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Author(s): Shefali Azad, Tammy Wactor and David Jachowski

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Relationship of Acorn Mast Production to Black Bear Population Growth Rates and Human–Bear Interactions in Northwestern South Carolina

Shefali Azad^{1,*}, Tammy Wactor², and David Jachowski¹

Abstract - Acorns represent a critical pulsed food source for American Black Bears (*Ursus americanus*) in the southern Appalachians, and represent their primary hibernation reserves. We used 20 years of acorn-mast data collected in northwestern South Carolina and examined time-lagged correlations to American Black Bear population growth rates (λ), human–bear interactions, and bear visitation to bait stations. Our goal was to assess the relative significance of annual indices monitored for state bear management. Our results indicated that λ was linked to acorn crop quality, although the correlation varied with oak species: positive with white oaks and negative with red oaks. Human–bear interactions were negatively correlated to mast in the same year. There was no significant relation between bait-station visitation and mast or λ . Overall our study reflects gaps in current monitoring practices, and we provide ideas towards refining them.

Introduction

A number of ecosystems are characterized by pulsed resources, which consist of periodic events of high resource availability followed by long inter-pulse events of reduced resource availability. Some well known examples include periodic outbreaks of insects (Hahus and Smith 1990, Yang 2004), increased plant growth following an El Niño event (Stapp and Polis 2003, Wright et al. 1999), seasonal transfer of nutrients from oceans to riparian systems by migrating salmon (Helfield and Naiman 2001), and mast fruiting by trees (Kelly 1994, Sork et al. 1993). Masting is defined as a synchronous seed production event by a large population of plants that is highly variable among years (Kelly 1994). Plants appear to track abiotic environmental variations to induce masting (Norton and Kelly 1988, Wright et al. 1999). Evolutionary benefits facilitated by masting include satiating predators during mast years to allow a substantial percentage of seeds available for germination and maintaining predator populations at lower levels in non-mast years (Isagi et al. 1997, Kelly 1994, Ostfield and Keesing 2000, Visser et al. 2011).

As mast fruits are an irregular resource, it is expected that the consumers most likely to respond to a masting event are trophic generalists (Ostfeld and Keesing 2000). In deciduous forest ecosystems, masting events can cause rapid population growth in generalists such as *Peromyscus leucopus* (Rafinesque) (White-footed Mouse; Elkinton et al. 1996, McShea 2000, Ostfeld et al. 1996), *Tamias striatus*

¹Department of Forestry and Environmental Conservation, Clemson University, Clemson, SC 29634-0317. ²Wildlife and Freshwater Fisheries, South Carolina Department of Natural Resources, Clemson, SC 29631. *Corresponding author - azadshfali@gmail.com.

(L.) (Eastern Chipmunk; McShea 2000, Wolff 1996), and *Odocoileus virginianus* (Zimmermann) (White-tailed Deer; McShea and Schwede 1993), with growth rates declining in interpulse intervals. *Ursus americanus* (Pallas) (American Black Bear, hereafter referred to as Black Bear), is also a generalist consumer, with a diet comprised generally of acorns, berries, grasses, insects, small rodents, birds, carrion, and foods from anthropogenic sources (Beeman and Pelton 1980). In the southern Appalachian region, they appear to prefer extensive, mature hardwood forests with late successional nut (acorn [*Quercus* spp.], hickory [*Carya* spp.], hazelnut [*Corylus* spp.]) trees and berry (blueberry [*Vaccinium* spp.], huckleberry [*Gaylussacia* spp.], service berry [*Amelanchier* spp.]) shrubs (Landers et al. 1979, Pelton 1989). However, acorns in particular represent the primary, energy-rich source of food and hibernation reserves for Black Bears in the Appalachians during the dormant season (Beeman and Pelton 1980, Clark 2004, Garner 1986). Pelton (1989) reports that Black Bears in the Appalachians may respond to mast failure by undertaking long-range movements, or intensively using small areas of high acorn concentration. Subsequently, this change in activity may lead to increased animal–human conflict (Baruch-Mordo et al. 2014; Obbard et al. 2014; Ryan et al. 2004, 2007; Sato and Endo 2006). Hard mast failure may also negatively impact female age of reproductive maturity, litter size, and barrenness, suggesting that population growth may subsequently fluctuate with success of mast crops (Beeman and Pelton 1980, Costello et al. 2003, Eagle and Pelton 1983, Eiler et al. 1989, Jonkel and Cowan 1971, McLean and Pelton 1994).

The direct correlation of masting to Black Bear population growth rates has been less well established. While studies have documented the influence of mast on seasonal movement and activity (Beeman and Pelton 1980, Kozakai et al. 2013, Schooley et al. 1994), reproductive potential (Costello et al. 2003, Eiler et al. 1989, Jakubas et al. 2005, Jonkel and Cowan 1971, Rogers 1976), mortality and human–bear conflict (Beeman and Pelton 1980; LaMere 2012; Obbard et al. 2014; Ryan et al. 2004, 2007), and bait-station visitation (Clark et al. 2005), few studies have assessed the relationship between production of acorn mast and population growth rates using long-term datasets (Clark et al. 2005, McLean and Pelton 1994, Reynolds-Hogland et al. 2007). In South Carolina, Black Bears are managed as a game species, and it is thus important for wildlife managers to understand the factors that affect their population growth rates and human–bear interactions (HBI). However, no study has evaluated Black Bear population dynamics in relation to acorn mast in South Carolina.

Our goal for this study was to analyze long-term datasets collected by state wildlife agencies to assess their statistical significance in the monitoring and management of Black Bear populations. We used a 15-year dataset of Black Bear abundance derived from harvest (Azad 2016) and long-term mast records from the South Carolina Department of Natural Resources (SCDNR) to examine the effect of acorn masting events on Black Bear population growth rates (λ) and HBI in northwestern South Carolina. Specifically, we hypothesized that a time-lagged, positive correlation exists between acorn mast indices and λ . We predicted

bear populations would increase ≥ 2 years after large mast events, considering recruitment of cubs born next spring to the harvestable population at ages ≥ 1 . We also hypothesized a negative correlation between acorn mast indices and HBI, likely due to increased foraging activity by bears in years of mast failure. Finally, bait-station visitation (the percentage of sardine bait stations visited by bears) was used as a population index for bears in South Carolina, but its efficacy had not been examined against lambda. Furthermore, Garshelis (1990) suggested visitation could vary with physiological and environmental covariates including food resource availability. Therefore, we hypothesized a positive correlation between visitation to bait stations and lambda, and predicted that visitation would be impacted positively by a reduction in mast availability.

Field-Site Description

Our study area primarily consisted of the region where bear harvest was permitted in northwestern South Carolina, defined by the SCDNR as “Game Zone 1” and extending over Oconee, Pickens, and Greenville counties (Fig. 1). Forest types in this region generally consisted of *Pinus echinata* Mill. (Shortleaf Pine), *Quercus prinus* Willd. (Chestnut Oak), *Q. coccinea* Muenchh. (Scarlet Oak), and hickory mixes on the southern Appalachian slopes (Butfiloski 1996, Willey 1995). Other notable masting species found in the region included *Q. alba* L. (White Oak), *Q. stellata* Wangenh. (Post Oak), *Q. rubra* L. (Northern Red Oak), and *Q. falcata* Michx. (Southern Red Oak). Biotic and climatic conditions were generally similar across all forested lands in the counties under consideration. For the response variable HBI, we also considered conflict reports from the counties of Anderson, Spartanburg, Laurens, and Cherokee that surrounded the primary study area.

Methods

Data collection

Hard-mast index (HMI). We used 23 years of hard-mast survey data collected by SCDNR, from 1993–2015. Surveys were conducted annually between August 15 and September 15 in established bear habitat, and survey routes were established such that altitudes varied as much as possible (Fig. 1; Greenberg and Warburton 2007). The elevation range of survey routes was between 305 and 915 m, with routes along mountainous ridges often varying by as much as 425 m along the same route. Routes were at least 16 km, with stops at 1.6-km intervals, and 2–4 tagged trees of each tree group were surveyed at each stop (new trees selected only if previously tagged trees were dead or logged, in the same vicinity and having dimensions similar to the old tree; Greenberg and Warburton 2007). Tree groups consisted of species preferred for consumption by Black Bears, divided up as follows: (1) white oak (White Oak and Post Oak) (2) red oak (Northern Red Oak, Southern Red Oak, Scarlet Oak, and *Q. velutina* Lam. [Black Oak]); (3) chestnut oak (Chestnut Oak), and (4) hickory (*Carya* spp.). Hickory data were only collected from 2004 onwards.

At each selected tree, acorn crops were rated on a relative scale using a hard-mast index (HMI). From 1993 to 2005, SCDNR personnel adopted the Whitehead (1969) survey method. This included for each tree a scored estimate of the percentage of tree crown with acorns (PCA), and counts of twigs, twigs bearing acorns, and number of acorns on a random subsample of 5 oak limbs >90 cm long. From 2006 to 2015, SCDNR personnel used the Greenberg and Warburton (2007) survey method that was standardized across the Southern Appalachian Black Bear Study Group (SABBSG 2007). This method estimated PCA by a surveyor looking up at the tree crown for 30 seconds, and proportion of trees bearing acorns (PBA) was then calculated as percentage of all trees sampled with PCA >35%. A regression equation was used to determine the HMI as:

$$y = b_0 + b_1x,$$

where y was the predicted HMI, x was PBA, and b_0 and b_1 were regression equation coefficients specific to the species of oak (Greenberg and Warburton 2007). The Greenberg and Warburton method was developed to predict the Whitehead method HMI using a less labor-intensive technique. We interpreted HMI of mast crop for each tree group as: 0.0–2.0 = poor, 2.1–4.0 = fair, 4.1–6.0 = good, and 6.1–8.0 = excellent (Greenberg and Warburton 2007).

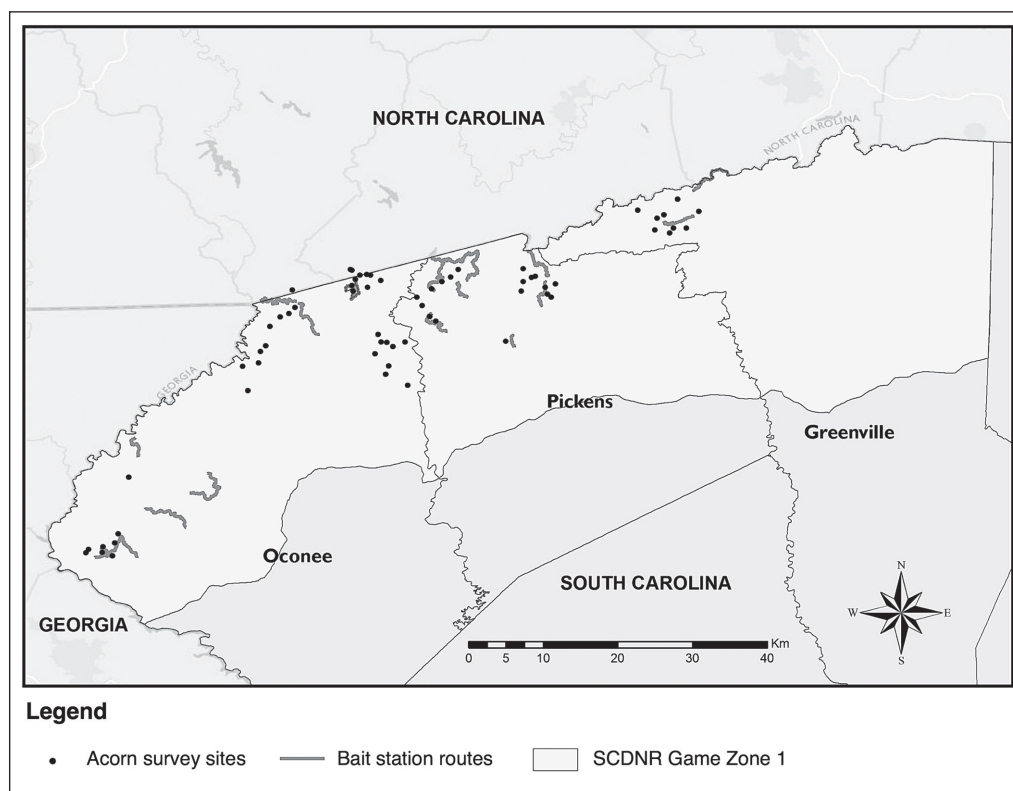


Figure 1. Location of acorn-mast survey site and bait-station routes, and the South Carolina Department of Natural Resources (SCDNR) defined Game Zone 1 (in white) for Black Bear harvest in the northwest region.

Black Bear data. We used 18 years of annual harvest records maintained by the SCDNR to reconstruct Black Bear populations in northwestern South Carolina (Azad 2016). Harvest records of both male and female bears of all ages and method of harvest from SCDNR Game Zone 1 were used in the reconstruction. Techniques essentially consisted of utilizing age- and sex-specific records of each harvested bear. The Downing (1980) method was used to perform a backwards reconstruction of cohorts, and subsequently estimate abundances, from harvest records (Azad 2016). We used the Python packages *xlrd* and *xlwt* to analyze harvest datasets (ver 2.7; van Rossum and Drake 1995). We augmented the dataset with non-harvest mortality, which consisted of road kill and euthanized nuisance bears. Due to lack of age estimates in data prior to 1998 and the limitations of the technique to reconstruct recent years, we reconstructed abundance for 1998–2013.

We generated an annual index of HBI for 2004–2015 based on nuisance bear reports submitted to SCDNR, typically contacted as the first responders to human–animal conflict in the region (SCDNR 2016a). This included visual sightings, property invasions, and crop damage by Black Bear. Reports of human–bear conflict were typically received as phone calls to the SCDNR, who recorded the geographical coordinates of the conflict points.

Bait station index (BSI). We used 23 years of bait-station visitation data spanning 1993–2015 that were collected by the SCDNR as part of a multi-state initiative to indicate changes in population (Clark et al. 2005). Bait-station surveys from 15 July to 10 August were conducted annually 1993–2005 and every 2 years during 2005–2015 (odd years). Bait-station transects, or bait lines, were established in accordance with Carlock et al. (1983) guidelines, and consisted of 10 stations spaced at 0.8-km intervals (Fig. 1). The elevation range of bait-station routes was between 365 and 915 m, with routes along mountainous ridges often varying by as much as 245 m along the same route. At each station, a partially opened can of sardines was suspended 2.5–3.0 m above ground level in a tree 10–20 cm in diameter, left for 5 nights, and then inspected for Black Bear visitation (Fendley 1991). Visitation was a presence–absence index determined by one or more of the following signs: tracks, bent/broken trees, claw marks on trees, and tooth punctures in sardine cans (Fendley 1991). The BSI was calculated as percentage of bait stations visited by bears that year.

Bait-station surveys were divided into core and fringe areas in accordance with historical records of core and fringe bear habitat by SCDNR, who subsequently monitored BSI separately for each region (SCDNR 2016b). We used the same division of core and fringe BSI in our analysis, and noted that bait-station visitation was typically lower in fringe bear areas. The core area was defined as traditional bear habitat based on harvest and nuisance records, and consisted of properties in the Jocassee Gorge in eastern Oconee and northern Pickens counties, and the Camp Greenville area in northern Greenville County. The fringe area consisted of areas west of Lake Jocassee in Oconee County and the remaining bear-inhabited areas of Pickens and Greenville counties (SCDNR 2016b).

Statistical analysis

In order to assess how resource limitations (mast availability) and monitoring indices (bait-station visitation, conflict reports) predict annual population growth (λ) and HBI, we first needed to determine the temporal delay (if any) with which these factors affected λ and HBI. Our goal was thus to identify the factors which, when accounted for a possible time lagged-response, had a statistically significant correlation to λ and HBI. We determined that a time-series analysis, a method of identifying the nature of a phenomenon represented by a sequence of observations, was most appropriate to identify these correlations, followed by a regression analysis to determine the effect size of the significant variables on λ and HBI. We first examined both auto-correlations within a dataset as well as cross-correlations between pairs of datasets. In order to minimize autoregression (the process used in stochastic analyses to account for dependence of a variable's value at year t on its own preceding values in previous years), we corrected for time-varying processes in all datasets by converting time-series data to percentage or rate change in annual values. We calculated the annual population growth rate over the study period using the formula $\lambda = N_t / N_{t-1}$. We converted BSI into Δ BSI using the formula Δ BSI $_t = \text{BSI}_t - \text{BSI}_{t-1}$. As BSI was collected every alternate year after 2005, Δ BSI values were only available until 2005.

We used the R package *forecast* (ver 7.1; Hyndman and Koehler 2006) to check datasets for need for further autoregression with the *acf* (autocorrelation) function, which is based on the standard Pearson's correlation coefficient (r) applied to a time series with a lag, and the *auto.arima* function to identify autoregressive transformations if any. We computed pairwise associations between each category of HMI (white oak, red oak, chestnut oak, hickory) and λ , Δ BSI, and HBI, using the function *ccf* to check for cross-correlation with a lag of up to 3 years. We chose a 0- to 3-year lag, considering that mast quality in year $_t$ could affect reproductive potential of females in year $_t$, subsequent birth of cubs in year $_{t+1}$, and recruitment into the harvestable population in year $_{t+2}$ (Clark et al. 2005, McLean and Pelton 1994). We also computed pairwise time-lagged cross-correlations between Δ BSI and λ to determine if change in population growth was reflected in bait-station visitation by bears. When carrying out cross-correlations, we trimmed datasets to match years of the shorter dataset, adjusting for the time lag chosen. For example, while examining correlations between white oak (data available 1993–2015) and λ (data available 1998–2013), we used white oak acorn indices from 1995–2013, assuming that acorn production prior to 1995 would not impact Black Bear population growth in or after 1998.

Next, we built a set of regression models with λ and HBI as response variables using a stepwise-regression approach by adding statistically significant correlations, defined as $|r| > 0.40$ (Evans and Over 1996), as dependent variables. For response variable λ , we considered all permutations of white oak, red oak, chestnut oak, and Δ BSI as dependent variables, and also applied 0- to 3-year time-lagged effects for each dependent variable. For response variable HBI, we considered all permutations of white oak, red oak, and chestnut oak. We only included a 0- to

1-year time-lagged effect for HBI, assuming that human interactions and road kills would not be affected by >1 year previous mast production.

After running all permutations of dependent variables, we eliminated statistically significant ($P < 0.1$) regression models having R^2 values < 0.5 . We also eliminated additive models (>1 covariate) where the addition of one covariate removed statistical support ($P < 0.1$) for all others. We compared the remaining models using Akaike's information criterion corrected for small samples (AICc) to determine relative statistical support. We did not model average because our primary goal was variable selection rather than parameter prediction (Arnold 2010).

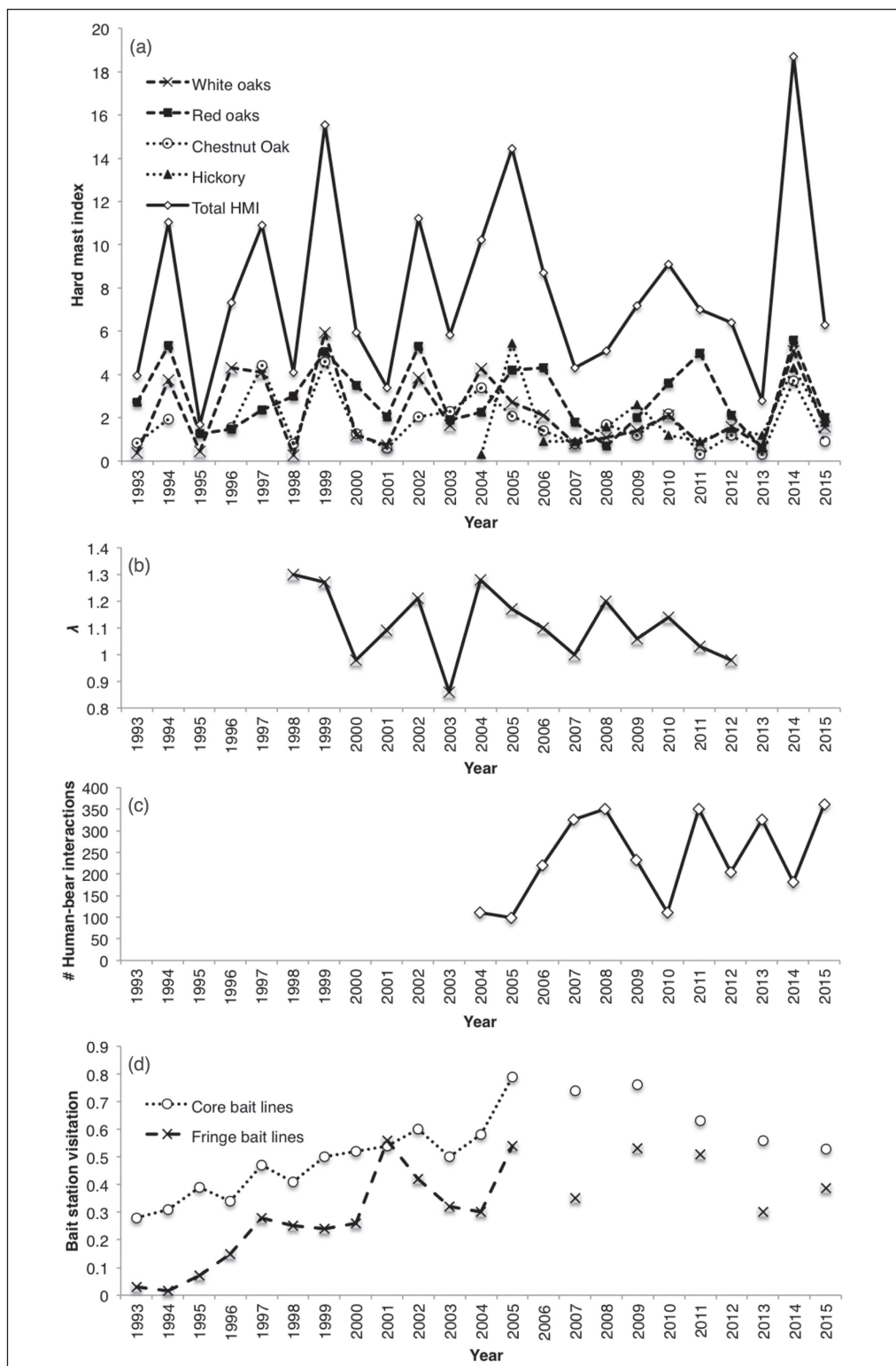
Results

Total HMI fluctuated from year to year, with consecutive peaks recurring every 2–4 years, where peaks were defined as a net change in HMI between 2 consecutive years that corresponded to a higher mast crop ranking on the Greenburg and Warburton (2007) scale (0.0–2.0 = Poor, 2.1–4.0 = Fair, 4.1–6.0 = Good, and 6.1–8.0 = Excellent). A notable exceptions was the period in 2004–2010 that had 6 years between peaks (Fig. 2a). Based on our criteria, white oak had consecutive peaks recurring approximately every 2–3 years, red oak every 2–4 years, chestnut oak every 2–5 years, and hickory every 4–5 years (Fig. 2a). White, red, and chestnut oaks also showed an overall decline in mast production in 2004–2009 (mean annual decline in HMI of 11.21% and 6.25% for white and chestnut oaks, respectively, in 2004–2009 and 7.66% for red oak in 2004–2008). λ and HBI fluctuated, with consecutive peaks recurring every 2–4 years (Fig. 2b,c); we assessed peaks visually where there was a net increase and subsequent decline in population growth rate or number of contact reports over 3 consecutive years. BSI showed a generally increasing trend from 1993 to 2009, and then declined from 2010 to 2015 (BSI_{core}: mean annual increase of 7.62% [SE = 17.61] in 1993–2009, mean annual decrease of 5.63% [SE = 5.79] in 2010–2015; BSI_{fringe}: mean annual increase of 41.74% [SE = 107.66] in 1993–2009, mean annual decrease of 2.60% [SE = 35.28] in 2010–2015; Fig. 2d).

Population growth

We found no significant autocorrelation ($|r| > 0.40$) in the λ time series as well as HMI time series of all tree groups (both individually and combined). This implied that although there was annual variation in λ and HBI, there was no overall trend in any variable over the time period examined. Mathematically, it suggested that we did not need to use the weighted sum of a variable at years $(t - 1)$, $(t - 2)$... to calculate the impact of the variable at year t . Pairwise cross-correlation suggested moderate correlation between λ and white oak at a 0- and 2-year time lag ($r = 0.486$ and $r = 0.470$, respectively), chestnut oak at a 0-year time lag ($r = 0.441$), and Δ BSI_{core} at a 0-year time lag ($r = 0.438$), and strong correlation between λ and red oak at a 1-year time lag ($r = 0.704$). Our top regression model predicting λ at year t was white oak with

Figure 2 (following page). Time series plots of (a) hard-mast index (HMI), (b) population growth rate (λ) of Black Bears, (c) number of human–bear interactions (HBI), and (d) percentage bait-station visitation (BSI) by bears in northwestern South Carolina in 1993–2015.



a 2-year time-lag + red oak with a 1-year time lag ($R^2 = 0.652$, $P < 0.001$, $AICc_w = 0.357$; Table 1). However there was significant uncertainty in which model best predicted λ because the next 2 models were also supported from AICc results (Model 2 $AICc_w = 0.339$, Model 3 $AICc_w = 0.263$; Table 1). White oak and chestnut oak had a moderate positive relationship to the population growth rate of Black Bears, whereas red oak had a strong negative relationship to the growth rate.

Human–bear interactions

We found no significant autocorrelation ($|r| > 0.40$) in the HBI time series, indicating that the number of conflict reports at any year t were not a weighted sum of conflict reports at years $(t - 1)$, $(t - 2)$... over the time period examined. Pairwise cross-correlation checks revealed moderate correlation between HBI and red oak at a 0-year time lag ($r = -0.415$), and strong correlation between HBI and white oak at a 0-year time lag ($r = -0.688$) and chestnut oak at a 0-year time lag ($r = -0.721$). Our top model predicting HBI at year t was white oak at a 0-year time lag ($R^2 = 0.67$, $P < 0.05$, $AICc_w = 0.567$), and white oak had a strong negative relationship to HBI (Table 2). Competing models from AICc rankings included total HMI across all species and chestnut oak as covariates, and both had a negative relationship to HBI.

Table 1. Covariates of the top regression models predicting population growth rate (λ) for Black Bears in northwestern South Carolina in 1998–2013. AICc is Akaike’s information criterion adjusted for small samples, $AICc_w$ is relative AICc support for models, and df is degrees of freedom in the model. + or - before a covariate indicate positive or negative regression slope. f ($t - n$) represents an effect at a time lag of n years.

Model covariates	Parameter estimate	df	AICc	$\Delta AICc$	$AICc_w$	R^2
White oak($t - 2$)	0.035	4	-26.198	0.000	0.357	0.652
Red oak($t - 1$)	-0.063					
White oak(t)	0.024	5	-26.090	0.108	0.339	0.720
White oak($t - 2$)	0.032					
Red oak($t - 1$)	-0.057					
White oak($t - 2$)	0.033	5	-25.583	0.615	0.263	0.711
Red oak($t - 1$)	-0.058					
Chestnut oak(t)	0.031					
White oak(t)	0.028	4	-25.866	4.332	0.041	0.536
Red oak($t - 1$)	-0.057					

Table 2. Covariates of the top regression models predicting human–bear interactions (BHI) for Black Bears in northwestern South Carolina in 2004–2015. AICc is Akaike’s information criterion adjusted for small samples, $AICc_w$ is relative AICc support for models, and df is degrees of freedom in the model. + or - before a covariate indicate positive or negative regression slope. (t) represents 0-yr time lag.

Model covariates	Parameter estimate	df	AICc	$\Delta AICc$	$AICc_w$	R^2
White oak(t)	-77.88	3	115.257	0.000	0.567	0.6707
Total HMI(t)	-30.64	3	116.583	1.307	0.295	0.6247
Chestnut oak(t)	-84.69	3	118.094	2.819	0.138	0.5634

Bait-station indices

Bait-station indices were autocorrelated, indicating that Black Bear visitation for year t was dependent on visitation in previous years. We received an ARIMA(0,1,0) value for the BSI_{core} and BSI_{fringe} time, indicating that using BSI_t as a dependent variable required the additive inclusion of BSI_{t-1} to correct for autocorrelation. However, we found no significant autocorrelation ($|r| > 0.40$) or need for autoregression in the ΔBSI time series. Pairwise cross-correlation suggested moderate correlation between ΔBSI_{core} and λ at a 0-year time lag ($r = 0.438$) and chestnut oak at a 0-year time lag ($r = 0.581$; Table 2). However, all statistically significant ($P < 0.1$) regression models could not sufficiently predict λ from visitation indices (i.e., model R^2 values were < 0.5).

Discussion

Our results supported the hypothesis that population growth rate of Black Bears in South Carolina was influenced by acorn mast availability in the region, although this relationship varied with oak species. A positive 2-year time-lagged relationship to white oak supported the biological hypothesis that female bears reared cubs with greater success of survival during a good mast year (Eiler et al. 1989, Pelton 1989), which when weaned 2 years later were likely represented in the harvested population that we used to derive abundance. However, we did not have corresponding survival data from dedicated studies on cub survival rates in the South Carolina population to confirm this hypothesis. In contrast to white oak mast production, the population showed a significant negative correlation, with a 1-year lag, to red oak mast. The differences observed between red and white oaks are likely due to preference of white acorns by Black Bears because red oak acorns are less palatable as a result of their higher tannin content (Clark 2004).

There is general agreement that resource limitation drives survival and recruitment, and eventually population growth rates of animals (Costello et al. 2003, McLean and Pelton 1994, Sinclair and Krebs 2002). However, these species–system ecological relationships are complex, and with trophic generalists such as the Black Bear, it may be especially difficult to identify factors that primarily drive population growth. Combined with results that HBI is negatively correlated to mast production, our results suggest that white oaks might impact population growth by increasing foraging activity and associated mortality following poor mast years. However, other studies emphasize the relationship of mast to reproductive success, suggesting that acorn mast regulates females' health and ability to nurse a cub over winter until soft mast production is available in the spring (Eiler et al. 1989, McLean and Pelton 1994). We postulate that the availability of other data such as movement, foraging-activity periods, time of torpor, and female fecundity and cub survival rates would present a more complete picture of how both hard and soft mast impact the expansion and growth of the population. Further, availability of soft mast is hypothesized to offset the impact of hard mast failure on population growth for Black Bears in the southern Appalachians (Eiler et al. 1989, Pelton 1989, Reynolds-Hogland et al.

2007), and it is likely that a future collection and inclusion of soft-mast data would also improve predictive models.

Although we determined that the Downing (1980) method was an appropriate technique of population reconstruction given the limitations of our harvest dataset, there are important assumptions and shortfalls of the method to consider (Azad 2016). Unlike later adaptations (for e.g., Creed et al. 1984, Fryxell et al. 1988, Roseberry and Woolf 1991) the Downing method does not take into account hunting effort, recruitment rates, or recruitment/survival vectors, all of which were unavailable for our population. We assumed that harvest was the primary source of mortality for the population (Klenzendorf 2002, Wooding and Hardisky 1994), and augmented our harvest dataset with secondary mortality reports including roadkills and illegal kills. We also assumed that harvest was non-differential towards age or sex, and improved robustness by inflating our dataset by the proportion of unaged bears harvested each year (Davis et al. 2007). It is important to note that the Downing method provides a minimum point estimate of abundances each year with no estimate of standard error. However, the method has been proven robust to population trajectories over a long-term dataset (Davis et al. 2007, Downing 1980, Tilton 2005), and harvest-reconstruction techniques are widely accepted in the Southeast as a means of establishing population dynamics of Black Bears (Noyce 2011, Tilton 2005). Thus we deemed our reconstruction results appropriate to use in a time-series analysis that compares trends in datasets to determine correlation. Given the uncertainty in abundance estimates, we emphasize again that our regression results be considered a baseline for variable selection rather than parameter prediction of lambda. We recommend that our abundance estimates be calibrated against future mark-recapture, reproductive tract, or similar studies for the population, and suggest that more-rigorous statistical reconstruction estimates be used when auxiliary data are available (Clawson 2015, Etter 2011, Fieberg et al. 2010).

The number of HBI were negatively correlated with acorn mast in the same year, which supports the biological hypothesis that bears increase foraging range and activity to meet nutritional needs during years of mast failure (Healy et al. 1997, Pelton 1989, Rogers 1987). This finding suggests that, similar to previous studies, Black Bears in this study area are less apt to forage near human settlements or scavenge personal property during good mast years, likely indicating an avoidance of human-dominated landscapes when natural food is available (Baruch-Mordo et al. 2014, Peine 2001). However, some studies have shown that variation in natural food does not explain variation in HBI when (1) bear populations are expanding their range, (2) urban development and recreation are expanding into bear habitat, or (3) bears become habituated to unnatural (human) food sources (Cotton 2008, McCarthy and Seavoy 1994, Obbard et al. 2014, Peine 2001, Schirokauer and Boyd 1998, Singer and Bratton 1980). Given the evidence from harvest records that the South Carolina bear population is expanding both numerically and geographically (Azad 2016), we thus recommend that agencies monitor mast-conflict correlations for signs of increasing interactions with no corresponding change in mast. Such signs may signify a need for increased conflict-mitigation training for wildlife

personnel and public educational programs. Furthermore, we recommend future spatio-temporal assessments of human–bear encounters as a potential index of change in habitat use and geographical spread of bears in the state.

Although BSI has been generally considered an indication of density since first established by Carlock et al. (1983), our results did not support a significant relationship between BSI and population growth rate. In addition, BSI showed no significant correlation to acorn mast, and a relative shift from core to fringe bait stations showed no correlation to increased HBI. Further, Pelton (1989) suggested that visitation can be affected by one individual visiting multiple stations, or merely portray bears' use of trails used to set up bait lines (LeCount 1982). It is important to note that Carlock et al. (1983), who established the technique, used bait stations as pre-baits for trap sites in their mark–recapture study, and thus visitation and capture rates used to predict abundance in that study were not independent (Pelton 1989). Other studies linking BSI to abundance or population growth rates have been inconclusive or negative (Clark et al. 2005, Garshelis 1990, Johnson 1989), and Rice et al. (2001) suggested that the statistical power required to detect long-term trends in visitation rates is not typically met by wildlife agencies' survey effort. Although, intuitively, BSI may represent changes in population, we likely lacked the required sample size as well as either supporting covariates or the spatial/temporal scales to determine the relationship (Clark et al. 2005). Further, while core and fringe routes were established in 1993, no study had been carried out since to determine the need for change in routes, addition or removal of survey lines, or updates in definition of an area as core or fringe. We suggest the need for state wildlife agencies to reevaluate the utility of using bait-station visitation as a population index, particularly where resources are insufficient to (1) survey at appropriate scales and (2) validate results against more robust population estimates.

While state wildlife agencies regularly collect annual environmental data or population indices, these data might not translate into rigorous scientific assessment of the relationship of these data to population management techniques (Clark et al. 2005, Garshelis and Hristienko 2006, Howe et al. 2010, Reynolds-Hogland et al. 2007). With a long-lived and elusive species such as the Black Bear, the impact of any demographic or environmental stochasticity might not be apparent to a wildlife agency until several years after the population has responded to the event (Brongo et al. 2005, Garshelis 1990, Garshelis and Hristienko 2006, McLean and Pelton 1994, Noyce and Garshelis 1997, Pelton 1989). Our study highlights several of these long-term patterns, and can aid state wildlife agencies in not only predicting the impact of the environmental variables they monitor; but also in minimizing the uncertainty in the population's response time to environmental stochasticity. We also illustrate several key gaps or problems that need to be further addressed to justify current monitoring practices. Given that these methodologies (e.g., acorn mast surveys, BSI, and bear–human conflict reports) are often used by management agencies to inform harvest regulations, habitat management, and predicting and preparing personnel for increased bear–human conflict in specific areas (Baruch-Mordo et al. 2008, Hristienko and McDonald 2007, Lewis et al. 2015, Obbard and

Howe 2008, Obbard et al. 2014), we stress the need for dedicated scientific studies that validate these monitoring practices and develop standardized robust predictive models of how populations respond to environmental and anthropological events.

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