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Original Article

Spent Lead Shot Availability and Ingestion by Ring-necked Pheasants in South Dakota

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ABSTRACT Lead is toxic to all vertebrate species and ingestion of lead ammunition has been reported in >130 avian species. Research has primarily focused on the effects and exposure of spent lead shot on waterfowl with little information about effects on upland game species, such as ring-necked pheasant (*Phasianus colchicus*). We collected 1,450 soil samples to estimate the availability of lead shot on 2 licensed shooting preserves in South Dakota, USA, 2012–2013. We concurrently collected gizzards from 660 hunter-harvested wild male pheasants from the shooting preserves and compared lead ingestion rates with those of 1,301 gizzards collected from nonpreserve areas. Spatial modeling showed the distribution of spent lead shot was associated with the systematic hunting pattern of each study site and, to a lesser extent, land-use type. Prevalence of ingested lead shot was 4.9 times greater for birds harvested on shooting preserves (3.9%, 95% CI = 2.7–5.7%) when compared with nonpreserve areas (0.8%, 95% CI = 0.4–1.4%) where lead shot availability was presumed less. Wild pheasants inhabiting areas of artificially high hunting intensity and lead deposition are at elevated risk of lead exposure and poisoning, although the consequences of lead ingestion in wild pheasants are unknown. © 2016 The Wildlife Society.

KEY WORDS availability, ingestion, lead, *Phasianus colchicus*, ring-necked pheasant, South Dakota, spent shot.

Lead is a nonspecific toxin to all vertebrate species (Eisler 1988, Murray et al. 2004) and ingestion of spent lead shot is the most common source of lead exposure in birds (Tranel and Kimmel 2009). Ingestion of lead ammunition (shot pellets, bullets and fragments, and prey contaminated with lead ammunition) has been documented in >130 avian species (Tranel and Kimmel 2009). Ingestion of lead causes reduced survival, poor body condition, behavioral changes, and impaired reproduction (reviewed by Tranel and Kimmel 2009). Prior to the 1987–1991 phased-in ban on lead ammunition for waterfowl hunting, an estimated 1.6–2.4 million waterfowl died annually from lead poisoning (Friend and Franson 1999). Additionally, an estimated 1.66 million mourning doves (*Zenaida macroura*) may die annually from ingesting lead pellets (Plautz et al. 2011). Both waterfowl and mourning doves are very susceptible to acute lead toxicosis, which causes reduced survival after ingestion of as few as 1–3 pellets (Jordan and Bellrose 1950, Schulz et al. 2006). In general, reported mortality from lead exposure is more common in waterfowl than nonmigratory upland game birds (Friend and Franson 1999).

Less is known about prevalence rates of ingested lead shot and the effects of lead poisoning on resident upland game birds. Prevalence rates of ingested lead pellets of 8% and 34% were documented in small samples of chukar (*Alectoris chukar*) and ring-necked pheasants (*Phasianus colchicus*; hereafter, pheasants), respectively, on a shooting estate in Canada (Kreager et al. 2008). Additional reported prevalence rates include 8.3% for chukar (Bingham et al. 2015), 1.3% for northern bobwhite (*Colinus virginianus*; Keel et al. 2002), 1.2% for ruffed grouse (*Bonasa umbellus*; Rodrigue et al. 2005), and 0.4% for scaled quail (*Callipepla squamata*; Best et al. 1992). Furthermore, 13% of wild turkeys (*Meleagris gallopavo*) sampled had elevated liver lead levels, possibly from ingesting lead shot (Kreager et al. 2008). Ingestion of lead shot and acute fatal poisoning in pheasants was reported as early as 1876 in Great Britain (Calvert 1876), and reduced reproductive parameters have been observed for captive pheasants dosed with lead pellets (Gasparik et al. 2012).

Studies on the availability of spent lead shot in upland habitats have been primarily limited to fields managed for mourning dove hunting. Pellet availability has been highly variable in managed dove fields and ranged from 0 to 860,000 pellets/ha (Lewis and Legler 1968, Anderson 1986, Castrale 1989, Best et al. 1992, Schulz et al. 2002, Douglass 2011). Among these studies, availability was shown to be greatest immediately posthunt and sometimes reduced by tillage. Holdner et al. (2004) found up to 560,000 pellets/ha in the top 10 cm of soil within a heavily hunted shooting estate in Canada. Interestingly, field studies have not shown a clear correlation between lead shot availability and ingestion in mourning doves, nor has this relationship

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been demonstrated in captive studies (Schulz et al. 2007, Plautz et al. 2011).

Wild pheasants are hunted in ≥25 states, with 1.1 million sportsmen harvesting an estimated 6.1 million birds annually (Midwest Pheasant Study Group 2012). Pheasants are the most abundant and most actively hunted upland game bird in South Dakota, USA (Flake et al. 2012). From 2000 to 2011, an average of 166,000 hunters harvested 1.68 million pheasants annually (Flake et al. 2012). In South Dakota, nontoxic shot is required for shotgun hunting of upland game on most public lands, which encompasses 1.6% of South Dakota's land area. Subsequently, the use of lead shot is allowed for pheasant hunting throughout most of South Dakota. Varying state-level restrictions on use of lead shot for upland game hunting exist, but lead shot is still used extensively for pheasant hunting. Additionally, pen-raised pheasants are commonly released throughout their range to augment wild populations for increased hunting opportunity. In areas where large numbers of released birds are hunted, wild pheasants may be exposed to greater amounts of deposited lead shot. For example, shooting preserves licensed with the South Dakota Department of Game, Fish and Parks may have elevated amounts of spent lead shot because they have liberal season length, relaxed bag limits, and release pen-raised birds for harvest. It is likely that wild pheasants are exposed to varying densities of deposited lead shot depending on local hunting pressure and laws related to the use of lead shot for hunting. Lead shot could be concentrated in very specific areas (e.g., food plots) where hunters are most likely to encounter pheasants. Our objectives were to 1) estimate and map the distribution and abundance of spent lead shot on 2 licensed shooting preserves; 2) develop a new method for collecting soil samples that can be utilized for untilled upland habitats (e.g., grass); and 3) estimate prevalence of ingested lead shot in pheasants harvested from the aforementioned shooting preserves and compare rates to birds harvested from nonpreserve areas.

STUDY AREA

Our study area consisted of 2 separate study sites located in the Northwestern Great Plains Eco-region, specifically the Sub-humid Pierre Shale Plains in Lyman County, South Dakota (Fig. 1; Bryce et al. 1996). Dominant soils were Millboro silty clay and Kolls silty clay soil type with 0–6% slope for Study Site 1 and Promise Clay with 0–3% slope and Millboro silty clay with 0–6% slope for Study Site 2 (Natural Resource Conservation Service 2012). These alkali (pH = 6.6–8.4) soils were neither saline nor sodic (total salinity <4 mmhos/cm, mostly <2) and did not have hard pans compared with other actual saline and sodic soils in South Dakota.

Study Site 1 was 248 ha and consisted of 68 ha of grassland (27%), 152 ha of cropland (62%), and 28 ha of food plots (11%). Study Site 2 was 378 ha and consisted of 237 ha of grassland (63%), 94 ha of food plots (25%), 38 ha of water–riparian area (10%), and 9 ha developed (2%). Food plots were tilled annually and consisted of unharvested grain sorghum or corn. All grasslands were predominately western

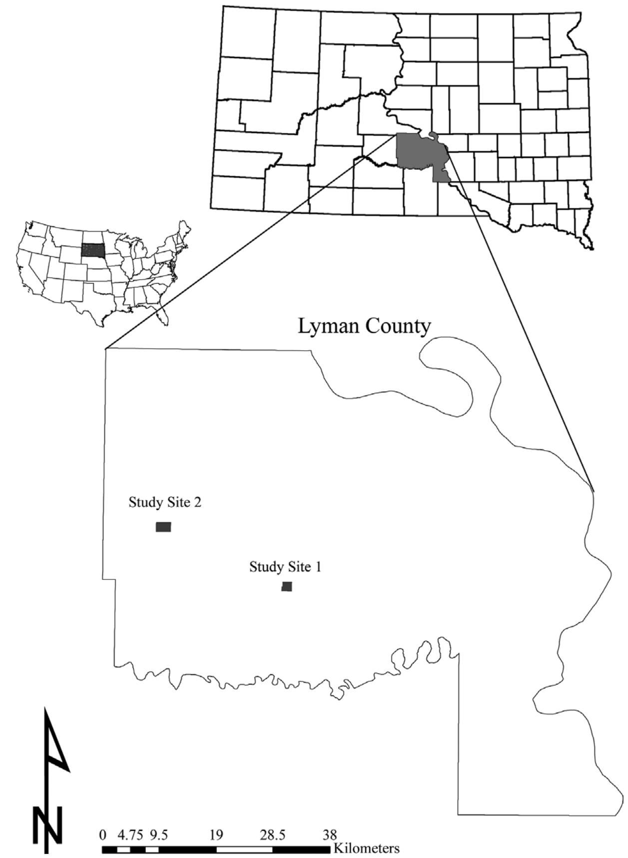

Figure 1. Map of study site locations where we studied availability and ingestion of lead shot by ring-necked pheasants in Lyman County, South Dakota, USA, 2012–2013.

wheatgrass (*Pascopyrum smithii*) and smooth brome (*Bromus inermis*), which were periodically managed by haying or mowing. The cropland in Study Site 1 was no-till farmed since 1996 with a rotation of corn, soy beans, wheat, sunflowers, and grain sorghum. Some of the land classified as grassland in Study Site 1 contained rows of recently planted eastern red cedar trees (*Juniperus virginiana*) that were <2 m tall.

Both study sites were private shooting preserves licensed with the South Dakota Department of Game, Fish and Parks. This designation allowed upland game hunting outside the regular upland-game season dates (7 months vs. 79 days for regular season), relaxed daily limits (15 daily vs. 3 daily for regular season), and required the release of at least as many pen-raised upland game birds as are harvested on the property. We selected private shooting preserves because they are subject to high levels of shooting intensity and offer a unique situation to estimate lead ingestion rates where lead availability was presumed to be much greater than other areas. Upland game bird release and harvest records were recorded by the private shooting preserve operator; thus, we also had a unique opportunity to compare lead availability between sites with varying lead deposition histories. Study Site 1 had been licensed as a shooting preserve since 2008 with an annual pheasant harvest of 5.4/ha from 2008 to 2011. Study Site 2 had been a licensed shooting preserve since 1996 with an annual pheasant harvest of 2.0/ha from 2005 to 2012.

Mean annual precipitation for both study sites was 50.3 cm (South Dakota Office of Climatology 2013). The study sites had a mean annual temperature of 8.3° C with temperatures ranging from 32° C to −15° C (South Dakota Office of Climatology 2013).

METHODS

Soil Sampling

We digitized our study sites by land-use type using National Agriculture Imagery Program compressed county mosaics as a reference for heads up digitizing in ArcMap (USDA 2012; ArcGIS 10.0; Environmental Systems Research Institute, Redlands, CA, USA). We classified land-use types as grassland, cropland, food plot, water, or developed (buildings, farm yard). In the first study site, we randomly generated 250 soil sample locations within grassland, cropland, and food plots (750 total) and collected soil samples during mid-July 2012. In the second study site, we randomly generated 450 soil-sample points in grasslands and 250 soil samples in food plots; we collected soil samples at Study Site 2 during mid-July 2013. We individually numbered and georeferenced each soil sample. We did not estimate spent-lead-shot abundance within water or developed land-use types because they were likely not important pheasant foraging areas.

We estimated spent-lead-shot availability by sampling the top 1.3 cm of the soil surface within 30.5-cm × 61.0-cm sample plots. Attempts to use soil sampling methods described by Castrale (1989) and Douglass (2011) were unsuccessful because of local soil and vegetation characteristics. We removed loose surface debris and standing vegetation by hand or with the use of pruning shears prior to sampling. We removed only large pieces of surface debris (e.g., crop residue), which were not expected to contain pellets, prior to sampling. In Study Site 1, we used a 30.5-cm × 61.0-cm metal quadrat in conjunction with a metal scraper blade, which was custom-made to extend 1.3 cm below the metal quadrat (Fig. 2A). We placed the quadrat flat on the soil surface and used the scraper blade to remove the top 1.3 cm of soil. We collected the loosened soil on a sheet of plastic at one end of the quadrat and then placed it into a labeled 3.78-L plastic freezer bag.

Our original soil collection method (Fig. 2A) was not feasible in the grassland because the soil was tightly bound by roots. To address this issue we developed a new method to collect the grassland soil samples in Site 1, which used an 18-V cordless reciprocating saw fitted with a 15.2-cm-wide Spyder Scraper® blade (SM Products, LLC, Kansas City, MO, USA) to loosen the top 1.3 cm of the soil surface within each quadrat. We removed vertical organic matter with shears and then outlined the boundary of the quadrat by operating the saw blade in a vertical orientation (Fig. 2B). We removed the quadrat and used the scraper blade horizontally to carefully loosen our best approximation of 1.3 cm of surface soil (Fig. 2C). We collected the loosened soil by hand and placed it into labeled 3.78-L plastic freezer bags (Fig. 2D). We considered this new soil-sample collection method an improvement over the original method because the motorized blade made it possible to collect tightly bound surface soil, and we used it for all samples in Study Site 2.

We removed any remaining large pieces of organic material that could be easily separated and that would not contain a pellet, then measured the volume of each soil sample to the nearest 50 mL in a graduated cylinder. We measured soil volume to assess the accuracy and precision of the methods used to collect the samples, and determine whether land-use type influenced sample volume. We washed each sample with water through a 2.0-mm test sieve, which isolated all pellets

Figure 2. Soil sampling procedures to determine lead availability to ring-necked pheasants in Lyman County, South Dakota, USA, 2012–2013. A metal quadrat with associated scraper blade was used to collect 1.3 cm of surface soil for cropland and food plot samples in Study Site 1 (A). A cordless reciprocating saw fitted with a scraper blade was used to collect soil samples from grassland samples in Site 1 and all samples in Site 2 (B–D).

of shot size #9 and larger. We tested pellets for their magnetic nature to determine ferrous composition and then classified malleable, nonmagnetic pellets as lead similar to Bingham et al. (2015).

Prevalence of Spent Lead Shot in Gizzards

All gizzards analyzed in this study were from wild male pheasants. We did not sample females because they are not legal for hunter harvest. We did not use gizzards from captive-raised pheasants because pheasants are often harvested soon after release and each bird would have an unknown amount of time to ingest lead in the wild. Pen-raised pheasants released into the wild are required by South Dakota codified law (41:09:01:03) to have a hind toe clipped or have an identifiable mark through the nares from an antipecking device. We did not collect gizzards from pheasants with either a hind toe clip or an identifiable mark through the nares. We collected gizzards from hunter-harvested wild pheasants from Study Site 1 from 19 October to 4 December 2012 and from Study Site 2 from 15 September to 15 December 2013. In addition to gizzards collected from the study sites, we collected gizzards from hunter-harvested pheasants from outside the study sites from 10 October 2013 to 5 January 2014 from throughout South Dakota on nonpreserve lands. Gizzards were collected by South Dakota Department of Game, Fish and Parks staff during personal, off-duty hunting trips, and by commercial bird processing facilities. We recorded date of harvest, age of bird (young of year vs. adult; Bihrle 1993), and whether or not lead shot was legal for upland game hunting where the bird was harvested. We recorded age of bird for gizzards collected from Study Site 1, but not Study Site 2. We radiographed gizzards to determine the presence of metal pellets; we necropsied gizzards that contained pellets to determine whether the pellets were ingested or embedded, and whether the pellets were lead or nontoxic. We did not consider pellets to be ingested if there was an entrance wound in the gizzard without a corresponding exit wound.

Spatial Variables for Estimating Spent Lead Shot Abundance

We hypothesized that spent lead shot could vary by land-use type because tillage regime and vegetation communities varied by land-use type. Land-use type was a 3-level factor variable (grassland, cropland, and food plot) for Study Site 1 and a 2-level factor variable (grassland and food plot) for Study Site 2. Because each study site was repeatedly hunted in a systematic pattern, we suspected pellet density may vary by distance to hunter locations. Both study sites were typically hunted with 2 types of hunters simultaneously. One set of hunters would actively walk linear habitat, mostly food plots, and were referred to as walkers. Another set of hunters, referred to as blockers, would be stationary at the end of the linear habitat being hunted by walkers. Shooting intensity is often concentrated where the 2 sets of hunters converge, although shooting did occur throughout both study sites. Each property owner provided the typical locations of walkers and blockers. Distances to nearest walker and blocker were used as variables when modeling spatial variation in spent-lead-shot abundance. After completion of collecting soil samples from Study Site 2, it was discovered that a portion of the study site was influenced by a recreational shooting area. To account for this major source of additional spent lead shot, we designated an area within 230 m of the recreational shooting area as an area of influence because it represented a distance that was 30 m greater than the maximum trajectory of number 6 lead shot (National Rifle Association of America 1991). We used a buffer distance greater than the maximum trajectory of common target shooting loads because it is possible some target shooting occurred with larger shot sizes, which would have a longer maximum trajectory. Thus, Study Site 2 had a 2-level factor variable of recreational shooting area. We did not include a model that included only the recreational shooting area in the model selection table because our interest was to evaluate spatial patterns in lead deposition across the study site. The area influenced by the recreational shooting accounted for 1.8% of the upland habitat on the study site.

Data Analysis

We conducted all statistical analyses using Program R (version 2.5.12; R Development Core Team 2012). We used a generalized linear mixed model with a negative binomial distribution to evaluate the influence of land-use type and hunting pattern on the density and distribution of spent lead shot in soil (Venables and Ripley 2002). The candidate model set for each study site consisted of all combinations of the limited number of predictor variables. We modeled each site separately in lieu of a single model set with a fixed effect for study area for several reasons. First, data were collected during different years so the effect of study site was confounded with year. Secondly, soil samples were not collected in the same manner in each site. Finally, because walker and blocker locations were determined by the preserve operators, they are inherently subjective and should be analyzed separately. We ranked each model from most to least support given the data using Akaike's Information Criterion corrected for small sample size (AIC_c; Burnham and Anderson 2002). We used Akaike weights (w_i) as an indication of support for each model and determining relative importance of variables within models. We reported coefficients and 95% confidence intervals for models within 2 ΔAIC_c of the top-ranked model for each study site. We used the raster calculator tool in ArcMap 10.0 (ArcGIS 10.0, ESRI) to create maps with weighted estimates of the mean predicted value of spent lead pellets per hectare based on the reported model set. We weighted model predictions by w_i, which was scaled to sum to 1 for each model set.

We constructed Wilson Score confidence intervals for estimates of prevalence rates of ingested spent lead shot. We used Fisher's exact test to determine differences in ingested-lead-shot prevalence between young-of-year birds and adults, between birds harvested on shooting preserves and nonpreserve areas, and between study sites. We used a Shapiro–Wilk test to determine whether our soil-sample volume data were normally distributed. We used a Kruskal–Wallis test to test for differences in soil sample volume

Table 1. Negative binomial-regression models predicting lead shot availability to ring-necked pheasants in Study Site 1 located in Lyman County, South Dakota, USA, 2012.

Model	K[a]	AIC_c[b]	ΔAIC_c[c]	w_i[d]
Distance blocker + distance walker + land-use type	5	635.55	0.00	0.43
Distance blocker	2	636.59	1.04	0.25
Distance blocker + land-use type	4	636.99	1.44	0.21
Distance blocker + distance walker	3	638.30	2.75	0.11
Distance walker + land-use type	4	647.63	12.08	0.00
Intercept only	1	648.21	12.66	0.00
Distance walker	2	649.34	13.79	0.00
Land-use type	3	649.95	14.40	0.00

[a] No. of parameters.
[b] Akaike's Information Criterion adjusted for small sample size.
[c] Difference of each model's AIC_c from that of the highest ranked model.
[d] Akaike wt.

among the 3 land-use types of Study Site 1. We used a Mann–Whitney *U*-test to test for differences in soil sample volume between the 2 land-use types of Study Site 2. If a difference among groups was detected (Site 1 only), we conducted nonparametric multiple tests for all-pairs comparisons (Gao et al. 2008, Konietschke 2012). We determined statistical significant using $\alpha = 0.05$.

RESULTS

Lead Availability

For Site 1, model-averaged predictions were based on 3 models within 2 ΔAIC_c of the top-ranked model and included the variables distance to blocker, distance to walker, and land-use type (Table 1). Spent lead shot declined as distance from blocker and walker locations increased, and was less abundant in food plots than cropland or grassland (Table 2). Distance to blocker was present in all models within 2 ΔAIC_c of the top model and had a coefficient twice as large as the variable distance to walker, which was only present in 1 competitive model. Summed model weights for models within 2 ΔAIC_c of the top model were greatest for models containing distance to blocker (0.89), followed by land-use type (0.64), and finally distance to walker (0.43). Predicted spent-lead-shot pellet density ranged from approximately 1,600 to 15,000 pellets/ha and was most influenced by distance to blocker and land-use type (Fig. 3). Overall lead availability (intercept-only model) was 6,187 (95% CI = 5,018–7,553) pellets/ha, but this model that assumed homogenous lead deposition was among our poorest performing models (Table 1).

For the second site, model-averaged predictions were based on 4 models within 2 ΔAIC_c of the top-ranked model and included the variables distance to blocker, distance to walker, and land-use type (Table 3). Spent lead shot declined as distance to blocker increased and distance to walker decreased (Table 4). Distance to blocker was present in all models within 2 ΔAIC_c of the top model while distance to walker was only present in 2 competitive models. The 95% confidence interval for the coefficient of distance to walker overlapped zero in both reported models, which suggests a spurious effect. Spent-lead-shot abundance was predicted to

Table 2. Coefficient (β) estimates and 95% confidence intervals for 3 models within 2 ΔAIC_c of the top-ranked model for estimating lead availability to ring-necked pheasants in Study Site 1 located in Lyman County, South Dakota, USA, 2012.

Covariate	β-estimate	95% CI Lower	95% CI Upper
Intercept	−0.842	−1.395	−0.309
Distance blocker	−0.002	−0.003	−0.001[a]
Land-use type			
Grass	−0.276	−0.777	0.222
Food plot	−0.781	−1.377	−0.188
Distance walker	−0.004	−0.008	0.000[a]
Intercept	−1.369	−1.722	−1.032
Distance blocker	−0.002	−0.002	−0.001[a]
Intercept	−1.151	−1.607	−0.718
Distance blocker	−0.002	−0.003	−0.001[a]
Land-use type			
Grass	−0.116	−0.589	0.354
Food plot	−0.480	−1.000	0.027

[a] Rounded.

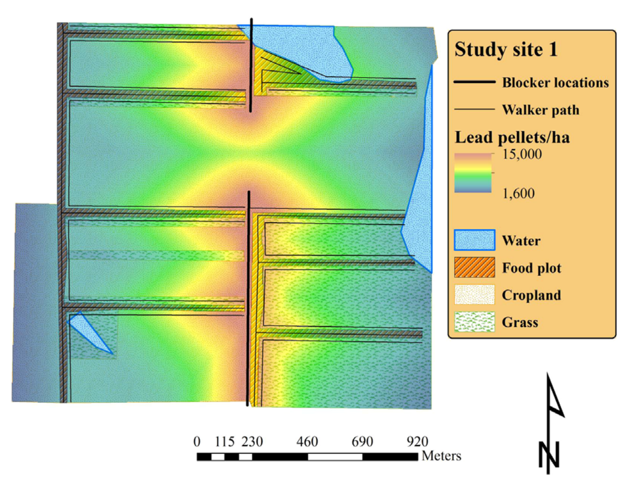

Figure 3. Estimated lead pellets per hectare available to ring-necked pheasants in Study Site 1, Lyman County, South Dakota, USA, 2012. Predictions are based on the weighted model-averaged predictions from models within 2 ΔAIC_c of the top model.

Table 3. Negative binomial-regression models predicting spent-lead-shot availability to ring-necked pheasants in Study Site 2 located in Lyman County, South Dakota, USA, 2013. All models except the intercept-only model include the 2-level factor variable target shooting area.

Model	K[a]	AIC_c[b]	ΔAIC_c[c]	w_i[d]
Distance blocker + distance walker	4	1,112.33	0.00	0.39
Distance blocker	3	1,112.88	0.56	0.29
Distance blocker + land-use type	4	1,113.96	1.63	0.17
Distance blocker + distance walker + land-use type	5	1,114.28	1.95	0.15
Land-use type	3	1,140.97	28.64	0.00
Distance walker	3	1,142.75	30.43	0.00
Distance walker + land-use type	4	1,142.77	30.44	0.00
Intercept only	1	1,214.63	102.30	0.00

[a] No. of parameters.
[b] Akaike's Information Criterion adjusted for small sample size.
[c] Difference of each model's AIC_c from that of the highest ranked model.
[d] Akaike wt.

be greater in grass than food plot, but this variable was only present in 2 models, both of which had coefficients with 95% confidence intervals that overlapped zero. Summed model weights for models within 2 ΔAIC_c of the top model were greatest for models containing distance to blocker (1.00), followed by distance to walker (0.54), and finally land-use type (0.32). Predicted spent-lead-shot pellet density ranged from approximately 1,100 to 42,000 pellets/ha and was most influenced by distance to blocker (Fig. 4). Overall lead availability (intercept-only model) outside of the recreation shooting area was 16,868 (95% CI = 13,891–20,492) pellets/ha, but this model that assumed homogenous lead deposition was our poorest performing model (Table 3).

Soil Volume

We used nonparametric tests to test for differences in soil volume between and among land-use types because our data were not normally distributed ($P \leq 0.01$ for all groups). The soil sample volume varied ($H = 114.52$, $P \leq 0.001$) by land-use type for Study Site 1 and was greater in grassland versus cropland ($T = 10.54$, $P \leq 0.001$) or food plot ($T = 10.06$, $P \leq 0.001$; Fig. 5); samples from cropland and food plots did not differ ($T = 0.41$, $P = 0.67$). For Study Site 2, the soil sample volume varied ($U = 79{,}038$, $P \leq 0.001$) by land-use type and was greater in samples collected in the food plots than grassland (Fig. 6).

Table 4. Coefficient (β) estimates and 95% confidence intervals for 4 models within 2 ΔAIC_c of the top-ranked model for estimating lead availability to ring-necked pheasants in Study Site 2 located in Lyman County, South Dakota, USA, 2013.

		95% CI	
Covariate	**β-estimate**	**Lower**	**Upper**
Intercept	−0.208	−0.585	0.180
Distance blocker	−0.005	−0.007	−0.003
Distance walker	0.003	−0.006	0.007
RSA[a]	2.912	2.053	4.027
Intercept	−0.099	−0.454	0.268
Distance blocker	−0.005	−0.006	−0.003
RSA	2.825	1.968	3.944
Intercept	−0.243	−0.696	0.223
Distance blocker	−0.005	−0.006	−0.003
Land-use type			
Grass	0.198	−0.199	0.594
RSA	2.833	1.979	3.945
Intercept	−0.246	−0.699	0.220
Distance blocker	−0.005	−0.007	−0.003
Distance walker	0.003	−0.001	0.007
Land-use type			
Grass	0.066	−0.380	0.509
RSA	2.907	2.048	4.023

[a] Recreational shooting area.

Lead Pellet Ingestion

For the first study site, 9 of 167 (5.4%, 95% CI = 2.9–9.9%) gizzards contained ≥ 1 ingested lead pellet ($\bar{x} = 1.55$, SE = 0.34, range = 1–4). We detected no difference ($P = 0.30$) in ingested-lead-pellet prevalence rates between adults and young-of-year birds. For the second study site, 17 of 493 (3.4%, 95% CI = 2.2–5.5%) gizzards contained ≥ 1 ingested lead pellet ($\bar{x} = 3.24$, SE = 0.97, range = 1–13). The prevalence rate of ingested lead pellets differed ($P \leq 0.001$) between Study Sites 1 and 2.

We collected gizzards from 1,301 pheasants throughout South Dakota from nonpreserve areas. We did not determine age of bird for 46 birds, and for 35 birds we did not know whether they were harvested from an area where lead shot

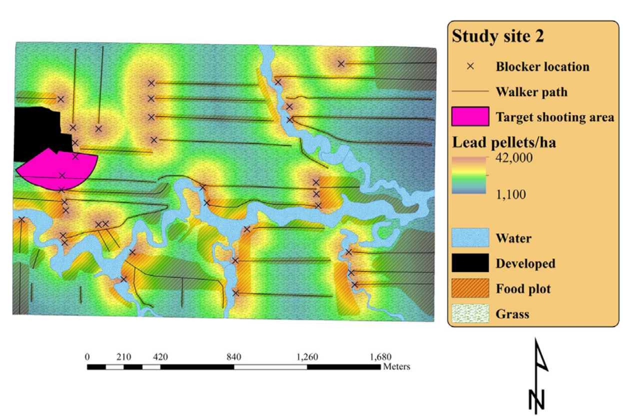

Figure 4. Estimated lead pellets per hectare available to ring-necked pheasants in Study Site 2, Lyman County, South Dakota, USA, 2013. Predictions are based on the weighted model-averaged predictions from models within 2 ΔAIC_c of the top model.

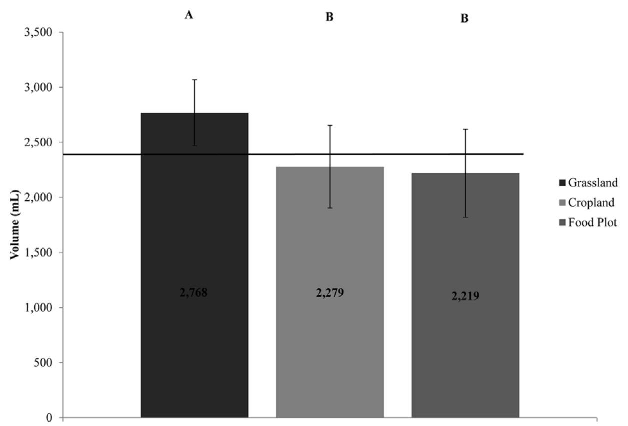

Figure 5. Mean soil-sample volume ± inter-quartile range for studying lead availability to ring-necked pheasants in Study Site 1, Lyman County, South Dakota, USA, 2012. Pairs with different letters differed ($P < 0.05$) according to multiple tests for all-pairs comparison (Gao et al. 2008). The horizontal black line indicates the target soil-sample volume of 2,419 mL according to a 30.5 × 61.0 × 1.3-cm sample.

was legal for upland game hunting. We excluded those samples from their applicable tests. Ten of 1,301 (0.8%, 95% CI = 0.4–1.4%) gizzards contained ≥1 ingested lead pellet ($\bar{x} = 2.4$, SE = 0.98, range = 1–11). There was no difference ($P = 0.75$) in prevalence rates of ingested lead shot between adult and young-of-year pheasants. We did not have an adequate sample size to test for differences in lead shot ingestion between areas where lead shot was allowed and areas where nontoxic shot was required. However, 1 of 150 (0.7%, 95% CI = 0.1–3.7%) gizzards collected from areas where lead shot was not allowed had a single ingested lead pellet. Nine of 1,116 (0.8%, 95% CI = 0.4–1.5%) gizzards

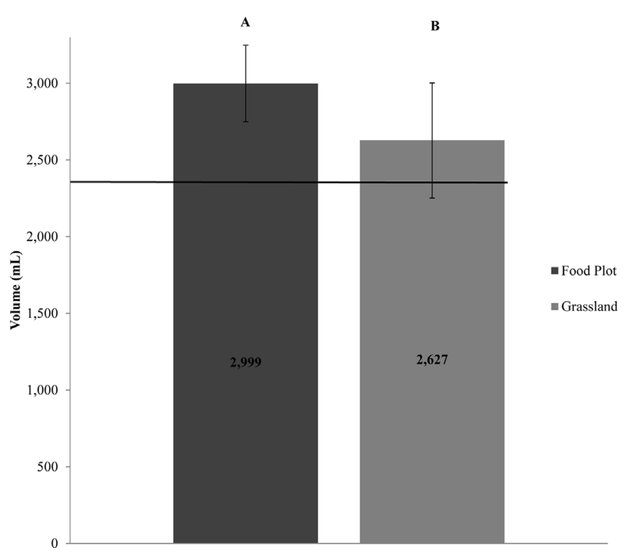

Figure 6. Mean volume ± inter-quartile range for studying lead availability to ring-necked pheasants in Study Site 2 in Lyman County, South Dakota, USA, 2013. Pairs with different letters differed ($P < 0.05$) according to multiple tests for all-pairs comparison (Gao et al. 2008). The horizontal black line indicates the target soil-sample volume of 2,419 mL according to a 30.5 × 61.0 × 1.3-cm sample.

collected from areas where lead shot was allowed had ≥1 ingested lead pellet(s) in the gizzard ($\bar{x} = 2.56$, SE = 1.08, range = 1–11). Prevalence of ingested lead shot was 4.9 times greater ($P \leq 0.001$) for birds harvested on shooting preserves (3.9%, 95% CI = 2.7–5.7%) when compared with non-preserve areas. The average number of pellets ingested for gizzards with ≥1 lead pellet was 2.65 (SE = 0.66, range = 1–13) for both preserve sites combined.

DISCUSSION

As expected, spent-lead-shot availability was strongly influenced by the systematic hunting pattern of each study site. Lead was highly concentrated near blocker locations in each study site. Spent shot was more abundant near walker locations for Study Site 1, but had limited influence for Study Site 2. For both study sites, food plots were predicted to have less lead than the other land-use types when the other spatial variables were equal. However, because most food plots were in close proximity to walker and blocker locations, portions of food plots still had substantial amounts of lead. This may increase the lead exposure risk to pheasants because food plots are foraging areas and contain bare ground where pheasants could easily mistake a lead pellet for a seed or grit. Our estimates of available lead pellets per hectare for areas used exclusively for hunting were well within the lower range reported for upland habitats in the literature (27,100–107,600, Lewis and Legler [1968]; 23,500–73,200, Anderson [1986]; 0–83,900, Castrale [1989]; 167,600–860,200, Best et al. [1992]; 353–6,342, Schulz et al. [2002]; and 67,813, Douglass [2011]).

We are aware of only a single study that estimated spent-lead-shot abundance in an area used for resident upland-game hunting (Holdner et al. 2004). However, their results are difficult to compare to ours because their estimates were based on samples of the top 10 cm of soil and very small sample sizes. Our greatest estimate of spent lead shot was located in an area used for recreational target shooting, which is similar to Holdner et al. (2004). Holdner et al. (2004) found spent-lead-shot abundance ranging from 0 pellets/ha to 205,100,000 pellets/ha based on only 14 samples. For fields used for hunting only, they reported 422,222 lead pellets/ha based on 9 samples.

More lead was available in Study Site 2, which had lower annual shooting intensity than Study Site 1, but had been a licensed shooting preserve for much longer. This suggests deposited lead shot accumulates over time and may be available for many years. Our study sites had alkali silty-clay soils with a pH of 6.6–8.4 (Schumacher 1987), which would mostly keep lead in a nonsoluble form (Casas and Sordo 2006). Other studies of managed mourning dove fields found the most recent year of gunning disproportionately contributed to the available lead shot, possibly due to tillage or pellets naturally moving down the soil profile (Lewis and Legler 1968, Anderson 1986, Castrale 1989, Best et al. 1992, Schulz et al. 2002, Douglass 2011). Our food plots had less lead and were the only tilled areas, which suggests annual tillage may have reduce lead availability in our study sites. The nonacidic, silty-clay soils of our study sites appear to have characteristics favorable for accumulation of lead at the surface over time, especially when untilled.

Past studies have assumed homogenous distribution of lead shot throughout study plots. Our intercept-only models, which assume homogenous shot distribution, were among our poorest performing models. Not accounting for spatial patterns in lead deposition could explain why estimates of available lead shot have yielded unexpected results. Schulz et al. (2002) estimated 1,086,275 lead pellets/ha were deposited within managed dove fields in Missouri, USA, estimated from shots fired by hunters. However, soil-sampling estimates (top 1.0 cm) only accounted for 0.6% (6,342 pellets/ha) of the estimated pellets. Observer bias, soil characteristics, and differing amounts of vegetation material on the soil surface were considered possible explanations (Schulz et al. 2002). It may be possible that lead shot deposition was highly clumped in relation to hunter locations and reporting lead shot abundance as if it were homogenously distributed biased estimates low. Schulz et al. (2002) reported 2,403 doves were harvested on the 36 ha of study fields, which was a much greater shooting intensity than our study sites, yet our estimates of spent lead shot were typically greater.

Our methods for collecting soil samples produced sample volumes that were reasonably close to the desired volume from a perfect 30.5 cm × 61.0 cm × 1.3 cm sample. For Study Site 1, samples from the grassland were 9.6% larger than desired and may have been caused by skimming >1.3 cm of soil with the reciprocating saw. For Site 2, samples from both grassland and food plots were larger (14% and 24%, respectively) than desired and also collected using the reciprocating saw. We suspect the sample volumes were slightly inflated because the soil was aerated during the collection process. Samples collected using the metal quadrat and scraper blade produced smaller than desired samples, which may have been caused by uneven soil surfaces resulting in <1.3 cm of soil being collected. We recommend the use of the reciprocating saw for collecting soil samples in both sod-bound and cropped soils because of its ease of use and reasonable sample-size volume collected in comparison to the desired amount.

We are unaware of any other study that examined the prevalence rate of ingested lead pellets for exclusively wild male ring-necked pheasants. Prevalence rates of ingested lead pellets in the gizzards of hunter-harvested pheasants from both shooting preserve study sites were slightly larger than estimates reported for male and female pheasants from shooting estates in Great Britain (3.0%; Butler et al. 2005). Unlike our samples, those samples likely contained a mixture of wild and captive-raised birds and collected throughout the spring, autumn, and winter (Butler et al. 2005). Our estimates were less than those reported (34%; captive-raised pheasants) from an intensely hunted island in Canada where intensive target shooting also may have contributed to the accumulation of spent lead shot (Kreager et al. 2008). In contrast, our estimates for pellet ingestion were greater than scaled quail (0.4%) and northern bobwhite (1.8%) in New Mexico, USA (Best et al. 1992), but less than for chukars (8.3%) in Utah, USA (Bingham et al. 2015).

Our estimates of prevalence rates of ingested lead shot from both study sites were greater than for mourning doves collected in the United States, including Tennessee (1.2%; Lewis and Legler 1968), Indiana (2.5%; Castrale 1991), New Mexico (0.2%; Best et al. 1992), Missouri (0.3%; Schulz et al. 2002), or Maryland, Virginia, North Carolina and South Carolina combined (2.4%; Kendall and Scanlon 1979). Locke and Bagley (1967) found 6.5% of mourning doves had ingested lead shot, but only 62 birds were collected. Mourning doves are highly susceptible to the acute effects of lead poisoning, which makes them unavailable for harvest; this may contribute to the relatively low prevalence of ingested lead shot in hunter-harvested mourning doves (Kendall et al. 1996, Schulz et al. 2002). Limited research on the acute effects of lead pellet ingestion on pheasants suggests they are not as vulnerable to the acute effects and that observed prevalence rates of ingested lead pellets may be closer to the true ingestion rate than for mourning doves (Gasparik et al. 2012, Runia and Solem 2014). This may partially explain why our samples of pheasants had greater prevalence rates of ingested lead pellets, whereas lead pellet availability was generally greater for areas where mourning doves were sampled.

Grain sorghum was a food-plot crop on both of our study sites. Although we did not conduct a diet analysis on harvested birds, grain sorghum seed was a common food item in the gizzards we necropsied. The dark-colored, round seeds are slightly larger than a lead pellet, but could be similar enough in appearance to be confused with lead pellets. This could have contributed to the relatively high prevalence rate of ingested lead on our study sites. Bingham (2011) found that captive chukars with a search image for seeds similar in shape, size, and color to lead pellets increased the ingestion rate of lead pellets.

The prevalence rate of ingested lead pellets was much less for birds collected throughout South Dakota from nonshooting preserve areas than those collected from the very heavily hunted study sites. Although we did not estimate the lead shot availability throughout South Dakota, we speculate that it is less than at our shooting preserve study sites. Our results suggest that ingestion of spent lead shot by pheasants may increase with pellet availability. Bingham (2011) documented greater lead ingestion rates for chukars harvested near water sources where lead availability was greatest. Interestingly, this relationship has not been demonstrated for mourning doves in the wild when comparing studies of pellet ingestion and availability, nor has this relationship been established in captive studies (Schulz et al. 2007, Plautz et al. 2011). The relationship between availability and prevalence of ingested lead shot may be easier to detect in pheasants because they do not exhibit large movements to foraging areas as mourning doves do. Spent-lead-pellet abundance was greater for Study Site 1 than Study Site 2, but prevalence of ingested lead pellets showed an opposite pattern, reaffirming that uncertainty exists in the relationship between lead shot availability and ingestion by upland foraging birds. Within our study sites, small portions of the landscape contained a majority of the lead pellets. It may be difficult to correlate lead shot availability to ingestion rate because the space-use patterns of pheasants within the study sites are unknown and would be difficult to determine. However, we have evidence that lead shot ingestion may occur at a lower rate on nonshooting preserve areas where lead shot deposition is presumed to be less intense.

More research is needed to determine whether lead shot ingestion occurs at a lower rate on areas where nontoxic shot is now required for hunting. Future research should assess the effects of lead shot ingestion on the survival and reproduction of free-ranging wild pheasants to increase knowledge of the overall impacts of lead exposure to wild pheasants. Information is also needed about lead ingestion rates outside of the pheasant hunting season to determine whether temporal patterns exist. Overall lead exposure in pheasants could be greater than we report because elevated liver lead levels have been documented in chukars without ingested lead pellets (Bingham et al. 2015).

MANAGEMENT IMPLICATIONS

Spent lead shot is available and ingested by wild male ring-necked pheasants in South Dakota. Managers should be aware that ingestion of lead shot likely occurs at a greater rate within licensed shooting preserves where hunting intensity is particularly high. Although the consequences of ingesting lead shot are unknown for wild pheasants, there is potential for lead poisoning. Annual tillage could be used as a management tool to reduce lead availability. This may be particularly important for food plots where pheasants often forage on sorghum seeds that resemble lead shot. Alternatively, the use of nontoxic shot for hunting could reduce lead availability over time, although the process could be slow for untilled lands.

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