence) as calculated in equation one.

A collision in this study was defined as detection of a whole carcass or a feather pile (> 5 feathers) within 15 m of the fence, or detection of feather tufts stuck in the barbed-wire fence. Despite this definition of a collision we were cautious when only feather sign was detected, and if a likely raptor plucking post was present we were conservative and did not classify these sites as collision locations. For example, plucking posts were common in some areas (mostly for passerine species) and were usually located at large wooden fence-posts, with the resulting feather piles scattered from the base of the post in the prevailing wind direction. In contrast, sites deemed collision locations based solely on feather-pile evidence commonly contained large numbers of feathers scattered in the prevailing wind direction from under the fence itself, or very close to the fence. Given this definition of a fence collision the only victims not accounted for were birds flying into fences and leaving no feathers either in the fence or on the ground, and no carcass, or victims whose evidence was removed prior to sampling. During our searches, feather tufts and piles were counted as collisions with no knowledge of the fate of the collision victim. Therefore our estimates are of the number of collision sites present at the time of the survey, and not of collision mortalities, as we had no way to assess the crippling bias caused by individual birds flying into fences and dying at a later time or in a different area (Bevanger 1999). Furthermore, this estimate is biased low due to a detection probability for collision evidence in sagebrush-steppe habitats that is influenced by vegetation characteristics at the site. The assumption that collision brings no benefit and is always associated with negative effects on the individual victims is inherent in this methodology. Thus, the relationship between the collision itself and the extent of the negative effects on the individual birds was left unstudied, as this is extremely difficult to accurately assess (Bevanger 1999).

Statistical estimation of global collision rates and total numbers of collision sites in the breeding areas once again followed from elementary sampling theory (Scheaffer et al. 2006). Estimates of the collision rates ($\bar{x} = \hat{\mu}$) and variance of the rates ($\hat{V}(\bar{x})$), as well as the total number of collision sites ($\hat{\tau}$) and variance of this estimate ($\hat{V}(\hat{\tau})$) within each strata were calculated using equations 1-6 similar to a 1-stage cluster sample. A global estimate of collision rates over all strata is analogous to a combined form ratio estimate, consisting of the ratio of the estimated average cluster total (i.e. average number of collisions detected per cluster) to an estimator of the average cluster size (i.e. average length of fence in kilometers per cluster). Estimates of the population average per linear kilometer of fence (i.e. population average collision rate) ($\bar{x}_c$) and its variance ($\hat{V}(\bar{x}_c)$) are as follows:

$$\bar{x}_c = \frac{\sum_{i=1}^{L} N_i \bar{x}_{ti}}{\sum_{i=1}^{L} N_i \bar{m}_i}$$

$$\hat{V}(\bar{x}_c) = \frac{1}{\hat{M}^2}\left\{\sum_{i=1}^{L}\frac{N_i(N_i - n_i)}{n_i}S_{ci}^2\right\}$$

Where:

$$S_{ci}^2 = \frac{\sum_{i=1}^{n}\left(x_{ji} - \bar{x}_c m_{ji}\right)^2}{n_i - 1}$$

$$\hat{M}^2 = \sum_{i=1}^{L} N_i \bar{m}_i$$

Notation used in these formulas represents the following values: N = total number of clusters in the L strata; N_i = total number of clusters in the ith strata; n_i = the number of clusters sampled in the ith strata; x_{ji} = the number of collisions detected in the jth cluster of the ith strata; m_{ji} = the length of fence in kilometers in the jth cluster of the ith strata; $\bar{x}_{ti}$ = average number of collision sites located in clusters of the ith strata (total number of collisions located / total number of clusters sampled); and $\bar{m}_i$ = average length of fence in kilometers of clusters in the ith strata (total length of fence / total number of clusters sampled). In the above equation for $\hat{V}(\bar{x}_c)$ the term $\hat{M}^2$ was used because the total length of fence in the population of clusters (M) was unknown.

The methods used will limit the inference of our estimates on breeding areas to fence collision rates within approximately 2.5 km of leks within a given region, however, this allowed us to sample a larger geographic extent across southern Idaho. We felt it was important to cover a broad geographic extent since collision risk across the landscape for sage-grouse has never been investigated, although this necessarily limited the intensity with which we could sample in any given region.

Once collision evidence was located during fence searches, biological, topographical and technical characteristics of the collision site were recorded. If collision sign was located without evidence present on the fence, all collision site measurements were made at the fence location perpendicular to the closest collision evidence. For example, feather pile evidence was often scattered by the wind from a location under or very close to fence, and the closest evidence (i.e. feather or group of feathers) was used to determine the point used for measurements. Biological variables collected at each collision site include the species, sex, and age of the victim (if possible), the type of evidence found, and the perpendicular distance to the closest, farthest, and 1st detected collision evidence located, as well as the vegetation characteristics at the collision site. Vegetation heights of the closest shrubs from each side of the fence not intersecting the plane of the fence, distances to the closest non-intersecting shrubs on each side of the fence, and canopy coverage along 10 m transects (Canfield 1941) in the cardinal directions centered on the collision locations were collected. We measured height of the closest shrub growing directly along the fence (i.e. intersecting the plane of the fence) within 5 m of the collision site in both directions to evaluate the influence of vegetation in the longitudinal direction of the fence.

Technical data collected at each collision site include fence height, fence type, and the distance between fence posts. Finally, we collected topographic data including UTM locations, and slopes both across and along the fence (measured over 20 m centered on the collision site). Additionally, aspect at each collision site and random sites will be obtained using a digital elevation model in ArcGIS. The lek count (provided by IDFG), and difference between vegetation height and the fence height will be calculated to include in modeling, and the distance of the collision site to the nearest known lek will be recorded at a later date using ArcGIS software.

Random points were selected on each study area to assess the significance of features recorded at collision locations for site scale analysis. We randomly generated one spatial location for each collision victim found on each study area within the sampling frame using the Hawthe's Tools extension (Beyer 2004) of ArcGIS software, and we used the closest fence segment to this location on the study area to measure site-scale variables that will be used in future modeling. We selected these random points from all available locations within sampling frame of each sampled lek route, and recorded data at the closest fence segment to this point.

Experimental Fence Marking

We conducted an experimental fence marking study on 8 lek routes in south-central and southeast Idaho to test the effectiveness of marking as a mitigation method to reduce sage-grouse fence collision rates. We found spatially aggregated collision sites (≥ 2 sites/km) during the 2009 field season, and used these sites in the fence marking study under a complete block design framework, with site as the blocking variable. We framed a 3 km fence segment around each collision site aggregation, and used the approximate center of these collision points as the center of the 3 km fence segments on each area. We subsequently divided each 3 km segment into 6 500 m segments, which served as the experimental units in this study. Fifty-meter buffer segments were placed between each 500-m segment to prevent birds from adjusting flight paths laterally along the fence and artificially increasing collision rates in unmarked segments by subsequently striking fences (Baines and Andrew 2003). Three of the 6 segments at each site were randomly selected to serve as marked treatments, while the remaining 3 segments at each site were used as unmarked controls. We constructed fence markers using vinyl siding undersill (Wolfe et al. 2009), and increased their visibility further by adding reflective metallic tape (Fig 1). We placed markers on the top strand of barbed-wire at approximate 1 m intervals, and did not mark subsequent strands of the fence. Wolfe et al. (2009) staggered markers on the first and third strands of fence for lesser prairie chickens in shortgrass prairie, however, we did not feel this was necessary due to the taller vegetation on Idaho rangelands. Further, fence marking will not be used as a mitigation strategy if it is not cost effective, therefore this strategy reduced both the number of markers and amount of materials needed to mark the fence. One-meter intervals were used to accommodate the approximate wingspan of sage-grouse. We initiated this study and completed all fence marking activities in February 2010.

We used a repeated measures design for fence sampling in this experiment. We sampled study fences at approximate 2-week intervals from 17 March – 26 May 2010, and all sites were sampled 5 times during the study. We conducted sampling as previously described for breeding

areas, with an observer walking each side of the fence in turn. Similarly, site data collected at collision points were identical to that collected during breeding area fence sampling, and we collected subsequent data at random fence points within the experimental areas as previously described. Traditional experimental design assumes homogenous experimental units to prevent erroneous inference in the presence of confounding variables. All fence segment experimental units in this study were not identical, therefore we collected data on covariates potentially influencing local scale collision at each segment. We used 1-in-5 systematic sampling (Schaeffer et al. 2006) on each of the 48, 500-m fence segments used in this study, where the first individual fence segment (defined as length of fence between 2 posts) sampled on each 500-m experimental unit was randomly selected from segments 1-5, and data was recorded at every 5th segment thereafter until the end of the 500-m segment. Covariate data collected at each segment included fence type, fence height, distance between fence posts, and the number of markers in the segment. We will use these data in future analysis to accommodate any confounding influences of these variables.

PRELIMINARY RESULTS

Fence Sampling and Collision Estimation

We sampled 63.3 km of fence in 14 lek routes during spring 2010 (Table 1). However, all lek routes except Crooked Creek, Medicine Lodge, and Red Road were sampled twice during the 2010 season. These 3 sites were only sampled once because they were not accessible early in the field season. Fence types sampled in sage-grouse breeding habitats were dominated by 4-strand barbed-wire, and other fence types were also common (Fig 2).

We located 50 collision sites during the 2010 fence surveys in sage-grouse breeding areas, which included 36 known sage-grouse, 13 unknown species and 1 chukar (*Alectoris chukar*) (Table 2). Twelve of the 50 collision locations (8 GRSG) found during the breeding season were not found in randomly selected sampling areas but while walking or driving through the study areas, and therefore were not used in collision rate estimation. We sent feather samples from all unknown avian collision victims to the feather identification lab at the Smithsonian Institution in attempt to identify these species. Some of these may be sage-grouse, which will increase collision rate estimates for that species. The composition of evidence types found at avian collision sites was dominated by feather piles; however, a large number of sites also contained feathers lodged in the associated fence (Fig 3).

Estimated collision rates showed strong spatial variation among study areas (Table 3). Estimated fence collision rates for all species ranged from 0-1.904 strikes/km, and appeared to be less in the second sampling round. Estimated collision rates for known sage-grouse ranged from 0-1.428 strikes/km, and showed the same reduction in subsequent sampling rounds. The estimated total number of fence collision sites present during sampling was also variable, and ranged from approximately 0-31 collision sites (Table 4). The estimated total number of sage-grouse collision sites present during sampling summed over both sampling rounds equals approximately 110 birds; however, this is a conservative estimate due to detection and survival bias. Global collision rate estimates treating lek routes as strata varied by sampling round from 0.162-0.486

(strikes/km) for all species, and 0.099-0.356 (strikes/km) for known sage-grouse (Table 5), however, as previously mentioned not all sampling rounds contained the same number of strata.

Experimental Fence Marking

We monitored 48, 500-m fence segment experimental units on 8 study areas over 5 sampling occasions during the spring 2010. We found a total of 60 avian collision sites during this sampling, 54 of which were sage-grouse (Table 6). We found evidence for both temporal variation in collision rates within the lekking season, and strong spatial variation in collision rates between the study sites (Fig 4). Our data suggest a peak in collision risk in late March and early April, which declines steadily towards the end of the lekking season (Fig 4). Furthermore, 3 times more collision sites were found at the Fingers Butte site (33) than at Lidy, the second highest site (11). The composition of evidence types found was dominated by feather piles, with a smaller number of sites also having feathers lodged in the fence (Fig 5). In addition to variation between the sites and within the season, we found differences in the number of collisions found by treatment type (Fig 6a). Sage-grouse fence collision rates pooled across all sites and times were approximately 5.74 times greater in unmarked control and buffer segments (3.33 strikes/km) than in marked treatment segments (0.58 strikes/km). Treatment effects may also have varied among sites (Fig 6b) and within the season (Fig 7), and may reflect changes in latent collision risk through space and time. The average distance between successive fence posts was 484.3 cm for the 48 500 m fence segments, with an average fence height of 110.5 cm.

FUTURE FIELDWORK

Future work for this project will focus on analysis of field data collected during the 2009 and 2010 field seasons, and completion of MS thesis no later than May 2011. Collision risk models will be built using classification and regression trees (CART) (Breiman et al. 1984), modeling collision risk as a function of biological, technical, and spatial factors at the site, patch and landscape scales. Experimental fence marking data will be analyzed using repeated measures mixed effects modeling to evaluate the influence of site, time, treatment, and fence covariate data on collision rates during this study.

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Table 1. Total fence length sampled for the presence of avian collision sites in each lek route sampled in southern Idaho during spring 2010.

Lek Route	Length
Antelope Pocket	7.3
Browns Bench	4.5
Big Desert #1	1.6
Big Desert #3	2.1
Crooked Creek	1.0
Fingers Butte	7.5
Lidy	5.3
Medicine Lodge	5.8
North Shoshone	4.3
Paddelford Flat	1.6
Picabo Hills	4.9
Plano	6.9
Red Road	6.2
Table Butte	4.2
Total	63.3

Table 2. Total number of avian collision sites found by species in randomly selected areas and opportunistically for each lek route sampled in southern Idaho during spring 2010.

Lek Route	CHKR[a]	GRSG[b]	UNK[c]	Total
Antelope Pocket	1	1	3	5
Browns Bench	-	3	-	3
Big Desert #1	-	-	-	0
Big Desert # 3	-	3	2	5
Crooked Creek	-	1	1	2
Fingers Butte	-	4	-	4
Lidy	-	6	1	7
Medicine Lodge	-	-	-	0
North Shoshone	-	-	-	0
Paddelford Flats	-	1	1	2
Picabo Hills	-	-	-	0
Plano	-	7	2	9
Red Road	-	4	1	5
Table Butte	-	6	2	8
Total	1	36	13	50

[a] CHKR = chukar.
[b] GRSG = greater sage-grouse.
[c] UNK = unknown species. All unknown species samples have been sent to the feather identification lab at the Smithsonian Institution for identification.

Table 3. Estimates of fence collision rates (strikes/km) by lek route and sampling round using only collision sites found in randomly selected areas for all sites sampled in southern Idaho during spring 2010.

Sampling Period/Area	n[a]	N[b]	Collision Rate (All Species)	95% CI	Collision Rate (Known GRSG)	95% CI
Round 1						
Antelope Pocket	10	42	0.275	(-0.009, 0.559)	0	N/A
Browns Bench	9	37	0	N/A	0	N/A
North Shoshone	4	14	0	N/A	0	N/A
Picabo Hills	5	19	0	N/A	0	N/A
Paddelford Flat	7	29	1.256	(0.627, 1.884)	0.628	(0.313, 0.942)
Fingers Butte	8	31	0.534	(0.068, 1.000)	0.534	(0.068, 1.000)
Big Desert #1	2	7	0	N/A	0	N/A
Big Desert #3	2	8	1.904	N/A	1.428	N/A
Crooked Creek	1	4	0	N/A	0	N/A
Medicine Lodge	8	32	0	N/A	0	N/A
Lidy	5	19	0.749	(-0.173, 1.671)	0.562	(-0.040, 1.163)
Table Butte	4	17	0.475	(-0.029, 0.979)	0.238	(-0.185, 0.660)
Plano	5	22	1.014	(0.319, 1.709)	0.869	(0.261, 1.477)
Red Road	10	42	0.802	(0.262, 1.342)	0.642	(0.149, 1.135)
Round 2						
Antelope Pocket	10	42	0.275	(0.031, 0.518)	0.137	(-0.117, 0.392)
Browns Bench	9	37	0	N/A	0	N/A
North Shoshone	4	14	0	N/A	0	N/A
Picabo Hills	5	19	0	N/A	0	N/A
Paddelford Flat	7	29	0	N/A	0	N/A
Fingers Butte	8	31	0	N/A	0	N/A
Big Desert #1	2	7	0	N/A	0	N/A
Big Desert #3	2	8	0.476	N/A	0	N/A
Lidy	5	19	0.562	(0.095, 1.028)	0.562	(0.095, 1.028)
Table Butte	4	17	0	N/A	0	N/A
Plano	5	22	0.290	(-0.031, 0.610)	0.145	(-0.129, 0.418)

[a] n = number of clusters sampled in lek route.
[b] N = total number of clusters in sampling frame for lek route.

Table 4. Estimated total number of collision sites present during sampling by lek route and sampling round using only collision sites found in randomly selected areas for all sites sampled in southern Idaho during spring 2010.

Sampling Period/Area	n[a]	N[b]	Estimated Total (All Species)	95% CI	Estimated Total (Known GRSG)	95% CI
Round 1						
Antelope Pocket	10	42	8.400	(-1.376, 18.176)	0	N/A
Browns Bench	9	37	0	N/A	0	N/A
North Shoshone	4	14	0	N/A	0	N/A
Picabo Hills	5	19	0	N/A	0	N/A
Paddelford Flat	7	29	8.286	(-6.148, 22.719)	4.143	(-3.074, 11.360)
Fingers Butte	8	31	15.500	(1.227, 29.773)	15.500	(1.227, 29.773)
Big Desert #1	2	7	0	N/A	0	N/A
Big Desert #3	2	8	16.000	N/A	12.000	N/A
Crooked Creek	1	4	0	N/A	0	N/A
Medicine Lodge	8	32	0	N/A	0	N/A
Lidy	5	19	15.200	(-3.820, 34.220)	11.400	(-1.648, 24.448)
Table Butte	4	17	8.500	(-0.083, 17.083)	4.250	(-3.183, 11.683)
Plano	5	22	30.800	(4.567, 57.033)	26.400	(3.847, 48.953)
Red Road	10	42	21.000	(1.296, 40.704)	16.800	(0.588, 33.012)
Round 2						
Antelope Pocket	10	42	8.400	(-1.376, 18.176)	4.200	(-3.132, 11.532)
Browns Bench	9	37	0	N/A	0	N/A
North Shoshone	4	14	0	N/A	0	N/A
Picabo Hills	5	19	0	N/A	0	N/A
Paddelford Flat	7	29	0	N/A	0	N/A
Fingers Butte	8	31	0	N/A	0	N/A
Big Desert #1	2	7	0	N/A	0	N/A
Big Desert #3	2	8	4.000	N/A	0	N/A
Lidy	5	19	11.400	(-1.648, 24.448)	11.400	(-1.648, 24.448)
Table Butte	4	17	0	N/A	0	N/A
Plano	5	22	8.800	(-0.674, 18.274)	4.400	(-3.336, 12.136)

[a] n = number of clusters sampled in lek route.
[b] N = total number of clusters in sampling frame for lek route.

Table 5. Global estimates of breeding season collision rates for sampling areas in southern Idaho 2010.

Sampling Round	Lek Routes Sampled	Collision Rate (All Species)	95 % CI	Collision Rate (Known GRSG)	95% CI
1	14	0.486	(0.298, 0.674)	0.356	(0.205, 0.506)
2	11	0.162	(0.065, 0.259)	0.099	(0.016, 0.183)

Table 6. Total number of avian collision sites found by species on marked treatment and unmarked control fence segments for each site sampled during experimental fence marking studies in southern Idaho during spring 2010.

Site	GRSG[a]	UNK[b]	Total
Browns Bench	1	-	1
Big Desert #1	1	-	1
Big Desert #3	8	-	8
Crooked Creek	4	1	5
Fingers Butte	31	2	33
Lidy	8	3	11
Table Butte	1	-	1
Total	54	6	60

[a] GRSG = greater sage-grouse.
[b] UNK = unknown species, all of which were subsequently sent to the feather identification lab at the Smithsonian Institution for identification.

a b

c d

Figure 1. Fence marker construction and completed product for markers used in experimental fence marking study in southern Idaho during spring, 2010. (a) Applying reflective metallic tape to vinyl undersill in fence marker construction. (b) Cutting reflective tape to allow vinyl undersill markers to clip onto fence. (c) Cutting 12 ft. reflective vinyl strips into 3 in. markers using table saw. (d) Example of finished marker product placed on the top strand of fence at approximate 1-m intervals on the Browns Bench site.

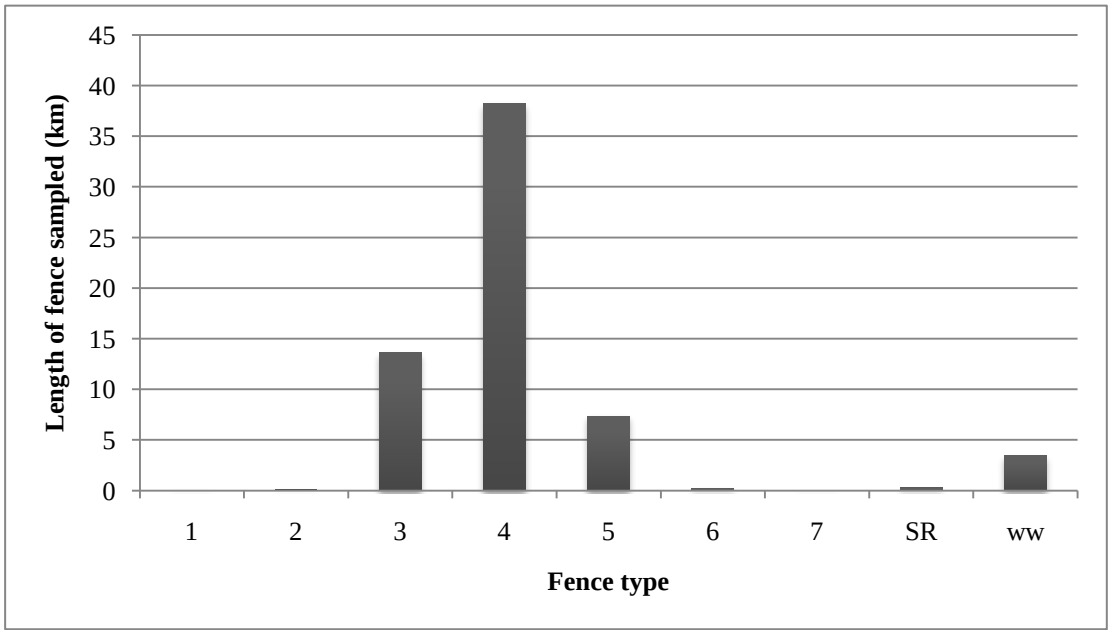


Figure 2. Length of each fence type found while sampling grater sage-grouse breeding areas for the presence of fence collision sites in southern Idaho during spring, 2010. Numbers are the number of strands of barbed-wire, SR = split rail fence, and WW = woven wire fence.

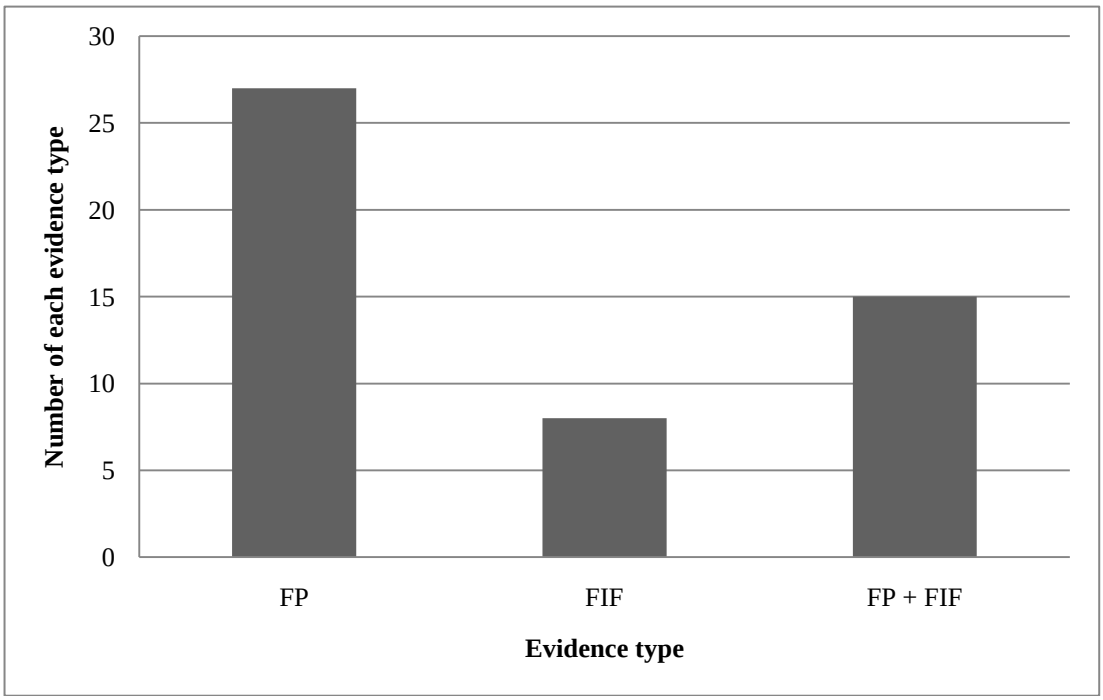


Figure 3. Proportion of each evidence type found at avian fence collision sites while searching greater sage-grouse breeding areas in southern Idaho during spring 2010. FP stands for feather piles found and FIF stands for feathers lodged in the fence.

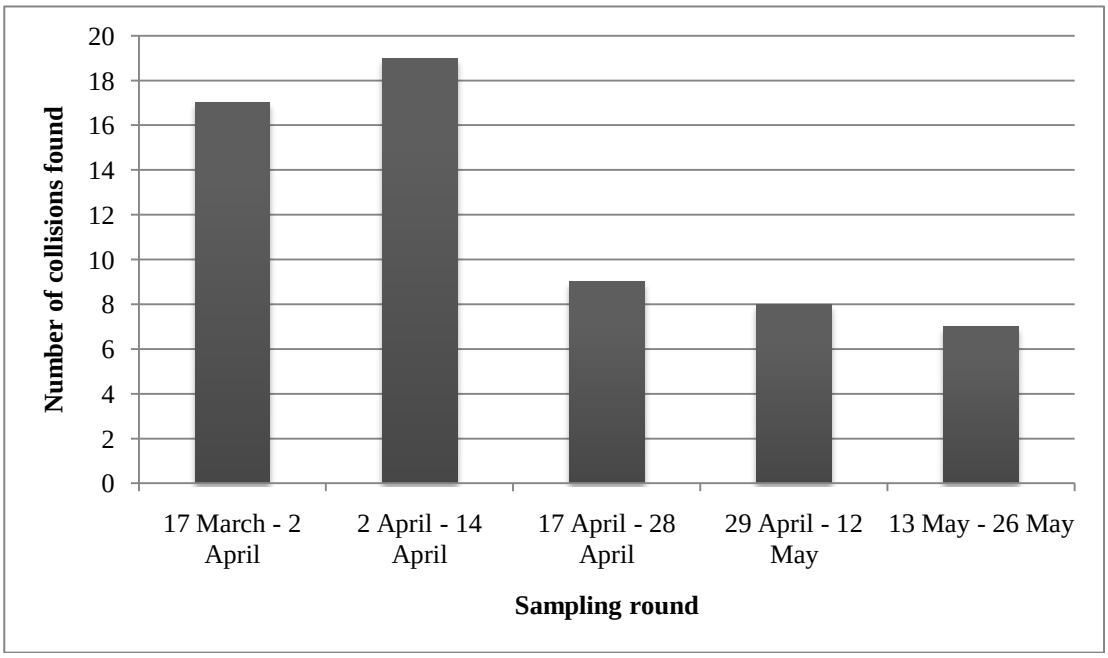
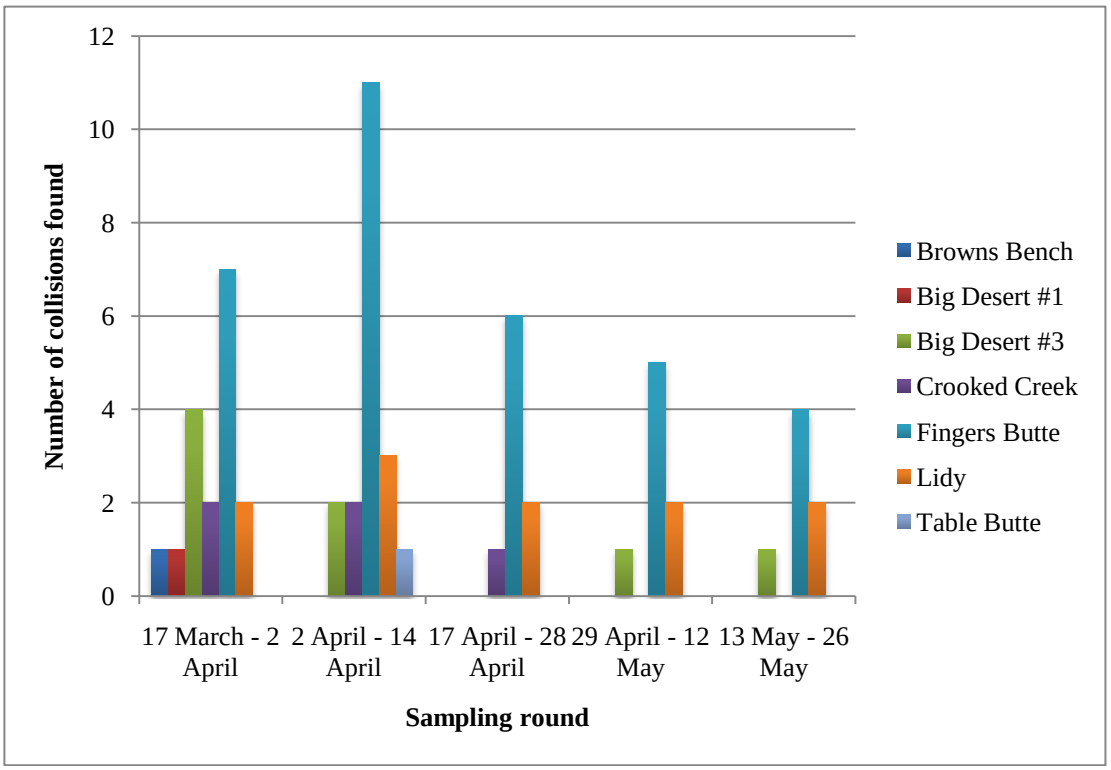

20
18
16
14
12
10
8
6
4
2
0
Number of collisions found
17 March - 2 April
2 April - 14 April
17 April - 28 April
29 April - 12 May
13 May - 26 May
Sampling round

Figure 4a



b

Figure 4a &b. Total number of avian collision sites located by time period during experimental fence marking studies in southern Idaho during spring 2010. (a) Total number of collision sites found for each sampling period all sites pooled. (b) Total number of collision sites found per site for each sampling period.

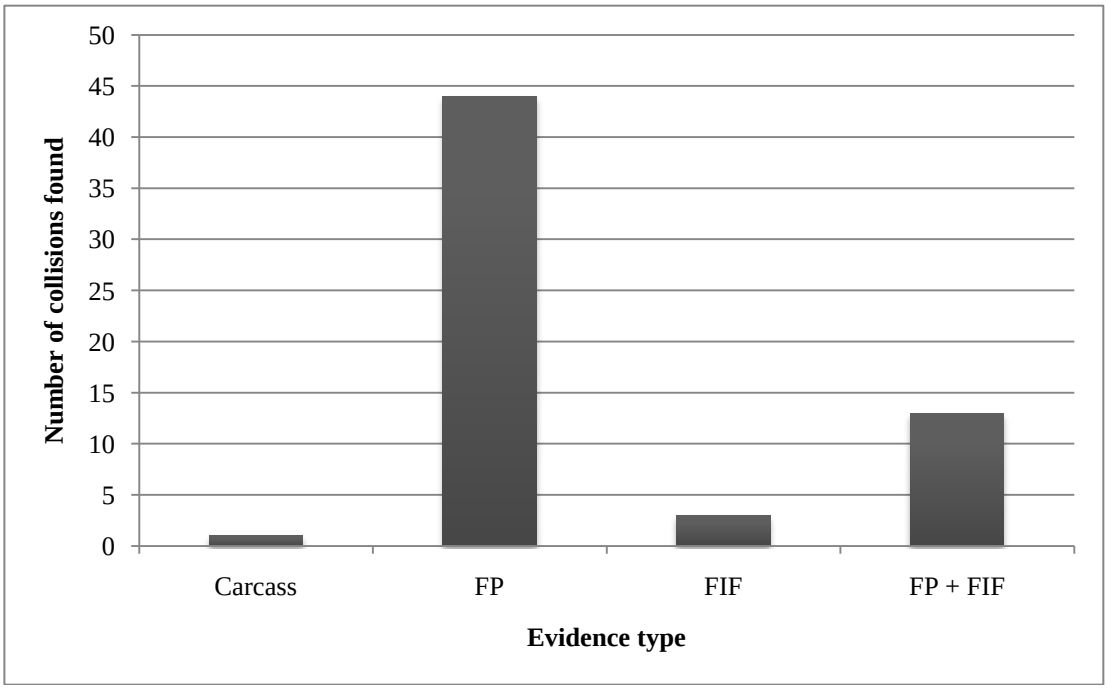


Figure 5. Proportion of each evidence type found at avian fence collision sites during experimental fence marking studies in southern Idaho during spring 2010. FP stands for feather piles found and FIF stands for feathers lodged in the fence.

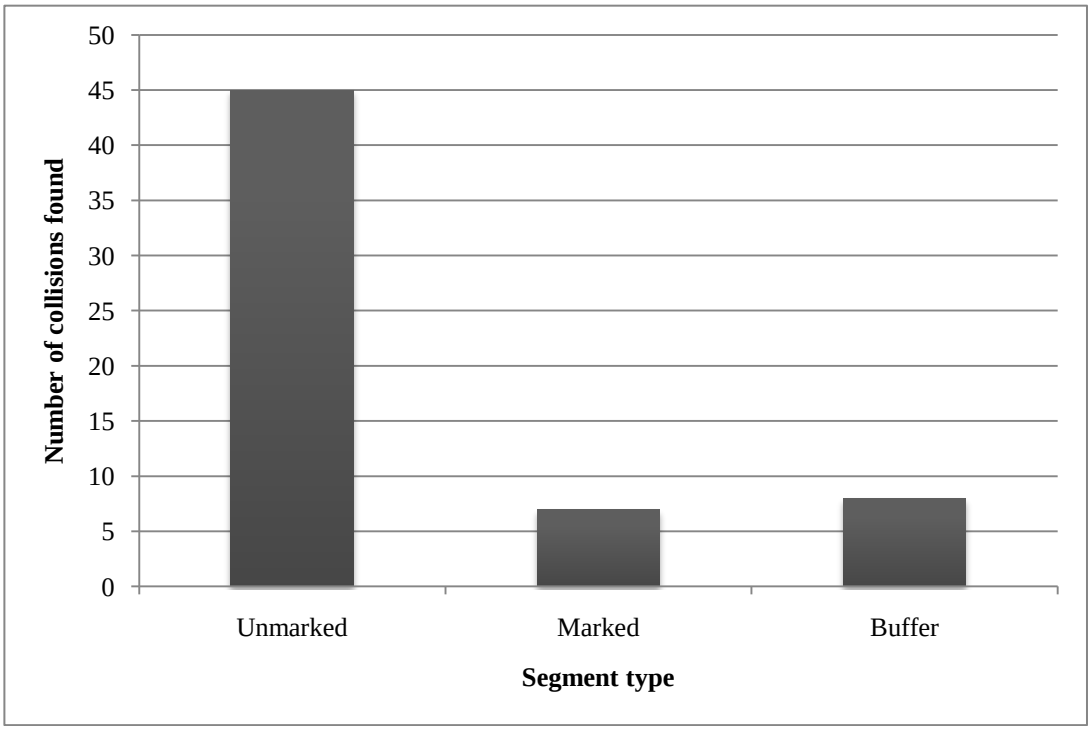

50
45
40
35
30
25
20
15
10
5
0
Number of collisions found
Unmarked
Marked
Buffer
Segment type

Figure 6a

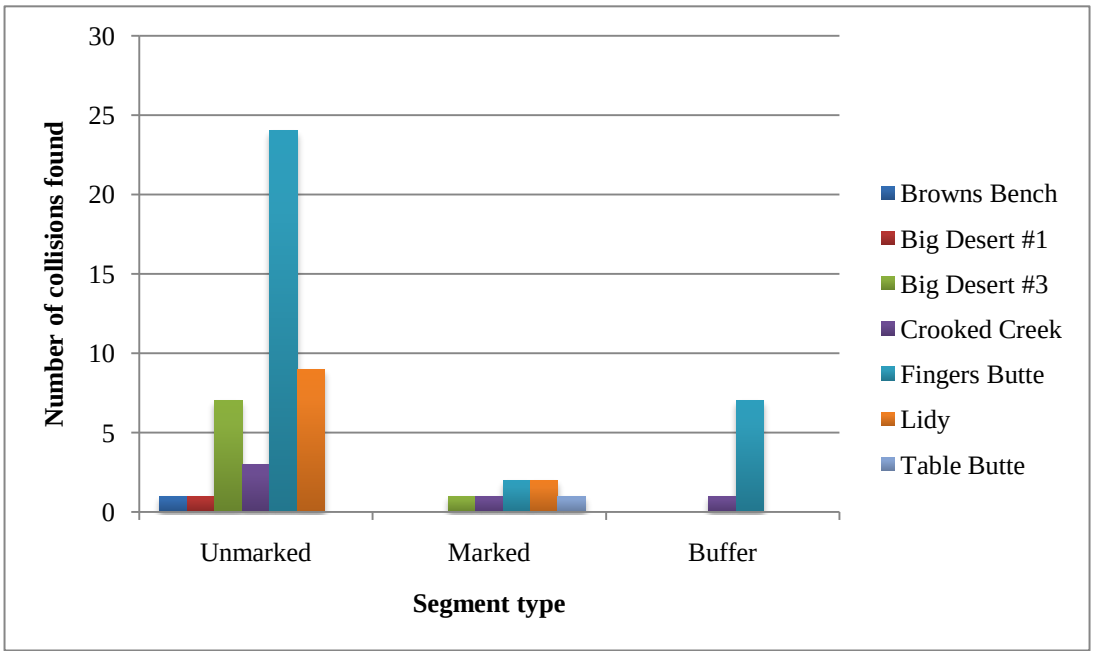


Figure 6a & b. Total number of avian collision sites located by treatment type during experimental fence marking studies in southern Idaho during spring 2010. (a) Total number of collision sites found for each treatment level all sites pooled. (b) Total number of collision sites found for each treatment level per site.

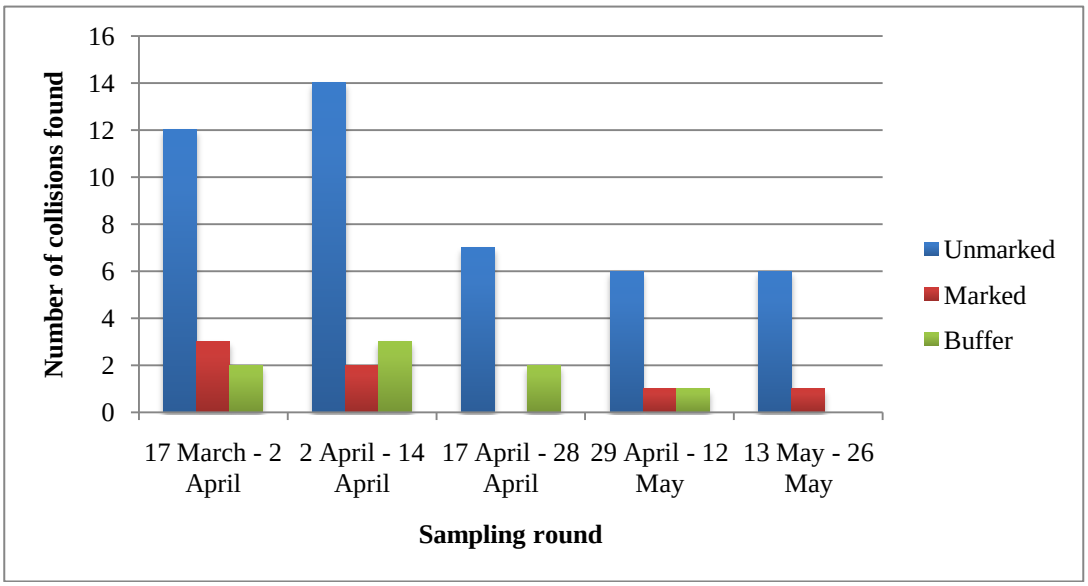


Figure 7. Total number of avian collision sites located per treatment type and time period duringexperimental fence marking studies in southern Idaho during spring 2010.

STUDY I; PART III

The following manuscript was accepted for publication in *Journal of Wildlife Management*.

Factors influencing survival and detectability bias of avian fence collision surveys in sagebrush steppe

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ABSTRACT We used hen ring-necked pheasant (*Phasianus colchicus*) carcasses as surrogates for greater sage-grouse (*Centrocercus urophasianus*) to study factors influencing survival and detection bias associated with avian fence collision surveys in southern Idaho, USA, during spring 2009. We randomly placed 50 pheasant carcasses on each of 2 study areas, estimated detection probability during fence-line surveys, and monitored survival and retention of carcasses and their associated sign over a 31-day period. Survival modeling suggested site and habitat features had little impact on carcass survival, and the constant survival models were most supported by the data. Model averaged carcass daily survival probability was low on both study areas, and ranged from 0.776-0.812. Survival of all carcass sign varied strongly by location, and the top sign survival model included a site effect parameter. Model averaged daily survival probability for collision sign on the 2 study sites ranged from 0.863-0.988, and varied between sites. Logistic regression modeling indicated detection probability of carcasses during fence-line surveys for avian collision victims was influenced by habitat type and microsite shrub height at the carcass location. Carcasses located in big sagebrush (*Artemisia tridentata*) habitats were detected at a lower rate (0.36) than carcasses in little and black sagebrush (*A. arbuscula, A. nova*) habitats (0.71). Increasing shrub height at the carcass location from the little sagebrush mean of 16.5 cm to the big sagebrush mean of 36.0 cm reduced the detection probability by approximately 30 percent. Avian fence collision surveys in sagebrush-steppe habitats should be conducted at a maximum of 2-week sampling intervals to reduce the impact of survival bias on collision rate estimates. Two-week sampling intervals may be too long in areas with low carcass and sign survival, therefore survival rates should be estimated on all study areas to determine the appropriate sampling interval duration. Researchers should be aware of the effects of local vegetation on detection probabilities, and methods to correct detection probabilities based on collision site attributes should be applied to ensure more accurate collision rate estimates. Additionally, caution should be used when aggregating or comparing uncorrected collision data from areas with differing vegetation, as detection probabilities are likely different between the sites.

KEY WORDS carcass survival, *Centrocercus urophasianus*, detectability, fence collisions, Idaho, sagebrush, sage-grouse, scavenging.

Avian collision with anthropogenic infrastructure has received considerable attention in recent years (Wolfe et al. 2007, Drewitt and Langston 2008, Smallwood et al. 2009, Gehring et al. 2009). Elevated structures known to cause avian collision mortality include fences (Baines and Summers 1997, Wolfe et al. 2007, Bevanger and Brøseth 2000), power lines (Bevanger 1995, Janss and Ferrer 2000), wind power turbines (Smallwood 2007, Smallwood and Thelander 2008, Smallwood et al. 2009), and communication towers (Avery et al. 1978, Gehring et al. 2009). Fence collision has been identified as a substantial source of mortality in some areas for lesser prairie chickens (*Tympanuchus pallidicinctus*) (Wolfe et al. 2007). Although elevated infrastructures such as fences and power lines are abundant on western rangelands, their impacts on sagebrush obligates like greater sage-grouse (*Centrocercus urophasianus*) (hereafter sage-grouse) are not well understood (Connelly et al. 2000, Connelly et al. 2004). Connelly et al. (2000) suggested flagging fences to increase their visibility near sage-grouse habitats if they appear hazardous, but no studies have evaluated the degree of hazard which fences present to sage-grouse. Many populations of shrub-steppe birds are believed to be declining (Knick et al. 2003), including sage-grouse (Connelly et al. 2004), and elevated structures have been hypothesized to contribute to sage-grouse declines (Braun 1998). Despite the lack of information concerning collision risk of sage-grouse, collision with fences (Flake et al. 2010) and power lines (Beck et al. 2006) has been documented, and both fences and power lines have proven a significant source of mortality for other species of tetraonids (Baines and Summers 1997, Bevanger and Brøseth 2000, Moss et al. 2000, Wolfe et al. 2007). Conservation concerns for sage-grouse have made it necessary to estimate avian collision rates in sagebrush habitats, and quantify collision risk across the landscape.

A commonly used method of estimating collision rates of birds with elevated infrastructure involves searchers walking along power line corridors or fence sections to locate collision sites (Baines and Summers 1997, Bevanger and Brøseth 2004). While this method is relatively effective for determining collision rates over large areas, there are 2 major biases associated with searching for collisions in this manner; detectability and scavenging bias (Bevanger 1999).

Detectability bias is common in mortality studies, and may be influenced by factors such as meteorological conditions, snow cover, size of the bird under study, local vegetation, and the ability of the field observers (Bevanger 1999). Scavenging bias refers to the "survival" of the collision evidence, which is often removed by scavenging animals or weather prior to observation during sampling (Bevanger 1999, Smallwood 2007). Because survival of collision evidence depends on more than scavenging alone, we chose to refer to this as survival bias.

Numerous attempts to quantify carcass retention rates have been made, and these usually involve monitoring collision victims (James and Haak 1979) or planted carcasses (Bevanger et al. 1994, Savereno et al. 1996). Most previous studies of carcass retention suggest the birds were placed "randomly"; however, these studies often lacked a strong experimental design or presented extremely vague descriptions of their methods (e.g. Pain 1991, Bevanger and Brøseth 2004). For example, Baines and Summers (1997) reported that 18 of 20 red grouse (*Lagopus lagopus scoticus*) carcasses placed near fences were gone after 1 month, however they did not provide a detailed description of field placement methods. The accuracy and applicability of scavenging rates calculated in many studies are also hindered by small sample sizes ($n \leq 20$) (e.g. Savereno

et al. 1996, Baines and Summers 1997). In addition to small sample sizes and vague descriptions of methodology, Smallwood (2007) suggested that much of the previous research has not asked the correct questions, often not attempting to address factors influencing carcass survival.

Although most studies estimating carcass retention only considered carcass survival as a function of time, several studies have evaluated the influence of other factors. Bumann and Stauffer (2002) suggested that birds with exposed viscera likely provided stronger olfactory stimulus to scavengers. One study using songbird carcasses reported differences in carcass retention between different habitat types (Kostecke et al. 2001). Furthermore, Smallwood (2007) found evidence for seasonal variation in scavenging rates of birds killed by wind facilities, with the greatest scavenging rates occurring in the fall.

Much like estimates of carcass retention, estimates of carcass detectability commonly accompany avian collision mortality studies (Anderson 1978, Savereno et al. 1996, Bevanger 1999). Although most of these studies have reported detectability rates to correct mortality estimates, many have failed to quantify the factors that influence detectability. Osborn et al. (2000) evaluated the influence of season, vegetation height, and snow cover on detectability of bird carcasses on wind facilities in Minnesota. Detectability was only influenced by size of the bird, with large birds having a higher detection rate (92.3%) than small birds (68.7%) (Osborn et al. 2000). Additionally, Smallwood (2007) examined influences of vegetation height and bird group on detectability of wind turbine collision victims and found that detectability varied by bird group (i.e. large raptors, large non-raptors, etc.), while vegetation height only influenced detection for small non-raptor birds.

Conservation concerns associated with elevated infrastructure and greater sage-grouse have made it desirable to quantify avian fence collision rates in sagebrush-steppe habitats. Proper estimation of collision rates requires quantifying various sources of bias associated with searching for and detecting collision victims along infrastructure (Bevanger 1999, Kuvlesky et al. 2007). Therefore, our objectives were to: 1) estimate surrogate sage-grouse carcass and collision sign survival and detectability in sagebrush-steppe habitats, and 2) determine the influence of habitat composition, distance of surrogate sage-grouse carcass from fence, and road presence on carcass and collision sign survival and detectability.

STUDY AREA

This study was replicated on 2 geographic regions of southern Idaho (Fig 1). We initiated studies to determine retention of pheasant carcasses on 27 March 2009 and 7 April 2009 on the Browns Bench (hereafter BB) and Upper Snake (hereafter US) study areas. The Browns Bench region is in southern Twin Falls County and borders Nevada. Elevations on the BB study area ranged from approximately 1,450-1,850 m, and vegetation was dominated by little sagebrush (*Artemisia arbuscula*) and black sagebrush (*Artemisia nova*) in the southern portion of the study area, and big sagebrush (*Artemisia tridentata*) in the northern areas. However, habitat conditions were variable and ranged from dense stands of sagebrush to bare pasture and large stands of crested wheatgrass (*Agropyron cristatum*). Cheatgrass (*Bromus tectorum*) was also common on the study area. Additionally, riparian areas in the southern portion of BB commonly contained stands of big sagebrush.

The US region study site occurred on the Table Butte and Crooked Creek areas of the Upper Snake River Plain, in Clark and Jefferson counties. Since big and little sagebrush dominated areas in the US region could not be located in as close proximity as the BB study site, the Table Butte area was selected for the big sagebrush habitat, and the Crooked Creek area for the little sagebrush habitat. These areas were approximately 10 km apart, with similar elevations ranging from approximately 1,520-1,825 meters. The big sagebrush study site was directly north of Table Butte, bounded on the east by U.S. Highway 15, and extended to the north and west of Table Butte by approximately 2.5 and 8 kilometers. This site was dominated by big sagebrush, with an approximately 900-ha stand of crested wheatgrass. The little sagebrush site in the Crooked Creek drainage was bounded on the south by Idaho Highway 22, and on the north, east, and west by the Beaverhead Mountains. This site was dominated by little sagebrush, with some large pasture and grassy areas intermixed. Furthermore, unlike the BB study site, the habitat types in the US sampling area were distinctly separated, such that carcasses placed in big and little sagebrush were not in as close proximity as those on the BB study site.

Avian, mammalian, and arthropod scavengers were common on the study areas. Common scavengers previously identified in the vicinity of our study sites (Coates et al. 2008) included coyotes (*Canis latrans*), American badgers (*Taxidea taxus*), weasels (*Mustela* spp.), common ravens (*Corvus corax*), American crows (*Corvus brachyrhynchos*), black-billed magpies (*Pica hudsonia*), and numerous species of small mammals (*Tamias* spp., *Spermophilus* spp., *Thomomys* spp., *Perognathus* spp, *Peromyscus* spp., *Lemmiscus* spp.). We observed additional predators on our study sites, including golden eagles (*Aquila chrysaetos*), and multiple species of hawks (*Buteo* spp.) and owls (*Asio* spp., *Athene* sp.). On the US study site we identified bobcat (*Lynx rufus*) tracks near carcass locations, and commonly observed scavenging by arthropods (Orders Coleoptera, Diptera, and Hymenoptera).

METHODS

Field Methods

We used pen-raised hen ring-necked pheasants (*Phasianus colchicus*) as the experimental units to evaluate factors influencing survival and detectability of fence collision victims in sagebrush-steppe habitats. We used a completely randomized design (Ott and Longnecker 2001) with 2 levels of treatment effects for habitat type (big sagebrush or little/black sagebrush), and carcass distance from the fence as a covariate, with carcasses placed at random distances of 0-15 m from a fence. We euthanized all birds used in this study using cervical dislocation, with approval from the University of Idaho Animal Care and Use Committee (Protocol 2009-21) prior to commencement of the study. We banded all carcasses for individual identification prior to field placement to aide in monitoring. We removed feathers from the front of the breast of each carcass to provide feather pile sign, and made two perpendicular 4 cm incisions, centered where the feathers were removed from the breast and intended to simulate collision with a barbed-wire fence. This was done to prevent scavenging bias associated with unrealistic carcass presentation (Bumann and Stauffer 2002). Carcasses were not frozen, but were placed in coolers on ice until field placement. A technician wearing rubber boots and gloves to minimize human scent on the carcasses (Whelan et al. 1994) placed each carcass and feather pile (n = 10-15 feathers) in the

field at night to minimize diurnal scavenger detection by simply observing field workers. Each carcass was placed at random distances perpendicular to the center point of the randomly selected fence segment to facilitate site relocation by researchers. Additionally, we placed 10-15 feathers immediately around each carcass. Lone feather piles of 10-15 breast feathers were placed similarly to carcasses to determine feather detectability, as it could differ from that of carcasses.

We measured the influence of the previously mentioned factors on detectability and longevity for the entire collision sign in addition to the carcasses. Placing feathers around each carcass prevented a carcass from being removed and leaving no visible sign, a scenario that is unlikely with collision victims as feathers will likely fall when birds strike fences (Flake et al. 2010). This design was replicated on both the BB and US study areas to allow detection of regional differences in carcass survival. Detectability was only measured on the BB site due to a lack of field personnel available to conduct detectability trials on the US study area.

We quantified available fences in each habitat type on each study area using ground searches, handheld global positioning system [GPS] units, and ArcGIS software [GIS]. Fence sections located in each habitat type were mapped by taking GPS waypoints along and at the ends of each fence section, then digitizing the fence segments in GIS. Because sagebrush-steppe habitats often have a patchy mosaic of sagebrush, pasture, and grassy areas, only fence segments traversing areas dominated on both sides by the desired habitat type were included in this study. We did not differentiate between little sagebrush and black sagebrush, and both species were grouped together in the little sagebrush level of habitat treatment.

Once fence sections were digitized, the GIS extension Hawthe's Tools (Hawth's analysis tools for ArcGIS, <http://www.spatialecology.com/htools.>, accessed 12 January 2009) was used to generate random fence points > 200 m apart in each habitat type. Bumann and Stauffer (2002) placed ruffed grouse carcasses > 100 m apart in their Appalachian study, however, sagebrush-steppe habitats are more open than deciduous forest, and larger inter-carcass distances are likely necessary. Therefore, our methods ensured that carcass locations were no closer than 200 m. We placed 50 bird carcasses on each study area, 25 replicates for each treatment level (big and little sagebrush). In addition to the 50 points generated on each area for placement of pheasant carcasses, we generated 50 points (25 for each treatment level) in a similar manner on the BB study area for placement of feather piles used in detectability trials. All randomly generated points were ground verified to be in the desired habitat type prior to field placement, and those points not in the desired habitat were discarded. We placed carcasses and feather piles on the BB site randomly along 31.7 km of fence (little sagebrush = 18.2 km, big sagebrush = 13.5 km). Less fence was available on the US site, and we placed carcasses randomly along 22.9 km of fence (little sagebrush = 11.2 km, big sagebrush = 11.7 km). We did not use fence sections running along paved roads, as this could bias results if scavengers used or avoided improved road corridors. We did include fence sections running along unimproved "2-track" or gravel roads due to the abundance of these roads on the study areas, and road presence was treated as a random covariate in this analysis. Thirty-two carcasses were placed along fences with a road present (US = 21, BB = 11), and more of these carcasses were located in little sagebrush (n = 23) than big sagebrush habitats (n = 9).

The day following placement of carcasses on the BB site, observers searched all study fence segments to estimate detection probabilities. Two observers walked each fence section (1 on each side of the fence, or 1 observer walked both sides in turn) searching for bird carcasses and sign within approximately 15 m of the fence, while monitoring the fence itself for presence of feathers or bird parts. Field observers searched all potential fence sections digitized for the study, both with and without planted carcasses and feather piles, to eliminate detection bias by workers expecting bird locations. Furthermore, field observers searched extra fence sections not included in the random point generation without knowledge of which sections were included in the study. The technician who planted the birds was not involved in searches but did verify the presence of birds not detected by the field searchers on day one.

After initial searches, carcasses and their remaining sign were monitored every 1-3 days until removal, for a maximum of 31 days. During each monitoring period we recorded one of the following: a) intact carcass, b) carcass scavenged but present, c) carcass removed but feathers or sign still present, or d) all collision evidence removed. Additionally, the observer described the qualitative description of the carcass sign and remaining feathers over time within approximately 5 m of the original carcass location. The observer also recorded the presence of any precipitation events that could influence carcass retention or detection of remaining sign, such as snow or rain, at the start of each search. Observers also noted any obvious scavenger sign or individual scavengers detected in the vicinity of the carcasses during each search.

We recorded microhabitat characteristics of carcass placement sites after carcass removal to determine the influences of vegetation on survival and detectability. Grass height and shrub height were measured at the carcass location, and 1 m from the carcass location in the cardinal directions (Hausleitner et al. 2005). We measured shrub canopy coverage on 2 perpendicular 4-m transects centered on the carcass location and oriented in the cardinal directions using the line-intercept method (Canfield 1941). Additionally, we used a 12x12 cm coverboard to estimate percent visual concealment at heights of 1.5 m and 1.0 m, at a distance of 10 m in the cardinal directions from the carcass location (Jones 1968, Hausleitner et al. 2005). Due to observer error, sign from one carcass on the BB study area was not monitored until complete removal of sign occurred; therefore all calculations regarding sign survival are with a sample size of 99 birds, whereas carcass survival analysis included all 100 experimental units.

Statistical Methods

We conducted survival analysis for hypothetical collision victims using the nest survival module in program MARK (White and Burnham 1999). Because survival of carcasses was monitored analogously to the monitoring of avian nests, with variable time intervals and binary Bernoulli trials for each monitoring event, the nest survival module was an appropriate model for rigorous survival estimation. We followed the terminology of Dinsmore et al. (2002), where daily survival rate is the probability the carcass will survive a single day, and survival probability is the probability of survival over the entire study period. All survival models in this study were generated using the logit link function, such that

$$\text{logit}\left(\hat{S}\right) = \ln\left(\frac{\hat{S}}{1-\hat{S}}\right) = \hat{\beta}_0 + \hat{\beta}_1 X_1 + \ldots + \hat{\beta}_k X_k$$

where $\hat{S}$ is the estimated daily survival rate, $\hat{\beta}_i$ are the linear model coefficients, and X_i are the independent predictor variables. Furthermore, we calculated reconstituted daily survival rate estimates by back transforming the given model, where

$$\hat{S} = \frac{e^{\hat{\beta}_0 + \hat{\beta}_1 X_1 + \ldots + \hat{\beta}_k X_k}}{1 + e^{\hat{\beta}_0 + \hat{\beta}_1 X_1 + \ldots + \hat{\beta}_k X_k}}$$

and the linear model is the generalized linear model under consideration. Lastly, the variances of reconstituted survival rates were calculated using the delta method (Seber 1982).

We conducted all modeling in this analysis within an information-theoretic model selection framework (Burnham and Anderson 2002). Hypothesized models were ranked using Akaike's Information Criterion corrected for small sample sizes [AIC_c] (Burnham and Anderson 2002). We estimated overdispersion [ ] in the carcass survival models by the deviance to deviance degrees of freedom ratio for the global model. This estimate is often biased high for small sample sizes (McCullagh and Nelder 1989), but is currently the only way to assess goodness of fit for the nest survival model in program MARK (Dinsmore et al. 2002). When modeling indicated overdispersion (> 1) we ranked models using quasi-AIC corrected for small sample size [$QAIC_c$] (Burnham and Anderson 2002). This model selection and ranking procedure was used instead of likelihood ratio testing to permit comparison of non-nested models. Additionally, we used normalized Akaike model weights [w_i] as a measure of strength of evidence for a given model, and generated model averaged daily survival rates and survival probabilities to account for uncertainty in model selection procedures (Burnham and Anderson 2002). Unconditional variances (Buckland et al. 1997) were calculated for model averaged survival rates in program MARK.

We constructed survival models for both planted carcasses and collision sign. We determined survival for a carcass by the length of time until a planted carcass was first found scavenged. Similarly, we determined survival for all collision sign by the length of time until < 5 feathers were found present at the original carcass location. Daily survival rate of all collision sign was of interest because this has direct application to the appropriate time-interval lengths between fence-line surveys for avian collision sites.

We constructed survival models using a priori hypothesized local-scale and microhabitat characteristics (Table 1). Local-scale factors used in model building included the habitat type treatment, and the random covariates for road presence and distance of the carcass from the fence. Site was also a 2-level factor included in the local-scale models because the field experiment was replicated on 2 study areas. Survival modeling for the local-scale factors included comparison of 16 additive models, using all combinations of the 4 independent variables, as well as the constant survival model (Appendix A). Furthermore, 3 biologically plausible 2-way interactions were hypothesized for these models (Site*Habitat, Site*Road, and Habitat*Road), and were added only when the terms in the interaction were in a model together among the top group of models ($\Delta AIC_c \approx \leq 2$) (Table 1).

We used a separate group of hypotheses and models to evaluate the importance of features at the microhabitat scale at carcass locations. Because these factors represented a different scale, and were not factors directly considered in the design of the study, these models represented a separate group of hypotheses and were not directly compared to the local-scale models. Variables included in microhabitat scale modeling were average grass height, average shrub height, percent visual concealment (measured as the total proportion of coverboard blocks concealed), and average percent canopy coverage (Table 1). We tested for correlation between microhabitat predictor variables using correlation t-tests. Microhabitat predictor variables that were significantly correlated ($P < 0.05$) were not included together in the same model. We hypothesized 2 biologically plausible 2-way interactions in the microhabitat models (Grass Height*Shrub Height, Grass Height*Canopy Coverage), and again added these terms only when the individual terms were in a model together among the top models (Table 1). Correlation in the microhabitat variables resulted in comparison of 10 models at this scale, with the global model included for the purpose of estimating overdispersion in the model set (Appendix A). Candidate model sets used for each group of hypotheses were identical for both carcass survival and sign survival modeling.

We conducted logistic regression modeling to determine the influence of features on carcass detectability using the known fate model in program MARK. Because this model assumes perfect detection of individuals, known fate survival estimation using only 1 time interval is identical to estimating "success" probability from a binomial likelihood model. For example, the maximum likelihood estimate of survival for a known fate model with 1 time interval is

$$\hat{S} = \frac{x}{n}$$

where n is the number of individuals alive at the start of the interval, and x is the number of individuals alive at the end of the interval. This is identical to the maximum likelihood estimator for a binomial model, and as such when used in conjunction with the logit link function is identical to logistic regression in standard statistical software. Program MARK was used specifically in this analysis to facilitate model selection, calculation of model weights, model averaged parameter estimates, and reconstituted parameter estimates.

We generated logistic regression models of carcass detection probabilities using both local-scale and microhabitat variables. Local-scale features hypothesized to influence detection probability during fence-line surveys included habitat type, distance of the carcass from the fence, and observer experience (experienced vs. inexperienced) (Table 1). Additionally, a snowstorm abruptly developed during one detectability trial, severely limiting visibility. Therefore, a parameter for snow conditions was included to accommodate this confounding factor. All 15 combinations of additive models and the constant detectability model were compared for the experimental factors, and no interaction terms were considered in this group of hypotheses due to small sample sizes (Appendix B). Microhabitat characteristics used in modeling were identical to those used in survival analysis (Table 1). We compared 9 models representing features at the microhabitat scale (Appendix B). Once again these models were constructed such that no

strongly correlated predictor variables were included in the same model, and 2 hypothesized interaction terms (Grass Height*Shrub Height, Grass Height*Canopy Coverage) were included when their constituent terms were together among the top group of models (Table 1). Generalized likelihood-ratio R^2 values (R_L^2) were calculated for each survival and logistic regression model to determine the performance of individual models relative to the null constant models (Menard 2000). Lastly, prediction success for each model was estimated using crossvalidation procedures in SAS (SAS Version 9.2, Cary, North Carolina). Crossvalidation procedures re-fit each model dropping each data point in turn, and subsequently predicted the probability of detection for each dropped data point with the re-fit model. If the probability of detection was > 0.5 the point was predicted to be observed, and the reported success probability is the proportion of correct classifications in this analysis.

RESULTS

Survival

We monitored persistence of 100 hen pheasant carcasses over 2 study areas, and measured local-scale and microhabitat characteristics of the carcass locations. Mean grass height at carcass locations was 10.5 cm (SD = 8.8, *n* = 99), mean shrub height was 23.6 cm (SD = 21.0, *n* = 100), and mean canopy coverage was 22.5% (SD = 13.9%, *n* = 100). Visual concealment at a height of 1.0 m was used for survival models, and averaged 88.4% (SD = 16.7%, *n* = 99), whereas visual concealment at 1.5 m averaged 71.0% (SD = 26.0%, *n* = 45) and was used in detectability modeling. One microhabitat measurement was missing each for grass height and visual concealment, thus those factors had a sample size of ninety-nine. Shrub height was correlated with canopy coverage (*r* = 0.71), and visual concealment was correlated with all other measurements (*r* = 0.24-0.40), therefore these variables were not included together in modeling.

Average time to detection of first scavenging of carcasses was 5.8 days (SD 2.9, *n* = 100) for both areas combined, and appeared similar between sites (BB: = 5.6, SD = 3.0, *n* = 50; US: = 6.0, SD = 2.8, *n* = 50). Average time to detection of first scavenging was 5.1 days for big sagebrush (SD = 2.6, *n* = 50) and 6.6 days for little sagebrush (SD = 3.0, *n* = 50) habitats. We observed differences between the study sites in persistence of collision sign. Average number of days until sign was no longer detected on the US site (= 27.2, SD = 6.8, *n* = 50) was greater than the BB site (= 8.8, SD = 2.0, *n* = 49), due to the large number of carcasses with sign surviving the entire study period on the US site (*n* = 32; little sagebrush = 18, big sagebrush = 14). In contrast, none of the BB carcasses had sign persist the entire 31-day sampling period. Due to the large number of carcasses with sign persisting the entire study period, the average sign persistence calculated for the US site is biased low, and the true average length of sign persistence is unknown.

Pheasant carcasses in little sagebrush habitats or on the BB study area were more likely to be directly removed during initial scavenging, while carcasses in big sagebrush habitats and those on the US site were more likely to be scavenged in their original location and not directly removed. Eighty percent of carcasses on the BB site were directly removed during initial scavenging (big sagebrush = 68%, little sagebrush = 92%), whereas 82% of carcasses on the US

site were scavenged in their original location (big sagebrush = 88%, little sagebrush = 76%). Furthermore, of the US carcasses that had sign persist the entire sampling period, 100% of those located in big sagebrush, and 78% of those located in little-sagebrush habitats were first scavenged in their original location, and not directly removed. Nine of the carcasses (BB = 3, US = 6) had evidence of small mammal scavenging, commonly in the form of tracks and scat on or near the carcass. These carcasses commonly had small amounts of feathers plucked or removed, and the underlying tissue appeared gnawed upon. Arthropods were observed scavenging at 26 of the US carcasses (big sagebrush = 19, little sagebrush = 7), whereas we did not observe scavenging by arthropods at the BB site. We also observed arthropod scavengers carrying feather evidence away from carcass locations on the US site.

None of the hypothesized local-scale parameters received strong support in the carcass survival modeling, and the null constant survival model was most supported by the data ($\Delta QAIC_c = 0$, $w_i = 0.237$) (Table 2). We found minimal support for both habitat type and road presence influences on carcass survival ($\Delta QAIC_c < 2$), however, these models performed nearly identical to the constant survival model in terms of proportional increase in likelihood ($R_L^2 = 0.005$-0.009). Model averaged survival estimates showed a trend of slightly lower carcass daily survival in big sagebrush habitats ($\Delta DSR = 0.022$), and areas with no roads present ($\Delta DSR = 0.013$-0.014) regardless of habitat, however, the precision was low and the confidence intervals were wide and overlapping (Table 3). Reconstituted daily survival rates for the constant survival model showed low daily survival rates (DSR = 0.794, 95% CI = 0.721-0.851), resulting from the speed at which the carcasses were scavenged.

Similarly, carcass survival models were generated to evaluate the influence of microhabitat characteristics on survival, and these factors received little support from the data (Table 2). The top microhabitat model suggested constant carcass survival ($\Delta QAIC_c = 0$, $w_i = 0.316$), and we found weak support for grass height, visual concealment, and shrub canopy coverage influences on carcass survival ($\Delta QAIC_c < 2$). Again, these models performed nearly identical to the null constant survival model in terms of proportional increase in likelihood ($R_L^2 = 0$-0.007).

Local-scale sign survival modeling suggested survival was most influenced by study site ($\Delta QAIC_c = 0$, $w_i = 0.341$), and the site effect model was approximately 2.4 times more likely than the second best model (Table 4). We also found minimal support for both habitat type and distance of the carcass from the fence effects on sign survival in addition to study area effects ($\Delta QAIC_c < 2$). Local-scale sign survival models that did not contain a site effect parameter received virtually no support in this analysis ($\Delta QAIC_c > 25$), and all of the top models performed better than the null constant survival model in terms of proportional increase in likelihood ($R_L^2 = 0.226$-0.228). Model averaged sign survival rates showed strong differences between study sites, with daily survival rates ranging from 0.987-0.988 on the US site, and from 0.863-0.872 on the BB site (Table 5). Differences in model averaged sign daily survival rates resulted in reduced model averaged sign survival probabilities for the entire 31-day period from the US (0.673-0.699) to BB (0.011-0.015) study sites. Regression coefficient estimates for the site effect model again show the strong influence of site on sign daily survival rates ($\beta_1 = -2.528$, 95% CI = -3.566-(-)1.490), with lower survival at the BB site.

Similarly, sign survival models evaluated the influence of fine scale microhabitat features on daily survival rates. None of the hypothesized microhabitat features received strong support in this analysis, and the null constant survival model was most supported by the data ($\Delta QAIC_c = 0$, $w_i = 0.228$) (Table 4). We found weak support for the influence of all the microhabitat variables on sign survival, but again these models performed similarly to the null model in terms of proportional increase in likelihood ($R_L^2 = 0.003\text{-}0.023$) (Table 4).

Detection

We used detectability trials to estimate detection probabilities of carcasses and sign associated with fence sampling methods in sagebrush habitats. Because carcasses were planted the night preceding detection trials, some carcasses were not present during the trials. Three carcasses were removed prior to detection trials (2 in big sagebrush, 1 in little sagebrush), and 1 carcass in big sagebrush was buried under a snow drift by the time of the trial, therefore 4 carcasses were not used in our calculations. Furthermore, 10 of the original feather piles placed were either blown away or covered with snow prior to initiating detection trials and were not used; however, the 3 carcasses removed all left feather piles and these feathers were used in the feather pile detection calculations. Detection probability for feather piles was extremely low, only 1 of 43 (2.3%) feather piles present were located. No detectability models were constructed for feather piles because such a small proportion was located. The total proportion of detected carcasses was 0.54 ($n = 46$), and was higher in little sagebrush (0.71) than big sagebrush (0.36) habitats. Detection probability for 6 of the 22 carcasses in big sagebrush and 1 of the 24 carcasses in little sagebrush may have been confounded by snowfall during sampling. If these carcasses are excluded, the proportion of carcasses in big sagebrush detected rises slightly to 0.44. The proportion of detected carcasses was similar for experienced (54.8%, $n = 31$ carcasses) and inexperienced (53.3%, $n = 15$ carcasses) field searchers.

We used logistic regression models to evaluate local-scale and microhabitat factors influencing carcass detection probability during fence-line surveys. Habitat type influences on detectability were most supported by the data at the local scale ($\Delta AIC_c = 0$, $w_i = 0.306$) (Table 6), however, we found additional weak support for the influence of snow presence and carcass distance from the fence on detectability ($\Delta AIC_c < 2$). The top local-scale models all predicted detection probabilities moderately (0.674), and all 3 models performed identically in this measure. Model averaged detection probabilities were 0.67 for little sagebrush (95% CI = 0.43-0.85) and 0.40 for big sagebrush (95% CI = 0.20-0.65) habitats, and the regression coefficient from the habitat type model showed a positive influence of little sagebrush habitats on detection probability ($\beta_1 = 1.447$, 95% CI = 0.210-2.684).

Similarly, we evaluated the influences of microsite habitat characteristics on detection of carcasses during fence line surveys, and the influence of shrub height on detectability was most supported by the data ($\Delta AIC_c = 0$, $w_i = 0.323$) (Table 6). We found less support for the influence of shrub canopy cover, and grass height in addition to shrub height on detectability ($\Delta AIC_c < 2$) (Table 6). Both shrub height and canopy cover reduced detection probability (Fig 2), and the confidence interval for the regression coefficient from the shrub height model did not include zero ($\beta_1 = -0.039$, 95% CI = -0.077-(-)0.002). Both shrub height and canopy cover predicted

detection probabilities moderately well, however, the canopy cover model performed slightly better (0.652) than the shrub height model (0.609).

DISCUSSION

Survival

Carcasses experienced rapid scavenging on both study areas, which produced low daily survival rates. While the speed at which the carcasses were scavenged was similar between study sites, the manner of scavenging, daily survival rate of the collision sign as a whole and subsequent longevity of the collision sign differed between study areas. Additionally, microhabitat characteristics performed poorly in both carcass and sign survival models, suggesting broad scale site or landscape features may have a stronger influence on survival of collision evidence.

Rapid scavenging occurred in many scavenging studies, and is common for both avian (Crawford 1971, Houston 1986, Peterson et al. 2001) and mammalian carcasses (Heinrich 1988, Travaini et al. 1998). Crawford (1971) reported 93% of bird carcasses planted around a television tower in Florida were scavenged during the first night of observation. Houston (1986) studied scavenging by vultures in tropical forests and reported 96% of chicken carcasses were scavenged within 3 days. In British Columbia, 52 of 54 waterfowl carcasses placed in agriculture fields to simulate poisoning were removed within 72 hours (Peterson et al. 2001). The ability to locate and consume carrion rapidly provides benefits to many predator and scavenger species, and most vertebrate predators will opportunistically scavenge fresh carrion when available (Devault et al. 2003). Both coyotes and common ravens are opportunistic scavengers (Hilton 1978, Heinrich 1988) common to many shrub-steppe dominated landscapes. Thus, the rapid scavenging observed in our study should not be surprising, and likely occurs on many areas within shrub-steppe habitats.

In contrast to the rapid rate of initial scavenging on both study areas, overall sign survival rates differed between study sites. Similarly, 50% of planted quail remains were completely removed within 4 days in Alabama, whereas only 13% of quail (*Colinus virginianus*) remains in Texas were completely removed in that period (Rosene and Lay 1963). In addition to differences in sign survival rates, we recorded differences in the way carcasses were scavenged between the study sites, resulting in differences in the remaining carcass evidence. Carcasses that were directly removed often had few or no feather evidence remaining at the site, whereas carcasses scavenged at their initial location often had large feather piles ($\geq$ 100 feathers) and bird pieces scattered around the site. Similarly, Rosene and Lay (1963) found that large feather piles from quail disappeared at a slower rate than small feather piles. This is consistent with our observations, and suggests the way a carcass is scavenged may influence overall survival of sign at the original location.

Although site scale differences between study areas had a large influence on sign survival, microhabitat characteristics performed poorly in both carcass and sign survival models. Previous research has produced variable results with respect to the influence of habitat features on avian carcass persistence. Pain (1991) reported mallard (*Anas platyrhynchos*) carcass longevity was significantly lower for exposed carcasses than those concealed by vegetation. In contrast,

Bumann and Stauffer (2002) found no relationships between scavenging of ruffed grouse carcasses and habitat characteristics.

Detection

In contrast to survival models, habitat characteristics did influence detection probability of carcasses during fence-line surveys. Similarly, Tobin and Dolbeer (1990) indicated that the lowest detection rate (50%) for songbird carcasses in New York fruit orchards occurred at the site with the heaviest ground cover. Smallwood (2007) summarized the results from 10 unpublished reports at wind facilities, and found detection appeared to vary by bird group (i.e. large raptors, large non-raptors, etc.), whereas vegetation only influenced detection for birds classified as "small birds". Overall, our detection rate (0.53) appears low when compared to previous published studies. Savereno et al. (1996) found 66% and 73% of planted bird carcasses in a power-line corridor in coastal South Carolina, and Osborn et al. (2000) reported detection rates varied from 68.7% for small birds to 92.3% for large birds on a Minnesota wind facility.

Our detection rate of 2.3% for feather piles may be unrealistically low, as we located 4 actual avian collision sites during the trials (B.S. Stevens, University of Idaho, unpublished data), 3 of which were feather piles. Alternatively, if our measured detection probability for feather piles does accurately represent true feather pile detection, this would suggest the presence of a relatively large number of collision sites.

The applicability of our results to fence-line surveys for sage-grouse collision victims relies on the assumption that detection and survival probabilities are similar for sage-grouse and hen pheasants. Smallwood (2007) suggested researchers use the species of interest to avoid misleading results and application. However, Gehring et al. (2009) successfully used surrogate songbird carcasses when evaluating avian collision with communication towers in Michigan. Conservation concerns surrounding sage-grouse prevented us from obtaining 100 grouse carcasses to use in this study, and greater sage-grouse were recently listed as warranted but precluded under the Endangered Species Act by the U.S. Fish and Wildlife Service (United States Department of the Interior 2010). Pheasant carcasses used were relatively similar in body size and cryptic plumage, which should eliminate potential bias caused by these factors (Osborn et al. 2000, Smallwood 2007). Although a potential source of error could arise from slight differences in coloration between hen pheasants and sage-grouse, we are unaware of any studies that fully quantify the effects of small changes in coloration on avian carcass detection probability. Linz et al. (1991) reported significantly more male (83%) than female (78%) red-winged blackbirds (*Agelaius phoeniceus*) were found during carcass searches in cattail marshes, however, these results were not consistent across trials.

Sample sizes and length of survival trials used in carcass studies can also introduce bias in survival estimates due to predator swamping and data censoring (Smallwood 2007, Smallwood et al. 2010). Carcasses in this study were distributed across relatively large geographic areas, which should have reduced the potential for predator swamping. Further, the ability of mammalian predators to remove and cache carcasses (Stoddart 1970, Prior and Weatherhead 1991) should reduce the influence of predator swamping on carcass removal rates. Sign survival on the US study site was high due to the large number of carcasses with evidence persisting

throughout the entire study. Censoring sign survival data at the end of the study could lead to biased survival estimates (Smallwood 2007), and caution should be used extrapolating our results past the 31-day sampling interval length. Smallwood (2007) suggested that carcass survival studies monitor all carcasses on a daily basis. However, rigorous statistical models are available for unbiased estimation of daily survival rates for variable time interval monitoring (Dinsmore et al. 2002, Rotella et al. 2004, Shaffer 2004), and longer intervals between monitoring periods would likely reduce the chances for observer effects on daily survival rates (e.g. Rotella et al. 2000).

MANAGEMENT IMPLICATIONS

This study provides the first estimates of avian carcass survival and detection probabilities associated with fence-line surveys in sagebrush-steppe habitats. Recommendations for standardized searches are difficult given the regional variability in collision sign survival documented in this study. However, low daily survival rates for carcasses and collision sign suggest time-interval lengths between surveys should be a maximum of 1-2 weeks to avoid the potential negative effects of survival bias on collision rate estimation. For small-scale studies it may be possible to sample on a weekly or bi-weekly basis, however, studies estimating collision rates at landscape scales may not be able to sample at such frequencies. Regional variation in sign survival also suggests the need to estimate survival on all study areas. Survey protocols should be standardized to avoid sampling when weather conditions are poor and could influence detection probabilities (i.e. snow cover, extreme wind or rain, etc.), and collision rates should be corrected based on identified site attributes influencing detection. Moreover, caution is warranted when aggregating or comparing un-corrected collision data from sites with varying vegetation characteristics, as detection probabilities are likely different between the sites.

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Table 1. Parameters used, and justification for parameter use, in survival and detectability modeling for carcasses placed along fences on the Browns Bench and Upper Snake regions of Idaho, USA, during spring 2009.

Parameter	Justification for hypothesized parameter
Local-scale Models	
Site[a]	Differences in landscape context
Habitat type[a,b]	Structural/concealment/height differences
Road presence[a]	Potential predator space use[d]
Distance of carcass from fence[a,b]	Potential predator space use[d]
Snow presence[b]	Covers collision evidence
Observer[b]	Experience during surveys may influence efficiency
Site*Habitat[a,c]	Landscape context effects may vary by vegetation community
Site*Road[a,c]	Road effects may vary by local community
Habitat*Road[a,c]	Road effects may vary by habitat type
Microhabitat Models	
Shrub height[a,b]	Documented effects on predation[e], visual obstruction
Shrub canopy coverage[a,b]	Documented effects on predation[f,g], visual obstruction
Grass height[a,b]	Documented effects on predation[f,h], visual obstruction
Visual concealment[a,b]	Documented effects on predation[g], visual obstruction
Grass height*Shrub height[a,b,c]	Shrub concealment effects may vary by grass height
Grass height*Canopy coverage[a,b,c]	Shrub coverage effects may vary by grass height

[a] Parameter used in survival modeling.
[b] Parameter used in logistic regression modeling.
[c] Interaction terms were only added when the individual terms were in a model together among the top group of models ($\Delta AIC_c \approx \leq 2$).
[d] Bradley and Fagre (1988).
[e] Gregg et al. (1994).
[f] Holloran et al. (2005).
[g] Coates et al. (2010).
[h] Moynahan et al. (2007).

Table 2. Top models of daily survival rate of carcasses placed along fences as a function of local-scale and microhabitat characteristics on the Browns Bench and Upper Snake regions of Idaho, USA, during spring 2009.

Model	QDeviance	K	QAICc	Δ QAICc	w_i	Likelihood	R_L^{2} [a]
Local-scale Models[b]							
$S_{(.)}$	96.135	1	98.145	0.000	0.237	1.000	N/A
$S_{(Habitat)}$	95.233	2	99.263	1.118	0.135	0.572	0.009
$S_{(Road)}$	95.637	2	99.667	1.522	0.111	0.467	0.005
$S_{(Site)}$	96.129	2	100.158	2.013	0.087	0.366	0.000
$S_{(Dist)}$	96.135	2	100.165	2.020	0.086	0.364	0.000
Microhabitat Models[c]							
$S_{(.)}$	95.720	1	97.729	0.000	0.316	1.000	N/A
$S_{(GH)}$[d]	95.041	2	99.070	1.341	0.162	0.512	0.007
$S_{(VC)}$[e]	95.614	2	99.643	1.914	0.122	0.384	0.001
$S_{(CC)}$[f]	95.690	2	99.720	1.990	0.117	0.370	0.000
$S_{(SH)}$[g]	95.719	2	99.748	2.019	0.115	0.364	0.000

[a] $R_L^2 = 1-(\ln(L_m)/\ln(L_o))$, where L_m = Maximized likelihood for model of interest, and L_o = Maximized likelihood for intercept only model (Menard 2000).
[b] Overdispersion ($\hat{c}$) was estimated for this group of models as the deviance divided by the deviance degrees of freedom. For this group of models $\hat{c}$ = 3.141.
[c] Overdispersion ($\hat{c}$) was estimated for this group of models as the deviance divided by the deviance degrees of freedom. For this group of models $\hat{c}$ = 3.155.
[d] GH = grass height microhabitat variable.
[e] VC = visual concealment microhabitat variable.
[f] CC = percent shrub canopy coverage microhabitat variable.
[g] SH = shrub height microhabitat variable.

Table 3. Model averaged estimates of local-scale carcass daily survival rates on the Browns Bench and Upper Snake regions of Idaho, USA, during spring 2009. Groups represent 8 combinations of 3 binary classification variables, representing site (BB = Browns Bench, US = Upper Snake), habitat type (LS = little sagebrush, BS = big sagebrush), and road presence (NR = no road, RP = road present).

		95% CI	
Group	Daily survival	Lower	Upper
BB LS NR	0.798	0.694	0.873
BB LS RP	0.811	0.699	0.888
BB BS NR	0.776	0.661	0.860
BB BS RP	0.789	0.655	0.881
US LS NR	0.798	0.691	0.875
US LS RP	0.812	0.703	0.887
US BS NR	0.777	0.661	0.861
US BS RP	0.790	0.661	0.879

Table 4. Top models of daily survival rate of collision sign placed along fences as a function of local-scale and microhabitat characteristics on the Browns Bench and Upper Snake regions of Idaho, USA, during spring 2009.

Model	QDeviance	K	QAICc	Δ QAICc	w_i	Likelihood	R_L^{2}[a]
Local-scale Models[b]							
$S_{(Site)}$	94.597	2	98.605	0.000	0.341	1.000	0.226
$S_{(Site+Dist)}$	94.298	3	100.313	1.708	0.145	0.426	0.228
$S_{(Site+Habitat)}$	94.322	3	100.337	1.732	0.143	0.421	0.228
$S_{(Site+Road)}$	94.596	3	100.611	2.007	0.125	0.367	0.226
$S_{(Site+Habitat+Dist)}$	94.002	4	102.027	3.422	0.062	0.181	0.231
Microhabitat Models[c]							
$S_{(.)}$	97.444	1	99.446	0.000	0.228	1.000	N/A
$S_{(GH)}$[d]	95.933	2	99.940	0.494	0.178	0.781	0.016
$S_{(CC)}$[e]	96.348	2	100.356	0.909	0.145	0.635	0.011
$S_{(SH)}$[f]	97.061	2	101.068	1.622	0.101	0.444	0.004
$S_{(VC)}$[g]	97.173	2	101.180	1.734	0.096	0.420	0.003
$S_{(GH+CC)}$	95.233	3	101.248	1.801	0.093	0.406	0.023

[a] $R_L^2 = 1-(\ln(L_m)/\ln(L_o))$, where L_m = Maximized likelihood for model of interest, and L_o = Maximized likelihood for intercept only model (Menard 2000).
[b] Overdispersion ($\hat{c}$) was estimated for this group of models as the deviance divided by the deviance degrees of freedom. For this group of models $\hat{c}$ = 3.216.
[c] Overdispersion ($\hat{c}$) was estimated for this group of models as the deviance divided by the deviance degrees of freedom. For this group of models $\hat{c}$ = 4.032.
[d] GH = grass height microhabitat variable.
[e] CC = percent shrub canopy coverage microhabitat variable.
[f] SH = shrub height microhabitat variable.
[g] VC = visual concealment microhabitat variable.

Table 5. Model averaged estimates of collision sign daily survival rates on the Browns Bench and Upper Snake regions of Idaho, USA, during spring 2009. Groups represent 8 combinations of 3 binary classification variables, representing site (BB = Browns Bench, US = Upper Snake), habitat type (LS = little sagebrush, BS = big sagebrush), and road presence (NR = no road, RP = road present).

		95% CI	
Group	Daily survival	Lower	Upper
BB LS NR	0.872	0.783	0.928
BB LS RP	0.872	0.764	0.935
BB BS NR	0.863	0.770	0.922
BB BS RP	0.863	0.746	0.931
US LS NR	0.988	0.970	0.996
US LS RP	0.988	0.969	0.996
US BS NR	0.987	0.967	0.995
US BS RP	0.987	0.966	0.995

Table 6. Top logistic regression models of carcass detection probability during fence collision surveys as a function of local-scale and microhabitat characteristics on the Browns Bench and Upper Snake regions of Idaho, USA, during spring 2009.

Model	Deviance	K	AICc	Δ AICc	w_i	Likelihood	R_L^{2a}	Prediction Success[b]
Local-scale Models								
$P_{(Habitat)}$	57.816	2	62.095	0.000	0.306	1.000	0.088	0.674
$P_{(Habitat+Snow)}$	57.146	3	63.717	1.623	0.136	0.444	0.099	0.674
$P_{(Habitat+Dist)}$	57.424	3	63.996	1.901	0.118	0.387	0.095	0.674
$P_{(Habitat+Observer)}$	57.795	3	64.367	2.272	0.098	0.321	0.089	0.674
$P_{(Snow)}$	61.178	2	65.457	3.362	0.057	0.186	0.035	0.609
Microhabitat Models								
$P_{(SH)}$[c]	57.329	2	61.608	0.000	0.323	1.000	0.096	0.609
$P_{(CC)}$[d]	57.931	2	62.210	0.602	0.239	0.740	0.087	0.652
$P_{(GH+SH)}$[e]	56.980	3	63.551	1.943	0.122	0.379	0.102	0.587
$P_{(GH+CC)}$	57.332	3	63.904	2.296	0.102	0.317	0.096	0.652
$P_{(VC)}$[f]	60.385	2	64.664	3.056	0.070	0.217	0.048	0.565

[a] $R_L^2 = 1-(\ln(L_m)/\ln(L_o))$, where L_m = Maximized likelihood for model of interest, and L_o = Maximized likelihood for intercept only model (Menard 2000).
[b] Prediction success calculated via crossvalidation in PROC LOGISTIC, PREDPROBS=CROSSVALIDATE output statement (SAS Version 9.2, Cary, North Carolina).
[c] SH = shrub height microhabitat variable.
[d] CC = percent shrub canopy coverage microhabitat variable.
[e] GH = grass height microhabitat variable.
[f] VC = visual concealment microhabitat variable.

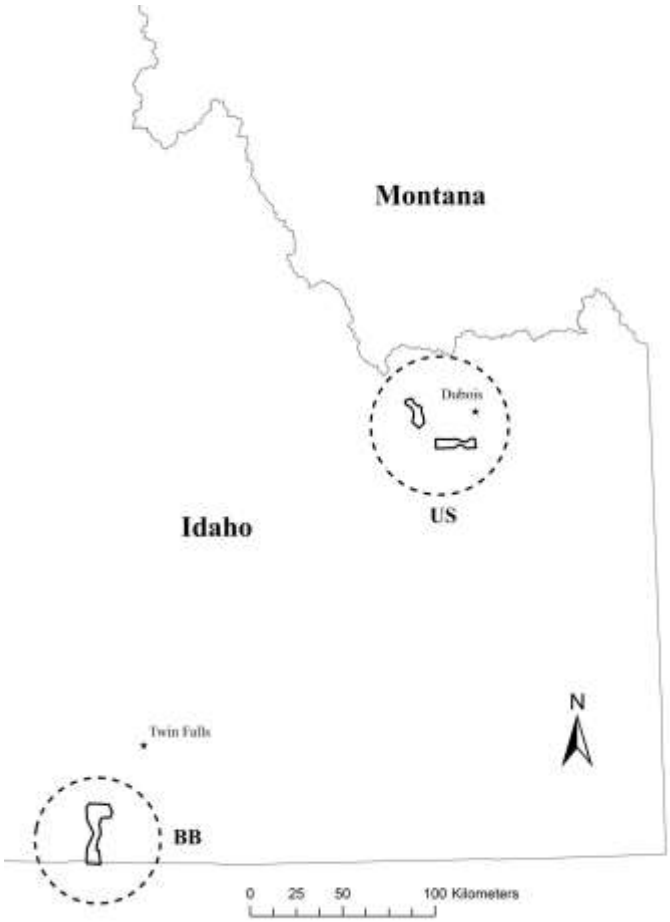
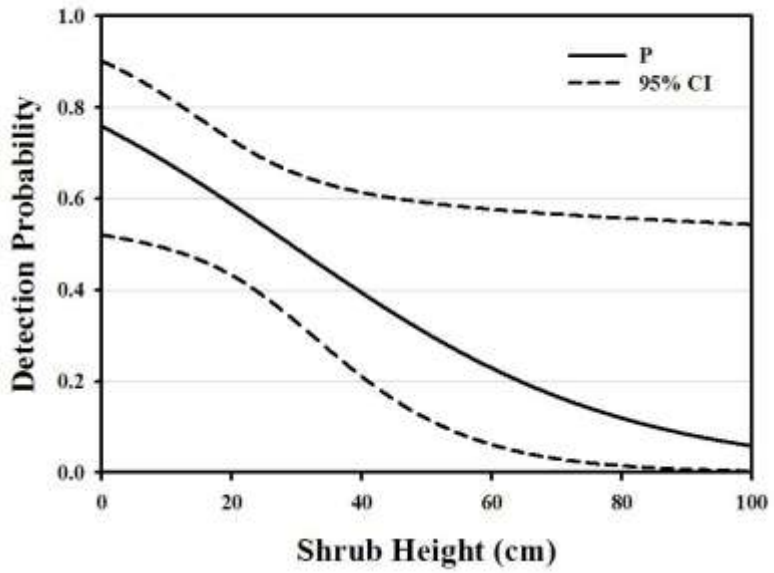
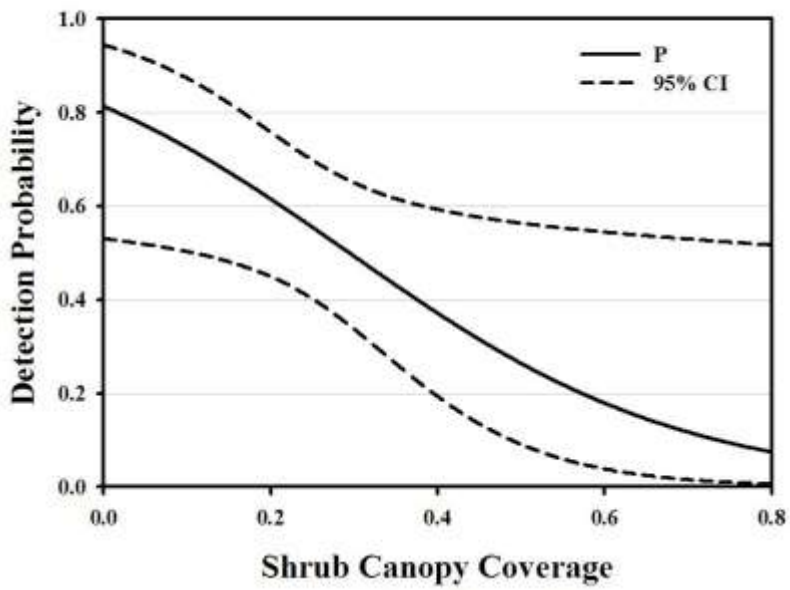


Figure 1. Southern Idaho, USA, study areas, where we studied survival and detectability of hen pheasant carcasses planted as hypothetical collision victims. Dashed lines represent the 2 study regions used in this study (BB = Browns Bench, US = Upper Snake) during the spring of 2009.



a



b

Figure 2. Plots of carcass detection probability during fence surveys as a function of microhabitat characteristics on the Browns Bench region of Idaho, USA, during 2009. (a) Carcass detection probability as a function of shrub height from the top microhabitat detectability model. (b) Carcass detection probability as a function of shrub canopy coverage from the second best microhabitat detectability model.

STUDY I; PART IV:

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Use of dwarf sagebrush by nesting greater sage-grouse

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ABSTRACT I investigated habitat characteristics for 156 Greater Sage-Grouse (*Centrocercus urophasianus*) nests and 138 random plots among eight populations in southern Idaho, US, during 2003-2005. In addition to traditional habitat measurement methods, I included two visual obstruction variables observed from the nesting female's perspective - effective height and horizontal cover. Independent, uncorrelated variables were obtained from principal components analysis for two sub-categories of habitat; habitat throughout southern Idaho and sites dominated by dwarf species of sagebrush (little sagebrush *Artemisia arbuscula* and black sagebrush *A. nova*). Height of shrubs and shrub canopy cover accounted for the single most variability among southern Idaho habitat and sites dominated by dwarf sagebrush, respectively. Shrub density and horizontal cover were the second and third factors contributing to the variability for both sub-categories. Grouse used sites with greater shrub height and canopy cover than available at random. Successful nests tended to have greater shrub density and somewhat greater horizontal cover than random points. Successful yearlings had the greatest horizontal cover among the age and fate categories of nests. A gradient of shrub heights was observed with dwarf sagebrush being the shortest while mesic sites with mixed shrubs and sites dominated by three-tip sagebrush (*A. tripartita*) being the tallest. Despite lacking height, dwarf sites maintained similar density and horizontal cover as other shrub sites. Logistic regression models were developed using variables obtained from principal components analysis. Horizontal cover 3 m from the center of the plot, alone, was the best model for separating nests from random plots in southern Idaho habitat and had the best classification rate among dwarf sagebrush habitat. Habitat measurements of dwarf sagebrush dominated nest sites were within the habitat guidelines used for managing Greater Sage-Grouse and is the first time this habitat type has been described. Nests (N = 46) in habitat dominated by dwarf sagebrush had 43% success and nests with dwarf sagebrush species concealing the nest bowl (N = 22) had 50% nest success. Certain grass and shrub species have similar overall heights but considerably different effective concealment cover. Grouse researchers should consider using effective height and horizontal cover habitat measurements in future research of shrub-steppe grouse.

KEY WORDS *Artemisia*, black sagebrush, *Centrocercus urophasianus*, effective height, Greater Sage-Grouse, habitat use, horizontal cover, little sagebrush, nest habitat.

Greater Sage-Grouse (*Centrocercus urophasianus*) populations have declined throughout their range (Connelly and Braun 1997, Connelly et al. 2004), and their distribution is greatly influenced by the occurrence of shrub-steppe habitat types, especially those dominated by sagebrush (Patterson 1952, Connelly and Braun 1997). Past research on nest habitat of Greater Sage-Grouse in Idaho (Connelly et al. 1991) and elsewhere (Hagen et al. 2007) has focused on landscapes dominated by taller structured sagebrush like Wyoming big sagebrush (*Artemisia*

tridentata wyomingensis), mountain big sagebrush (*A. t. vaseyana*), and silver sagebrush (*A. cana*). Little is known about nesting by Greater Sage-Grouse in habitat dominated by dwarf sagebrush like little sagebrush (*A. arbuscula*) and black sagebrush (*A. nova*) which are more associated with winter cover (Dalke et al. 1963, Beck 1977). Gregg et al. (1994) sampled three successful nests in little sagebrush habitat and Popham and Gutiérrez (2003) observed four nests in little sagebrush. Current guidelines for managing Greater Sage-Grouse habitat are also based on studies conducted in areas largely dominated by taller sagebrush species (Connelly et al. 2000). Providing values for nest use in dwarf sagebrush habitat and comparing with current guidelines will assist in managing this unique habitat to benefit Greater Sage-Grouse.

Traditional habitat measurement methods have focused on droop height of the tallest structure of a grass or shrub (Connelly et al. 2003), overlooking the possible importance of visual obstruction provided by individual plants. Plants can have the same height but provide different horizontal concealment cover depending on their growth structure, soil and climate differences, and land management practices. Concealment cover is important for several aspects of grouse survival (Mussehl 1963, Bergerud and Gratson 1988), especially for nesting Greater Sage-Grouse (Hagen et al. 2007).

Nesting Greater Sage-Grouse females are known to take periodic incubation breaks to forage and defecate away from the nest (Coates and Delehanty 2008). These forays risk exposing the female and her nest to predation especially if concealment cover is lacking. Measuring horizontal cover from the nesting female's perspective is a novel approach and may explain selection of a nest site for optimum concealment during incubation breaks.

My objective was to describe habitat use immediately around nests of Greater Sage-Grouse in southern Idaho, focusing on areas dominated by dwarf sagebrush species for comparisons to sites dominated by other shrub species. I also wanted to determine the usefulness of two new habitat measurement methods to describe cover measured from a nesting female's perspective in addition to methods previously used.

STUDY AREAS

This research was conducted on multiple study areas throughout southern Idaho (Fig. 1), previously described by other researchers (Klott et al. 1993, Wik 2002, Lowe 2006, Shepherd 2006), and ranging in elevation from 1,600-2,400 m. Most of the 20-40 cm of annual precipitation in southern Idaho occurs during the winter and spring. The landscape in general, is flat to rolling shrub-steppe habitat on valley floors, steeper slopes along valley edges, increasing in elevation to base of mountain ranges. Mesic sites are typically on west sides of mountain ranges with deeper soils whereas xeric sites are on east sides within the rain shadow in shallow soils. Depending on the moisture regime and soil depth, several sagebrush species dominate the cover including Wyoming big sagebrush, mountain big sagebrush, basin big sagebrush (*A. t. tridentata*), three-tip sagebrush (*A. tripartita*), little sagebrush, and black sagebrush. Non-sagebrush shrubs include antelope bitterbrush (*Purshia tridentata*), green rabbitbrush (*Chrysothamnus viscidiflorus)*, saltbush (*Atriplex* spp.), spiny hopsage (*A. spinosa*), and horsebrush (*Tetradymia canescens*). Common grasses include: bluebunch wheatgrass (*Pseudoroegneria spicata*), Idaho fescue (*Festuca idahoensis*), cheatgrass (*Bromus tectorum*),

western wheatgrass (*Agropyron smithii*), crested wheatgrass (*A. cristatum*), basin wildrye (*Elymus cinereus*), Sandberg bluegrass (*Poa secunda*), Indian ricegrass (*Achnatherum hymenoides*), bottlebrush squirreltail (*Elymus elymoides*), and needle-and-thread grass (*Stipa comata*). Common forbs include: western yarrow (*Achillea millefolium*), wild onion (*Allium* spp.), pussytoes (*Antennaria* spp.), arrow-leaf balsamroot (*Balsamorhiza sagittata*), hawksbeard (*Crepis* spp.), buckwheat (*Eriogonum* spp.), lettuce (*Lactuca* spp.), lupine (*Lupinus* spp.), phlox (*Phlox* spp.), salsify (*Tragopogon* spp.), and mules ears (*Wyethia amplexicaulis*).

METHODS

Habitat Sampling

This research was conducted on eight breeding populations of Greater Sage-Grouse among two moisture regimes and three shrub species categories distributed throughout southern Idaho. Xeric sites have annual precipitation <30 cm/yr whereas Mesic sites have >30 cm/yr and are dominated by a variety of sagebrush and non-sagebrush species (Hironaka et al. 1983). Sites were also categorized according to dominance of sagebrush height based on grouping by Hironaka et al. (1983); tall (*A. tridentata* spp.) and dwarf (little and black sagebrush). I also considered three-tip sagebrush as a separate category. The shrub species with the largest canopy cover at a plot was considered dominant.

Greater Sage-Grouse females were captured by night lighting (Giesen et al. 1982, Wakkinen et al. 1992) and fitted with 16.5 g necklace style radio transmitters (Riley and Fistler 1992) and monitored during laying and incubation as part of other ongoing studies. Age-class was categorized as yearling (<1 yr old) and adult (>1 yr old) and determined by wing characteristics (Eng 1955, Dalke et al. 1963). A nest was considered successful if at least one egg hatched. Random point coordinates, independent from nest sites, were generated using ArcView Spatial Analyst (ESRI, Redlands CA, 92373) and located in the field with hand-held GPS units. Random points were centered on the coordinates to avoid observer bias and eliminate over estimation of available shrub canopy cover and density. Random plots were sampled within known boundaries of nesting Greater Sage-Grouse based on past and current research in Idaho.

Habitat was measured at nest sites within one week after females ceased nesting efforts or when abandoned or depredated nests would have hatched based on initiation dates and a 27 day incubation period (Patterson 1952). Random plots were measured throughout the hatching season and at least one plot was measured within a week of measuring a nest plot. Habitat measurements were sampled along four, 10 m transects placed at right angles radiating from the center of the nest and oriented in a random direction similar to Wakkinen (1990), Gregg et al. (1994), and Musil et al. (1994). Droop height of the closest shrub and grass was measured for each plant species within 1 m of the transect at 1, 3, and 5 m from the center of the plot for each of the four transects. Droop height is defined as the tallest naturally growing portion of the plant (Connelly et al. 2000; 2003). Sagebrush was identified according to a dichotomous key (Hironaka et al. 1983) but subspecies and hybrids were possibly misidentified by not following more discriminating methods (Stevens and McArthur 1974, Shumar et al. 1982, Rosentreter and Kelsey 1991). Droop height of residual (previous season growth excluding flower stalks), live (current green and growing leaves), flower stalk (tallest flower structure), and number of flower

stalks was measured for each grass species separately. Maximum height (tallest structure of entire plant) was generated post hoc by selecting maximum measurements from those recorded in the field. Effective height was measured by placing a meter stick behind the grass or shrub opposite the nest and estimating the tallest height concealing >50% of the 2.5 cm wide meter stick.

Horizontal cover using a cover pole outside of the nest bowl was measured at 1, 3, and 5 m from the plot center along the transects and read from 20 cm above the ground immediately outside of the nest shrub or at the center of a random plot (Robel et al. 1970, Fig. 2). At least one-half of a 2.54 cm tall segment (48 segments/pole) had to be obscured by vegetation to be counted as covered. Measurements were separated post hoc according to the following segment categories: low (0-18 cm), medium (19-61 cm), high (62-122 cm), ground (0-61 cm) and total (0-122 cm). Readings from the 1 m position were later discarded so nests and random plots could be compared.

Shrub canopy cover and shrub density was measured along the 10 m transects and segmented into 0-1, 1-3, 3-5, 5-10, and 1-10 m increments (Canfield 1941). To eliminate over-sampling the center and to compare random plots to nests, the 0-1 m increment was discarded during subsequent analysis. Gaps in the canopy >5 cm were excluded (Connelly et al. 2003). Shrub density was sampled by counting the number of plants of each shrub species intersecting or within 0.5 m on both sides for the length of the transects and separated into the same segments used for canopy coverage.

Understory cover for each forb and grass species was measured with a 40 x 50 cm modified cover frame (Daubenmire 1959) at 1, 3, and 5 m from the center on each of the four transects. Cover canopies were modified from Daubenmire (1959) to include more sensitivity for lower cover values for the following cover classes: 1 (0-1%), 2 (2-5%), 3 (6-25%), 4 (26-50%), 5 (51-75%), and 6 (76-100%). Slope and aspect were measured using a clinometer and compass, respectively. Elevation was estimated by plotting locations on 7.5 minute quadrangle maps.

Statistical Analysis

Principal components analysis was used to reduced the set of correlated habitat variables into independent components using SAS (SAS Institute 2001, McGarigal et al. 2000). My goal was to test the statistical hypothesis there is no difference in vegetation between use and non-use by nesting Greater Sage-Grouse. The biological hypothesis tested was that Greater Sage-Grouse were selecting nesting habitat for concealment cover. Varimax rotation facilitated interpretation of the variables within the components (O'Rourke et al. 2005). Meaningful factor loadings were set at ±0.40. Variables were considered complex and were removed if they significantly loaded on >1 component. Analyses were run sequentially until all complex variables were identified and eliminated. Post hoc analyses were also conducted by selecting a subset of the data to determine variables best describing nest use for habitat dominated by dwarf sagebrush.

In an attempt to avoid model dredging (Guthery et al. 2005), I identified variables from the first principal components describing at least 50% of the cumulative variability for subsequent use in developing predictive models of nest use analyzed with logistic regression (Afini and Clark

1996). One variable with the greatest loading from each principal component was selected to represent the component in the model. When loadings were tied, the variable with the greatest ease of interpretation for land management was selected. Once a list of variables was chosen, I conducted a model search using a SAS macro (All Possible Logistic Regressions; available from C. T. Moore – see White et al. 2005) and used corrected Akaike's Information Criterion (AIC_c) scores to rank the models (Burnham and Anderson 2002), according to methods described by White et al. (2005). The models were also ranked according to Akaike weights (w_i) and the best model picked with the largest weight (Burnham and Anderson 2002). For each model, I used a SAS macro (Monte Carlo Cross-Validation for Logistic Regression; available from C. T. Moore – see White et al. 2005) to determine the predictive accuracy for each model $\leq \Delta AIC_c = 2$ of the best model and set the number of iterations at 1,200. I set the proportion of the data withheld based on the number of variables in the model using criteria described by White et al. (2005): 50%, 40%, and 35% of the data withheld for 1-, 2-, and 3-variable models, respectively..

RESULTS

I measured 156 Greater Sage-Grouse nests and 138 random plots from eight grouse breeding populations during 2003-2005 in southern Idaho, including 89 vegetation variables. Forty-four percent (N = 68) of the 156 nests successfully produced chicks. Adults comprised 77% of the nests sampled and of those, 43% were successful whereas 40% of the 35 yearling nests were successful. Overall apparent success of nests sampled was 34% (N = 62), 49% (N = 46), and 44% (N = 48) in 2003, 2004, and 2005, respectively. For nest sites in dwarf sagebrush dominated habitat, I sampled 21 successful and 24 unsuccessful nests. Of the 22 nests with dwarf sagebrush directly over the nest bowl, 11 were successful. Analyses are separated between the data for plots sampled throughout all habitat types in southern Idaho and for the subset of plots dominated by dwarf sagebrush.

Principal components analysis

Southern Idaho habitat. − Eleven variables associated with horizontal cover near the ground as well as live, effective, and maximum grass heights were identified as complex variables and were removed from principal components analysis, but 78 other variables were retained. Sixteen principal components met the minimum Eigenvalue of 1.0 and described 84% of the variance in the data. A majority of the variance (55%) was described by the first five principal components (Table 1). Eight shrub height variables were associated with the first principal component (Prin I) and accounted for 23% of the variance, Prin II had eight variables of shrub density accounting for 11% of variance, Prin III with eight variables of horizontal cover and 8% of variance, Prin IV with eight variables of shrub canopy cover and 7% of variance, and Prin V had five forb cover variables describing 6% of the variance.

Viewing the first three components three-dimensionally (Fig. 3a), nests were separated from random plots by having greater shrub height (Prin I) and shrub density (Prin II). Successfully hatched nests of yearling females appeared to have the greatest horizontal cover (Prin III) and shrub density. Successful adult nests had shorter shrub heights than successful yearlings, while maintaining similar shrub density. Unsuccessful nests of both adults and yearlings tended to have less horizontal cover than successful nests but similar values to random plots.

Grouping plots by sagebrush type and moisture regimes for the first three principal components showed a gradient of shrub height from dwarf dominated plots to mesic and three-tip sagebrush plots (Fig. 4). There also appears to be a gradient along the Prin III axis with mesic sites having the least and three-tip sagebrush having the most horizontal cover. Mesic random plots had the least shrub density while maintaining tall shrub height along with mesic nest sites. Dwarf sagebrush sites, though lacking the height, maintained similar shrub density and horizontal cover to the taller sites.

Dwarf sagebrush habitat. − Analysis of 46 nests and 43 random plots dominated by dwarf sagebrush cover showed 12 variables were considered complex and were removed leaving 77 variables in the analysis. Complex variables were associated with the number of grass and forb species and heights of shrubs and grasses. Ten principal components met the minimum Eiganvalue of 1.0 and accounted for 80% of the variability. A majority (53%) of the variance was described by the first four principal components (Table 2). Prin I had eight variables associated with shrub canopy cover and accounted for most (19%) of the variability. Shrub density was described by eight variables for Prin II and accounted for 15% of the variability. Prin III was associated with seven variables of horizontal cover with 10% of the variability. Five variables of grass height and one variable of horizontal cover were associated with Prin IV and described 9% of the variability.

Comparing the first three principal components three-dimensionally (Fig. 3b), nests of unsuccessful yearling females tended to have the least shrub density but similar horizontal cover to unsuccessful adults, whereas successful yearlings had the greatest horizontal cover and highest shrub canopy cover. Successful adult nests tended to have the greatest shrub density but shrub canopy cover was more similar to random plots than other nests. Random dwarf plots tended to have the lowest shrub canopy coverage but similar horizontal cover to adult nests.

Logistic regression

Southern Idaho habitat. − The five variables with highest loading on the first five principal components, one from each component, were used to develop and compare models from all possible combinations of these variables to separate nests from random plots. Even though both variables had identical loadings for Prin II, density of sagebrush 1-10 m from center of plot was chosen to represent Prin II instead of sagebrush density at 3-5 m due to greater ease of interpretability for landscape management. The model with the lowest AIC_c score and greatest weight (w_i) had one variable (horizontal cover 3 m from center of plot) and had a classification rate of 59.3% (Table 3). Additional variables did not improve the classification rate.

Dwarf sagebrush habitat. − Horizontal cover 3 m from center of the plot and height of grass flower stalks best separated nests from random plots in dwarf sagebrush dominated habitat (Table 4). Horizontal cover alone had a better classification rate than the minimum AIC_c model. Models with additional variables had higher AIC_c and lower w_i values as well as poorer classification rates.

Dwarf sagebrush nest characteristics

I sampled 46 Greater Sage-Grouse nests and 43 random plots that were dominated by dwarf sagebrush canopy cover (Table 5). Dwarf sagebrush tended to be shorter than other sagebrush by about 20 cm at nest sites. Nest sites tended to have greater overall shrub height than available at random. Plant height over the nest was greater than average shrub heights 1-10 m from the nest. Females were using about 20 cm taller dwarf sagebrush to conceal over the nest than the average height of dwarf sagebrush in the vicinity of the nest. Nest sites had >2 cm taller effective and residual grass heights than random plots but had similar maximum grass heights. Shrub densities, canopy cover, and grass and forb cover were similar between nests and random points. Nests appeared to have greater horizontal cover 3 m from the center of the plot than available at random. Slope and aspect were similar but nests were located at slightly lower elevations than random plots.

Use of effective height as a variable

Depending on the grass species, tall grasses like bluebunch wheatgrass, crested wheatgrass, and Idaho fescue tended to have a greater proportion of the maximum height providing effective horizontal cover than shorter stature species like cheatgrass and Sandberg bluegrass (Table 6). Needle-and-thread and Indian ricegrass, though species of taller stature, provided less horizontal cover than bottlebrush squirreltail, a shorter species with dense foliage. Crested wheatgrass provided the tallest horizontal cover whereas cheatgrass provided the shortest.

Effective height for shrub species was related to the overall height (Table 7). Little sagebrush, black sagebrush, spiny hopsage, and green rabbitbrush were low stature and tended to have shorter effective heights. The proportion of the plant providing effective cover also varied with the flower structure. Shrubs with short flower stalks, like antelope bitterbrush and spiny hopsage which have flowers embedded within the branches, tended to have a higher proportion of the plant providing concealment cover. Species with long flower stalks providing maximum height well above the main structure of the shrub, like basin big sagebrush and Wyoming big sagebrush, tended to have a lower proportion of the overall height providing effective concealment cover. Mountain big sagebrush had a greater effective concealment cover percentage than other species of sagebrush.

Nest habitat use

Descriptions of Greater Sage-Grouse nests in dwarf sagebrush dominated habitat have been previously limited to few nest sites (Gregg et al. 1994, Popham and Gutiérrez 2003) or at the landscape scale (Shepherd 2006). Popham and Gutiérrez (2003) found Greater Sage-Grouse avoided dwarf sagebrush nesting habitat but Shepherd (2006) found almost equal proportions of nests in little and big sagebrush landscapes. In my study, nest sites in dwarf sagebrush habitat accounted for 29% of the nests sampled and were within the established guidelines for sage-grouse breeding habitat characteristics (Connelly et al. 2000), despite being dominated by shorter stature species of sagebrush. Instead of shrub height, shrub canopy coverage was the factor most important in describing the variation in the data. Females also used taller sagebrush immediately over the nest for concealment, as has been found for studies in habitats dominated by taller

sagebrush species (Wallestad and Pyrah 1974, Petersen 1980, Wakkinen 1990, Musil et al. 1994, Apa 1998). Females in dwarf sagebrush also tended to use taller grass, similar to results of Wakkinen (1990), Gregg et al. (1994), and Sveum et al. (1998). Use does not necessarily correlate with fitness at a landscape scale (Aldridge and Boyce 2007), but 43% of the nests we sampled in dwarf sagebrush habitat successfully hatched as did 50% of the nests concealed by dwarf sagebrush over the nest bowl. These rates are average for nest success of Greater Sage-Grouse (Bergerud 1988, Schroeder et al. 1999), showing that dwarf sagebrush can provide productive nesting habitat.

This study is the first to compare Greater Sage-Grouse nesting habitat use across multiple habitat regimes. The models did not perform well having a <70% correct classification rate between nest site and random plots. Also, the top variables used in the models accounted for relatively low amounts of the variability in the data, perhaps due to similarity in available vegetation within the habitat types. Klebenow (1969) had a similar problem when describing Greater Sage-Grouse nest habitat in Idaho. Despite these shortcomings, some interesting patterns were revealed with my data. It appears Greater Sage-Grouse obtain concealment cover for nests depending on the height of the dominant sagebrush species available. Within tall sagebrush, females chose sites with greater shrub density than random points, but with similar shrub heights. The same was true for mixed shrubs on mesic sites except this habitat provided taller shrubs, probably because robust antelope bitterbrush was more common. Three-tip sagebrush nest sites had shrub heights similar to mesic sites but were among the densest shrub values across the habitat types sampled. Females selected nest sites in dwarf sagebrush habitat with more overhead cover, from the shrub canopy, possibly to compensate for lack of brush height. Herman-Brunson et al. (2009) found higher nest success on sites with shorter sagebrush but greater canopy cover.

Greater Sage-Grouse females are likely selecting sites with adequate concealment within 10 m of the nest, but I did not discern at what distance cover might open for better views of approaching predators (Götmark et al. 1995). Females require concealment while exiting from and returning to the nest during incubation breaks to avoid attracting predators. Coates and Delehanty (2008) determined yearling female Greater Sage-Grouse take more frequent and longer incubation breaks than adults, thus exposing themselves and their nests to greater depredation. Incubation rhythms could explain why successful yearlings tended to have greater horizontal cover in my study in both overall habitat use and sites dominated by dwarf sagebrush. Coupling less incubation constancy (Coates and Delehanty 2008) with apparently less horizontal cover, unsuccessful yearling females are more exposed during their frequent movements to and from the nest and are more easily detected by predators, thus lending themselves to higher nest failure. Wiebe and Martin (1998) found a fitness trade-off, though, where concealed nests of white-tailed ptarmigan (*Lagopus leucurus*) were less likely to be detected by predators but had higher risks of depredation on incubating females because approaching predators could not be easily seen. I had 2 (1%) females depredated on nests, similar to low values found in other studies (Coates et al. 2008, Moynahan et al 2007), so it is probably more beneficial for Greater Sage-Grouse to find nest locations with more concealment than a view of approaching predators.

Use of two new habitat measurement variables

Effective height and horizontal cover are not necessarily new methods for measuring habitat but they have not been previously used in conjunction to describe Greater Sage-Grouse nest habitat. Rotenberry and Wiens (1980) used effective height to measure bird habitat, some of which was in shrub-steppe cover and it loaded significantly on a component they described as vertical cover. Similarly, Mussehl (1963) measured a form of effective height for Blue Grouse (*Dendragapus obscurus*) habitat and determined its importance for concealment during brood rearing but did not describe the protocol used to measure effective height, especially the angle of view. Visual obstruction methods have been used to describe Greater Sage-Grouse nest habitat and have been found to be important indicators of habitat use (Popham and Gutiérrez 2003, Moynahan et al. 2007, Herman-Brunson et al. 2009, Kaczor et al. this volume) but have not been recorded from the point of view of nesting females.

Horizontal cover, as I measured it with a cover pole, proved to be the most important metric occurring in every top predictive model. I did record the gaps among the different levels above ground but did not discern among obstruction by shrubs, grasses, or forbs. In hindsight, horizontal cover should be categorized by the type of plant obscuring the pole and at what level above ground it occurs. Also, horizontal cover should be measured from different positions rather than just from the plot center since females require concealment while traveling to and from nests during incubation breaks, not just while sitting on the nest. Cover at 5 m from the nest was typically masked by cover at 3 m except when cover was lacking at 3 m, so observations should be no more than 2 m from the pole. Comparisons should be conducted between observing the pole from ground level and from the traditional 1 m height above ground to see if the two metrics are correlated. Observing from the female's perspective likely includes more obstruction than from traditional measurements taken 1 m above ground.

I found differences of effective heights among different species of grasses and shrubs. Sandberg bluegrass, as with other *Poa* spp., may have tall flower stalks but the cover they provide remains in the blades at the base, providing little concealment when compared to bottlebrush squirreltail, which has similar overall height but dense cover throughout the plant structure. Interspecific differences are illustrated in Figure 5 which compares two bunch grasses with similar heights but different concealment qualities. A site dominated by plants with taller effective heights would provide more concealment for nests and likely better nest success. I conclude that vegetation height alone does not adequately describe habitat for nesting Greater Sage-Grouse, quality of the plant structure is more important. Effective shrub height may not measure the true concealment by shrubs because only cover at crown height is measured. Species like basin big sagebrush tend to have a more open structure near ground level whereas dwarf sagebrush and mountain big sagebrush have cover throughout the vertical structure thereby providing more concealment of nests.

I recommend biologists include two techniques I used when monitoring nesting habitat for grouse populations in shrub-steppe habitat. Effective height and horizontal cover measure concealment by visual obstruction, the former for quality of individual plant species and the latter for quality of the landscape in general. A combination of the two methods provided important information for modeling the nesting habitat of Greater Sage-Grouse across a range of

habitats in my study. I believe measuring habitat from the grouse's perspective adds strength to detecting relationships with the environment, especially in habitat that is naturally sparse, such as shrub-steppe, but may not be applicable to habitats with dense understories as in prairie grasslands.

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Table 1. Variables loading on first principal components comprising majority of variability of Greater Sage-Grouse nesting habitat in Idaho, US, 2003-2005.

Principal Component I		Principal Component II		Principal Component III		Principal Component IV		Principal Component V	
Variables[a]	%[b]	Variables[c]	%	Variables[d]	%	Variables[e]	%	Variables[f]	%
HTSHRUBEF	89	DENSAGE110	94	HCOVTT3	89	CCSAGE110	93	CCFORBTT	94
HTSHRUBLV	87	DENSAGE35	94	HCOVMED3	88	CCSAGE35	86	CCFORB1	87
HTSHRUBEF3	86	DENSAGE510	93	HCOVTT5	88	CCSHRUB110	82	CCFORB5	87
HTSHRUBEF1	85	DENSHRUB35	92	HCOVGRD3	87	CCSAGE510	82	CCFORB3	85
HTSHRUBLV1	82	DENSHRUB110	90	HCOVHIGH5	86	CCSHRUB510	78	CCFORBSPP	59
HTSHRUBLV3	81	DENSHRUB510	89	HCOVHIGH3	80	CCSAGE13	78		
HTSHRUBEF5	80	DENSAGE13	72	HCOVMED5	79	CCSHRUB35	76		
HTSHRUBLV5	74	DENSHRUB13	68	HCOVLOW5	43	CCSHRUB13	70		
(23 % of variability)		(11 % of variability)		(8 % of variability)		(7 % of variability)		(6 % of variability)	

[a] Shrub height abbreviations: EF is effective height, LV is height of live portion of tallest branch measured 3 or 5 m from center of plot.
[b] Percent loading of variable onto component, only those variables loading >40% were retained.
[c] Density of shrubs: SAGE is any sagebrush species, SHRUB is all shrub species including sagebrush measured at segments of 1-3 (13), 3-5 (35), 5-10 (510) and 1-10 (110) m from center of plot.
[d] Horizontal cover: LOW is measured from ground level to 18 cm, MED is measured from 18 – 61 cm above ground level, HIGH is from 61-122 cm, GRD is from ground level to 61 cm, and TT is entire cover pole measured 3 and 5 m from center of plot.
[e] Canopy coverage of shrubs: same designations as density.
[f] Canopy coverage of forbs for 1, 3, and 5 m from center of plot and for all distances totaled (TT).

Table 2. Variables loading on first principal components comprising majority of variability of Greater Sage-Grouse nesting habitat dominated by dwarf sagebrush in Idaho, US, 2003-2005.

Principal Component I		Principal Component II		Principal Component III		Principal Component IV	
Variables[a]	%[b]	Variables[c]	%	Variables[d]	%	Variables[e]	%
CCSHRUB110	95	DENSAGE110	98	HCOVTT3	95	HTGRASSFLWTT	91
CCSAGE110	94	DENSAGE510	97	HCOVHIGH5	92	HTGRASSFLW5	87
CCSHRUB510	92	DENSAGE35	96	HCOVTT5	90	HTGRASSFLW1	82
CCSAGE510	89	DENSHRUB35	95	HCOVMED3	86	HTGRASSMAX5	81
CCSAGE35	87	DENSHRUB110	94	HCOVHIGH3	85	HTGRASSFLW3	75
CCSHRUB35	86	DENSHRUB510	94	HCOVGRD3	83	HCOVLOW5	41
CCSAGE13	79	DENSAGE13	71	HCOVMED5	76		
CCSHRUB13	78	DENSHRUB13	68				
(19 % of variability)		(15 % of variability)		(10 % of variability)		(9 % of variability)	

[a] Shrub canopy cover abbreviations: SAGE is any sagebrush species, SHRUB is all shrub species including sagebrush measured at segments of 1-3 (13), 3-5 (35), 5-10 (510) and 1-10 (110) m from center of plot.
[b] Percent loading of variable onto component, only those variables loading >40% are retained.
[c] Density of shrubs: same designations as canopy coverage.
[d] Horizontal cover (HCOV): MED is measured from 18 – 61 cm above ground level, HIGH is from 61-122 cm, GRD is from ground level to 61 cm, and TT is entire cover pole measured 3 and 5 m from center of plot.
[e] Droop height (HT) of grass flowers (GRASSFLW) at 1, 3, and 5 m from center of plot, and for all distances totaled (TT). HCOVLOW5 is the horizontal cover measured from ground level to 18 cm with a cover pole at 5 m from center of plot.

Table 3. Models of Greater Sage-Grouse nesting habitat in Idaho, US, 2003-2005.

					Correctly classified (%)[b]	
Model[a]	Estimate	SE	AIC_c	w_i	$\bar{X}$	SE
Intercept	-0.954	0.258	386.9	0.332	59.3	0.1
HCOV3	0.024	0.005				
Intercept	-0.908	0.261	387.1	0.289	59.4	0.1
HCOV3	0.026	0.006				
CCFORB	-0.016	0.012				
Intercept	-1.035	0.319	388.6	0.132	58.6	0.1
DENSAGE110	0.030	0.069				
HCOV3	0.024	0.005				
Intercept	-0.914	0.289	388.7	0.125	59.6	0.1
HTSHRUBEF	-0.002	0.006				
HCOV3	0.024	0.005				
Intercept	-0.932	0.286	388.8	0.122	59.0	0.1
HCOV3	0.024	0.005				
CCSAGE110	-0.003	0.014				

[a] Variable abbreviations: HCOV3 is horizontal cover at 3 m from center of plot, CCFORB is canopy cover of forbs throughout plot, DENSAGE110 is density of sagebrush 1-10 m from center of plot, HTSHRUBEF is the effective height of all shrubs, CCSAGE110 is canopy cover of sagebrush 1-10 m from center of plot.

[b] Cross-validation based on 1,200 iterations and withholding 50% and 40% of data to validate the 1- and 2-variable models, respectively.

Table 4. Models of Greater Sage-Grouse nesting habitat dominated by dwarf sagebrush in Idaho, US, 2003-2005.

					Correctly classified (%)[b]	
Model[a]	Estimate	SE	AIC_c	w_i	$\bar{X}$	SE
Intercept	-1.148	0.726	106.6	0.394	65.9	0.2
HCOV3	0.080	0.020				
HTGRASSFLW	-0.053	0.033				
Intercept	-1.937	0.560	107.3	0.285	68.4	0.2
HCOV3	0.065	0.017				
Intercept	-0.772	0.841	108.1	0.189	65.1	0.2
DENSAGE110	-0.0824	0.092				
HCOV3	0.081	0.021				
HTGRASSFLW	-0.060	0.034				
Intercept	-1.179	0.923	108.8	0.132	64.9	0.2
CCSHRUB110	0.002	0.029				
HCOV3	0.079	0.022				
HTGRASSFLW	-0.053	0.034				

[a] Variable abbreviations: HCOV3 is horizontal cover at 3 m from center of plot, HTGRASSFLW is height of grass flower, DENSAGE110 is density of sagebrush 1-10 m from center of plot, CCSHRUB110 is canopy cover of shrubs 1-10 m from center of plot.
[b] Cross-validation based on 1,200 iterations and withholding 50%, 40%, and 35% of data to validate the 1- , 2-, and 3-variable models, respectively.

Table 5. Characteristics of Greater Sage-Grouse nests in habitat dominated by dwarf sagebrush in southern Idaho, US, 2003-2005.

Variable	Nests (N = 46) $\bar{X}$	SE	Random (N = 43) $\bar{X}$	SE
Dwarf sagebrush height (cm)	30.6	1.2	25.8	1.2
Non-dwarf Sagebrush height (cm)	51.2	4.6	44.2	5.7
All sagebrush height (cm)	35.7	1.8	28.4	1.6
Shrub height (cm) all shrubs combined	30.9	1.7	22.6	1.7
Effective dwarf sagebrush height (cm)	21.9	1.2	18.3	1.3
Effective non-dwarf sagebrush height (cm)	35.4	5.2	34.5	5.5
All sagebrush effective height (cm)	25.2	1.7	20.6	1.6
Effective shrub height (cm)	24.3	2.0	18.3	1.9
Plant height (cm)[a] over nest by species				
Dwarf sagebrush (22 nests)	51.5	2.3	-	-
Non-dwarf sagebrush (10 nests)	69.5	8.5	-	-
Other shrub (11 nests)	73.5	24.8	-	-
Non-shrub (2 nests)	44.0	27.0	-	-
Grass height maximum (cm)	27.1	1.3	26.9	1.5
Effective grass height (cm)	9.3	0.7	6.8	0.6
Residual grass height (cm)	11.7	1.3	9.1	0.8
Live grass height (cm)	14.4	0.6	12.8	0.7
Grass flower height (cm)	23.1	1.2	23.2	1.4
Shrub density (#/m^2)	4.2	0.5	4.4	0.4
Sagebrush density (#/m^2)	3.2	0.5	3.5	0.3
Dwarf sagebrush density (#/m^2)	3.1	0.4	3.4	0.3
Shrub canopy cover (%)	22.8	1.3	19.3	1.3
Sagebrush canopy cover (%)	18.6	1.6	18.2	1.7
Dwarf sagebrush canopy cover (%)	17.4	1.4	16.3	1.6
Grass cover (%)	15.1	1.6	14.9	1.5
Forb cover (%)	10.5	1.6	9.2	1.5
Horizontal cover (%) 3 m from center	37.8	2.3	25.4	2.2
Slope (degrees)	3.5	0.7	3.9	0.6
Aspect (degrees)	161.7	16.4	141.5	14.2
Elevation (m)	1678	25	1776	28

[a] Maximum droop height of plant over nest. One nest was in open with no plants over the nest.

Table 6. Effective height (cm) compared to traditional maximum droop height for selected grass species in Greater Sage-Grouse habitat, Idaho, US, 2003-2005.

Species	N[a]	Effective Height $\bar{X}$	Effective Height se	Maximum Height $\bar{X}$	Maximum Height se	% Effective of Maximum $\bar{X}$	% Effective of Maximum se
Bluebunch wheatgrass	179	12.9	0.7	31.6	1.3	48.6	4.0
Bottlebrush squirreltail	175	10.6	0.5	20.6	0.8	57.8	3.2
Cheatgrass	152	1.2	0.3	12.0	0.5	13.2	3.3
Crested wheatgrass	23	16.3	1.6	34.3	2.4	50.7	5.9
Idaho fescue	52	11.6	0.7	24.5	1.8	57.8	5.1
Indian ricegrass	59	6.6	0.6	12.9	1.0	62.1	6.0
Needle-and-thread grass	30	7.5	0.7	25.9	3.6	41.2	6.9
Sandberg bluegrass	248	4.1	0.2	23.2	0.8	25.0	1.2

[a] Number of nests and random plots combined. Maximum number of plants measured is 12/plot, grazed plants were not measured.

Table 7. Effective height (cm) compared to traditional maximum droop height for selected shrub species in Greater Sage-Grouse habitat, Idaho, US, 2003-2005.

Species	N[a]	Effective Height $\bar{X}$	Effective Height se	Maximum Height $\bar{X}$	Maximum Height se	% Effective of Maximum $\bar{X}$	% Effective of Maximum se
Little sagebrush	128	22.4	0.8	29.4	0.9	79.3	5.4
Black sagebrush	25	15.6	1.1	25.2	1.9	59.9	1.7
Basin big sagebrush	47	48.8	4.0	66.5	4.3	69.7	2.6
Three-tip sagebrush	42	30.5	1.6	47.0	2.2	64.7	1.5
Mountain big sagebrush	36	63.8	4.2	69.1	4.8	88.1	3.5
Wyoming big sagebrush	130	41.9	1.8	51.1	1.6	78.2	1.9
Antelope bitterbrush	31	76.2	7.5	79.4	7.9	92.6	3.1
Spiny hopsage	35	28.0	2.6	32.7	2.9	88.3	9.0
Green rabbitbrush	162	22.6	0.9	28.4	0.9	76.3	1.6

[a] Number of nests and random plots combined. Maximum number of plants measured is 12/plot, grazed plants were not measured.

Figure 1. Eight populations of Greater Sage-Grouse sampled for nest habitat in southern Idaho, US, 2003-2005. Populations are: 1) Washington County, 2) Cow Creek, 3) Oreana/Big Springs/Shoofly, 4) Sheep Creek, 5) Browns Bench/Shoshone Basin, 6) Laidlaw Park, 7) Birch Creek, and 8) Little Lost Creek. Elevation is depicted with shading from low (black) to high (white) elevations.

Figure 2. Measurement of horizontal cover at a Greater Sage-Grouse nest site, Idaho, US. Photo A: observer reading cover pole from immediately outside the nest bowl with eye level 20 cm above ground. Pole is 3 m distance from center of nest. Photo B: view of cover pole measured in Photo A.

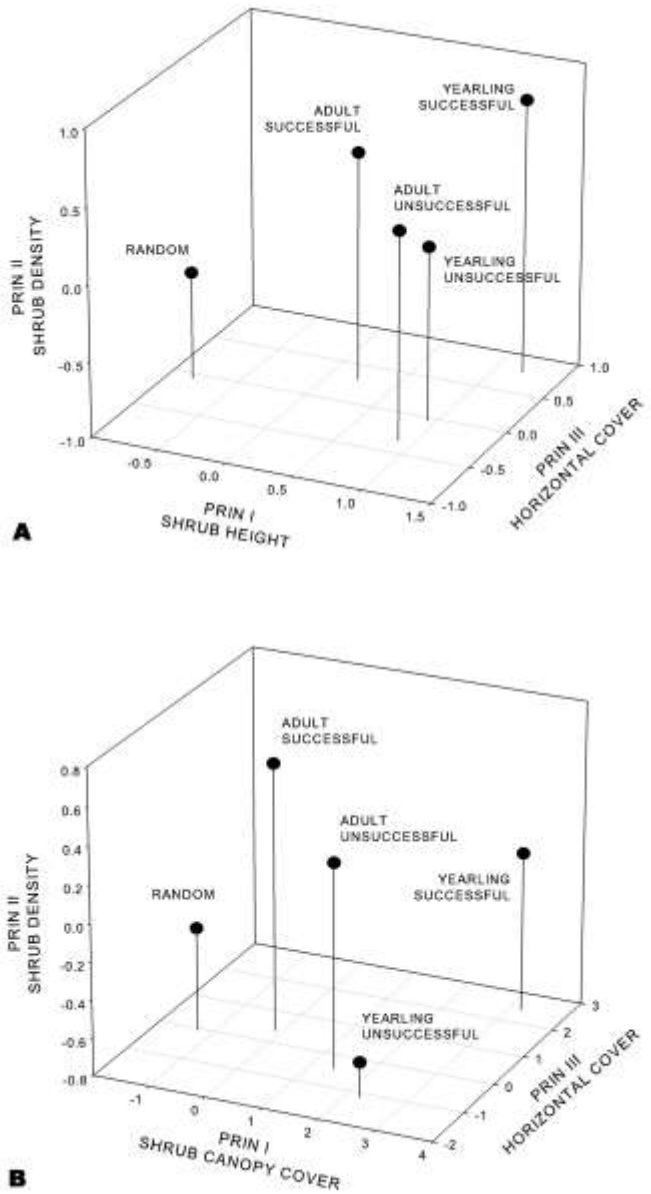

Figure 3. Habitat relationship of first three principal components for age-class of Greater Sage-Grouse females, nest fate, and random plots in Idaho, US, 2003-2005. A: Habitat throughout southern Idaho. B: Habitat dominated by dwarf sagebrush.

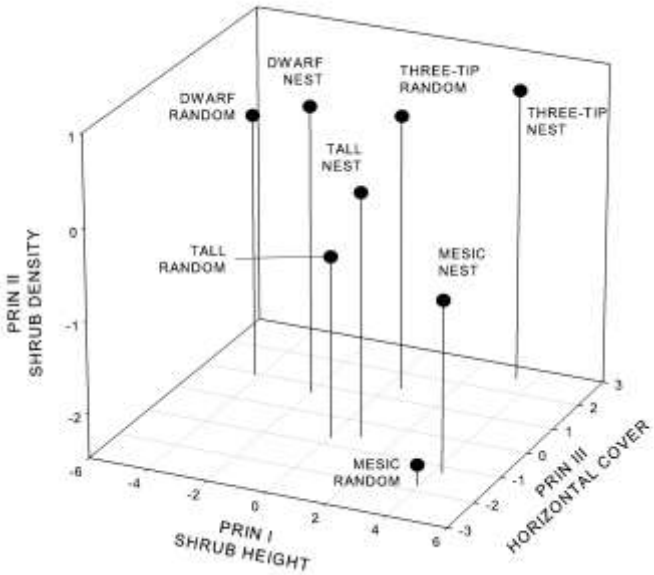

Figure 4. Habitat relationship for first three principal components among Greater Sage-Grouse nest sites and random plots by moisture category and shrub size in Idaho, US, 2003-2005.

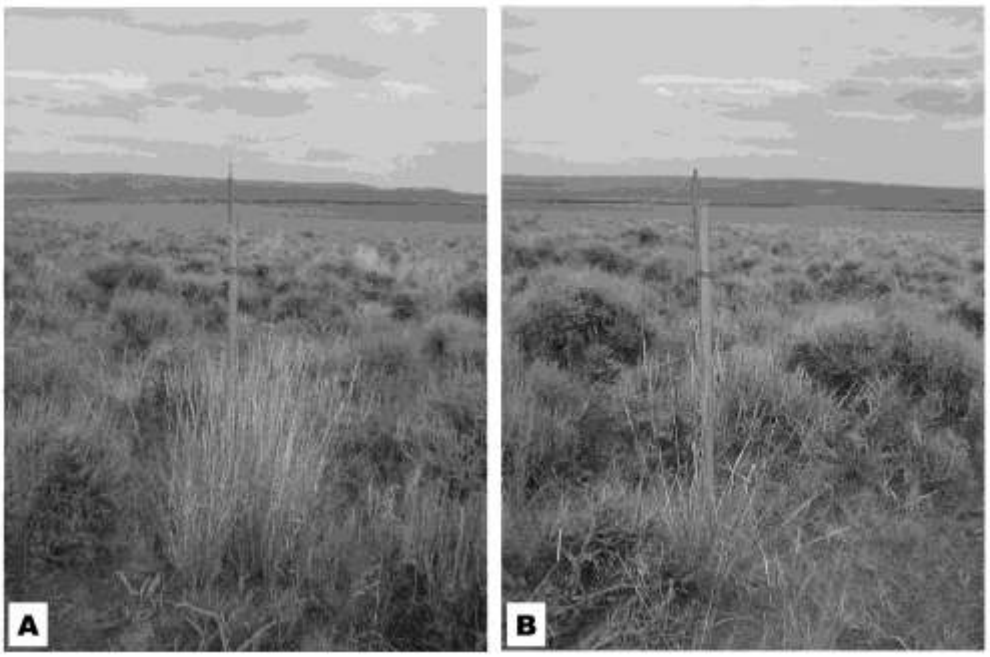

Figure 5. Example of two grasses with similar heights but different levels of horizontal concealment cover. Grass A (bluebunch wheatgrass) has taller concealment cover than B (needle-and-thread grass) but both species have relatively identical droop heights of residual, live, and flower structures. Meter stick is supported with a metal rod.

STUDY II: Population Characteristics and Habitat Use of Exploited Forest Grouse Populations

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ABSTRACT Forest grouse (dusky grouse [*Dendragapus obscurus*], ruffed grouse [*Bonasa umbellus*], and spruce grouse [*Falcipennis canadensis*]), are popular among gamebird hunters in Idaho. Unfortunately, abundance, population trends, and harvest rates are largely unknown in the state. We established 12 random line transects and 2 management roadside surveys to estimate ruffed grouse density during the last 3 years in the Squaw Creek drainage of game management unit 32A. Densities have averaged 5.05 ± 0.41 males/km^2 (mean ± 95% CI) for transects and 4.15 ± 2.60 males/km^2 for roadsides. Twenty-six ruffed grouse males have been captured, radio-collared, and monitored for survival and movements. Adult male home ranges (24 ± 13 ha, n=14) were similar (P = 0.156) to yearling males (62 ± 77 ha, n=5) as were dispersal from drumming logs (AD 123 ± 37 m, JV 208 ± 132 m, P=0.096). Survival of 19 radio-marked males averaged 56.7 ± 0.1%/year with only 1 male being harvested. A hunter check station was operated for 18 days during the 2009 hunting season with 498 hunters checked and 96 forest grouse (69 ruffed grouse, 27 dusky grouse) harvested. Hunters were categorized by primary species hunted; 70% big game, 19% grouse, 5% predator, 2% turkey, and 4% by recreationists scouting for big game or woodcutting. Grouse only hunters accounted for 69% of the forest grouse harvest, 22% by big game, 9% recreationists (big game scouting and firewood cutting), 0% predator, and 0% by turkey hunters. During the first 2 weeks of the hunting season, 69% of the harvest was juvenile birds and 51% of the entire juvenile harvest occurred during this time. A hatch curve was developed with 37% of the hatch occurring during the second week of June. Less than 10% of the radio-marked males have been harvested, possibly due to road closures to motorized vehicles in the study area.

KEY WORDS Forest grouse, ruffed grouse, dusky grouse, spruce grouse, surveys

Forest grouse (ruffed grouse [*Bonasa umbellus*], dusky grouse [*Dendragapus obscurus*], and spruce grouse [*Falcipennis canadensis*]), are among the most popular game birds in Idaho, annually averaging 148,000 harvested birds (1990-2006) and averaging 23% of the upland game bird harvest, second highest of all species (IDFG 2007). Unfortunately, abundance, population trends, and harvest rates are largely unknown. Forest grouse are also dependent on habitat often affected by land management decisions (e.g., riparian zones, mixed shrub uplands, and older growth timber). To properly manage these game birds and assess responses to habitat change and harvest by hunters, reliable survey techniques need to be developed and tested. In Idaho, no standardized routes are monitored before the hunting season to track population trends or forecast harvest. Forest grouse populations in close proximity to urban areas or other landscape features (e.g. roads) may have higher harvest rates than more remote populations. Also, forest management is changing to increase health of stands and prescriptions to reduce threats of wildfire, both of which may impact forest grouse.

As forest grouse become more popular as a hunted game species because other upland species are declining and changes in hunter interests, determining accurate population levels is critical for proper management. Currently, Idaho relies on incidental pre-hunting season observations by

biologists and conservation officers to estimate population trend and to forecast harvest opportunity for hunters. Eastern states with ruffed grouse monitor population trends with spring drumming counts and assume a 1:1 male to female ratio. This estimate is prior to nesting and does not reflect late spring/early summer weather affecting chick survival. Occasionally, Idaho has monitored drumming routes but these have not been consistent or standardized. No standardized surveys have been conducted in Idaho for dusky or spruce grouse.

Historically, the length of the forest grouse hunting season in Idaho was conservative (mid-September to end of November). During the 1980s, the seasons gradually increased to the current length started in 1990 (1 September – 31 December). It is unknown if this has had an effect on the harvest of hens with broods but has been speculated as the cause for reductions in populations of ruffed grouse in areas close to urban centers. The daily bag and possession limits have been held constant at 4 birds/day, 8 birds in possession in aggregate. It is unknown if the 3 species of forest grouse in Idaho are affected by this harvest strategy. Devers et al. (2007) present a thorough review of effects of harvest on forest grouse, mainly ruffed grouse and dusky grouse, and found equivocal evidence. Hunting may be compensatory up to a certain point, then become additive but depends on several factors including landscape attributes and hunter behavior. Monitoring harvest rates in Idaho will be critical to determine hunting pressure on the different species and age/sex classes.

STUDY AREA

The Second Fork of Squaw Creek is within the Boise National Forest in Gem County of west central Idaho and within Game Management Unit 32A (Fig. 1). The study area is dominated by ponderosa pine (*Pinus ponderosa*) with patches of quaking aspen (*Populus tremuloides*). Open areas are dominated by sagebrush (*Artemisia tridentata*) and antelope bitterbrush (*Purshia tridentata*). Riparian zones are dominated by currants (*Ribes* spp), chokecherry (*Prunus virginiana*) and hawthorn (*Crataegus spp.*). The study area has a west-southwest aspect and elevation ranges 1160-1524 m. Two main gravel roads, FS 653 and FS 626, provide access to the Second Fork and Sage Hen Reservoir drainages, respectively. Several 2-track logging roads intersect the main roads and some are closed to year-round motorized traffic.

METHODS

Capture and telemetry

To capture male ruffed grouse, we used mirror traps with similar dimensions to Gullion (1965) and in design to Bendell and Fowle (1950) but constructed ours of plastic drainpipe rather than a wooden frame wrapped in hardware cloth (Fig. 2a). Traps were deployed on active drumming logs found during surveys along random line transects and roadsides. Traps were checked twice daily by 1000 and 2100 hrs. Necklace style (Riley and Fistler 1992) 12 g radio transmitters $\leq$2% of grouse body mass (Kenward 1987) were used and programmed with 4 hour mortality sensors (Advanced Telemetry Systems, Isanti, MN 55040). Birds were also weighed with 1000 g spring scales. Age was determined by wear and shape of primary feathers 9 and 10 as well as caliper measurements of central tail feather diameters (Dorney and Holzer 1957).

To capture female ruffed grouse during June-August, we attracted hens with playback recordings of chick-distress calls and used 4.9 m noosepoles (Zwickel and Bendell 1967, Schroeder 1986) or foot snares of monofilament fishing line loops tied atop 2 m x 30 cm of carpet (personal design). The foot snare technique involved placing the carpet of snares in the path of the approaching hen while backing up with the recoding playing. During August-September, we used cover-leaf funnel traps placed near springs and wet meadows (Fig. 2b) similar to Chambers and English (1958) and Backs et al. (1985).

We tracked radio-marked grouse using hand-held 3-element collapsible Yagi antennas and obtained visuals or enclosed the location within a 50 m radius circle. Locations were obtained at least 1/week during May – October and 1/month during November-April. Home range was estimated using the 95% fixed kernel method (Hooge and Eichenlaub 2000). Dispersal was estimated by calculating distances between coordinates of the capture log and subsequent radio-locations after mid-May to avoid over representation during the breeding season.

Surveys

Roadside surveys - Roadside surveys for ruffed grouse, similar to Petraborg et al. (1953) and Dorney et al. (1958), were conducted April-June depending on road access conditions. Routes were at least 5 km long, started half hour before sunrise, and ended <2 hours after sunrise. Observation points were at 800 m intervals and observed for 5 minutes/stop. Stops were spaced to avoid overlap of observations if roads or trails switchback due to topography. Routes were not run when wind was >15 km/hr or precipitation was heavy.

Similar surveys for brood rearing female ruffed grouse were conducted on USFS side roads and trails to avoid traffic on the 2 main roads. Females were attracted by playback of chick-distress calls played for 4-5 minutes similar to Healy et al. (1980) and attending hens, hens with chicks, and number of chicks were counted. Observation points were spaced at approximately 100 m intervals, similar to spacing of point intersection transects.

Point intersection transect surveys - Random start points were generated within the study area using ArcGIS (ESRI, Redlands, CA) and spaced 400 m apart. Random directions (azimuth) for each transect were generated and spaced >400 m from the nearest transect. Transects were 1000 m in length and marked with surveyor flagging at each observation point at 100 m spacing for 11 points/transect and 5 minute listening bouts for each point. Azimuth towards each drumming ruffed grouse were recorded. Each transect were run from 1 half hour before sunrise to 2 hours after sunrise. Transects were conducted with the same weather protocol as for roadside surveys. Individual displaying males were determined by plotting intersections of directions from each observation point where the drumming male was heard and visually by approaching drummers.

The same transects were used to survey brood rearing female ruffed grouse by playing a recording of a chick-distress call during June-August as described above.

Habitat measurements

We measured habitat on the random point intersection transects to determine canopy cover, tree and shrub density by diameter and height, and basal area for each species of conifer and

deciduous tree, and density of shrubs for each plant species. We placed 4, 11m belt transects radiating from the center of each observation point (11 plots/transect) in the 4 cardinal directions. Numbers of conifers and deciduous trees were tallied for plants ≤ 2m and > 2m, and among the following diameter breast heights (cm): <2.54, 2.55-9, 9.1-15, >15. A 20 gauge basal area factor was used for conifers and 5 gauge for deciduous trees out to 11m from plot center. Shrub stem density was measured from the center of the plot out to 5m within a 1m wide belt in the 4 cardinal directions and tallied separately for each species ≤2m and >2m height. Slope (degrees) and aspect (azimuth) were also measured. A concave densitometer measurement was taken from the center of the plot in the 4 cardinal directions.

Hunter surveys

A hunter check station was conducted at the intersection of Second Fork Squaw Creek (FS 653) and Sagehen Reservoir Road (FS 626). The check station was operated opening day and weekend of the grouse season with random weekdays and weekends sampled until early November. Opening days and weekends of big game seasons were also targeted for check stations. Check stations were operated from 1000 to 2100 (or sunset). Hunters were asked what primary species was hunted (big game, grouse, turkey, etc). If the primary hunted species was not grouse, the hunters were asked if they had the opportunity, would they attempt to harvest a grouse. Method of take (bow, shot gun, rifle, etc), method of travel (foot, motorized vehicle), number of grouse seen by species, and hours hunted (grouse hunters only) and/or days hunted (other primary species) were also determined for each hunter. Harvested grouse were weighed if the carcass was not field dressed. One wing, entire tail, and rump feathers were collected from each ruffed grouse to determine age and sex (Roussel and Ouellet 1975) and to estimate hatch dates (Bump et al. 1947).

RESULTS

Capture and telemetry

A total of 26 drumming male ruffed grouse have been captured and radio tracked since 2007 (Table 1). Home ranges and dispersal have been calculated for 19 males during 2007-2009 but not yet for 7 males captured in 2010. Adult males are significantly heavier than yearlings but no differences could be detected in home range size or dispersal at the 0.05 level of significance. Males have been predominantly grey phase for both age classes.

Survival estimates have not been determined for the radio-marked male ruffed grouse but apparent annual survival for 19 tracked 2007-2009 is 56.7 ± 0.1%. For those males that died during the study, they have averaged a minimum life span of 2.2 ± 0.4 years. Two males captured as adults in 2007 were still alive in 2010 so are at least 5 years old. One of these males has been recaptured twice on his drumming logs but the other male has only been identified visually since 2008 and avoided mirror traps for 2 seasons after battery failure of his transmitter. For the 8 mortalities documented; 4 occurred between mid-August and late-September (3 apparently caused by predator depredation and 1 by transmitter complications), 1 harvested by a big game hunter, and 3 died during the winter (cause of death undetermined). Harvest rates for

17 radio-marked males (24 cumulative samples) available for harvest during the 2007-2009 hunting seasons averaged 3.3 ± 6.5%/year.

No females have been capture with clover-leaf traps in 2007 nor noose-poling in 2008-2010. The summer of 2007 was considered drought conditions with springs and creeks going dry early in July thereby eliminating our chances to capture grouse at concentrated areas. We placed several trap lines in areas where we had incidentally encountered hens with broods but were unsuccessful. During 2008, we tested the use of chick-distress playback calls during late July through late September with responses only during late July and the hens did not approach close enough for noosing. Noose-poling was attempted again in 2009 after transects and roadside routes were first run for play-back surveys during June-early July. We returned to 3 of the 5 sites within 3 days where hens had been attracted but they did not respond as readily as the first time. During 2010, we attracted 6 hens with the recordings and tried noose-poling immediately as we encountered them along the survey routes but still could not get close enough for noose-poling. The hens were attracted to within 6 m but not close enough for the 4.9 m noose-pole. When I eventually developed the foot snare technique and approached several of the hens that were still in the same areas in late-July, they were again, more reluctant to approach close enough for capture.

Surveys

Ruffed grouse drumming surveys were established on 12 line transects and 2 roadsides (Figure 3). Three line transects, intentionally run along creek drainages in 2007 were changed to random directions in 2008 to comply with line transect protocol (Buckland et al. 2001). Two roadside surveys were conducted on 2 separate roads (FS 653 and FS 626). Program DISTANCE (Buckland et al. 2001) has not been used yet to estimate density. Instead, density was determined by assuming drumming males were heard up to 200 m (Gullion 1966, Zimmerman and Gutierrez 2007), and estimated for roadsides and random transects (Figure 4). Roadside survey densities (4.15 ± 2.60 males/km^2, mean ± 95% CI) were more variable than for point intersection transects (5.05 ± 0.41 males/km^2) during 2007-2010. Transect surveys had an average gain of 7% annually in drumming male densities whereas roadside surveys were very erratic and trend could not be determined. We detected an average of 31.5 ± 3.8 drummers/yr (range 26-35) on transects and 10 ± 6.5 drummers/yr (range 4-18) on roadside routes.

Without knowing how far away hens are attracted to the chick distress calls, we cannot determine hen density but can use the surveys as indices (Table 2). We detected 5 hens and no broods along the 12 point intersection transects in 2009 and 4 hens and 1 brood in 2010. For roadside routes, we detected 2 hens and 1 brood on 6 routes in 2009 and 5 hens with 2 broods on 8 routes in 2010.

Habitat use

The habitat measurements on each observation point have been entered but not summarized or analyzed. Digital cover maps with stand composition and age structure have not been obtained from the US Forest Service.

Hunter surveys

We operated the 2nd Fork Squaw Creek check station 18 days (9 weekdays, 9 weekend days) for 10% of the available weekdays and 26% of the available weekends for the 122 day, 2009 season. A total of 498 hunters were interviewed and categorized by primary species hunted: 70% big game, 19% grouse, 5% predator, 2% turkey, and 4% by recreationists scouting for big game or woodcutting.

Of all the hunters checked, 27% said they would harvest a grouse if they had an opportunity during their trip and 37% of all hunters checked successfully harvest at least 1 grouse. Few big game (6%) and predator (4%) hunters targeted grouse, whereas 55% of recreationists and 33% of turkey hunters would try to harvest a grouse if seen.

For the 96 forest grouse brought through the check station, 69% were harvested by grouse hunters, 22% big game, 9% recreationists, 0% predator, and 0% by turkey hunters. The majority (72%) of the grouse harvested were ruffed grouse, the rest were dusky grouse and no spruce grouse. It took more hours to bag a dusky grouse than a ruffed grouse within the study area (Table 3).

Only 4 types of weapons were used: shotgun (93%), big game rifle (4%), archery (2%), and .22 rifle (2%). Most grouse hunters (97%) and big game hunters (65%) use only shotguns while all recreationists, predator, and turkey hunters strictly use shotguns. Grouse hunters also use archery (1%) and .22 rifles (2%) while 29% of big game hunters use rifles and 6% used archery.

Grouse hunters (including primary and secondary grouse hunters) traveling by foot (47%) harvested 58% of the grouse while those traveling roads and trails on motorized vehicles (53%) harvested 42% of the grouse. Eleven percent of grouse hunters use dogs and account for 8% of the total grouse harvest.

For 53 harvested ruffed grouse which age and sex could be determined, 66% of the harvest were juvenile birds (43% JV females, 23% JV males), 17% were adult females, and 17% were adult males. The juvenile:adult female ratio was 2.6:1 while the juvenile:adult ratio was 1.94:1.

Grey phase color morphs dominate among females in the harvest (69% grey) where only 48% of the males harvested were grey phase. Two males and 1 female were intermediate between grey and red phase with 60% of all harvested birds being grey phase, 34% red phase, and 6% intermediate phase.

During the first 2 weeks of the hunting season, 69% of the harvest was juvenile birds and 51% of the entire juvenile harvest occurred during this time. Weight of juvenile birds harvested during the first 2 weeks of the season (444 ± 34 g) were significantly lighter ($n = 28$, $P = 0.019$, Kruskal-Wallis 1-way AOV) than those harvested $\geq$ 3 weeks (495 ± 24 g) after the season opened September 1.

We developed a hatch curve from 38 wings of juvenile birds (Fig. 5). Peak hatch appears to occur during the second week of June with 37% of the juveniles hatching then. Weeks 3 and 4 of June have 29% and 21% of the hatch, respectively. The juvenile grouse harvested during the

first 2 weeks of the hunting season are 83 ± 3 days (11-12 weeks) old whereas those harvested ≥ 3 weeks after the opening day are 110 ± 6 days (15-17 weeks) old.

The peak grouse harvest was by grouse-only hunters and occurred during the first week of the season (Fig. 6) and mostly on Labor Day weekend. The next peak in harvest began to increase during weeks 4 and 5 prior to opening of the deer season. After the deer opener on 10 October, all the harvest was by big game hunters. No check stations were run during weeks 8 and 9, it was run again during week 10, but not the rest of the season.

Most grouse-only hunters (90%) returned home after a single day afield whereas 10% camped and hunted multiple days. For those grouse hunters day-tripping afield, few filled the 4-bag limit (Table 4).

DISCUSSION

Density of drumming male ruffed grouse in our study area appears to be similar to low cycles in British Columbia (Davies and Bergerud 1988) and within the ranges, but lower than average, to those across North America (Rusch et al. 2000). We consistently saw the same drumming logs used annually with immediate replacement the following year when a radio-marked male died. Therefore, optimal habitat could be limited and patchy, which is possibly why our density is consistently low compared to Eastern U.S. values where habitat is more abundant.

Survival appears somewhat greater than reported in Eastern U.S. studies (Rusch et al. 2000) but has not been separated by age classes due to low sample sizes in our study area. Males use relatively small home ranges year-round and do not migrate within the study area. Though adult male home range and dispersal was not significantly different from yearlings, it does appear adults are more sedentary on a daily basis than yearlings. Yearlings tended to make large movements between consecutive locations whereas adults were more centrally located around drumming logs year-round.

Delaying drumming counts due to poor road access appears to alter density estimates. During 2 years where roadside counts were delayed into mid-May (2008 and 2010), the counts were considerably lower than the random transects. During years where roadside counts were conducted in late-April/early-May, the counts were biased higher than random transects. This could be because roadside surveys follow creek drainages where densities might be greater due to better habitat near riparian zones. It appears the chick-distress playback survey technique has merit as a useful management tool but our study population may be too small to adequately test the technique.

It is difficult to speculate why almost twice as many juvenile females were harvested than juvenile males, while adult females and males were in equal proportions. Determining sex of young grouse without observing gonads could have caused errors in assessing the sex ratio.

Poorer forest grouse hunting occurred within our study area than outside, especially for dusky grouse. This is likely related to our study area being at a lower elevation where dusky grouse are less likely to be seen in the fall since they migrate to higher elevations before winter (Miyasaki 2001). Also, the habitat within the study area might be poorer quality or in lower quantity than

outside and hunters have learned to avoid the lower elevations in attempts to optimize their effort, especially for ruffed grouse. The lower elevations also tend to be hotter, early in the season, and this may cause the avoidance by hunters.

Few big game hunters admit to hunting grouse but do harvest a significant portion of grouse taken. Big game hunters do appear to avoid harvesting grouse during the later seasons and instead are concentrating only on elk whereas grouse are more likely to be harvested by early season big game hunters. Grouse-only hunters likely attempt to hunt again prior to the opening of deer season for one last chance before the increase in hunting pressure by deer hunters. It is unknown how many grouse-only hunters return to the field after big game seasons close since we did not sample during late-November or December. Snow conditions likely limit late-season grouse hunting.

MANAGEMENT IMPLICATIONS

Random point intersection transects, though more time consuming, provide a more precise estimate of male ruffed grouse densities. If roadside surveys are implemented statewide, routes should be carefully planned to avoid inaccessible roads that could delay counts, thereby missing peak drumming.

It is unknown if changing opening day of harvest to mid-September would increase population levels of ruffed grouse in GMU 32A. Young of the year might be just as vulnerable if the season was delayed 2 weeks, but the body size would be heavier and tails more fully developed. Most hunters do not fill bag limits of either species so reducing limits would likely not affect overall harvest or increase population levels if harvest is affecting growth. It is unknown what affect lengthening the season will have on harvest in northern Idaho, but shortening the season would likely have little effect since a majority of the harvest occurs early in the season, similar to that found by Palmer and Bennett (1963).

Since a majority of the hunters admitted to hunting grouse along forest roads, road closures for motorized vehicle use might be limiting vulnerability within the study area, at least it appears so for males and may account for the gradual increase in male densities since 2007. The only harvested radio-marked male was shot along a motorized accessible road. Past studies have documented the effects of roads on harvest (Dorney 1963, Fischer and Keith 1974) but have not determined the effect of road closures to vehicle traffic. Some hunters on foot still access areas via closed roads but these areas likely do not receive the same amount of pressure as open road areas.

FUTURE WORK

Habitat mapping and GIS analysis within 200 m of observation points for roadside and transect counts has yet to be accomplished. Also, program DISTANCE analysis to determine detectability functions based on geography (eg. areas masked by ridges blocking hearing) need to be accomplished. Further check station interviews of hunters and continuation of transects/roadside surveys will be accomplished as funds are available. Surveys need to be developed in other habitat types and areas with greater grouse densities to determine the

relationship between convenience sampling along roads and random transects. Low densities in our study area have precluded a proper test of survey techniques for dusky and spruce grouse. Eventually, annual counts need to be conducted statewide to better monitor forest grouse populations throughout Idaho.

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Table 1. Male ruffed grouse captured in Game Management Unit 32A during 2007-2010, Idaho.

Age	N	Weight (g)[1]	Grey phase (%)	Home Range (ha)[2]	Dispersal(m)[3]
Adult	17	543 ± 18	65	24 ± 13	123 ± 37
Yearling	9	497 ± 22	67	62 ± 77	208 ± 132
		P = 0.006[4]		*P*=0.156	*P*=0.096

[1] $\bar{X}$ ± 95% CI
[2] 95% fixed kernal home range estimate, 2007-2009 males (n = 14 adults, 5 yearlings).
[3] Distance from point of capture to locations off drumming log, 2007-2009 males (n = 14 adults, 5 yearlings).
[4] All comparisons are Kruskal-Wallis 1-way nonparametric AOV tests.

Table 2. Surveys of hen and brood ruffed grouse attracted during playback of chick-distress recordings along random transects and roadsides, 2009-2010, Idaho.

Survey/Year	Routes	Points[1]	Hens	Broods	Chicks	HPP[2]	BPP[3]	CPP[4]
Transects								
2009	12	132	5	0	0	0.038	0.000	0.000
2010	12	132	4	1	5	0.030	0.008	0.038
Roadsides								
2009	6	68	2	1	2	0.029	0.015	0.029
2010	6	68	2	1	2	0.029	0.015	0.029

[1] Total observation points; transects have 11 points/transect, roadsides vary by route.
[2] Hens per point index
[3] Broods per point index
[4] Chicks per point index

Table 3. Harvest effort by grouse-only hunters for Squaw Creek check station, Idaho, 2009.

Area, grouse harvested	Birds/day	Hours/bird	Birds/hunter
Within study area[1]			
Ruffed grouse	0.35	11.2	0.42
Dusky grouse	0.19	20.6	0.23
All forest grouse	0.55	7.3	0.65
Outside study area[2]			
Ruffed grouse	0.38	8.0	0.38
Dusky grouse	0.23	13.3	0.23
All forest grouse	0.62	5.0	0.62
All areas/all grouse	0.62	6.2	0.69

[1] Hunting within boundaries of study area: 52 hunters, 247 hours, 62 days, 22 ruffed grouse and 12 dusky grouse harvested.

[2] Hunting outside boundaries of study area: 52 hunters, 159 hours, 52 days, 20 ruffed grouse and 12 dusky grouse harvested.

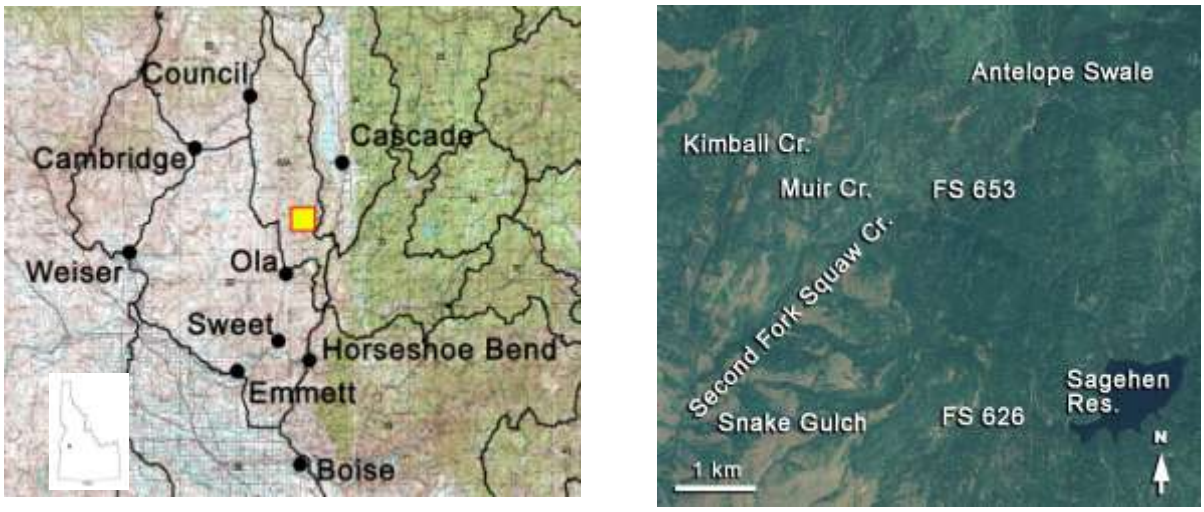
[3] Hunting by all grouse hunters passing through check station whether hunting within or outside of study area: 96 hunters, 406 hours, 107 days, 66 grouse harvested. Note: some hunters hunted both within and outside the study area so their trip was kept as 1 hunter and 1 day,. Hours hunted are cumulative. Also, some hunter trips were multiple days but kept as 1 hunter.

Table 4. Distribution of forest grouse bag limits among grouse hunters interviewed during 2009.

Primary species hunted, grouse harvested	Bag limit (%)[1]				
	0	1	2	3	4
Grouse-only (90) [2]					
Ruffed grouse	77	12	6	2	3
Dusky grouse	81	14	3	1	0
All forest grouse	62	20	11	1	6
Big game (14)					
Ruffed grouse	50	29	14	0	7
Dusky grouse	86	14	0	0	0
All forest grouse	43	36	7	7	7
Other (15)					
Ruffed grouse	80	7	0	13	0
Dusky grouse	93	7	0	0	0
All forest grouse	73	13	0	13	0
All grouse hunters (119)					
Ruffed grouse	74	13	6	3	3
Dusky grouse	83	13	3	1	0
All forest grouse	61	21	9	3	5

[1] % of hunters within group that harvested 0, 1, 2, 3, or 4 grouse during the day afield. Row adds to 100%. Some rounding errors may occur.

[2] Number of hunters interviewed, in parentheses, that would harvest a grouse if given the opportunity. "Other" combines hunters primarily hunting predators and turkeys, scouting for big game, or cutting wood.



A B

Figure 1. Forest grouse study area. A) Yellow box indicates study area north of Ola and southwest of Cascade, Idaho, within Game Management Unit 32A. B) Landmarks within study area. "FS" are U.S. Forest Service roads.

A B

Figure 2. Traps used to capture ruffed grouse. A) Modified mirror-trap deployed on an active ruffed grouse drumming log. B) Clover style funnel-trap placed near water sources and constructed of welded-wire sides and nylon netting ceilings with directional wings made of chicken wire connecting 4-8 traps in a set line.

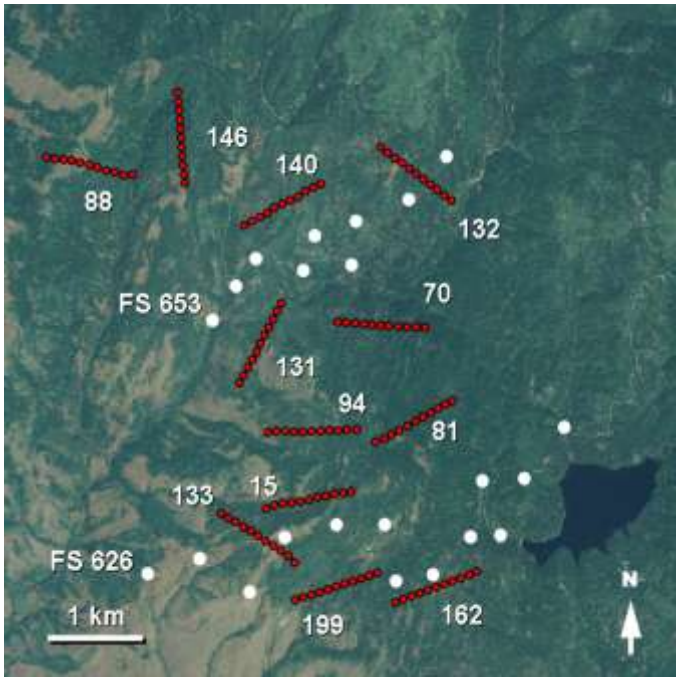


Figure 3. Two roadside survey routes (large white dots) along US Forest Service Roads (FS 653 and FS 626) and 12 point intersection transects (small red dots) with 11 listening posts per transect.

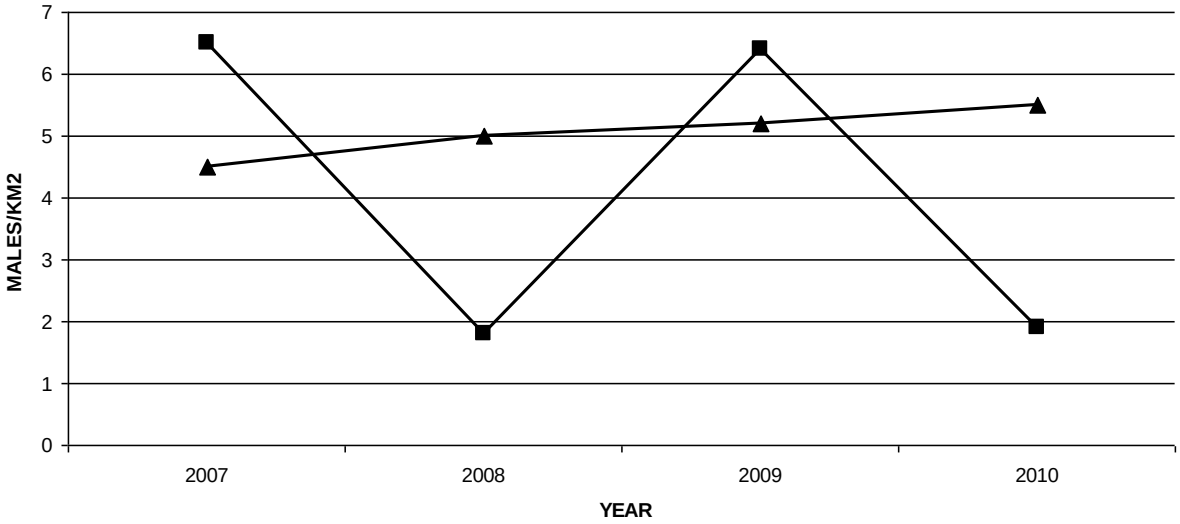


Figure 4. Density of drumming male ruffed grouse surveyed along 2 roadsides (squares) and 12 point intersection transects (triangles) in Game Management Unit 32A, 2007-2010.

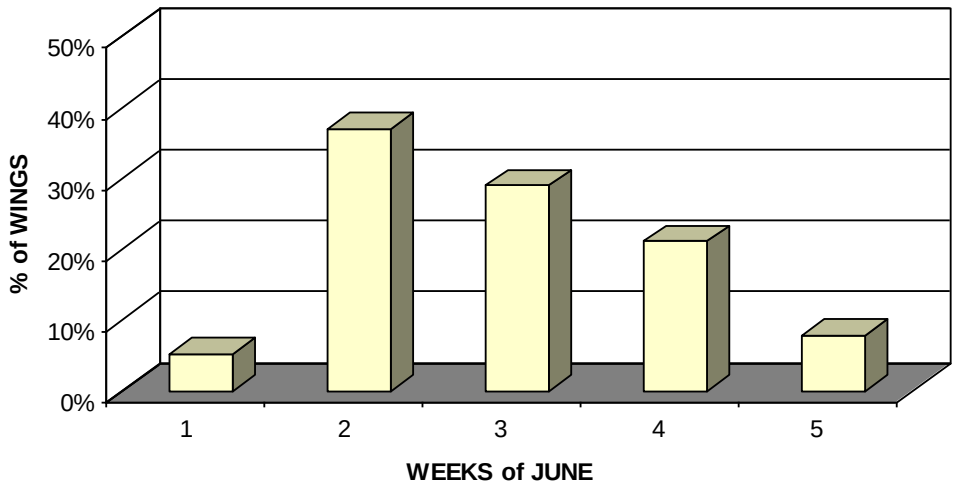


Figure 5. Hatch dates for juvenile ruffed grouse (n = 38) harvested during 2009.

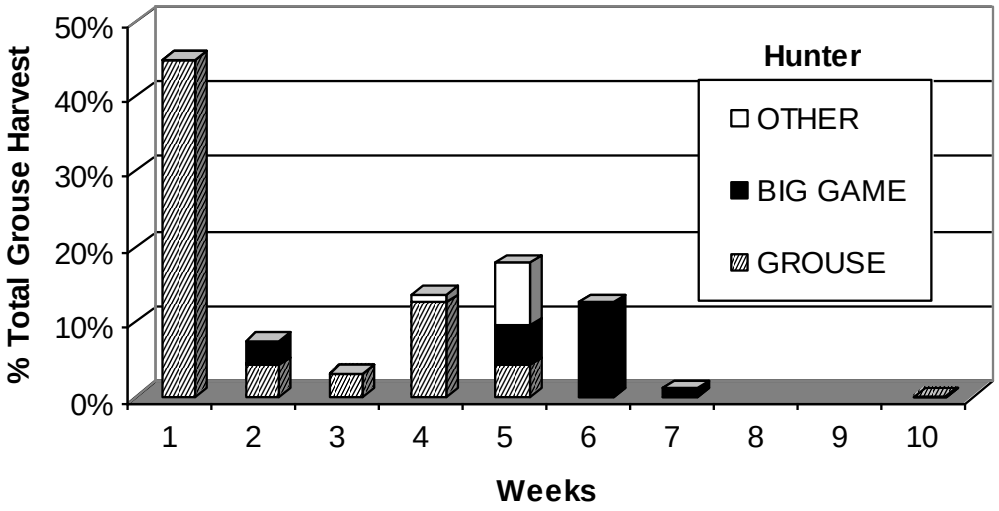


Figure 6. Distribution of grouse harvest by hunter category during first 10 weeks of the 2009 hunting season. "Other" hunters is a combination of those scouting for big game, cutting wood, or hunting predators.

STUDY III: TRANSLOCATION OF MOUNTAIN QUAIL INTO HISTORIC HABITATS

Completion report - Master's thesis.

Post-release performance of mountain quail translocated to the Northeastern periphery of historic range

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ABSTRACT Our objectives were to augment a founding population of mountain quail released in the Bennett Hills in 2006-2007 through subsequent releases of mountain quail in 2008-2009 and, assess radio-marked quail survival and reproduction. . I attempted to detect the establishment of a self-sustaining population. High mortality rates immediately following release influenced survival functions. Mountain quail survival rates were generally high during autumn (>85%). Winter survival rates represented the lowest of the year (≤ 40%). Winter stochastic events were the ultimate factor influencing mountain quail survival. The majority of translocated mountain quail (63%) did not move from breeding ranges although. We observed average mountain quail productivity despite a seeming shift in mountain quail reproductive strategy. A preponderance of female parental care coupled with seemingly promiscuous behavior by multiple males and females is evidence that care-giving males may be plastic in their parental and mating effort.

KEYWORDS Idaho**,** Mountain quail, movements, *Oreortyx pictus*, reproduction, survival

Of the 6 New World quail (Odontophoridae) species found north of Mexico, mountain quail are the only species native to Idaho. Among these 6 species, mountain quail are the largest, have the most northerly distribution, occupy and breed at the highest elevations, and are the least studied (Gutierrez and Delehanty 1999). Consequently, many gaps remain in our understanding of mountain quail breeding biology and general life history. Mountain quail (*Oreortyx pictus*) were once widespread and common in western and southwestern Idaho. By 1990, mountain quail occupied no more than 5% of their historic Idaho range (Brennan 1990) and now appear to be reduced to small, relict populations within remaining Idaho range (Reese et al. 2005).

Male and female mountain quail are generally monomorphic. Subtle sex differences can be identifiable with close observation (Pine 1981, Brennan and Block 1985) but plumage-based gender identification is inaccurate within wild populations (Delehanty et al. 1995). Mountain quail generally are socially monogamous (Gutierrez and Delehanty 1999) and can employ simultaneous multiple-clutching where the female will lay a clutch of eggs which the male will incubate (Delehanty 1995) and then the female will lay another clutch (Pope and Crawford 2001). Studies in which male-female social pairs are inferred by association during the breeding season and nesting proximity, indicate that multi-clutching male and female mountain quail will nest approximately 200 m from each other. Each adult incubates a clutch uniparentally.

Mountain quail typically inhabit early seral stage habitats and these habitats can be diverse (Gutiérrez and Delehanty 1999) such as mixed desert scrub of the Mojave Desert, California, USA, or dense conifer forests of the Cascade Mountains, Oregon, USA. (Mclean 1930, Aldrich and Duvall 1955, Crawford 2000).

Because of declining populations, mountain quail are currently classified as a "Species of Greatest Conservation Need" by The Idaho Department of Fish & Game (IDFG), and as a "Sensitive Species" by the Bureau of Land Management (BLM) and by Region 4 of the U.S. Forest Service (USFS). The continued decline of mountain quail after the hunting ban in 1984 suggests habitat degradation caused their decline (Brennan 1990, Gutierrez and Delehanty 1999). Conservation groups petitioned the United States Department of the Interior (USDI) to list mountain quail as a threatened or endangered species due to acute population declines in the eastern portions of their range (USDI Federal Register 2003). Although the petition was denied in 2003, it serves as an indication of mountain quail population decline sufficient to cause public concern. Since the 1980's there has been interest by state and federal agencies and citizen conservation organizations in re-establishing mountain quail within their historic range.

Further research is needed to determine if mountain quail can be restored throughout their historic range. Reasons for their precipitous decline are not clear, but new world quail are well known for population fluctuations (Gutierrez and Delehanty 1999) that make them vulnerable at small population sizes. Translocation – the deliberate capture, transport, and release or introduction or reintroduction of wild animals – has become a common technique to attempt to reestablish or to supplement avian populations for many species including galliform species like mountain quail (Scott and Carpenter 1987). A recent translocation of mountain quail occurred in western Idaho at the Craig Mountain Wildlife Management Area as part of a study by Stephenson (2008) to reestablish a self-sustaining population.

Archeological reports identified mountain quail remains east of the Bennett Hills in Jerome County, Idaho and were dated within the last 700-1600 years (Gruhn 1961). Furthermore, Murphey (1991) described a pictograph found in Jerome County, Idaho of a gallinaceous-like bird similar to mountain quail. Mountain quail were observed in the Bennett Hills as recently as 1986 (Peter Janss, pers. comm.). During calling surveys conducted in 2005 by the Bureau of Land Management, mountain quail crow vocalizations described in the literature were heard on the east fork of Cold Springs creek (43°8'N,115°20'W) in the Bennett Hills (pers. comm. Jill Holderman), but mountain quail are generally believed to be extirpated from here.

In an effort to restore mountain quail to an area of extirpation within their historic range, a habitat suitability analysis was conducted by Troy and Gamo (2004) and indicated the Bennett Hills was suitable habitat for mountain quail according to a model developed by Brennan et al. (1986). Thus, 125 mountain quail were translocated to the Bennett Hills (43°11'N,115°20'W) from 2 different source populations in a study conducted cooperatively by Idaho Department of Fish and Game and Idaho State University in April 2006 and 2007. These actions have led to a small, vulnerable population that was closely monitored from 2006 to 2007. Detailed results of this study are not yet available, but the release appeared to have resulted in a small number of mountain quail persisting locally in the region of the release.

My objective was to augment a nascent population of mountain quail released in the Bennett Hills in 2006 and 2007 through subsequent releases of mountain quail in 2008 and 2009. I was interested in how survival and reproductive rates in the eastern periphery of historic range would compare with other translocations (Pope 2001, Nelson 2007, Stephenson 2008). Furthermore, I investigated how productivity of mountain quail in a spatially and temporally heterogeneous environment differed in survival and reproductive effort of quail released at different sites. Although my main goal was to monitor mountain quail during the breeding-brooding season, another goal was to monitor mountain quail beyond the traditional 150-day field season and document basic mountain quail ecology during autumn and winter. I envisioned that an innovative trapping method for mountain quail (Troy et al. in review) would provide me the opportunity to monitor mountain quail annually and for multiple breeding seasons.

In my study, mountain quail were released in the spring of 2008 and 2009 at 2 different release sites within the Bennett Hills. I used two sites and one source population for three reasons. First, by using one source population and 2 release sites, I hoped to be able to parse out release site effects from source population effects. Second, during the 2006-2007 study mountain quail were occupying inaccessible private property and we were unable to gauge reproductive efforts. It was judged that adding a second release site would minimize the number of birds occupying inaccessible private property and allow us to determine reproductive rates. Third, this was an applied conservation project with the express goal of restoring mountain quail across the landscape more broadly in the third and fourth years of releases with the hope of causing a broader zone of occupation by mountain quail.

STUDY AREA

The Bennett Hills are located about 70 km southeast of Boise, Idaho, USA (Fig 1) at the eastern periphery of historic mountain quail range (Ormiston 1966). Elevation ranges from 900 to 2300 m. The study area encompassed approximately 63,000 ha adjacent to the Sawtooth National Forest and the northern Rocky Mountains (USA) to the north and the Snake River Plain to the south (Fig 1). Annual precipitation ranges from 23 to 40 cm and annual snowfall ranges from 90 to 130 cm. The minimum temperature during the 2-year study period was -18°C and maximum temperature was 33°C (http://www.wcc.nrcs.usda.gov).

The Bennett Hills are rugged and remote but are far from pristine. The study area was subject to annual grazing by sheep and cattle and contains many, roads and, water impoundments. Additionally, range fires followed by cheatgrass invasion altered vegetative communities, especially at low elevations when post fire reseeding does not occur.

Five nonnative galliform (order Galliformes) species have been introduced into the Bennett Hills,: California quail (*Callipepla californica*), ring-necked pheasant (*Phasianus colchicus*), gray partridge (*Perdix perdix*), chukar (*Alectoris chukar*) and wild turkey (*Meleagris gallopavo*). In recent years, owners of private land within the Bennett Hills have attempted to alter and improve wildlife habitat in the Bennett Hills through seeding of mixed grasses following fires and weed control. In 2008, the Bureau of Land Management enforced vehicle route restrictions in the 11,860 ha King Hill Wilderness Study Area. The Bennett Hills are a wildlife migration corridor for cervids (Bishop et al 2005), predatory mammals, and raptors. The study area is host

to a suite of predators including the gray wolf (*Canus lupus*), coyote (*Canis latrins*), gray fox (*Vulpes velox*), black bear (*Ursus americanus*), rattlesnakes (*Crotalus spp.*), mountain lion (*Puma concolor*), wolverine (*Gulo gulo*), badger (*Taxidea taxus*), bobcat (*Lynx rufus*), pine marten (*Martes americana*), striped skunk (*Mephitis mephitis*), long-tailed weasel (*Mustela frenata*), golden eagle (*Aquila chrysaetos*), red-tailed hawk (*Buteo jamaicensis*), Swainson's hawk (*Buteo swainsoni*), prairie falcon (*Falco mexicanus*), northern harrier (*Circus cyaneus*), short-eared owl (*Asio flammeus*), and 3 accipiter hawks: sharp-shinned hawk (*Accipiter striatus velox*), Cooper's hawk (*Accipiter cooperii*), and the northern goshawk (*Accipiter gentilis*).

The Bennett Hills support a rich mix of bunchgrass, perennial herbs and shrubs (Errter and Moseley 1992). Its floristic distinctiveness was not discovered until numerous studies of mule deer winter range began in the 1970's (Errter and Mosely 1992). Low elevations are xeric with few perennial springs and streams. The dominant terrain consists of large plateaus and outcrops of igneous rock dissected by narrow, steep canyons (Bishop et al. 2005) with ephemeral streams. Low elevation vegetation consists of shrub-steppe except in areas of disturbance, consisting of grassland dominated by exotic annual grasses and forbs. Common species include sagebrush (*Artemisia spp.*), antelope bitter brush (*Purshia tridentata*), serviceberry (*Amelanchier alnifolia*), bluebunch wheatgrass (*Pseudoroegneria spicata*), fern-leaved desert-parsley (*Lomatium dissectum*), and cheat grass (*Bromus tectorum*). Other less common plants include rabbitbrush (*Chrysothamnus spp.*), currant (*Ribes spp.*), syringa (*Philadelphus lewisii*), bitter cherry (*Prunus emarginata*), quaking aspen (*Populus tremuloides*), low sagebrush (*Artemesia arbusculus*), and buckwheat (*Eriogonum sp.*). Some riparian woodland exists at low elevations. High elevations are more mesic than low elevations and possess many perennial springs and streams. The terrain is characterized by steep slopes and basaltic outcrops. Open ridges are covered with shrub-steppe and draw bottoms are lined with deciduous shrubs. North-facing slopes contain Douglas fir (*Pseudotsuga menziesii*) forest and deciduous shrubs. Cover types include a mosaic of sagebrush, antelope bitter brush, snowbrush (*Ceanothus velutinis*), choke cherry (*Prunus virginiana*), bitter cherry, rocky mountain maple (*Acer glabrum*), and Douglas fir. Other less common plants include snowberry (*Symphoricarpos albus*), dogwood (*Cornus sericea*), elderberry (*Sambucus racemosa*), mountain alder (*Alnus incana*), currant, syringa, quaking aspen, buckwheat (*Eriogonum sp.*), blue-bunch wheatgrass and cheat grass.

The restoration of mountain quail began in spring 2006 (Troy et al. 2007). This study is a continuation of that project. Releases took place in spring 2008 and 2009 on the Bennett Hills. The Bennett Hills are located about 70 km southeast of Boise, Idaho. The elevation ranges from 900 to 2300 m. The study area encompasses approximately 63,000 ha and is a wildlife migration corridor for many species inhabiting the Sawtooth National Forest of the northern US Rocky Mountains. The annual precipitation ranges from 23 to 40 cm and annual snowfall ranges from 90 to 130 cm.

There were 2 different release sites on the Bennett Hills. Site 1 was adjacent to Bennett mountain road (Easting – 634869, Northing – 4783556, Universal Transverse Mercator WGS 84) and site 2 was approximately 20 km east on the west fork of Dempsey creek (Easting – 653905 Northing – 4778959 Universal Transverse Mercator WGS 84). Site 1 is approximately 250 m higher in elevation than site 2. There are more springs, streams, and cover types at site 1 than site 2 within a 5-km radius. The additional cover type was Douglas fir at site 1, whereas the

closest conifer cover type is approximately 6 km from site 2. The permanence of watercourses is greater at site 1 than site 2, with drainages dominated by perennial as opposed to ephemeral streams.

Predators in the Bennett Hills include the gray wolf (*Canus lupus*), black bear (*Ursus americanus*), golden eagle (*Aquila chrysaetos*), gray fox (*Vulpes velox*), mountain lion (*Puma concolor*), wolverine (*Gulo gulo*), badger (*Taxidea taxus*), bobcat (*Lynx rufus*), pine marten (*Martes americana*), red-tailed hawk (*Buteo jamaicensis*), swainson's hawk (*Buteo swainsoni*), prairie falcon (*Falco mexicanus*) and all 3 of the *Accipiter* genera: sharp-shinned hawk (*Accipiter striatus velox*), Cooper's hawk (*Accipiter cooperii*), and the goshawk (*Accipiter gentilis*). Introduced species of Galliformes include the California quail (*Callipepla californica*), ring-necked pheasant (*Phasianus colchicus*), gray partridge (*Perdix perdix*), and chukar (*Alectoris chukar*).

Low elevations are xeric with few springs and perennial streams; the dominant terrain consists of large plateaus of igneous rock dissected by narrow, steep canyons (Bishop et al. 2005) with ephemeral streams. Low elevations are composed of shrub-steppe except in areas of disturbance, which have converted to annual grassland. Major habitat types include big sagebrush (*Artemisia tridentata*), antelope bitterbrush (*Purshia tridentata*), blue-bunch wheatgrass (*Pseudoroegneria spicata*), and cheatgrass (*Bromus tectorum*). Other less common plants include serviceberry (*Amelanchier alnifolia*), rabbitbrush (*Chrysothamnus nauseosus*), fern-leaved desert-parsley (*Lomatium dissectum*), low sagebrush (*Artemesia arbusculus*), and buckwheat (*Eriogonum sp.*). Some riparian woodland exists at low elevations.

High elevations are mesic with many springs and perennial streams; the dominant terrain consists of steep slopes and basalt outcrops. Open ridges are covered with shrub steppe, draw bottoms are lined with deciduous shrubs, and north-facing slopes contain coniferous forest and deciduous shrubs. Major habitat types include a mosaic of big sagebrush, antelope bitter brush, snowbrush (*Ceanothus velutinis*), choke cherry (*Prunus virginiana*), bitter cherry (*Prunus emarginata*) and Douglas fir (*Pseudotsuga menziesii*). Other less common plants include snowberry (*Symphoricarpos albus*), rocky mountain maple (*Acer glabrum*), buckwheat (*Eriogonum*), quaking aspen (*Populus tremuloides*), blue-bunch wheatgrass, and cheat grass.

The Little Canyon release site (43°11'N,115°20'W) received mountain quail in the initial releases of 2006 and 2007 and supported some mountain quail prior to our release. In other words, with our release at this site, we were attempting to augment a nascent mountain quail population. Our other release site was approximately 20 km east on the west fork of Dempsey Creek (43°8'N,115° 6'W) and 250 m lower in elevation (1,500 m). The two release sites differed in 3 important ways. First, during the first year of my releases (2008) the Little Canyon site likely contained a small number of resident quail, unlike the Dempsey Creek site. Second, the Little Canyon release site contains more watercourse permanence, possessing perennial streams and a greater number of springs and seeps than the Dempsey release site. In short, Little Canyon is a mesic release site compared to the xeric Dempsey release site. Third, the Little Canyon release site has more vegetative cover types within a 5-km radius than Dempsey creek including greater availability of Douglas fir cover.

In addition to physical differences between sites, I was interested in the possible social effects the first year of the release. Mountain quail remain in family groups following hatch and are somewhat gregarious during the non-breeding season. Post release movement, reproduction, and other behaviors could be influenced by the prior presence or absence of mountain quail at the release site. Mountain quail released into an area might adjust their post release movements if they encounter and associate with mountain quail already occupying the area. A possible site effect in the first year of the study was that Little Canyon had a history of recent releases (2006, 2007) whereas Dempsey had not yet been occupied by mountain quail.

Mountain quail were captured in the Cascade and Coastal ranges of western Oregon USA in late autumn through early winter 2007-2008 and 2008-2009. Mountain quail were captured by a private trapper contracted through the Oregon Department of Fish and Wildlife using funnel traps, and held in captivity in Roseburg, Oregon USA until their transfer to an AAALAC certified outdoor aviary on the campus of Idaho State University Pocatello, Idaho USA. Upon receiving mountain quail from Oregon, I extracted 2– ul of blood from juvenile quail for gender determination and additional blood samples were taken from quail during captivity for health assessment by the Idaho Fish and Game Wildlife Health Laboratory in Caldwell, Idaho.

I housed mountain quail in an outdoor aviary in 2 groups, 1 group for each release site. My goal was for mountain quail to have the opportunity to form social relationships such as dominance hierarchies, pair bonds, or forms of familiarity prior to release. I sought to avoid, pair bond formation between 2 birds that would be released into separate sites.

To minimize stress in captivity, I entered the pens in the same manner each time mountain quail were given food and water. Various shrubs, evergreen trees and boughs were dispersed throughout the pens to provide ample hiding cover and thermal protection. Providing escape cover curtailed mountain quail from flushing, thereby minimizing injury of quail from flying into pen walls constructed of mesh netting.

We controlled for diet by feeding all mountain quail ad libitum a complete game bird diet further augmented with ad libitum hard seed and green foodstuffs. We fed mountain quail Purina Game Bird Flight Maintenance (Nestle Purina Pet Care Company, St. Louis, Missouri USA) game bird chow with 18% protein. Female mountain quail fed diets supplemented with xanthophyll, a class of carotenoid pigment found in green vegetation, have been found to lay eggs at a greater rate and more eggs overall during the breeding season than control females (Delehanty 1997). Hence, for 4 weeks prior to release mountain quail diet was supplemented with broccoli crowns to augment carotenoid and other nutrient uptake and to physiologically prepare their gut for natural forage in the wild. We also timed the release date to follow release site snow melt, at time when forbs and grasses have begun new green growth and when release sites were accessible with all-terrain vehicles.

I fitted Mountain quail with 6.3 g necklace-style radio transmitters (RI-2BM Holohil, Inc., Carp, Ontario Canada), less than 3% of the average body mass. Transmitter antennae were bent to orient along the back of the quail. In 2008, mountain quail were released on 1 May at Little Canyon creek, and 2 May at Dempsey creek. We released 22 mountain quail equipped with radio-transmitters at Little Canyon, and 50 mountain quail, of which 23 were equipped with

radio-transmitters at Dempsey (Table 1). At Little Canyon in 2008, we fitted all mountain quail released with radio transmitters for two reasons. First, resident mountain quail from the previous year's releases could have been present. We wanted to detect for the establishment of a nascent population and releasing birds without radio transmitters in 2008 would have caused obscurity. Second, we wanted to detect differences in survival, movement, and reproduction as a possible function of social differences between release sites. In 2009, releases took place at both sites on 24 April . We released 39 birds of which 23 were equipped with radio-transmitters at each site (Table 1).

I used a stratified random sample by separating mountain quail by age and randomly selecting male and female yearlings for radio-marking while maintaining similar sex ratios by site. In 2009, there were more male than female yearlings and all yearling females were radio-marked. I controlled for age by only selecting yearling males and females for radio-marking.

I divided the thesis into 2 main chapters to discuss vital rates in the context of other translocation studies. It is my intent to submit these chapters for publication. Hence, I use journal submission format and have chosen the "we" form to accommodate the multiple authors of the future submission for each chapter. However, each chapter represents work that is centrally mine as a thesis.

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Table 1. The age and gender of all released quail during 2008 and 2009 in the Bennett Hills, Idaho USA. Gender was known for yearling mountain quail but not adults.

	2008		2009	
	Little Canyon	Dempsey	Little Canyon	Dempsey
Adults	0	19	8	8
Yearlings	22	31	31	31
Male	11	15	18	21
Female	11	16	10	10
Unknown	0	19	11	8

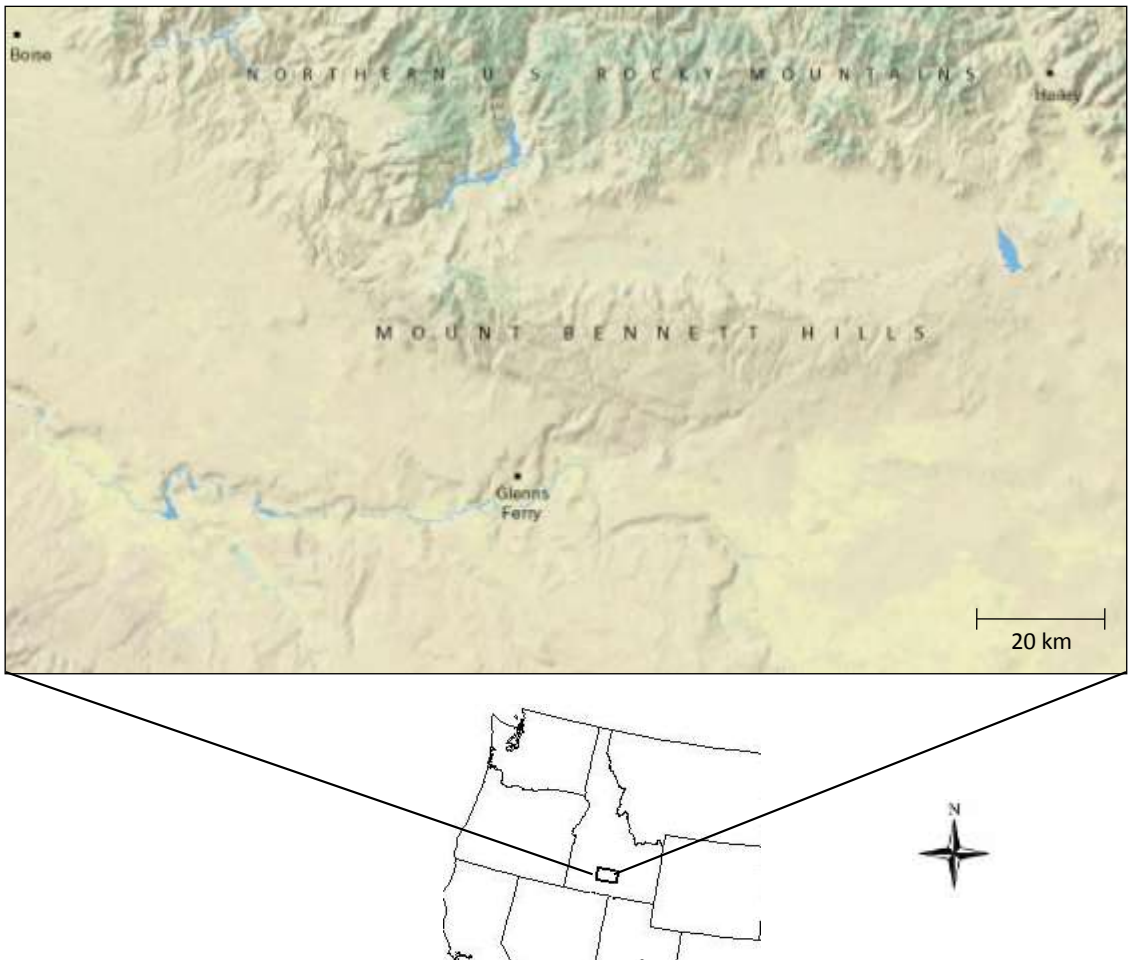

Figure 1. Map of the Bennett Hills study area in south-central Idaho

Chapter 1: Survival and movement of mountain quail translocated to northeastern edge of historic range

Translocation is a common conservation technique for restoring populations of threatened species (Wolf et al. 1998, Dickens et al. 2009). Restoration through translocation generally means that a population is founded with a small number of individuals. The behavior and reproductive success of a small number of individuals and their encounter rate with conspecifics occupying vast spaces becomes centrally important. Hence, closely monitoring the performance of translocated individuals is paramount in understanding the process of population establishment or failure.

Monitoring how survival and reproduction varies across the lifespan of an individual helps us understand population dynamics (Cole 1954). Delineating the dynamic nature of mountain quail populations has been deemed a priority for future research (Guttierrez and Delehanty 1999, Pope 2002, Nelson 2007). Biotic and abiotic factors differentially affect survival of organisms living in a seasonal environment (Carlson and Letcher 2003). Relating seasonal variation in biotic and abiotic factors to seasonal variation in survival is common for some species (Letcher et al. 2002, Mitro and Zale 2002, Carlson and Letcher 2003) but rare for mountain quail. Indeed, distinguishing between biotic and abiotic affects on population variation has become a focal point for understanding population dynamics (Royama 1992, Williams et al. 2003). Moreover, such affects may vary across the range of a species (Nerc 1996**,** Williams et al. 2003). Abiotic factors are widely thought to decrease population size in the range periphery (Dale 2001, Garcia 2001, Bahn et al. 2006). Nonetheless, how population variability changes across the range of a species can be unclear because of destabilizing processes like environmental stochasticity, predation, or habitat condition (Williams et al. 2003).

Within-season variation in survival of mountain quail has been documented (Delehanty 1997, Pope 2002, Reese et al. 2005, Nelson 2007, Stephenson 2008, Troy 2009). Survival trends have been contingent on release date. Most studies are conducted for approximately 150 days – beginning in early spring and ending in late summer when transmitter batteries fail. Observing the number of individuals that survive to reproduce is a good short-term indicator of translocation success (Scott and Carpenter 1987).

Seasonal variation in survival for most populations of mountain quail is poorly documented (Pope 2002), especially variation in survival of a population across multiple seasons and years. Without knowing survival rates for multiple seasons it is difficult to identify factors influencing population dynamics and assess the success or failure of mountain quail translocations leading to population establishment. A few studies have documented survival during autumn and winter (Delehanty 1997, Reese et al. 1999, Pope 2002). However, mountain quail locations are usually less frequent during autumn and winter, making meaningful survival estimates and comparisons of movement across studies difficult. Most information concerning population dynamics of mountain quail includes only survival and reproduction during spring and summer (Reese et al. 2005, Nelson 2007, Stephenson 2008).

Little published information exists regarding seasonal movements of mountain quail and existing information is largely based on anecdotal evidence. Mountain quail are frequently described as

elevational migrants (Grinnell and Storer 1924, Bent 1932, Enderlin 1947, Sumner and Dixon 1953, Edminster 1954, Ormiston 1966, Johnsgard 1973, Leopold et al. 1981, Ehrlich et al. 1988, Brennan 1990, Gutierrrez and Delehanty 1999, Delehanty 2001, Pope 2002, Reese et al. 2005). However, few studies have tracked individual quail during autumn and winter to document timing, rate, and distance of movement or the percentage of migrants within a given population.

Telemetry-based survival studies can be limited (Murray 2006) for a species like mountain quail despite technological advances. Mountain quail declines at the periphery of their range, small body size (240 g), secretive nature, and occupation of remote, steep terrain has made long-term monitoring difficult. The small size of mountain quail limits the size of radio transmitters used to relocate quail, which in turn limits transmitter capabilities and especially transmitter battery life. This means that quail cannot be monitored for more than 6-14 months following radio-marking with a battery powered radio-transmitter. To obtain the repeated sampling necessary for long-term monitoring and document variation in seasonal survival per annum a new method is needed.

There is considerable variation in survival within the breeding season for mountain quail. The first 2 weeks following release explained 75% of survival across a 150-day study period by Nelson (2007). Survival rates have varied from 15% (Stephenson 2008) to 72% (Herman et al. 2002) during the breeding season. Autumn was noted as the period of greatest environmentally driven variability in population dynamics for 3 small vertebrate species (Williams et al. 2003). In contrast, survival rates for mountain quail were similar during winter, spring, and summer in southwestern and northeastern Oregon, USA (Pope 2002). Irruptive population changes may go undetected in year-to-year comparisons of survival (Pope 2002); hence, documenting long-term population trends would be valuable (Nelson 2007). We hypothesize that survival rates vary significantly from year-to-year during autumn and winter for a species, such as mountain quail, that possess many characteristics of an *r*-selected species.

The literature is inconsistent on sex-biased survival with examples of male-biased survival (Pope 2002), female-biased survival (Nelson 2007) or no bias (Stephenson 2008). Habitat quality and predator abundance may influence survival as a function of precipitation and raptor migrations, respectively, to explain differences in survival rates between years (Nelson 2007). High winter precipitation has been suggested as positively influencing survival (Herman et al. 2002). Low survival rates were reported (approximately 20%) for a study of mountain quail released in Western Idaho and Eastern Washington with a negative correlation between movement rate and survival (Stephenson 2008). One of the few consistent trends is that mountain quail have high survival rates during incubation periods and late summer (Nelson 2007, Stephenson 2008). Most hypotheses explaining the cause for variation in survival of mountain quail invoke predation by raptors (Pope 2002, Nelson 2007, Stephenson 2008). Surprisingly, this hypothesis has not been tested directly. Furthermore, proximate factors indirectly related to predation such as movement, predator abundance, habitat condition, and anti-predator behaviors have not been adequately studied (Pope 2002) for mountain quail.

We hypothesized seasonal variation in survival within years and within-seasons. Specifically, we predicted that translocated mountain quail would have high survival rates during periods of incubation (summer) and autumn. We hypothesized that release sites of mountain quail that

differ topographically and edaphically would differ in movements by mountain quail, which would influence survival rates. We predicted that birds released at a low-elevation site (1500 m) would move farther and have a lower survival rate than a high-elevation release site (1750 m). We also postulated that survival rates during winter and spring would vary from year to year as a function of snow accumulation and movement.

The repeated sampling necessary to assess variation in annual survival rates of a particular population of mountain quail has been difficult to attain in areas of low density. Using a unique and innovative method for trapping in late summer and early autumn, we captured mountain quail and monitored them through autumn and winter seasons by replacing their transmitters prior to battery failure. Our objectives included: monitoring movements, reproduction, and survival of a founding population of quail for 2 years, directly testing some of the conflicting hypotheses explaining variation in mountain quail survival during the breeding season, providing basic life-history information lacking for mountain quail, including dispersal distances, age-specific mortality, and lifespan.

STUDY AREA

The Bennett Hills is a rich mix of bunchgrass, perennial herbs and shrubs (Errter and Moseley 1992). Its floristic distinctiveness was not discovered until numerous studies of mule deer winter range began in the 1970's (Errter and Moseley 1992). Low elevations are xeric with few perennial springs and streams. The dominant terrain consists of large plateaus and outcrops of igneous rock dissected by narrow, steep canyons (Bishop et al. 2005) with ephemeral streams. Low elevation vegetation is mostly shrub-steppe except in areas of disturbance, consisting of grassland dominated by exotic annual grasses and forbs. Common species include sagebrush (*Artemisia spp.*), antelope bitter brush (*Purshia tridentata*), serviceberry (*Amelanchier alnifolia*), bluebunch wheatgrass (*Pseudoroegneria spicata*), fern-leaved desert-parsley (*Lomatium dissectum*), and cheat grass (*Bromus tectorum*). Other less common plants include rabbitbrush (*Chrysothamnus spp.*), currant (*Ribes spp.*), syringa (*Philadelphus lewisii*), bitter cherry (*Prunus emarginata*), quaking aspen (*Populus tremuloides*), low sagebrush (*Artemesia arbusculus*), and buckwheat (*Eriogonum sp.*). Some riparian woodland exists at low elevations. There are no junipers (*Juniper spp*) in the study area.

High elevations are more mesic than low elevations and possess many perennial springs and streams. The terrain is characterized by steep slopes and basaltic outcrops. Open ridges are covered with shrub-steppe and draw bottoms are lined with deciduous shrubs. North-facing slopes contain Douglas fir (*Pseudotsuga menziesii*) forest and deciduous shrubs. Cover types include a mosaic of sagebrush, antelope bitter brush, snowbrush (*Ceanothus velutinis*), choke cherry (*Prunus virginiana*), bitter cherry, rocky mountain maple (*Acer glabrum*), and Douglas fir. Other less common plants include snowberry (*Symphoricarpos albus*), dogwood (*Cornus sericea*), elderberry (*Sambucus racemosa*), mountain alder (*Alnus incana*), currant, syringa, quaking aspen, buckwheat (*Eriogonum sp.*), blue-bunch wheatgrass and cheat grass.

Quail releases

In our study, mountain quail were released in the spring of 2008 and 2009 at 2 different release sites within the Bennett Hills. We used 2 sites and 1 source population for 3 reasons. First, by

using 1 source population and 2 release sites, we hoped to be able to parse out release site effects from source population effects. Second, during the 2006-2007 study mountain quail were occupying inaccessible private property and we were unable to gauge reproductive efforts. We judged that adding a second release site would minimize the number of birds occupying inaccessible private property and allow us to determine reproductive rates. Third, this was an applied conservation project with the express goal of restoring mountain quail across the landscape more broadly in the third and fourth years of releases with the hope of developing a broader zone of occupation by mountain quail.

The Little Canyon release site (43°11'N, 115°20'W) received mountain quail in the initial releases of 2006 and 2007 and supported some mountain quail prior to our release. Thus, with our release at this site, we attempted to augment a nascent mountain quail population. Our other release site was approximately 20 km east on the west fork of Dempsey Creek (43°8'N,115° 6'W) and 250 m lower in elevation (1,500 m). The 2 release sites differed in 3 important ways. First, during the first year of releases (2008) the Little Canyon site likely contained a small number of resident quail, unlike the Dempsey Creek site. Second, the Little Canyon release site contains more watercourse permanence, possessing perennial streams and a greater number of springs and seeps than the Dempsey release site. In short, Little Canyon is a mesic release site compared to the xeric Dempsey release site. Third, the Little Canyon release site has more vegetative cover types within a 5-km radius than Dempsey creek including greater availability of Douglas fir cover.

METHODS

Mountain Quail Capture, Captivity and Radiotelemetry

Mountain quail were captured in the Cascade and Coastal ranges of western Oregon USA in late autumn through early winter 2007-2008 and 2008-2009. Mountain quail were captured by a private trapper contracted through the Oregon Department of Fish and Wildlife using funnel traps, and held in captivity in Roseburg, Oregon USA until their transfer to an AAALAC certified outdoor aviary on the campus of Idaho State University Pocatello, Idaho USA.

I housed mountain quail in an outdoor aviary. I housed mountain quail in two groups, one group for each release site. My goal was for mountain quail to have the opportunity to form social relationships such as dominance hierarchies, pair bonds, or forms of familiarity prior to release. I sought to avoid pair bond formation between 2 birds that I knew would be released into separate sites.

To minimize stress in captivity, I entered pens in the same manner each time mountain quail were given food and water. Various shrubs, evergreen trees and boughs were dispersed throughout the pens to provide ample hiding cover and thermal protection. Providing escape cover curtailed mountain quail from flushing, thereby minimizing injury of quail from flying into pen walls constructed of mesh netting.

We controlled for diet by feeding all mountain quail ad libitum a complete game bird diet further augmented with ad libitum hard seed and green foodstuffs. We fed mountain quail Purina Game

Bird Flight Maintenance (Nestle Purina Pet Care Company, St. Louis, Missouri USA) game bird chow with 18% protein. Female mountain quail fed diets supplemented with xanthophyll, a class of carotenoid pigment found in green vegetation, have been found to lay eggs at a greater rate and more eggs overall during the breeding season than control females (Delehanty 1997). Hence, for 4 weeks prior to release mountain quail diet was supplemented with broccoli crowns to augment carotenoid and other nutrient uptake and to physiologically prepare their gut for natural forage in the wild. We also timed the release date to follow release site snow melt, at time when forbs and grasses have begun new green growth and when release sites were accessible with all-terrain vehicles.

I weighed and fitted mountain quail in captivity pens just prior to transport and release with 6.3 g necklace-style radio transmitters (RI-2BM Holohil, Inc., Carp, Ontario Canada), less than 3% of the average body mass. Transmitter antennae were bent to orient along the back of the quail. In 2008, mountain quail were released on 1 May at Little Canyon creek, and 2 May at Dempsey creek. We released 22 mountain quail equipped with radio-transmitters at Little Canyon, and 50 mountain quail, of which 23 were equipped with radio-transmitters at Dempsey (Table 1). At Little Canyon in 2008, we fitted all released mountain quail with radio transmitters for 2 reasons. First, resident mountain quail from the previous year's releases could have been present. We wanted to detect for the establishment of a nascent population and releasing birds without radio transmitters in 2008 would have caused obscurity. Second, we wanted to detect for differences in survival, movement, and reproduction as a possible function of social differences between release sites. In 2009, releases took place at both sites on 24 April. We released 39 birds of which 23 were equipped with radio-transmitters at each site (Table 1).

I used a stratified random sample by separating mountain quail by age and randomly selecting male and female yearlings for radio-marking while maintaining similar sex ratios by site. In 2009, there were more male than female yearlings and all yearling females were radio-marked. I controlled for age by only selecting yearling males and females for radio-marking.

Following release, mountain quail were located on average every 4 days through ground and aerial radiotelemetry. In the first and second months following translocation, aerial surveys were conducted every 2 weeks and once a month thereafter for the remainder of the breeding season. Upon locating a mountain quail from the ground, we recorded location (UTM) using handheld GPS devices, elevation (m), slope (%), slope directional aspect (degrees), habitat type, distance from release site (m), and distance to surface water (m).

To provide an opportunity to monitor released quail for longer than the typical summer period, we recaptured mountain quail and replaced their radio-transmitter during new moon phases of September, October, and November as weather conditions permitted in 2007-2009. We used night-netting to recapture quail. Briefly, this entailed locating roosting quail at night using a combination of radiotelemetry and spotlighting, then capturing quail with a modified net on the end of a long pole. We report survival of translocated populations of mountain quail during autumn and winter for 4 different years. Mountain quail from the 2006-2007 study were released near the Little Canyon release site and consisted of mountain quail from 2 source populations. During autumn and winter, quail locations were less frequent than the breeding season and were obtained once per month via aerial telemetry and once per month via telemetry on the ground.

Monitoring during winter was conducted with snowmobiles, cross-country skis, and snowshoes to access remote areas occupied by mountain quail.

Movement

We assembled an initial movement profile for each mountain quail beginning on the day of release through 150 days of monitoring. The minimum total distance a bird moved was the cumulative sum of straight-line distances between serial sightings through 150 days of monitoring or until quail death within the first 150 days following release. Minimum rate of movement was calculated as the straight-line distance between resightings divided by the number of days between locations through 150 days of monitoring or until death. Movement direction from release location to breeding location and from breeding location to winter range was calculated using National Geographic TOPO Version 4.2.8 mapping software. We separated movement direction into 4 quadrants: NE (1-90°), SE (91-180°), SW (181-270°), and NW (271-360°) to assign mountain quail movement away from the release location to directional categories.

Statistical Analysis

Overall, data for movement had equal variances but was non-normally distributed. To test for differences in movement distance and rate we used a Wilcoxon-Mann-Whitney rank sum test in the program R (R-Statistics 2004). Where data were both nonnormal and heteroscedastic, we used a non-parametric Brunner-Munzel test using the statistical program R (Aho in review). Where data were only marginally non-normal ($0.10 > p > 0.05$), we performed *t*-tests to verify if conclusions would have been different than Brunner-Munzel tests. To address sensitivity to non-normality we used the Fligner-Killeen test for heteroscedasticity (Aho in review).

Survival estimates for the breeding season were calculated with the Kaplan Meier estimator (Kaplan and Meier 1958). To compare estimated survival functions, we used a log rank test for staggered entry design (Pollock et al. 1989). We conducted logistic regression in program R to analyze survival (Aho in review). Logistic models consisted of 7 explanatory variables; time, sex, mass (the day before transport and release), year, release site, minimum distance moved, and minimum movement rate.

To develop hazard models, we used maximum partial likelihood estimation (Cox 1972) in program R and setting $\alpha \leq 0.05$ for all statistical comparisons. We tested for uniformity in movement direction from release site to breeding range and from breeding range to winter range using a Rayleigh test (Zar 1999). We did not use a modification of the Rayleigh test (Durand and Greenwood 1958) to test for movement in a preferred direction because we had no *a priori* hypothesis that birds would move a particular direction. Furthermore, in 2008 we did not observe a preferred direction of movement within a range of 30° (Durand and Greenwood 1958) to establish an *a priori* hypothesis about movement in 2009. We used the Fisher exact test to compare the frequency of quail that exhibited seasonal movement and quail that did not move seasonally during autumn and winter months between years.

Data for snowpack were obtained from the United States Department of Agriculture (USDA) Natural Resources Conservation Sources (NRCS). The USDA NRCS collect data at the end of December, January, February, and March (http://www.wcc.nrcs.usda.gov). We report data collected at three different sno-tel sites we refer to as low (1200 m), medium (1350 m), and high (1525 m) elevation in the Bennett Hills.

RESULTS

Movement during the breeding season

Most mountain quail moved from 50 m to 23 km during the first 4 weeks following release; after establishing a home range quail moved very little during the remainder of the 150 day breeding season. On those occasions that quail did move after establishing a home range, the movement usually occurred during the hottest days of the year and quail moved to areas where surface water was readily available and remained in that area.

We found no statistical difference for the total distance moved between Little Canyon and Dempsey over the 2-year study, (W = W = 849, df = 87, p = 0.268; Table 1). The average total minimum distance moved by male (8.4 ± 0.9 km) mountain quail relative to female (6.2 ± 1.0 km) mountain quail was significantly greater (W = 746, df = 87, p = 0.029; Table 1) during the 2-year study.

Average total minimum distance mountain quail moved post-release at Little Canyon (5.1 ± 1.1 km) was less than Dempsey (11.4 ± 1.5 km) in 2008; mountain quail moved significantly farther at the low elevation site (W = 109, df = 41, p = 0.002; Fig 1). Furthermore, median movement at Dempsey was much higher than at Little Canyon (Fig 1). Average male movement (9.0 ± 1.3 km) did not differ from average female movement (7.6 ± 1.6 km) in 2008 (W = 198, df = 41, p = 0.108; Table 1).

In 2009, the average total distance mountain quail moved at Little Canyon (7.3 ± 1.0 km) was numerically greater than Dempsey (6.3 ± 1.4 km) following release but not significantly so (W = 203.5, df = 44, p = 0.092; Table 1). The numerically greater mean male movement (8.3 ± 1.2 km) versus mean female movement (4.4 ± 0.6 km) was significant in 2009 (W = 173, df = 44, p = 0.039; Table 1).

Mountain quail moved significantly less at Little Canyon in 2008 (5.1 ± 1.1 km) than 2009 (7.3 ± 1.0 km, W = 158, df = 43, p = 0.0160; Table 1). The opposite trend occurred at Dempsey. The average total distance mountain quail moved at Dempsey in 2008 (11.4 ± 1.5 km) was significantly greater than at Dempsey in 2009 (6.3 ± 1.4 km, W = 134.5, df = 44, p = 0.006; Table 1).

We only report statistical comparisons for total distance mountain quail moved. Statistical conclusions for movement rate and total distance moved were different only once when comparing the Dempsey site between years; total distance moved was statistically different between years (p = 0.006) but movement rate was not (p = 0.086).

Movement direction from the release was uniform only when pooling data for 2008 ($0.10 > p > 0.05$; Table 2). Otherwise, movement direction by year and site were not uniform in distribution ($p < 0.05$; Table 2). Mountain quail did not appear to move a single directional quadrant by year or by release site (Table 3). Movement direction from release site to home range was northwest or southwest for 67% of mountain quail compared to 33% dispersing northeast or southeast. At Little Canyon only 2% of birds dispersed in a northeast direction and only 4% of birds moved in a southeast direction at Dempsey (Table 3).

Autumn movement

Most mountain quail surviving through the breeding season did not move from their breeding range. During autumn 2007, 2008, and 2009 we observed 13 of 35 (37%) monitored mountain quail move from their breeding range an average of 6.1 km (range: 1.7 – 15.0 km). We observed similar kinds of movements during winter. Mountain quail moved to lower and higher elevations during autumn. Mountain quail that moved to lower elevation descended an average of 323 m (n=7, range: 86 – 548 m). However, some quail moved up an average of 151 m (n=6, range: 52 – 252 m). Movement from breeding range to winter range was not uniform in distribution ($p < 0.02$). We did not detect autumn movements in any particular cardinal direction.

Six mountain quail moved during autumn 2007, one mountain quail moved during the autumn 2008, and 6 mountain quail moved during autumn 2009. The frequency of migrating birds was greater during 2007 (6 of 11) and 2009 (6 of 11) than 2008 (1 of 13) ($p = 0.020$; Table 9). In 2007, all migration movements were during October or November. The single mountain quail migration in 2008 began in September. Lastly, in 2009, all 6 migrations occurred at the end of September and first week of October following an early season snow storm on September 23, 2009 with snow accumulations up to 41 cm at 1,838 m in elevation (pers. comm. Lee Blackwell).

Mountain quail released at Dempsey moved from breeding ranges more frequently (6 of 11, 55%) than mountain quail released at Little Canyon (7 of 24, 29%) but movement frequency was not significantly different ($p = 0.258$; Table 9). Mountain quail at Dempsey gained elevation (mean = 95 m) with autumn movements compared to a decrease in elevation at Little Canyon (mean = 274.4 m). During the autumn and winter of 2006-07 mountain quail did not have transmitters with mortality signals and we were unable to do aerial telemetry consistently to monitor movements.

Survival during the breeding-brooding season

We monitored 91 birds during the 150-day breeding seasons of 2008 (n = 45) and 2009 (n = 46). Seven of 91 (8%) radio-marked quail were censored during the breeding season. Thirty-two of 91 (35%) mountain quail survived that time period. Mountain quail had the lowest survival rates during May in 2008 (32 of 45; 71%) and 2009 (30 of 46; 65%).

We did not observe differences in Kaplan Meier survival estimates between release sites ($X^2 = 0.96$, $p = 0.328$; Table 4). Nor did we observe significant differences in survival between any combination of site or year. Across years, the Kaplan Meier survival estimates, which take into

account censored birds, was 34% at Little Canyon and 28% at Dempsey. In 2008, Kaplan Meier estimates for 150 days were 37% at Little Canyon and 34% at Dempsey, not significantly different between sites ($X^2 = 0.01$, $p = 0.932$; Table 4). In 2009, the probability of an individual bird surviving to 150 days was 32% at Little Canyon and 12% at Dempsey, not significantly different between sites ($X^2 = 2.50$, $p = 0.114$; Table 4).

We observed a difference in survival between sexes ($X^2 = 9.0$, $p = 0.003$; Table 4). Over the 2-year study, Kaplan Meier estimates for the 150-day breeding season were 6% for females and 35% for males (Table 4). Each site each year exhibited male-biased survival (Table 5). There were 3 censored birds in 2008 during the breeding season and 4 censored birds in 2009.

Logistic regression survival models

Movement rate and total distance moved by mountain quail were important parameters in logistic regression models explaining survival. Distance moved and movement rate were significant predictors of survival and present in the most competitive models in 2008, 2009, and when combining years (Table 6, Table 7, Table 8). Overall, movement rate was the most important single parameter explaining survival (Table 8). The interaction of distance moved and movement rate and the interaction of sex, movement rate, and distance moved were competitive models explaining survival for both years combined (Table 8). There was a negative correlation of movement rate with survival and a positive correlation of distance moved with survival.

In 2008, sex was the most important single parameter explaining survival (Table 6) and present in the most competitive models. The interaction of sex, distance moved, and movement rate was the most competitive model explaining survival in 2008 (Table 6). In 2009, movement rate was the most important single parameter explaining survival and present in all of the competitive models (Table 7). The interaction of movement rate and distance moved was the best model explaining survival (Table 7).

Snow accumulation and survival during autumn and winter

During the winter of 2006-07, average monthly snow accumulation was 6.8 cm and slightly below the 10-year average (7.6 cm, Fig 3). Four of 10 mountain quail survived from October 2006 until March 2007. Mountain quail did not have transmitters equipped with mortality signals and we were unable to conduct aerial telemetry consistently to determine monthly survivorship during that time and determine if mountain quail perished in sequence with stochastic weather events or predation from October 2006 to March 2007.

During winter 2007-2008 the average monthly snow accumulation was 17.1 cm. One of 12 birds survived from October 2007 until March 2008. Mountain quail had high survival rates during autumn (85%) and low survival rates during winter (10%). The average snow accumulation during 2007-2008 was substantially higher than the year before, the year after, and the 10-year average (Fig 3). Furthermore, snow accumulations at low elevation sno-tel sites were as high as 800% of the 10-year average. The only bird surviving that winter migrated 3.1 km from its breeding range and descended 284 m in elevation. An adult female and hatch-year bird migrated 5.6 km from their breeding range and descended 548 m in elevation. They survived high snow

accumulations in December and January until the second week of February when they were apparently killed by a predator at their nocturnal roost sight. The remaining quail died during winter months during the period of high snow accumulation.

The 2008-2009 average monthly snow accumulation was 9.1 cm and slightly above the 10-year average (Fig 3). One of 12 mountain quail survived from October 2008 until March 2009. The Bennett Hills received almost no snow accumulation from September to December 2008 and mountain quail survival during autumn has high (94%). Survival rates were low during winter (8%). A storm system in mid-December accumulated almost a meter in snow over 48 hours and all winter mortalities occurred during this month (Fig 4). The following January at Little Canyon we were able to recover 6 of 11 mortalities. Dead mountain quail were found mostly intact under an average snow depth of 56.1 cm (range: 15.2 – 130.8 cm). Necropsy of recovered winter mortalities revealed that half the quail (n = 3) died directly from snow accumulation – ultimately starving to death under a meter of snow. The other half (n = 3) of winter mortalities contained external puncture wounds presumably from a predator, one of which was cached by a small mammal and found with the body in apparent good condition.

DISCUSSION

A central issue in population dynamics is separating the roles of density-dependent and density-independent processes in determining the variability of population abundance (Royama 1992). Because population variation is driven by the interaction between environmental variability and density dependence in some species (Ives 1995, Turchin 1995), it has been difficult to isolate their separate effects (Sharov 1992, Ranta etal. 1997).

Without following this translocated mountain quail population through multiple seasons we would have had reason to believe, based on high survival rates and reproductive rates during the breeding season, that a founding population would become self-sustaining. Long-term monitoring of marked individuals helps to understand how populations respond to environmental change (Duriez et al. 2009) and has obvious benefits in understanding survivorship. A modified trapping technique enabled us to monitor post-release performance of mountain quail for multiple seasons. Monitoring mountain quail for multiple seasons revealed the majority of quail were not moving from breeding ranges prior to stochastic winter events and were suffering high winter mortality. Furthermore, biotic and abiotic factors differentially affected mountain quail survival in a seasonal environment at the northeastern periphery of historic range.

During both years of the study survival rates were lowest during the first month following release, consistent with other mountain quail studies (Nelson 2007, Stephenson 2008). High mortality rates immediately following release (2-4 weeks) continue to impact population dynamics and establishment of a self-sustaining population of mountain quail. Mechanisms influencing high mountain quail mortality rates immediately following translocation are unclear and need further research.

Survival rates were high during periods of incubation in late June and early July. This trend represents one of the few generalizations that can be made about population dynamics of mountain quail throughout their range (Pope 2002, Nelson 2007, Stephenson 2008). Spring and

summer weather patterns may directly and indirectly affect population dynamics. Studies have noted spring and summer weather patterns may be a mechanism driving seasonal variation in survival by decreasing survival rates (Nelson 2007, Stephenson 2008). For example, the first two weeks after the release in 2009 and during mountain quail dispersal from the release site to breeding range, the Bennett Hills received high precipitation for successive days. Mountain quail were inactive during spring rains while using dense shrub cover. Quail commenced long distance movements immediately following rain subsidence. The highest rate of mortality in 2009, which we attributed to mammalian predators, was during successive days of spring rains when quail were stationary, occupying heavy cover. We also observed high rates of mortality that we attributed to raptors immediately following rainstorms when both mountain quail and raptors were active again. The effect of mammalian and avian predators as a function of high precipitation for successive days may have lead to the 10% higher mortality rates we observed for May and June 2009 compared to 2008. Thus, abiotic perturbations appeared to indirectly influence survival rates during the breeding season.

Movement-associated survival has been reported for reptiles (Bonnet el al. 1998; French snakes, *Coluber spp., Elaphe spp., Vipera spp., Natrix spp.*, Reinert and Rupert 1999: rattlesnakes, *Crotalus horridus* Gruber and Henle 2008 geckos, *Gehrya variegate*, Hester et al. 2008; eastern box turtle, *Terrapene Carolina*,) and birds (Warner et al. 1999: pheasants, *Phasianus colchicus*, Williams et al. 2000: quail, *Colinus virginianus*). Furthermore, small mammals (Byrom and Krebs 1999: ground squirrels, *Spermophilus parryii*, Schwartz et al. 1998: marmots, *Marmota flaviventris*,), ungulates (McNay and Voller 1995: black-tailed deer, *Odocoileus hemionus columbianus*, Petersburg et al. 2000; elk, *Cervus elaphus*,), and other mammalian predators (Chamberlain et al. 1999: bobcats, *Lynx rufus*, Moehrenschlager & Macdonald 2003: foxes, *Vulpes velox*, Benson and Chamberlain 2007: black bears, *Ursus americanus luteolus*, Spinola et al. 2008: river otters, *Lontra canadensis*) exhibit movement-associated survival. For mountain quail, Gutierrez (1980)suggested that increased movement decreases survival . Stephenson (2008) noted that movement rate was negatively correlated with survival and advised managers to minimize post-release movements through release technique.

Associating movement with survival can be misleading because of complex interactions. For example, the total minimum distance mountain quail moved was positively correlated with survival while minimum movement rate was negatively correlated with survival. Minimum distance moved and minimum movement rate were important parameters in logistic regression models explaining survival in 2008, 2009, and when combining years. We observed mountain quail moving across barren lava reefs all seasons of the year at Dempsey similar to observations by Brennan et al. (1987) on the Modoc plateau. Furthermore, mountain quail at Dempsey maintained survival rates consistent with mountain quail that moved significantly less and across a Little Canyon landscape with relatively contiguous cover during the same time period. Specifically, mountain quail released at the xeric, low-elevation Dempsey site in 2008 moved 5 km farther on average than mountain quail at the mesic, high-elevation Little Canyon release site during the breeding season. When movement was significantly different between release sites or years, survival was not significantly different for the same comparison between sites or years. In contrast, males and females were the only comparison where a significant difference in survival and movement occurred simultaneously. Male mountain quail moved significantly further than females and survival rates were significantly higher for males than females. Regardless, Kaplan

Meier survival estimates were similar between release sites throughout the 150-day breeding season both years despite significant differences in minimum distance moved and minimum movement rates.

Few studies have investigated mountain quail survival and movement during autumn and winter. Trapping efforts in late summer enabled us to monitor beyond the 150-day spring and summer field season. Although autumn and winter monitoring was not the main objective of our study, we observed within-season variation in survival between years. Autumn 2007 and 2008 survival rates were high (>85%) and winter survival rates were low (≤10%). During translocations in 2007 and 2008 the prevailing winter condition appeared to be the ultimate factor influencing survival of mountain quail. However, low survival rates during autumn 2009 (54%) and over-winter survival of 40% during 2006-2007 suggests within-season variation in survival for autumn and winter. In 2006, predation pressure appeared to be the ultimate factor influencing population dynamics with 4 of 10 mountain quail surviving the winter.

The pattern of high winter mortality during 2007-08 and 2008-09 is not a direct result of predation or habitat condition but a classic example in conservation biology of how small populations are susceptible to abiotic stochastic events (Letcher et al. 2002, Mitro and Zale 2002, Carlson and Letcher 2003). The Bennett Hills varies more in elevation and receives more snow fall than other recent mountain quail study areas (Pope and Crawford 2001, Reese et al. 2005, Stephenson 2008, http://www.wcc.nrcs.usda.gov). However, mountain quail may be snow-limited in many areas at the periphery of historic mountain quail range.

We relate seasonal variation in biotic and abiotic factors to seasonal variation in survival (Letcher et al., 2002; Mitro & Zale, 2002). The winters of 2007-2008 and 2008-2009 support a prediction by Garcia (2001) that variables such as weather or other abiotic factors should have a greater influence on demographic rates in border than in central areas of the distribution of a species, provided that climatic adaptation might be important in determining range borders. As a result of high winter mortality during 2007-2008 and 2008-2009, we were unable to document movement from winter range to breeding ranges or basic life-history information such as lifespan, dispersal, and age-specific mortality by following individuals for successive years.

The majority of mountain quail translocated to the Bennett Hills did not move from breeding ranges. Movement from breeding ranges during autumn may increase the probability of survival. Whether mountain quail migration is primarily innate, depends on environmental cues, or is influenced by cultural transmission is unknown. It is also unclear how much flexibility exists in migration strategy. The differential mountain quail movement we observed in autumn suggests behavioral plasticity in migration. Many factors of life history can influence how a bird travels through a diverse landscape including natal habitat, fledging date, molting patterns, wintering location, flight capacity, and migration timing (Kelly et al. 1999, Delong et al. 2008, Carlisle et al. 2009). The autumn movements in 2009 immediately following an early season snowstorm as well as the lack of periodicity in movement between years may be evidence that environmental variables control mountain quail migration. Short-distance migration is generally influenced by environmental variables whereas long-distance migration is influenced by endogenous, circannual rhythms (Gwinner 1972, Berthold 1973, 1975) but few field studies have distinguished the two (Meunier et al. 2008). Species that are short-distance migrants should be

able to respond to a labile environment (Carlisle et al. 2009). Mountain quail appeared to respond to the early season snowstorm in 2009. Further research is needed to determine how mountain quail use habitat during seasonal movements, how seasonal movements, or the lack thereof, influences survival, the connectivity between breeding, migrating and wintering areas, and to what degree mountain quail alter their behavior in response to a changing environment. Each is part of a broader construct of emerging questions and migration research needs for avian species (Skagen et al. 2004, Carlisle et al. 2009).

Management Implications

For most species of new world quail, predation is the primary cause of mortality (Rollins and Carrol 2001, Pope 2002) and biotic factors are often the likely culprit for mortality in a wide range of taxa. Our current understanding about the influence of environmental stochasticity on population dynamics of mountain quail is limited. We observed low winter survival when monthly snow accumulations during winter were above the 10-year average (7.5 cm). Most studies of mountain quail have focused on biotic effects during traditional spring-summer field season**.** In either instance (biotic or abiotic), to document the population dynamics of a nascent population and determine translocation success, mountain quail must be followed through time (Ehrlich et al. 1972) on an annual basis. If the trend of high mountain quail mortality rates immediately following release is not subdued, at least 45 mountain quail need to be fitted with radio-transmitters prior to release to monitor a translocated population of mountain quail per annum.

We believe the release site is a key to mountain quail translocation success. High mortality rates following release may be a function of chronic stress resulting from translocation (Cyr et al. 2009, Dickens et al. 2009). Release technique could reduce mortality rates. An innovative soft-release could decrease the stress of a novel release habitat and allow individuals time to recover physiologically to baseline levels after the stress of capture, handling, and transport (Dickens et al. 2009). Managers should take care to reduce stress during all phases of translocation including capture, handling, captivity, transport, and release. Lastly, low-elevation releases may allow mountain quail to learn migration corridors and suitable winter habitat, thereby, increasing the likelihood of migration during autumn and, in turn, increasing survival during winter.

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Table 1. The mean distance mountain quail moved during the study and the Wilcoxon Mann-Whitney rank sum test comparing mountain quail movement between categories.

Category		n	Mean Distance Moved (km)	SE
Overall		89	7.4	0.66
2008		43	8.2	1.04
	Site 1	22	5.1	1.13
	Site 2	21	11.4	1.50
2009		46	6.8	0.83
	Site 1	23	7.3	0.98
	Site 2	23	6.3	1.36
Site 1		45	6.2	0.76
Site 2		44	8.7	1.07
Males		50	8.4	0.88
Females		39	6.2	0.96

Comparisons		W	P
2008	2009	899	0.231
Site 1	Site 2	849	0.268
Site 1 2008	Site 2 2008	109	0.002
Site 1 2009	Site 2 2009	204	0.095
Males	Females	746	0.029
Site 1 2008	Site 1 2009	158	0.016
Site 2 2008	Site 2 2009	134.5	0.006

Table 2. Rayleigh test for uniformity in dispersal direction from release site to home range. A *p*-value less than 0.05 signify no uniformity in dispersal direction.

	n	*p* - value
Overall	88	< 0.001
2008	45	> 0.05
2009	43	< 0.001
Site 1	42	< 0.001
Site 2	46	< 0.001
Site 1 2008	22	< 0.002
Site 2 2008	23	< 0.001
Site 1 2009	20	< 0.001
Site 2 2009	23	< 0.001

Table 3. The percent dispersal direction from release site to home range for mountain quail in the Bennett Hills, Idaho, USA.

	n	Northwest	Southwest	Northeast	Southeast
Overall	88	0.35	0.32	0.24	0.09
Site 1	42	0.29	0.52	0.02	0.17
Site 2	46	0.48	0.20	0.28	0.04
2008	45	0.38	0.24	0.27	0.11
Site 1	22	0.23	0.50	0.05	0.23
Site 2	23	0.52	0.00	0.48	0.00
2009	43	0.40	0.47	0.05	0.09
Site 1	20	0.35	0.55	0.00	0.10
Site 2	23	0.43	0.39	0.09	0.09

Table 4. Kaplan Meier survival estimate and log-rank chi-square tests comparing survival between sites, year, and sex for mountain quail translocated to the Bennett Hills, Idaho, USA in 2008 and 2009.

	Survival Estimate
Overall	0.29
2008	0.37
Site 1	0.37
Site 2	0.34
2009	0.26
Site 1	0.32
Site 2	0.12
Females	0.06
Males	0.35

Comparison		X^2	*p*
Site 1 2008	Site 2 2008	0.01	0.932
Site 1 2009	Site 2 2009	2.50	0.114
Site 1 2008	Site 1 2009	0.06	0.799
Site 2 2008	Site 2 2009	3.25	0.071
2008	2009	0.68	0.411
Males	Females	9.00	0.003

Table 5. The percent survivorship for males and females by year and site in the Bennett Hills, Idaho, USA in 2008 and 2009.

	n	Survivorship
Females	36	22%
Males	46	48%
2008 Females	20	30%
2008 Males	20	60%
2009 Females	16	13%
2009 Males	26	38%
Females Site 1	17	24%
Males Site 1	25	56%
Females Site 2	19	21%
Males Site 2	21	38%

Table 6. The most competitive logistic regression models explaining survival during the 150-day breeding-brooding season in 2008 using Akaike's Information Criterion with correction term for small sample sizes for mountain quail translocated to the Bennett Hills, Idaho, USA. *p*- value is the result of type II sums of squares Anova for each parameter in the model.

Model		K	AIC_c	ΔAIC_c	w_i
Sex + DM + MR		4	39.01	0.00	0.36
DM + MR		3	40.36	1.35	0.18
Sex + DM + MR + Weight		5	40.46	1.45	0.18
Sex + DM + MR + Place		5	41.60	2.59	0.10
DM + MR + Weight		4	42.67	3.66	0.06
Sex + DM + MR + Weight + Place		6	42.82	3.81	0.05
DM + MR + Place		4	42.89	3.88	0.05
Sex + DM + Weight		4	45.29	6.28	0.02
Sex + MR		3	53.90	14.89	0.00
Sex + MR + Weight		4	55.87	16.86	0.00
MR=Movement rate, DM=Distance moved					
Parameter	*p*	K	AIC_c	ΔAIC_c	w_i
Sex	0.021	2	56.94	0.00	0.63
Movement rate	< 0.0001	2	58.75	1.81	0.25
Distance moved	< 0.0001	2	61.13	4.19	0.08
Weight	0.206	2	63.77	6.83	0.02
Place	0.501	2	64.07	7.13	0.02

Table 7. The most competitive logistic regression models explaining survival during the 150-day breeding-brooding season using Akaike's Information Criterion with correction term for small sample sizes for mountain quail translocated to the Bennett Hills, Idaho, USA in 2009. *p*- value is the result of type II sums of squares Anova for each parameter in the model.

Model	K	AIC_c	ΔAIC_c	w_i
MR + DM	3	38.75	0.00	0.27
MR + DM + Place	4	39.02	0.27	0.23
MR + DM + Weight	4	39.42	0.67	0.19
MR + DM + Weight + Place	5	41.16	2.41	0.08
MR + DM + Sex	4	41.28	2.53	0.08
MR + DM + Sex + Place	5	41.31	2.56	0.07
MR + DM + Sex + Weight	5	42.07	3.32	0.05
MR + Sex + Place	4	43.71	4.96	0.02
DM + Weight	3	50.98	12.23	0.00
MR	2	51.23	12.48	0.00

DM = Distance moved, MR = Movement rate

Parameter	*p*	K	AIC_c	ΔAIC_c	w_i
Movement rate	< 0.0001	2	50.66	0.00	0.83
Distance moved	< 0.0001	2	54.11	3.45	0.15
Weight	0.528	2	58.75	8.09	0.01
Sex	0.622	2	61.89	11.23	0.00
Place	0.283	2	62.79	12.13	0.00

Table 8. The most competitive logistic regression models explaining survival during the 150-day breeding-brooding season in 2008 and 2009 using Akaike's Information Criterion with correction term for small sample sizes for mountain quail translocated to the Bennett Hills, Idaho, USA. *p*- value is the result of type II sums of squares Anova for each parameter in the model.

Model	K	AICc	ΔAICc	wi
DM + MR	3	72.31	0.00	0.21
DM + MR + Sex	4	72.46	0.15	0.20
DM + MR + Sex + Place	5	72.82	0.51	0.17
DM + MR + Place	4	73.14	0.83	0.14
DM + MR + Weight	5	73.94	1.63	0.09
DM + MR + Sex + Weight	5	74.28	1.97	0.08
DM + MR + Year	4	74.47	2.16	0.06
DM + MR + Sex + Weight + Place	6	75.04	2.73	0.05
DM + MR + Sex + Year + Place	6	75.15	2.84	0.05
DM + MR + Place + Weight	5	75.17	2.86	0.05
DM = Distance moved, MR = Movement rate				

Parameter	*p*	K	AIC_c	ΔAIC_c	w_i
Movement rate	< 0.0001	2	105.66	0.00	0.97
Distance moved	< 0.0001	2	112.67	7.01	0.03
Sex	0.116	2	117.45	11.79	0.00
Weight	0.209	2	120.08	14.42	0.00
Place	0.730	2	122.10	16.44	0.00
Year	0.939	2	123.38	17.72	0.00

Table 9. The percentage of mountain quail monitored during autumn and winter that migrated and the mean distance and change of altitude during migration in the Bennett Hills from autumn 2007 to winter 2009. Results of a Fisher exact test comparing frequency of migrating birds.

	n	Migration (%)	Mean Distance (km)	Mean Δ Altitude (m)
Overall	35	0.37	6.1	-104.0
Little Canyon	24	0.29	4.9	-274.7
Dempsey	11	0.55	7.6	95.0
2007	11	0.55	5.1	-362.5
2008	13	0.08	3.0	86.0
2009	11	0.55	7.7	122.7

Comparisons	*p* – Value
2007 vs. 2008	0.027
2007 vs. 2009	1.000
2008 vs. 2009	0.020
Dempsey vs. Little Canyon	0.258

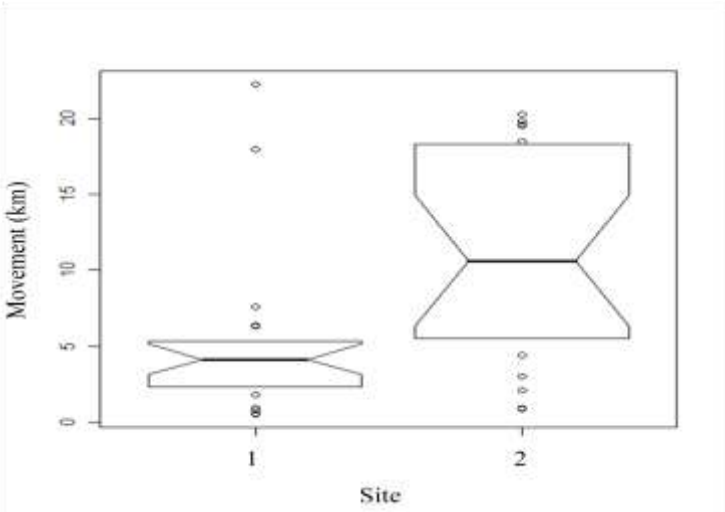

Figure 1. Distance mountain quail moved at site 1 (*n* = 22) and site 2 (*n* =21) during 150 days of the breeding-brooding season during 2008, Bennett Hills, Idaho., USA. Notches represent 95% confidence intervals and bold horizontal lines represent the median movement.

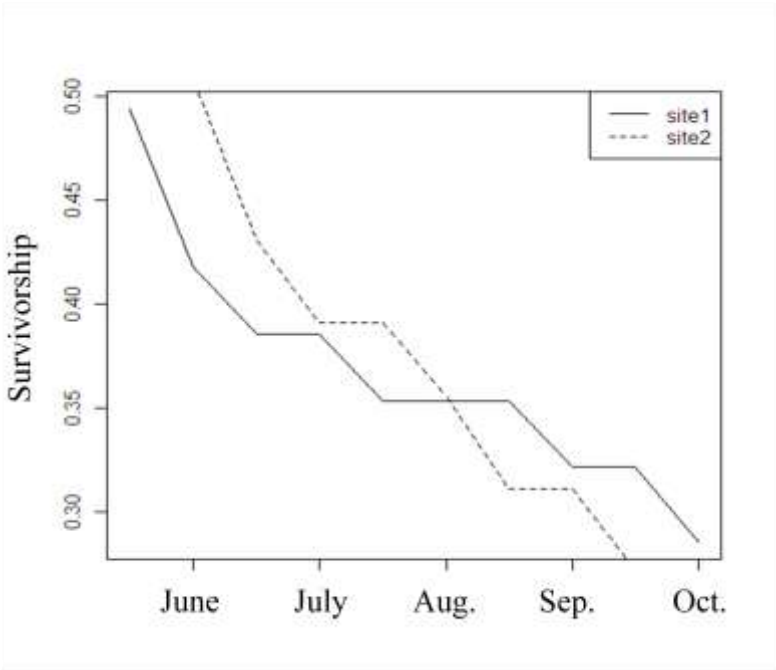

Figure 2. Kaplan-Meier survival estimate at site 1 and site 2 during the 150 day breeding-brooding season. Each age class represents 15 days beginning May 1st during 2008, Bennett Hills, Idaho, USA

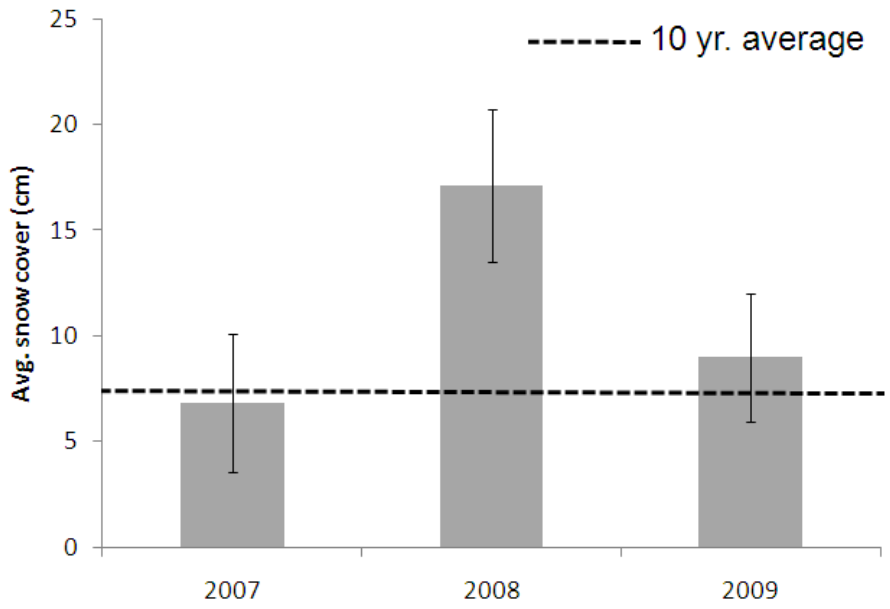

Figure 3. The average snow cover in the Bennett Hills, Idaho, during 2007-09. Error bars = 1 SE.

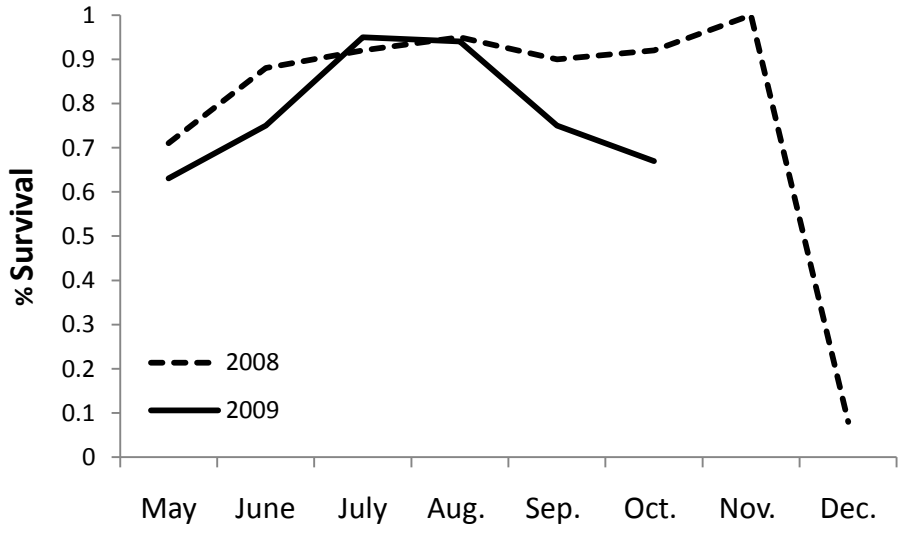

Figure 4. The percent survival of mountain quail by month for 2008 in the Bennett Hills, Idaho, USA. The decrease in sample size from September to October is a function of failed transmitter batteries on birds that we were unable to replace in the autumn, and probably is not a result of mortality.

Chapter 2: Reproductive pattern and nest site characteristics of mountain quail translocated to Southern Idaho, USA.

Mountain quail (*Oreortyx pictus*) occupy areas with significant differences in topographic and edaphic conditions throughout their range (Brennan 1987, Pope 2002). For example, mountain quail occupy the arid Mojave Desert and the moist west-slope of the Cascade Mountains. Regional environmental differences likely influence mountain quail populations (Gutierrez and Delehanty 1999). To understand population dynamics it is helpful to know how survival and reproduction vary across the lifespan of an individual (Cole 1954). Identifying factors that impact survival and reproduction can be especially useful as an aid to wildlife managers when seeking to reestablish self-sustaining populations or protecting small populations – typical circumstances in avian restoration attempts. Mountain quail restoration efforts are underway in central and eastern Oregon, southern Idaho, and northern Nevada in areas experiencing reduction in distribution and abundance (pers. comm. Jenny Barnett, Espinosa 2008, Abel 2008). Translocation is the main restoration strategy used.

Predator abundance, spring precipitation, winter precipitation, habitat quality, raptor predation and movement rate previously have been identified as important factors influencing mountain quail survival (Herman et al. 2002, Pope 2002, Nelson 2007, Stephenson 2008). Few of these factors affecting survival rates have been consistent across studies (Herman et al. 2003, Reese et al. 1999, Nelson 2007,Stephenson 2008, chapter 1, above). In some cases, conclusions are conflicting. For example, Herman et al. (2003) concluded that high winter precipitation positively influenced survival; however, we observed a negative correlation between snow accumulation and survival.

Mountain quail are consistent in many factors influencing reproduction such as reproductive strategy, parental care, nest success, nest site characteristics, and clutch size (Gutierrez 1980, Pope and Crawford 2001, Reese et al. 2005, Nelson 2007, Stephenson 2008). Mountain quail males and females essentially are monomorphic although the sexes can exhibit subtle dimorphic in plumage and mass (Pine 1981, Brennan and Block 1985, Delehanty et al. 1995) and socially monogamous (Pope 2002). Male mountain quail can contribute prominently in all phases of reproduction, including uniparental incubation and shared or uniparental brood rearing (Delehanty 19995, Gutierrez and Delehanty 1999, Pope and Crawford 2001, Beck et al. 2005, Nelson 2007, Stephenson 2008). Specifically, it is common for male mountain quail to incubate a separate clutch (Heekin 1993, Delehanty 1995) and rear a brood concurrently with a female (Pope and Crawford 2001). Most research indicates male and female mountain quail share nearly equal effort in incubation and brood rearing with little variation among studies (Pope 2002, Beck et al. 2005, Nelson 2007, Stephenson 2008) although Reese et al. (2005) observed a preponderance of female incubation in a 2-year study.

Following a life history review of mountain quail (Gutierrez and Delehanty 1999), there have been numerous studies describing nest site characteristics. Several studies have reported mountain quail nests in mixed evergreen forests with a shrub understory (Pope 2002, Reese et al. 2005, Nelson 2007, Stephenson 2008). Canopy cover and perimeter shrub height appear to be important factors in nest site selection (Pope 2002, Nelson 2007) and influence nest success (Stephenson 2008). Mountain quail nests usually are located underneath some form of overhead

cover such as rock outcrops, living or dead vegetation including – trees, hollow logs, bunchgrasses, and herbaceous or shrubby vegetation.

The consistency in parameters associated with mountain quail nest site selection and nest success reflect a successful reproductive strategy that may center on a high rate of nest success. Mountain quail nest success ranges from 62%-100% with most studies documenting nest success between 75-85 percent (Pope 2001, Beck et al. 2005, Nelson 2007, Stephenson 2008). For many birds, nest failure due to predation is the primary cause of reproductive failure (Ricklefs 1969, Martin 1995, Fontaine and Martin 2006). The direct and indirect effects of nest predation on behavior can be complex (Fontaine and Martin 2006), however, predation of mountain quail nests is low compared to other US quail (Rollins and Carrol 2001, Rader et al. 2007) and gallinaceous birds (Hewett et al. 2001, Popham and Gutierrez 2003, Tirpak et al. 2006, Coates et al. 2008)

A comprehensive review of mountain quail life history (Gutierrez and Delehanty 1999) indicated that only sparse data were available for clutch size. Clutch size can range from 6-14 eggs (Miller 1950). Recent studies documenting clutch size in native and translocated populations of mountain quail in the last decade have reported a trend towards smaller clutch size for translocated mountain quail. Studies of translocated quail reported less than 10 eggs per nest (Nelson 2007 and Stephenson 2008) versus clutch sizes of 11-12 eggs in established populations (Pope and Crawford 2001, Beck et al. 2005). Furthermore, most studies have found that clutch size among nests incubated by males is at least one egg greater than for nests incubated by females (Pope 2002, Beck et al. 2005, Nelson 2007, Stephenson 2008).

Our objective was to augment a small and probably vulnerable population of mountain quail founded 2 years previously by translocation of mountain quail into the Bennett Hills (releases in 2006 and 2007) through two subsequent releases of mountain quail (in 2008 and 2009) and to assess population dynamics following the 2008 and 2009 releases. We sought to measure productivity of mountain quail in a spatially and temporally heterogeneous environment and examine any differences in reproductive effort between birds released at sites that differed in surface water availability and habitat. We measured nest site characteristics, incubation by males and females, nest and brood success, clutch size and observed apparent reproductive strategy to quantify productivity.

METHODS

We monitored each radio-marked quail creating a behavioral record for each quail consisting of general movement patterns, breeding behavior, and associations with other mountain quail. Mountain quail were closely monitored during the breeding season to document breeding activity and reproductive success or failure.

On average, mountain quail were located every 3-4 days for 150 days following release. Mountain quail are secretive and sensitive to disturbance, thus, we tried to minimize disturbing mountain quail with our monitoring activities. Of the 2 locations per week, 1 was a specific location accompanied with a visual resighting of the quail. The relocation consisted of a 2 or 3 line-of-sight triangulation using unidirectional radio signal strength from a distance from the

quail of approximately 50 m. When possible, sightings were made with binoculars or a spotting scope. Visual sightings provided irregular information about behavior such as whether or not the quail was part of a group.

Transmitters were equipped with mortality signals and helped minimize disturbance and at the same time monitor their status. In the first and second months of the translocation, aerial surveys were conducted bi-weekly and once a month thereafter for the remainder of the breeding-brooding season.

Nest and Brood Success

Incubating mountain quail were discovered both through visual sightings with binoculars and radio-telemetry. Once a mountain quail was located on a nest, the bird was monitored to determine point of incubation, on the second or third week of incubation, the bird was prompted off the nest and we recorded clutch size (Nelson 2007). Generally, incubating quail remained within a few meters during egg count, although some flushed a short distance away from the nest. We considered nests to be successful if at least one egg hatched (Butler and Rotella 1998, Beck et. al 2005). Nests were considered unsuccessful if we were unable to find evidence that one or more eggs hatched. To quantify and compare nest attempts between year and site, we reported the proportion of nesting birds that survived to at least 1 June (Stephenson 2008).

Broods were considered successful if at least one chick survived to 28 days post-hatch. Because adult quail locations were monitored an average of every 4 days, we were able to confidently estimate within 2 days of when eggs hatched. When circumstances allowed, we located broods using binoculars or spotting scope once weekly until 28 days post-hatch. We intentionally flushed broods at 28 days post-hatch to estimate the number of offspring that survived.

Nest and Random Site Habitat Analysis

We analyzed nest site characteristics at both microhabitat and macrohabitat scales. Our nest habitat surveys were consistent with the protocol established by Pope (2002) and adapted by other mountain quail studies (Reese et al. 2005, Nelson 2007, Stephenson 2008). We conducted microhabitat surveys at nest sites in July and August, after mountain quail had successfully hatched their young and left the area. We also conducted identical microhabitat surveys at a random site for each nest site surveyed. The random site was determined by generating a random compass bearing and random distance between 25-200 m from the corresponding nest site but within the same cover type. We measured perimeter shrub height, or visual obstruction, using a 3-m tall and 3-cm wide Robel pole with alternating decimeters marked black and white (Robel et al. 1970). The Robel pole was placed in the nest bowl and viewed 4-m and 8-m from the nest in each of the cardinal directions at a height of 1 meter. We measured understory using a 20 x 50 cm Daubenmire frame centered on the nest bowl and at 2-m and 4-m in each of the cardinal directions for a total of 9 measurements per site (Daubenmire 1959). Understory measurements included the proportion of shrubs, grass, forbs, bare ground, litter, and rock.

We estimated overstory, or the percent canopy coverage, with a convex spherical densiometer on top of the nest bowl and on the ground 8-m from the nest in each of the cardinal directions

(Lemmon 1957). We recorded the species and height of the tallest and shortest shrub within 1 meter of the nest center and 4-m from the nest in each of the cardinal directions. We used the average values for the 5 measurements of canopy coverage and tallest/shortest shrub for statistical comparisons between the true vs. random and successful vs. unsuccessful nest sites. We used known locations of seeps or springs and National Geographic TOPO Version 4.2.8 mapping software to determine distance to surface water. We made one measurement of slope (degrees) 10-m below each nest site using a clinometer.

Statistical Analysis

In general, data for habitat variables had sufficiently equal variances but was not normally distributed. We used a Wilcoxon-Mann-Whitney rank sum test in the program R (R-Statistics 2004) to test for differences in habitat variables between nest sites and random sites and between successful nests and unsuccessful nests. In cases where assumptions of normal distribution and homoscedasticity were violated, we used the non-parametric Brunner-Munzel test in program R for comparisons. To address sensitivity to non-normality we used the Fligner-Killeen test for heteroscedasticity (Aho in review). We were unable to compare habitat variables statistically by sex of the incubating adult due to insufficient sample sizes.

We analyzed differences in nest site characteristics between nest and random sites and between successful and unsuccessful nests using logistic regression in the program R (R Statistics 2004). Pearson's product moment correlation was used to test for multicollinearity among explanatory variables. We used the Fisher exact test to compare the frequency of nesting birds between sites, year, and sex.

When birds abandon a nest or a nest is depredated, the average hatch number reported can be ambiguous because it is unclear if depredated eggs or abandoned eggs are included in the average hatch number. We did not include depredated eggs in the average hatch number for nests that were depredated. We included abandoned eggs in the average hatch number for nests that were abandoned. For example, if an incubating bird abandoned the nest, the number of eggs hatched (zero) *was* included in the average hatch number for each category. Due to small sample sizes we did not statistically compare the influence of site, year, or sex on clutch size or hatch number.

RESULTS

Proportion of Nesting Mountain Quail Surviving Until June 1st

Seventeen of 46 (37%) mountain quail monitored after June 1 nested during the 2-year study (Table 1). Ten of 26 (38%) mountain quail nested during 2008, similar to 7 of 20 (35%) mountain quail during 2009 (odds ratio = 1.157, $p = 1.000$; Table 2). Across years, nesting frequency did not differ among monitored mountain quail occupying Little Canyon creek where 10 of 24 (42%) nested compared with 7 of 22 (32%) quail occupying Dempsey creek that nested (odds ratio = 1.516, $p = 0.552$; Table 2). The proportion of nesting birds was not significantly different at Little Canyon or Dempsey release sites between years ($p > 0.05$; Table 2). Among radio-marked mountain quail known to be alive during the nesting period 4 of 25 (16.0%) males

incubated nests, significantly fewer than 13 of 21 (62%) monitored females (odds ratio = 0.124, $p = 0.002$; Table 3).

Nest and Brood Success

In 2008, we detected 10 nests incubated by birds with radio-transmitters, 2 of which were incubated by males, both at Little Canyon (Table 4). Six of the 10 nests were successful. Among the 4 unsuccessful nests, 2 females abandoned incubation, 1 female was found dead away from the nest, 1 female incubated for approximately 41 days without hatch. Both nests incubated by males were successful. Average clutch size was 10.2 eggs (SE = 0.7, n = 10; Table 4). Among successful nests (n = 6), 59 of 60 (98%) eggs hatched.

In 2009, we detected 7 nests incubated by radio-marked quail (Table 4). Three nests were within the Little Canyon release area and 4 at Dempsey. Four of the 7 (57%) nests were successful. Two of the 3 unsuccessful nests were incubated by males and were depredated, but both birds survived. The other unsuccessful nest was incubated by a female and she was found dead away from the nest after 2 weeks of incubation. Her nest was not depredated and eggs were partially developed upon inspection. We obtained clutch size data for 2 nests incubated by males in 2009 and for 3 incubated by females. Clutch size averaged 8.2 eggs (SE = 0.6; Table 4). Thirty-three of 35 (94%) eggs hatched among the 4 successful nests in 2009.

Overall nest success was 59% (10 of 17) and was numerically higher at Little Canyon (7 of 10) than Dempsey (3 of 7) but the difference was not statistically significant (odds ratio = 2.897, $p = 0.350$). Nest success was similar between years (odds ratio = 1.117, $p = 1.000$) and sex (odds ratio = 1.593, $p = 1.000$; Table 4). For the 2-year study period, 2 of 17 (12%) nests were depredated, 2 of 17 (12%) nests were abandoned, 1 of 17 nests was infertile (6%), and 2 of 17 (12%) incubating birds were predated when absent from the nest. Overall mean clutch size was 9.5 ± 0.6 eggs (n = 15, range: 6-14) and was similar between release sites and sex (Table 4). The hatch date ranged from 28 June to 22 July (mean: 11 July).

Four of 6 successful nests in 2008 resulted in successful broods, two at each release site. We did not observe any broods with unmarked birds in 2008. In 2009, at least 3 of the 4 successful nests produced successful broods. One of the 4 adults that incubated a successful nest was censored (the radio-transmitter failed) during incubation. We observed the quail on the nest and could see the radio-transmitter but were unable to receive a signal with telemetry equipment. We observed 2 broods in the area of the censored quail's nest 28-days following nest hatching. One of the adults accompanying those broods bore a radio-transmitter that we could observe visually but from which we were unable to detect a signal. In addition, in Little Canyon, we observed a minimum of 4 broods tended by adults that were not radio-marked and 1 brood at Dempsey tended by an adult that was not radio-marked. Cumulatively, there were at least 8 successful broods in the study area in 2009 and 51 chicks that survived to 28 days.

We also observed intraspecific brood amalgamation of mountain quail as well as interspecific brood amalgamation of mountain quail and California quail in 2009. Three adult mountain quail with 23 chicks were observed in one covey at a spring. None of the adults bore radio-transmitters. At another location we observed an adult mountain quail with 8 chicks and an adult California quail with 2 chicks in the same covey at a water source. The California quail chicks

were much smaller and lacked the broad dorsal chestnut stripe characteristic of mountain quail chicks.

Brood success was 7 of 9 (78%) overall and female brood success was 7 of 7 (100%) while male brood success was 0 of 2 (0%; Table 4). For the 2 study years combined, the average brood number at 28 days post-hatch was 4.9 ± 0.9 (2008 mean = 5.0 ± 1.5, 2009 mean = 4.8 ± 1.3).

Nest and Random Site Habitat Analysis

Comparing nest plots and random plots we found the top model consisted of visual obstruction (VO) and percent bare ground (B, Table 5). Percent shrub cover was significantly greater at nest sites (45.8 ± 3.9) than random sites (32.8 ± 5.0) during the 2-year study (W = 90, df = 32, p = 0.024; Table 6) but was not retained in competitive models. Perimeter shrub height at nest sites (96.1 ± 7.2) was significantly higher than random sites (63.4 ± 7.5, W = 60, df = 32, p = 0.002; Table 6). Percent bare ground at nest sites (7.6 ± 1.9) was significantly lower than random sites (19.0 ± 3.0, W = 61, df = 32, p = 0.002; Table 6). Finally, slope at nest sites (19.4 ± 2.0) was higher than random sites (15.2 ± 1.5; W = 95.5, df = 32, p = 0.047; Table 5). No other habitat variables were significantly different between nest and random sites. Almost half of nest and random plots (16 of 36) had 0% canopy coverage at 5 locations within the plot.

Nest site slope and percent bare ground were the most important logistic regression reproductive parameters differentiating successful and unsuccessful nests. They were the only variables consistently in the most informative model (Table 6). Slope at successful nests (22.4 degrees ± 2.0) was significantly greater than unsuccessful nests (15.0 degrees ± 3.4, W = 16.5, df = 15, p = 0.031, Table 7). No other habitat variables were significantly different between successful and unsuccessful nests. Percent bare ground, which was an important parameter in logistic regression models, was higher at successful nests (10.6%) than unsuccessful nests (3.3%, W = 20, df = 15, p = 0.120, Table 7). Distance to 2-track roads at successful nests (1531.4 m ± 428.1) was farther than unsuccessful nests (755 m ± 222.2) but it was marginally significant (p = 0.089, Table 7). Distance to surface water at successful nests (305.4 m ± 97.9) was closer than unsuccessful nests (580.0 m ± 188.2) but was also marginally significant (W = 20.5, df = 15, p = 0.086, Table 7).

Mating Strategy

Nine of 17 (53%) nesting attempts involved a male and/or female associating with 2 or more mountain quail of the opposite sex. Five of 17 (29%) nesting attempts we were not able to monitor closely enough to observe associations during nest attempts. Mountain quail were considered a pair if they were observed together for 2 weeks or more without associating with other mountain quail. Three of 17 (18%) nesting attempts only involved associations between mated pairs. We identified 2 mated pairs where the male and female incubated clutches simultaneously during the 2-year study. Furthermore, we observed 5 females paired with multiple males for successive weeks during the nesting season prior to nesting and 6 males paired with multiple females for successive weeks during the nesting season.

We observed several interesting social associations. In instance 1 at Little Canyon, a male and a female associated exclusively prior to the female laying a clutch of eggs which the male incubated. The day following incubation initiation by the male, the female left the area and began associating with a radio-marked male. During the 2-week association, the female laid a second clutch of 6 eggs. We observed the female at this second nest for only one day and no other quail incubated the clutch. The following week the female was associating with a 3rd radio-marked male and during that week a seventh egg was added to the clutch. She abandoned the 2nd nest she had laid but remained in the area. These interactions have additional complexities, however.

Instance 2 – another female at Little Canyon was initially paired with the 3rd radio-marked male from instance 1 for 3 weeks prior to his association with the female from instance 1. She subsequently left the male and associated with a second radio-marked male for 10 days until she began incubating a nest. Instance 3 – a third female at Little Canyon was paired with the second male (from Instance 2) for 3 weeks and then she associated with a different radio-marked male for 2 weeks; afterward, switching associations between these 2 males for 2 weeks until incubating a nest. The majority of this polygynandrous group of 6-8 individuals spent the brooding season together after nesting. One brood appeared to be jointly raised by the group.

We observed apparent polygyny in 2008. A male and 2 females remained with each other for 3 weeks before 1 of the females began incubating a clutch of eggs. She abandoned the nest shortly thereafter and the trio was together for 3 more weeks when the other female of the trio was found incubating a clutch of eggs in the same area. She successfully incubated the clutch and joined the male and female that abandoned the nest to raise her brood. We found the eggs in the abandoned nest to be fertile and believe the male in the trio fertilized both clutches, but he was never observed incubating a nest.

We observed apparent polyandry in 2009. A female paired with a male for 2 weeks until the male left the area. She then paired with an unmarked mountain quail of unknown sex for 3 weeks; after which, she paired with a third radio-marked male for 2 weeks before incubating a clutch. Near the Dempsey release site we observed a female paired with a mountain quail of unknown sex for 3 weeks and appeared to mate-switch between the radio-marked male and an unmarked quail for the next 6 weeks prior to incubating a nest. Also near the Dempsey release site, a female was paired with a male for 6 weeks until the male began incubating a nest. She remained in the area and was found with 2 males on one occasion before she too began incubating a nest. This was the only observed simultaneous multiple clutch in 2009.

DISCUSSION

We observed many aspects of reproduction that were different from the pattern described in recent mountain quail studies. Over half of nesting attempts appeared to involve a polygynous or polyandrous mating strategy. If a simultaneous multiple clutch is a function of a predominant monogamous mating system in mountain quail (Pope and Crawford 2001), then one would expect, in a population exhibiting polygamy, to observe few simultaneous multiple clutches. We observed one simultaneous multiple clutch per year. Males incubated only 4 of 17 (24%) nest attempts, results similar to those of Reese et al. (2005) and unlike most studies which have found

the role of incubation and brood rearing to be nearly equal between sexes (Pope and Crawford 2001, Beck et al. 2005, Nelson 2007, Stephenson 2008). Nonetheless, reproductive strategy was flexible enough to achieve reproduction despite low population size. The factors that influence reproductive plasticity are unclear. Reese et al. (2005) trapped and studied local populations of mountain quail in west-central Idaho and few similarities exist between his study and ours that may have lead to decreased male participation in reproduction.

The preponderance of female incubation and brood rearing we observed, in addition to what was documented by Reese et al. (2005), is evidence to support the hypothesis by Ketterson and Nolan (1994) that socially monogamous care-giving males may be plastic in their parental and mating effort according to hormonal responses from changing environmental cues. Mountain quail were released later in the year relative to other studies (1 May in 2008 and 24 April in 2009) and may have experienced different environmental cues as a result. Furthermore, various studies of passerines support the hypothesis that differences in paternal care reflect differences in circulating hormones (Van Roo et al. 2003, O'Neal et al. 2008); specifically, increased testosterone or decreased prolactin inhibits male incubation (Goldsmith 1992, Van Roo et al. 2003, Schwagmeyer et al. 2005, Pinxten et al. 2007, Schwagmeyer et al. 2008). The factors driving the hormone-behavior interaction can be complex and the interaction an important component of proximate mechanisms by which birds breed successfully in a capricious environment (Wingfield and Hunt 2002). However, this hypothesis has not been directly tested in a mountain quail population.

A late spring release may not provide mountain quail with enough time to establish sufficient pair bonds in the wild prior to the nesting season despite an effort to control for sociality in captivity. Birds are capable of assessing nest predation risk which influences reproductive strategy (Fontaine and Martin 2006); hence, male mountain quail may be assessing certainty of paternity based on the strength of the pair bond in their decision to incubate. Male incubation comes with a cost: lost mating opportunities reduce the variability of reproductive success (Ketterson and Nolan 1994), lost feeding opportunities, and predation risk. The cost of incubation coupled with a weak pair bond may prevent a male mountain quail from incubating. Lastly, high mortality rates in the first few weeks following release (Nelson 2007, Stephenson 2008) that we observed likely disrupted pair bonds. Factors affecting the decision of male mountain quail to actively participate in nesting and brooding are unknown. The lack of male parental care and the apparent polygamous mating system observed warrant further investigation of phenotypic plasticity in mountain quail reproductive strategy.

We observed substantial behavioral flexibility in reproduction in a small population of mountain quail newly founded through translocation. This is an important finding because simultaneous multiple clutches have been described as the main reproductive strategy for mountain quail (Pope and Crawford 2001, Beck et al. 2005, Stephenson 2008) and Beck et al. (2005) this was an effective adaptation for living in an environment with a short breeding season. In some mountain quail populations males may forego incubation and brood rearing and simultaneous multiple clutches become rare in the local population. The lack of male parental care could have detrimental effects on reproductive rates and influence local population dynamics. Male-biased survival in a population with few males participating in parental care is evidence that incubation and brood rearing may be a significant mortality risk. Further research is needed to determine

when simultaneous multiple clutches are the main reproductive strategy, how plastic mountain quail are in their mating strategy and how mating strategy affects reproduction, survival and, ultimately, population dynamics.

Few, if any, researchers have reported observing a seemingly polygamous mating strategy in mountain quail or such a low degree of incubation and brood rearing by male mountain quail. We observed multiple individuals that appeared to be participating in a complex mating system or some form of social polygamy during the 2-year study. We never observed territorial behavior. We were not able to conduct a genetic analysis of offspring to quantify the degree of polygamy within this translocated population.

Nest success was low (59%) compared to other studies (range: 62-83%). Nonetheless, the proportion of birds surviving to 1 June that nested (37%) was higher than a translocation study (29%) in eastern Washington and western Idaho (Stephenson 2008). The main reasons for nest failure were: predation, abandonment, infertility, or the adult was depredated when absent from the nest. Pope (2002) reported considerable nest predation (up to 38%) but Beck et al. (2005) reported 12% and Stephenson (2008) found no nest predation. Nelson (2007) did not report nest predation rates but found 78% nest success, hence, nest predation could not have been more than 22% and was probably lower. Nest predation or failure does not appear to limit productivity in many mountain quail populations.

In our study, nest success was influenced by nest attentiveness and abandonment as much or more than predation. Little is known about factors influencing nest attentiveness. Pope (2002) postulated mountain quail translocated to the Idaho/Oregon border of USA that were absent from the nest rarely, only for short time periods, and during early morning hours was a strategy to reduce predation risk. In our study, nest success was higher at the mesic site (70%) than the xeric site (43%), however, sample sizes were low. Pope (2002) found that water was an important variable when comparing nests and random sites and that water may be more important for nest sites selected in semi-arid landscapes. Adults that were depredated during incubation recess appeared to be nesting far from surface water (>100m). In xeric environments nest attentiveness may be an important factor influencing nest success. Further research is needed to determine factors influencing nest attentiveness which in turn may influence nest predation and adult predation when absent from the nest, ultimately influencing nest success.

Reese et al. (2005) compared nest and brood habitat characteristics but there is a paucity of information regarding brood rearing and success. Stephenson (2008) defined a successful brood as at least one fledgling surviving to 28 days and documented 61% brood success over the course of their study with 5.8 ± 0.7 chicks per successful brood. We found brood success to be higher (78%) and chicks per successful brood to be lower (4.9 ± 0.9) than mountain quail translocated to eastern Washington and western Idaho (Stephenson 2008). Brood success may limit productivity in mountain quail populations, but little is known about factors affecting chick survival or vital rates. Evidence suggests that water is an important factor influencing chick survival (Edminster 1954, Johnsgard 1973, Brennan 1987) but this hypothesis has not been tested directly. In 2009, we observed intraspecies brood amalgamation of unmarked mountain quail with California quail in mid-July within 2 meters of the only known surface water in the drainage they were occupying. Thus, the importance of protecting seeps, springs, and streams

from grazing is crucial during late summer when birds are congregated around water sources. This overlap in habitat is evidence to support a hypothesis by Gutierrez (1980) that drinking water may lead to intraspecies habitat overlap.

We recognize that the average number of chicks per brood that survived to 28 days was a conservative estimate. On 2 different occasions we assumed that broods were unsuccessful when broods were consistently in dense vegetation and were probably too cryptic to observe. First, while trapping at night in autumn 2008 we discovered chicks with marked birds that we incorrectly identified as having unsuccessful broods a month earlier. Second, in 2009, we discovered a winter mortality of a female we had trapped in autumn and identified as unsuccessfully rearing a brood, buried under snow with a hatch year bird. The possibility of post-hatching brood amalgamation coupled with the difficulty of observing chicks in the wild could lead to bias in chick survival estimates (Faircloth et al. 2005) for mountain quail.

The mean hatch date (July 11) was later than other studies (range: 30 June – 6 July) with few simultaneous multiple clutches that one would expect to expedite mean hatch date (Pope and Crawford 2001, Beck et al. 2005, Stephenson 2008). The average clutch size (9.5 ± 0.6) was similar to other translocations and consistent with the trend of smaller clutch sizes compared to native populations. Smaller clutch size relative to native populations could be a function of translocation stress (Stephenson 2008). Chronic stress can disrupt the reproductive hormone axis and reproductive behavior (Berga 2008, Dickens et al. in review) which may explain smaller clutch size in translocated populations compared to native populations. Alternatively, poor body condition because of an inability to find resources efficiently in a novel environment could explain smaller clutch size in translocated populations. Regardless, further investigation is needed to determine if translocation affects clutch size in mountain quail.

The Bennett Hills study area is further east in longitude, higher in elevation, varies more in elevation, the maximum and minimum temperatures per year are more extreme, and they receive less annual precipitation but more snow fall than most recent mountain quail study areas (Pope and Crawford 2001, Reese et al. 2005, Stephenson 2008, http://www.wcc.nrcs.usda.gov). The heterogeneous environment of the Bennett Hills is conducive to reproduction for translocated mountain quail. We observed a high proportion of nesting mountain quail and high brood success despite a seeming shift in mating and reproductive strategies. The mesic Little Canyon release site numerically had more nests and higher nest success than the xeric Dempsey Meadows release site. The Dempsey Meadows release site had higher brood success and hatch number than Little Canyon. However, none of these differences in productivity were significant. Productivity was similar between release sites despite topographic and edaphic differences.

Canopy coverage has been identified as an important parameter describing nest site selection (Pope 2002, Nelson 2007, Stephenson 2008); purportedly for the benefit of thermal and hiding cover (Pope 2002). We did not find percent canopy coverage to be an important variable that described nest sites or differentiated between successful and unsuccessful nests. This may have been a function of limited canopy coverage available since 56% of nest and random sites analyzed did not have any canopy coverage. Nonetheless, mountain quail nest success was similar to other studies and thermal protection could only have had a direct effect on nest viability in one instance where eggs were infertile. We found that perimeter shrub height and

visual obstruction to be an important microhabitat variable distinguishing nest and random sites as have several previous studies (Pope 2002, Nelson 2007, Stephenson 2008). Clearly, mountain quail select some form of dense cover for nesting. Also, percent bare ground was significantly less at nest sites compared with random sites, which is consistent with Nelson (2007). Lastly, slope was significantly greater at successful nests compared to unsuccessful nests and was an important parameter in the top models differentiating successful and unsuccessful nests. Gutierrez (1980) noted slope as an important habitat characteristic influencing mountain quail reproduction. Nests located on steep slopes probably receive fewer visitations by terrestrial predators decreasing nest predation and nest abandonment.

MANAGEMENT IMPLICATIONS

Among mountain quail translocated to a high elevation site within the interior portion of their historic range, we observed high proportion of nesting quail, high brood success and nest success rates similar to other studies. These results indicate potential for restoring mountain quail close to former distributional boundaries in the interior American west. Although mountain quail made behavioral adjustments to reproduce despite a short breeding window, few males participated in parental care phases limiting the per capita rate of population increase during the establishment phase. This effect argues for releases of larger numbers of individuals. Where possible, earlier spring releases of mountain quail also might increase male participation in incubation and brood rearing.

Managers should be careful not to preclude conservation approaches because of perceived behavioral limitations of a species. At any given time, mountain quail may express only a subset of a broader behavioral repertoire. Behavioral plasticity may aid translocation success. For example, canopy cover has consistently been found to be an important variable in nest site selection. This could provide a false pretense that mountain quail will not nest or successfully nest without high canopy coverage. Many areas in the Bennett Hills are void of classic mountain quail canopy coverage described in the literature, however, mountain quail nested successfully.

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Table 1. Proportion of radio-marked mountain quail nesting that survived to at least June 1 in the Bennett Hills, Idaho in 2008 and 2009. Sample size (n) represents the number of mountain quail monitored.

	Nested	n	%
Total	17	46	37
Little Canyon	10	24	42
Dempsey	7	22	32
Males	4	25	16
Females	13	21	62
2008	10	26	38
Little Canyon	7	14	50
Dempsey	3	12	25
2009	7	20	35
Little Canyon	3	10	30
Dempsey	4	10	40

Table 2. Results of a Fisher exact test comparing the frequency of nesting birds in the Bennett Hills, Idaho in 2008 and 2009

Comparisons	P-value
Little Canyon vs. Dempsey	0.552
Males vs. Females	0.002
2008 vs. 2009	1.000
Little Canyon vs. Dempsey 2008	0.248
Little Canyon vs. Dempsey 2009	1.000
Little Canyon 2008 vs. 2009	0.421
Dempsey 2008 vs. 2009	0.652

Table 3. The number of radio-marked males and females nesting during 2008 and 2009 immediately following release to the Bennett Hills, Idaho USA.

	Little Canyon		Dempsey		
	Male (11)	Female (13)	Male (13)	Female (9)	Total
2008	2	5	0	3	10
2009	0	3	2	2	7

Table 4. Reproduction during the breeding season by radio-marked mountain quail translocated into the Bennett Hills, Idaho immediately following release during spring 2008 and 2009.

		Clutch size		Hatch number		Nest success		Brood success	
	nests	mean	SE	mean	SE	successful nests	%	successful broods	%
Overall	17	9.5	0.6	7.1	1.1	10	59	8	80
2008	10	10.2	0.7	6.6	1.7	6	60	4	67
2009	7	8.2	0.6	8.3	0.6	4	57	4	100
LC	10	9.4	0.7	6.7	1.3	7	70	5	71
D	7	9.8	1.0	8.0	2.4	3	43	3	100
Male	4	9.5	1.5	9.5	1.5	2	50	0	0
Female	13	9.5	0.6	6.6	1.3	8	62	8	100

LC = Little Canyon release site, D = Dempsey Canyon release site

Table 5. Ranking of best approximating models using logistic regression and Akaike's Information Criterion with correction term for small sample sizes (AIC_c) for nest site and random site characteristics of translocated mountain quail to the Bennett Hills, Idaho, in 2008 and 2009.

Model	K	AIC_c	ΔAIC_c	w_i
B + VO	3	41.63	0.00	0.20
B	2	42.47	0.84	0.13
VO + F + B	4	43.15	1.52	0.09
VO	2	43.21	1.58	0.09
VO + SL + B	4	43.28	1.65	0.09
VO + TS + B	4	43.52	1.89	0.08
VO + L + B	4	44.12	2.49	0.06
F + B + TS + VO	5	44.39	2.76	0.05
VO + G + B	4	44.39	2.76	0.05
F + B + SL + VO	5	44.86	3.23	0.04
F + B + TS + SS + VO	6	45.44	3.81	0.03
SL + B + TS + VO	5	45.47	3.84	0.03
CC + VO	3	45.47	3.84	0.03
S	2	47.84	6.21	0.01
F + L + B + TS + SS + VO	7	48.04	6.41	0.01
F + B + TS + SS +SL	6	48.63	7.00	0.01
F + B + TS + SS + VO + SL + CC	8	49.80	8.17	0.00
CC	2	51.48	9.85	0.00
DW	2	51.93	10.30	0.00
Global	13	73.85	32.22	0.00

B=Bare ground (%), F=Forb (%), L=Litter (%), G=Grass (%), CC = Canopy cover (%), S=Shrub(%), VO=Visual obstruction (cm), TS=Tall shrub (cm), SS=Short shrub (cm), SL=Slope (degrees), DW=Distance to water (m)

Table 6. Ranking of best approximating models using logistic regression and Akaike's Information Criterion with correction term for small sample sizes (AIC_c) for successful nest and unsuccessful nest site characteristics of translocated mountain quail to the Bennett Hills, Idaho, in 2008 and 2009.

Model	K	AIC_c	ΔAIC_c	w_i
Bare + Slope	3	21.37	0.00	0.38
Bare	2	24.52	3.15	0.08
Bare + Slope + Tall shrub	4	24.80	3.43	0.07
Bare + Slope + Shrub	4	24.81	3.44	0.07
Slope	2	24.84	3.47	0.07
Bare + Slope + Forbs	4	25.19	3.82	0.06
Bare + VO + Slope	4	25.46	4.09	0.05
Bare + DW	3	25.59	4.22	0.05
Litter + Bare	3	26.04	4.67	0.04
Slope + Forbs	3	26.34	4.97	0.03
Bare + Grass	3	26.57	5.20	0.03
DW	2	26.82	5.45	0.03
Grass	2	27.04	5.67	0.02
VO + Slope	3	27.36	5.99	0.02
Shrub + Slope + CC	4	27.68	6.31	0.02

CC = Canopy cover (%), VO=Visual obstruction (cm), DW=Distance to water (m), Shrub=% cover, Bare=% bare ground, Grass=% grass cover, Forbs=% forb cover

Table 7. Habitat variable means and standard errors (SE) for nest and random sites in the Bennett Hills, Idaho in 2008 and 2009

	Nest Sites (n = 17)		Random Sites (n = 17)		
	Mean	SE	Mean	SE	*p*-value
Shrub cover (%)	45.8	3.9	32.8	5.0	0.024
Grass cover (%)	5.6	2.3	6.3	1.9	0.264
Forb cover (%)	4.2	2.1	4.0	1.5	0.458
Litter (%)	12.2	2.6	13.8	2.3	0.235
Bare ground (%)	7.6	1.9	19.0	3.0	0.002
Rock (%)	12.7	3.4	10.1	3.3	0.239
Canopy Cover (%)	11.4	4.1	7.5	4.1	0.120
Height of tallest shrub (cm)	112.7	9.4	104.7	14.1	0.151
Height of shortest shrub (cm)	50.0	6.1	39.6	5.6	0.087
Perimeter shrub height (cm)	96.1	7.2	63.4	7.5	0.002
Distance to Water (m)	418.5	98.9	434.2	100.7	0.500
Slope (degrees)	19.4	2.0	15.2	1.5	0.055

Table 8. Habitat variable means and standard errors (SE) for successful and unsuccessful nests in the Bennett Hills, Idaho in 2008 and 2009.

	Successful (n = 10)		Unsuccessful (n = 7)		
	Mean	SE	Mean	SE	*p* –value
Shrub cover (%)	43.1	6.0	49.8	4.5	0.212
Grass cover (%)	3.1	1.5	9.1	5.1	0.227
Forb cover (%)	5.6	3.4	2.1	1.6	0.259
Litter (%)	10.8	2.8	14.3	5.0	0.279
Bare ground (%)	10.6	2.9	3.3	1.3	0.120
Rock (%)	13.9	4.8	11.1	5.1	0.597
Canopy Cover (%)	11.0	3.6	10.9	9.0	0.129
Height of tallest shrub (cm)	111.4	14.5	114.6	11.2	0.437
Height of shortest shrub (cm)	51.2	9.8	48.3	6.2	0.412
Perimeter shrub height (cm)	92.3	11.6	101.4	6.0	0.335
Distance to Water (m)	305.4	97.9	580.0	188.2	0.086
Distance to Road (m)	1531.4	428.1	755.0	222.2	0.089
Slope (degrees)	22.4	2.0	15.0	3.4	0.031

Table 9. Habitat variable means for nest site categories in the Bennett Hills, Idaho in 2008 and 2009.

	Overall	2008	2009	Little Canyon	Dempsey
Shrub (%)	39	49	42	48	42
Grass (%)	6	6	5	34	13
Forbs (%)	4	3	5	6	2
Litter (%)	13	14	10	14	9
Bare ground (%)	13	7	8	8	7
Rock (%)	11	11	16	13	12
Tallest shrub (cm)	107.6	107.1	120.7	108.5	118.7
Shortest shrub (cm)	4.5	46.7	54.7	49	51.6
Visual obstruction (cm)	79.7	99.9	90.6	99.8	90.7
Distance to water (m)	400	440	388	362	500
Slope (degrees)	17	21	16	22	15
Canopy cover (%)	9	14	7	16	4

STUDY IV: Pheasant ecology and management

Territorial male pheasant density response to habitat changes.

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ABSTRACT Surveys of pheasants (*Phasianus colchicus*) were conducted during the springs of 1994-1999 on 13, one-mile sections of agricultural land in Gooding County, Idaho. Crop reports and aerial photos are being used to determine cover types within the sections throughout the sampling years. The data has been edited and is now being analyzed.

KEYWORDS pheasants, *Phasianus colchicus* , territories, habitat, density

One aspect of pheasant ecology that may be limiting populations is habitat available for territorial males during spring breeding season (Robertson et al. 1993, Robertson 1996). Males display in open areas adjacent to heavier cover. The open cover provides for optimum displaying and attracting mates while adjacent heavier vegetation provides escape security from predators (Leif 2005). As habitat is limited in agricultural areas, densities of territorial male pheasants may also be limited, reducing availability for mating with females and ultimately reducing the population size. Affects of habitat change can be better understood by examining pheasant numbers over time as they relate to landscape change. Relating these changes to territorial male densities can provide important information allowing managers to determine factors limiting population levels.

STUDY AREA

Pheasant surveys were conducted on square mile sections (259 ha) of agricultural land in northern Gooding County, Idaho. Gooding County is in south-central Idaho within the Snake River Plain. Dominant crops include alfalfa, corn, small grain, potatoes, sugar beets, pastures, and beans. Dispersed between the crops are idle areas of annual herbaceous cover, sagebrush (*Artemisia* spp.), irrigation canal banks of grass, riparian and wetland areas, and grass ditch banks along roads. The topography is relatively flat averaging 1,000 m elevations. The mean annual precipitation is 26.7 cm and annual temperatures average 8.7º C.

METHODS

Pheasants were counted from the ground according to protocol described by Robertson et al. (1993) and P. Robertson (pers. comm.). Observations were made a half hour before sunrise and 2 hours before sunset. Locations were plotted on aerial photos. An effort was made to observe every portion of the section by moving to strategic positions, observing with spotting scopes and/or binoculars, and listening for crowing and wing flapping. Counts were conducted 3 times each spring, once during each of the following periods: 15-30 April, 1-14 May, and 15-31 May. Pheasants were classified into 3 groups: females, non-territorial males, and territorial males.

Territorial males crow and wing-flap during display. All males with accompanying females were considered territorial. Males displaying outside the section but within 107 m (320 ft) were also mapped.

Field edges were mapped with hand-held global positioning systems in 1999. These were overlain onto base maps of 1987 orthophoto quadrangle imagery from Idaho Department of Lands using ArcView (ESRI, Redlands CA 92373). Crop types were determined from Farm Services Administration (FSA) databases and field edges corrected for each year from FSA's annual aerial photos. Cover types were also determined 107 m (320 ft) and 214 m (640 ft) outside of the sections to include territories observed outside of the section.

Pheasant locations were transferred from field maps made by observers to global information systems (GIS) maps. P. Robertson (pers. comm.) estimated territories were 3 ha (7 ac) in Nevada. Therefore, we plotted 3 ha circles around each territorial male location so underlying cover can be measured.

RESULTS

Thirteen sections were surveyed for pheasant densities during 1994-1999. Data entry and GIS map editing is complete – analysis is nearing completion.

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STUDY IV: Pheasant ecology and management

A web based information brochure for upland game birds of Idaho

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ABSTRACT Ring-necked pheasants (*Phasianus colchicus*) as well as other upland game birds are a popular species among Idaho hunters. Research conducted in Idaho has not been adequately provided to the general public. Compilation of the work presented on the Department's webpage will allow interested publics to learn ecology and management. This will be expanded to include all 12 upland game birds in Idaho.

Ring-necked pheasants are a popular game bird among Idaho hunters. Unfortunately, populations have declined throughout the state and some controversy has arisen regarding the reasons. Considerable research has been done to better understand pheasant population dynamics and the influence of various factors on pheasant numbers but has not been adequately presented to the general public explaining the plight of this game bird. Creating a webpage with ecology and management information for not only pheasants, but for each of Idaho's upland game birds, will provide the general public and resource professionals with much needed information. This will be an ongoing project with 2 species added annually for a total of 12 upland game bird species. Pages will be updated regularly as new information or status becomes available.

RESULTS

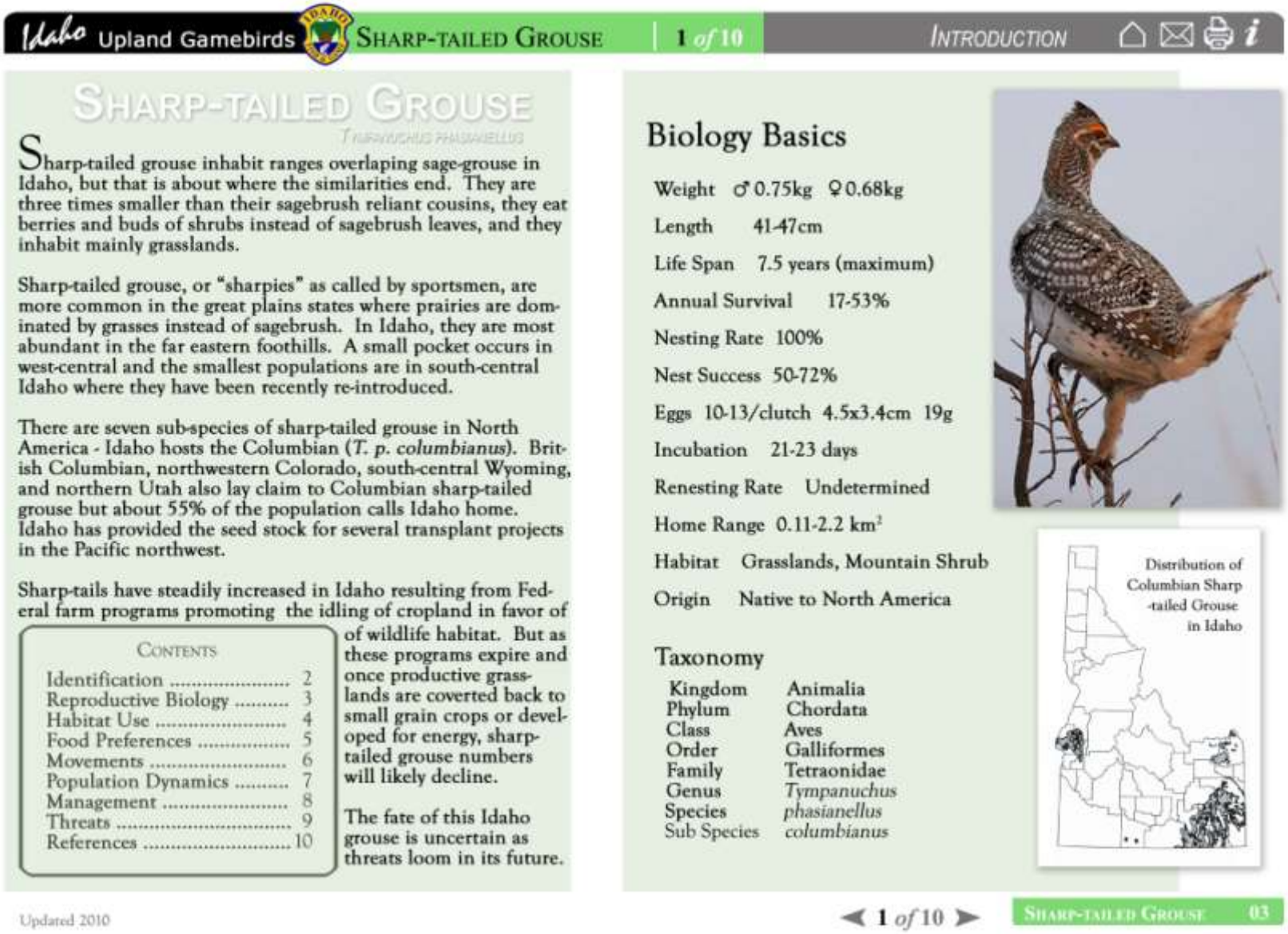
Results from research projects on upland game birds in Idaho, including university studies, are being compiled and a non-technical text is being written for the general public. Information on ecology and management from other literature sources is also included. Each species has standardized 10 pages of content. The pages are formatted to be downloaded and printed by users with enough space on the top margin to accept a 3 ring binder for shelf access.

Pages for greater sage-grouse and ring-necked pheasants are nearing completion – we are awaiting approval for use of copyrighted photographs and internal review of content before the pages are published on-line. Work has begun on pages for Columbian sharp-tailed grouse (*Tympanuchus phasianellus*, Fig. 1) and ruffed grouse (*Bonasa umbellus*, Fig. 2). A schedule for completion has been developed (Table 1) with the final pages estimated to be available by 2016.

Table 1. Estimated time-line for production of upland game bird series of web-based information on ecology and management in Idaho.

Year	Species

2011	Greater Sage-Grouse & Ring-necked Pheasants
2012	Columbian Sharp-tailed Grouse & Ruffed Grouse
2013	Mountain Quail & Dusky Grouse
2014	Chukar & Spruce Grouse
2015	Grey Partridge & California Quail
2016	Wild Turkey & Mourning Dove



Idaho Upland Gamebirds | Sharp-tailed Grouse | 1 of 10 | Introduction

Sharp-tailed Grouse

Tympanuchus phasianellus

Sharp-tailed grouse inhabit ranges overlaping sage-grouse in Idaho, but that is about where the similarities end. They are three times smaller than their sagebrush reliant cousins, they eat berries and buds of shrubs instead of sagebrush leaves, and they inhabit mainly grasslands.

Sharp-tailed grouse, or "sharpies" as called by sportsmen, are more common in the great plains states where prairies are dominated by grasses instead of sagebrush. In Idaho, they are most abundant in the far eastern foothills. A small pocket occurs in west-central and the smallest populations are in south-central Idaho where they have been recently re-introduced.

There are seven sub-species of sharp-tailed grouse in North America - Idaho hosts the Columbian (*T. p. columbianus*). British Columbian, northwestern Colorado, south-central Wyoming, and northern Utah also lay claim to Columbian sharp-tailed grouse but about 55% of the population calls Idaho home. Idaho has provided the seed stock for several transplant projects in the Pacific northwest.

Sharp-tails have steadily increased in Idaho resulting from Federal farm programs promoting the idling of cropland in favor of of wildlife habitat. But as these programs expire and once productive grasslands are coverted back to small grain crops or developed for energy, sharp-tailed grouse numbers will likely decline.

The fate of this Idaho grouse is uncertain as threats loom in its future.

Contents

Identification 2
Reproductive Biology 3
Habitat Use 4
Food Preferences 5
Movements 6
Population Dynamics 7
Management 8
Threats 9
References 10

Biology Basics

Weight ♂ 0.75kg ♀ 0.68kg

Length 41-47cm

Life Span 7.5 years (maximum)

Annual Survival 17-53%

Nesting Rate 100%

Nest Success 50-72%

Eggs 10-13/clutch 4.5x3.4cm 19g

Incubation 21-23 days

Renesting Rate Undetermined

Home Range 0.11-2.2 km²

Habitat Grasslands, Mountain Shrub

Origin Native to North America

Taxonomy

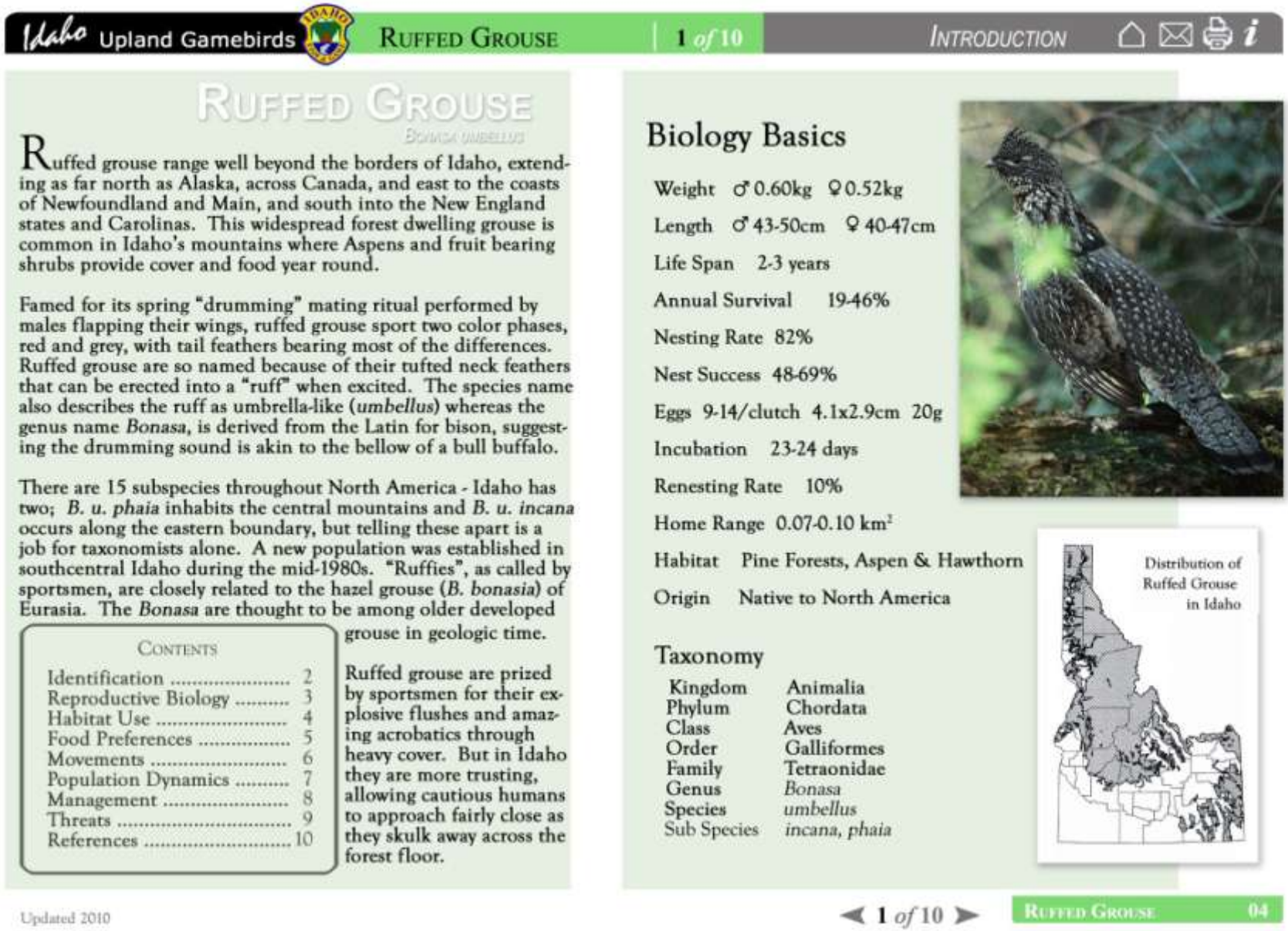
Kingdom	Animalia
Phylum	Chordata
Class	Aves
Order	Galliformes
Family	Tetraonidae
Genus	*Tympanuchus*
Species	*phasianellus*
Sub Species	*columbianus*

Distribution of Columbian Sharp-tailed Grouse in Idaho

Updated 2010

1 of 10 | Sharp-tailed Grouse 03

Figure 1. Front page for Columbian sharp-tailed grouse in web-based series of upland game bird ecology and management in Idaho.



Idaho Upland Gamebirds | Ruffed Grouse | 1 of 10 | Introduction

Ruffed Grouse

Bonasa umbellus

Ruffed grouse range well beyond the borders of Idaho, extending as far north as Alaska, across Canada, and east to the coasts of Newfoundland and Main, and south into the New England states and Carolinas. This widespread forest dwelling grouse is common in Idaho's mountains where Aspens and fruit bearing shrubs provide cover and food year round.

Famed for its spring "drumming" mating ritual performed by males flapping their wings, ruffed grouse sport two color phases, red and grey, with tail feathers bearing most of the differences. Ruffed grouse are so named because of their tufted neck feathers that can be erected into a "ruff" when excited. The species name also describes the ruff as umbrella-like (*umbellus*) whereas the genus name *Bonasa*, is derived from the Latin for bison, suggesting the drumming sound is akin to the bellow of a bull buffalo.

There are 15 subspecies throughout North America - Idaho has two; *B. u. phaia* inhabits the central mountains and *B. u. incana* occurs along the eastern boundary, but telling these apart is a job for taxonomists alone. A new population was established in southcentral Idaho during the mid-1980s. "Ruffies", as called by sportsmen, are closely related to the hazel grouse (*B. bonasia*) of Eurasia. The *Bonasa* are thought to be among older developed grouse in geologic time.

Contents

Identification 2
Reproductive Biology 3
Habitat Use 4
Food Preferences 5
Movements 6
Population Dynamics 7
Management 8
Threats 9
References 10

Ruffed grouse are prized by sportsmen for their explosive flushes and amazing acrobatics through heavy cover. But in Idaho they are more trusting, allowing cautious humans to approach fairly close as they skulk away across the forest floor.

Biology Basics

Weight ♂ 0.60kg ♀ 0.52kg

Length ♂ 43-50cm ♀ 40-47cm

Life Span 2-3 years

Annual Survival 19-46%

Nesting Rate 82%

Nest Success 48-69%

Eggs 9-14/clutch 4.1x2.9cm 20g

Incubation 23-24 days

Renesting Rate 10%

Home Range 0.07-0.10 km²

Habitat Pine Forests, Aspen & Hawthorn

Origin Native to North America

Distribution of Ruffed Grouse in Idaho

Taxonomy

Kingdom	Animalia
Phylum	Chordata
Class	Aves
Order	Galliformes
Family	Tetraonidae
Genus	*Bonasa*
Species	*umbellus*
Sub Species	*incana, phaia*

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◄ 1 of 10 ► Ruffed Grouse 04

Figure 2. Front page for ruffed grouse in web-based series of upland game bird ecology and management in Idaho.