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Resource use by American black bears in suburbia: a landholder step selection approach

Farshid S. Ahrestani, Pennsylvania Cooperative Fish and Wildlife Research Unit, 436 Forest Resources Building, The Pennsylvania State University, University Park, PA 16802, USA

Mark A. Ternent, Bureau of Wildlife Management Pennsylvania Game Commission, 2001 Elmerton Avenue, Harrisburg, PA 17110, USA

Matthew J. Lovallo, Bureau of Wildlife Management Pennsylvania Game Commission, 2001 Elmerton Avenue, Harrisburg, PA 17110, USA

W. David Walter, U.S. Geological Survey, Pennsylvania Cooperative Fish and Wildlife Research Unit, 403 Forest Resources Building, The Pennsylvania State University, University Park, PA 16802, USA *wdw12@psu.edu*

Abstract: Range expansion of American black bears (*Ursus americanus;* bear) and residential development have increased the bear presence in suburbia. Suburban landscapes exhibiting patchworks of variable-sized parcels and habitats and owned by landowners with diverse values can create large areas of suitable habitats with limited public access. These landscapes may limit the effectiveness of hunting as a traditional bear population management tool. Managers require better information regarding landowner attitudes about hunting before implementing harvest regulations intended to mitigate conflicts in suburban areas. To address this need, in 2013, we surveyed landowners to identify properties that allowed bear hunting in 3 suburban areas of Pennsylvania, USA where bear sightings or human–bear conflicts have increased. We then used location data obtained for 29 bears equipped with global positioning system transmitters from 2010 to 2012 to model their resource selection in the study area. We assessed the influence of hunting access, housing density, land cover, and topographic variables on radio-marked black bears monitored 10 days before, during, and after the bear hunting season. We found that resource selection of radio-marked bears was similar for all 3 periods and bears selected for forested land in all 3 seasons and herbaceous cover in the pre-hunting and hunting periods. Resource selection by bears was not influenced by hunting access in the pre-hunting and hunting periods. For the post-hunting period, lands closed to hunting had support as the second-best model. All of the radio-marked bears in our study were vulnerable to harvest. However, they did not change resource selection during the hunting season, nor did they avoid areas open to hunting. Integrating human dimension data with bear habitat use studies, especially in suburban landscapes, has the potential to address bear space use and population management needs often overlooked by traditional research designs.

Key words: American black bear, global positioning system, housing density, human–bear conflict, hunting, landholder, Pennsylvania, *Ursus americanus*

Black bear (*Ursus americanus*; bear; Figure 1) populations are increasing in many U.S. jurisdictions (Hristienko and McDonald 2007). Concomitantly, suburban development is also encroaching and fragmenting historical bear habitat (Tri et al. 2016, Loosen et al. 2019). The increasing overlap between humans and bears has increased the potential for conflict (Conover 2001, Woodroffe et al. 2005). Major contributors to these conflicts included increased available food sources such as crops, apiaries, bird feeders, livestock, and refuse in suburban landscapes (Beckmann and Berger 2003, Merkle et al. 2011). Managing human–bear interactions in suburban areas has now become commonplace and is integral to the overall management of bear populations at state and national levels (Hopkins et al. 2010).

Suburban lands pose a challenge to managing wildlife populations, as only a limited number of strategies can be employed in areas with human residents (Merkle et al. 2011). Regulated hunting has been recommended as a tool for reducing some types of human–bear conflicts (Conover 2001, Ziegltrum 2004, Treves et al. 2010). Harvest vulnerability may vary with forest composition, hunter density, and snow cover and by sex and age class (Diefenbach et al. 2004, Malcolm and Van Deelen 2010). Bears living near residential areas may have low susceptibility to harvest

Figure 1. American black bears (*Ursus americanus*) in a tree within a study site in Centre County, Pennsylvania, USA (*photo courtesy of M. Marshall*).

because of limited hunter access (Wolgast et al. 2005). Harvest vulnerability for bears living near residential areas may further be influenced by other factors such as property size, local ordinances that prohibit discharge of firearms, and laws that prevent hunting near occupied buildings or from roadways.

Movements and resource use by large mammals may be affected by hunter activity (Millspaugh et al. 2000, Bowman 2012, Lovely et al. 2013). In response to increased hunter activity, large mammals may increase their use of dense vegetation (Bowman 2012), refugia offered by private or public property closed to hunting (Burcham et al. 1999), and relocate to areas where disturbance or hunter access is further limited (Kilpatrick and Lima 1999). For example, white-tailed deer (*Odocoileus virginianus*) vacated a residential neighborhood within days of the onset of a controlled hunt (Kilpatrick and Lima 1999). Bears also have adjusted their use of risky habitats near roads in response to initiation of hunting seasons (Stillfried et al. 2015). Previous research in a variety of landscapes has documented that the availability of land open to hunting is of equal or greater importance than hunting seasons due to the aforementioned shifts in behavior of large mammals to disturbance (Kilpatrick and Lima 1999, Bowman 2012, Stillfried et al. 2015).

Multiple studies have investigated bear resource use in diverse landscapes in the United States. In Michigan, USA, bears selected for forested wetlands while avoiding roads and developed lands (Carter et al. 2010). In North Carolina, USA, bears selected for habitats that were both steeper and at lower elevations (Powell and Mitchell 1998). In Washington, USA, bears selected for riparian and deciduous forests, meadows, and shrub-fields (Lyons et al. 2003). In Montana, USA, bears selected areas with intermediate housing densities close to forested patches and watercourses (Merkle et al. 2011). These studies collectively highlight the importance of both anthropogenic and environmental covariates when evaluating bear

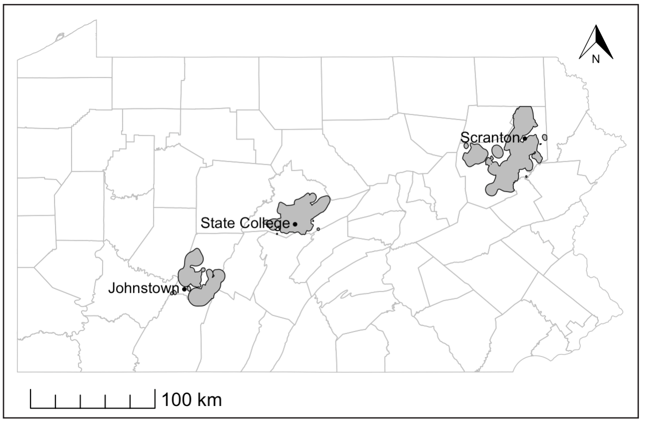

Figure 2. County outlines where tax parcel data was available statewide for surveys to be mailed to landowners in Pennsylvania, USA in 2013. Three study areas (gray polygons) were created using 95% fixed kernel density estimator smoothed by an ad hoc reference bandwidth for all American black bear (*Ursus americanus*) locations in Cambria County (903 km^2), Centre County (952 km^2), and Lackawanna/Luzerne/Wyoming counties (1,823 km^2).

resource selection. In suburban landscapes, hunter access to areas occupied by bears may be an equally important covariate.

To our knowledge, no prior study has integrated human dimension surveys with bear resource selection modeling. Our goal was to determine if bear use of suburban landscapes before, during, and after the hunting season could provide an index of harvest susceptibility. Unlike previous bear resource selection analyses, we included a spatial layer generated from landowner responses regarding hunting access as one of our environmental and anthropogenic covariates. We attempted to identify if different patterns of bear resource selection existed and if they could be attributed to hunting access. To our knowledge, this study is the first to incorporate landholder survey data into a framework for modeling resource selection by radio-marked bears in suburbia.

Study area

Our study was conducted in 3 study sites located across the state of Pennsylvania, USA, 1 study site each in the eastern, central, and south-central regions of the state (Figure 2). These sites were chosen because they were known to have both growing bear populations and increasing complaints of bear (Tri et al. 2017). The study sites consisted of a mix of public and privately owned properties. The vegetation in all 3 sites was similar, broadly characterized by forested and shrublands interspersed with cultivation and grasslands.

Eastern. This study area was 1,823 km^2, near the cities of Scranton and Wilkes-Barre, and included portions of Luzerne, Wyoming, and Lackawanna counties. The mean human density was 38.7 km^{-2} and was 81% forested/shrubs, 14% urban-suburban, and 5% cultivated/grasses.

Central. This study area was 952 km^2 and located in Centre County. The mean human density was 17.9 km^{-2} and had the highest (23%) proportion of cultivated/grasses, the least amount (71%) of forested/shrubs, and 6% was urban-suburban.

South-Central. This study area was the smallest of the 3 sites (903 km^2), lies within Cambria County, had a mean human density of 11.6 km^{-2}, and was the least developed with only 3% being urban-suburban, 6% cultivated/grasses, and 91% forested/shrubs. The climate in all 3 sites was similar; mean annual rainfall was 90–100 cm, and mean annual number of sunny days was ~170 days.

Methods

Because we were interested in determining the contribution of hunter access and resource selection to bear harvest vulnerability, we evaluated separate step selection functions (SSFs) over 3 10-day periods in succession: (1) pre-hunting as 10 days prior to opening day of hunting season, (2) hunting as the 10 days open to the hunting of bears every year, and (3) post-hunting as 10 days after the bear hunting season closed. The exact dates of all 3 periods combined varied by year: November 5–December 4, 2010; November 4–December 3, 2011; and November 2–December 1, 2012.

Bear location data

We captured bears opportunistically in all 3 study sites from 2009 to 2011 using barrel or culvert-style traps at locations in areas bears were observed or human–bear conflicts reported. All capture and handling methods were carried out by trained Pennsylvania Game Commission staff in strict accordance of SOP 40.9 (Standard Operating Procedures of the Pennsylvania Game Commission) that follows guidelines outlined by the American Society of Mammalogists (Sikes et al. 2011).

After capture, we deployed global positioning system (GPS) collars (Vectronics, Berlin, Germany; Lotek, New Market, Ontario, Canada) on bears >45 kg and released them as soon as possible close to the capture sites. For each year in the study, 2010–2012, we programmed collars to record bear locations every hour between 0600 and 1800 hours—the time when hunting typically occurs—during our annual 30-day study period. To ensure we captured a meaningful representation of resource areas selected by individual bears, we only included bears for which we had ≥50 locations in each of the 3 study periods in our analyses.

Anthropogenic covariates

We used all bear GPS locations in each study site for the duration of the study to determine an overall use area for each study site using 95% fixed kernel density estimator smoothed by an ad hoc reference bandwidth (Figure 2). We did this to estimate the overall area encompassed by each subpopulation of bears to administer landowner surveys. To ensure we sampled an adequate area in each study site, we used the liberal fixed kernel density estimator as it is known to overestimate the size of a home range (Walter et al. 2011).

We estimated the anthropogenic variable housing density using a reclassification of census block group (Theobald et al. 1997, Theobald 2001). We preferred housing unit density to metropolitan statistical units (i.e., census block groups) because it provided a better measure of urban density and sprawl compared to simple measures of human density (Theobald et al. 1997, Theobald 2001). The 3 housing density categories we defined were: suburban (≥0.062 units per ha); rural (<0.062 units per ha); and undeveloped (no housing structures; Theobald 2001). We created a 30 × 30-m raster of our study sites categorized into the different housing density levels and included this layer as "housing density."

Within the polygons of the overall home ranges of bear in each study area, we solicited and received tax parcel data from the Geographic Information System departments of all overlapping government counties. We placed a 4-ha minimum parcel size threshold on survey recipients for 3 reasons: (1) hunting on parcels <4 ha were unlikely to occur because of hunting restrictions near roads and structures, (2) it was logistically not possible to survey every landowner in the study area, and (3) detecting bear use of parcels smaller than 4 ha was less likely with our collection schedule and resolution of spatial data. We spatially linked multiple parcels owned by the same individual so that only 1 survey form was received by the landowner, and their response was linked to all parcels owned by that landowner.

We did not send surveys to landowners for parcels of land on which we knew hunting was allowed, such as public land and parcels included in hunter access programs, thus coding them as huntable in analysis. The final mailing list was derived after correcting or omitting incorrect and undeliverable addresses. The survey was implemented by the Pennsylvania Game Commission using the tailored design method to maximize response rate (Dillman et al. 2009).

Of the 6,746 landowners with properties ≥4 ha surveyed across all 3 study areas, we received 4,760 responses that corresponded to a 70.6% response rate. There were 53 responses that were unusable due to missing or unreadable data, which resulted in 4,707 responses for analysis. We then created 4 categories for the huntable covariate in properties ≥4 ha: (1) publicly-owned or privately-owned property that permitted bear hunting; (2) publicly-owned or privately-owned property that did not permit bear hunting; (3) privately owned properties for which there was no response from owners; and (4) privately-owned properties that we did not survey (e.g., wrong address in landowner records or were <4 ha). We created a raster of the parcel data based on these 4 huntable categories and included this layer as "huntable" in our models.

Environmental covariates

We identified 3 topographical covariates that have been shown to be important for bears that included 30 × 30-m rasters of elevation, slope, and aspect (Clark et al. 1993, Powell and Mitchell 1998, Lee and Vaughan 2003, Lyons et al. 2003). We obtained an elevation layer from the United States Department of Agriculture (USDA) Natural Resources Conservation Services and National Cartography and Geospatial Center; both slope and aspect data were derived from this elevation layer (USDA 1998). Elevation and slope were continuous variables, while aspect was classified as either North (315–45°), East (45–135°), South (135–225°), or West (225–315°).

Because we focused on suburban lands, we obtained land cover from the National Land Cover Data raster data layer (Homer et al. 2015), then reclassified land cover across our study areas in the following 3 categories: (1) developed, which included either urban or suburban land; (2) forest, which included all forest types, shrub, and woody wetlands; and (3) grass, which included agriculture, pasture hay, graminoids, and herbaceous vegetation. We created 30 × 30-m rasters of the land cover data based on these 3 categories and included this layer as "land cover." All raster data for anthropogenic and environmental covariates was rescaled to a resolution of 30 × 30 m to be similar to the National Land Cover Database (Homer et al. 2015).

Statistical analysis

We examined continuous covariates for collinearity in each of the 3 periods: pre-hunting, hunting, and post-hunting. A correlation test found that the 2 continuous variables, elevation and slope, were not correlated in any of the periods, and estimates of η2 from ANCOVA tests (Maher et al. 2013) confirmed that neither elevation nor slope were related to the other categorical covariates (i.e., aspect, land cover, housing density, and huntable). We standardized the values of the 2 continuous covariates, slope and elevation. All the other covariates were categorical in nature.

We used SSF, first reported by Fortin et al. (2005), that were similar to traditional resource selection functions (RSFs) with a key difference that SSFs connect consecutive locations along the movement path of an animal. This differs from traditional RSFs that did not incorporate a temporal component to location data nor a movement parameter in resource selection. Relatedly, SSF and traditional RSF approaches differ in how available resource data are identified. With RSF, available resources are typically selected from a polygon that encompasses all recorded locations (Manly et al. 2002), whereas in SSF, available resources are identified at each recorded step and are chosen from the distributions of step angles and lengths along the travel path to determine which habitat/environmental covariates are relevant to a species (Thurfjell et al. 2014). The used locations (i.e., locations where bears were recorded by their GPS collars) were coded as 1 and represented steps for our SSF, and each used location was paired with 10 *available* locations that were randomly selected using the *rdSteps* function of the *hab* library (Basille 2015) in the R statistical environment (R Core Development Team 2018). The function *rdSteps* selects points from a distribution of lengths and turning angles of recorded steps, which in our case was from all bears in the study.

To ensure that no improbable step lengths were used, we omitted any GPS locations from the dataset that were recorded after an inter-step interval >1 hour (1 hour was the time interval for which we derived location fixes of bears [i.e., after any location fixes were missed during data collection]). Each available location was coded as 0. We then used mixed conditional logit models to determine which covariates were relevant and tested these models using functions from the *mclogit* library (Elff 2018) in the R statistical environment (R Core Development Team 2018).

With 7 independent variables, we had the choice of testing 127 models. We chose to evaluate a set of only 8 candidate models for each of the 3 study periods because each model represented a meaningful *a priori* combination of covariates that may influence resource selection in our study area (Table 1). We combined data across study areas and years for the SSF analysis because of our limited sample sizes, homogeneity of habitats (>70% forested) surrounding the urban-suburban study areas, and no difference in habitats used across the region documented in a previous study (Tri et al. 2016).

To account for the variation in the number of locations for each individual bear, we included an identifier for each bear as a random effect in the models (Gillies et al. 2006). Thurfjell et al. (2014) point out that with increasing fix rate, positional data of animals become increasingly autocorrelated. Thurfjell et al. (2014) go on to say that temporally autocorrelated data do not affect the value of estimated coefficients of other predictor variables but may underestimate their variance. Furthermore, it is likely that bears occupied small areas during our

Table 1. Candidate set of 8 models selected *a priori* to evaluate the influence of different combinations of environmental and anthropogenic covariates on resource selection by American black bears (*Ursus americanus*), 2010–2012, Pennsylvania, USA.

	Model terms
Model 1:	Elevation + Aspect + Slope
Model 2:	Huntable + Elevation + Aspect + Slope
Model 3:	Aspect + Elevation + Housing density + Slope
Model 4:	Land cover + Elevation + Aspect + Slope
Model 5:	Land cover + Huntable + Housing density
Model 6:	Huntable
Model 7:	Housing density
Model 8:	Land cover

Table 2. Models with the most support using Akaike's Information Criteria (AICc) adjusted for small sample size and model weights (weight) evaluating the influence of environmental and anthropogenic covariates on resource selection by American black bears (*Ursus americanus*), 2010–2012, across 3 sampling periods (pre-hunting, hunting, post-hunting) in Pennsylvania, USA. No additional models tested were >2.0 ΔAICc and were therefore not reported.

Model terms	AICc	Weight
Pre-hunting		
Land cover + Elevation + Aspect + Slope	26546.3	0.999
Hunting		
Land cover + Elevation + Aspect + Slope	20021.7	0.994
Post-hunting		
Land cover + Elevation + Aspect + Slope	9210.4	0.616
Huntable + Housing density + Land cover	9211.7	0.324

short study periods (10 days each), which would have, by default, resulted in some of the data being temporally autocorrelated. We, therefore, tested our models with time of the bear locations as a predictor variable but found no change in the values nor the variance of the coefficients of the other predictor variables estimated by the model.

We followed the informational theoretic approach recommended by Burnham and Anderson (2002) and selected best-fit models based on lowest AICc value in each of the 3 periods: pre-hunting, hunting, and post-hunting. We assessed the validity of the top model in each period using a k-folds cross-validation and Spearman-rank correlation test across 5 randomly selected training sets (Boyce et al. 2002). We used a model training-to-testing ratio of 80:20 for the 5 random sets by fitting the top model with all data, then using the estimated coefficients to obtain predicted values for both training and testing datasets.

Results

We analyzed location data of 29 bears (11 females and 18 males) with individual bear location data in a given period ranging from 61–217. The data included 5,546, 4,196, and 1,926 location points of 29, 24, and 11 bears in the pre-hunting, hunting, and post-hunting periods, respectively. Of the 29 bears we monitored during the study, 8 bears were harvested by hunters (3 bears in 2010, 4 bears in 2011, 1 bear in 2012) and others were lost to vehicle mortality, dropped collars, or when they left the study area. This resulted in uneven sample sizes across periods and years.

In both the pre-hunting and hunting seasons, the model that included land cover with all 3 topographical covariates was the most supported model with 99% of model weight (Table 2). In the post-hunting season, the model that included land cover with all 3 topographical covariates was also the most supported model, but the model that included land cover, housing density, and huntable also showed strong support and was within 2.0 ΔAICc (Table 2).

Forested landscapes dominated all covariates in all periods, and we found evidence that bears also selected for herbaceous covered lands in the pre-hunting and hunting periods (Table 3). Bear locations were negatively related to west- and north-facing slopes in all 3 periods in the pre-hunting and hunting seasons and positively related to west-facing slopes in the post-hunting period (Table 3). While steeper slopes were positively related to bear locations in the hunting period, bears appeared to be selecting for less steeper slopes in the pre-hunting period (Table 3). The second most supported model for the post-hunting period indicated that bear locations were positively related to parcels of land ≥4 ha that did not allow hunting (Table 3). Cross-validation indicated that models performed well for each of the 3 periods (Table 4).

Discussion

Bear were harvested in our study area during the hunting period, indicating that lack of selection for properties open to hunting was not necessarily in response to management activities (i.e., hunting). Bears did not appear to alter resource selection during management activities, suggesting that the use of active management as a tool in urban/suburban areas is possible within our study area.

We expected that bears would prefer sloping forested habitats because of the need to gain mass prior to the energetically demanding denning season (Robbins et al. 2007). Tree species found in forested habitats that produce hard-mast, such as oaks (*Quercus* spp.), have been shown to be the single most important food resource for bears (Beeman and Pelton 1980, Inman and Pelton 2002). The need for bears to maximize consumption of this important resource before denning for winter, coupled with a desire to seek cover as much as possible in areas in relatively close proximity to people, may explain why we found bear selecting forested patches in all 3 periods. Our findings were similar to results reported from an urban bear study in the mid-Atlantic region (Tri et al. 2016).

In the northern hemisphere, west- and north-facing slopes receive less sunlight than south-facing slopes (Nowacki and Abrams 1992, Beaty and Taylor 2001). It is possible that because less sunlight is received, trees on these slopes have a reduced amount of mast available, which may explain why we found bears not preferring west- and north-facing slopes in the pre-hunting and hunting periods. It is worth noting that analyzing remotely-sensed data on tree greenness or composition, such as normalized difference vegetation index, was not an option for this study, as the period during which we analyzed data coincided with the time when all non-coniferous trees had lost foliage.

Our study, though, was not without caveats. Logistics and lack of survey responses prevented us from actually sampling if hunting occurred on a parcel, sampling all landowners whose land was used by radio-marked bears. Thus, we were unable to determine hunting access for every single parcel of land used by radio-marked bears. In addition, recording the number of hunters, access points to hunted parcels, and greater details on levels of disturbance by hunters would be beneficial in future research designs. To increase the accuracy of inferences from analyzing bear resource selection, subsequent research could focus on a single area with more radio-marked bears and a more thorough survey to document land available for hunting.

Documenting the use of non-huntable lands after the hunting season, however, is difficult to interpret because it is unclear if bears use these general areas for forage consumption to prepare for the onset of winter or as refugia in response to the pressure of the hunting season over the previous 10 days. Nevertheless, the ability of our study design to identify bear selection for non-huntable lands after the hunting season closed has important methodological implications. This conclusion is supported by our models that identified areas of hunting access and an important covariate.

It is worth noting that the high response rate for the landowner survey in our study was comparable to and even slightly higher than response rates to a survey in Minnesota, USA, re-

Table 3. Parameters, model coefficients (estimates), standard error (SE), and 95% confidence intervals (CI) for the most supported model based on 8 candidate models used to assess resource selection by American black bears (*Ursus americanus*) in 10-day periods prior to (pre-hunting), during (hunting), and after the firearm hunting season (post-hunting), 2010–2012, across Pennsylvania, USA.

Parameters	Estimates	SE	CI
Pre-hunting			
Forest	0.59	0.09	0.42 to 0.77
Grass	0.42	0.11	0.20 to 0.64
Elevation	-0.17	0.50	-1.16 to 0.81
Aspect (North)	-0.18	0.06	-0.31 to 0.06
Aspect (South)	-0.05	0.05	-0.16 to 0.05
Aspect (West)	-0.23	0.06	-0.35 to -0.11
Slope	-0.65	0.23	-1.10 to -0.19
Hunting			
Forest	0.75	0.10	0.55 to 0.94
Grass	0.29	0.13	0.04 to 0.54
Elevation	-0.24	0.53	-1.27 to 0.80
Aspect (North)	-0.22	0.08	-0.37 to 0.06
Aspect (South)	0.06	0.08	-0.06 to 0.17
Aspect (West)	-0.15	0.07	-0.29 to -0.01
Slope	0.96	0.23	0.50 to 1.41
Post-hunting			
Model 4			
Forest	0.46	0.14	0.19 to 0.72
Grass	-0.11	0.19	-0.50 to 0.26
Elevation	0.37	0.74	-1.08 to 1.82
Aspect (North)	0.02	1.03	-0.18 to 0.22
Aspect (South)	-0.30	0.10	-0.50 to -0.10
Aspect (West)	0.05	0.10	-0.16 to 0.26
Slope	0.35	0.31	-0.26 to 0.96
Model 5			
Forest	0.45	0.14	0.18 to 0.72
Grass	-0.15	0.19	-0.53 to 0.23
Huntable: hunting not permitted based on survey	0.39	0.19	0.01 to 0.76
Huntable: non-respondent to survey	-0.08	0.15	-0.37 to 0.21
Huntable: not surveyed	0.12	0.37	-0.49 to 0.97
Housing density: rural	0.17	0.18	-0.33 to 0.38
Housing density: suburban	0.13	0.16	-0.07 to 0.55

Table 4. The predictive ability of the top resource selection models for each sample period using k-folds cross-validated Spearman-rank correlations across 5 training data sets (set) for American black bears (*Ursus americanus*), 2010–2012, Pennsylvania, USA.

Set	r_s	*P*
Pre-hunting		
1	0.95	<0.001
2	0.90	<0.001
3	0.90	<0.001
4	0.86	0.007
5	0.90	<0.001
Hunting		
1	0.95	<0.001
2	0.93	<0.001
3	0.86	0.007
4	0.93	0.007
5	0.95	0.007
Post-hunting		
Model 4		
1	0.93	<0.001
2	0.93	<0.001
3	0.93	<0.001
4	0.90	<0.001
5	0.83	0.01
Model 5		
1	0.97	0.004
2	1.0	<0.001
3	1.0	<0.001
4	0.99	<0.001
5	1.0	<0.001

garding bear conflict (Garshelis et al. 1999) and a survey regarding public perceptions of bear in Arkansas and Mississippi, USA (Bowman et al. 2001). The high response rate is encouraging because it demonstrated that a relatively high percentage of landowners were engaged with public or government agencies in the study sites. Expanding bear hunting in Pennsylvania may have increased public support, as 70% of adult residents surveyed agreed with using legal regulated hunting to manage bear populations (Duda et al. 2008). Furthermore, maximizing the number of landowners, both public and private, who are willing to implement strategies to manage bears, and the fact that hunters are willing to pay to travel and hunt on lands open to bear hunting, is an obvious focus of future management strategies (Mozumder et al. 2007).

Management implications

The novel approach of our study assessed resource selection of bears by obtaining information from a survey of landowners with properties large enough to potentially accommodate management activities. Combining a survey of landowners with an SSF approach enabled spatial analysis on resource selection of bears during 3 10-day sampling periods focused around the hunting season in our study sites. Our study refined approaches to understanding resource selection of bears in the urban-suburban landscape. By including a spatial layer of properties that permit hunting of bears, we were able to specifically assess use of lands open to hunting that were previously not considered in bear management strategies. While the preference of forested lands by bears in suburbia is apparent and has been confirmed by multiple studies, management of bears in suburbia would continue to benefit from subsequent research focused at determining the role that lands open to hunting play in resource selection of bears in urban-suburban landscapes.

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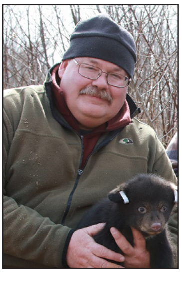
Associate Editor: Carl W. Lackey

Farshid S. Ahrestani currently works as a data scientist and conservationist in South Asia. He holds a Ph.D. degree in ecology, is a Fulbright Scholar, and is an expert on South Asian large herbivores. His research has included investigating how climate change influences dynamics in large herbivore populations, how to improve estimating populations of difficult-to-detect mammals in tropical forests, developing integrated state-space models for North American songbird populations, and exploring the interface of the semantic web and ecological research.



Mark A. Ternent is currently the wildlife management supervisor for the Pennsylvania Game Commission's northcentral region. For nearly 20 years, he also coordinated the statewide black bear research and monitoring program. He holds wildlife conservation degrees from Frostburg State University (B.S., 1991) and the University of Minnesota (M.S., 1995). Before joining the Pennsylvania Game Commission, he also worked as a trophy game biologist with the Wyoming Game and Fish Department. He is a member of the International Association for Bear Research and Management and previous chair of the Northeast Bear Technical Committee.

Matthew J. Lovallo is the chief of administration within the Pennsylvania Game Commission's Bureau of Wildlife Management. He received a Ph.D. degree (1999) from The Pennsylvania State University and M.S. (1993) and B.S. degrees (1990) from The University of Wisconsin – Stevens Point. In his prior positions within the Pennsylvania Game Commission, he spent 20 years as a field biologist coordinating a variety of field research investigations throughout the Commonwealth. His research was primarily focused on furbearer management and the development of sustainable harvest management programs.

W. David Walter is the assistant unit leader at the U.S. Geological Survey, Pennsylvania Cooperative Fish and Wildlife Research Unit and also serves as adjunct associate professor of wildlife ecology in the Department of Ecosystem Science and Management at The Pennsylvania State University. He received his Ph.D. degree from Oklahoma State University (2006), M.S. degree from the University of New Hampshire (2000), and his B.S. degree from the State University of New York-College of Environmental Science and Forestry (1995). His research program focuses on compiling and evaluating datasets at the landscape or regional level for a more complete understanding of ecology of a variety of organisms that includes spatial ecology, disease epidemiology, and landscape genetics.