



Research Article

Survival and Detectability Bias of Avian Fence Collision Surveys in Sagebrush Steppe

BRYAN S. STEVENS,¹ *Department of Fish and Wildlife Resources and Department of Statistics, P.O. Box 441136, University of Idaho, Moscow, ID 83844, USA*

KERRY P. REESE, *Department of Fish and Wildlife Resources, P.O. Box 441136, University of Idaho, Moscow, ID 83844, USA*

JOHN W. CONNELLY, *Idaho Department of Fish and Game, 1345 Barton Road, Pocatello, ID 83204, USA*

ABSTRACT We used female 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 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 to 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 to 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 (*A. arbuscula*) and black sagebrush (*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 detection probability by approximately 30%. Avian fence collision surveys in sagebrush-steppe habitats should be conducted at ≤ 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 sites. © 2011 The Wildlife Society.

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, Bevanger and Brøseth 2000, Wolfe et al. 2007), 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, 2004). Connelly et al. (2000) suggested flagging fences to increase their visibility near sage-grouse habitats if the fences appear hazardous, but no studies have evaluated the degree of hazard fences present to sage-grouse. Many populations of shrub-steppe birds are believed to be declining (Knick et al. 2003), including sage-grouse, and elevated structures have been hypothesized to contribute to sage-grouse declines (Braun 1998, Connelly et al. 2004). Despite the lack of information concerning collision risk of sage-grouse, collision with fences and power lines has been documented (Beck et al. 2006, Flake et al. 2010), 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 to quantify collision risk across the landscape.

Received: 29 January 2010; Accepted: 4 July 2010

Additional Supporting Information may be found in the online version of this article.

¹E-mail: stev8930@vandals.uidaho.edu

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). Although this method is 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 observers (Bevanger 1999). Scavenging bias refers to survival of the collision evidence (i.e., temporal longevity of evidence at a collision site), 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 refer to this as survival bias.

Numerous attempts to quantify carcass retention rates have been made and usually involve monitoring collision victims or planted carcasses (James and Haak 1979, Bevanger et al. 1994, Savereno et al. 1996). Most previous studies of carcass retention suggest birds were placed randomly; however, these studies often lacked a strong experimental design or presented 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 those authors did not provide a detailed description of field placement methods. 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 among 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 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 (e.g., large raptors, large non-raptors), whereas 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

We replicated our study 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 and Upper Snake study areas. The Browns Bench region was in southern Twin Falls County and bordered Nevada. Elevations on Browns Bench ranged from approximately 1,450–1,850 m, and vegetation was dominated by little sagebrush (*Artemisia arbuscula*) and black sagebrush (*A. nova*) in the southern portion of the study area and big sagebrush (*A. 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 Browns Bench commonly contained stands of big sagebrush.

The Upper Snake study site occurred on the Table Butte and Crooked Creek areas of the Upper Snake River Plain, in Clark and Jefferson counties. We selected the Table Butte area for big sagebrush habitat and the Crooked Creek area for little sagebrush habitat because we could not locate big and little sagebrush-dominated areas at the same site in Upper Snake region. The Table Butte and Crooked Creek areas were approximately 10 km apart, with similar elevations ranging from approximately 1,520–1,825 m. The big sagebrush study site was directly north of Table Butte, bounded on the east by United States Highway 15, and extended to the north and west of Table Butte by approximately 2.5 km and 8 km, respectively. 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

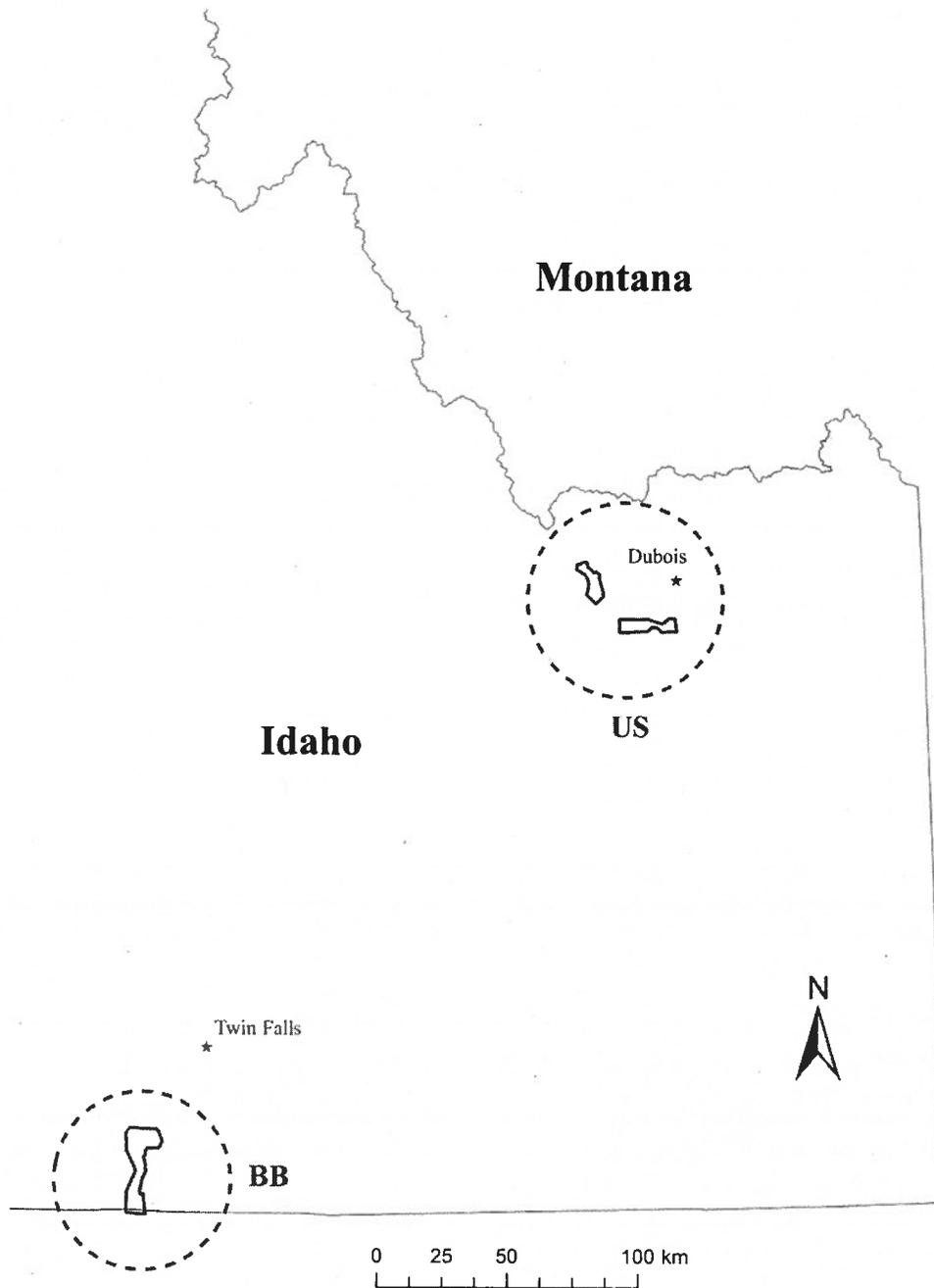


Figure 1. Southern Idaho, USA, study areas, where we studied survival and detectability of female pheasant carcasses planted as hypothetical collision victims. Dashed lines represent our 2 study regions (BB, Browns Bench; US, Upper Snake) during spring of 2009.

sagebrush, with some large pasture and grassy areas inter-mixed. Furthermore, unlike the Browns Bench study site, habitat types in the Upper Snake study area were distinctly separated, such that carcasses placed in big sagebrush were not as near to those placed in little sagebrush as on the Browns Bench study site.

Avian, mammalian, and arthropod scavengers were common on the study areas. Common scavengers previously identified in the vicinity of our study sites included coyotes (*Canis latrans*), American badgers (*Taxidea taxus*), weasels (*Mustela* spp.), common ravens (*Corvus corax*), American crows (*Corvus brachyrhynchos*), and black-billed magpies (*Pica hudsonia*; Coates et al. 2008). Small mammal species

previously identified included least chipmunks (*Tamias minimus*), Wyoming ground squirrels (*Spermophilus elegans*), Piute ground squirrels (*S. mollis*), northern pocket gophers (*Thomomys talpoides*), Great Basin pocket mice (*Perognathus parvus*), deer mice (*Peromyscus maniculatus*), and sagebrush voles (*Lemmys curtatus*; Coates et al. 2008). 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 Upper Snake 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 female ring-necked pheasants (*Phasianus colchicus*) as 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 and 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 using cervical dislocation, with approval from the University of Idaho Animal Care and Use Committee (Protocol 2009-21).

We banded all carcasses for individual identification prior to field placement to aide in monitoring. To prevent scavenging bias associated with unrealistic carcass presentation (Bumann and Stauffer 2002), we removed feathers from the front of the breast of each carcass and made 2 perpendicular 4-cm incisions, centered where the feathers were removed from the breast, to simulate collision with a barbed-wire fence. We placed carcasses 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 ($n = 10\text{--}15$ feathers) in the field at night to minimize diurnal scavenger detection by simply observing field workers. We placed each carcass 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. We also placed lone piles of 10–15 breast feathers to determine feather-pile detectability, as it could differ from carcass detectability.

We measured the influence of habitat type and distance from fence on detectability and longevity for the entire collision sign in addition to carcasses. Placing feathers around each carcass prevented it from being removed and leaving no visible sign, an unlikely scenario with collision victims, as feathers will likely fall when birds strike fences (Flake et al. 2010). We replicated this design on both the Browns Bench and Upper Snake study areas to allow detection of regional differences in carcass survival. We only measured detectability on the Browns Bench site due to a lack of field personnel available to conduct detectability trials on the Upper Snake 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 Geographic Information System (GIS) software. We mapped fence sections in each habitat type 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, we only included fence segments traversing areas dominated on both sides by the desired habitat type. We did not differentiate between little sagebrush and black sagebrush, and we grouped both species together in the little sagebrush level of habitat treatment.

Once we digitized fence sections, we used the GIS extension Hawth's Tools (Hawth's analysis tools for ArcGIS version 3, <<http://www.spatial ecology.com/htools>>, accessed 12 Jan 2009) 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, so larger inter-carcass distances are likely necessary. Therefore, we ensured that carcass locations were ≥ 200 m apart. We placed 50 bird carcasses on each study area with 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 similarly generated 50 points (25 for each treatment level) on the Browns Bench study area for placement of feather piles used in detectability trials. We ground verified all randomly generated points were in the desired habitat type prior to field placement, and discarded points not in the desired habitat. We placed carcasses and feather piles on the Browns Bench site randomly along 31.7 km of fence (little sagebrush = 18.2 km, big sagebrush = 13.5 km). Less fence was available on the Upper Snake site, so 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, which could bias results if scavengers used or avoided improved road corridors. We did include fence sections running along unimproved 2-track dirt or gravel roads due to the abundance of these roads on the study areas, and we treated road presence as a random covariate in this analysis. We placed 32 carcasses along fences with a road present (Upper Snake = 21, Browns Bench = 11), with more in little sagebrush ($n = 23$) than in big sagebrush habitats ($n = 9$).

The day following placement of carcasses on the Browns Bench 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 field searchers on day one.

After initial searches we monitored carcasses and their remaining sign 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 qualitatively described the carcass sign and remaining feathers over time within approximately 5 m of the

original carcass location. The observer also recorded 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 near carcasses during each search.

We recorded microhabitat characteristics of carcass placement sites after carcass removal to determine influences of vegetation on survival and detectability. We measured grass height and shrub height at the carcass location and 1 m from the carcass location in each cardinal direction (Hausleitner et al. 2005). We measured shrub canopy coverage on 2 perpendicular 4-m transects centered on the carcass location and oriented in each cardinal direction using the line-intercept method (Canfield 1941). Additionally, we used a 12 cm × 12 cm coverboard to estimate percent visual concealment at heights of 1.5 m and 1.0 m, at a distance of 10 m in each cardinal direction from the carcass location (Jones 1968, Hausleitner et al. 2005). Due to observer error, sign from one carcass on the Browns Bench study area was not monitored until complete removal 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 we monitored survival of carcasses analogously to 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 one day, and survival probability is the probability of survival over the entire study period. We generated all survival models using the logit link function, such that

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

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

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

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

We conducted all modeling within an information-theoretic model selection framework (Burnham and Anderson 2002). We ranked hypothesized models using Akaike's

Information Criterion corrected for small sample sizes (AIC_c; Burnham and Anderson 2002). We estimated overdispersion (\hat{c}) in 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 ($\hat{c} > 1$) we ranked models using quasi-AIC corrected for small sample size (QAIC_c; Burnham and Anderson 2002). We used information-theoretic methods to compare models 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). We calculated unconditional variances (Buckland et al. 1997) 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 it 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 we replicated the field experiment on 2 study areas. Survival modeling for 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, available online at www.onlinelibrary.wiley.com). Furthermore, we hypothesized 3 biologically plausible 2-way interactions for these models (site × habitat, site × road, and habitat × road) and added them only when the terms in the interaction were in a model together among the top group of models ($\Delta\text{AIC}_c \approx \leq 2$; 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 we directly considered in the design of the study, these models represented a separate group of hypotheses that we did not directly compare to the local-scale models. Variables included in microhabitat scale modeling were average grass height, average shrub height, percent visual concealment (measured as 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. We did not include microhabitat predictor variables that were

Table 1. Parameters we used, and justification for each parameter, in survival and detectability modeling for female pheasant 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, ht 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 ht ^{a,b}	Documented effects on predation ^e , visual obstruction
Shrub canopy coverage ^{a,b}	Documented effects on predation ^{e,g} , visual obstruction
Grass ht ^{a,b}	Documented effects on predation ^{e,h} , visual obstruction
Visual concealment ^{a,b}	Documented effects on predation ^e , visual obstruction
Grass ht × shrub ht ^{a,b,c}	Shrub concealment effects may vary by grass ht
Grass ht × canopy coverage ^{a,b,c}	Shrub coverage effects may vary by grass ht

^a Parameter used in survival modeling.

^b Parameter used in logistic regression modeling.

^c We added interaction terms when 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 and Delehanty (2010).

^h Moynahan et al. (2007).

significantly correlated ($P < 0.05$) together in the same model. We hypothesized 2 biologically plausible 2-way interactions in 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 microhabitat variables resulted in comparison of 10 models at this scale, with the global model included to estimate overdispersion in the model set (Appendix A, available online at www.onlinelibrary.wiley.com). Candidate model sets we 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 (i.e., probability given event occurred). 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. We specifically used Program MARK in this analysis to facilitate model selection and 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, we included a parameter for snow conditions to accommodate this confounding factor. We compared all 15 combinations of additive models and the constant detectability model for the experimental factors, and we did not consider any interaction terms in this group of hypotheses due to small sample sizes (Appendix B, available online at www.onlinelibrary.wiley.com). Microhabitat characteristics we used in modeling were identical to those we used in survival analysis (Table 1). We compared 9 models representing features at the microhabitat scale (Appendix B, available online at www.onlinelibrary.wiley.com). Once again we constructed these models such that we did not include strongly correlated predictor variables in the same model, and we included 2 hypothesized interaction terms (grass height × shrub height, grass height × canopy coverage) when their constituent terms were together among the top group of models (Table 1). We calculated generalized likelihood-ratio r -squared values (R_L^2) for each survival and logistic regression model to determine the performance of individual models relative to the null constant models (Menard 2000). Lastly, we estimated prediction success for each model using cross-validation procedures in SAS Version 9.2 (SAS, Cary, NC).

Crossvalidation procedures re-fit each model dropping each data point in turn and subsequently predicted probability of detection for each dropped data point with the re-fit model. If 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 female 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%, $n = 100$). For survival models we used visual concealment at a height of 1.0 m, which averaged 88.4% (SD = 16.7%, $n = 99$), whereas visual concealment at 1.5 m, which we used in detectability modeling, averaged 71.0% (SD = 26.0%, $n = 45$). One microhabitat measurement was missing each for grass height and visual concealment, thus for those factors $n = 99$. 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 we did not include these variables 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 (Browns Bench: $\bar{x} = 5.6$, SD = 3.0, $n = 50$; Upper Snake: $\bar{x} = 6.0$, 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 study sites in persistence of collision sign. Average number of days until sign was no longer detected on the Upper Snake site ($\bar{x} = 27.2$, SD = 6.8, $n = 50$) was greater than for the Browns Bench site ($\bar{x} = 8.8$, SD = 2.0, $n = 49$), due to many carcasses with sign surviving the entire study period on the Upper Snake site ($n = 32$; little sagebrush = 18, big sagebrush = 14). In contrast, none of the Browns Bench carcasses had sign persist the entire 31-day sampling period. Due to many carcasses with sign persisting the entire study period, average sign persistence calculated for the Upper Snake site was biased low, and the true average length of sign persistence is unknown.

Pheasant carcasses in little sagebrush habitats or on the Browns Bench study area were more likely to be directly removed during initial scavenging, whereas carcasses in big sagebrush habitats and those on the Upper Snake site were more likely to be scavenged in their original location and not directly removed. On the Browns Bench site 80% of carcasses were directly removed during initial scavenging (big sagebrush = 68%, little sagebrush = 92%), whereas 82% of carcasses on the Upper Snake site were scavenged in their original location (big sagebrush = 88%, little sagebrush = 76%). Furthermore, of the Upper Snake 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. We found evidence of small mammal scavenging at 9 carcasses (Browns Bench = 3, Upper Snake = 6), commonly in the form of tracks and scat on or near the carcass. Carcasses with apparent small mammal scavenging also had patches of feathers plucked or removed, and the underlying tissue appeared gnawed upon. We observed arthropods scavenging at 26 of the Upper Snake carcasses (big sagebrush = 19, little sagebrush = 7), whereas we did not observe scavenging by arthropods at the Browns Bench site. We also observed arthropod scavengers carrying feather evidence away from carcass locations on the Upper Snake 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 ($\Delta\text{QAIC}_c = 0$, $w_i = 0.238$; Table 2). We found minimal support for both habitat type and road presence influences on carcass survival ($\Delta\text{QAIC}_c < 2$), 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 slightly lower carcass daily survival in big sagebrush habitats ($\Delta\text{DSR} = 0.021$ – 0.022) and areas with no roads present ($\Delta\text{DSR} = 0.013$ – 0.014) regardless of habitat, however, precision was low and 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, we generated carcass survival models 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 ($\Delta\text{QAIC}_c = 0$, $w_i = 0.318$), and we found weak support for grass height, visual concealment, and shrub canopy coverage influences on carcass survival ($\Delta\text{QAIC}_c \leq 2$). 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 ($\Delta\text{QAIC}_c = 0$, $w_i = 0.341$), 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 ($\Delta\text{QAIC}_c < 2$). Local-scale sign survival models that did not contain a site effect parameter received virtually no support in this analysis ($\Delta\text{QAIC}_c > 25$), 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 to 0.988 on the Upper Snake site and from 0.863 to 0.872 on the Browns Bench site (Table 5). Differences in model averaged sign daily survival rates resulted in reduced model averaged sign survival

Table 2. Top models of daily survival rate of female pheasant 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. We ranked and compared models using quasi-Akaike's Information Criterion corrected for small sample sizes (QAIC_c) and normalized Akaike model weights (w_i ; Burnham and Anderson 2002).

Model	QDeviance ^a	K ^b	QAIC _c	ΔQAIC _c	w_i	Likelihood ^c	R_L^{2d}
Local-scale models ^e							
$S_{(.)}$	96.135	2	100.165	0.000	0.238	1.000	N/A
$S_{(Habitat)}$	95.233	3	101.293	1.128	0.135	0.569	0.009
$S_{(Road)}$	95.637	3	101.696	1.532	0.111	0.465	0.005
$S_{(Site)}$	96.129	3	102.188	2.023	0.087	0.364	0.000
$S_{(Dist)}$	96.135	3	102.194	2.030	0.086	0.363	0.000
Microhabitat models ^f							
$S_{(.)}$	95.720	2	99.749	0.000	0.318	1.000	N/A
$S_{(GH)}^g$	95.041	3	101.100	1.351	0.163	0.509	0.007
$S_{(VC)}^h$	95.614	3	101.673	1.924	0.121	0.382	0.001
$S_{(CC)}^i$	95.690	3	101.749	2.000	0.117	0.368	0.000
$S_{(SH)}^j$	95.719	3	101.778	2.029	0.115	0.363	0.000

^a QDeviance = quasi-deviance (Burnham and Anderson 2002).

^b K = no. of parameters in model.

^c Likelihood = w_i/w_{top} , where w_i = normalized Akaike model weight for model of interest, and w_{top} = normalized Akaike model weight for the top model (i.e., $\Delta QAIC_c = 0$; Burnham and Anderson 2002).

^d $R_L^2 = 1 - [\ln(L_m)/\ln(L_o)]$, where L_m = maximized likelihood for model of interest, and L_o = maximized likelihood for intercept only model (Menard 2000).

^e We estimated overdispersion (\hat{c}) for this group of models as the deviance divided by the deviance df. For this group of models $\hat{c} = 3.141$.

^f We estimated \hat{c} for this group of models as the deviance divided by the deviance df. For this group of models $\hat{c} = 3.155$.

^g GH = grass ht microhabitat variable.

^h VC = visual concealment microhabitat variable.

ⁱ CC = % shrub canopy coverage microhabitat variable.

^j SH = shrub ht microhabitat variable.

probabilities for the entire 31-day period from the Upper Snake (0.673–0.699) to Browns Bench (0.011–0.015) study sites. Regression coefficient estimates for the site effect model again showed the strong influence of site on sign daily survival rates ($\beta_1 = -2.528$, 95% CI = -3.566 to -1.490), with lower survival at the Browns Bench 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 microhabitat variables on sign survival, but again these models performed similarly to the null model in terms of proportional increase in likelihood ($R_L^2 = 0.003$ – 0.023 ; Table 4).

Table 3. Model averaged estimates of female pheasant 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).

Group	Daily survival	95% CI	
		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

Detection

Because we planted carcasses the night preceding detection trials, some carcasses were not present during 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 we excluded 4 carcasses from our calculations. Furthermore, we did not use 10 of the original feather piles placed, which were either blown away or covered with snow prior to initiating detection trials; however, the 3 carcasses removed all left feather piles and we used these feathers in feather pile detection calculations. Detection probability for feather piles was extremely low, only 1 of 43 (2.3%) feather piles present was located. We constructed no detectability models 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 in 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. Excluding these carcasses, the proportion of carcasses in big sagebrush detected rises 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

Table 4. Top models of daily survival rate of female pheasant 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. We ranked and compared models using quasi-Akaike's Information Criterion corrected for small sample sizes (QAIC_c) and normalized Akaike model weights (w_i ; Burnham and Anderson 2002).

Model	QDeviance ^a	K ^b	QAIC _c	ΔQAIC _c	w_i	Likelihood ^c	R _L ^{2d}
Local-scale models ^e							
S _(Site)	94.597	3	100.612	0.000	0.341	1.000	0.226
S _(Site+Dist)	94.298	4	102.322	1.711	0.145	0.425	0.228
S _(Site+Habitat)	94.322	4	102.347	1.735	0.143	0.420	0.228
S _(Site+Road)	94.596	4	102.621	2.009	0.125	0.366	0.226
S _(Site+Habitat+Dist) ^f	94.002	5	104.039	3.427	0.061	0.180	0.231
Microhabitat models ^f							
S _(.)	97.444	2	101.451	0.000	0.228	1.000	N/A
S _(GH) ^g	95.933	3	101.948	0.496	0.178	0.780	0.016
S _(CC) ^h	96.348	3	102.363	0.912	0.145	0.634	0.011
S _(SH) ⁱ	97.061	3	103.076	1.624	0.101	0.444	0.004
S _(VC) ^j	97.173	3	103.188	1.736	0.096	0.420	0.003
S _(GH+CC)	95.233	4	103.258	1.806	0.093	0.405	0.023

^a QDeviance = quasi-deviance (Burnham and Anderson 2002).

^b K = no. of parameters in model.

^c Likelihood = w_i/w_{top} , where w_i = normalized Akaike model weight for model of interest, and w_{top} = normalized Akaike model weight for the top model (i.e., ΔQAIC_c = 0; Burnham and Anderson 2002).

^d R_L² = 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).

^e We estimated overdispersion (\hat{c}) for this group of models as the deviance divided by the deviance df. For this group of models \hat{c} = 3.216.

^f We estimated \hat{c} for this group of models as the deviance divided by the deviance df. For this group of models \hat{c} = 4.032.

^g GH = grass ht microhabitat variable.

^h CC = % shrub canopy coverage microhabitat variable.

ⁱ SH = shrub ht microhabitat variable.

^j VC = visual concealment microhabitat variable.

(ΔAIC_c < 2). The top local-scale models all predicted success moderately (0.674; Table 6), and all 3 top 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 (β_1 = 1.447, 95% CI = 0.210–2.684).

In the analysis of microsite habitat characteristics on detection of carcasses during fence-line surveys the influence of shrub height on detectability was most supported by the data (ΔAIC_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 (ΔAIC_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 (β_1 = -0.039, 95% CI = -0.077 to -0.002). Both shrub height and canopy cover predicted detection success 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. Although the speed at which 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, Travani 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 turkey vultures (*Cathartes aura*) 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

Table 5. Model averaged estimates of female pheasant 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).

Group	Daily survival	95% CI	
		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 female pheasant 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. We ranked and compared models using Akaike's Information Criteria corrected for small sample sizes (AIC_c) and normalized Akaike model weights (w_i ; Burnham and Anderson 2002).

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

^a K = no. of parameters in model.

^b Likelihood = w_i/w_{top} , where w_i = normalized Akaike model weight for model of interest, and w_{top} = normalized Akaike model weight for the top model (i.e., ΔQAIC_c = 0; Burnham and Anderson 2002).

^c $R_L^2 = 1 - [\ln(L_m)/\ln(L_o)]$, where L_m = maximized likelihood for model of interest, and L_o = maximized likelihood for intercept only model (Menard 2000).

^d We calculated prediction success via crossvalidation in PROC LOGISTIC, PREDPROBS = CROSSVALIDATE output statement (SAS Version 9.2, Cary, NC).

^e SH = shrub ht microhabitat variable.

^f CC = % shrub canopy coverage microhabitat variable.

^g GH = grass ht microhabitat variable.

^h VC = visual concealment microhabitat variable.

fields to simulate poisoning were removed within 72 hr (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 common to shrub-steppe dominated landscapes (Hilton 1978, Heinrich 1988, Coates et al. 2008). Thus, the rapid scavenging we observed 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 bobwhite quail (*Colinus virginianus*) remains were completely removed within 4 days in Alabama, whereas only 13% of bobwhite quail 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 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 (≥100 feathers) and bird pieces scattered around the site. Similarly, Rosene and Lay (1963) found that large feather piles disappeared at a slower rate than small feather piles, which 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 (*Bonasa umbellus*) 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 (e.g., large raptors, large non-raptors), whereas vegetation only influenced detection for birds classified as small birds. Overall, our detection rate (0.53) appears low compared to previous published studies in other habitats. 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 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, it suggests presence of many collision sites.

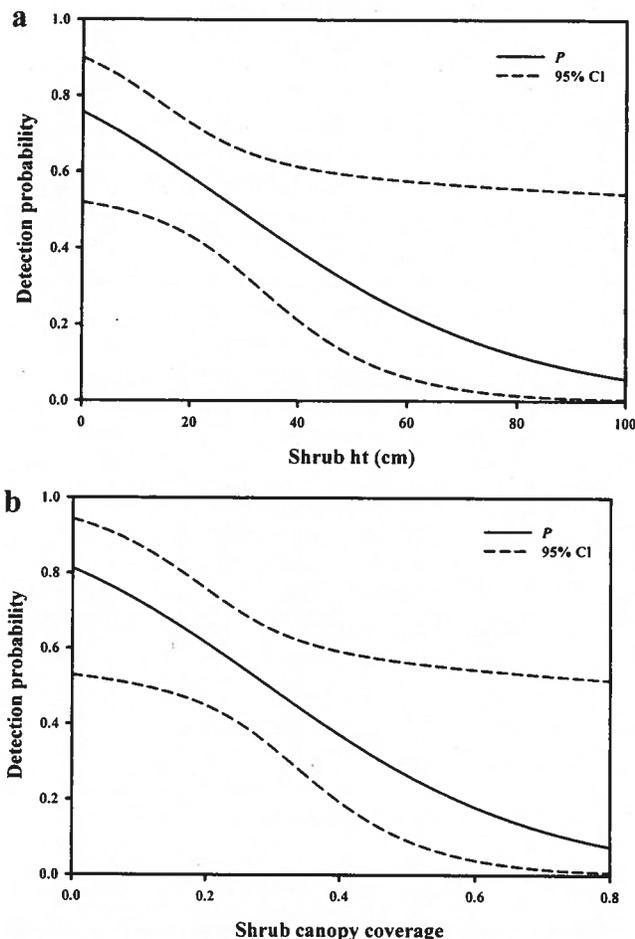


Figure 2. Plots of female pheasant carcass detection probability during fence surveys as a function of microhabitat characteristics on the Browns Bench region of Idaho, USA, during 2009. (a) Carcass detection probability as a function of shrub height from the top microhabitat detectability model. (b)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

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 female 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, and greater sage-grouse were recently listed as warranted but precluded under the Endangered Species Act by the United States Fish and Wildlife Service (United States Department of the Interior 2010). Pheasant carcasses we used were similar to female sage-grouse 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 differences in coloration between female pheasants and male 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). We distributed carcasses across 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 Upper Snake site was high due to many 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 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

We provide 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 we documented. However, low daily survival rates for carcasses and collision sign suggest time-interval lengths between surveys should be $\leq 1-2$ weeks to avoid 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 (e.g., snow cover, extreme wind, or rain), and collision rates should be corrected based on identified site attributes influencing detection. Moreover, caution is warranted when aggregating or comparing uncorrected collision data from sites with varying vegetation characteristics, as detection probabilities are likely different between sites.

ACKNOWLEDGMENTS

This research was partially supported by grant no. 09659 from the Student Grant Program at the University of Idaho. Additional funding and field support for this project was provided by Idaho Department of Fish and Game and the United States Bureau of Land Management. We thank D.

Ayers, S. Jackson, C. Cardinal, A. Locatelli, and N. Muhn, for assisting in field data collection, and J. Baumgardt, D. Musil, C. Hendricks, and L. Cross for additional field support, without which this project would not have been possible. Pheasants we used were provided by Magic Valley Gamebirds, Larnjo Upland Gamebirds, and Western Wings Upland Gamebirds. This is contribution 1044 of the University of Idaho Forest, Wildlife and Range Experiment Station and Idaho Federal Aid in Wildlife Restoration Project W-160-R. We thank Drs. B. Dennis, E. O. Garton, K. Vierling, and L. Brennan, as well 2 anonymous reviewers, whose suggestions and helpful comments improved this manuscript.

LITERATURE CITED

- Avery, M. L., P. F. Springer, and J. F. Cassel. 1978. The composition and seasonal variation of bird losses at a tall tower in southeastern North Dakota. *American Birds* 32:1114-1121.
- Anderson, W. L. 1978. Waterfowl collisions with power lines at a coal-fired power plant. *Wildlife Society Bulletin* 6:77-83.
- Baines, D., and R. W. Summers. 1997. Assessment of bird collisions with deer fences in Scottish forests. *Journal of Applied Ecology* 34:941-948.
- Beck, J. L., K. P. Reese, J. W. Connelly, and M. B. Lucia. 2006. Movements and survival of juvenile greater sage-grouse in southeastern Idaho. *Wildlife Society Bulletin* 34:1070-1078.
- Bevanger, K. 1995. Estimates and population consequences of tetraonid mortality caused by collisions with high tension power lines in Norway. *Journal of Applied Ecology* 32:745-753.
- Bevanger, K. 1999. Estimating bird mortality caused by collision with power lines and electrocution: a review of methodology. Pages 29-56 in M. Ferrer and G. F. E. Janss, editors. *Birds and power lines: collision, electrocution and breeding*. Quercus, Madrid, Spain.
- Bevanger, K., and H. Brøseth. 2000. Reindeer *Rangifer tarandus* fences as a mortality factor for ptarmigan *Lagopus* spp. *Wildlife Biology* 6:121-127.
- Bevanger, K., and H. Brøseth. 2004. Impact of power lines on bird mortality in a subalpine area. *Animal Biodiversity and Conservation* 27:67-77.
- Bevanger, K., Ø. Bakke, and S. Engen. 1994. Corpse removal experiments with willow ptarmigan (*Lagopus lagopus*) in power-line corridors. *Ökologie der Vögel* 16:597-607.
- Bradley, L. C., and D. B. Fagre. 1988. Coyote and bobcat responses to integrated ranch management practices in south Texas. *Journal of Range Management* 41:322-327.
- Braun, C. E. 1998. Sage-grouse declines in western North America: what are the problems? *Proceedings of the Western Association of State Fish and Wildlife Agencies* 78:139-156.
- Buckland, S. T., K. P. Burnham, and N. H. Augustin. 1997. Model selection: an integral part of inference. *Biometrics* 53:603-618.
- Bumann, G. B., and D. F. Stauffer. 2002. Scavenging of ruffed grouse in the Appalachians: influences and implications. *Wildlife Society Bulletin* 30:853-860.
- Burnham, K. P., and D. R. Anderson. 2002. Model selection and multi-model inference: a practical information theoretic approach. Second edition. Springer-Verlag, New York, New York, USA.
- Canfield, R. H. 1941. Application of the line interception method in sampling range vegetation. *Journal of Forestry* 39:388-394.
- Coates, P. S., and D. J. Delehanty. 2010. Nest predation of greater sage-grouse in relation to microhabitat factors and predators. *Journal of Wildlife Management* 74:240-248.
- Coates, P. S., J. W. Connelly, and D. J. Delehanty. 2008. Predators of greater sage-grouse nests identified by video monitoring. *Journal of Field Ornithology* 79:421-428.
- Connelly, J. W., M. A. Schroeder, A. R. Sands, and C. E. Braun. 2000. Guidelines to manage sage grouse populations and their habitats. *Wildlife Society Bulletin* 28:967-985.
- Connelly, J. W., S. T. Knick, M. A. Schroeder, and S. J. Stiver. 2004. Conservation assessment of greater-sage grouse and sagebrush habitats. Western Association of Fish and Wildlife Agencies, Cheyenne, Wyoming, USA.
- Crawford, R. L. 1971. Predation on birds killed at a TV tower. *Oriole* 36:33-35.
- DeVault, T. L., O. E. Rhodes, and J. A. Shivik. 2003. Scavenging by vertebrates: behavioral, ecological, and evolutionary perspectives on an important energy transfer pathway in terrestrial ecosystems. *Oikos* 102: 225-234.
- Dinsmore, S. J., G. C. White, and F. L. Knopf. 2002. Advanced techniques for modeling avian nest survival. *Ecology* 83:3476-3488.
- Drewitt, A. L., and R. H. W. Langston. 2008. Collision effects of wind-power generators and other obstacles on birds. *Year in Ecology and Conservation Biology* 2008. *Annals of the New York Academy of Sciences* 1134:233-266.
- Flake, L. D., J. W. Connelly, T. R. Kirschenmann, and A. J. Lindbloom. 2010. Grouse of plains and mountains: the South Dakota story. South Dakota Department of Game, Fish, and Parks, Pierre, USA.
- Gehring, J., P. Kerlinger, and A. M. Manville. 2009. Communication towers, lights, and birds: successful methods of reducing the frequency of avian collisions. *Ecological Applications* 19:505-514.
- Gregg, M. A., J. A. Crawford, M. S. Drut, and A. K. DeLong. 1994. Vegetational cover and predation of sage grouse nests in Oregon. *Journal of Wildlife Management* 58:162-166.
- Hausleitner, D., K. P. Reese, and A. D. Apa. 2005. Timing of vegetation sampling at greater sage-grouse nests. *Rangeland Ecology and Management* 58:553-556.
- Heinrich, B. 1988. Winter foraging at carcasses by three sympatric corvids, with emphasis on recruitment by the raven, *Corvus corax*. *Behavioral Ecology and Sociobiology* 23:141-156.
- Hilton, H. 1978. Systematics and ecology of the eastern coyote. Pages 220-227 in M. Bekoff, editor. *Coyotes: biology, behavior, and management*. The Blackburn Press, Caldwell, New Jersey, USA.
- Holloran, M. J., B. J. Heath, A. G. Lyon, S. J. Slater, J. L. Kuipers, and S. H. Anderson. 2005. Greater sage-grouse nesting habitat selection and success in Wyoming. *Journal of Wildlife Management* 69:638-649.
- Houston, D. C. 1986. Scavenging efficiency of turkey vultures in tropical forest. *Condor* 88:318-323.
- James, B. W., and B. A. Haak. 1979. Factors affecting avian flight behavior and collision mortality at transmission lines. Bonneville Power Administration, Portland, Oregon, USA.
- Janss, G. F. E., and M. Ferrer. 2000. Common crane and great bustard collision with power lines: collision rate and risk exposure. *Wildlife Society Bulletin* 28:675-680.
- Jones, R. E. 1968. A board to measure cover used by prairie grouse. *Journal of Wildlife Management* 32:28-31.
- Knick, S. T., D. S. Dobkin, J. T. Rotenberry, M. A. Schroeder, W. M. Vander Haegen, and C. van Riper. 2003. Teetering on the edge or too late? conservation and research issues for avifauna of sagebrush habitats. *Condor* 105:611-634.
- Kostecke, R. M., G. M. Linz, and W. J. Bleier. 2001. Survival of avian carcasses and photographic evidence of predators and scavengers. *Journal of Field Ornithology* 72:439-447.
- Kuvlesky, W. P., L. A. Brennan, M. L. Morrison, K. K. Boydston, B. M. Ballard, and F. C. Bryant. 2007. Wind energy development and wildlife conservation: challenges and opportunities. *Journal of Wildlife Management* 71:2487-2498.
- Linz, G. M., J. E. Davis, Jr., R. M. Engeman, D. L. Otis, and M. L. Avery. 1991. Estimating survival of bird carcasses in cattail marshes. *Wildlife Society Bulletin* 19:195-199.
- McCullagh, P., and J. A. Nelder. 1989. *Generalized linear models*. Second edition. Chapman and Hall, New York, New York, USA.
- Menard, S. 2000. Coefficients of determination for multiple logistic regression analysis. *American Statistician* 54:17-24.
- Moss, R., N. Picozzi, R. W. Summers, and D. Baines. 2000. Capercaille *Tetrao urogallus* in Scotland—demography of a declining population. *Ibis* 142:259-267.
- Moynahan, B. J., M. S. Lindberg, J. J. Rotella, and J. W. Thomas. 2007. Factors affecting nest survival of greater sage-grouse in northcentral Montana. *Journal of Wildlife Management* 71:1773-1783.

- Osborn, R. G., K. F. Higgins, R. E. Usgaard, C. D. Dieter, and R. D. Neiger. 2000. Bird mortality associated with wind turbines at the Buffalo Ridge Wind Resource Area, Minnesota. *American Midland Naturalist* 143:41–52.
- Ott, R. L., and M. Longnecker. 2001. An introduction to statistical methods and data analysis. Fifth edition. Wadsworth Group-Thompson Learning, Inc., Pacific Grove, California, USA.
- Pain, D. J. 1991. Why are lead-poisoned waterfowl rarely seen? The disappearance of waterfowl carcasses in the Camargue, France. *Wildfowl* 42:118–122.
- Peterson, C. A., S. L. Lee, and J. E. Elliot. 2001. Scavenging of waterfowl carcasses by birds in agricultural fields of British Columbia. *Journal of Field Ornithology* 72:150–159.
- Prior, K. A., and P. J. Weatherhead. 1991. Competition at the carcass: opportunities for social foraging by turkey vultures in southern Ontario. *Canadian Journal of Zoology* 69:1550–1556.
- Rosene, W., and D. W. Lay. 1963. Disappearance and visibility of quail remains. *Journal of Wildlife Management* 27:139–142.
- Rotella, J. J., M. L. Taper, and A. J. Hansen. 2000. Correcting nest-success estimates for observer effects: maximum-likelihood estimates of daily survival rates with reduced bias. *Auk* 117:92–109.
- Rotella, J. J., S. J. Dinsmore, and T. L. Shaffer. 2004. Modeling nest-survival data: a comparison of recently developed methods that can be implemented in MARK and SAS. *Animal Biodiversity and Conservation* 27:187–205.
- Savereno, A. J., L. A. Savereno, R. Boettcher, and S. M. Haig. 1996. Avian behavior and mortality at power lines in coastal South Carolina. *Wildlife Society Bulletin* 24:636–648.
- Seber, G. A. F. 1982. The estimation of animal abundance and related parameters. Second edition. Macmillan Book, New York, New York, USA.
- Shaffer, T. L. 2004. A unified approach to analyzing nest success. *Auk* 121:526–540.
- Smallwood, K. S. 2007. Estimating wind turbine-caused bird mortality. *Journal of Wildlife Management* 71:2781–2791.
- Smallwood, K. S., and C. Thelander. 2008. Bird mortality in the Altamont Pass Wind Resource Area, California. *Journal of Wildlife Management* 72:215–223.
- Smallwood, K. S., L. Ruge, and M. L. Morrison. 2009. Influence of behavior on bird mortality in wind energy developments. *Journal of Wildlife Management* 73:1082–1098.
- Smallwood, K. S., D. A. Bell, S. A. Snyder, and J. E. DiDonato. 2010. Novel scavenger removal trials increase wind turbine-caused avian fatality estimates. *Journal of Wildlife Management* 74:1089–1097.
- Stoddart, L. C. 1970. A telemetric method for detecting jackrabbit mortality. *Journal of Wildlife Management* 34:501–507.
- Tobin, M. E., and R. A. Dolbeer. 1990. Disappearance and recoverability of songbird carcasses in fruit orchards. *Journal of Field Ornithology* 61:237–242.
- Travaini, A., J. A. Donazar, A. Rodríguez, O. Ceballos, M. Funes, M. Delibes, and F. Hiraldo. 1998. Use of European hare (*Lepus europaeus*) carcasses by an avian scavenging assemblage in Patagonia. *Journal of Zoology* 246:175–181.
- United States Department of the Interior. 2010. Endangered and threatened wildlife and plants; 12-month finding for petitions to list greater sage-grouse as threatened or endangered. *Federal Register* 75:3909–14014.
- Whelan, C. J., M. L. Dilger, D. Robson, N. Hallyn, and S. Dilger. 1994. Effects of olfactory cues on artificial-nest experiments. *Auk* 111:945–952.
- White, G. C., and K. P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. *Bird Study* 46(Suppl): 120–138.
- Wolfe, D. H., M. A. Patten, E. Shochat, C. L. Pruett, and S. K. Sherrod. 2007. Causes and patterns of mortality in lesser prairie-chickens *Tympanuchus pallidicinctus* and implications for management. *Wildlife Biology* 13(Suppl 1): 95–104.

Associate Editor: Leonard A. Brennan.