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Integrating Black Bear Behavior, Spatial Ecology, and Population Dynamics in a Human-Dominated Landscape: Implications for Management

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INTEGRATING BLACK BEAR BEHAVIOR, SPATIAL ECOLOGY, AND POPULATION DYNAMICS IN A HUMAN-DOMINATED LANDSCAPE: IMPLICATIONS FOR MANAGEMENT

by

Jarod D. Raithel

A dissertation submitted in partial fulfillment
of the requirements for the degree

of

DOCTOR OF PHILOSOPHY

in

Ecology

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ABSTRACT

Integrating Black Bear Behavior, Spatial Ecology, and Population Dynamics in a Human-Dominated Landscape: Implications for Management

by

Jarod D. Raithel, Doctor of Philosophy

Utah State University, 2017

Major Professor: Dr. Lise M. Aubry
Department: Wildland Resources

The New Jersey Division of Fish and Wildlife (NJDFW), in collaboration with Bear Trust International, presented us an opportunity to examine a long-term (33 years) American black bear (*Ursus americanus*) data set from northwestern New Jersey (NJ), USA. State agencies continue to grapple with uncertainty about the efficacy of socially divisive management actions such as recreational harvest and lethal control as tools to reduce escalating human-bear conflicts. We applied multistate capture-reencounter models to a large sample of black bear captures (>5,000) and dead recoveries (>1,300) between 1981 – 2014 to estimate cause-specific mortality and spatial dynamics between wildland and anthropogenic habitats. Additionally, we assessed temporal correlations between more than 26,500 reported human–black bear interactions and mortality rates. Adult females were twice as likely (0.163 ± 0.014) as males (0.087 ± 0.012) to be harvested, and cubs (0.444 ± 0.025) and yearlings (0.372 ± 0.022) had a high probability of dying, primarily from vehicle strikes. Nuisance behaviors reported declined with increasing harvest and lethal management ($P = 0.028$, $R^2 = 0.338$). Adult bears previously designated as a nuisance and/or threat (hereafter, “problem”) were more likely to be harvested (0.176 ± 0.025) than those with no conflict history (0.109 ± 0.010). Combined legal kills and vehicle strikes, the two greatest mortality causes for marked bears, occurred significantly less than expected per unit area in urban and agricultural areas, and more than expected in the wildland-urban interface and wildland habitats. Across all age-classes, problem bears were significantly more likely to transition to anthropogenic habitats, yet they died at lower rates than conspecifics with no history of conflict in wildlands. Cubs and yearlings died at significantly higher rates than adults in the risky interface habitat, corroborating independent estimates of their increased susceptibility to harvest and vehicle strikes. Ultimately, wildland habitats represented a population source ($\lambda = 1.133$) and anthropogenic habitats a sink ($\lambda = 0.945$). Harvest represents an important management tool to help meet population targets and decrease human-bear conflicts by disproportionately removing problem bears.

(234 pages)

PUBLIC ABSTRACT

Integrating Black Bear Behavior, Spatial Ecology, and Population Dynamics in a Human-Dominated Landscape: Implications for Management

Jarod D. Raithel

The American black bear (*Ursus americanus*) has made a robust recovery within the human-dominated, social-ecological systems characterizing the Mid-Atlantic United States. For example, in northwestern New Jersey (NJ), USA, black bear abundance increased from an estimated 450-500 in 1996 to 3200-3400 in 2010. Bear recovery coincided with increasing human populations, coupled with shifting settlement patterns toward sprawling suburban communities. Consequently, conflicts have rapidly proliferated over the past three decades and resulted in >1400 incidents of verified property damage, >400 livestock kills, >250 pet attacks and/or kills, seven human attacks and one human fatality since 2001. The New Jersey Division of Fish and Wildlife (NJDFW) has spent in excess of $9 million USD on black bear management and has concluded that this level of conflict is fiscally and culturally untenable. Conservation efforts must now pivot toward shaping bear behavior to facilitate human-bear coexistence within the increasingly shared landscapes of the Anthropocene.

We assessed whether NJDFW's newly implemented black bear harvest was effective in curbing bear population growth and mitigating increasing human-bear conflicts. Adult females and bears with a history of conflict with humans (i.e., "problem" bears) were disproportionately harvested. Problem bears, across all age classes, were significantly more likely to be recaptured in urban and wildland-urban interface habitats. During harvest years, the population growth rate of bears in wildland habitats stabilized, while the anthropogenic component of the population decreased dramatically. We recommend that a carefully regulated harvest continue to be part of an integrated management strategy that includes education and incident-response protocols, which collectively will help reduce human- black bear conflicts.

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From the moment our paths crossed, I was struck by the irrepressible enthusiasm and unwavering resolve by which Melissa approaches the many roles she takes on. She has reinvigorated my belief that now, more than ever, conservation hinges on translating the work of academics into meaningful, ecological curricula for educators and all students. Her example speaks volumes – as ecologists we have a shared responsibility in ensuring that a conservation ethic begins at home with our own children, and extends not just to university students, but to all children at our K-12 public schools and community colleges. While I will never be able to fully reciprocate for all you have done on my behalf Lise and Melissa, I promise you that I will carry your examples in paying it forward to my future students, thank you :)

I give special thanks to my dad, aka, granddad, for all of your support in helping us raise our beautiful and "spirited" children. Bodhi and Sage, should you read this someday, know that my greatest desire in this world is for you two to live healthy, peaceful, and joyous lives. To my wife, traveling companion, and best friend Heather, I love you now more than ever, thank you for this.

Jarod Raithel

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CHAPTER 1

INTRODUCTION

Human activities and anthropogenic landscape transformation induce pervasive ecological ramifications, including altering animal behavior (Ditchkoff et al. 2006, Lowry et al. 2013), habitat use and resource selection (Markovchick-Nicholls et al. 2008, Bateman and Fleming 2012), and population dynamics (Fischer et al. 2015, Šálek et al. 2015). The classical approach of portraying ecosystem patterns and processes as predominantly a function of physical geography, climate, and biotic interactions becomes increasingly insufficient wherever human activities and their associated landscape transformations occur (Hobbs et al. 2006, Ellis and Ramankutty 2008). Across the biosphere, ecological dynamics are now principally driven by the type, intensity and historical duration of human interactions with the system (Ellis et al. 2010). Sustaining the structure, function, identity, and feedbacks inherent to social-ecological systems requires recognizing the magnitude of human influence on the system, as well as, the stakeholders' capacity to affect system resilience (Walker et al. 2004).

While conserving contiguous wildland tracts and networks that allow for the persistence of biological diversity continues to be of prime importance (Soulé and Simberloff 1986, Kingsland 2002), today a mere 10.1 – 15.5% of the world's terrestrial landmass is afforded some level of protected status (Soutullo 2010). With forecasts of burgeoning human population growth, paired with increases in per capita consumption and urban expansion (Seto et al. 2012), acquiring large refugia will become increasingly difficult (Shafer 2008). Therefore, applied ecology must continue to readjust its focus toward systems from across the wildland to urban landscape gradient, within which understanding human-wildlife interactions may be paramount in effectively informing conservation and management efforts (Alberti et al. 2003).

Status and Ecological Importance of Large Carnivores

Despite being comprised of some of the world's most iconic species, numerous ecologically-influential, apex mammals within the order Carnivora continue to experience precipitous population and geographic range declines (Morrison et al. 2007, Di Marco et al. 2014). The intensity of human threats differs inter-regionally, but globally, carnivore declines are consistently associated with anthropogenic habitat degradation, direct persecution and utilization, and diminished prey bases (Ripple et al. 2014). Increasingly, large carnivores are recognized as exerting disproportionate influence on ecosystem structure and function via trophic cascades which can extend beyond community dynamics and affect wildfire regimes, carbon sequestration, and biogeochemical cycles (Estes et al. 2011).

Given their high energetic demands (Carbone et al. 2007), large carnivores inherently exist at low population densities and range widely, increasing exposure to high-density human populations (Cardillo et al. 2004). Yet, populations of several species of large carnivores appear stable or increasing in Europe (Chapron et al. 2014) and North America (Gompper et al. 2015), suggesting that coexistence is attainable in human-dominated landscapes. Ensuring the long-term viability of these charismatic, ecologically important species demands solutions for their management within anthropogenic landscapes, as evidenced by the recent proliferation of studies demonstrating their use of human-altered systems (Gese et al. 2012, Dellinger et al. 2013, Merkle et al. 2013, Johnson et al. 2015).

The American Black Bear in New Jersey, USA

American black bears (*Ursus americanus*; hereafter black bears) followed a similar trajectory as other large carnivores up until the mid-20th century, having been extirpated, or nearly so, from much of their pre-European North American range (Hall 1981). However, the IUCN now lists black bears as a species of least concern, with stable or modestly increasing populations in North America (Garshelis and Hristienko 2006), bolstered by resilient populations in the mid-Atlantic states (Hurst et al. 2012).

Black bears were abundant throughout New Jersey (NJ), USA prior to European settlement (Abbot 1894, Regensburg 1978). However by the mid-1900's, unregulated killing coupled with habitat loss resulting from two centuries of timber extraction and agricultural conversion had severely reduced black bear population size to less than an estimated 100 individuals in northern counties (Lund 1980, McConnell et al. 1997). The NJ Fish and Game Council granted black bears "game animal" status in 1953, a protection that likely prevented the extirpation of the species from the state. From 1958-1970, 46 bears were reported harvested, and from 1971-2002 regulated hunting was closed altogether (Wolgast et al. 2010). Since its low point during the 1950s, the NJ black bear population has greatly increased in abundance, density, and in the extent of its spatial distribution (Carr and Burguess 2011). Multiple factors likely contributed to this robust recovery including: i) the 32-year hunting moratorium, ii) bear immigration from concurrently increasing populations in the adjacent states of Pennsylvania and New York, and iii) improved habitat quality associated with the maturation and integrated management of mid-Atlantic deciduous forests (McConnell et al. 1997, Carr and Burguess 2004).

Expanding black bear populations in recent decades have coincided with increasing human population densities coupled with a shift in human settlement patterns away from urban centers toward sprawling suburban, exurban, and rural communities across the northeastern USA (Hurst et al. 2012). Black bears are adaptive, opportunistic generalists, and as such exhibit a diversity of responses to changes in habitat quality resulting from forest management (Mitchell and Powell 2003). Further, they are capable of utilizing fragmented habitats in close proximity to high human densities and/or high anthropogenic disturbance by exploiting human-derived food sources (Merkle et al. 2013, Baruch-Mordo et al. 2014) and protected patchworks as reported in New Jersey (Fimbel et al. 1991).

Consequently, human-bear conflicts in NJ have rapidly proliferated over the past three decades and resulted in >1400 incidents of verified property damage, >400 livestock kills, >250 pet attacks and/or kills, seven human attacks, and one human fatality since 2001 (Carr and Burguess 2011). The New Jersey Division of Fish and Wildlife (NJDFW) spent in excess of $9 million USD on black bear management between fiscal years 2001-09, responding to over 26,500 human-black bear incidents, and has concluded that this level of human-bear conflict is fiscally and culturally untenable (Wolgast et al. 2010). As NJ possesses some of the highest black bear densities recorded (Huffman et al. 2010, Carr and Burguess 2011) and the greatest human densities in the USA, this social-ecological system provides an ideal model to evaluate how anthropogenic activities impact the ecological dynamics of a large carnivore species.

The Ecological Consequences of Anthropogenic Activities: Linking Wildlife Behavior, Spatial Ecology, and Population Dynamics

Wildlife Behavioral Responses to Human-Induced Rapid Environmental Change

An individual's interaction with its environment is mediated by its behavior (Sih et al. 2011); thus, individuals frequently exhibit behavioral modifications as an initial response to human-altered conditions (Tuomainen and Candolin 2011). Behavioral adjustments may prove adaptive by increasing survival, as evidenced by spatiotemporal shifts in habitat use and/or activity patterns to avoid humans (Wong and Candolin 2015). For example, male European red deer (*Cervus elaphus*) expeditiously switch to dense concealing habitat with the onset of hunting season (Lone et al. 2015), African lions (*Panthera leo*) alter their habitat use to evade seasonal movements of the Maasai and their livestock (Schuette et al. 2013), and urban red foxes (*Vulpes vulpes*) more frequently cross roads during periods of low traffic flow (Baker et al. 2007).

Adaptive benefits may also be conferred by behavioral responses that increase reproductive success in response to human activities (Wong and Candolin 2015). Examples include: North Atlantic right whales (*Eubalaena glacialis*) increasing the amplitude of their calls in response to maritime noise (Parks et al. 2011), male European tree frogs (*Hyla arborea*) ceasing calling activity during periods of loud traffic roar (Lengagne 2008), and urban great tits (*Parus major*) singing with a higher minimum frequency relative to wild conspecifics to distinguish their calls from the low-frequency anthropogenic background din (Slabbekoorn and Peet 2003).

However, human-induced rapid environmental change can also promote maladaptive behavioral scenarios (i.e., ‘evolutionary traps’), where there is a mismatch between environmental cues and conditions that evolutionarily may have bestowed high-quality habitats, mates, and/or food items, but now decrease realized fitness in human-dominated landscapes (Sih 2013). Evolutionary traps can result from maladaptive habitat selection (i.e., ‘ecological traps’), foraging behavior, navigation, oviposition, and mate selection (Robertson et al. 2013). Ecological traps can have especially pernicious consequences, as anthropogenic activities act to uncouple the cues individuals use to discern high-quality habitat from the positive outcomes historically associated with given cues (Robertson and Hutto 2006). For instance, increased prey availability near the border of the protected Phinda-Mkhuze Complex, South Africa, creates a ‘vacuum effect,’ persistently attracting leopards (*Panthera pardus*) from the reserve’s core who then experience substantially greater mortality risk from persecution (Balme et al. 2010).

Regardless of whether behavioral modifications resulting from anthropogenic environmental change prove adaptive or maladaptive, the extent to which they are even possible is ultimately determined by the plasticity of the behavior, which varies widely across behaviors and species. Behavioral plasticity, i.e., the extent in which animals may, or may not, modify their behaviors in response to heterogeneous environmental conditions, results from complex interactions between pre-programmed cue-response behaviors and learning from cumulative experiences (Mery and Burns 2009). Interindividual variation in behavioral tendencies (e.g., boldness, aggressiveness, activity exploration, sociability, etc.) that are consistently repeatable within individuals, stable over time, and correlated across contexts allows for the categorization of ‘animal personalities’ (Wolf and Weissing 2012). Even single personality dimensions can be indicative of fitness, as demonstrated by a comprehensive meta-analysis where ‘bold’ males across a diversity of taxa derived increased reproductive success, but incurred a cost in decreased survival probability

‘Behavioral syndromes’ arise when there exists between-individual consistency in the correlation of behavioral tendencies such as boldness and aggressiveness (Sih et al. 2004, Sih and Bell 2008). Individuals who tend to be more aggressive toward conspecifics also frequently respond more boldly to predation risk, as first established forty years ago in three-spined sticklebacks (*Gasterosteus aculeatus*) (Huntingford 1976). If single personality dimensions and/or behavioral syndromes have fitness consequences and an underlying genetic basis, they can then be viewed as phenotypic distributions apt to change akin to a conventional trait. In species with high behavioral plasticity, how might human-induced rapid environmental change be shifting these distributions?

Ursid Behavioral Responses to Anthropogenic Activities

Given their generalist life-history strategies and behavioral plasticity, both black and brown bears (*Ursus arctos*) are capable of rapidly modifying their behaviors in response to anthropogenic activities. Perhaps the most conspicuous and well-documented bear behavioral response to anthropogenic landscape transformation is their capacity to exploit human-derived foods including: garbage, agricultural crops, ornamental fruit trees, apiaries, livestock, bird feeders, pet food, bait stations, etc. (Davenport 1953, Horstman and Gunson 1982, Mattson 1990, Beckmann and Berger 2003a, Merkle et al. 2013, Baruch-Mordo et al. 2014, Hopkins et al. 2014, Massé et al. 2014, Johnson et al. 2015). The prevalence of individuals utilizing anthropogenic foods can quickly escalate within bear populations, as food-conditioned foraging behaviors are transmitted vertically from maternal sows to cubs (Mazur and Seher 2008). Black bear cubs become food conditioned through social learning via imitation of their mothers and/or trial and error; cubs reared in urban areas have a high probability of continuing to forage in urban areas when they become independent (Mazur and Seher 2008). However, this behavioral plasticity also allows bears to readily revert to reliance on natural foods in response to management efforts minimizing access to human-derived foods as evidenced in Yosemite (Hopkins et al. 2014) and Yellowstone (Cole 1974) National Parks.

When compared to 'wild' conspecifics, black bears that chronically forage on garbage are active for significantly fewer hours per day (8.5 vs. 13.3), shift those activities from crepuscular to nocturnal periods, enter dens later, and remain denned for significantly fewer days (Beckmann and Berger 2003a). Across seasons, both male and female black bears tend to be most active in urban landscapes (Lyons 2005) and in campgrounds (Ayres et al. 1986) during late night periods when human activity is lowest; however, subadult male brown bears were less risk-averse than adult females, more willing to exploit high-quality habitat adjacent to the high-speed, high-volume TransCanada Highway during time periods with less human activity (Gibeau et al. 2002).

Following experimental approaches by humans, GPS-collared Scandinavian brown bears avoided approaching observers by seeking dense, concealing cover, and subsequently altered their foraging and resting routines, also increasing movement during night-time hours (Ordiz et al. 2013b). Similarly, black bears outfitted with GPS and biologgers demonstrated a stress response, as indicated by elevated heart rates, when traversing agricultural areas lacking food and cover (Ditmer et al. 2015). However, in the absence of negative or positive stimuli, repeated neutral encounters between bears and humans, such as observing bears from a close distance, can foster ‘habituation,’ whereby bears mute their reactions and tolerate humans (Herrero et al. 2005). This capacity to habituate to human activity coupled with the behavioral plasticity evident in their activity and foraging patterns, has allowed recovering bear populations to occupy increasingly anthropogenic areas across the wildland-urban landscape gradient and resulted in escalating human-bear conflict.

Wildlife Habitat Use in Response to Human Landscape Transformation

The importance of examining patterns in habitat selection, and the representative resources therein, has long been recognized in ecology (Lack 1933, MacArthur and Pianka 1966). However, the recent wide-spread application of GPS technology, coupled with advancements in statistical methods, has produced a proliferation of work highlighting the need for precise definitions and appropriate inference in spatial ecology (Lele et al. 2013). Evaluating habitat and resource selection, home range dynamics, and landscape connectivity across the wildland-to-urban landscape gradient is further complicated by the reality that contemporary urban areas are hastily expanding in spatially complex, non-linear arrangements compared with slower growth in past centuries that primarily occurred linearly by the addition of concentric rings of development (Ramalho and Hobbs 2012). While recognizing that no single definition of a wildland-to-urban gradient is wholly adequate, using a two-dimensional continuum of dominant land cover coupled with human population density can help categorize degrees of human influence upon the landscape (e.g., Wildland, Exurban, Rural, Suburban, and Urban; Marzluff et al. 2008).

Human management of anthropogenic landscapes frequently produces more continuously available resources than spatially and temporally patchy resources found in adjacent wildlands (Shochat et al. 2006). Seasonal changes in the availability of food and water are dampened by extended growing seasons in temperate cities, year-round irrigation of perennial grasslands in arid cities, and direct and/or indirect feeding across urban areas (Shochat et al. 2004, Parris and Hazell 2005). As reviewed by Bateman and Fleming (2012), a multitude of medium-sized carnivore species have colonized and continuously occupy anthropogenic landscapes including: raccoons (*Procyon lotor*; Gross et al. 2012), badgers (*Meles meles*; Davison et al. 2008), gray foxes (*Urocyon cinereoargenteus*; Riley 2006), red foxes (*Vulpes vulpes*), coyotes (*Canis latrans*), and bobcats (*Lynx rufus*; Gehrt et al. 2010). Although they may not live exclusively within urban areas, numerous large carnivore species incorporate human-dominated areas within their home ranges including: both black and brown bears (reviewed below), cougar (*Puma concolor*), leopards (*Panthera pardus*), and spotted hyenas (*Crocuta crocuta*; Yirga et al. 2016).

Carnivores exhibit a diversity of selection patterns and home range modifications resulting from human transformation of the landscape, as illustrated by the following examples. Red wolves (*Canis rufus*) selected for human-associated land-cover types (i.e., agricultural fields, pine plantations, and early successional fields) over 'natural' land-cover types, as well as areas near secondary roads at the landscape level; however, avoidance of natural land-cover decreased as human densities increased (Dellinger et al. 2013). Urban coyotes avoided land-use types associated with human activity within their home ranges, but maintained home ranges twice as large as conspecifics in less-developed areas (Gehrt et al. 2009, Gese et al. 2012). Conversely, in a recent meta-analysis compiled from 411 articles, home range sizes of carnivores significantly decrease in six of eight species across three categories of landscape classification (i.e., natural, suburban, urban), including: striped skunk (*Mephitis mephitis*), stone marten (*Martes foina*), badger, red fox, coyote (different result than above, as meta-analysis reflects 46 studies), and bobcat (Šálek et al. 2015).

As home ranges are simply spatial representations of the composite of resources that carnivores select to meet specific life-history needs, decreasing home ranges across the wildland-urban gradient reflect shifts in the types and/or distributions of resources selected. Further, individual variation in selection or avoidance of anthropogenic resources is linked to the plasticity of the behavior for which the resource was selected, as evidenced by individual variation in the use of anthropogenic foraging sites by black bears.

Ursid Spatial Ecology in Human-Dominated Landscapes

Comparing three developed areas in the western US surrounded by mesic, high-quality bear habitat (Aspen, Colorado), moderately productive habitat (Durango, Colorado), and relatively xeric, poor-quality habitat (Lake Tahoe, Neveda), Johnson et al. (2015) found black bear resource selection for human development to be highly dynamic. Black bears increasingly selected anthropogenic areas in years when natural food production (e.g., acorns, serviceberries, and chokecherries) was low, and seasonally increased use of human development throughout summer-fall with the onset of hyperphagia. Female bears were more likely to select developed areas as they aged, and males in Aspen persistently used areas of intermediate development, although use was more pronounced in poor natural food years. Of interest here, individual bears across sites displayed varied responses in selection for anthropogenic areas, yet, population-level analyses suggested that bears in poor quality habitat may more consistently select for development across seasons and years (Johnson et al. 2015).

Seasonal differences were also observed in black bear selection of foraging sites in developed areas within Missoula, Montana, where the probability of bears foraging near houses increased during urban spring green-up and apple seasons, with males again, more frequently exploiting these resources. In contrast here though, black bears invariably selected these human-derived resources even when wild foods (five native berry species) were readily abundant; the authors conclude that the availability of certain anthropogenic resources, such as fruit trees, may represent strong attractants that outweigh associated risks (Merkle et al. 2013). These studies suggest that individual variation in the selection of foraging sites in anthropogenic areas may be associated with intrinsic factors (e.g., sex, age, body condition during hyperphagia) as well as extrinsic factors (e.g., variation in natural foods, quality of adjacent bear habitat, availability of strong attractants).

Black bears structure their home ranges to optimize resource use by incorporating resource-maximizing areas efficiently within an area-minimizing strategy (Mitchell and Powell 2007). Returning to Aspen, black bears that used urban areas in poor natural food years exhibited smaller home ranges and more nocturnal activity than in good natural food production years (Baruch-Mordo et al. 2014). Yet, in the comparatively diverse, deciduous forests of NJ, Pennsylvania, and West Virginia, black bears have been frequently shown to be resident on the edge of human-dominated areas (< 5 km) across all seasons, and no shifts in home ranges closer to developed areas were detected during food shortages (Tri 2013). In a striking example, black bears with ≥90% of their locations in urban areas in the western Great Basin, Nevada, and Lake Tahoe basin, California, displayed home ranges reduced by 90% for males and 70% for females compared to wild conspecifics (Beckmann and Berger 2003b). Whether bears disproportionately select anthropogenic areas in response to poor mast years (CO) or persistently do so (NJ, NV, CA), these studies demonstrate that urbanization consistently results in the selection of different resources with dissimilar spatial distributions as reflected in smaller home ranges. However, habituation and food-conditioning may only partly explain the spatial configuration of bears in and around anthropogenic development, as the territoriality of large, male bears also greatly influences spatial distributions.

While the occurrence of bears near human populations can result from habituation, and the persistent use of human-derived foods can be explained by food-conditioning, these may be considered only proximate mechanisms (Elfström et al. 2014a, Elfström et al. 2014b), because both are contingent upon bears having previously experienced humans and/or anthropogenic foods. Elfström et al. (2014a,b) contend that the 'despotic distribution hypothesis' represents the ultimate mechanism driving bear occupancy patterns in and around human settlements; the distribution of bears across the wildland-urban gradient is foremost a response to intraspecific predation avoidance and/or interference competition. Beckmann and Berger (2003b) argue that sex ratios skewed 4.25 times more toward males in urban-interface areas in the western Great Basin-Lake Tahoe region was best explained by population reallocation resulting from the despotic distribution model.

While habituation, food-conditioning, and intraspecific dynamics may alter black bear spatial ecology in human-dominated landscapes, ultimately we are interested in how individual behavioral variation, as reflected in selection for anthropogenic habitats and their associated resources, scales up to influence demography and population-level dynamics.

Wildlife Population Dynamics across the Wildland-Urban Landscape Gradient

In a recent meta-analysis, population densities increased with the degree of urbanization for three of six carnivore species including raccoon, red fox, and coyote

(Šálek et al. 2015). However, as high densities of individuals occupying low-quality habitat may result from despotic distributions (Andren 1990) and ecological traps (Battin 2004), Fischer et al. (2015) recently proposed the term *urban dweller* to represent species whose population growth rates are ≥ 1 in anthropogenic landscapes regardless of persistence in adjacent natural areas (i.e., species whose populations are stable or growing independent of immigration from wildlands; Fischer et al. 2015). Whereas *urban utilizers* occupy anthropogenic areas as foragers, but populations rely upon breeders dispersing from adjacent natural areas to persist (Fischer et al. 2015). Although initially the difference here may appear subtle, the management implications are important as conserving urban utilizers within the wildland-urban interface requires assessing limiting factors, spatial dynamics, and demography within and between both natural and anthropogenic landscapes.

Bears select natural and anthropogenic edges (Stewart et al. 2013), frequently occupy human-dominated areas across North America (Bateman and Fleming 2012), and achieve high densities within the wildland-urban interface in some landscapes, as is occurring in northwestern NJ (Huffman et al. 2010, Carr and Burguess 2011). However, questions remain in NJ regarding whether these densities are the result of an inherent demographic response to urbanization (i.e., increased fertility and/or decreased mortality rates) or are the result of a landscape-level reallocation, where urban areas may operate as population sinks.

Ursid Population Ecology in Human-Dominated Landscapes

Black bears in urban areas within the Great Basin-Lake Tahoe region attained densities three times greater than historical densities from the same areas, and urban-interface females had a higher proportion of potentially reproductive years producing three times the number of cubs as wild conspecifics; yet, rates of successful dispersal in these urban juveniles were half those of wild juveniles (Beckmann and Berger 2003b). Beckmann and Lackey (2008) report that higher age-specific fecundity rates in these urban females did confer increased fitness, given their increased age-specific mortality rates. They conclude that urbanization in the Lake Tahoe Basin is creating a population sink ($\lambda = 0.749$) and resulting in spatial reallocation from wildland to urban areas. Florida black bears (*U. americanus floridanus*) exhibited substantially higher adult female survival rates in the contiguous Ocala National Forest (0.966) than in the adjacent residential community of Lynne (0.776), but here cub survival was higher in the suburban (0.507) than in the natural (0.282) area (Hostetler et al. 2009). Similar to Lake Tahoe though, the population growth rate was less than one in the human-dominated area, and exceeded one for the wildland population (Hostetler et al. 2009). These three studies suggest that the increased fecundity or cub survival rates associated with black bear urbanization does not impact the population growth rate to the extent that declines in adult survival rates do.

The observed relationship in both Lake Tahoe and Florida tightly linking variation in adult female survival to changes in population growth rate is consistent with elasticity patterns reported in other black bear populations. Adult female survival was identified as having the greatest impact on black bear population growth rate in the Southeastern Coastal Plain, USA, amid growing concerns over reductions in this vital rate resulting from ongoing habitat fragmentation and human disturbance (Freedman et al. 2003). Similarly, population growth rate was most sensitive to changes in adult female survival in Banff National Park, Canada; however, here adult female survival was heavily influenced by management status, with problem bears exhibiting lower survival (0.66) than in adjacent hunted populations (Hebblewhite et al. 2003). Simulation approaches parameterized with the aforementioned Aspen black bear population data indicated that the lethal management of adult females, given their high elasticity, offset increased cub production resulting from the exploitation of human-derived foods in poor mast years, and high-removal scenarios induced rapid population declines (Lewis et al. 2014).

Although the high elasticity of adult female black bear survival has been identified across studies, it is important to note that natural selection has buffered this vital rate against temporal variability (Gaillard and Yoccoz 2003), and it was relatively invariant compared to the spatiotemporal variation in recruitment documented in a protected area of the Appalachian Mountains, North Carolina (Mitchell et al. 2009). A recent meta-analysis (Beston 2011) determined that despite the high elasticity of adult survival, differences between eastern and western black bear population growth rates were fundamentally driven by differences in reproduction, and highlighted that western populations tended to have higher survival (including 34% where $\lambda > 1$) whereas eastern populations where characterized by higher fecundity (including 55% where $\lambda > 1$).

Ultimately, human activities and landscape transformation profoundly affect black bear population dynamics, and as a result, conservation efforts must now prioritize innovative interventions outside of protected areas.

Summary of Chapter Objectives

Ultimately, managing black bears across human-dominated landscapes, requires an integrated approach founded on reducing benefits for bears in urban landscapes (Baruch-Mordo et al. 2013) and incorporating resident attitudes toward management actions (Don Carlos et al. 2009, Lowery et al. 2012) to balance the preservation of viable black bear populations, protect human welfare and property, and meet the needs of diverse stakeholders in a cost-effective manner (Hristienko and McDonald 2007). Questions remain however regarding what role harvest may (or may not) play as a component of an integrated management strategy to reduce human-bear conflicts in anthropogenic landscapes. As previously outlined, if we view individual variation in the propensity to exploit human-derived foods as a behavioral phenotypic distribution, and we recognize that human predators have the capacity to rapidly shift this distribution (Coltman et al. 2003, Darimont et al. 2009), how might harvest management be applied as a tool to reduce human-bear conflicts?

Cromsigt et al. 2013 recently made the argument that applied ecologists might do well to consider promoting “hunting for fear,” i.e., using approaches traditionally considered unethical (e.g., dogs, targeting calves, year-round seasons) as a pragmatic means to limit negative human-ungulate interactions. Similarly, the NJDFW policy of allowing bear hunters to use bait, may be reprehensible to some, but it also may be pragmatic, in reducing harvest success uncertainty (a concern of Bischof et al. 2012), and may also promote the disproportionate take of food-conditioned, nuisance bears. In addition to the direct removal of problem bears, harvest may shift behavioral distributions indirectly as the ecology of fear likely applies to large, terrestrial carnivores whose activities are shaped by a distinct cause of fear, human predation (Oriol-Cotterill et al. 2015).

Conserving large carnivores in human-dominated landscapes may require a 'Landscape of Coexistence,' whereby refugia with low human-caused mortality risk are allowed to persist, and the fear of humans is allowed to dominate in areas with high human-caused mortality risk (Oriol-Cotterill et al. 2015). However, Ordiz, Bischof, and Swenson 2013a have expressed concern that management attempting to instill fear into apex predators may perversely limit their capacity to create the sought-after landscape of fear, with all of its ecological reverberations. Although recent work suggests otherwise, cougars actually increased kill rates, and decreased site fidelity and overall carcass consumption, as a function of increasing housing density (used as a proxy for human-induced fear; Smith et al. 2015). The extent and rate in which NJ harvest regimes may be influencing black bear behavior remains unexamined.

Herein, I link the ecological consequences, including behavioral, spatial, and demographic responses, of black bears to anthropogenic activities and landscape transformation within an archetypal human-dominated landscape. My second chapter examines how nuisance and threatening black bear behaviors, as well as age and sex, relate to the probability of harvest, lethal management, and other sources of mortality, such as vehicle strikes. I also assessed correlations between temporal trends in human-bear conflict reports and harvest and lethal management rates. My third chapter evaluates the intrinsic factors (e.g., sex, age, conflict history) associated with black bear spatial transitions across the wildland-urban landscape gradient. My fourth chapter quantifies how human landscape transformation influences black bear fertility and ultimately creates a source-sink dynamic between wildland and anthropogenic habitats. Building upon previous chapters, we demonstrate the importance of regulated harvest in reducing urban bear populations and associated human-bear conflicts. My fifth chapter was developed as an educational "case-study" to be used in undergraduate ecology courses to demonstrate the importance of objective population ecology in guiding real-world wildlife management issues.

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CHAPTER 2

RECREATIONAL HARVEST AND INCIDENT-RESPONSE MANAGEMENT REDUCE HUMAN-CARNIVORE CONFLICTS IN AN ANTHROPOGENIC LANDSCAPE[1]

Summary

1. Conserving viable large carnivore populations requires managing their interactions with humans in increasingly anthropogenic landscapes. Faced with declining budgets and escalating wildlife conflicts, agencies in North America continue to grapple with uncertainty surrounding the efficacy of socially divisive management actions such as harvest to reduce conflict.
2. We used multistate capture–reencounter methods to estimate cause-specific mortality for a large sample (>3500) of American black bears *Ursus americanus* in north-western New Jersey, USA over a 33-year period. Specifically, we focused on factors that might influence the probability of bears being harvested, lethally managed, or dying from other causes. We further analysed temporal correlations between >26,000 human–black bear incidents reported between 2001–2013 and estimates of total mortality rates, and specifically, rates of harvest from newly implemented public hunts and lethal management.

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[1] Raithel, Jarod D., Melissa J. Reynolds-Hogland, David N. Koons, Patrick C. Carr, and Lise M. Aubry (2016). Recreational harvest and incident-response management reduce human-carnivore conflicts in an anthropogenic landscape. *Journal of Applied Ecology*. DOI: 10.1111/1365-2664.12830

3. Adult females were twice as likely (0.163 ± 0.014) as adult males (0.087 ± 0.012) to be harvested during the study period. Cubs (0.444 ± 0.025) and yearlings (0.372 ± 0.022) had a higher probability of dying from other causes, primarily vehicle strikes, than adults (0.199 ± 0.008). Reports of nuisance behaviours in year $t + 1$ declined with increasing mortality resulting from harvest plus lethal management in year t ($P = 0.028$, $R^2 = 0.338$). Adult bears previously designated as a nuisance and/or threat were more likely to be harvested (0.176 ± 0.025) than those never identified as a problem (0.109 ± 0.010). Across age classes, individuals assigned problem status, were significantly more likely to be lethally controlled.
4. *Synthesis and applications.* Given continuing failures in conserving exploited carnivores, their recreational harvest and lethal management remain polarizing. Within this social-ecological system, the well-regulated harvest of carefully monitored black bear populations represents a pragmatic approach to achieve population objectives. Further, the integration of harvest and incident-response management (both lethal and non-lethal practices) with educational programs aimed at reducing anthropogenic attractants can result in subsequent reductions in problem behaviours reported.

Introduction

As humans continue to rapidly transform landscapes into novel social-ecological systems (Fischer *et al.* 2015), conservation in the Anthropocene (Corlett 2015) will increasingly entail minimizing human-wildlife conflicts (Soulsbury & White 2016). One of the complexities inherent to these systems is that ecological and cultural carrying capacities can radically differ, yet dynamically interrelate (Levin *et al.* 2013). Thus, managers are not only charged with ensuring wildlife population viability, but must also heed shifting stakeholder acceptance (Marchini 2014). Some proactive solutions to human-wildlife conflicts are publically acceptable such as intensively managing raptors nesting on power infrastructure in South Africa (Jenkins *et al.* 2013) or applying deterrents to reduce elephant crop raiding in Kenya (Graham & Ochieng 2008). However, other management actions, such as creating 'landscapes of fear' to reduce human-ungulate (Cromsigt *et al.* 2013) and human-carnivore (Oriol-Cotterill *et al.* 2015) conflicts are controversial.

Many iconic members of the order Carnivora continue to experience precipitous population and range declines globally (Di Marco *et al.* 2014). However, populations of several species of large carnivores appear stable or increasing in Europe (Chapron *et al.* 2014) and North America (Gompper, Belant & Kays 2015), suggesting that coexistence is attainable in human-dominated landscapes. Nevertheless, the use of recreational hunting to manage large carnivores tends to elicit strong emotional responses from the public (Slagle, Bruskotter & Wilson 2012) whose attitudes are strikingly bimodal (Smith, Nielsen & Hellgren 2014), can become more entrenched as tolerance diminishes (Treves, Naughton-Treves & Shelley 2013), and split along broad cultural lines (Gangaas, Kaltenborn & Andreassen 2015). Lethal management to remove problem individuals is viewed more favourable by those living alongside carnivores, but may not improve tolerance (Browne-Nuñez *et al.* 2015) as liberalizing management culling may result in the increased acceptability of poaching (Chapron & Treves 2016). Sociological analyses aside, the functional removal of apex predators can have unanticipated and far-reaching ecological consequences (Estes *et al.* 2011). Managing carnivores via recreational harvest has recently been criticized for i) not incorporating uncertainty in estimated harvest, ii) failure to maintain sustainable populations, iii) failure to reduce conflicts over property loss and competition for game species, and iv) not building political support for carnivore conservation (Treves 2009; Bischof *et al.* 2012). For example, recent work examining the use of recreational harvest to reduce human-bear conflicts (hereafter conflicts) is equivocal. Higher bear harvests did not reduce conflicts at the landscape scale in Ontario (Obbard *et al.* 2014), nor state wide in Wisconsin (Treves, Kapp & MacFarland 2010); however, it did prove effective regionally in Pennsylvania (Ternent 2008).

American black bears *Ursus americanus* (hereafter black bears) followed a similar trajectory as other large carnivores up until the mid-20th century, having been extirpated, or nearly so, from much of their pre-Columbian North American range. However, the IUCN now lists black bears as a species of least concern, with stable or modestly increasing populations in North America (Garshelis & Hristienko 2006). Since its low point during the 1950's, the New Jersey (NJ), USA, black bear population has greatly increased in abundance, density, and in the extent of its spatial distribution (Carr & Burguess 2011). Black bear recovery has coincided with increasing human population densities, coupled with a shift in settlement patterns away from urban centres toward sprawling suburban communities. Consequently, conflicts in NJ have rapidly proliferated over the past three decades and resulted in >1400 incidents of verified property damage,

>400 livestock kills, >250 pet attacks and/or kills, seven human attacks and one human fatality since 2001. The New Jersey Division of Fish and Wildlife (NJDFW) spent in excess of $9 million USD on black bear management between 2001-2009, and has concluded that this level of conflict is fiscally and culturally untenable (Wolgast *et al.* 2010).

Black bear harvest often represents a source of additive mortality (Obbard & Howe 2008) that negatively influences population growth rate given its sensitivity to small changes in adult female survival (Hebblewhite, Percy & Serrouya 2003). Interestingly, bear populations managed for sustained harvest appear less prone to population declines compared with those where hunting is prohibited, as regulated harvest may decrease illicit take by enlisting consumers with long-term interests in the use of the resource (Garshelis 2002). Attitudes surrounding the introduction of black bear harvest and lethal management in NJ are nuanced, and more complex than a simple distillation into pro- and anti-hunting perspectives (Johnson & Sciascia 2013). Despite this diversity of opinions, the majority of stakeholders seek science-based information from wildlife managers (Campbell & Mackay 2009). Questions remain regarding the efficacy of hunting in reducing property damage given the presumed difficulty hunters would face in targeting offending individuals (Treves 2009), disconnects between the age- and sex-classes of harvested animals versus those of offending individuals (Treves, Kapp & MacFarland 2010), and localized age-structure perturbations resulting from spatio-temporal dynamics initiated by harvest (Robinson *et al.* 2008).

Our objective is to quantify cause-specific mortality rates of black bears from harvest and lethal management, relative to other sources of mortality, as a function of sex, age-class, and assigned behavioral classification (problem vs. normal status) by utilizing long-term capture-reencounter data from northwestern NJ. We also examine the extensive NJDFW incidents database to determine if temporal trends in normal/nuisance/threatening behaviors reported since 2001 are associated with annual variation in harvest, lethal management, and total mortality rates. If we observe declines in reports of problem behaviors following increases in annual harvest and lethal management rates, we expect that problem bears should be disproportionately harvested and lethally controlled, relative to bears never exhibiting undesirable behaviors. Further, the age-sex profiles of bears captured in response to problem incidents should be congruent with those of harvested bears. It is important to note here that NJDFW's comprehensive black bear management policy has always included educational programs, and a substantial investment in outreach was made during 2007-2014. This social-ecological system provides a model to test whether recreational harvest and incident-response management, when coupled with sustained educational outreach, help reduce undesirable bear behaviors in a landscape with high black bear densities and the greatest human densities in the USA.

Materials and methods

Study area

Data were collected as part of the long-term research and management of black bears by the NJDFW, primarily in northwestern NJ, USA (41˚04' N, 74˚40' W; Fig. 2-1). The study area is described in detail in Makkay (2010) . Black bear abundance increased from 450-500 in 1996 (McConnell *et al.* 1997) to 3200-3400 in 2010 (Carr & Burguess 2011). A limited black bear harvest was first reinstated in NJ in 2003, following closure for over three decades. The 2004 season was closed by NJ Supreme Court order, in response to public objection. A 2005 harvest occurred under the 2003 parameters, but was again closed from 2006-2009 pending the development and approval of the NJ Fish and Game Council Comprehensive Black Bear Management Policy, reopened in 2010, and continues today (Wolgast *et al.* 2010). The NJ black bear hunting season is a lottery framework that lasts six days in early December, concurrent with the firearm deer season. Participants must possess a permit and are limited to one bear per season. Participants may employ bait while hunting from the ground and from elevated stands at least 300 feet from the bait. There are no restrictions on age, size or sex of targeted bears, or on females with cubs; however, taking/disturbing bears in dens or on open nests is prohibited. Successful hunters must take the harvested bear to a designated hunter check station the day of the kill where NJDFW personnel record sex, weight, and extract a tooth for cementum analysis.

Data collection

From 1981-2014, NJDFW personnel conducted 5,185 black bear captures, marking 3,533 unique individuals (1614 females, 1919 males), including 1,344 cubs of the year, 877 yearlings, and 1,312 adults. 1,256 of the young bears reached an age >2 years; thus, our adult age class includes 2,568 unique individuals. The cause of mortality was documented for 1,338 of these marked individuals, consisting of 556 hunter harvests, 396 management mortalities (158 euthanized, 238 agricultural depredation permits), and 386 other mortalities (primarily composed of 271 vehicle strikes and 58 illegal kills). Current capture protocols are described in detail in Appendix 1. In November 2000, the NJDFW implemented the Black Bear Rating and Response Criteria (BBRRC), a standardized framework for responding to bears deemed a threat to human safety, agricultural crops and/or property, or exhibiting nuisance behaviour. All bears from this point forward, as well as all captures dating back to 1987, were consistently designated by NJDFW managers as one of the following three behavioral categories: I) Threat: including human, livestock, and unprovoked pet attacks, home entries, and agricultural/property damage >$500 USD; II) Nuisance: including habitual visits to garbage containers, dumpsters and/or birdfeeders, and property damage <$500 USD; and III) Normal: including bears observed by hunters, hikers, or campers in bear habitat, or dispersing animals that wander through rural and suburban communities. Threatening bears are lethally controlled as soon as possible throughout the year. Nuisance bears, if trapped, are aversively conditioned on-site using rubber buckshot, pyrotechnics, and bear dogs. Additionally, NJDFW received 26,582 incident reports from the general public between 2001-2013, and categorized those according to BBRRC as 2,277 Threats, 12,013 Nuisances, and 12,292 Normal interactions.

Capture-reencounter model for cause-specific mortality

We analyzed the capture-reencounter data using a multistate framework with an alive state (A), and three dead states for individuals that were harvested (H) (i.e., legally taken by the public during 2003, 2005, or 2010-2012), lethally controlled via NJDFW personnel or agricultural depredation management (M), or died from any other cause (D) following Bischof *et al.* (2009) and Koons, Rockwell & Aubry (2014). Fixing survival probabilities for individuals in state A to 1, and H, M, and D to 0, allowed us to estimate the probability of individual *i* dying from cause *k* between year *t* and *t*+1 ($\mu_{i,t}^{k}$) via the 'transition' probabilities (and thus survival becomes 1- $\sum_k \mu^k$; Fig. 2-2). Additionally, transition probabilities between dead states (H, M, D) were fixed to 0 because each dead state is an absorbing state (see Schaub & Pradel 2004). Transition probabilities from the A to H state were fixed to 0 in years when harvest moratoria were in place. We estimated $\mu_{i,t}^{k}$ conditional on state-specific probabilities of recapturing each live individual *i* in state A in year *t* ($p_{i,t}^{\mathrm{A}}$) and the probabilities of recovering an individual who died from cause *k* ($r_{i,t}^{k}$). As hunters, NJDFW personnel, and farmers are required to report all harvests, euthanized individuals, and authorized depredation kills, respectively, the $r_{i,t}^{\mathrm{H}}$ and $r_{i,t}^{\mathrm{M}}$ detection probabilities were fixed to 1.

Data analyses

We used package RMark (Laake 2013) within Program R version 3.1.2 (R Core Team 2016) to estimate multistate model parameters, and calculate Akaike's information criterion adjusted for sample size ($\mathrm{AIC_c}$) to compare the predictive performance of hypothesized models (annotated R code available in Appendix 2.9; Burnham & Anderson 2002). We applied simulated annealing in an effort to estimate global maximum likelihoods and avoid convergence on local maxima. An initial exploration of full time variation in $p_{i,t}^{\mathrm{A}}$ and $r_{i,t}^{\mathrm{D}}$ indicated that both recapture and recovery probabilities were relatively high during the 1980s (mean $p_{i,1981-89}^{\mathrm{A}}$ = 0.299 ± 0.106; mean $r_{i,1981-89}^{\mathrm{D}}$ = 0.432 ± 0.164) when the bear population was small and geographically restricted. Detection probabilities decreased throughout the 1990s as the population grew and expanded, but capture efforts remained constant (mean $p_{i,1990-99}^{\mathrm{A}}$ = 0.179 ± 0.039; mean $r_{i,1990-99}^{\mathrm{D}}$ = 0.278 ± 0.098). During the 2000s, as the population increased three-fold, dead recovery probabilities declined again (mean $r_{i,2000-14}^{\mathrm{D}}$ = 0.222 ± 0.052), but recapture probabilities remained unchanged (mean $p_{i,2000-14}^{\mathrm{A}}$ = 0.180 ± 0.025), as NJDFW appropriated greater resources toward black bear research and management beginning in 2001. Rather than expending degrees of freedom on a fully time-dependent model and losing precision in parameter estimates, we assessed temporal variation in detection probabilities by comparing the following parameterizations: three decadal time bins (1981-1989; 1990-1999; 2000-2014), two time bins (pre and post-2001), as well as quadratic and cubic time trend functions. Following selection of the best time-varying parameterization for $p_{i,t}^{\mathrm{A}}$ and $r_{i,t}^{\mathrm{D}}$, we next incorporated potentially influential covariates

(Garshelis & Noyce 2006), as well as interactions between sex, age-class (recaptured/recovered at age 1, 2, and 3+), and NJDFW's BBRRC (we collapsed bears classified as threats and/or nuisances into a single categorical variable, designated as 'problem' behaviour). After establishing the best performing model for detection and recovery probabilities, we retained this parameterization while modelling cause-specific mortality probabilities $\mu_{i,t}^{\mathrm{H}}$, $\mu_{i,t}^{\mathrm{M}}$, and $\mu_{i,t}^{\mathrm{D}}$. We initially assessed temporal variation in transition probabilities by parameterizing time, as described above. We constructed the final candidate model set using our best time-varying parameterization for transition probabilities in combination with sex, age-class, and behavioral variables, including ecologically-meaningful interactions.

We included total incidents reported annually between December 1st 2000 and November 31st 2013 by BBRRC behavioral category (threat, nuisance, normal) and all sub-categories (e.g., garbage visits, home entries). Incorporating the best performing model for detection/recovery probabilities and full-time variation in transition probabilities, we estimated annual cause-specific mortality probabilities during 2001-2012 by backtransforming multinomial logit ('mlogit') link estimates (see C.17, Cooch & White 2012). We then regressed change in BBRRC incidents reported by behavioral category between year t and $t + 1$ against both annual total mortality and harvest plus management mortality probabilities, in year t. Normality assumptions were met, as assessed by Lilliefors' test using R package nortest ($p = 0.477$, $p = 0.239$, respectively; Gross & Ligges 2015). To account for uncertainty in the relationship between conflict records and annual estimates of cause-specific mortality, we used a Monte Carlo simulation approach. We sampled 1000 cause-specific mortality probabilities from beta distributions defined by respective estimates of standard error. For each iteration, we estimated the intercept and slope of the relationship between conflicts and sampled mortality probabilities, and then generated a mean slope relationship with associated 95% confidence intervals from the 1000 iterations (Wolfe *et al.* 2015). Lastly, we used χ^2 tests to compare sex- and age-class ratios of bears trapped in response to complaints with those of harvested bears in our marked sample.

Results

A quadratic time trend in both $p_{i,t}^{\mathrm{A}}$ and $r_{i,t}^{\mathrm{D}}$ best explained time variation in these parameters ($w_i = 0.748$; see Appendix 2.3), and was appreciably more parsimonious than full-time specificity. Further, sex was identified as the most important covariate for both $p_{i,t}^{\mathrm{A}}$ ($w_i = 0.710$) and $r_{i,t}^{\mathrm{D}}$ ($w_i = 0.290$; see Appendix 2.4). The best overall model for $p_{i,t}^{\mathrm{A}}$ and $r_{i,t}^{\mathrm{D}}$ included a quadratic time trend for $r_{i,t}^{\mathrm{D}}$, and retained sex for $p_{i,t}^{\mathrm{A}}$ ($w_i > 0.999$; see Appendix 2.5). The temporal pattern in detection probabilities is consistent with an increasing and expanding black bear population, somewhat mitigated by substantial increases in trap effort post-2001 (see Appendix 2.2). Mean female recapture probability (0.492 ± 0.040) exceeded that of males (0.252 ± 0.019). When incorporating the best-performing model structure for detection probabilities, the top-ranked model for cause-specific mortality included an interaction between age and sex for harvest mortality, and age alone for predicting the probability of being lethally managed and dying from all other causes ($w_i = 0.990$; Table 2-1; see Appendix 2.6 and 2.7). Adult females (> 2 years-old; 0.163 ± 0.014) and yearling males (0.233 ± 0.031) were more likely to be harvested than adult males (0.087 ± 0.012), and ‘problem’ adults were more likely to be harvested (0.176 ± 0.025) than ‘normal’ adults (0.109 ± 0.010; Table 2). Cubs (0.444 ± 0.025) and yearlings (0.372 ± 0.022) were more likely to die from all other causes than adults (0.199 ± 0.008; Table 2-2).

In years immediately following the reintroduction of recreational harvest, the total number of nuisance incidents reported, including all nuisance subcategories (e.g., garbage visits, property damage < $500), consistently and substantially declined (Fig. 2-3; see Appendix 2.8). In years immediately following the suspension of harvest, total nuisances across all subcategories consistently rose (Fig. 2-3; see Appendix 2.8). Threat behaviors were less frequently reported, but exhibited analogous patterns. The proportion of bears reported to NJDFW displaying normal behavior relative to problem behaviors began to increase after 2008, three years following the second harvest suspension, and one year following significant NJDFW investment in a concerted educational outreach campaign, and continued throughout the extent of this study (Fig. 2-3). The change in nuisance behaviors reported between year t and $t + 1$ was negatively correlated with increasing harvest plus management mortality in year t ($P = 0.028$; $R^2 = 0.338$); this relationship was further supported when we accounted for uncertainty in cause-specific mortality probabilities using Monte Carlo simulations (Fig. 2-4). However, the change in nuisance behaviors reported between year t and $t + 1$ was only weakly correlated with increasing total mortality in year t ($P = 0.081$; $R^2 = 0.201$; Fig. 2-4). The proportion of cubs/yearlings captured compared to adults captured as part of the BBRRC incident-response protocol (30.4% young: 69.6% adult, $n = 872$) did not differ ($\chi^2 = 0.508$, $P =$

0.476) from those harvested in our marked sample (32.6% young: 67.4% adult, $n = 556$). Adult harvested bears that were previously marked were predominately female (68.8%, $n = 375$, $\chi^2 = 26.7$, $P < 0.001$), as were adult bears captured in response to incidents (54.7%, $n = 607$, $\chi^2 = 2.5$, $P = 0.114$).

Discussion

This study provides evidence that the introduction of a recreational black bear harvest can be an effective tool to help managers achieve population objectives. Under the aforementioned harvest regulations, adult female black bears were almost twice as likely to be harvested as adult males (Table 2-2). A harvest which disproportionately decreases the survival of adult females, a vital rate with high elasticity selected to exhibit low variance, will appreciably impact the population growth rate, and thus, requires vigilant monitoring. Age-class was identified as an important predictor of all sources of mortality, and this was most evident in young bears which were more susceptible to mortality from other causes, primarily vehicle strikes. Additionally, young males were more likely to be harvested than adult males (Table 2-2). These results were unsurprising, as black bears are capable of experiential learning, evidenced by alterations in their activity patterns in response to human-induced perturbations (Beckmann & Berger 2003a). This behavioral plasticity, so advantageous in undisturbed habitats, may ultimately be highly detrimental in NJ, as young bears reared on anthropogenic food sources are more likely to continue to do so as independent subadults (Mazur & Seher 2008), experiencing greater road exposure and mortality risk from vehicle strikes (Beckmann & Berger 2003b).

Total nuisance behaviors, including every nuisance sub-category, repeatedly declined in the year following each of the 5 harvests (e.g., -37% in 2004; Fig. 2-3). The years immediately following harvest moratoria then exhibited mirrored increases in nuisances reported (e.g., +37% in 2005; see Appendix 2.8). However, the relationship between changes in nuisances reported between year *t* and *t* + 1 was only marginally correlated with total mortality in year *t* (Fig. 2-4). The change in nuisances reported in year *t* + 1 was better explained by harvest plus lethal management mortality rates in year *t* (Fig. 2-4), suggesting that problem bears were being disproportionately harvested and lethally controlled. The best performing multistate model which included behavioral covariates indicated that adult problem bears were significantly more likely to be harvested than adults never having been designated a problem (Table 2-2). However, the opposite pattern was detected for yearlings as very few independent individuals were trapped as a problem prior to their harvest. Adult bears within 5 km of urban areas in NJ are capable of shifting from areas of relatively higher to lower harvest vulnerability at the initiation of the hunting season (Tri 2013); however, our analyses suggest that food-conditioned bears may be less apt to do so. Unsurprisingly, problem behavior increased the probability of being lethally controlled by 2-6 times. The significant increase in the probability of ‘normal’ yearlings lethally controlled was due to the large number of yearlings critically injured by vehicle strikes and subsequently euthanized.

In examining the mechanisms underlying bear occupancy in and around anthropogenic areas, it is meaningful to distinguish between proximate and ultimate drivers. While the occurrence of bears near humans can result from habituation, and the persistent use of anthropogenic foods can be explained by food-conditioning, these may be considered only proximate mechanisms because both are contingent upon bears having previously experienced humans and/or their foods (Elfström *et al.* 2014b). The distribution of despots (i.e., intraspecific predation avoidance and/or interference competition) may be the ultimate mechanism driving bear occupancy patterns in and around settlements (Beckmann & Berger 2003b; Elfström *et al.* 2014a). Further, the disproportionate occurrence of sex, age and reproductive classes exploiting urban areas can be informative about the availability of strong attractants and the habitat quality of adjacent wildlands (Elfström *et al.* 2014b). Treves, Kapp & MacFarland (2010) reported no relationship between harvest and subsequent reductions in conflicts in Wisconsin; however, the age and sex profiles of black bears trapped following complaints were incongruent with those of harvested bears. In NJ, however, age-sex profiles of incident-response captures were consistent with those of the marked individuals harvested.

Obbard *et al.* (2014) also found no correlation between prior harvest and ensuing conflicts; instead, conflict was associated with variation in natural food availability across Ontario. Similarly, in the western USA, inter-seasonal and -annual variation in black bear use of human-derived resources is inversely related to variation in the production of natural forage (Johnson *et al.* 2015). We acknowledge that if bears only exploit anthropogenic foods during years of scarcity, then an individual previously designated a nuisance will not necessarily exhibit problem behavior(s) during the year of its harvest. Yet, in the comparatively diverse deciduous forests of NJ, Pennsylvania, and West Virginia, black bears have been frequently shown to be resident on the edge of human-dominated areas (< 5 km) across all seasons, and no shifts in home ranges closer to developed areas were detected during natural food shortages (Tri 2013). Further, highly desirable anthropogenic attractants, like fruit trees, likely increase the probability of conflict, regardless of whether or not natural foods are readily available (Merkle *et al.* 2013). Additional work remains to fully understand the causes associated with, and frequencies of, bears transitioning across the wildland-urban gradient in NJ.

It is important to note that public complaints about nuisance bear activity are a function of both the frequency of interactions, *and* the rate at which people report events. The latter can be heavily influenced by how people perceive controversial management decisions (Howe *et al.* 2010) as occurred with the reintroduction and subsequent rapid closure of bear harvest in NJ. Although we observed a numeric decrease in complaints recorded following the first two harvests, we did not observe a decline in the proportion of problem relative to normal bears reported (Fig 2-3). This may be partly a consequence of local stakeholder anger over 2004 and 2006 closures, resulting in decreased tolerance and increased reporting rates of problem behaviours. However, beginning after 2008, and continuing through 2013, there was a consistent decline in the proportion of problem relative to normal behaviors reported. Notably in 2008, NJDFW substantially invested in delivering 204 educational outreach presentations reporting 24,215 attendees, and continued these efforts throughout the study duration (NJDFW 2015). Despite these correlations, we cannot exclude the alternative explanation that changes in reporting rates are ultimately driven by public perception and not underlying changes in bear behavior.

Reducing available anthropogenic foods by 55-70% is the most cost-effective strategy in eliminating most bears from risking entering urban landscapes (Baruch-Mordo *et al.* 2013). Unfortunately, NJDFW does not have the authority to require bear-resistant garbage containers within residential communities, and purchasing appropriate receptacles remains voluntary. While harvest may represent one management tool to disproportionately remove bears currently a nuisance, conflict will continue until strong attractants are substantially reduced. Relative to normal bear behaviors, we did not observe declines in reported problem behaviors prior to the concerted educational outreach campaign perhaps because home ranges vacated by harvested nuisance sows were quickly occupied by offspring reared on readily available human-derived foods (J.D. Raithel, unpublished data). Our top-ranked model, which received overwhelming support in model selection (w_i = 0.990; Table 2-1), indicated the disproportionate harvest of adult females. Given the magnitude of these harvest estimates, it is plausible that the removal of adult females was inducing subsequent declines in abundance, and the associated declines in problem behaviors were simply a numeric response. However, following the harvests between 2010-2012, a period which also included substantial educational outreach, reports of nuisance bears fell more sharply than those of ‘normal’ bears (Fig. 2-3), suggesting declines in conflict may be driven by more than declining abundance alone. Educational outreach may have resulted in increased containment of bear attractants and when coupled with the disproportionate harvest of ‘problem’ bears, may help explain decreasing conflicts. In addition, reoccurring public hunts may be establishing a ‘landscape of fear’ for these large carnivores, promoting spatio-temporal avoidance of habitats in and near anthropogenic areas with high harvest vulnerability, and thereby decreasing their probability of engaging in nuisance behaviors (Oriol-Cotterill *et al.* 2015).

With average human population densities exceeding 467 individuals/km^2 and outward urban expansion now consuming land at more than double the per-capita consumption of development prior to 1986 (Hawkins *et al.* 2006), New Jersey today represents a harbinger of the anthropogenic transformation coming in future decades across much of North America. Here, there simply are not enough large, contiguous tracts of wildlands remaining to alone support viable carnivore populations, necessitating that conservation approaches in NJ focus on coexistence (Chapron *et al.* 2014). Densities of large mammals inhabiting the matrix of wild and developed areas will ultimately be determined by cultural carrying capacities, and managers' ability to achieve these population targets given budgetary constraints. Assessing means to increase cultural carrying capacities was beyond the scope of this study, but involves recognizing that local stakeholders' perceptions depend upon their knowledge of carnivores, ability to participate in management decisions, and economic factors (see Young *et al.* 2015).

Hristienko & McDonald (2007) conducted a comprehensive meta-analysis of trends in black bear populations and conflicts, in relation to varied management approaches to harvest, across 52 states and provinces in North America. They propose that managing black bears in the 21st century requires agencies to balance preservation of viable black bear populations, protecting human welfare and property, and meeting the needs of their diverse stakeholders in a cost-effective manner. Our case-study supports the authors’ conclusions that this balance is achievable, even in human-dominated landscapes, when management integrates public harvest, incident-response protocols for applying non-lethal and lethal management, and continued investment in educational outreach regarding waste management. We recommend carefully-regulated and adaptive harvest for black bears in anthropogenic landscapes be considered foremost a tool to meet cultural carrying capacity, and when coupled with incident-response management, an additional means to reduce problem bears. We suggest implementing bear harvest only when consistent monitoring, coordinated educational programming, and an incident-response framework are already in place, and encourage agencies already successfully managing sustainable harvests to continue to emphasize minimizing available anthropogenic foods.

Data accessibility

Multistate capture-mark-reencounter histories: Dryad entry DOI:10.5061/dryad.08fc8 (Raithel *et al.* 2016)

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Tables and figures

Table 2-1. Candidate model ranking for mortality (μ) of black bears dying from harvest (*H*), lethal management (*M*), or 'dead other' (*D*) between 1981-2014 in New Jersey, USA. Explanatory variables include age class (cub, yearling, adult), sex, and behavior (problem individuals, and bears never having been classified as a nuisance and/or threat). Time is parameterized with a cubic function, as selected from previous analyses. All candidate models below include the top-ranked model for probabilities of live recapture ($p_{i,t}^{\mathrm{A}}$) and dead recovery ($r_{i,t}^{\mathrm{D}}$). Interactions are represented by a colon, and the top model is denoted in bold.

Mortality (transition to state H, M, or D)			**Model performance**			
To Harvest ($\mu_{i,t}^{\mathrm{H}}$)	**To Management ($\mu_{i,t}^{\mathrm{M}}$)**	**To All Other ($\mu_{i,t}^{\mathrm{D}}$)**	**Δ AIC$_c$**	**NP**	***w$_i$***	**Deviance**
age: sex	**age**	**age**	**0.000**	**19**	**0.990**	**470099.7**
age	age	age	9.221	16	0.010	470115.0
age: sex	time	age	24.063	23	0.000	470115.7
age	time	age	33.085	20	0.000	470130.8
age: behavior	age: behavior	age	54.146	22	0.000	470147.8
age: sex	age: behavior	age	56.416	22	0.000	470150.1
age: behavior	age: behavior	time	132.097	26	0.000	470217.7
sex: behavior	age: behavior	time	143.540	24	0.000	470233.2
sex: behavior	behaviour	age	163.431	16	0.000	470269.2
sex: behavior	time	time	238.296	22	0.000	470332.0
age: behavior	time	time	245.830	24	0.000	470335.5
age: sex	age	time	260.723	23	0.000	470352.4
age: sex	time	time	264.480	24	0.000	470354.1
age: behavior	age	time	264.783	23	0.000	470356.4
age	age	time	269.255	20	0.000	470367.0
sex: behavior	age	time	274.460	21	0.000	470370.1
age	time	time	283.500	21	0.000	470379.2
null	time	null	285.701	16	0.000	470391.5
time	null	time	287.100	19	0.000	470386.8
time	time	null	287.100	19	0.000	470386.8
null	time	time	287.100	19	0.000	470386.8
null	null	null	327.992	10	0.000	470445.8

Table 2-2. Black bear cause-specific mortality estimates in New Jersey, USA monitored between 1981-2014. Parameter estimates ($\mu^{\mathrm{H}}_{i,t}$, $\mu^{\mathrm{M}}_{i,t}$, $\mu^{\mathrm{D}}_{i,t}$) are derived from the top-ranked multi-state model, and from the best-fitting behavioral model.
* denotes significant differences between factors.

Top-ranked model:
[($p^{\mathrm{A}}_{i,t}$: sex) + $r^{\mathrm{H}}_{i,t}$ + $r^{\mathrm{M}}_{i,t}$ + ($r^{\mathrm{D}}_{i,t}$ * Time + (Time^{2}))] [($\mu^{\mathrm{H}}_{i,t}$: age: sex) + ($\mu^{\mathrm{M}}_{i,t}$: age) + ($\mu^{\mathrm{D}}_{i,t}$: age)]

Mortality cause	**Age class**	**Sex**	**Estimate**	**SE**	**95% l CI**	**95% u CI**
Harvest ($\mu^{\mathrm{H}}_{i,t}$)	Cub	Female	0.166	0.029	0.110	0.223
	Yearling	Female	0.167	0.028	0.111	0.222
	Adult	Female*	0.163	0.014	0.136	0.189
	Cub	Male	0.131	0.025	0.081	0.180
	Yearling	Male	0.233	0.031	0.172	0.293
	Adult	Male*	0.087	0.012	0.063	0.112
Management ($\mu^{\mathrm{M}}_{i,t}$)	Cub		0.052	0.006	0.040	0.064
	Yearling		0.069	0.007	0.055	0.082
	Adult		0.052	0.003	0.046	0.059
All Other ($\mu^{\mathrm{D}}_{i,t}$)	Cub		0.444	0.025	0.395	0.492
	Yearling		0.372	0.022	0.328	0.416
	Adult*		0.199	0.008	0.184	0.215

Best-fitting behavioral model:
[($p^{\mathrm{A}}_{i,t}$: sex) + $r^{\mathrm{H}}_{i,t}$ + $r^{\mathrm{M}}_{i,t}$ + ($r^{\mathrm{D}}_{i,t}$ * Time + (Time^{2}))] [($\mu^{\mathrm{H}}_{i,t}$: age: behavior) + ($\mu^{\mathrm{M}}_{i,t}$: age: behaviour) + ($\mu^{\mathrm{D}}_{i,t}$: age)]

Mortality cause	**Age class**	**Behavior**	**Estimate**	**SE**	**95% l CI**	**95% u CI**
Harvest ($\mu^{\mathrm{H}}_{i,t}$)	Cub	Normal	0.106	0.016	0.075	0.137
	Yearling*	Normal*	0.203	0.023	0.157	0.249
	Adult	Normal*	0.109	0.010	0.090	0.128
	Cub	Problem	0.146	0.040	0.068	0.223
	Yearling*	Problem*	0.033	0.013	0.008	0.058
	Adult	Problem*	0.176	0.025	0.128	0.224
Management ($\mu^{\mathrm{M}}_{i,t}$)	Cub	Normal*	0.038	0.005	0.029	0.048
	Yearling*	Normal*	0.082	0.008	0.065	0.098
	Adult	Normal*	0.040	0.003	0.034	0.046

Table 2-2 cont.

	Cub	Problem*	0.302	0.046	0.212	0.392
	Yearling*	Problem*	0.172	0.028	0.118	0.227
	Adult	Problem*	0.312	0.023	0.266	0.357
All Other ($\mu_{i,t}^{\mathrm{D}}$)	Cub		0.472	0.024	0.425	0.520
	Yearling		0.435	0.022	0.392	0.479
	Adult*		0.223	0.009	0.207	0.241

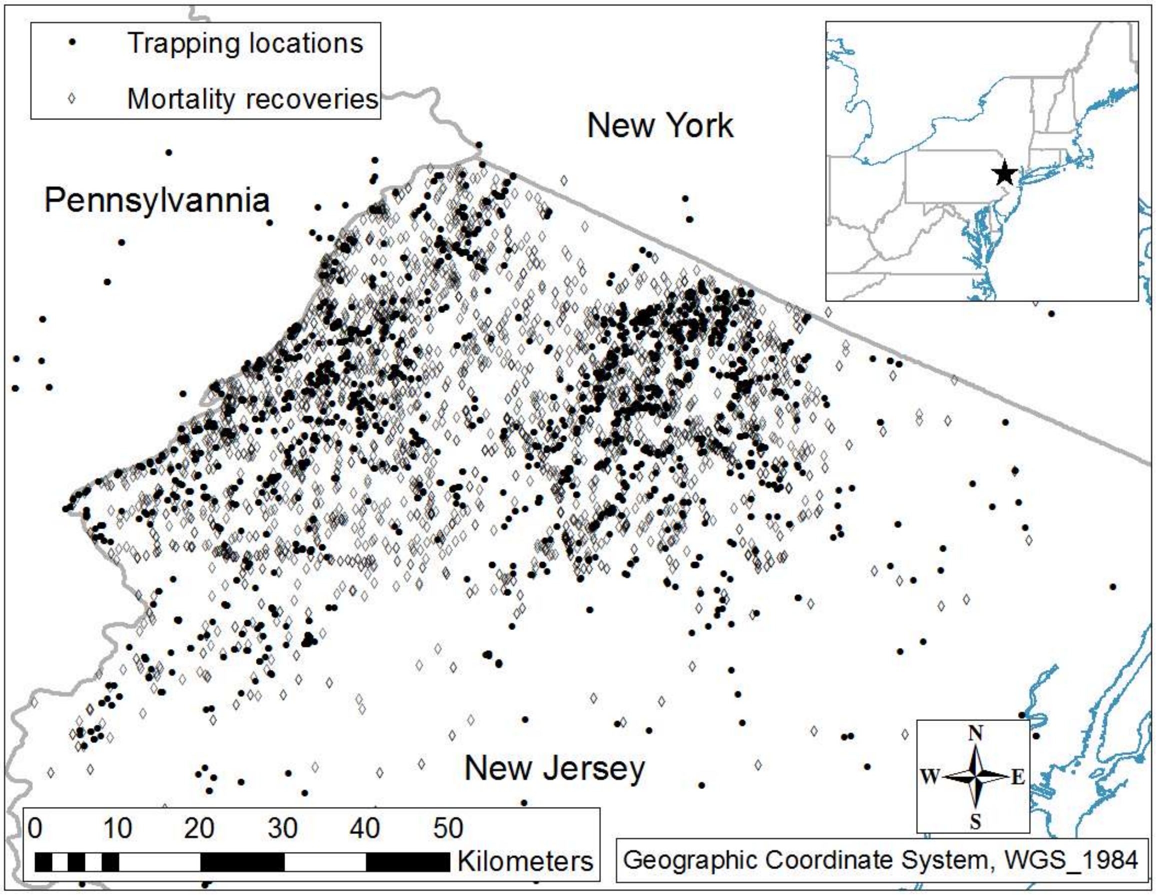

Fig. 2-1. Spatial distribution of the majority of New Jersey Division of Fish and Wildlife black bear encounter data from 1981-2014. Encounter data are comprised of 5,185 captures and 1,338 mortality recoveries.

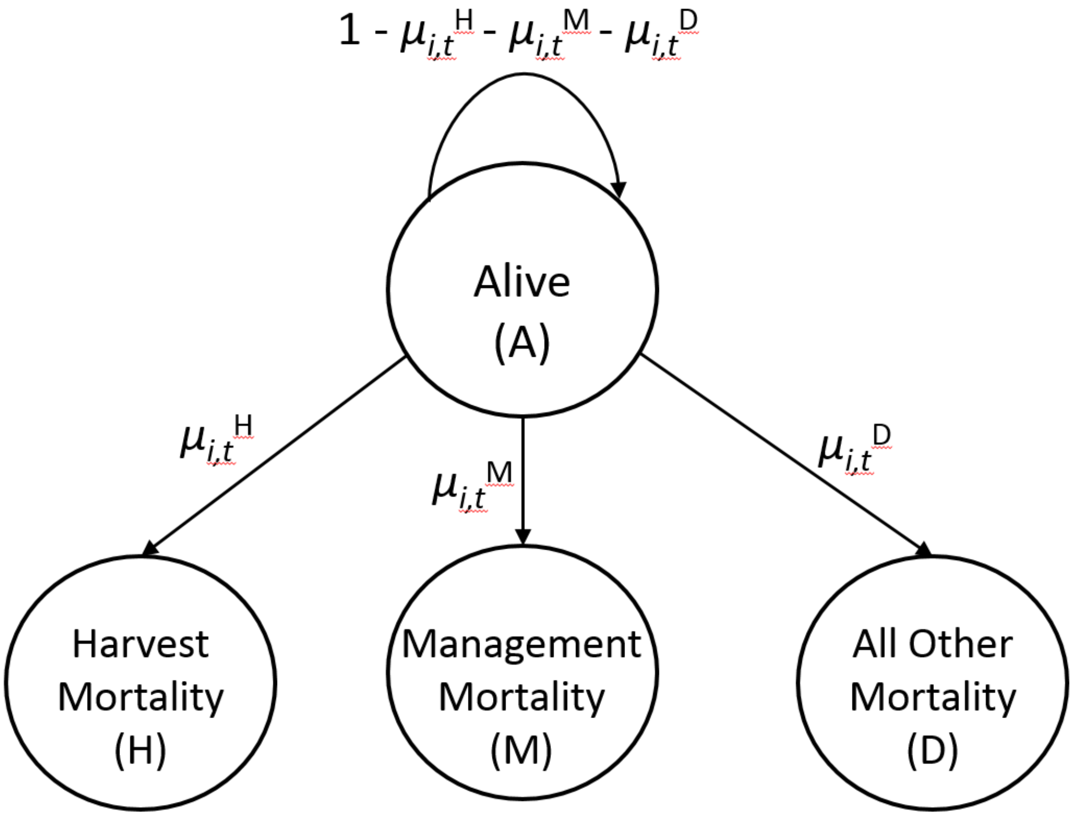

Fig. 2-2. Demographic transitions of remaining alive (*A*), dying from harvest (*H*), lethal management (*M*), or all other sources of mortality combined (*D*). Here μ represents cause-specific mortality probabilities and *H*, *M*, and *D* are absorbing states.

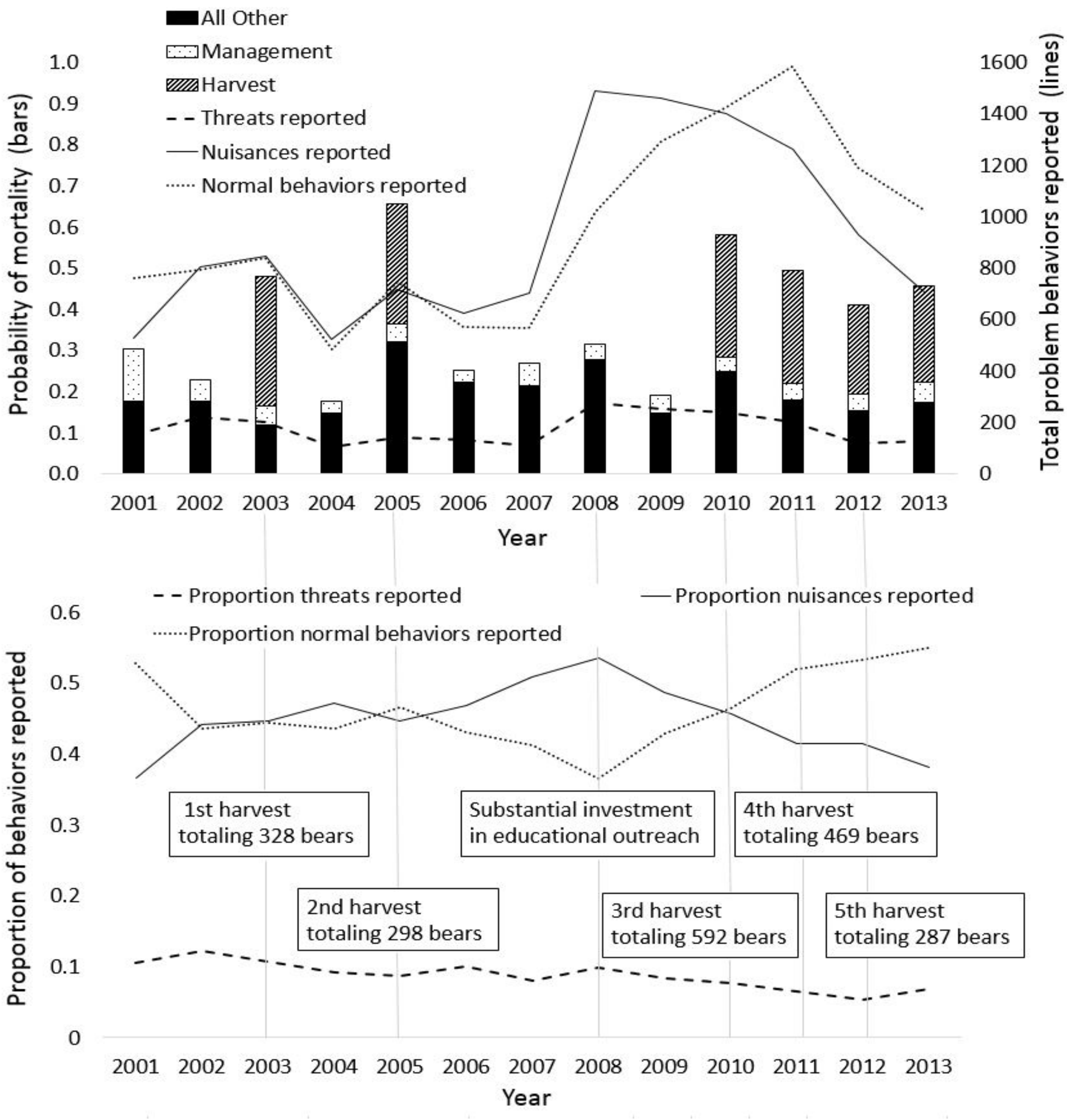

Fig. 2-3. Black bear cause-specific mortality estimates lumping sex and age-classes, and Black Bear Rating and Response Criteria incidents reported to the New Jersey Division of Fish and Wildlife between 2001-2013 (top panel), and relative proportion of normal, nuisance, and threat incidents reported relative to harvests which occurred in 2003, 2005, 2010-2012, and investment in educational campaign in 2008. (bottom panel).

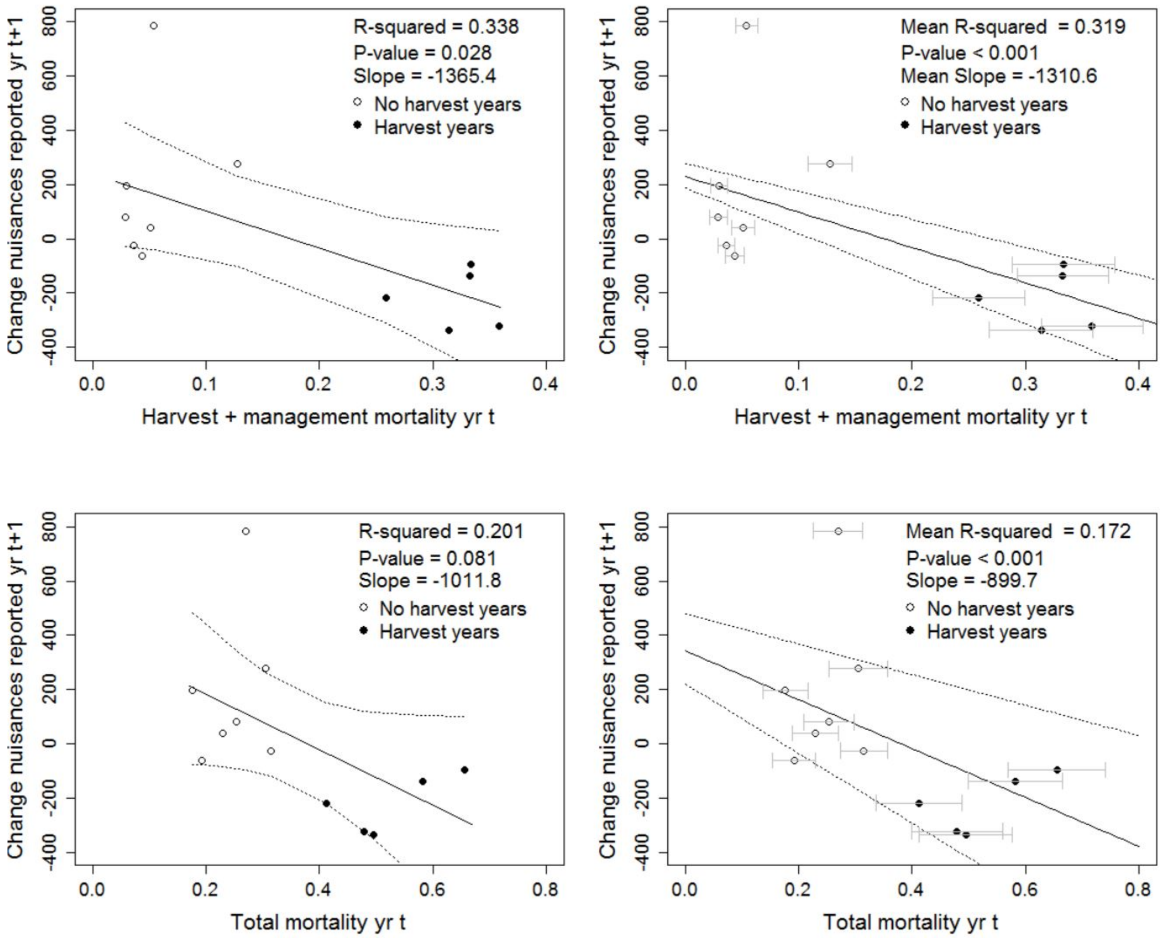

Fig. 2-4. Change in nuisance black bears reported regressed against estimates of harvest plus management mortality, and total mortality, bound by 95% confidence intervals (left plots), and including uncertainty in mortality estimates using Monte Carlo simulations (right plots).

CHAPTER 3

BLACK BEAR STRATEGIES FOR EXPLOITING ANTHROPOGENIC HABITATS DIFFER BASED ON THEIR PAST CONFLICT HISTORY WITH HUMANS

Abstract

Conserving large carnivores in the transformed landscapes of the Anthropocene hinges on increasing cultural tolerance for these occasionally dangerous predators. This need is especially evident in Mid-Atlantic North America, where the robust recovery of black bears, coupled with accelerating suburban sprawl, has resulted in unprecedented levels of human-bear conflict. We used multistate capture-mark-recapture models for a large sample of spatially-explicit bear captures (3,712) over a 14-year period in northwestern New Jersey, USA, to estimate how conflict behaviors (individuals previously designated a nuisance and/or threat), age-class, and sex influenced the probability bears would transition between wildland, agricultural, wildland-urban interface, and/or urban habitats. Across all age-classes, problem bears were significantly more likely to transition to urban and interface habitats, and they died at significantly lower rates than conspecifics displaying ‘normal’ behaviors (i.e., no history of conflict with humans) in wildland habitats. Legal kills (531) and vehicle strikes (118) combined, the two greatest mortality causes for marked bears, occurred significantly less than expected per unit area in urban and agricultural areas, and more than expected in the interface zone and wildlands. Cubs and yearlings died at significantly higher rates than adults in the risky interface habitat, corroborating independent estimates of their

increased susceptibility to harvest and vehicle strikes. These behavioral differences highlight the importance of averting initial food-conditioning, as it induces long-lasting changes in how bears utilize human-dominated landscapes.

Highlights

- Problem bears more frequently transitioned to urban and interface habitats.
- Urban and agricultural areas were safer than interface and wildland habitats.
- Problem bears had higher survival than bears with no history of conflict in wildlands.
- Young bears had lower survival than adult bears in the risky interface.
- Preventing initial food-conditioning key to reducing anthropogenic transitions.

1. Introduction

One of the immense challenges of the Anthropocene is conserving large, potentially dangerous, carnivores at densities that increase the resilience of novel social-ecological systems (Corlett, 2015; Kuijper et al., 2016). Expansive, contiguous networks of protected areas that sustain habitat integrity and provide security from human persecution continue to be unequivocally important in conserving large mammals (Craigie et al., 2010; Geldmann et al., 2013). Yet, forecasts of burgeoning human population growth and landscape transformation (Seto et al., 2012) indicate the acquisition of large terrestrial refugia sufficient for large predators will become increasingly difficult in future decades (Shafer, 2008). Further, reliance on protected areas alone to ensure viable carnivore populations is complicated by their life histories: large predators range widely given their high energetic demands (Carbone et al., 2007), often encounter high-density human developments (Cardillo et al., 2004), and have inherently low population densities that can result in inbreeding depression and unacceptable extinction risks should refugia become isolated (Benson et al., 2016). Ultimately, ecological dynamics across the biosphere are now principally driven by anthropogenic landscape transformation (Ellis et al., 2010). Sustaining the structure, function, identity, and feedbacks that emerge in these novel social-ecological systems requires recognizing the magnitude of human influence on the system, as well as the stakeholders' capacity to affect system resilience (Walker et al., 2004). As such, the conservation of large, ecologically influential carnivores now necessitates prioritizing innovative interventions outside of protected areas (Di Minin et al., 2016).

In contrast to the wide-spread and accelerating decline of other large mammals (Di Marco et al., 2014), the populations of several carnivore species have stabilized or are increasing within human-dominated landscapes in Europe (Chapron et al., 2014) and North America (Gompper et al., 2015). While the 'separation model' may produce benefits for both people and predatory wildlife in Africa (Packer et al., 2013), facilitating coexistence between humans and carnivores likely represents the only realistic way forward in ensuring carnivore persistence in heavily transformed, novel social-ecological systems (López-Bao et al., 2015). Optimizing human-carnivore coexistence in these shared landscapes demands mutualistic co-adaptation, i.e., both humans and carnivores learning from experience and altering their behaviors to minimize negative impacts on each other (Carter and Linnell, 2016). Large carnivores have interacted with humans for millennia, and some of their adaptations in response to human activities such as spatial avoidance (Gese et al., 2012; Wilmers et al., 2013) and altered activity schedules (Carter et al., 2012; Odden et al., 2014a) offer promise. However, the widespread exploitation of anthropogenic foods documented for 36 terrestrial predators in 34 countries (Newsome et al., 2015) suggests that deterring food-conditioning may be paramount in facilitating coexistence. Notably, the reliable availability of easily-accessible, high-caloric anthropogenic foods may be inducing rapid eco-evolutionary changes in carnivores, shifting phenotypic distributions such as body size (Yom-Tov, 2003).

Conserving the American black bear (*Ursus americanus*; hereafter black bears) in the human-dominated, social-ecological systems emerging in mid-Atlantic North America may require implementing management approaches that shape bear behavior to facilitate coexistence, thereby balancing the conservation of viable black bear populations with human welfare and property (Hristienko and McDonald, 2007). Black bears exhibit high behavioral plasticity (McCullough, 1982), and their conspicuous capacity to exploit anthropogenic resources has long been documented (Davenport, 1953; Horstman and Gunson, 1982). The remarkably rapid behavioral modifications of bears in response to landscape transformation (Beckmann and Berger, 2003a), as well as the inter-seasonal and -annual variation in their resource selection for human developments (Johnson et al., 2015) have increasingly been documented. In the Western USA, black bear age and sex interacted with habitat quality, natural food production, and the energetic demands of hyperphagia in driving dynamic selection for anthropogenic habitats (Baruch-Mordo et al., 2014; Johnson et al., 2015). Yet, strong anthropogenic attractants, such as fruiting ornamental trees, were invariably exploited during years with readily abundant natural foods in Montana (Merkle et al., 2013).

Bear habituation to human activities (Herrero et al., 2005) and learned food-conditioning (Mazur and Seher, 2008) may be proximate mechanisms in explaining their occurrence near human settlements (Elfström et al., 2014). The 'despotic distribution hypothesis' (Fretwell and Lucas, 1969) has been posited as the underlying driver of bear occupancy patterns in and around developed areas, given bear distributions are ultimately governed by intraspecific predation avoidance and/or interference competition (Beckmann and Berger, 2003b; Elfström et al., 2014). Mechanistically, this may occur as adult males transmit information about the nutritional landscape via chemical communication along a network of travel routes (Noyce and Garshelis, 2014). Regardless, anthropogenic landscape transformation profoundly complicates black bear conservation in a myriad of ways by: reshaping bear activity schedules, altering denning chronology, reducing home range size, increasing localized densities, promoting highly male-skewed sex ratios, increasing fertility (via earlier primiparity and greater fecundity), suppressing realized recruitment, and increasing mean body mass (Beckmann and Berger, 2003a, b; Beckmann and Lackey, 2008).

Utilizing a long-term, spatially-explicit, black bear capture-mark-recapture (CMR) dataset, we examine the extent in which intrinsic factors (sex, age-class, and behavior) influence black bear spatial transitions between four habitat states (Urban, Interface, Agricultural, and Wildland) in northwestern New Jersey (NJ), USA. Additionally, we quantify the mortality costs associated with utilizing these different habitats along a wildland-urban landscape gradient. NJ is an archetypal human-dominated landscape, characterized by high black bear densities, the greatest human densities in the USA, and unprecedented levels of human-bear conflict. This social-ecological system provides a model to test how ‘problem’ versus ‘normal’ bears may adopt differing strategies for exploiting anthropogenic habitats in a heavily human-dominated landscape.

2. Materials and methods

2.1 Study area

The New Jersey Division of Fish and Wildlife (NJDFW) collected data as part of the long-term research and management of black bears, primarily in northwestern NJ, USA (41˚04’ N, 74˚40’ W; Fig. 3-1). Detailed boundaries and bear habitat selection are described in Tri et al. (2016), and bear food habits are quantified in Makkay (2010) . Black bear abundance increased from 450-500 in 1996 (McConnell et al., 1997) to 3200-3400 in 2010 (Carr and Burguess, 2011).

2.2 Black bear capture and recovery data

From 2000 to 2014, NJDFW recorded GPS locations for 3,712 black bear captures, marking 2,718 unique individuals including 1,323 females, 1,395 males; 1,035 adults, 708 yearlings, and 975 cubs. Locations were obtained for 667 mortality recoveries predominately composed of 531 legal kills (harvest and depredation permits) and 118 vehicle strikes. Live captures were comprised of annual research trapping, incident response tapping, and den surveys. Current NJDFW capture and handling protocols are detailed in Appendix 1. All bears captured were assigned a behavioral category (Threat, Nuisance, or Normal) according to standardized criteria described in Raithel et al. (2016). As determined by the NJDFW incident response framework, if captured, threatening bears were euthanized, and nuisance bears were aversively conditioned on-site. In our analyses, we collapsed threatening and/or nuisance bears into a single category, designated as 'problem' behavior. The 2,718 unique capture histories included 505 problem bears captured in response to a reported incident, and 2,213 bears never having been identified as a threat and/or nuisance, designated herein as bears exhibiting 'normal' bear behaviors, captured as part of the NJDFW long-term monitoring program.

2.3 Assigning habitat states and estimating habitat-specific mortality risk

While urbanization during the last century occurred relatively slowly by the addition of concentric rings of development, contemporary urban areas like those in NJ during 2000-2014 are hastily expanding in spatially complex, non-linear arrangements (Ramalho and Hobbs, 2012). Further, no standardized designations exist for the variable classifications recently applied to human-altered landscapes (e.g., urban core, exurban, wildland-urban interface; McCleery et al., 2014). Thus, we explicitly derived habitat states (Urban, Interface, Agricultural, and Wildland) from 2001, 2006, and 2011 National Land Cover Databases (NLCD; Homer et al., 2015) at a spatial resolution of 30 m. We applied: NLCD 2001 to encounters between 2000-2003; NLCD 2006 to encounters between 2004-2008; and NLCD 2011 to encounters between 2009-2014. Using ArcGIS 10.2.2 (ESRI, 2011), we categorized each encounter (captures and recoveries) as follows:

encounters located within Developed (Low, Medium and High Intensity) pixels including a one pixel buffer (30 m) as Urban; encounters located within Pasture/Hay and Cultivated Crops pixels including a one pixel buffer as Agriculture; encounters located between 30-600 m from Developed (Low, Medium and High Intensity) pixels (not previously designated as Agriculture) as Interface; and all remaining encounters as Wildland (primarily composed of Deciduous Forest, Mixed Forest, Woody Wetlands, and Emergent Herbaceous Wetlands).

We expanded on the existing approach of categorizing black bear habitat as wildland or urban (Beckmann and Lackey, 2008; Merkle et al., 2013) by also explicitly evaluating bear space use in an interface zone consisting of natural land cover types immediately adjacent to developed areas. Herein, the ‘Interface’ represents the ‘green space’ within which bears encountered readily access human foods in adjoining development. We conservatively delineated the Interface from the Wildland state with a 600 m buffer around developed areas based on the movement behavior of 35 GPS-instrumented bears known to use developed areas. Previous efforts demonstrated that bears trapped in NJ urban areas were resident in close proximity to development; the median distance from the center of 54 seasonal home ranges to the nearest urban areas was < 1 km for both sexes (Tri, 2013). Further analyses suggested that this 600 m interface zone would identify bears captured in natural land cover types with a high probability of recently exploiting anthropogenic resources, as this buffer reflects the mean daily linear distance traveled (584 ± 246 m) by GPS-instrumented bears (6,857 bear days and 107,344 locations averaged across 4 seasons; J.D. Raithel, *Unpublished results*).

Thus, the 4,379 spatially-explicit black bear encounters (3,712 captures, 667 mortality recoveries) were categorized as: 1,521 Wildland (40.9%), 1,502 Interface (40.5%), 425 Agriculture (11.5%) and 263 Urban (7.1%; Fig. 3-1).

To estimate the functional boundary of the landscape, we needed to account for the 72 bears documented both within and outside of NJ (as far away as 135 km from the state boundary). Thus, we circumscribed all encounters with a minimum convex polygon to define the functional boundaries of the study area. The resultant landscape was comprised of 36.2% Interface, 27.0% Wildland, 18.6% Urban, and 18.2% Agriculture habitats. In addition to the survival rates estimated from CMR models, we evaluated habitat-specific mortality risk from legal kills and vehicle strikes, the two greatest documented sources of mortality, as the total proportional observed mortality for these sources, relative to the total proportional landscape coverage.

2.4 Multistate capture-reencounter model

To evaluate black bear transition probabilities between habitats, we analyzed the CMR data using a multistate framework with Urban (U), Interface (I), Agriculture (A), and Wildland (W) states. Each ‘year’ in our analyses consisted of captures occurring between February 15 – December 31, as bears give birth during January and February in winter dens. We estimated the probability of individual *i* transitioning from one state to another (e.g., probability of transitioning from Wildland to Interface states = $\psi_{i,t}^{\mathrm{WI}}$), or remaining in the same state (e.g., probability of remaining in the Wildland state = $\psi_{i,t}^{\mathrm{WW}}$) between occasion t and $t + 1$. We estimated all habitat transition and stasis probabilities conditional on state-specific survival and live recapture probabilities (probability of recapturing live individual *i* in habitat state U, I, A, or W during occasion *t* (e.g., probability of recapture in Wildland state = $p_{i,t}^{\mathrm{W}}$).

2.5 Data analyses

We used package RMark (Laake, 2013) in Program R version 3.3.1 (R Core Team, 2016) to estimate multistate model parameters, and calculate Akaike's information criterion adjusted for sample size (AIC$_c$; Akaike, 1973) used to rank models in our model set. Previous analyses had identified sex as the most important covariate explaining variability in live recapture probabilities ($p_{i,t}^{a}$; Raithel et al., 2016). Initially we modeled recapture probabilities, and subsequently survival probabilities, in each habitat state (U, I, A, and W), as a function of combinations of demographic covariates of interest (sex, age-class, and behavior). Age was ascertained via cementum analysis. Bears entered multistate framework based on their age-class at time of initial marking, as: 0 – < 1 years as cub; 1 – < 2 years as yearling; and ≥ 2 years as adult. Bears trapped in response to an incident and identified as a problem (i.e., nuisance and/or threat) retained this designation regardless of subsequent capture habitat. After selection of the best parameterization for detection and survival probabilities, we modeled habitat transitions ($\psi_{i,t}^{ab}$), and stasis ($\psi_{i,t}^{aa}$) probabilities as a function of single covariates (i.e., sex, age-class, and behavior). We constructed the final candidate model set for all transition probabilities using the best-performing covariates and ecologically meaningful interactions between these covariates.

3. Results

3.1 Risk and intrinsic factors associated with habitat transitions probabilities

Relative to their coverage within the northwestern NJ landscape, the Interface (1.231 % risk/% coverage) and Wildland (1.314 % risk/ % coverage) states were more "dangerous" (i.e. highest risk: highest proportion of mortalities relative to the proportion of landscape coverage measured in km^2), and the Urban (0.421 % risk/% coverage) and Agriculture (0.465 % risk/% coverage) states were significantly safer, with respect to the two greatest sources of mortality, legal kills and vehicle strikes (Table 3-1).

The best-performing model for recapture probabilities (w_i = 1.000) identified sex as the most important covariate within all habitat states (Table 3-2). Mean female recapture probability exceeded that of males in both the Wildland (0.453 ± 0.063, 0.152 ± 0.024, respectively) and Interface states (0.366 ± 0.047, 0.153 ± 0.023; Table 3-3).

The best-performing model for survival probabilities (w_i = 1.000) identified behavior as the most important covariate within Wildland and Agriculture states and age within Interface and Urban states (Table 3-2). Mean survival probability of problem bears greatly exceeded that of normal bears in Wildlands (0.909 ± 0.103, 0.495 ± 0.032, respectively), and mean adult bear survival exceeded that of both cubs and yearlings in the Interface (0.692 ± 0.034, 0.374 ± 0.048, 0.294 ± 0.047 respectively; Table 3-3). Although behavior was not the best predictor for survival in the Interface and Urban states, in the best-performing model that included behavior for these states, the pattern was analogous. Problem bear survival exceeded that of normal bears in the Interface (0.668 ± 0.048, 0.445 ± 0.036, respectively). In the best-performing model that included age as a predictor of survival in all habitat states, adults consistently survive at higher rates than young bears.

After incorporating the best-performing model structure for detection and survival probabilities described above, the top-ranked model for habitat transitions included an age effect for the probability of transitioning to Wildland and Agriculture states, and a behavior effect for the probability of transitioning to Interface and Urban states (w_i = 0.890; Table 3-2). Problem bears, relative to normal bears, were more likely to transition to Urban (0.137 ± 0.017, 0.040 ± 0.010, respectively) and Interface (0.489 ± 0.037, 0.237 ± 0.032, respectively; Table 3-3) states. Specifically, the mean probabilities of problem bears transitioning to the Interface from Wildland (0.370 ± 0.059), Agriculture (0.384 ± 0.057), and Urban (0.384 ± 0.056) states were all significantly greater than those of bears with no past history of conflict, respectively (0.129 ± 0.027, 0.183 ± 0.028, 0.117 ± 0.023; Fig. 3-2). Further, the mean probabilities of problem bears transitioning to the Urban state from Wildland (0.141 ± 0.039), Agriculture (0.146 ± 0.040), and Interface (0.192 ± 0.046) states were all significantly greater than those of normal bears, respectively (0.029 ± 0.011, 0.041 ± 0.015, 0.029 ± 0.011; Fig. 3-2).

4. Discussion

This study provides evidence that bears with a history of conflicts with humans (i.e., bears that are likely food-conditioned; *n* = 505 ‘problem’ bears) and bears with no history of conflicts (*n* = 2,213 ‘normal’ bears) exhibit differing strategies in exploiting anthropogenic habitats in human-dominated landscapes. Relative to normal conspecifics, black bears previously trapped in response to nuisance complaints were more likely to be subsequently re-encountered in Urban and Interface habitats (Table 3-3; Fig. 3-2). These results imply that preventing *initial* food-conditioning in bears is critical, as past nuisance behavior(s) was a strong predictor (w_i = 1.00; Table 3-2) of the probability that bears would transition across, and disproportionally utilize, anthropogenic habitats.

We demonstrate that distinct anthropogenic habitats, i.e., developed areas compared with the 'doughnut' of immediate green space ringing development (i.e., the Interface), may pose appreciably different levels of risk. Significantly fewer than expected mortality recoveries, per unit area, occurred in the Urban state, with respect to the two greatest documented causes of bear mortality, i.e., legal kills and vehicle strikes. However, the juxtaposed Interface state, where vehicular speed limits and harvest vulnerability both increase, produced greater than expected mortality recoveries, per unit area (Table 3-1). Survival estimates derived from live CMR models (an independent dataset) support these mortality recovery data; across all age classes (cubs, yearlings, and adults), bears survived at higher rates in the Urban (0.551 ± 0.156, 0.632 ± 0.166, 0.738 ± 0.102, respectively) relative to Interface state (0.374 ± 0.048, 0.294 ± 0.047, 0.692 ± 0.034, respectively; Table 3-3). This difference in survival between Urban and Interface habitats was especially pronounced for yearlings (0.294 vs. 0.632), corroborating independent estimates of their significantly greater susceptibility to vehicle strikes and harvest (Raithel et al., 2016). We documented a similar mortality risk pattern between the Wildland and Agriculture states; bears were vulnerable to harvest in wildlands, while agricultural areas, like urban developments, may serve as refugia. Survival estimates from the independent CMR dataset supported this observation as well, as both problem and normal bears exhibited increased survival in the Agriculture relative to the Wildland state (Table 3-3).

Interestingly, problem bears who transition more frequently to Urban and Interface habitats in NJ, were also significantly more likely to survive than normal bears in the Wildland state (0.909 ± 0.103 vs. 0.495 ± 0.032, respectively). This, coupled with the importance of age-class in predicting mortality risk in Urban and Interface habitats (Table 3-2), highlight the importance of experiential learning for ursids. The ‘cognitive map’ bears create (Gilbert, 1999) to access spatially and temporally variable resources (McCall et al., 2013) may also apply to assessing risk while navigating the complex spatial heterogeneity of this anthrome.

As adult males were significantly less likely to be harvested than adult females and yearling males during the NJ black bear harvests implemented between 2003-2014 (Raithel et al., 2016), we anticipated adult males would be frequently re-encountered in habitats with low human-caused mortality risk, potentially forcing females and young bears into riskier habitats. Although imprecise state-specific transition probabilities estimated by sex precluded any significant differences, this pattern of male use of refugia was evident in their increased use of the Agriculture state relative to females (Fig. 3-3). Further, males consistently transitioned less frequently to the riskier Interface and Wildland states (Fig. 3-3).

The accessibility of human foods continues to be a pernicious conservation concern ubiquitous within the novel social-ecological systems characterizing the Anthropocene (Oro et al., 2013), inducing rapid and profound behavioral changes in black bears. When compared to wild conspecifics, black bears that chronically forage on garbage are active for significantly fewer hours per day (8.5 vs. 13.3), shift their activities from crepuscular to nocturnal periods, enter dens later, and remain denned for significantly fewer days (Beckmann and Berger, 2003a). Further, the prevalence of individuals utilizing anthropogenic foods can quickly escalate within bear populations, as foraging behaviors are transmitted vertically from maternal sows to cubs, and juveniles reared in urban habitats have a high probability of continuing to forage in developed areas when they become independent (Mazur and Seher, 2008). Reducing available anthropogenic attractants by 55-70% is unequivocally the most cost-effective management action in deterring most bears from using developed areas (Baruch-Mordo et al., 2013). However, those state agencies that lack the legislative authority to mandate bear-resistant garbage receptacles in residential communities, such as NJDFW, must rely solely on educational outreach to encourage their voluntary adoption.

Results from this study have important management implications. Within this social-ecological system, characterized by high densities of black bears and high levels of human-bear conflict, bears that have been previously identified as a nuisance are likely to be reencountered in anthropogenic areas. That said, bears exhibit remarkable behavioral plasticity, and we do not contend that the undesirable behaviors of these nuisance bears are irreversible, given preceding evidence to the contrary (Cole, 1974; Hopkins et al., 2014; Johnson et al., 2015). Yet, in the absence of high rates of voluntary compliance with respect to bear-proof residential receptacles, managers must recognize that problem bears are more likely to frequent anthropogenic habitats than bears without a history of conflict (i.e., less likely to be food-conditioned). Thus, to achieve a balance between bear conservation and human safety and property protection (Hristienko and McDonald, 2007), integrated management in this system should continue to invest heavily in educational outreach to prevent food-conditioning, but should also include the lethal control of threatening individuals, as well as recreational harvest applied to maximize bear fear of human-caused mortality, to “keep wild bears wild.” Additionally, we recommend attempting to eliminate urban and agricultural refugia by employing harassment techniques in these areas during closely-monitored harvest periods. Although these CMR data are composed of > 3,700 black bear captures over 14 years, they are limited in providing inference with respect to continuous movement patterns. Additional work remains examining how behavior may, or may not, influence fine-scale habitat use in this human-dominated landscape.

Lastly, we must recognize that we have transformed the Mid-Atlantic deciduous forest into a novel human-dominated landscape, and black bears are responding. The provisioning of food subsidies over the past four decades coupled with exponential population growth (Carr and Burguess, 2011) is likely inducing eco-evolutionary changes in the behavioral and phenotypic distributions of this species. Thus, we agree that novel management techniques, such as altering the timing of harvest periods to mitigate the seasonal use of refugia by problem species (Cromsigt et al., 2013), should be carefully explored to promote a landscape of coexistence (Oriol-Cotterill et al., 2015), and that conservation efforts on behalf of Mid-Atlantic black bears must now pivot toward our ability to shape bear behavior, and increase the cultural carrying capacities of local stakeholders (Young et al., 2015).

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Tables and Figures

Table 3-1

Habitat state classification for locations of legal kills and vehicle strikes for marked black bears in northwestern New Jersey, USA from 2000-2014, relative to coverage of each habitat state across the landscape. Legal kills and vehicle strikes represent the two greatest sources of documented mortality. Legal kills are comprised of harvest and depredation permits. Habitat states include: Urban, Interface, Agriculture, and Wildland.

Habitat State	Landscape Coverage	Mortality Cause	Total Mortalities	% Mortality	% Risk / % Coverage
Urban	18.6%	Legal Kill	10	1.5%	0.083
		Vehicle Strike	41	6.3%	0.339
		Legal Kill + Vehicle Strike	**51**	**7.9%**	**0.421**
Interface	36.2%	Legal Kill	230	35.4%	0.980
		Vehicle Strike	59	9.1%	0.251
		Legal Kill + Vehicle Strike	**289**	**44.5%**	**1.231**
Agriculture	18.2%	Legal Kill	69	10.6%	0.584
		Vehicle Strike	10	1.5%	0.085
		Legal Kill + Vehicle Strike	**79**	**12.2%**	**0.465**
Wildland	27.0%	Legal Kill	222	34.2%	1.268
		Vehicle Strike	8	1.2%	0.046
		Legal Kill + Vehicle Strike	**230**	**35.4%**	**1.314**
Combined	100%	Legal Kill	531	81.8%	
		Vehicle Strike	118	18.2%	
		Legal Kill + Vehicle Strike	649	100%	

Table 3-2

Candidate model ranking for recapture (*p*), survival (*S*), and transition probabilities (*Ψ*) of black bears within and to four habitat states: Wildland (*W*), Agriculture (*A*), Interface (*I*), and Urban (*U*) between 2000-2014 in New Jersey, USA. Explanatory variables include age class (cub, yearling, adult), sex, and behavior (problem individuals, and bears never having been classified as a nuisance and/or threat). Interactions are represented by a colon, and the top model is denoted in bold.

Parameter	**Habitat State**				**Model Performance**		
	Wildland (*W*)	Agriculture (*A*)	Interface (*I*)	Urban (*U*)	Δ AIC_c	Weight (w_i)	Number of Parameters
Recapture (*p*)	**sex**	**sex**	**sex**	**sex**	**0.000**	**1.000**	**16**
	sex	sex	stratum	stratum	56.631	0.000	14
	sex	sex	age	age	57.624	0.000	16
	sex	stratum	stratum	stratum	67.101	0.000	13
	age	age	sex	sex	72.945	0.000	16
	stratum	sex	sex	sex	74.742	0.000	15
	stratum	stratum	sex	sex	86.034	0.000	14
	age	age	age	age	126.754	0.000	16
	stratum	stratum	stratum	stratum	138.878	0.000	12
Survival (*S*)	**behavior**	**behavior**	**age**	**age**	**0.000**	**1.000**	**22**
	age	age	age	age	33.49	0.000	24
	age	age	behavior	behavior	35.60	0.000	22
	behavior	behavior	behavior	behavior	39.52	0.000	20
	age	behavior	behavior	behavior	44.07	0.000	21
	stratum	age	age	age	54.37	0.000	22
	age	stratum	stratum	stratum	106.68	0.000	18
	sex	sex	sex	sex	131.79	0.000	20

Table 3-2 cont.

	stratum	stratum	stratum	stratum	135.79	0.000	16

Parameter	**Habitat Transitions**				**Model Performance**		
	To Wild ($\Psi^{\mathrm{W}}_{i,t}$)	To Agriculture ($\Psi^{\mathrm{A}}_{i,t}$)	To Interface ($\Psi^{\mathrm{I}}_{i,t}$)	To Urban ($\Psi^{\mathrm{U}}_{i,t}$)	Δ AIC$_c$	Weight (w_i)	Number of Parameters
Transition (Ψ)	**age**	**age**	**behavior**	**behavior**	**0.000**	**0.890**	**28**
	behavior	behavior	behavior	behavior	5.204	0.066	26
	age	behavior	behavior	behavior	6.689	0.031	27
	age:behavior	age:behavior	age:behavior	age:behavior	8.626	0.012	42
	behavior	age	age	age	16.369	0.000	29
	behavior	behavior	age	age	20.064	0.000	28
	age	age	age	age	23.740	0.000	30
	sex	sex	sex	sex	30.746	0.000	26
	stratum	stratum	stratum	stratum	41.174	0.000	22

Table 3-3

Mean black bear state-specific recapture (*p*), survival (*S*), and transition probability (*Ψ*) estimates derived from the top-ranked multi-state model in New Jersey, USA between 2000-2014. * denotes significant differences between factors.

		Factor							
		Female				**Male**			
Parameter	Habitat State	**Mean**	SE	95% l CI	95% u CI	**Mean**	SE	95% l CI	95% u CI
Recapture (*p*)	Wildland (*W*)	**0.453***	0.063	0.329	0.577	**0.152***	0.024	0.104	0.200
	Agriculture (*A*)	**0.134**	0.030	0.075	0.192	**0.055**	0.013	0.028	0.081
	Interface (*I*)	**0.366***	0.047	0.273	0.458	**0.153***	0.023	0.108	0.198
	Urban (*U*)	**0.194**	0.045	0.105	0.282	**0.194**	0.045	0.106	0.282

		Normal				**Problem**			
		Mean	SE	95% l CI	95% u CI	**Mean**	SE	95% l CI	95% u CI
Survival (*S*)	Wildland (*W*)	**0.495***	0.032	0.431	0.559	**0.909***	0.103	0.706	0.999
	Agriculture (*A*)	**0.622**	0.050	0.524	0.721	**0.773**	0.116	0.545	0.999

		Cub				**Yearling**				**Adult**			
		Mean	SE	95% l CI	95% u CI	**Mean**	SE	95% l CI	95% u CI	**Mean**	SE	95% l CI	95% u CI
	Interface (*I*)	**0.374**	0.048	0.279	0.468	**0.294**	0.047	0.201	0.386	**0.692***	0.034	0.626	0.758
	Urban (*U*)	**0.551**	0.156	0.246	0.856	**0.632**	0.166	0.306	0.958	**0.738**	0.102	0.538	0.939

Table 3-3 cont.

		Cub				Yearling				Adult			
		Mean	SE	95% l CI	95% u CI	**Mean**	SE	95% l CI	95% u CI	**Mean**	SE	95% l CI	95% u CI
Transition (Ψ)	To Wildland	**0.229**	0.031	0.169	0.289	**0.179**	0.021	0.138	0.221	**0.225**	0.020	0.185	0.266
	To Agriculture	**0.246**	0.043	0.162	0.331	**0.472***	0.089	0.297	0.646	**0.149***	0.070	0.070	0.227

		Normal				Problem			
		Mean	SE	95% l CI	95% u CI	**Mean**	SE	95% l CI	95% u CI
	To Interface	**0.237***	0.032	0.174	0.300	**0.489***	0.037	0.417	0.562
	To Urban	**0.040***	0.010	0.020	0.060	**0.137***	0.017	0.103	0.171

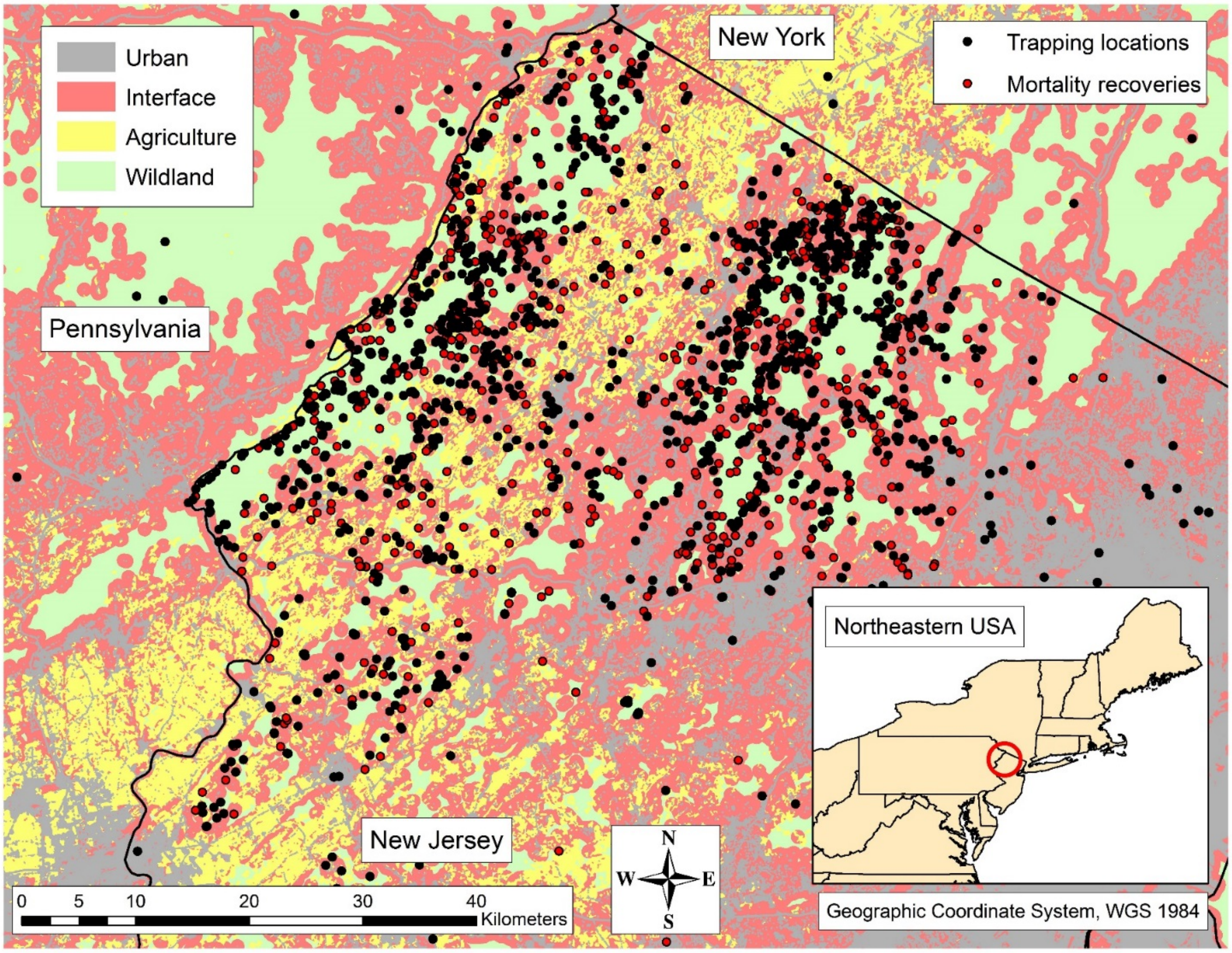

Fig. 3-1. Spatial distribution of the majority of New Jersey Division of Fish and Wildlife black bear encounter data from 2000-2014 overlaid on habitat states from National Land Cover Data 2011.

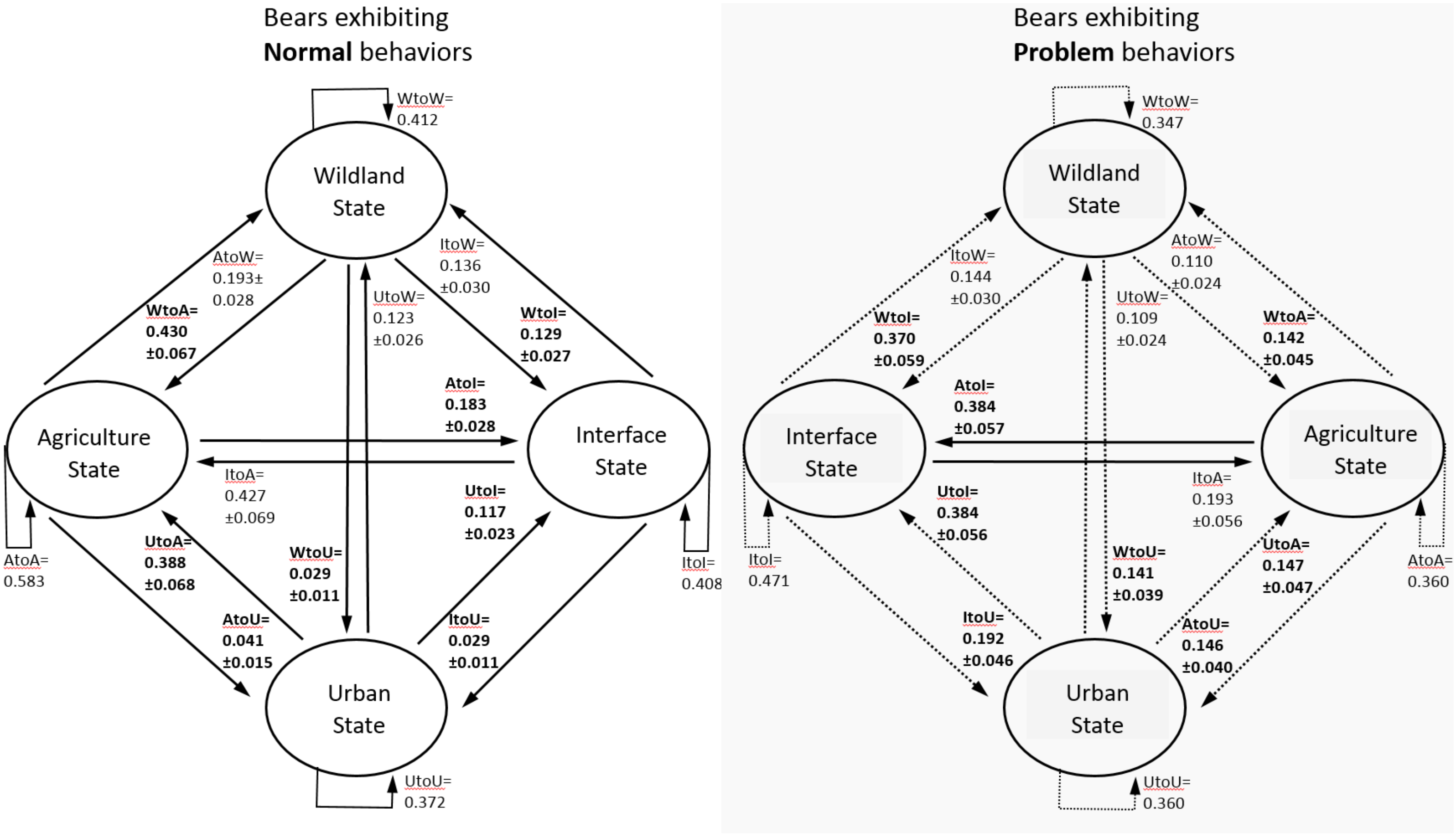

Fig. 3-2. Black bear state-specific transition estimates by behavioral group in New Jersey, USA between 2000-2014. Bears with no conflict history (i.e., 'Normal behaviors') on left and bears with documented conflict history (i.e., 'Problem behaviors') on right. Bold denotes significant differences between groups.

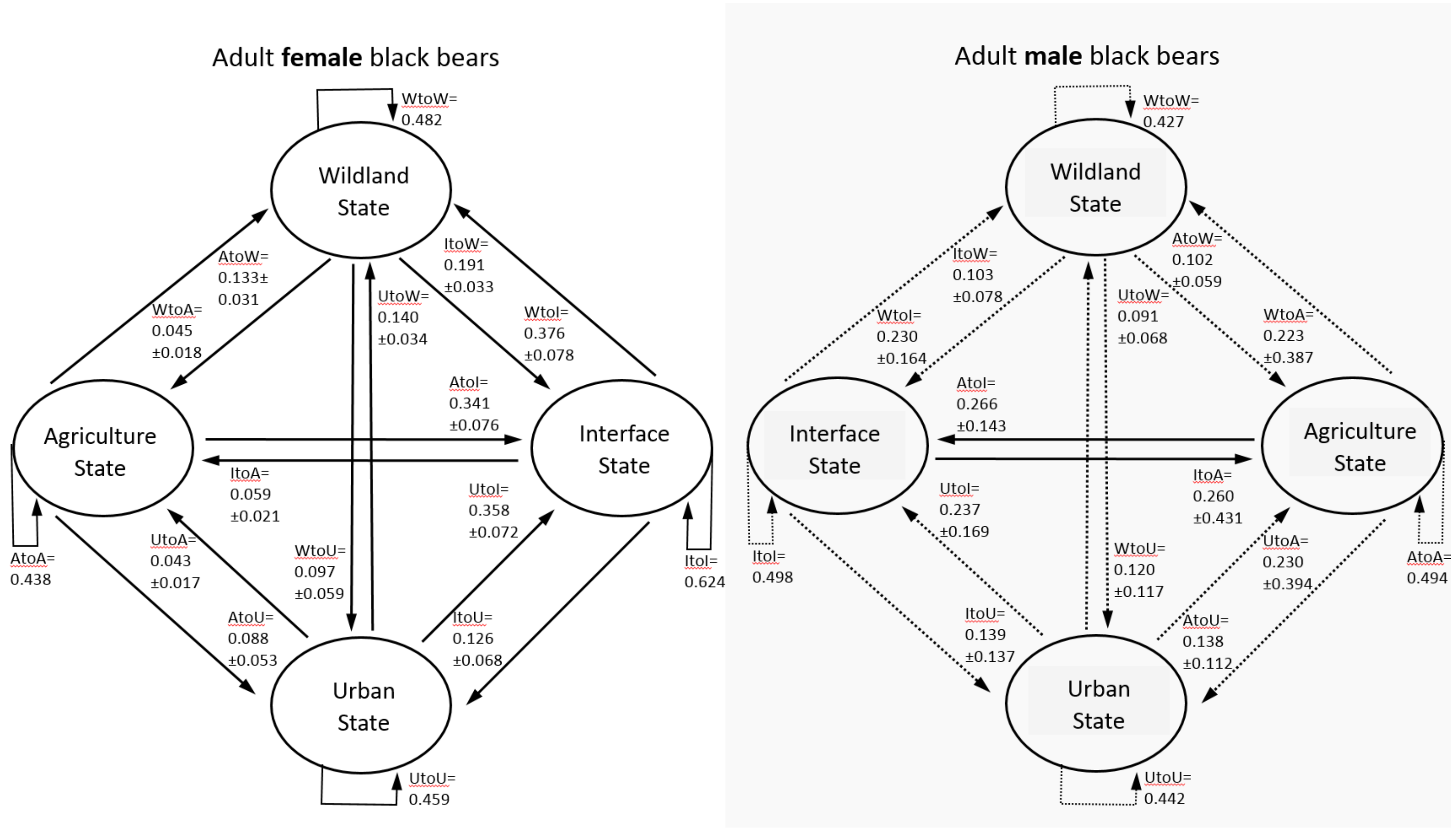

Fig. 3-3. Adult black bear state-specific transition estimates by sex in New Jersey, USA between 2000-2014. Adult female bears on left and adult male bears on right.

CHAPTER 4

BLACK BEAR POPULATION DYNAMICS IN ANTHROPOGENIC AND WILDLAND HABITATS IN THE MID-ATLANTIC UNITED STATES

Abstract

The robust recovery of the American black bear (*Ursus americanus*) in the human-dominated landscapes typifying the Mid-Atlantic United States has been accompanied by unsustainable levels of human-black bear conflict. Conservation efforts must now pivot toward identifying management practices that promote coexistence between humans and this ecologically-important species. We assessed bear population dynamics within wildland and anthropogenic habitats in northwestern New Jersey, USA, in response to a new bear harvest implemented to stabilize population growth and mitigate conflicts. We parametrized wildland and anthropogenic matrix population models and habitat transition rates with 1,312 female encounter histories and 259 adult ($\geq$ 3 years-old) female den surveys. Although females that denned in anthropogenic areas exhibited greater age-specific fecundity ($P = 0.014$), breeding earlier and senescing later than wildland conspecifics, they also experienced substantially lower survival across all age-classes. Between 2000 –2013, including six harvest and seven harvest-moratoria years, wildland habitats represented a population source ($\lambda_W = 1.133$, 95% CI 1.025–1.213) and anthropogenic habitats, a sink ($\lambda_A = 0.945$, 95% CI 0.848–1.034). However, during harvest moratoria, both wildland ($\lambda_{W\,No\,H} = 1.264$) and anthropogenic ($\lambda_{A\,No\,H} =$

1.197) components of the population grew rapidly. During harvest years, the wildland ($\lambda_{W\ Harv} = 1.011$) component stabilized and the anthropogenic component decreased dramatically ($\lambda_{A\ Harv} = 0.697$). Abundance projections derived from these matrix population models and habitat transition rates closely mirrored two independent abundance estimates that occurred in 1996 (450–500 bears) and 2009–10 (3200–3400 bears). Observed human-bear conflicts were highly correlated ($P < 0.001$; $R^2 = 0.698$) with projections of anthropogenic bear abundance. We recommend a carefully regulated harvest integrated within a black bear management strategy that includes incident-response protocols, and educational programs aimed at reducing anthropogenic attractants. Innovative harvest management that further expands the take of bears in the wildland-urban interface will help reduce the use of anthropogenic habitats by black bear, and ultimately decrease human-bear conflicts.

Introduction

One of the fundamental challenges impeding human-carnivore coexistence is the propensity of predators to exploit human-derived foods, a ubiquitous pattern documented in a multitude of carnivore taxa across regions (Newsome et al. 2015). This concern is especially pronounced in the Mid-Atlantic United States, where the rapid recovery of the American black bear (*Ursus americanus*; hereafter, black bear) coincided with accelerating anthropogenic landscape transformation, and has resulted in unprecedented levels of human-bear conflict. For example, in northwestern New Jersey (NJ), the state with the highest human densities in the United States, black bear abundance exponentially increased from an estimated 450–500 in 1996 (McConnell et al. 1997) to

3200–3400 in 2009–10 (Huffman et al. 2010, Carr and Burguess 2011), with densities in some areas exceeding 1.3 bears/km^2. It is unclear whether this six-fold population increase, occurring in less than 15 years, may have been driven by the intrinsic demographic responses (i.e., increased fertility and/or survival) of bears capitalizing on anthropogenic food subsidies, or is ultimately the result of a landscape-level redistribution.

Black bears exhibit remarkable plasticity in foraging behaviors (McCullough 1982), rapidly modifying their activity budgets, diel patterns, and denning chronology in response to the availability of human-derived foods (Beckmann and Berger 2003a). Around several developed areas in the Western USA, black bear selection for anthropogenic resources was highly dynamic across seasons and years (Baruch-Mordo et al. 2014, Johnson et al. 2015), while in other developments they predictably exploited strong seasonal attractants, such as fruiting ornamental trees (Merkle et al. 2013). Generalizing bear population-level responses to anthropogenic landscape transformation is confounded by acute behavioral heterogeneity, as individual bears may exhibit highly variable levels of habituation to human activities (Herrero et al. 2005) and learned food-conditioning (Mazur and Seher 2008). In northwestern NJ, bears with a history of conflict transitioned more frequently from wildland to anthropogenic habitats relative to conspecifics never identified as a problem (see Chapter 3). Additionally, bear occupancy patterns in, and around, anthropogenic areas may also be influenced by intraspecific predation avoidance and/or interference competition (Elfström et al. 2014a, Elfström et al. 2014b).

Anthropogenic habitats provide reliable, easily-accessible, high-caloric food subsidies relative to the spatially and temporally patchy resources found in adjacent wildlands (Shochat et al. 2004), possibly depressing density-dependent reproductive limitations that manifest in wildlands (Rodewald and Gehrt 2014). Female black bears utilizing anthropogenic areas in the Lake Tahoe Basin, Nevada, exhibited earlier primiparity (Beckmann and Berger 2003b) and increased age-specific fecundity (Beckmann and Lackey 2008), ultimately resulting in reproductive output three times that of wild conspecifics. Yet, this increased fecundity did not translate into realized recruitment and population growth, given low rates of successful juvenile dispersal due to vehicle strikes (Beckmann and Berger 2003b) and high age-specific mortality (Beckmann and Lackey 2008). Similarly, simulation models parameterized with black bear data from Aspen, Colorado, demonstrated that the increased fecundity of sows exploiting anthropogenic foods during natural food failures was negated by the lethal control of adult females during these periods (Lewis et al. 2014).

Human-induced rapid environmental change can propagate mismatches between environmental conditions that may have historically bestowed wildlife with high-quality habitats, mates, and/or food items, but now decrease realized fitness in human-dominated landscapes (Sih 2013). These maladaptive behavioral scenarios can result in “ecological traps” with pernicious consequences, as anthropogenic activities act to uncouple the cues individuals use to discern high-quality habitat from the positive outcomes evolutionarily associated with given cues (Robertson and Hutto 2006). The aforementioned urbanization of the Lake Tahoe Basin has created a population sink ($\lambda = 0.749$) resulting in the spatial reallocation of black bears from wildland to anthropogenic areas. Similarly, Florida black bear (*U. americanus floridanus*) subpopulations were stable in the contiguous Ocala National Forest (λ = 1.014–1.100), but declined in a nearby residential community (ʎ = 0.917–0.969; Hostetler et al. 2009). Evidence for an ecological trap for grizzly bears (*U. arctos*) was recently demonstrated in a mountain valley in British Columbia, Canada, rich in berry resources but with high human densities and traffic volume (Lamb et al. 2017). Questions remain regarding the extent to which anthropogenic areas in the heavily human-dominated landscape of northwestern NJ may be inducing similar population dynamics.

Between 2000 and 2013, >26,000 human-black bear incidents were reported to the New Jersey Division of Fish and Wildlife (NJDFW), including >1400 incidences of verified property damage, >400 livestock kills, >250 pet attacks and/or kills, seven human attacks and one human fatality. A regulated harvest was reintroduced in northwestern NJ in 2003 as one component of an integrated management plan to stabilize bear population growth rates and mitigate conflicts, resulting in an acrimonious public debate. Utilizing a long-term, spatially-explicit, black bear capture-reencounter data set, our objectives are to: 1) quantify the relative contributions of the wildland and anthropogenic components of the population to landscape-level population growth in response to harvest; and 2) project black bear abundance within a metapopulation system connecting both wild wildland and anthropogenic habitats, and associated rates of human-bear conflict under a range of harvest scenarios. NJ represents an archetypical human-dominated, social-ecological system, and is an ideal model to test the efficacy of a regulated black bear harvest in curbing an increasing anthropogenic bear subpopulation and escalating human-bear conflicts.

Methods

Study area

NJDFW collected these data as part of the long-term research and management of black bears, primarily in northwestern NJ, USA (41˚04’ N, 74˚40’ W). The study area is detailed in Tri et al. (2016) and Makkay (2010). A regulated black bear harvest was first reinstated in NJ in 2003, closed by the NJ Supreme Court in 2004, occurred again in 2005, the moratorium reinstated from 2006-2009 pending the approval of the NJ Fish and Game Council Comprehensive Black Bear Management Policy, reopened in 2010, and continues today (Wolgast et al. 2010). Harvest regulations are described in detail in Chapter 2.

Data collection

From 2000 to 2014, NJDFW recorded GPS locations for 1,984 female black bear captures, marking 1,312 unique females including 997 adults (≥ 2 years-old), 379 yearlings, and 608 cubs of the year (hereafter, cubs). Locations were obtained for 397 female mortality recoveries, predominately composed of 285 legal kills (harvest and depredation permits), 47 lethal controls, and 44 vehicle strikes. Current NJDFW black bear capture and handling protocols are detailed in Appendix 1. Ages were acquired for all bears captured and recovered as determined by cementum analysis performed by the Gary Matson laboratory (Missoula, Montana). Age at first capture for female bears ranged from newborn cubs to 28 years-old. All bears were assigned a behavioral category (Threat, Nuisance, or Normal) according to standardized criteria described in Raithel et al. (2016) (Chapter 2). These analyses included 1,042 female black bears with no history of conflict and 270 females previously designated a nuisance and/or threat. Additionally, NJDFW received 26,582 incident reports from the general public between 2001 – 2013, and categorized those as 2,277 Threats, 12,013 Nuisances, and 12,292 Normal interactions (protocol detailed in Chapter 2).

From 1987 to 2014, NJDFW personnel conducted 462 female black bear den surveys between late January and mid-April to determine female and male cub production. Of these surveys, 50 sows were recorded with yearlings (individuals that just turned 1 year-old) present within the den. 378 of the denned females without yearlings present were determined to be of breeding age (≥ 3 years-old). Interestingly, 2 of 13 dens surveyed of females that just turned 2 years-old each produced 1 female cub, indicating these females bred the previous summer as yearlings. To our knowledge, this is the first documentation of the American black bear achieving reproductive maturity this early. Spatial coordinates were recorded for 259 of these 378 dens of females ≥ 3 years-old (Fig. 4-1).

Assigning habitat states

Using ArcGIS 10.2.2 (ESRI 2011), we categorized each encounter (i.e., capture, recovery, den survey) as occurring in either a wildland or anthropogenic habitat state using 2001, 2006, and 2011 National Land Cover Databases (NLCD; Homer et al. 2015) at a spatial resolution of 30 m. The wildland habitat state was primarily composed of deciduous forest, mixed forest, woody wetlands, and emergent herbaceous pixels. The anthropogenic habitat state included low-, medium- and high-intensity development pixels, pasture/hay, and cultivated crops pixels. We incorporated agricultural pixels within our anthropogenic habitat state because crop and livestock depredation continues to be an NJDFW management concern (Wolgast et al. 2010) We incorporated the wildland-urban interface (hereafter, interface) within our anthropogenic with a 500 m buffer around all developed areas (comparable to Chapter 3). The resultant landscape was comprised of 41.4% wildland and 58.6% anthropogenic habitats, and included 1081 wildland and 903 anthropogenic encounters, and 115 wildland and 144 anthropogenic den surveys.

Estimating age- and habitat-specific fecundity patterns

We fit smoothing splines to the age-specific fecundity data obtained from the den surveys using the most parsimonious number of knots ($n = 4$) in Program R version 3.3.1 (R Core Team 2016). Initial fecundity estimates represent total cub production, including both male and female cubs, produced annually by black bears between 3 – 23 years-old in both wildland and anthropogenic habitats. Additionally, we determined the mean age of maternal sows, and assessed the mean number of female and male cubs produced per den across the 259 adult female dens with spatial coordinates, and tested for differences in habitat-specific means using *t*-tests. Cub sex ratios were estimated as the mean number of female cubs produced relative to male cubs per den in each habitat.

Estimating age-class- and habitat-specific survival and transition rates

We delineated black bear age-classes as: cubs (0 – 1 year-old), yearlings (1 – 2 years-old), sub-adults (2 – 4 years-old), prime-age adults (4 – 13 years-old), and old adults (> 13 years-old) based partly on the existing literature from western populations (Hovey and McLellan 1996, Hebblewhite et al. 2003). However, we additionally included an old adult age-class to account for the decline in fecundity occurring in wildland bears > 13 years-old. Sub-adults were demarcated from prime-age adults at 4 instead of 6 years-old, given that eastern black bear populations are characterized by earlier investment in reproduction and higher fecundity than western populations (Beston 2011). We estimated annual black bear survival probabilities for these age-classes within each habitat, and evaluated annual transition probabilities between habitats using a multistate framework as described in Chapter 3; however, herein we collapse Urban, Interface, and Agriculture states into one Anthropogenic state. Exploratory analysis initially indicated an average *net* annual movement rate from wildland-to-anthropogenic habitats of 0.147 ± 0.052 between 2000-01 and 2011-12 in northwestern NJ, and a harvest effect. Herein, during harvest years, we set the average *net* annual movement rate from wildland-to-anthropogenic habitats to 0.057. In harvest-moratoria years, we set the average *net* annual movement rate from wildland-to-anthropogenic habitats to 0.193 (J. D. Raithel, *Unpublished results*).

Constructing habitat-specific mean matrix population models

We constructed age-structured, post-breeding birth-pulse, Leslie matrix population models for female black bears in both wildland and anthropogenic habitats from mean survival and fertility rates estimated across 2000 – 2013 (Caswell 2006). Dimensions of the matrices (28 x 28) captures the age range of first captures and year-specific fecundity estimates; however, the sub-diagonals of the matrices were parameterized with appropriate age-class survival rates (Fig. 4-3). Fertility rates (top row of the matrices) were calculated as the product of: age-specific fecundity rate (Fig. 4-2), female cub survival rate (μ_c^{WD} = 0.749, μ_c^{AD} = 0.562), female cub sex ratio (0.529 in wildland component, 0.476 in anthropogenic component), and a habitat-specific inter-birth interval (0.572 in wildland component, 0.640 in anthropogenic component. As black bears with surviving cubs breed inter-annually (every other year), previous efforts have used a constant 0.5 multiplier to account for the inter-birth interval (Hovey and McLellan 1996, Hebblewhite et al. 2003). However, we modified the inter-birth interval to reflect the high cub mortality previously documented (see Chapter 2), allowing the proportion of maternal sows who lose cubs of the year to breed the following year, as: 1 / ((μ_c^{hD} x 2) + ((1 – μ_c^{hD}) x 1)). We included the small fertility contribution of ‘yearlings’ (0.027 – 0.036; Fig. 4-3) to account for the documented proportion of sows that bred as yearlings and produced cubs at 2 years-old (0.154). To quantify uncertainty in estimates of population growth rates, we estimated $\lambda_{95\%\ \mathrm{l\ CI}}$ and $\lambda_{95\%\ \mathrm{u\ CI}}$ using the lower and upper bounds of the 95% confidence intervals bounding age-class-specific survival estimates (Table 4-2).

Constructing year-specific harvest and harvest-moratoria matrices

As the wildland and anthropogenic component mean population matrices were derived from survival estimates, including six harvest and seven harvest-moratoria years, we next constructed harvest-year specific matrices (approximated from the estimated proportion of mortality attributable to harvest in that year, see Chapter 2) and mean harvest-moratoria year matrices for both wildland and anthropogenic habitats. For example, in 2010 harvest represented 51.1% of the total mortality; therefore, mean prime-age adult survival across 2000 – 2013 in wildlands (0.873) was adjusted downward by 0.065 (total age-class specific mortality x proportion of mortality attributable to harvest [0.127 x 0.511]) to 0.808. Similarly in harvest-moratoria years (2000-02, 2004, 2006-09), mean prime-age adult survival across 2000 – 2013 in wildlands (0.873) was adjusted upward by 0.068 (total age-class specific mortality x mean proportion of mortality attributable to harvest [0.127 x 0.537]) to 0.934.

Testing for correlations between reported problem bear behaviors and habitat-specific abundance projections

Between 2001 and 2013, NJDFW recorded 14,290 negative human-black bear interactions (hereafter conflicts), ranging from a low of 626 reported in 2004 (the year following the reintroduction of harvest) to a high of 1765 in 2008 (three years following the second moratorium). We previously demonstrated that the change in conflicts reported between year t and $t + 1$ was negatively correlated with increasing harvest plus management mortality in year t ($P = 0.028$; $R^2 = 0.338$; see Chapter 2). Thus, herein we regressed annual total conflicts reported in year t on both annual estimates of wildland and anthropogenic bear abundance in year t, between 2001 – 2013. Normality assumptions were met, as assessed by Lilliefors' test using R package nortest ($p = 0.632$, $p = 0.446$, respectively; Gross and Ligges 2015).

Projecting habitat-specific abundance and conflicts under future harvest scenarios

Incorporating the net annual wildland-to-anthropogenic transition probabilities in harvest and harvest moratoria years, and the relationship between conflicts observed and anthropogenic bear abundance, we projected wildland and anthropogenic bear abundance and associated conflicts from 2014 to 2025 under the following scenarios: a) eliminating the NJDFW black bear harvest (i.e., applying the mean harvest-moratoria years matrix models and transition rates from 2000-02, 2004, 2006-09); b) implementing an intermittent harvest as occurred from 2000 – 2013 (i.e., applying the mean matrix models and transition rates); c) applying current harvest rates (i.e., applying the mean harvest-year matrix models and transition rates from 2003, 2005, 2010-13); d) applying current harvest rates and achieving a 50% net reduction in annual wildland-to-anthropogenic transition probabilities (i.e., from 0.057 to 0.028); and e) increasing harvest rates by 10% above mean harvest-year matrix models (i.e., adjusting survival downward by 10% for all age-classes) and achieving a 50% net reduction in annual wildland-to-anthropogenic transition probabilities.

Results

Age- and habitat-specific vital rate patterns

Between 1987 – 2014 in northwestern NJ, adult (≥ 3 years-old) female black bears who denned in the anthropogenic (*n* = 144) habitats produced a greater number of total cubs per den (2.73 ± 0.096 vs. 2.35 ± 0.123; *P* = 0.014), invested in greater male cub production (1.28 ± 0.130 vs. 0.93 ± 0.114; *P* = 0.045), and were younger than wildland conspecifics (7.26 ± 0.346 vs. 8.61 ± 0.468; $P = 0.019$; Table 4-1). Anthropogenic bear fecundity was greater across all ages, rapidly increasing at 4 years-old, and did not exhibit reproductive senescence until > 18 years-old (Fig. 4-2). Documented female to male cub sex ratios were greater in wildland (52.9%: 47.1%) when compared to anthropogenic habitats (47.6%: 52.4%; Table 4-1).

Between 2000 – 2013 in northwestern NJ, female black bears in anthropogenic habitats exhibited lower survival rates across all age-classes than wildland conspecifics (Table 4-2); 95% confidence intervals about survival did not overlap for anthropogenic vs. wildland sub-adults (0.696 ± 0.029, 0.842 ± 0.031), or prime-age adults (0.747 ± 0.017, 0.873 ± 0.022).

Habitat-specific mean matrix population models

The increase in fecundity observed in the anthropogenic state was negated by decreased anthropogenic female survival across all age-classes. Mean matrix population models indicated that the anthropogenic habitat component of the population had a high probability of decline ($\lambda_{A} = 0.945$, 95% CI 0.848–1.034) and the wildland habitat component was increasing ($\lambda_{W} = 1.133$, 95% CI 1.025–1.213; Fig. 4-3). However during harvest-moratoria years, both wildland ($\lambda_{W\ No\ H} = 1.264$) and anthropogenic ($\lambda_{A\ No\ H} = 1.197$) components of the population rapidly grew. Adjusting mean survival downward by the proportion of mortality attributable to harvest (see Chapter 2) during harvest years, stabilized the wildland component ($\lambda_{W\ Harv} = 1.011$) and resulted in a rapid decline in the anthropogenic component ($\lambda_{A\ Harv} = 0.697$).

Habitat-specific population dynamics and conflicts

Applying year-specific harvest and harvest-moratoria matrices, and net annual wildland-to-anthropogenic transition probabilities in harvest and harvest-moratoria years, we projected total female abundance of 455 in 2000 increased to 1592 in 2009, which closely aligns with two independent abundance estimates occurring in 2009-10 (1545 and 1865 females, respectively; Huffman et al. 2010, Carr and Burguess 2011). This overall increase was primarily driven by increasing black bear abundance in anthropogenic habitats (Fig. 4-4). While we projected the wildland component of the population grew from 410 females in 2000 to 558 in 2009, we projected the anthropogenic component grew from 45 females in 2000 to 1,034 in 2009. As a result, annual conflicts reported were significantly correlated with annual estimates of anthropogenic bear abundance ($P < 0.001$; $R^2 = 0.698$); the inclusion of wildland bear abundance ($P = 0.536$; $R^2 = 0.681$) did not improve the strength of the correlation.

Eliminating the NJDFW black bear harvest resulted in an exponential increase in the anthropogenic component of the population and resultant conflicts between 2014 and 2025 (Fig. 4-4 a). Applying an intermittent harvest regime as occurred between 2000 and 2013 also resulted in increased abundance in both the wildland and anthropogenic components of the population as well as a projected increase in human-bear conflicts (Fig. 4-4 b). Although projecting current black bear harvest rates resulted in declines in anthropogenic bear growth ($\lambda_A = 0.697$) and conflicts, the wildland component, though intrinsically stable ($\lambda_W = 1.011$), also declined due to an annual net wildland-to-anthropogenic transition rate of 0.057 (Fig. 4-4 c). However, current black bear harvest rates coupled with a 50% reduction in annual net wildland-to-anthropogenic transition rate (from 0.057 to 0.028) resulted in a steep decline in anthropogenic bear growth and associated conflicts, and a relatively stable wildland component through 2025 (Fig. 4-4 d). A 10% increase in harvest over current rates coupled with a 50% reduction in annual net wildland-to-anthropogenic transition rate also resulted in a steep decline in anthropogenic bear growth ($\lambda_A = 0.672$) and resultant conflicts, and reduced the intrinsic wildland population growth rate to about 1 ($\lambda_W = 0.998$; Fig. 4-4 e).

Discussion

This study provides evidence that the introduction of a regulated black bear harvest in northwestern NJ succeeded in rapidly curbing the population growth rate of the anthropogenic component of the bear population (reducing $\lambda_{A\ No\ H} = 1.197$ to $\lambda_{A\ Harv} = 0.697$), and was effective in reducing annual human-bear conflicts from a high of 1,765 reported in 2008 to 837 conflicts reported in 2013 (Fig. 4-4). We demonstrate that the abundance of black bears within anthropogenic habitats is a strong predictor of human-bear conflicts ($P < 0.001$; $R^2 = 0.698$). The efficacy of the NJDFW black bear harvest, integrated into a comprehensive black bear management strategy that includes incident-response protocols and educational outreach aimed at reducing anthropogenic attractors, is driven by:

1) bears with a conflict history are disproportionately harvested relative to bears with no history of conflict (0.176 ± 0.025 vs. 0.109 ± 0.010; see Chapter 2);

2) problem bears consistently use the wildland-urban interface, a habitat with high harvest vulnerability (see Chapter 3);

Female bears that denned in anthropogenic habitats exhibited appreciably greater fecundity rates than wildland conspecifics across all ages (Fig. 4-2). It has long been understood that black bear reproductive maturation and fecundity rates are positively related to body weight (Jonkel and Cowan 1971, Stringham 1990); thus, it is unsurprising that the provisioning of reliable, high-caloric foods in anthropogenic areas induces earlier primiparity (Beckmann and Berger 2003b) and greater age-specific fecundity rates (Beckmann and Lackey 2008). New Jersey represents one of the most heavily human-dominated landscapes in North America, within which a sizeable contingent of black bears consistently exploit human-derived foods. Based on the locations of some of the capture sites deep within urban areas (see Chapter 3) and the comparatively minuscule median annual home range size previously documented in urban NJ females (11.37 km^2; Tri 2013), some individuals may be almost exclusively doing so.

To our knowledge, this study is the first to document that black bear yearlings are capable of breeding and producing offspring at 2 years of age. By 4 years of age, the anthropogenic bear fecundity rate sharply exceeded that of wildland females by 0.71 cubs per den. Further, while wildland bears are already reproductively senescing by 13 years-old, anthropogenic females did not abruptly reproductively decline until > 18 years of age; between the ages of 15 – 20 years-old, mean anthropogenic fecundity rates exceeded those of wildlands by a remarkable 1.28 ± 0.07 cubs per den. This increase in anthropogenic, relative to wildland, fecundity rates was characterized by markedly greater investment in male cub production (1.28 ± 0.130 vs. 0.93 ± 0.114, respectively; $P = 0.045$) and is consistent with the prediction that mothers in good nutritional condition should bias the sex ratio of their offspring toward males, given that males exhibit greater variation in reproductive value (Trivers and Willard 1973, Veller et al. 2016).

However, these increased anthropogenic fecundity rates did not translate into greater realized recruitment, given the low survival rates of anthropogenic, relative to wildland, cubs (0.562 ± 0.051 vs. 0.749 ± 0.049), yearlings (0.581 ± 0.055 vs. 0.763 ± 0.050), and sub-adults (0.696 ± 0.029 vs. 0.842 ± 0.031; Table 4-2). This dynamic, where increased black bear reproductive output in anthropogenic areas is offset by poor juvenile dispersal due to vehicle collisions and decreased age-specific survival (Beckmann and Berger 2003b, Beckmann and Lackey 2008), may be the paradigm in highly novel and heterogeneous anthropogenic landscapes. Age-class is consistently identified as an important predictor of black bear mortality risk (see Chapters 2 and 3), highlighting the importance of experiential learning in assessing risk in ursids (Gilbert 1999). Yet the learning curve for juvenile bears may be steep and unforgiving with respect to vehicular collisions, as they represent the greatest cause-specific source of mortality and limitation to population growth for carnivore populations in human-dominated landscapes (Rodewald and Gehrt 2014). In northwestern NJ, the probability of juvenile bears dying from harvest (0.131 – 0.233) and lethal control (0.052 – 0.069) was swamped by all other causes of mortality (0.372 – 0.444), primarily consisting of vehicle strikes (see Chapter 2). Although vehicle strikes undoubtedly contributed to lower intrinsic population growth rates in anthropogenic relative to wildland habitats, both the anthropogenic ($\lambda_{A\ No\ H}$ = 1.197) and wildland ($\lambda_{W\ No\ H}$ = 1.264) components of the NJ bear population rapidly grew in the absence of harvest.

The high elasticity of adult female black bear survival identified across regions (Freedman et al. 2003, Hebblewhite et al. 2003) indicates that natural selection has buffered this vital rate against spatiotemporal variability (Gaillard and Yoccoz 2003). In black bear populations protected from hunting, adult female survival was relatively invariant compared to the high spatial and temporal variation in observed recruitment rates (Mitchell et al. 2009). These elasticity patterns suggest that management actions that influence adult female survival can have consequential impacts on population dynamics. Adult female black bears were twice as likely to be harvested as adult males in northwestern NJ, (0.163 ± 0.014 vs. 0.087 ± 0.012; see Chapter 2), which confirms that harvesting adult females is the most effective approach to curbing population growth.

The mean wildland and anthropogenic population matrix models indicated that wildland habitats served as a population source (λ_W = 1.133, 95% CI 1.025–1.213) bolstering anthropogenic sinks (λ_A = 0.945, 95% CI 0.848–1.034; Fig. 4-3). Importantly, these projections assume that hunters will continue to take a disproportionate number of bears from the wildland-urban interface, included herein within the anthropogenic habitat state. Given the efficacy of a regulated harvest in mitigating conflicts in northwestern NJ, it will likely continue to be an important component of an integrated strategy that facilitates human-bear coexistence in this human-dominated, social-ecological system. Ultimately, this tool must be carefully applied and monitored, as the stability of the population at the landscape level hinges on reducing movement from wildland to anthropogenic habitats.

Management implications

Unequivocally, the most cost-effective management action in deterring most bears from using anthropogenic areas is to reduce available attractants by 55-70% (Baruch-Mordo et al. 2013). State wildlife agencies like NJDFW, which lack the legislative authority to mandate bear-resistant garbage receptacles in residential communities, must continue to substantially invest in educational outreach programs that encourage their voluntary adoption. Faced with declining budgets, escalating wildlife conflicts, and increasingly polarized constituencies (Johnson and Sciascia 2013), state agencies must balance the preservation of viable black bear populations with the protection of human welfare and property in a cost-effective manner (Hristienko and McDonald 2007). Given forecasts of agricultural and urban expansion, protected areas alone will be inadequate to ensure carnivore conservation in the Anthropocene; thus, innovative interventions must be explored to reduce human-carnivore conflicts outside of protected areas (Di Minin et al. 2016). The recreational bear harvest in northwestern NJ exemplifies one such intervention, inducing a demographic response in black bears that reduces human-black bear conflicts in an anthropogenic landscape.

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Tables and Figures

Table 4-1. Black bear fecundity from 259 adult (≥ 3 years-old) female den locations in wildland and anthropogenic habitats in New Jersey, USA from 1987 – 2014. For comparative purposes, 2 year-olds (yearlings during the last breeding season) and sows with yearlings present at the den site have been removed. Significant differences (at $\alpha < 0.05$) are denoted with an *.

	Den Location Habitat State		
	Wildland (n = 115)	Anthropogenic (n = 144)	P-value
Mean – Total Cubs	2.35 ± 0.123 *	2.73 ± 0.096 *	0.014 *
Range – Total Cubs	0 – 5	0 – 5	
Mean – Female Cubs	1.04 ± 0.133	1.16 ± 0.137	0.526
Range – Female Cubs	0 – 4	0 – 4	
Mean – Male Cubs	0.93 ± 0.114 *	1.28 ± 0.130 *	0.045 *
Range – Male Cubs	0 – 3	0 – 4	
Female : Male Sex Ratio	52.9% : 47.1%	47.6% : 52.4%	
Mean – Sow Age (years)	8.61 ± 0.468 *	7.26 ± 0.346 *	0.019 *
Range – Sow Age (years)	3 – 28	3 – 26	

Table 4-2. Mean survival estimates for 1,312 female black bears by age-class and habitat state (i.e., wildland and anthropogenic habitat components) in New Jersey, USA, from 2000 – 2013. Significant differences (at $\alpha < 0.05$) are denoted with an *.

	Habitat State							
	Wildland Habitats				Anthropogenic Habitats			
Age-class	Mean Survival	SE	95% l CI	95% u CI	Mean Survival	SE	95% l CI	95% u CI
Cubs (0 – 1 years)	0.749	0.049	0.641	0.833	0.562	0.051	0.461	0.659
Yearlings (1 – 2 years)	0.763	0.050	0.652	0.847	0.581	0.055	0.471	0.683
Sub-Adults (2 – 4 years)	0.842*	0.031	0.771	0.893	0.696*	0.029	0.635	0.750
Prime Adults (4 – 13 years)	0.873*	0.022	0.822	0.911	0.747*	0.017	0.713	0.778
Old Adults (> 13 years)	0.858	0.037	0.769	0.916	0.722	0.049	0.616	0.808

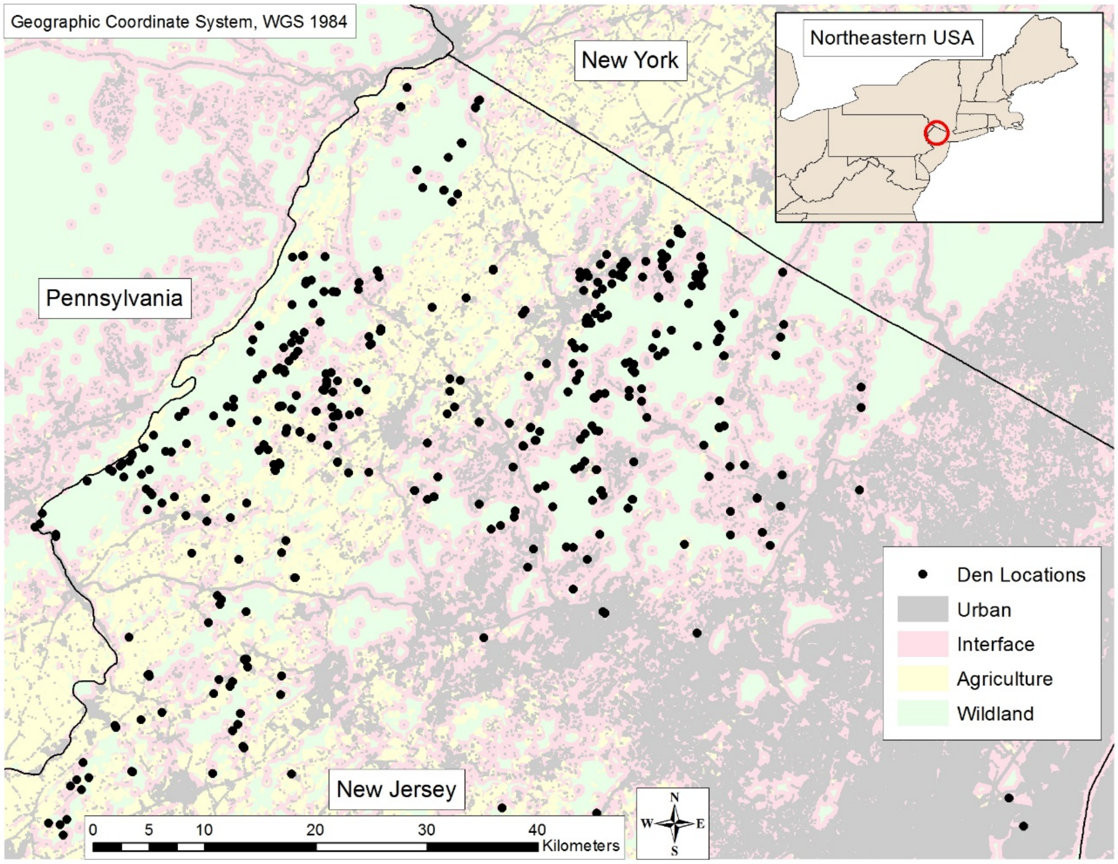


Figure 4-1. Spatial distribution of New Jersey Division of Fish and Wildlife black bear den site locations from 1987 – 2014 overlaid on habitat states from National Land Cover Data 2011.

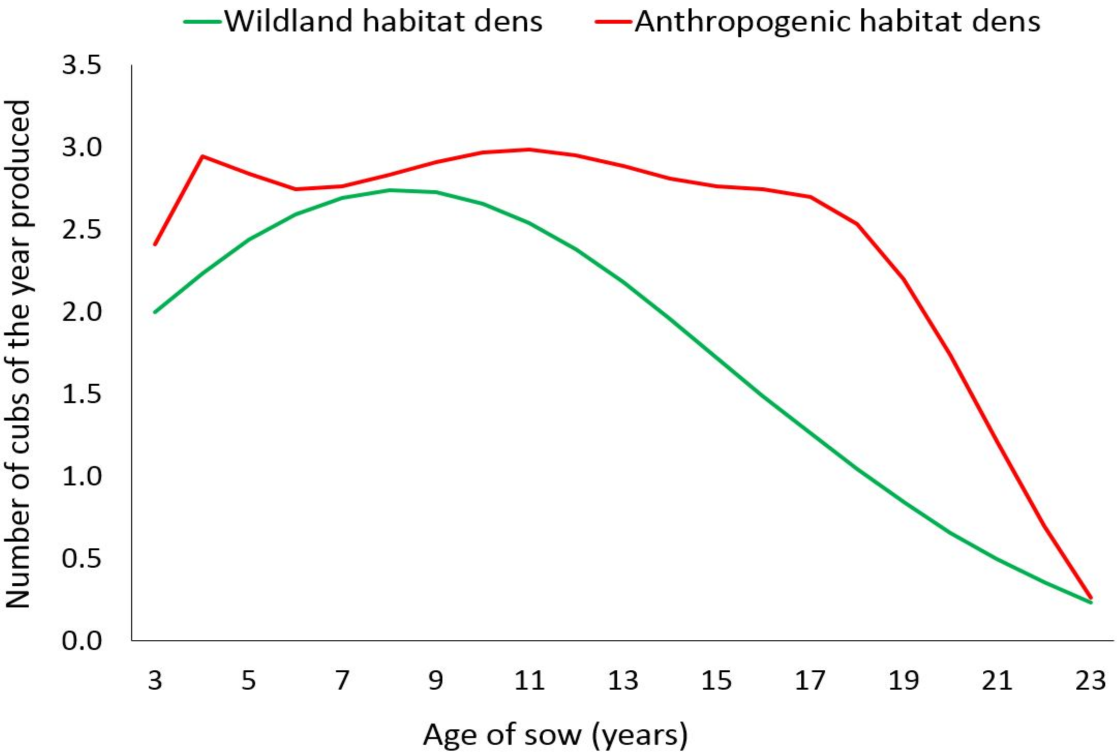


Figure 4-2. Black bear age-specific fecundity from wildland and anthropogenic habitat den locations in New Jersey, USA, from 1987 – 2014. Models were fit using smoothing splines with the most parsimonious number of knots ($n = 4$) to produce a cubic function.

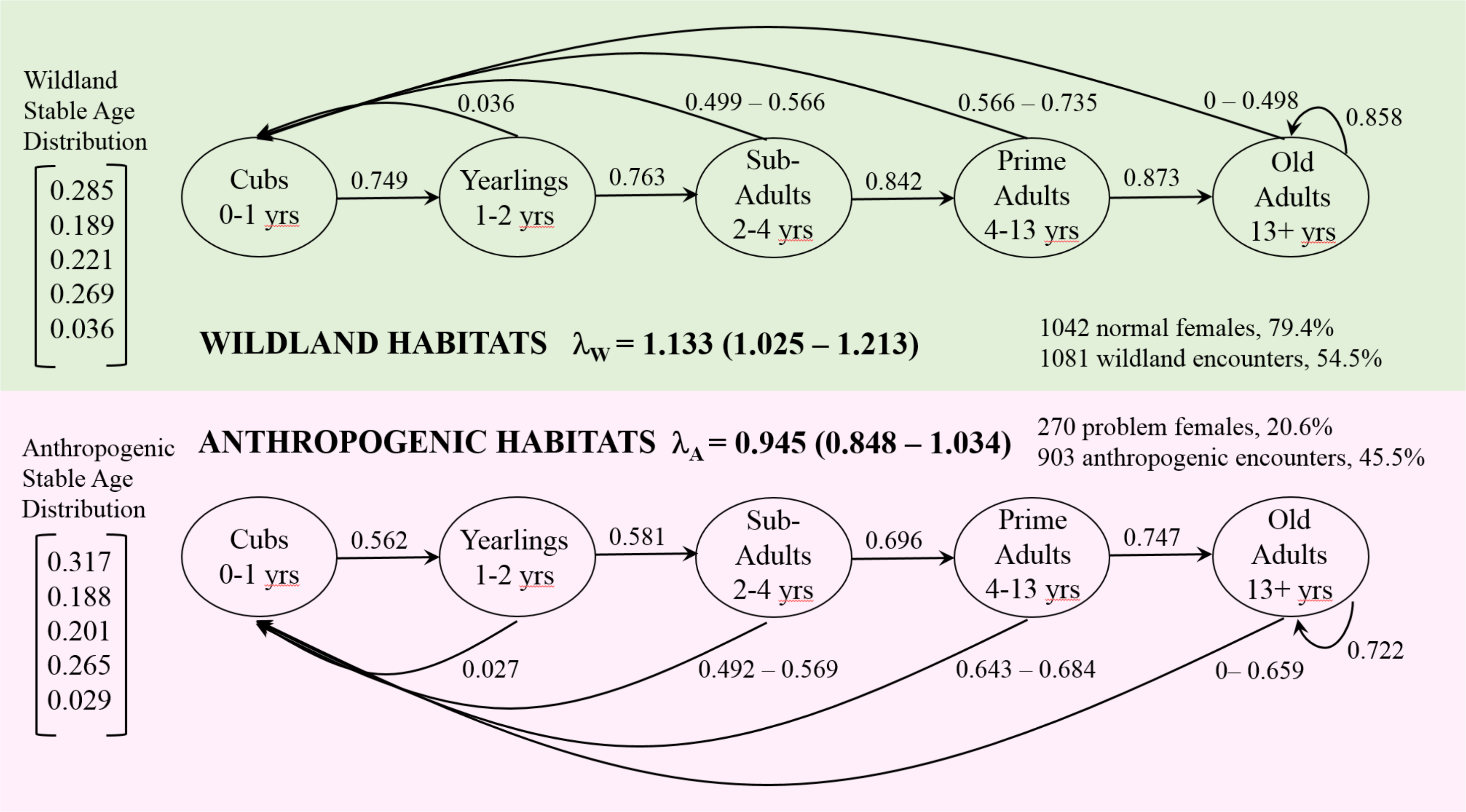
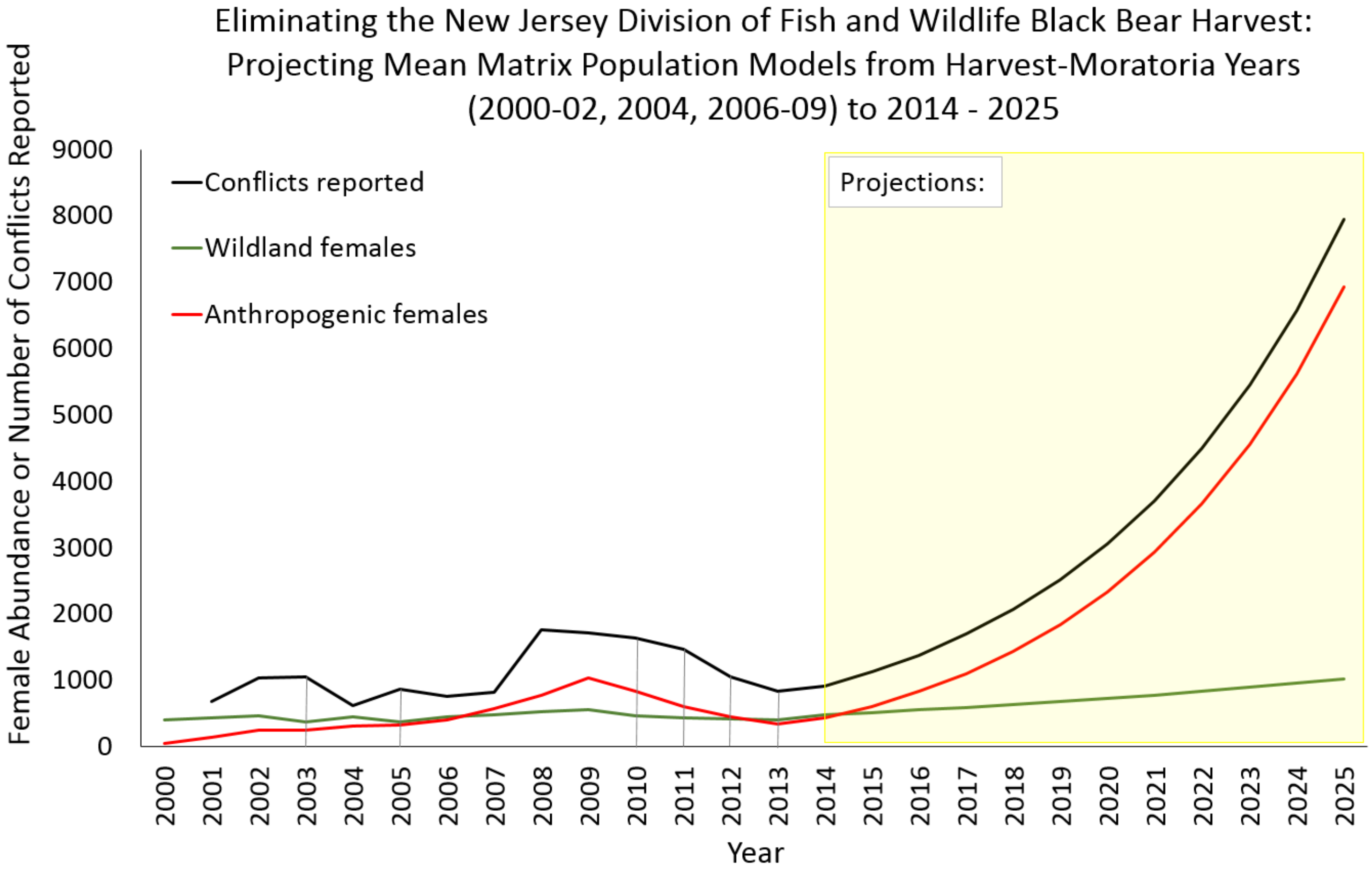
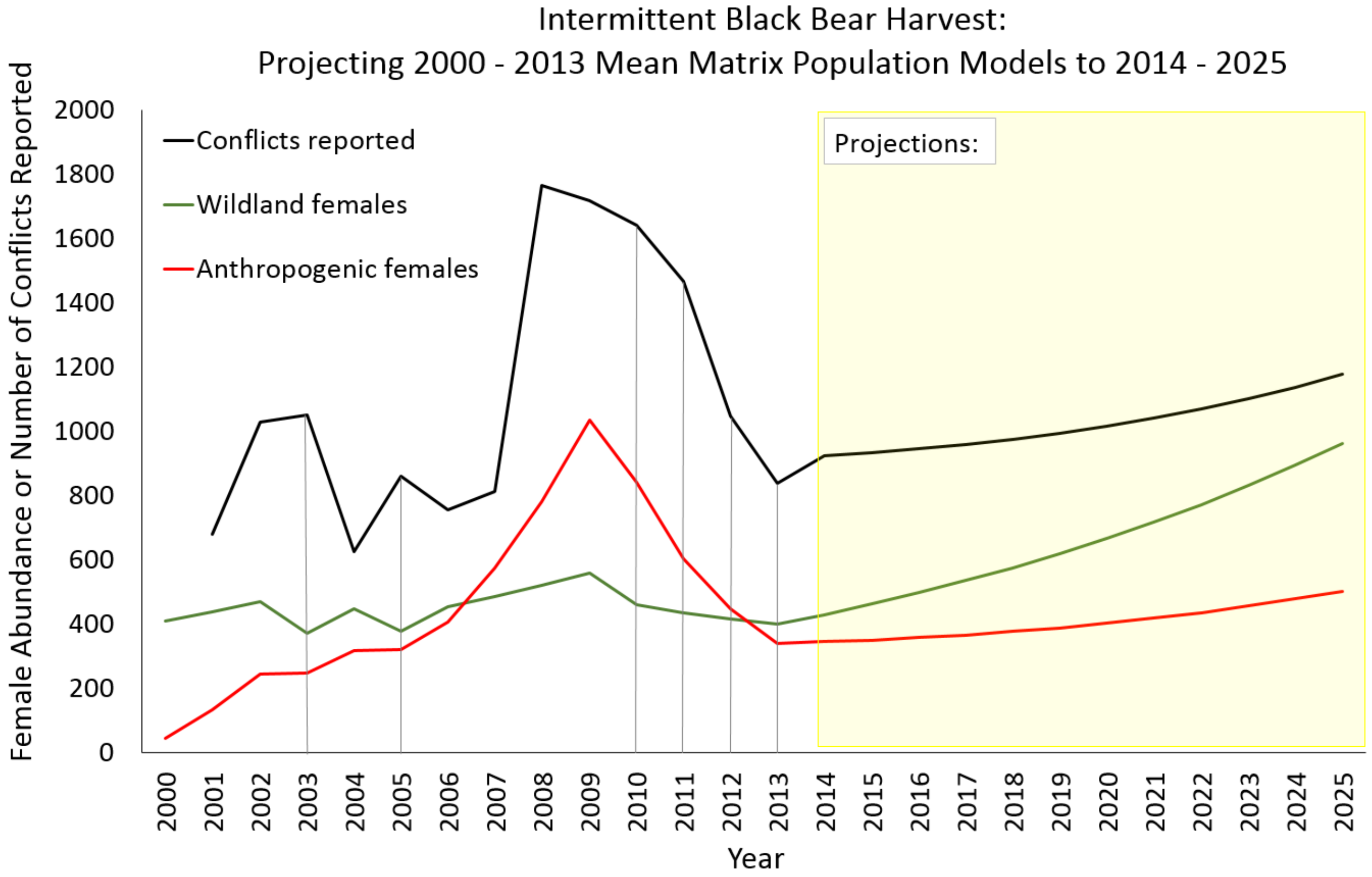

Wildland Stable Age Distribution
0.285
0.189
0.221
0.269
0.036
Cubs 0-1 yrs
Yearlings 1-2 yrs
Sub-Adults 2-4 yrs
Prime Adults 4-13 yrs
Old Adults 13+ yrs
0.749
0.763
0.842
0.873
0.036
0.499 – 0.566
0.566 – 0.735
0 – 0.498
0.858
WILDLAND HABITATS $\lambda_W = 1.133$ (1.025 – 1.213)
1042 normal females, 79.4%
1081 wildland encounters, 54.5%
Anthropogenic Stable Age Distribution
0.317
0.188
0.201
0.265
0.029
ANTHROPOGENIC HABITATS $\lambda_A = 0.945$ (0.848 – 1.034)
270 problem females, 20.6%
903 anthropogenic encounters, 45.5%
Cubs 0-1 yrs
Yearlings 1-2 yrs
Sub-Adults 2-4 yrs
Prime Adults 4-13 yrs
Old Adults 13+ yrs
0.562
0.581
0.696
0.747
0.027
0.492 – 0.569
0.643 – 0.684
0– 0.659
0.722

Figure 4-3. Female survival and fertility estimates by age-class used to parameterize mean population matrix models in wildland and anthropogenic habitat components of the black bear population in New Jersey, USA, from 2000 - 2013. Estimated long-term geometric rates of population growth (λ_h), and stable age distributions are presented for each habitat state (i.e., wildland and anthropogenic components). Survival estimates were derived from 1,312 female black bear encounter histories, including 1,081 wildland and 903 anthropogenic habitat encounters (i.e., live recaptures and dead recoveries). Fertility estimates were derived from 259 adult female ($\geq$ 3 years-old) den surveys incorporating habitat-specific fecundity estimates adjusted by habitat-specific cub survival, female cub production, and an inter-annual breeding factor. Yearling (individuals 1-2 years-old) fertility estimates were derived from observations of 13 individuals, 2 of which (0.154) produced 1 female cub, at the beginning of their second year.

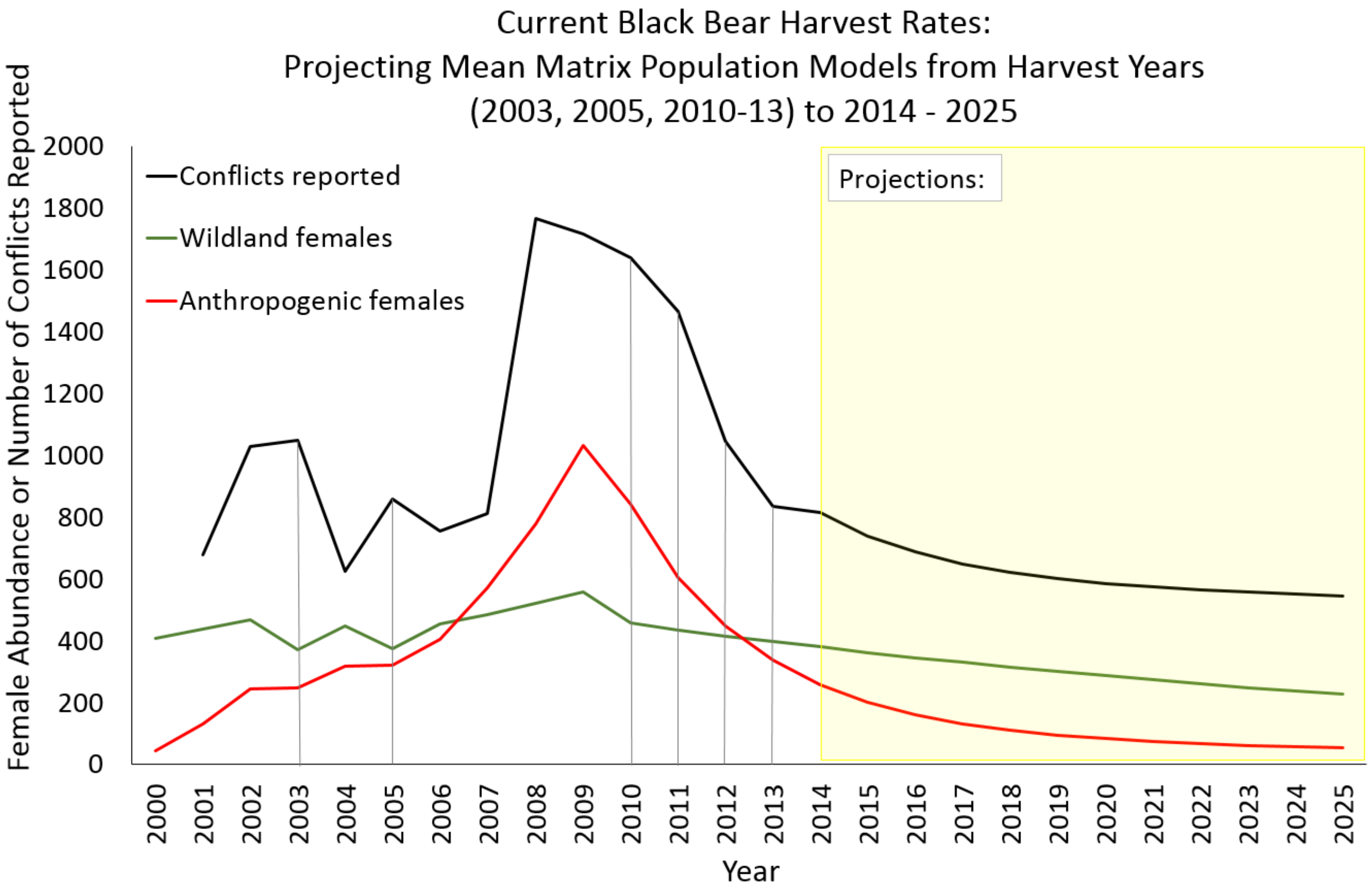
a)


Eliminating the New Jersey Division of Fish and Wildlife Black Bear Harvest:
Projecting Mean Matrix Population Models from Harvest-Moratoria Years
(2000-02, 2004, 2006-09) to 2014 - 2025
Female Abundance or Number of Conflicts Reported
9000
8000
7000
6000
5000
4000
3000
2000
1000
0
Conflicts reported
Wildland females
Anthropogenic females
Projections:
2000
2001
2002
2003
2004
2005
2006
2007
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
2018
2019
2020
2021
2022
2023
2024
2025
Year

b)


Intermittent Black Bear Harvest:
Projecting 2000 - 2013 Mean Matrix Population Models to 2014 - 2025
Female Abundance or Number of Conflicts Reported
2000
1800
1600
1400
1200
1000
800
600
400
200
0
Conflicts reported
Wildland females
Anthropogenic females
Projections:
2000
2001
2002
2003
2004
2005
2006
2007
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
2018
2019
2020
2021
2022
2023
2024
2025
Year

c)


Current Black Bear Harvest Rates:
Projecting Mean Matrix Population Models from Harvest Years
(2003, 2005, 2010-13) to 2014 - 2025
Female Abundance or Number of Conflicts Reported
2000
1800
1600
1400
1200
1000
800
600
400
200
0
Conflicts reported
Wildland females
Anthropogenic females
Projections:
2000
2001
2002
2003
2004
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2007
2008
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2016
2017
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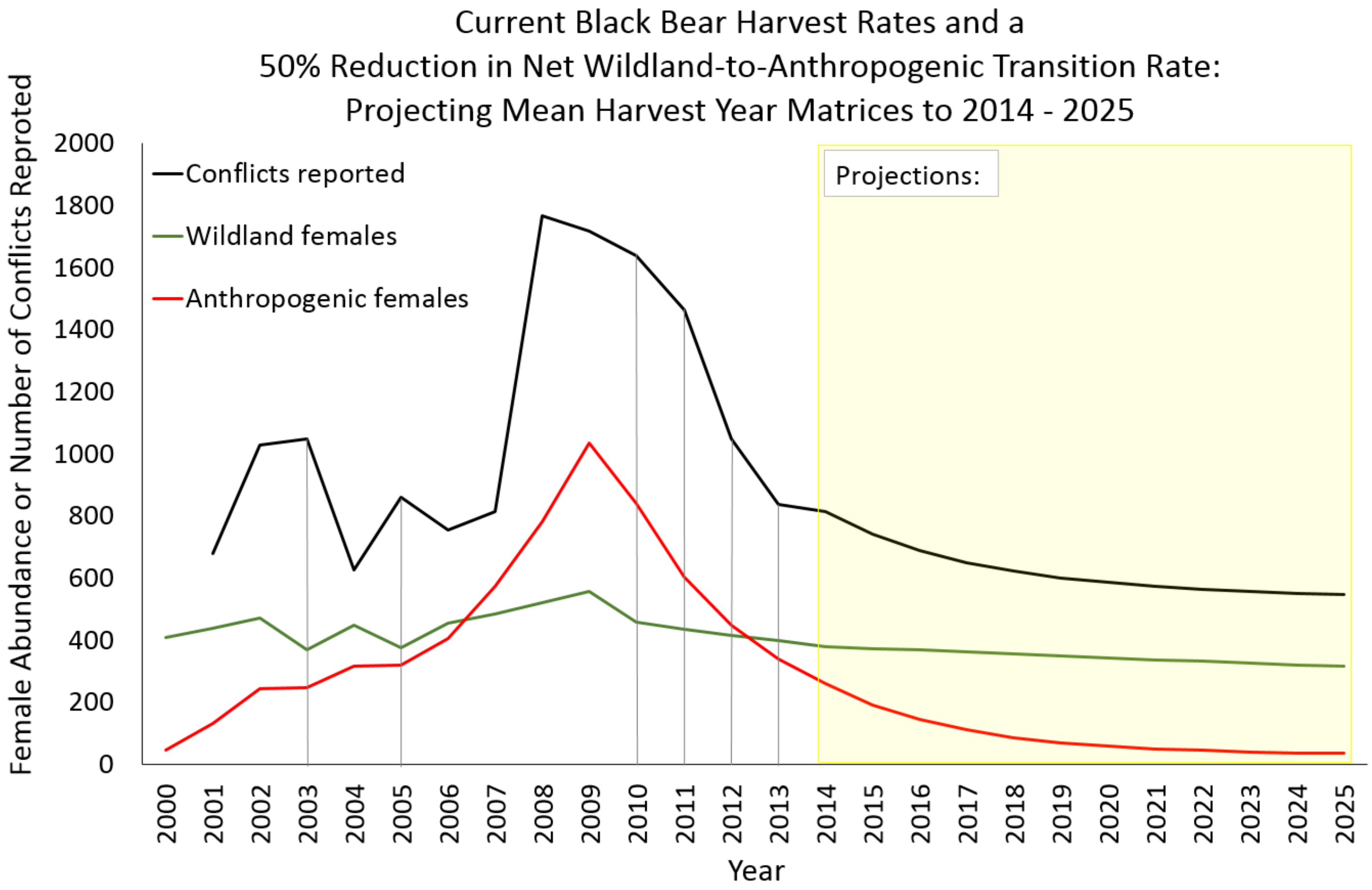
2019
2020
2021
2022
2023
2024
2025
Year

d)


Current Black Bear Harvest Rates and a
50% Reduction in Net Wildland-to-Anthropogenic Transition Rate:
Projecting Mean Harvest Year Matrices to 2014 - 2025
Female Abundance or Number of Conflicts Reproted
2000
1800
1600
1400
1200
1000
800
600
400
200
0
Conflicts reported
Wildland females
Anthropogenic females
Projections:
2000
2001
2002
2003
2004
2005
2006
2007
2008
2009
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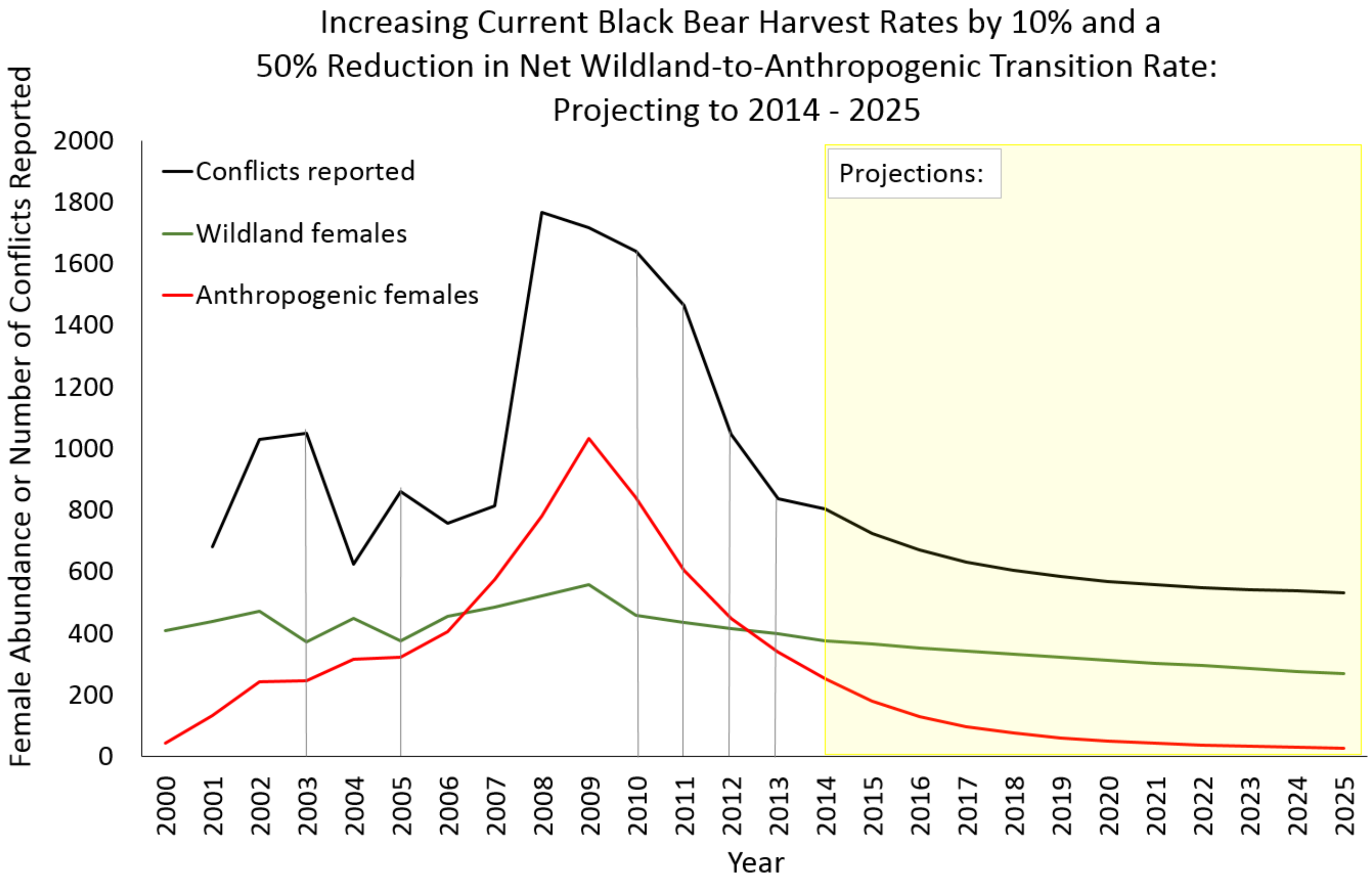
2024
2025
Year

e)


Increasing Current Black Bear Harvest Rates by 10% and a
50% Reduction in Net Wildland-to-Anthropogenic Transition Rate:
Projecting to 2014 - 2025
Female Abundance or Number of Conflicts Reported
2000
1800
1600
1400
1200
1000
800
600
400
200
0
Conflicts reported
Wildland females
Anthropogenic females
Projections:
2000
2001
2002
2003
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2024
2025
Year

Figure 4-4. Reported human-black bear conflicts in New Jersey, USA, between 2001 – 2013, and projections of estimated conflicts associated with habitat-specific female population dynamics from 2014 – 2025. Projections are derived from wildland and anthropogenic mean population matrices in non-harvest years coupled with harvest year-specific matrices. Transition rates between wildland and anthropogenic habitats were derived from means across harvest and harvest-moratoria years estimated from a fully time-dependent model. Projections in wildland and anthropogenic female black bear abundances and predicted conflict rates were extrapolated between 2014 – 2025 under the following scenarios: a) eliminating the NJDFW black bear harvest (i.e., applying the mean harvest-moratoria years matrix models and transition rates from 2000-02, 2004, 2006-09); b) implementing an intermittent harvest as occurred from 2000 – 2013 (i.e., applying the mean matrix models and transition rates); c) continuing at current harvest rates (i.e., applying the mean harvest-year matrix models and transition rates from 2003, 2005, 2010-13); d) continuing current harvest rates and achieving a 50% net reduction in annual wildland-to-anthropogenic transition probabilities (i.e., from 0.057 to 0.028); and e) increasing harvest rates by 10% above mean harvest-year matrix models (i.e., adjusting survival downward by 10% for all age-classes) and achieving a 50% net reduction in annual wildland-to-anthropogenic transition probabilities. Harvest years are represented by gray vertical lines.

CHAPTER 5

WHY DOES THE REGULATED HARVEST OF BLACK BEARS AFFECT THE RATE OF HUMAN-BEAR CONFLICTS IN NEW JERSEY?[2]

ABSTRACT

Humanity has a miserable track record in conserving large carnivores: from Paleolithic hunters skinning the enormous cave lion 15,000 years ago to the contemporary loss of the marsupial Tasmanian tiger. Today, several iconic members of the order *Carnivora* are on the brink of extinction (Amur leopards, Asiatic cheetahs), and over 75% of the world's 31 large carnivore species have experienced alarming population declines, often directly from human persecution. Yet, several species of large predators have dramatically rebounded (European gray wolf, American black bear) in the most unlikely of places: heavily human-dominated landscapes. For example, the black bear population in northwestern New Jersey (NJ), the state with the highest human densities in the United States, has exponentially increased over sixfold in just 15 years. During this period of unprecedented suburban sprawl in NJ there have been over 26,500 reported human-bear interactions including seven attacks on humans and one human fatality. Given accelerating anthropogenic landscape transformation, there simply are not enough large tracts of wildlands remaining to alone support expanding bear populations. Thus, American black bear conservation in the Anthropocene may ultimately depend upon society's tolerance for this large carnivore in areas where people live, work, and recreate.

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[2] Raithel, Jarod D., Melissa J. Reynolds-Hogland, Patrick C. Carr, and Lise M. Aubry (2017). *In Press.* Why does the regulated harvest of black bears affect the rate of human-bear conflicts in New Jersey? *Case Studies in the Enviroment*

In an effort to curb bear population growth and reduce conflicts, the first regulated NJ black bear harvest in over three decades was held in 2003 resulting in an acrimonious public debate. How can objective population ecology help us make informed decisions about management actions that elicit such strong emotional responses among different stakeholder groups?

LEARNING OUTCOMES

Students will evaluate how sex, age-class, and behavior (problem vs. normal) affect the probability that black bears in northwestern NJ die from harvest, lethal control, and other causes of mortality like vehicle strikes. Given these results, students will then propose possible explanations for the observed correlation between bear harvest rates and subsequent declines in nuisance bear behaviors reported. Informed by this remarkable dataset comprised of over 3,500 individual bears collected over 33 years, students will ultimately have a meaningful discussion about whether a carefully regulated bear harvest should be included in an integrated management strategy to conserve American black bears.

Classroom Tested? Yes

INTRODUCTION

Large, terrestrial predators (e.g., wolves, big cats, and bears) are some of humanity's most beloved animals. Ironically, they are also some of our planet's most imperiled species (Appendix 5.4, Slide #2) as 24 of the 31 largest carnivore species are experiencing population declines, and they currently occupy an average of only 47% of their historical ranges [1]. This is especially disconcerting to ecologists as the functional removal of apex predators can trigger trophic cascades in food webs resulting in unanticipated and far-reaching ecological consequences [2] (Appendix 5.4, Slide #3). Surprisingly though, populations of several species of large carnivores have stabilized and are increasing in landscapes that have been heavily transformed by humans such as the European gray wolf (*Canis lupus*) [3] and the American black bear (*Ursus americanus*) [4] (Appendix 5.4, Slide #4). Black bears were almost extirpated in New Jersey (NJ), United States, during the 1950s reduced to less than an estimated 100 individuals. Yet the population recovered to 450–500 bears by the mid-1990s, and then exponentially increased in abundance and spatial range reaching 3,200–3,400 bears by 2010, with densities in some areas exceeding 1.3 bears/km^2 [5] (Appendix 5.4, Slide #5).

Black bears are opportunistic generalists that exhibit high behavioral plasticity; they are remarkably capable of exploiting human-derived food sources such as garbage, agricultural crops, ornamental fruit trees, apiaries, livestock, bird feeders, pet food, etc. [6]. Black bear recovery in NJ coincided with a rapidly increasing human population and a shift in settlement patterns toward sprawling suburban communities (Appendix 5.4, Slide #6). Since 2001, the New Jersey Division of Fish and Wildlife (NJDFW) has spent over US$9 million on black bear management, responding to over 26,500 human-bear interactions, including >1,400 incidents of verified property damage, >400 livestock kills, >250 pet attacks and/or kills, and seven human attacks, including one fatality (Appendix 5.4, Slide #7). In their comprehensive black bear management report, NJDFW concluded that this level of human-bear conflict is both culturally and fiscally unsustainable [7].

With large, potentially dangerous predators, the cultural carrying capacity (the number of black bears humans will tolerate) is often much lower than the ecological carrying capacity (the number of black bears the environment can support) [8]. Given the robust recovery of black bears across the Mid-Atlantic United States, conservation efforts must now pivot from facilitating population growth toward shaping both bear and human behaviors to promote coexistence in shared landscapes. By reducing human-bear conflicts, we can ultimately bolster the long-term viability of this species by increasing the abundance of bears local stakeholders will tolerate outside of protected areas [7].

In an attempt to curb bear population growth, reduce conflicts with humans, and achieve a bear population that is culturally acceptable, NJDFW reinstated a limited, lottery-based, six-day black bear hunt in December 2003—the first such harvest in NJ in over three decades. Following objections by some stakeholder groups (Appendix 5.4, Slide #8), a harvest moratorium occurred in 2004, the hunt was reopened in 2005, then closed again between 2006 and 2009, and has been reopened since 2010. For those familiar with experimental design, this intermittent harvest represents a treatment that allows us to test the effects of bear harvest on nuisance complaints reported in this social-ecological system. It is important to note that this harvest is adaptive in that quotas, i.e. harvest limits, are set annually based on the estimated rate of population growth, and the hunt is immediately closed if that quota is reached prior to the close of the six-day black bear hunting season. Further, recreational harvest is just one component in NJDFW's integrated black bear management plan which also includes educational outreach programs (detailed here: http://www.state.nj.us/dep/fgw/bearfacts_education.htm) and an incident-response protocol for managers and public safety officers (described below, and detailed here: http://www.nj.gov/dep/fgw/bearfacts_resandmgt.htm). For example beginning in 2008, NJDFW began heavily investing in bear educational resources including informational brochures, classroom kits, and over 100 public presentations annually reaching over 10,000 in attendance [7].

Our research team, comprised of bear managers from NJDFW, researchers and donors from the non-profit organization Bear Trust International, and population ecologists from Utah State University, sought to understand if the socially divisive management actions harvest and lethal control (i.e., humanely euthanizing threatening bears), when coupled with existing educational and incident response protocols, were effective in reducing human-bear conflicts (Appendix 5.4, Slide #9) [9]. Our analyses were guided by the following three, big questions:

1. At what rates do black bears die from the following mortality causes: a) regulated bear harvest, b) lethal control by managers, and c) all other sources combined (e.g., vehicle strikes, illegal kills, intraspecific infanticide, etc.)?
2. How does bear sex (female or male), age-class (cub, yearling, or adult), and behavior (problem or normal bear) affect the probability of dying from the different causes of mortality listed above? In other words, if you are an adult male bear, are you more or less likely to be harvested than a yearling male? If you become a "problem" bear, are you more or less likely to be lethally controlled by managers?
3. Is there a relationship between the rates of bears harvested and lethally controlled in year t and the number of nuisance complaints that are reported in year $t + 1$?

CASE EXAMINATION

Data Collection

From 1981 to 2014, NJDFW personnel conducted 5,185 black bear captures, marking 3,533 different individuals with unique metal ear tags (1,614 females, 1,919 males), including 1,344 cubs of the year, 877 yearlings, and 1,312 adults (Appendix 5.4, Slide #10). NJDFW employs a standardized incident-response framework for responding to bears deemed a threat to human safety, agricultural crops and/or property, or exhibiting nuisance behavior. All captured bears are designated by NJDFW managers as one of the following behavioral categories: I) Threat: including human, livestock, and unprovoked pet attacks, home entries, and agricultural/property damage >US$500; II) Nuisance: including habitual visits to garbage containers, dumpsters and/or birdfeeders, and property damage <US$500; and III) Normal: including bears observed by hunters, hikers, or campers in bear habitat, or dispersing animals that wander through rural and suburban communities. Threatening bears are lethally controlled as soon as possible throughout the year. Nuisance bears, if trapped, are aversively conditioned on-site. Aversive conditioning entails using non-lethal stimuli, including rubber buckshot, pyrotechnics, and bear dogs, to ensure that nuisance bears associate undesirable behaviors (e.g., foraging in anthropogenic habitats) with a negative experience.

NJDFW received 26,582 incident reports from the general public between 2001 and 2013 and categorized those as 2,277 Threats, 12,013 Nuisances, and 12,292 Normal interactions. The cause of mortality was later documented for 1,338 of the 3,533 captured and marked individuals, consisting of 556 hunter harvests, 396 management mortalities

(158 euthanized by managers and police officers, 238 lethally controlled with agricultural depredation permits), and 386 other mortalities (primarily composed of 271 vehicle strikes and 58 illegal kills). Agricultural depredation permits are granted to farmers, livestock producers, and apiarists who sustain >US$500 in loss, as verified by NJDFW bear managers, to personally control problem bears. All controlled bears are reported to NJDFW and included in the analyses herein.

Estimating Cause-Specific Mortality Probabilities

To address questions 1 and 2, we employed a demographic modeling tool used to estimate survival and cause-specific mortality rates of mobile species, aptly named, capture-mark-recapture (CMR) methods [10] (Appendix 5.4, Slide #11). The technique CMR initially allows us to estimate the probability that an animal, a black bear in our case, previously marked with a unique metal ear tag will be recaptured on another trapping occasion or physically recovered if it dies. Based on these “detection probabilities,” we then estimate the probability that a bear will transition from an “Alive State” to one of our three defined dead states (Harvested, Lethally Managed, or Dead All Other Causes) in any given year. This probability of transitioning to a dead state is one technique population ecologists employ to estimate “cause-specific mortality rates” of organisms as a function of sex, age, and physical and/or behavioral traits. For example, we now have an empirically derived estimate of the probability that an adult male will die from harvest in a given year during the study period (Appendix 5.4, Slide #12).

Estimating If There is a Correlation Between Harvest and Lethal Control Rates in Year t, and Problem Behaviors Reported in Year t + 1

To achieve objective 3, we first plotted all bear behaviors reported (i.e., Threat, Nuisance, and Normal), as well as our cause-specific mortality rates (described above) for each year we had reporting data (2001 through 2013; Appendix 5.4, Slide #13). We also plotted the proportion of each bear behavior relative to other behaviors during these same years (Appendix 5.4, Slide #14). We then used the statistical tool linear regression to determine if there was a significant ($\alpha = 0.05$) relationship between the rates of bears dying from all causes combined (total mortality) in year t and the number of nuisance behaviors reported in year $t + 1$. Additionally, we used the same methods to evaluate if just harvest and lethal control rates in year t explained variation in the number of nuisance behaviors reported during year $t + 1$ (Appendix 5.4, Slide #15). Note, these rates were coupled to assess the additional influence of harvest on the background, inter-annual lethal control rates, and ultimately, increase our sample size (see Teaching Notes for more detail).

A Final Note on “Significance” and Uncertainty in Social-Ecological Systems

In the ensuing discussion it is important to remind students that a defining attribute of the scientific process is an explicit attempt to quantify uncertainty. For example, we present a “significant” correlation between increasing harvest and lethal control rates and subsequent declines in nuisance behaviors reported by expressing a low calculated probability (i.e., p-value = 0.028) which informs the audience that these data are highly unlikely if the null hypothesis is true, i.e., there is no relationship between our predictor and response variables. That said, there is much we do not understand as ecological drivers (e.g., increasing bear densities, climate variability) interact in complex ways with the social dimensions inherent to the system (e.g., the probability that a person who experiences a negative bear interaction reports this interaction). As an example, untangling how much of the observed decline in nuisance bear behaviors is due to humans altering their behavior in response to educational outreach versus bears adapting to the presence of a new predator in the system, i.e., the human hunter, is likely inextricably confounded with this dataset. However, it could be addressed with the collection of additional data within a well-executed experimental design.

Case-Study Questions

1. Examine our tables of cause-specific mortality estimates to answer the following questions:
    a. How does the age-class (cub, yearling, and adult) and sex of the black bear (female vs. male) interact to influence its probability of being harvested? Remember to examine whether 95% confidence intervals overlap to determine if differences are significant. (Appendix 5.4, Slide #16)
    b. What is the dominant source of mortality for young bears (cubs and yearlings) in this human-dominated landscape? Why do you think this category is so high? (Appendix 5.4, Slide #16)
    c. How does behavior affect the probability that black bears in northwestern NJ are harvested and lethally controlled? (Appendix 5.4, Slide #17)

2. Examine our temporal line/bar graph on Slide #13 (Appendix 5.4): Do you see a relationship between nuisance behaviors reported and the implementation of harvest in years 2003, 2005, and 2010–2013?
3. Examine our temporal line graph on Slide #14 (Appendix 5.4): In what year does the proportion of normal behaviors reported begin increasing relative to problem behaviors (that begin decreasing at this inflection point)? What event occurred during this time, and what are the implications for bear management?
4. Examine our linear regressions in Slide #15 (Appendix 5.4): Is the change in nuisance behaviors reported in year $t + 1$ more strongly correlated with total mortality in year t or harvest + management mortality rates in year t?
5. Provide three hypotheses explaining the observed correlation between implementing the new bear harvest and subsequent declines in nuisance bear behaviors reported. They do not have to be “mutually-exclusive,” i.e., two or more hypotheses could be occurring simultaneously.

CONCLUSION

The conservation of large, ecologically-influential carnivores necessitates prioritizing innovative interventions outside of protected areas as coexistence between humans and carnivores likely represents the only realistic way forward in ensuring carnivore persistence in heavily transformed, novel social-ecological systems [11]. Optimizing human-carnivore coexistence in these shared landscapes demands mutualistic coadaptation, i.e., both humans and carnivores learning from experience and altering their behaviors to minimize negative impacts on each other [12] (Appendix 5.4, Slide #18).

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CHAPTER 6

CONCLUSIONS

The recovery of the American black bear (*Ursus americanus*; hereafter black bear) in the Mid-Atlantic United States is due in large part to the stewardship of state wildlife management agencies like the New Jersey Division of Fish and Wildlife (NJDFW). However, *adaptive* wildlife management necessitates meeting the new challenges presented by the Anthropocene (Corlett 2015); for instance, the most pressing threat to black bear conservation in northwestern New Jersey (NJ), USA, is now escalating levels of human- black bear conflict (hereafter conflict; Carr and Burguess 2011). Accelerating anthropogenic landscape transformation, coupled with increasing and expanding bear populations, have resulted in conflict levels that are both culturally and fiscally unsustainable (Wolgast et al. 2010). State wildlife agencies in this region have the formidable task of balancing bear population viability with the protection of the welfare and property of an increasingly polarized public (Hristienko and McDonald 2007), all while grappling with diminishing budgets. In NJ, protected areas alone are insufficient to ensure viable black bear populations, thus, wildlife managers must employ strategies that promote human-bear coexistence outside of public lands across the wildland-urban landscape gradient (Carter and Linnell 2016).

One such management tool, regulated sport harvest, has long been intertwined with the North American Model of Wildlife Conservation (Geist et al. 2001), a framework some argue is too overly-simplistic to capture the complexities of modern wildlife management (Peterson and Nelson 2017). Further, new hunter recruitment and retention has been declining across the United States since the early 1980s (Larson et al. 2014). The reintroduction of a recreational black bear harvest in NJ in 2003 initiated an acrimonious debate among stakeholder groups with complex attitudes, more nuanced than simple pro- versus anti-hunting perspectives (Johnson and Sciascia 2013). While the majority of stakeholders are uncommitted with respect to their views on hunting, they expect wildlife managers to communicate sound, balanced, empirically-derived information such that the public can derive their own conclusions (Campbell and Mackay 2009). Informed by the remarkable, 33-year NJ black bear data set, we sought to provide just that to NJDFW. Herein, I have attempted to integrate black bear behavior, spatial ecology, and population dynamics within this human-dominated landscape, to assess the influence of harvest in mitigating conflicts.

In Chapter 2, we established that much of the inter-annual variation in the >12,000 nuisance black bear behaviors reported to NJDFW between 2001 and 2013, is explained by variation in harvest plus lethal management rates the preceding year ($P = 0.028$, $R^2 = 0.338$). Following the reintroduction of a six-day black bear hunt in 2003, 2005, and 2010-12, reported nuisance complaints decreased by an average of 27.7% ± 7.4% the following year (Appendix 2.8). This consistent decline in conflicts is likely driven, in part, by the disproportionate harvest probability of adult bears previously designated as a nuisance and/or threat (i.e., “problem bears,” 0.176 ± 0.025) compared with those never identified as a problem (i.e., “normal bears,” 0.109 ± 0.010). Further, problem bears are between 2 and 8 times more likely to be lethally controlled than normal bears (Table 2-2).

In Chapter 3, we demonstrated that the habitats with the greatest harvest pressure, per unit area, are a 600 m “doughnut” surrounding developed areas (i.e., the wildland-urban interface; Table 3-1) and wildlands. Problem bears were significantly more likely to be recaptured in this risky interface area, as well as urban habitats, which likely serve as refugia. This finding has meaningful conservation implications, as it suggests the importance of preventing the *initial* food-conditioning that occurs in these anthropogenic habitats and induces rapid behavioral and ecological repercussions (Beckmann and Berger 2003). Once it has transpired, this food-conditioning alters the landscape-level habitat use of problem bears (Table 3-3; Fig. 3-2), as they were consistently more likely to be reencountered in anthropogenic areas after being trapped in response to a nuisance complaint.

In Chapter 4, we confirmed that between 2000 and 2013, both the anthropogenic and wildland components of the NJ black bear population rapidly grew during harvest moratoria ($\lambda_{A\ No\ H} = 1.197$; $\lambda_{W\ No\ H} = 1.264$, respectively). Reported conflicts were highly correlated ($P < 0.001$; $R^2 = 0.698$) with projections of anthropogenic female bear abundance (Fig. 4-5). However, during harvest years, the wildland ($\lambda_{W\ Harv} = 1.011$) component stabilized and the anthropogenic component decreased dramatically ($\lambda_{A\ Harv} = 0.697$). The sensitivity of the black bear population growth rate to reductions in female survival is unsurprising given that adult female survival is consistently identified as the vital rate with the highest elasticity (Freedman et al. 2003, Hebblewhite et al. 2003, Mitchell et al. 2009). Ultimately, the NJ black bear harvest represents a cost-effective tool to meet population objectives given that adult female black bears were twice as likely to be harvested as adult males between 2000 and 2013 (0.163 ± 0.014 vs. 0.087 ± 0.012; Table 2-2).

In the human-dominated landscape of northwestern NJ, a carefully regulated black bear harvest represents a valuable component of an integrated management strategy that includes coordinated incident-response protocols among wildlife and law enforcement agencies, and a substantial investment in educational outreach aimed at reducing anthropogenic attractants. The viability of this recovered black bear population now hinges on increasing the cultural carrying capacity of those that live, work and recreate alongside black bears by reducing negative human-black bear interactions. My hope is that this research bolsters understanding regarding the importance of an adaptive black bear harvest in: 1) reducing reported human- black bear conflicts; 2) disproportionately removing bears previously trapped in response to nuisance complaints; 3) disproportionately removing bears from the wildland-urban interface; 4) driving rapid population declines in the anthropogenic component of the bear population responsible for the majority of conflicts; and 5) stabilizing the wildland habitat component of the black bear population in northwestern NJ.

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APPENDICES

Appendix 1. New Jersey Division of Fish and Wildlife Black Bear Capture and Handling protocol. Additional definitions of terms associated with multistate capture-reencounter modeling framework.

New Jersey Division of Fish and Wildlife personnel began capturing, marking and recovering bears in April 1981; these analyses incorporate research and incident-response management trapping, den surveys, and mortality recoveries from 1981-2014. Current capture protocols include using barrel-style, culvert-style, or Aldrich wrist-snare trap sets, or via free-range darting. Anesthetic is administered using a New Dart hand projector or Dan-Inject dart rifle (Wildlife Pharmaceuticals, Inc., Fort Collins, CO, USA) and consists of ketamine hydrochloride (4.4 mg/kg) and xylazine hydrochloride (1.7 mg/kg) based on estimated body weight. Subjects are tagged in both ears using self-piercing, numbered, metal tags, style 56-L, size 36.5 x 9.5 mm (Hasco Tag Co., Dayton, KY, USA), and tattooed on the inside of the lip with the ear tag identification number. With the exception of cubs of the year, a premolar is extracted for age determination. Additionally, date, time, capture location, sex, weight, morphometric data, and reproductive status (estrous, lactating, descended testes) are recorded.

In our multistate capture-reencounter models intial ‘capture’ occurred via den surveys, research, or management trapping. ‘Reencounters’ consisted of physical recaptures either via research or incident-response management trapping ($p_{i,t}^{\mathrm{A}}$) or dead recoveries ($r_{i,t}^{\mathrm{D}}$).

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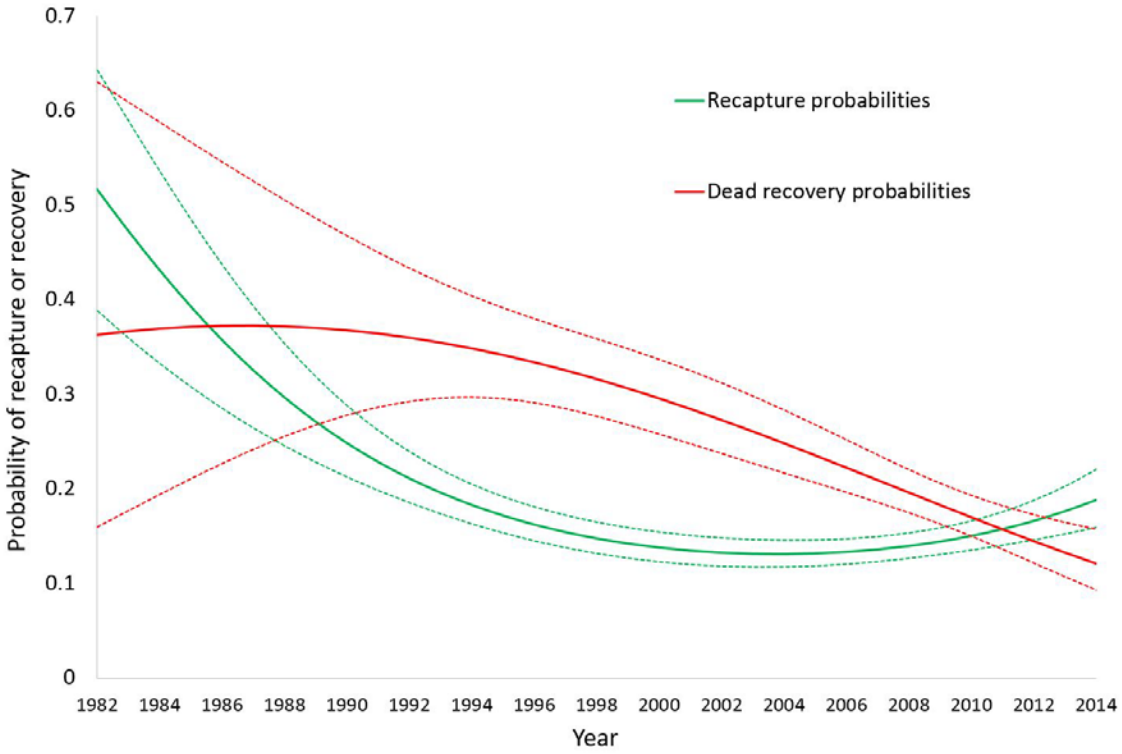
Appendix 2.2. Black Bear Recapture and Recovery Probabilities. Black bear recapture ($p_{i,t}^{\mathrm{A}}$) and recovery ($r_{i,t}^{\mathrm{D}}$) probabilities as a function of time for bears that died from all sources of mortality excluding harvest, and management mortality from 1982 – 2014 in northwestern New Jersey, USA. Dashed lines delineate the 95% CI associated with the estimates, color-coded accordingly.



Appendix 2.3. Recapture Model Selection as a Function of Time. Exploring $p_{i,t}^{\mathrm{A}}$ and $r_{i,t}^{\mathrm{D}}$ as a function of time. Time parameters represented below as follows: (Time + I(Time^2)) = quadratic time function; (Time + I(Time^2) + I(Time^3)) = cubic time function; tbin_dec = decadal time bins; tbin_2001 = two time bins pre- and post-2001. Top models denoted in **bold**.

Candidate Models	**Δ AIC$_c$**	**NP**	w_i	**Deviance**
strA * (Time + I(Time^2)) + strH + strM + strD	**0.000**	**7**	**0.374**	**460417.1**
strA + strH + strM + strD * (Time + I(Time^2))	**0.000**	**7**	**0.374**	**460417.1**
strA * (Time + I(Time^2) + I(Time^3)) + strH + strM + strD	2.672	9	0.098	460415.7
strA + strH + strM + strD * (Time + I(Time^2) + I(Time^3))	2.672	9	0.098	460415.7
strA * (Time + I(Time^2)) + strH + strM + strD * (Time + I(Time^2))	4.012	9	0.050	460417.1
strA * (Time + I(Time^2) + I(Time^3)) + strH + strM + strD * (Time + I(Time^2) + I(Time^3))	8.696	12	0.005	460415.7
strA:tbin_dec + strH + strM + strD:tbin_dec	15.650	7	0.000	460432.7
strA:tbin_dec + strH + strM + strD	36.931	5	0.000	460458.0
strA + strH + strM + strD:tbin_dec	38.241	5	0.000	460459.3
strA + strH + strM + strD:tbin_2001	41.627	4	0.000	460464.7
strA:tbin_2001 + strH + strM + strD:tbin_2001	41.841	5	0.000	460462.9
strA + strH + strM + strD	59.564	3	0.000	460484.7
strA:tbin_2001 + strH + strM + strD	60.147	4	0.000	460483.2

Appendix 2.4. Recapture Model Selection as a Function of Covariates. Exploring $p_{i,t}^{\mathrm{A}}$ and $r_{i,t}^{\mathrm{D}}$ as a function of covariates sex, age, and behavior. Covariates represented below as follows: f = female, m = male; cy = captured as cubs, yy = captured as yearlings, ay = captured as adults; b = bear classified as normal behavior, p = bear classified as problem behavior. Top models denoted in **bold**.

Candidate Models	**Δ AIC$_c$**	**NP**	w_i	**Deviance**
strA:f + strA:m + strH + strM + strD	**0.000**	**4**	**0.710**	**471546.5**
strA:f + strA:m + strH + strM + strD:f + strD:m	**1.794**	**5**	**0.290**	**471546.3**
strA:cy + strA:yy + strA:ay + strH + strM + strD:cy + strD:yy + strD:ay	22.733	7	0.000	460318.7
strA:cy + strA:yy + strA:ay + strH + strM + strD	36.864	5	0.000	460336.8
strA + strH + strM + strD:cy + strD:yy + strD:ay	47.754	5	0.000	460347.7
strA + strH + strM + strD	59.877	3	0.000	460363.9
strA + strH + strM + strD:f + strD:m	61.830	4	0.000	471608.4
strA + strH + strM + strD:b + strD:p	158.920	4	0.000	471705.5
strA:b + strA:p + strH + strM + strD	208.250	4	0.000	471754.8
strA:b + strA:p + strH + strM + strD:b + strD:p	316.664	5	0.000	471861.2

Appendix 2.5. Final Recapture Candidate Model Set. Modeling $p_{i,t}^{\mathrm{A}}$ and $r_{i,t}^{\mathrm{D}}$ as a function of the best-fitting time parameterization (quadratic function) and covariate (sex). Time parameters and covariates represented as described above. Top models denoted in **bold**.

Candidate Models	**Δ AIC$_c$**	**NP**	**w_i**	**Deviance**
strA:f + strA:m + strH + strM + strD * (Time + I(Time^2))	**0.000**	**8**	**1.00**	**471460.0**
strA * (Time + I(Time^2)) + strH + strM + strD	73.164	7	0.00	460290.5
strA + strH + strM + strD * (Time + I(Time^2))	73.164	7	0.00	460290.5
strA * (Time + I(Time^2)) + strH + strM + strD:f + strD:m	75.060	8	0.00	471535.0
strA * (Time + I(Time^2)) + strH + strM + strD * (Time + I(Time^2))	77.177	9	0.00	460290.5
strA:f + strA:m + strH + strM + strD	78.571	4	0.00	471546.5
strA:f + strA:m + strH + strM + strD:f + strD:m	80.365	5	0.00	471546.3
strA + strH + strM + strD	138.448	3	0.00	460363.9
strA + strH + strM + strD:f + strD:m)	140.401	4	0.00	471608.4

Appendix 2.6. Transitions Model Selection as a Function of Time. Exploring $\mu_{i,t}^{\mathrm{H}}$, $\mu_{i,t}^{\mathrm{M}}$, and $\mu_{i,t}^{\mathrm{D}}$ as a function of time. Time parameters represented below as follows: (Time + I(Time^2)) = quadratic time function; (Time + I(Time^2) + I(Time^3)) = cubic time function; tbin_dec = decadal time bins; tbin_2001 = two time bins pre- and post-2001. . Top models denoted in **bold**.

Candidate Models	**Δ AIC$_c$**	**NP**	**w_i**	**Deviance**
toH + toM * (Time + I(Time^2) + I(Time^3)) + toD	**0.000**	**16**	**0.257**	**470671.6**
toH * (Time + I(Time^2) + I(Time^3)) + toM * (Time + I(Time^2) + I(Time^3))	**1.339**	**19**	**0.132**	**470666.9**
toH * (Time + I(Time^2) + I(Time^3)) + toM + toD * (Time + I(Time^2) + I(Time^3))	**1.349**	**19**	**0.131**	**470666.9**
toH + toM * (Time + I(Time^2) + I(Time^3)) + toD * (Time + I(Time^2) + I(Time^3))	**1.349**	**19**	**0.131**	**470666.9**
toH + toM * (Time + I(Time^2)) + toD	**1.578**	**14**	**0.117**	**470677.2**
toH * (Time + I(Time^2)) + toM + toD * (Time + I(Time^2))	2.700	16	0.067	470674.3
toH * (Time + I(Time^2)) + toM * (Time + I(Time^2)) + toD	2.700	16	0.067	470674.3
toH + toM * (Time + I(Time^2)) + toD * (Time + I(Time^2))	2.700	16	0.067	470674.3
toH + toM + toD * (Time + I(Time^2))	6.488	14	0.010	470682.1
toH * (Time + I(Time^2)) + toM * (Time + I(Time^2)) + toD * (Time + I(Time^2))	6.725	18	0.009	470674.3
toH + toM + toD * (Time + I(Time^2) + I(Time^3))	7.120	16	0.007	470678.7
toH * (Time + I(Time^2) + I(Time^3)) + toM * (Time + I(Time^2) + I(Time^3)) + toD * (Time + I(Time^2) + I(Time^3))	7.394	22	0.006	470666.9
toH + toM:tbin_2001 + toD	22.840	11	0.000	470704.5
toH * (Time + I(Time^2) + I(Time^3)) + toM + toD	22.860	16	0.000	470694.4
toH:tbin_2001 + toM:tbin_2001 + toD:tbin_2001	23.859	12	0.000	470703.5
toH + toM:tbin_dec + toD	24.239	12	0.000	470703.8
toH + toM:tbin_dec + toD:tbin_dec	25.678	14	0.000	470701.3

toH + toM + toD:tbin_2001	27.650	11	0.000	470709.3

Appendix 2.6 cont.

toH + toM + toD:tbin_dec	27.659	12	0.000	470707.3
toH:tbin_2001 + toM + toD	28.672	10	0.000	470712.3
toH + toM + toD	28.672	10	0.000	470712.3
toH * (Time + I(Time^2)) + toM + toD	29.778	14	0.000	470705.4

Appendix 2.7. Transitions Model Selection as a Function of Covariates. Exploring $\mu_{i,t}^{\mathrm{H}}$, $\mu_{i,t}^{\mathrm{M}}$, and $\mu_{i,t}^{\mathrm{D}}$ as a function of covariates sex, age, and behavior. Covariates represented below as follows: f = female, m = male; c= captured as cubs, y = captured as yearlings, a = captured as adults; b = bear classified as normal behavior, p = bear classified as problem behavior.

Candidate Models	**Δ AIC$_c$**	**NP**	***w_i***	**Deviance**
toH:c + toH:y + toH:a + toM:c + toM:y + toM:a + toD:c + toD:y + toD:a	**0.000**	**16**	**1.000**	**470115.0**
toH + toM:c + toM:y + toM:a + toD:c + toD:y + toD:a	44.037	14	0.000	470163.0
toH:c + toH:y + toH:a + toM + toD:c + toD:y + toD:a	66.537	14	0.000	470185.5
toH + toM + toD:c + toD:y + toD:a	94.948	12	0.000	470218.0
toH + toM + toD:b + toD:p	147.859	11	0.000	470272.9
toH + toM:f + toM:m + toD:f + toD:m	177.148	12	0.000	470300.2
toH:f + toH:m + toM:f + toM:m + toD:f + toD:m	177.787	13	0.000	470298.8
toH + toM + toD:f + toD:m	240.829	11	0.000	470365.9
toH:f + toH:m + toM + toD:f + toD:m	242.588	12	0.000	470365.6
toH:b + toH:p + toM:b + toM:p + toD:b + toD:p	245.127	13	0.000	470366.1
toH + toM:b + toM:p + toD:b + toD:p	257.908	12	0.000	470380.9
toH:f + toH:m + toM:f + toM:m + toD	295.688	12	0.000	470418.7
toH + toM:f + toM:m + toD	296.329	11	0.000	470421.4
toH:c + toH:y + toH:a + toM:c + toM:y + toM:a + toD	299.567	14	0.000	470418.6
toH + toM:c + toM:y + toM:a + toD	309.658	12	0.000	470432.7
toH:c + toH:y + toH:a + toM + toD	311.758	12	0.000	470434.8
toH:f + toH:m + toM + toD	315.549	11	0.000	470440.6

Appendix 2.7 cont.

toH + toM + toD	318.771	10	0.000	470445.8
toH:b + toH:p + toM + toD	326.249	11	0.000	470451.3
toH:b + toH:p + toM:b + toM:p + toD	399.718	12	0.000	470522.7
toH + toM:b + toM:p + toD	400.349	11	0.000	470525.4

Appendix 2.8. Annual Reported Black Bear Incidents. In November 2000, the New Jersey Division of Fish and Wildlife (NJDFW) implemented the Black Bear Rating and Response Criteria (BBRRC), an effective and standardized framework for responding to bears deemed a threat to human safety, agricultural crops and/or property, or exhibiting nuisance behavior. Between 2001 – 2013, NJDFW received 26,582 reports of human-bear interactions from the general public, and categorized those according to the BBRRC as: I) Threat = 2277; II) Nuisance = 12,013; and III) Normal = 12,292. We present the subcategories that comprise these totals below, and calculate the percent change from Dec 1 – Nov 30 of the subsequent year following the introduction of harvest or harvest moratoria.

Behavior Category Type	**Pre-harvest**	**Following Harvest I (Dec 2003)**		**Following No harvest**		**Following Harvest II (Dec 2005)**		**Following No harvest**		**Pre-harvest**	**Following Harvests III-V (Dec 2010-2012)**	
	Mean ARI 2001-03	ARI 2003-2004	**% change**	ARI 2004-2005	**% change**	ARI 2005-2006	**% change**	ARI 2006-2007	**% change**	Mean ARI 2007-10	Mean ARI 2010-13	**% change**
Category I – Threat	214	103	**-52**	140	**+36**	133	**-5**	110	**-17**	256	149	**-42**
Home entries (attempted & successful)	72	32	**-55**	45	**+41**	58	**+29**	47	**-19**	87	49	**-44**
Property damage >$500 (including vehicle entries)	21	5	**-76**	11	**+120**	12	**+9**	9	**-25**	16	6	**-65**
Aggressive bears (including tent entries)	26	9	**-65**	25	**+178**	14	**-44**	11	**-21**	29	7	**-75**
Agricultural damage, livestock attacks/kills	96	57	**-40**	59	**+4**	49	**-17**	43	**-12**	123	87	**-29**
Category II – Nuisance	825	523	**-37**	719	**+37**	623	**-13**	703	**+13**	1451	967	**-33**
Garbage visits	374	270	**-28**	316	**+17**	288	**-9**	314	**+9**	542	417	**-23**
General Nuisance (including campsite visits)	358	211	**-41**	336	**+59**	282	**-16**	322	**+14**	771	441	**-43**
Property damage <$500	94	42	**-55**	67	**+60**	53	**-21**	67	**+26**	137	109	**-20**
Problem Behaviors (Categories I and II Combined)	1039	626	**-40**	859	**+37**	756	**-12**	813	**+8**	1706	1116	**-35**

Appendix 2.9. Annotated Program R Code for Chapter 2

Program R scripts composed in version 3.1.2.

```r
# These scripts model:
# 1) recapture probability (p) in state A and recovery probability (rD) in state D with respect to time, and covariates
# 2) transition probabilities (psi) from state A to H, M, and D with respect to time, covariates, and covariate interactions
# All psi models apply best p rD model: p_sex_rD_quad
# S = survival probability; p = recapture probability in A state; rD = recovery probability in D state; Psi = transition probability
# A = alive; H = harvested; M = management mortality; D = dead other
# Default link functions
# Recovery probability of management mortality (rM) fixed to 1
# Recovery probability of harvest mortality (rH) fixed to 1 in harvest yrs; 0 in non-harvest yrs
# load the RMark package
library(RMark)
# Read in capture history text file
MS_harvest <- read.delim(". . .txt")
num_MS_harvest <-dim(MS_harvest)[1]
# For loop to identify ch as character vector
for(i in 1:num_MS_harvest){
  MS_harvest[i,3] <- as.character(MS_harvest[i,2])
}
is.character(MS_harvest[i,3])
##########################################################################################
# An initial age is defined as 2 for adult, 0 for cub, and 1 for yearling.
# They are assigned in that order because they are assigned in alphabetical order of the group variable. Here, it does not matter that adults         # could be a mixture of ages because we will only model cub year (0), yearling year (1), and  adult (2+) effects.
MS_harvest_group_class.process <- process.data(MS_harvest, model = "Multistrata", begin.time = 1981,
              group=c("class_first_cap"), age.var=1, initial.age=c(2,0,1))
MS_harvest_group_class.ddl <- make.design.data(MS_harvest_group_class.process)
```

```
######################################################################################################
# Create time bins for p/rD and Psi by adding design data
# See pg 6 cran.r RMark .pdf
# Add a field for 3 time bins for p and r from 1981-89; 1990-2000; 2001-2014
MS_harvest_group_class.ddl=add.design.data(MS_harvest_group_class.process,
        MS_harvest_group_class.ddl, parameter="p", type="time",
        bins=c(1981,1989,2000,2014),name="tbin_dec", replace=TRUE)
# Add a field for 2 time bins for p and r pre and post 2001
MS_harvest_group_class.ddl=add.design.data(MS_harvest_group_class.process,
        MS_harvest_group_class.ddl, parameter="p", type="time",
        bins=c(1981,2000,2014),name="tbin_2001", replace=TRUE)
# We applied the same parameterization format to Psi . . .
######################################################################################################
# Creating 4 variables associated with State for estimating p and rD
MS_harvest_group_class.ddl$p$strA=0
MS_harvest_group_class.ddl$p$strA[MS_harvest_group_class.ddl$p$stratum=="A"]=1
MS_harvest_group_class.ddl$p$strH=0
MS_harvest_group_class.ddl$p$strH[MS_harvest_group_class.ddl$p$stratum=="H"]=1
# We applied the same parameterization format to strM and strD . . .
######################################################################################################
# Add dummy variables for operating on specific states or transitions
# Create variable (toH, toM, toD) which allows us to eliminate possible transition to H, M, D
MS_harvest_group_class.ddl$Psi$toH=0
MS_harvest_group_class.ddl$Psi$toH[MS_harvest_group_class.ddl$Psi$stratum=="A"&
        MS_harvest_group_class.ddl$Psi$tostratum=="H"]=1
# We applied the same parameterization format to toM and toD . . .
######################################################################################################
# Create variable (fromH, fromM, fromD) which allows us to eliminate possible transition out of H, M, D
state
MS_harvest_group_class.ddl$Psi$fromH=0
MS_harvest_group_class.ddl$Psi$fromH[MS_harvest_group_class.ddl$Psi$stratum=="H"]=1
# We applied the same parameterization format to fromM and fromD . . .
######################################################################################################
```

```
# Add dynamic dummy variable age class fields to the design data for p/rD

# cy=recapture for yrlngs captured first as cubs;

# yy=recapture for adults captured first as yrlngs;

# ay=recapture for adults captured first as adults

MS_harvest_group_class.ddl$p$cy=0

MS_harvest_group_class.ddl$p$yy=0

MS_harvest_group_class.ddl$p$ay=0

MS_harvest_group_class.ddl$p$cy[MS_harvest_group_class.ddl$p$age==1]=1

MS_harvest_group_class.ddl$p$yy[MS_harvest_group_class.ddl$p$age==2]=1

MS_harvest_group_class.ddl$p$ay[MS_harvest_group_class.ddl$p$Age>=3]=1

# Add dynamic dummy variable age class fields to the design data for Psi

# c=bears aged 0-1 years

# y=bears aged 1-2 years

# a=bears aged 2+ years

MS_harvest_group_class.ddl$Psi$c=0

MS_harvest_group_class.ddl$Psi$y=0

MS_harvest_group_class.ddl$Psi$a=0

MS_harvest_group_class.ddl$Psi$c[MS_harvest_group_class.ddl$Psi$age==0 &
MS_harvest_group_class.ddl$Psi$stratum=="A"]=1

MS_harvest_group_class.ddl$Psi$y[MS_harvest_group_class.ddl$Psi$age==1 &
MS_harvest_group_class.ddl$Psi$stratum=="A"]=1

MS_harvest_group_class.ddl$Psi$a[MS_harvest_group_class.ddl$Psi$Age>=2 &
MS_harvest_group_class.ddl$Psi$stratum=="A"]=1

#####################################################################################################
# Add individual covariates for Psi and p/rD

# Code sex, where f=female, m=male

MS_harvest_group_class.ddl$p$f=0

MS_harvest_group_class.ddl$p$f[MS_harvest$female==1]=1

MS_harvest_group_class.ddl$p$m=0

MS_harvest_group_class.ddl$p$m[MS_harvest$male==1]=1

# We applied the same parameterization format to Psi . . .

#####################################################################################################

# Code behavior for p_rD_behav . . .
```

```
# b = normal bear beahvior; tn = threat or nuisance; n = nuisance; t = threat

MS_harvest_group_class.ddl$p$b=0

MS_harvest_group_class.ddl$p$b[MS_harvest$threat_or_nuis==0]=1

MS_harvest_group_class.ddl$p$tn=0

MS_harvest_group_class.ddl$p$tn[MS_harvest$threat_or_nuis==1]=1

# We applied the same parameterization format to Psi . . .

###########################################################################################
# Define model structures for S

# Calculating number of A, H, M, Ds

SA=as.numeric(row.names(MS_harvest_group_class.ddl$S[MS_harvest_group_class.ddl$S$stratum=="A",]))

# We applied the same parameterization format to SH, SM, SD . . .

# Create vector of that length populated by ones or zeros

SAval=rep(1,length(SA))

SHval=rep(0,length(SH))

# We applied the same parameterization format to SM, SD . . .

# Fix S to 1 for A and 0 for dead states H,M,D - modeling S is NOT important here because we are interested in transition probabilities

S_fix <- list(formula=~stratum,fixed=list(index=c(SA,SH,SM,SD),value=c(SAval,SHval,SMval,SDval)))

#############################################################################################
# Fixing values for r

# Setting rH = probability of dead recovery from HARVEST, to 0 for stratum H in non-harvest yrs

# Note: there were harvests in 2003, 2005, 2010, 2011, 2012, 2013 AND 2014; however we do NOT have 2014 harvest data so fixed r to 0 for 2014 too

rH1982=as.numeric(row.names(MS_harvest_group_class.ddl$p[MS_harvest_group_class.ddl$p$time==1982 &

		MS_harvest_group_class.ddl$p$stratum=="H",]))

rH1982val=rep(0,length(rH1982))

rH1983=as.numeric(row.names(MS_harvest_group_class.ddl$p[MS_harvest_group_class.ddl$p$time==1983 &

		MS_harvest_group_class.ddl$p$stratum=="H",]))

rH1983val=rep(0,length(rH1983))

# We applied the same parameterization format to rH 1984-2009 (with the exception of 2003, 2005, see below) . . .
```

```
# Fixing rH to 1 in harvest years, because hunters must report all harvests to check stations; previous analysis estimating rH as 0.9999 . . .

rH2003=as.numeric(row.names(MS_harvest_group_class.ddl$p[MS_harvest_group_class.ddl$p$time==2003 &

            MS_harvest_group_class.ddl$p$stratum=="H",]))

rH2003val=rep(1,length(rH2003))

rH2005=as.numeric(row.names(MS_harvest_group_class.ddl$p[MS_harvest_group_class.ddl$p$time==2005 &

            MS_harvest_group_class.ddl$p$stratum=="H",]))

rH2005val=rep(1,length(rH2005))

# We applied the same parameterization format to rH 2010-2013 . . .

# Fixing rM to 1 in ALL years, b/c if managers and farmers must report all management mortalities

rM1982=as.numeric(row.names(MS_harvest_group_class.ddl$p[MS_harvest_group_class.ddl$p$time==1982 &

            MS_harvest_group_class.ddl$p$stratum=="M",]))

rM1982val=rep(1,length(rM1982))

rM1983=as.numeric(row.names(MS_harvest_group_class.ddl$p[MS_harvest_group_class.ddl$p$time==1983 &

            MS_harvest_group_class.ddl$p$stratum=="M",]))

rM1983val=rep(1,length(rM1983))

# We applied the same parameterization format to rM 1984-2014 . . .

# Fix rH, where probability of Harvest recovery 0 in non-harvest yrs, 1 in harvest yrs; rM, probability of management recovery 1 in all yrs.

rHM_fix <- list(formula=~time,
fixed=list(index=c(rH1982,rH1983,rH1984,rH1985,rH1986,rH1987,rH1988,rH1989,rH1990,rH1991,rH1992,rH1993,rH1994,rH1995,rH1996,
rH1997,rH1998,rH1999,rH2000,rH2001,rH2002,rH2003,rH2004,rH2005,rH2006,rH2007,rH2008,rH2009,
rH2010,rH2011,rH2012,rH2013,rH2014,
rM1982,rM1983,rM1984,rM1985,rM1986,rM1987,rM1988,rM1989,rM1990,rM1991,rM1992,rM1993,rM1994,rM1995,rM1996,rM1997,
rM1998,rM1999,rM2000,rM2001,rM2002,rM2003,rM2004,rM2005,rM2006,rM2007,rM2008,rM2009,rM2010,rM2011,rM2012,rM2013, rM2014),
value=c(rH1982val,rH1983val,rH1984val,rH1985val,rH1986val,rH1987val,rH1988val,rH1989val,rH1990val,rH1991val,rH1992val,rH1993val,
rH1994val,rH1995val,rH1996val,rH1997val,rH1998val,rH1999val,rH2000val,rH2001val,rH2002val,rH2003val,rH2004val,rH2005val,rH2006val, rH2007val,rH2008val,rH2009,rH2010val, rH2011val, rH2012val,
rH2013val, rH2014val,
rM1982val,rM1983val,rM1984val,rM1985val,rM1986val,rM1987val,rM1988val,rM1989val,rM1990val,rM1991val,rM1992val,rM1993val,
rM1994val,rM1995val,rM1996val,rM1997val,rM1998val,rM1999val,rM2000val,rM2001val,rM2002val,rM2003val,rM2004val,rM2005val, rM2006val,rM2007val,rM2008val,rM2009val,rM2010val, rM2011val,
rM2012val, rM2013val, rM2014val)))
```

```
##########################################################################################
# Fixing values for Psi

# These variables prevent transitions from H,M and D; i.e., once individual is dead, remains dead

PsiH=as.numeric(row.names(MS_harvest_group_class.ddl$Psi[MS_harvest_group_class.ddl$Psi$stratum=
="H",]))

# We applied the same parameterization format to PsiM and PsiD . . .

PsiHval=rep(0,length(PsiH))

# We applied the same parameterization format to PsiMval and PsiDval . . .

# Create variables that eliminate the possibility of transitioning into H in non-harvest years

Psi1981=as.numeric(row.names(MS_harvest_group_class.ddl$Psi[MS_harvest_group_class.ddl$Psi$time=
=1981 &

            MS_harvest_group_class.ddl$Psi$stratum=="A" &
MS_harvest_group_class.ddl$Psi$tostratum=="H",]))

Psi1982=as.numeric(row.names(MS_harvest_group_class.ddl$Psi[MS_harvest_group_class.ddl$Psi$time=
=1982 &

            MS_harvest_group_class.ddl$Psi$stratum=="A" &
MS_harvest_group_class.ddl$Psi$tostratum=="H",]))

# We applied the same parameterization format to Psi1983 – Psi2014, with the exception of harvest years
(2003, 2005, 2010-2013)  . . .

# Fixing Psi from H, M and D to 0; and Psi to H to 0 in non-harvest years

Psi_fix <- list(formula=~stratum, fixed=list(index=c(PsiH,PsiM,PsiD,Psi1981, Psi1982, Psi1983, Psi1984,
Psi1985, Psi1986, Psi1987, Psi1988, Psi1989, Psi1990, Psi1991, Psi1992, Psi1993, Psi1994, Psi1995,
Psi1996,Psi1997, Psi1998, Psi1999, Psi2000, Psi2001, Psi2002, Psi2004, Psi2006, Psi2007, Psi2008,
Psi2009, Psi2014), value=0))

##########################################################################################
# Defining model structures and running models for p and rD

# Stratum

stratum <- list(formula=~-1 + strA + strH + strM + strD,
fixed=list(index=c(rH1982,rH1983,rH1984,rH1985,rH1986,rH1987,rH1988,rH1989,rH1990,rH1991,rH19
92,rH1993,rH1994,rH1995,rH1996,
rH1997,rH1998,rH1999,rH2000,rH2001,rH2002,rH2003,rH2004,rH2005,rH2006,rH2007,rH2008,rH2009,
rH2010,rH2011,rH2012,rH2013,rH2014,
rM1982,rM1983,rM1984,rM1985,rM1986,rM1987,rM1988,rM1989,rM1990,rM1991,rM1992,rM1993,rM
1994,rM1995,rM1996,rM1997,
rM1998,rM1999,rM2000,rM2001,rM2002,rM2003,rM2004,rM2005,rM2006,rM2007,rM2008,rM2009,rM
2010,rM2011,rM2012,rM2013, rM2014),
value=c(rH1982val,rH1983val,rH1984val,rH1985val,rH1986val,rH1987val,rH1988val,rH1989val,rH1990
val,rH1991val,rH1992val,rH1993val,
rH1994val,rH1995val,rH1996val,rH1997val,rH1998val,rH1999val,rH2000val,rH2001val,rH2002val,rH20
03val,rH2004val,rH2005val,rH2006val, rH2007val,rH2008val,rH2009val,rH2010val, rH2011val,
rH2012val, rH2013val, rH2014,
rM1982val,rM1983val,rM1984val,rM1985val,rM1986val,rM1987val,rM1988val,rM1989val,rM1990val,r
```

```
M1991val,rM1992val,rM1993val,
rM1994val,rM1995val,rM1996val,rM1997val,rM1998val,rM1999val,rM2000val,rM2001val,rM2002val,r
M2003val,rM2004val,rM2005val, rM2006val,rM2007val,rM2008val,rM2009val,rM2010val, rM2011val,
rM2012val, rM2013val, rM2014val)))

S_fix.stratum.Psi_fix <-
mark(MS_harvest_group_class.process,MS_harvest_group_class.ddl,model.parameters=list(S=S_fix,
p=stratum, Psi=Psi_fix))

#

# Example of one model including Quadratic time function for p and rD

p_rD_quad <- list(formula=~-1 + strA*(Time+I(Time^2)) + strH + strM + strD*(Time+I(Time^2)),

fixed=list(index=c(rH1982,rH1983,rH1984,rH1985,rH1986,rH1987,rH1988,rH1989,rH1990,rH1991,rH19
92,rH1993,rH1994,rH1995,rH1996,
rH1997,rH1998,rH1999,rH2000,rH2001,rH2002,rH2003,rH2004,rH2005,rH2006,rH2007,rH2008,rH2009,
rH2010,rH2011,rH2012,rH2013,rH2014,
rM1982,rM1983,rM1984,rM1985,rM1986,rM1987,rM1988,rM1989,rM1990,rM1991,rM1992,rM1993,rM
1994,rM1995,rM1996,rM1997,
rM1998,rM1999,rM2000,rM2001,rM2002,rM2003,rM2004,rM2005,rM2006,rM2007,rM2008,rM2009,rM
2010,rM2011,rM2012,rM2013, rM2014),
value=c(rH1982val,rH1983val,rH1984val,rH1985val,rH1986val,rH1987val,rH1988val,rH1989val,rH1990
val,rH1991val,rH1992val,rH1993val,
rH1994val,rH1995val,rH1996val,rH1997val,rH1998val,rH1999val,rH2000val,rH2001val,rH2002val,rH20
03val,rH2004val,rH2005val,rH2006val, rH2007val,rH2008val,rH2009val,rH2010val, rH2011val,
rH2012val, rH2013val, rH2014,
rM1982val,rM1983val,rM1984val,rM1985val,rM1986val,rM1987val,rM1988val,rM1989val,rM1990val,r
M1991val,rM1992val,rM1993val,
rM1994val,rM1995val,rM1996val,rM1997val,rM1998val,rM1999val,rM2000val,rM2001val,rM2002val,r
M2003val,rM2004val,rM2005val, rM2006val,rM2007val,rM2008val,rM2009val,rM2010val, rM2011val,
rM2012val, rM2013val, rM2014val)))

S_fix.p_rD_quad.Psi_fix <-
mark(MS_harvest_group_class.process,MS_harvest_group_class.ddl,model.parameters=list(S=S_fix,
p=p_rD_quad, Psi=Psi_fix))

#
# Example of one model including Cubic time function for p and rD

p_rD_cube <- list(formula=~-1 + strA*(Time+I(Time^2)+I(Time^3)) + strH + strM +
strD*(Time+I(Time^2)+I(Time^3)),

fixed=list(index=c(rH1982,rH1983,rH1984,rH1985,rH1986,rH1987,rH1988,rH1989,rH1990,rH1991,rH19
92,rH1993,rH1994,rH1995,rH1996,
rH1997,rH1998,rH1999,rH2000,rH2001,rH2002,rH2003,rH2004,rH2005,rH2006,rH2007,rH2008,rH2009,
rH2010,rH2011,rH2012,rH2013,rH2014,
rM1982,rM1983,rM1984,rM1985,rM1986,rM1987,rM1988,rM1989,rM1990,rM1991,rM1992,rM1993,rM
1994,rM1995,rM1996,rM1997,
rM1998,rM1999,rM2000,rM2001,rM2002,rM2003,rM2004,rM2005,rM2006,rM2007,rM2008,rM2009,rM
2010,rM2011,rM2012,rM2013, rM2014),
value=c(rH1982val,rH1983val,rH1984val,rH1985val,rH1986val,rH1987val,rH1988val,rH1989val,rH1990
val,rH1991val,rH1992val,rH1993val,
rH1994val,rH1995val,rH1996val,rH1997val,rH1998val,rH1999val,rH2000val,rH2001val,rH2002val,rH20
03val,rH2004val,rH2005val,rH2006val, rH2007val,rH2008val,rH2009val,rH2010val, rH2011val,
```

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rH2012val, rH2013val, rH2014,
rM1982val,rM1983val,rM1984val,rM1985val,rM1986val,rM1987val,rM1988val,rM1989val,rM1990val,r
M1991val,rM1992val,rM1993val,
rM1994val,rM1995val,rM1996val,rM1997val,rM1998val,rM1999val,rM2000val,rM2001val,rM2002val,r
M2003val,rM2004val,rM2005val, rM2006val,rM2007val,rM2008val,rM2009val,rM2010val, rM2011val,
rM2012val, rM2013val, rM2014val)))

S_fix.p_rD_cube.Psi_fix <-
mark(MS_harvest_group_class.process,MS_harvest_group_class.ddl,model.parameters=list(S=S_fix,
p=p_rD_cube, Psi=Psi_fix))

#______________________________________________________________________________
# Example of one model including time bins (by decade) for p and rD

p_r_tbin_dec <- list(formula=~-1 + strA:tbin_dec + strH + strM + strD:tbin_dec,

fixed=list(index=c(rH1982,rH1983,rH1984,rH1985,rH1986,rH1987,rH1988,rH1989,rH1990,rH1991,rH19
92,rH1993,rH1994,rH1995,rH1996,
rH1997,rH1998,rH1999,rH2000,rH2001,rH2002,rH2003,rH2004,rH2005,rH2006,rH2007,rH2008,rH2009,
rH2010,rH2011,rH2012,rH2013,rH2014,
rM1982,rM1983,rM1984,rM1985,rM1986,rM1987,rM1988,rM1989,rM1990,rM1991,rM1992,rM1993,rM
1994,rM1995,rM1996,rM1997,
rM1998,rM1999,rM2000,rM2001,rM2002,rM2003,rM2004,rM2005,rM2006,rM2007,rM2008,rM2009,rM
2010,rM2011,rM2012,rM2013, rM2014),
value=c(rH1982val,rH1983val,rH1984val,rH1985val,rH1986val,rH1987val,rH1988val,rH1989val,rH1990
val,rH1991val,rH1992val,rH1993val,
rH1994val,rH1995val,rH1996val,rH1997val,rH1998val,rH1999val,rH2000val,rH2001val,rH2002val,rH20
03val,rH2004val,rH2005val,rH2006val, rH2007val,rH2008val,rH2009val,rH2010val, rH2011val,
rH2012val, rH2013val, rH2014,
rM1982val,rM1983val,rM1984val,rM1985val,rM1986val,rM1987val,rM1988val,rM1989val,rM1990val,r
M1991val,rM1992val,rM1993val,
rM1994val,rM1995val,rM1996val,rM1997val,rM1998val,rM1999val,rM2000val,rM2001val,rM2002val,r
M2003val,rM2004val,rM2005val, rM2006val,rM2007val,rM2008val,rM2009val,rM2010val, rM2011val,
rM2012val, rM2013val, rM2014val)))

S_fix.p_r_tbin_dec.Psi_fix <-
mark(MS_harvest_group_class.process,MS_harvest_group_class.ddl,model.parameters=list(S=S_fix,
p=p_r_tbin_dec, Psi=Psi_fix))

#______________________________________________________________________________
# Example of one model including covariate (age) for p and rD

p_rD_age <- list(formula=~-1 + strA:cy + strA:yy + strA:ay + strH + strM + strD:cy + strD:yy + strD:ay,

fixed=list(index=c(rH1982,rH1983,rH1984,rH1985,rH1986,rH1987,rH1988,rH1989,rH1990,rH1991,rH19
92,rH1993,rH1994,rH1995,rH1996,
rH1997,rH1998,rH1999,rH2000,rH2001,rH2002,rH2003,rH2004,rH2005,rH2006,rH2007,rH2008,rH2009,
rH2010,rH2011,rH2012,rH2013,rH2014,
rM1982,rM1983,rM1984,rM1985,rM1986,rM1987,rM1988,rM1989,rM1990,rM1991,rM1992,rM1993,rM
1994,rM1995,rM1996,rM1997,
rM1998,rM1999,rM2000,rM2001,rM2002,rM2003,rM2004,rM2005,rM2006,rM2007,rM2008,rM2009,rM
2010,rM2011,rM2012,rM2013, rM2014),
value=c(rH1982val,rH1983val,rH1984val,rH1985val,rH1986val,rH1987val,rH1988val,rH1989val,rH1990
val,rH1991val,rH1992val,rH1993val,
rH1994val,rH1995val,rH1996val,rH1997val,rH1998val,rH1999val,rH2000val,rH2001val,rH2002val,rH20
03val,rH2004val,rH2005val,rH2006val, rH2007val,rH2008val,rH2009val,rH2010val, rH2011val,
```

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rH2012val, rH2013val, rH2014,
rM1982val,rM1983val,rM1984val,rM1985val,rM1986val,rM1987val,rM1988val,rM1989val,rM1990val,r
M1991val,rM1992val,rM1993val,
rM1994val,rM1995val,rM1996val,rM1997val,rM1998val,rM1999val,rM2000val,rM2001val,rM2002val,r
M2003val,rM2004val,rM2005val, rM2006val,rM2007val,rM2008val,rM2009val,rM2010val, rM2011val,
rM2012val, rM2013val, rM2014val)))
```

```
S_fix.p_rD_age.Psi_fix <-
mark(MS_harvest_group_class.process,MS_harvest_group_class.ddl,model.parameters=list(S=S_fix,
p=p_rD_age, Psi=Psi_fix))
```

```
#_______________________________________________________________________________________
# Top ranked model including covariate (sex) and quadratic function for p and rD
```

```
p_sex_rD_quad <- list(formula=~-1 + strA:f + strA:m + strH + strM + strD*(Time+I(Time^2)),
```

```
fixed=list(index=c(rH1982,rH1983,rH1984,rH1985,rH1986,rH1987,rH1988,rH1989,rH1990,rH1991,rH19
92,rH1993,rH1994,rH1995,rH1996,
rH1997,rH1998,rH1999,rH2000,rH2001,rH2002,rH2003,rH2004,rH2005,rH2006,rH2007,rH2008,rH2009,
rH2010,rH2011,rH2012,rH2013,rH2014,
rM1982,rM1983,rM1984,rM1985,rM1986,rM1987,rM1988,rM1989,rM1990,rM1991,rM1992,rM1993,rM
1994,rM1995,rM1996,rM1997,
rM1998,rM1999,rM2000,rM2001,rM2002,rM2003,rM2004,rM2005,rM2006,rM2007,rM2008,rM2009,rM
2010,rM2011,rM2012,rM2013, rM2014),
value=c(rH1982val,rH1983val,rH1984val,rH1985val,rH1986val,rH1987val,rH1988val,rH1989val,rH1990
val,rH1991val,rH1992val,rH1993val,
rH1994val,rH1995val,rH1996val,rH1997val,rH1998val,rH1999val,rH2000val,rH2001val,rH2002val,rH20
03val,rH2004val,rH2005val,rH2006val, rH2007val,rH2008val,rH2009val,rH2010val, rH2011val,
rH2012val, rH2013val, rH2014,
rM1982val,rM1983val,rM1984val,rM1985val,rM1986val,rM1987val,rM1988val,rM1989val,rM1990val,r
M1991val,rM1992val,rM1993val,
rM1994val,rM1995val,rM1996val,rM1997val,rM1998val,rM1999val,rM2000val,rM2001val,rM2002val,r
M2003val,rM2004val,rM2005val, rM2006val,rM2007val,rM2008val,rM2009val,rM2010val, rM2011val,
rM2012val, rM2013val, rM2014val)))
```

```
S_fix.p_sex_rD_quad.Psi_fix <-
mark(MS_harvest_group_class.process,MS_harvest_group_class.ddl,model.parameters=list(S=S_fix,
p=p_sex_rD_quad, Psi=Psi_fix))
```

```
##########################################################################################
# Final model selection for best fit of time and covariates with respect to p and rD
```

```
p_rD_time_cov.results <- collect.models(type="Multistrata", table=TRUE)
```

```
p_rD_time_cov_AIC <- model.table(p_rD_time_cov.results, type="Multistrata", sort = TRUE )
```

```
write.table(p_rD_time_cov_AIC, "p_rD_time_cov_AIC.txt", sep = "\t" )
```

```
# View parameter estimates of best model, p_sex_rD_quad
```

```
p_sex_rD_quad_betas <- S_fix.p_sex_rD_quad.Psi_fix$results$beta
```

```
write.table(p_sex_rD_quad_betas, "p_sex_rD_quad_betas.txt", sep = "\t" )
```

```
p_sex_rD_quad_reals <- S_fix.p_sex_rD_quad.Psi_fix$results$real
```

```
write.table(p_sex_rD_quad_reals, "p_sex_rD_quad_reals.txt", sep = "\t" )
```

```
######################################################################################
# Defining model structures and running models for Psi, applying best p rD model: p_sex_rD_quad

# Example of one model including Quadratic time function for toH, toM, toD

HMD_quad <- list(formula=~-1 + toH*(Time+I(Time^2))  + toM*(Time+I(Time^2))  +
toD*(Time+I(Time^2))  + fromH + fromM + fromD,

fixed=list(index=c(PsiH,PsiM,PsiD,Psi1981, Psi1982, Psi1983, Psi1984, Psi1985, Psi1986, Psi1987,
Psi1988, Psi1989, Psi1990, Psi1991, Psi1992, Psi1993, Psi1994, Psi1995, Psi1996, Psi1997, Psi1998,
Psi1999, Psi2000, Psi2001, Psi2002, Psi2004, Psi2006, Psi2007, Psi2008, Psi2009, Psi2014), value=0))

S_fix.p_sex_rD_quad.HMD_quad <-
mark(MS_harvest_group_class.process,MS_harvest_group_class.ddl,model.parameters=list(S=S_fix,
p=p_sex_rD_quad, Psi=HMD_quad))

#_____________________________________________________________________________________
# Example of one model including covariate (age) for toH, toM, toD

HMD_age <- list(formula=~-1 + toH:c + toH:y + toH:a + toM:c + toM:y + toM:a + toD:c + toD:y + toD:a
+ fromH + fromM + fromD,

fixed=list(index=c(PsiH,PsiM,PsiD,Psi1981, Psi1982, Psi1983, Psi1984, Psi1985, Psi1986, Psi1987,
Psi1988, Psi1989, Psi1990, Psi1991, Psi1992, Psi1993, Psi1994, Psi1995, Psi1996, Psi1997, Psi1998,
Psi1999, Psi2000, Psi2001, Psi2002, Psi2004, Psi2006, Psi2007, Psi2008, Psi2009, Psi2014), value=0))

S_fix.p_sex_rD_quad.HMD_age <-
mark(MS_harvest_group_class.process,MS_harvest_group_class.ddl,model.parameters=list(S=S_fix,
p=p_sex_rD_quad, Psi=HMD_age))

#_____________________________________________________________________________________
# Example of one model including covariate interaction (age * sex) for toH, and Cubic time function for
# toM, toD

H_age_sex_MD_cubic <- list(formula=~-1 + toH:c:f + toH:y:f + toH:a:f + toH:c:m + toH:y:m + toH:a:m +
toM*(Time+I(Time^2)+I(Time^3)) + toD*(Time+I(Time^2)+I(Time^3)) + fromH + fromM + fromD,

fixed=list(index=c(PsiH,PsiM,PsiD,Psi1981, Psi1982, Psi1983, Psi1984, Psi1985, Psi1986, Psi1987,
Psi1988, Psi1989, Psi1990, Psi1991, Psi1992, Psi1993, Psi1994, Psi1995, Psi1996, Psi1997, Psi1998,
Psi1999, Psi2000, Psi2001, Psi2002, Psi2004, Psi2006, Psi2007, Psi2008, Psi2009, Psi2014), value=0))

S_fix.p_sex_rD_quad.H_age_sex_MD_cubic <-
mark(MS_harvest_group_class.process,MS_harvest_group_class.ddl,model.parameters=list(S=S_fix,
p=p_sex_rD_quad, Psi=H_age_sex_MD_cubic))

#_____________________________________________________________________________________
# Top ranked model including covariate interaction (age * sex) for toH, and age covariate for toM, toD

H_age_sex_M_age_D_age <- list(formula=~-1 + toH:c:f + toH:y:f + toH:a:f + toH:c:m + toH:y:m +
toH:a:m + toM:c + toM:y + toM:a + toD:c + toD:y + toD:a + fromH + fromM + fromD,

fixed=list(index=c(PsiH,PsiM,PsiD,Psi1981, Psi1982, Psi1983, Psi1984, Psi1985, Psi1986, Psi1987,
Psi1988, Psi1989, Psi1990, Psi1991, Psi1992, Psi1993, Psi1994, Psi1995, Psi1996, Psi1997, Psi1998,
Psi1999, Psi2000, Psi2001, Psi2002, Psi2004, Psi2006, Psi2007, Psi2008, Psi2009, Psi2014), value=0))
```

```
S_fix.p_sex_rD_quad.H_age_sex_M_age_D_age <-
mark(MS_harvest_group_class.process,MS_harvest_group_class.ddl,model.parameters=list(S=S_fix,
p=p_sex_rD_quad, Psi=H_age_sex_M_age_D_age))

#_____________________________________________________________________________________
# Best behavioral model including covariate interaction (age * behavior) for toH and toM, and age
# covariate for toD

H_age_bp_M_age_bp_D_age <- list(formula=~-1 + toH:c:b + toH:y:b + toH:a:b + toH:c:p + toH:y:p +
toH:a:p + toM:c:b + toM:y:b + toM:a:b + toM:c:p + toM:y:p + toM:a:p + toD:c + toD:y + toD:a + fromH +
fromM + fromD,

fixed=list(index=c(PsiH,PsiM,PsiD,Psi1981, Psi1982, Psi1983, Psi1984, Psi1985, Psi1986, Psi1987,
Psi1988, Psi1989, Psi1990, Psi1991, Psi1992, Psi1993, Psi1994, Psi1995, Psi1996, Psi1997, Psi1998,
Psi1999, Psi2000, Psi2001, Psi2002, Psi2004, Psi2006, Psi2007, Psi2008, Psi2009, Psi2014), value=0))

S_fix.p_sex_rD_quad.H_age_bp_M_age_bp_D_age <-
mark(MS_harvest_group_class.process,MS_harvest_group_class.ddl,model.parameters=list(S=S_fix,
p=p_sex_rD_quad, Psi=H_age_bp_M_age_bp_D_age))

######################################################################################
# We applied the same procedure as above for final model selection for best fit of time, covariates, and
# covariate interactions with respect to Psi, and to call parameter estimates of best model,
# H_age_sex_M_age_D_age, and best behavioral model, H_age_bp_M_age_bp_D_age.


######################################################################################
######################################################################################

 # These scripts

# 1) regress changes in nuisance behaviors reported on estimated mortality components

# 2) model uncertainty in mortality component estimates using Monte Carlo simulations

# 3) generate Fig. 4 in manuscript

# clear

rm(list=ls())

#Read in mortality components file including means, se, and variances for 2001-2012

hmd_se_var_ordered_yr_2001_2012 <- read.delim(" . . .txt")

#Read in NJFDW conflict reports with changes in magnitude and percent change between years

conflict_change_2002_2013 <- read.delim(" . . .txt")

######################################################################################
# Generate regression model, regressing magnitude change nuisances reported in year t+1 on harvest +
management mortality in year t

est_yrs <- list(2001:2012)

# Define mean, standard errors, and variance for harvest plus management mortality years 2001-2012

hm_mn <- hmd_se_var_ordered_yr_2001_2012$hm_mn
```

```
# We applied the same parameterization format to hm_se, hm_var, hmd_mn(total mortality means), hmd_se, hmd_var . . .

# Define the magnitude of change in nuisance behaviors between each year from 2002-2013

n_mag_ch <- conflict_change_2002_2013$n_mag_ch

# Create a data frame including estimate years, mortality component means and standard errors, and change in nuisance behaviors

hmd_mn_se <- data.frame(est_yrs, hm_mn, hm_se, hmd_mn, hmd_se, n_mag_ch)

# Order years by increasing harvest + management mortality

hm_mn_se_ordered <- hmd_mn_se[order(hm_mn),]

# Define independent variable in regression, harvest + management mortality

hm_mn_ordered <- hm_mn_se_ordered$hm_mn

# Define dependent variable in regression, n_mag_ch_ordered

n_mag_ch_ordered <- hm_mn_se_ordered$n_mag_ch

# Generate regression model, including adjusted r^2, p-value, and 95% confidence interval

n_mag_hm_model = lm(n_mag_ch_ordered ~ hm_mn_ordered)

n_mag_hm_model_modsum <- summary(n_mag_hm_model)

r2_n_mag_hm_model = n_mag_hm_model_modsum$adj.r.squared

p_n_mag_hm_model = n_mag_hm_model_modsum$coefficients[2,4]

conf_n_mag_hm_model <- predict(n_mag_hm_model, interval = "confidence", level =0.95)

# Define years within which harvest occurred for graphic below

hm_mn_se_ordered$X2001.2012 <- c("F","F","F","F","F","F","F","T","T","T","T","T")

display_harvest_yrs <- hm_mn_se_ordered$X2001.2012 =="F"


##########################################################################################
# Incorporate uncertainty in harvest + management mortality components using Monte Carlo simulations

# Create a data frame including estimate years, mortality component means, standard errors, variances, and change in nuisance behaviors

hm_mn_se_var <- data.frame(est_yrs, hm_mn, hm_se, hm_var, n_mag_ch)

# Order years by increasing harvest + management mortality

hm_mn_se_var_ordered <- hm_mn_se_var[order(hm_mn),]

# Define independent variable in initial regression, harvest + management mortality

hm_mn_ordered_mc <- hm_mn_se_var_ordered$hm_mn

# Define dependent variable in initial regression, n_mag_ch_ordered
```

```
n_mag_ch_ordered_mc <- hm_mn_se_var_ordered$n_mag_ch

# Define number of simulations, 1000

sim <- 1000

# Create empty matrix with same dimensions as hm_mn_se_var

MhmMC_n <- matrix(0,sim,dim(hm_mn_se_var)[1])

# Write a function to get a and b parameters of a Beta distribution to parametrically describe uncertainty in
# mortality probabilities

a.start = 0.25

Beta_parm = function (par,mn,variance){

  a = par

  b = (a/mn)-a

  (((a*b)/((a+b)^2*(a+b+1)))-variance)^2

}

# Create list of appropriate dimensions

L <- dim(hm_mn_se_var_ordered)[1]

# For loop to iteratively generate harvest + management mortality mean estimates for years 2001-2012
# from Beta distributions defined by                  # means and variances in respective year

for (i in 1:L) {

  if (hm_mn_se_var_ordered[i,2]>0) {

    # solve for the a and b values for S1

    # sd / mean = coefficient of variation,  take percentage of that ==> new CV ==> get new variance from
there

    # percent multiplier = 1-j*0.1

    a_parmMH =
optim(a.start,Beta_parm,mn=hm_mn_se_var_ordered[i,2],variance=hm_mn_se_var_ordered[i,4],method='
BFGS')

    a_mortMH = a_parmMH$par

    b_mortMH = (a_mortMH/hm_mn_se_var_ordered[i,2])-a_mortMH

    MhmMC_n[,i] = rbeta(sim, a_mortMH, b_mortMH)

  }

  else MhmMC_n[,i] <- MhmMC_n[,i]

  }

# Compare original estimates to those generated from simulation

hm_mn_se_var_ordered[,2]
```

```
head(MhmMC_n)

# Create matrix of response variables of appropriate size to iteratively estimate 1000 regression models

Mhm_change_n <- matrix(n_mag_ch_ordered_mc,nrow=1,ncol=12)

hm_change_n_matrix <- matrix(Mhm_change_n,nrow=1000,ncol=length(Mhm_change_n),byrow=TRUE)

# Create empty vectors to population with estimates generated from Monte Carlo simulations

IntMhmreal_n <- rep(0,sim)          # intercept real scale

SlMhmreal_n <- rep(0,sim)           # slope real scale

r2_Mhm_n <- rep(0,sim)

p_Mhm_n <- rep(0,sim)

# Generate 1000 regression models including intercept, slope, r^2, and p-values, iteratively regressing hm_change_n_matrix on MhmMC_n

for (j in 1:sim) {

  IntMhmreal_n[j] <- lm(hm_change_n_matrix[j,] ~ MhmMC_n[j,])$coefficients[1]

  SlMhmreal_n[j] <- lm(hm_change_n_matrix[j,] ~ MhmMC_n[j,])$coefficients[2]

  r2_Mhm_n[j] = summary(lm(hm_change_n_matrix[j,] ~ MhmMC_n[j,]))$adj.r.squared

  p_Mhm_n[j] = summary(lm(hm_change_n_matrix[j,] ~ MhmMC_n[j,]))$coefficients[2,4]

}

#________________________________________________________________________________________
# Get 95% confidence intervals from simulated Monte Carlo values

IntMhmfitted_n <- mean(IntMhmreal_n)

IntMhmlower_n <- quantile(IntMhmreal_n, 0.025)

IntMhmupper_n <- quantile(IntMhmreal_n, 0.975)

SlMhmfitted_n <- mean(SlMhmreal_n)

SlMhmlower_n <- quantile(SlMhmreal_n, 0.025)

SlMhmupper_n <- quantile(SlMhmreal_n, 0.975)

x_hm_n <- seq(0.0,0.65, 0.01)

Mhmpred_n <- length(x_hm_n)

Mhmpredlower_n <- length(x_hm_n)

Mhmpredupper_n <- length(x_hm_n)

for (i in 1:length(x_hm_n)) {

  Mhmpred_n[i] <- IntMhmfitted_n+SlMhmfitted_n*x_hm_n[i]

  Mhmpredlower_n[i] <- IntMhmlower_n+SlMhmlower_n*x_hm_n[i]

  Mhmpredupper_n[i] <- IntMhmupper_n+SlMhmupper_n*x_hm_n[i]
```

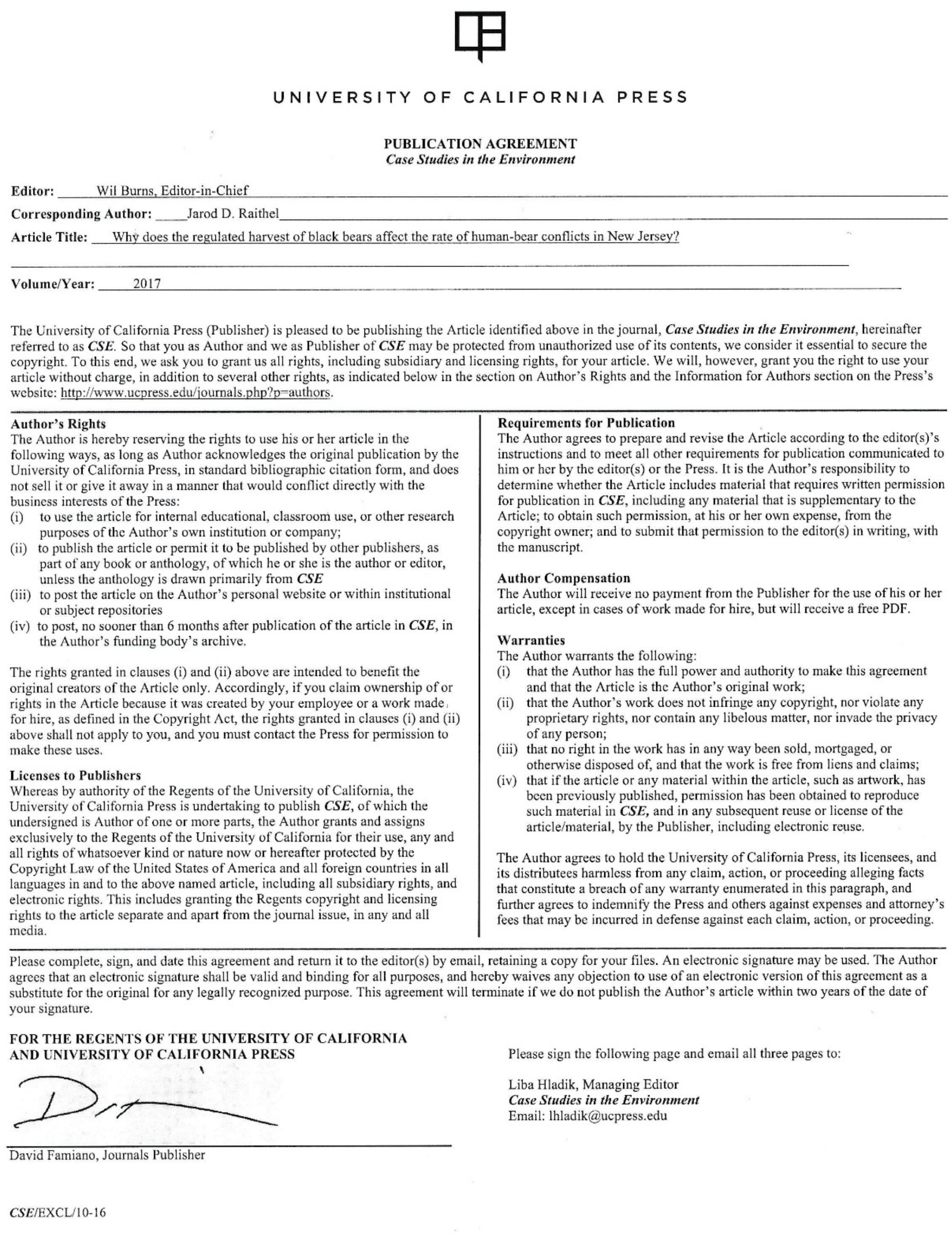
```
}

#########################################################################################

# We applied the same procedure as above to generate regression model and incorporate uncertainty in
# estimates using Monte Carlo simulations to total mortality in year t (harvest + management + dead all
# other) . . .

###########################################################################################

# Generate Fig. 4 in manuscript

par(mfrow=c(2,2))

plot(hm_mn_ordered, n_mag_ch_ordered, xlab = "Harvest + management mortality year t", ylab= "Change nuisances reported yr t+1",

    ylim=c(-400, 800), xlim=c(0,0.4), cex = 1.25, cex.axis = 1.35, cex.lab = 1.6, pch=ifelse(display_harvest_yrs=="FALSE", 19, 1))

text(x = 0.23, y = 780, labels = "R-squared = 0.338", cex = 1.4, adj = 0)

text(x = 0.23, y = 690, labels = "P-value = 0.028", cex = 1.4, adj = 0)

text(x = 0.23, y = 600, labels = "Slope = -1365.4 ", cex = 1.4, adj = 0)

lines(hm_mn_ordered, conf_n_mag_hm_model[,2], lty=3)

lines(hm_mn_ordered, conf_n_mag_hm_model[,3], lty=3)

X_hm_change_nuis <- c(0.02,0.36)

Y_hm_change_nuis <- predict(n_mag_hm_model, newdata=data.frame(hm_mn_ordered=X_hm_change_nuis))

lines(X_hm_change_nuis, Y_hm_change_nuis)

legend(x = 0.22, y = 590, legend = c("No harvest years", "Harvest years"), pch =c(1,19), bty = "n", cex = 1.4)


plot(hm_mn_ordered_mc, n_mag_ch_ordered, xlab= "Harvest + management mortality year t", ylab= "Change nuisances reported yr t+1",

    cex = 1.25, cex.axis = 1.35, cex.lab = 1.6, pch=ifelse(display_harvest_yrs=="FALSE", 19, 1), ylim=c(-400, 800), xlim=c(0.0,0.40))

lines(x_hm_n, Mhmpred_n, type="l")

lines(x_hm_n, Mhmpredlower_n, lty=3)

lines(x_hm_n, Mhmpredupper_n, lty=3)

arrows(hm_mn_ordered_mc[1]-hm_se_bars[1],80, hm_mn_ordered_mc[1]+hm_se_bars[1],80, length=0.1, angle=90, code=3, lty=1, col="gray")

arrows(hm_mn_ordered_mc[2]-hm_se_bars[2],196,hm_mn_ordered_mc[2]+hm_se_bars[2],196, length=0.1, angle=90, code=3, lty=1, col="gray")
```

```
arrows(hm_mn_ordered_mc[3]-hm_se_bars[3],-25,hm_mn_ordered_mc[3]+hm_se_bars[3],-25,
length=0.1, angle=90, code=3, lty=1, col="gray")

arrows(hm_mn_ordered_mc[4]-hm_se_bars[4],-62,hm_mn_ordered_mc[4]+hm_se_bars[4],-62,
length=0.1, angle=90, code=3, lty=1, col="gray")

arrows(hm_mn_ordered_mc[5]-hm_se_bars[5],40,hm_mn_ordered_mc[5]+hm_se_bars[5],40, length=0.1,
angle=90, code=3, lty=1,   col="gray")

arrows(hm_mn_ordered_mc[6]-hm_se_bars[6],785, hm_mn_ordered_mc[6]+hm_se_bars[6],785,
length=0.1, angle=90, code=3, lty=1, col="gray")

arrows(hm_mn_ordered_mc[7]-hm_se_bars[7],277,hm_mn_ordered_mc[7]+hm_se_bars[7],277,
length=0.1, angle=90, code=3, lty=1, col="gray")

arrows(hm_mn_ordered_mc[8]-hm_se_bars[8],-218,hm_mn_ordered_mc[8]+hm_se_bars[8],-218,
length=0.1, angle=90, code=3, lty=1, col="gray")

arrows(hm_mn_ordered_mc[9]-hm_se_bars[9],-336,hm_mn_ordered_mc[9]+hm_se_bars[9],-336,
length=0.1, angle=90, code=3, lty=1, col="gray")

arrows(hm_mn_ordered_mc[10]-hm_se_bars[10],-137,hm_mn_ordered_mc[10]+hm_se_bars[10],-137,
length=0.1, angle=90, code=3, lty=1, col="gray")

arrows(hm_mn_ordered_mc[11]-hm_se_bars[11],-96,hm_mn_ordered_mc[11]+hm_se_bars[11],-96,
length=0.1, angle=90, code=3, lty=1, col="gray")

arrows(hm_mn_ordered_mc[12]-hm_se_bars[12],-322,hm_mn_ordered_mc[12]+hm_se_bars[12],-322,
length=0.1, angle=90, code=3, lty=1, col="gray")

text(x = 0.19, y = 780, labels = "Mean R-squared  = 0.319", cex = 1.4, adj=0)

text(x = 0.19, y = 690, labels = "P-value < 0.001", cex = 1.4, adj=0)

text(x = 0.19, y = 600, labels = "Mean Slope = -1310.6 ", cex = 1.4, adj=0)

legend(x = 0.18, y = 590, legend = c("No harvest years", "Harvest years"), pch =c(1,19), bty = "n", cex =
1.4)


plot(hmd_mn_ordered, n_mag_ch_ordered_hmd, xlab = "Total mortality year t", ylab= "Change nuisances
reported yr t+1",

    ylim=c(-400, 800), xlim=c(0,0.80), cex = 1.25, cex.axis = 1.35, cex.lab = 1.6,
pch=ifelse(display_harvest_yrs=="FALSE", 19, 1))

text(x = 0.47, y = 780, labels = "R-squared = 0.201", cex = 1.4, adj = 0)

text(x = 0.47, y = 690, labels = "P-value = 0.081", cex = 1.4, adj = 0)

text(x = 0.47, y = 600, labels = "Slope = -1011.8 ", cex = 1.4, adj = 0)

lines(hmd_mn_ordered, conf_n_mag_hmd_model[,2], lty=3)

lines(hmd_mn_ordered, conf_n_mag_hmd_model[,3], lty=3)

X_hmd_change_nuis <- c(0.17,0.67)
```

```
Y_hmd_change_nuis <- predict(n_mag_hmd_model,
newdata=data.frame(hmd_mn_ordered=X_hmd_change_nuis))

lines(X_hmd_change_nuis, Y_hmd_change_nuis)

legend(x = 0.45, y = 590, legend = c("No harvest years", "Harvest years"), pch =c(1,19), bty = "n", cex =
1.4)


plot(hmd_mn_ordered_mc, n_mag_ch_ordered_hmd, xlab= "Total mortality year t", ylab= "Change
nuisances reported yr t+1",

    cex = 1.25, cex.axis = 1.35, cex.lab = 1.6, pch=ifelse(display_harvest_yrs=="FALSE", 19, 1), ylim=c(-
400, 800), xlim=c(0,0.8))

lines(x_hmd_n, Mhmdpred_n, type="l")

lines(x_hmd_n, Mhmdpredlower_n, lty=3)

lines(x_hmd_n, Mhmdpredupper_n, lty=3)

arrows(hmd_mn_ordered_mc[1]-hmd_se_bars[1],196,hmd_mn_ordered_mc[1]+hmd_se_bars[1],196,
length=0.1, angle=90, code=3, lty=1, col="gray")

arrows(hmd_mn_ordered_mc[2]-hmd_se_bars[2],-62,hmd_mn_ordered_mc[2]+hmd_se_bars[2],-62,
length=0.1, angle=90, code=3, lty=1, col="gray")

arrows(hmd_mn_ordered_mc[3]-hmd_se_bars[3],40,hmd_mn_ordered_mc[3]+hmd_se_bars[3],40,
length=0.1, angle=90, code=3, lty=1, col="gray")

arrows(hmd_mn_ordered_mc[4]-hmd_se_bars[4],80, hmd_mn_ordered_mc[4]+hmd_se_bars[4],80,
length=0.1, angle=90, code=3, lty=1, col="gray")

arrows(hmd_mn_ordered_mc[5]-hmd_se_bars[5],785, hmd_mn_ordered_mc[5]+hmd_se_bars[5],785,
length=0.1, angle=90, code=3, lty=1, col="gray")

arrows(hmd_mn_ordered_mc[6]-hmd_se_bars[6],277,hmd_mn_ordered_mc[6]+hmd_se_bars[6],277,
length=0.1, angle=90, code=3, lty=1, col="gray")

arrows(hmd_mn_ordered_mc[7]-hmd_se_bars[7],-25,hmd_mn_ordered_mc[7]+hmd_se_bars[7],-25,
length=0.1, angle=90, code=3, lty=1, col="gray")

arrows(hmd_mn_ordered_mc[8]-hmd_se_bars[8],-218,hmd_mn_ordered_mc[8]+hmd_se_bars[8],-218,
length=0.1, angle=90, code=3, lty=1, col="gray")

arrows(hmd_mn_ordered_mc[9]-hmd_se_bars[9],-322,hmd_mn_ordered_mc[9]+hmd_se_bars[9],-322,
length=0.1, angle=90, code=3, lty=1, col="gray")

arrows(hmd_mn_ordered_mc[10]-hmd_se_bars[10],-336,hmd_mn_ordered_mc[10]+hmd_se_bars[10],-
336, length=0.1, angle=90, code=3, lty=1, col="gray")

arrows(hmd_mn_ordered_mc[11]-hmd_se_bars[11],-137,hmd_mn_ordered_mc[11]+hmd_se_bars[11],-
137, length=0.1, angle=90, code=3, lty=1, col="gray")

arrows(hmd_mn_ordered_mc[12]-hmd_se_bars[12],-96,hmd_mn_ordered_mc[12]+hmd_se_bars[12],-96,
length=0.1, angle=90, code=3, lty=1, col="gray")

text(x = 0.39, y = 780, labels = "Mean R-squared  = 0.172", cex = 1.4, adj=0)
```

```
text(x = 0.39, y = 690, labels = "P-value < 0.001", cex = 1.4, adj=0)

text(x = 0.39, y = 600, labels = "Slope = -899.7 ", cex = 1.4, adj=0)

legend(x = 0.37, y = 590, legend = c("No harvest years", "Harvest years"), pch =c(1,19), bty = "n", cex = 1.4)
```

Appendix 5.1. Permissions Case Studies in the Environment



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Case Studies in the Environment
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CSE/EXCL/10-16

Appendix 5.2. Teaching Notes

Article Case Long Title:

Why does the regulated harvest of black bears affect the rate of human-bear conflicts in New Jersey?

Target Group:

We feel this case study would be most appropriate for introductory-level Ecology and Wildlife Management courses typically taken during the first two years of an undergraduate curriculum.

Learning Objectives and Key Issues:

Students will evaluate how sex, age-class, and behavior (problem vs. normal) affect the probability that black bears in northwestern New Jersey die from harvest, lethal control, and other causes of mortality like vehicle strikes. Given these results, students will then propose possible explanations for the observed correlation between bear harvest rates and subsequent declines in nuisance bear behaviors reported. Informed by this remarkable dataset comprised of over 3,500 individual bears collected over 33 years, students will ultimately have a meaningful discussion about whether a carefully regulated bear harvest should be included in an integrated management strategy to conserve American black bears.

Teaching Strategy:

It is important to acknowledge at the onset of exploring this case study that recreational hunting of large carnivores, like the American black bear, may induce strong emotional reactions in students just as it does in the diverse stakeholders in the region. Managing black bears in the 21st century requires that wildlife managers balance the preservation of viable black bear populations (i.e., reduce the probability of extinction to within acceptable limits) with the protection of human welfare and property in a cost-effective manner. Tolerance for this species can vary widely across the constituents that mangers serve, including those that rarely interact with bears in urban areas like adjacent New York City, and those that frequently interact with bears in relatively rural northwestern New Jersey. Our role as ecological researchers is to provide the objective information that wildlife managers require to take informed actions, regardless of our personal beliefs.

To illustrate just how captivating, adaptive, and resilient this species is, we recommend initially viewing the immensely popular YouTube video of "Pedals" the beloved, bipedal NJ black bear (https://www.youtube.com/watch?v=ZVsA5vlFV4E). To emphasize how capable black bears are at exploiting anthropogenic resources, we also engage students with this short video clip (https://www.youtube.com/watch?v=jMRmMLBaYt4).

Additional detail on the small sample size of "problem" yearlings within this modeling framework

'Problem' cubs and yearlings are the known young of problem sows (often times caught with them in culvert traps or marked in dens). However, unmarked 'yearlings', independent from their mothers for just one summer/fall prior to being subject to the December harvest simply do not have much time to be captured in response to nuisance complaints, resulting in a very small sample size of problem yearlings and increasing our uncertainty in this estimate.

Addressing the similarities and differences in temporal patterns in nuisance versus threat behaviors

We focus on nuisance behaviors reported in subsequent analyses as these behaviors are 4-6 times more frequent than threat behaviors. We simply do not have strong statistical power with threats reported – although the temporal patterns between nuisances and threats reported do mirror one another. Important here, whether we look at nuisances alone, or nuisances + threats (all problems combined) there is a statistically significant relationship between increasing harvest + lethal control rates and subsequent declines in problem behaviors. Many of the 'threatening' bears are initially nuisance bears that become progressively bolder. Bears tend to be successful capitalizing on garbage cans, bird feeders, pet food, etc., before they risk breaking into homes. A reduction in nuisance bears will likely, ultimately, translate to a reduction in threat bears as well.

Additional explanation on the coupling of harvest and lethal control rates in Question 3 / Slide #15

Lethal management rates are coupled with harvest rates so that we have non-zero mortality estimates in non-harvest years, increasing the sample size in our regression from 5 to 12 data points (years herein). These are the two primary tools managers employ to reduce conflict. Lethal control, which is even more targeted than harvest at removing problem individuals, is unable to alone reduce conflict (see “No harvest years” in Slide #15). Important here, integrated management including lethal control, regulated harvest and education *together* have resulted in reductions in conflicts.

Update on the “Landscape of Fear”

Since submission of our initial case study, we now have strong evidence that indicates that hunting is indeed altering the behavior of ‘normal’ bears (i.e., bears without a history of conflict) in that these bears avoid anthropogenic habitats in years where harvest occurred and when mast conditions (i.e., natural bear foods) are at or above average. Normal bears are significantly more likely to transition from wildland to anthropogenic habitats in years were harvest did not occur and in years when mast conditions were poor.

Appendix 5.3. Case Study Questions Answer Key

Article Case Long Title:

Why does the regulated harvest of black bears affect the rate of human-bear conflicts in New Jersey?

Case study questions:

1. Examine our tables of cause-specific mortality estimates to answer the following questions:
    a. How does the age-class (cub, yearling, and adult) and sex of the black bear (female vs. male) interact to influence its probability of being harvested? Remember to examine whether 95% confidence intervals overlap to determine if differences are significant. (Slide #16)
        i. *Answer: Females, regardless of age-class, have an ~16% chance of being harvested. However, adult males are significantly less likely to be harvested than either adult females or yearling males. There is no significant difference between male vs. female cubs as 95% CIs overlap.*

    b. What is the dominant source of mortality for young bears (cubs and yearlings) in this human-dominated landscape? Why do you think this category is so high? (Slide #16)
        i. *Answer: 'All Other Sources' category, which describes the high probability that young bears are struck and killed by vehicles in human-dominated landscapes.*

c. How does behavior affect the probability that black bears in northwestern NJ are harvested and lethally controlled? (Slide #17)

    i. *Answer: Adult bears that have been previously designated a problem (assigned nuisance or threat status) are significantly more likely to be harvested. We see the opposite pattern for yearlings, but this is an artifact of our models, i.e., most yearling bears are harvested before they have a chance to become a problem. For all age-classes of bears, those individuals previously designated a problem are significantly more likely to be lethally controlled by management action.*

2. Examine our temporal line/bar graph on Slide #13: Do you see a relationship between nuisance behaviors reported and the implementation of harvest in years 2003, 2005, and 2010-2013?

    i. *Answer: In all years following harvest (2004, 2006, 2011-2013), the number of problem behaviors reported declines, by an average of 29%. In all years following harvest moratoria (2005, 2007-2010), the number of problem behaviors reported increases, by an average of 23%.*

3. Examine our temporal line graph on Slide #14: In what year does the proportion of normal behaviors reported begin increasing relative to problem behaviors (that begin decreasing at this inflection point)? What event occurred during this time, and what are the implications for bear management?

   i. *Answer: The inflection occurs during the year 2008, following a sizeable investment by the NJDFW in educational materials and outreach to inform the public about how to properly manage attractants (e.g., garbage, bird feeders, pet food) when living and recreating in bear habitat. The implication is that educational outreach that alters human behaviors with respect to waste management is an important component of an integrated bear management strategy.*

4. Examine our linear regressions in Slide #15: Is the change in nuisance behaviors reported in year *t + 1* more strongly correlated with total mortality in year *t* or harvest + management mortality rates in year *t*?

   i. *Answer: 33.8% of the variation in change in nuisances reported in year t + 1 is explained by just harvest + management mortality rates alone in year t, compared to only 20.1% of this variation explained by total mortality. This is reflected in the smaller p-value (more significant relationship) in the regression on the left. Our model predicts that for every 0.1 increase in harvest and management mortality rate in year t, we would expect to see ~137 less nuisances reported in year t + 1.*

5. Provide three hypotheses explaining the observed correlation between implementing the new bear harvest and subsequent declines in nuisance bear behaviors reported. They do not have to be “mutually-exclusive,” i.e., two or more hypotheses could be occurring simultaneously.

a. *Answers:*

   i. *As total mortality increases, driven by harvest and lethal control, there are less bears in the northwestern NJ population to get in trouble, so less problem behaviors are reported.*

   ii. *Harvest and lethal control disproportionately removes problem individuals, thereby selecting for bears that stay out of trouble. Over time this has resulted in a reduction in problem bears relative to normal bears on the landscape.*

   iii. *Hunting by humans is creating a "landscape of fear," causing nuisance bears to change their behavior and avoid humans and their anthropogenic resources.*

   iv. *The substantial investment by NJDFW in educating humans has resulted in humans changing their behavior. Reducing anthropogenic attractants around human homes, businesses, campgrounds, etc. has reduced the number of nuisance bears attracted to these areas.*

   v. *Reporting of nuisance bear behaviors is subjectively influenced by human attitudes. For example, in years following harvest moratoria, disgruntled pro-hunting advocates may have been more likely to report nuisance bear behaviors in response to anger over the rapid closure of the bear hunting season.*

6. If the objective of management is to reduce the black bear population in NJ, but maintain its long-term viability (ensure that it does not go extinct), discuss whether or not the science indicates that harvest can be used as a conservation tool.
    i. *Answer: See Research and Management Conclusions (Slide #18). The NJ black bear harvest, as it is currently administered, disproportionately removes adult females, as well as, adult nuisance bears from the population. In large, long-lived vertebrates, adult females drive population dynamics because they have high survival and high reproductive value (i.e., from a demographer's perspective, "males are cheap!"). Therefore, the NJ harvest is likely to exert a strong influence on curbing population growth and reducing population size, helping managers achieve "culturally carrying capacity." Further, harvest may represent a powerful conservation tool, as hunters disproportionately kill problem bears, resulting in a relative increase in bears that do not exploit anthropogenic resources, and ultimately increasing the culture's carrying capacity for this remarkable species.*

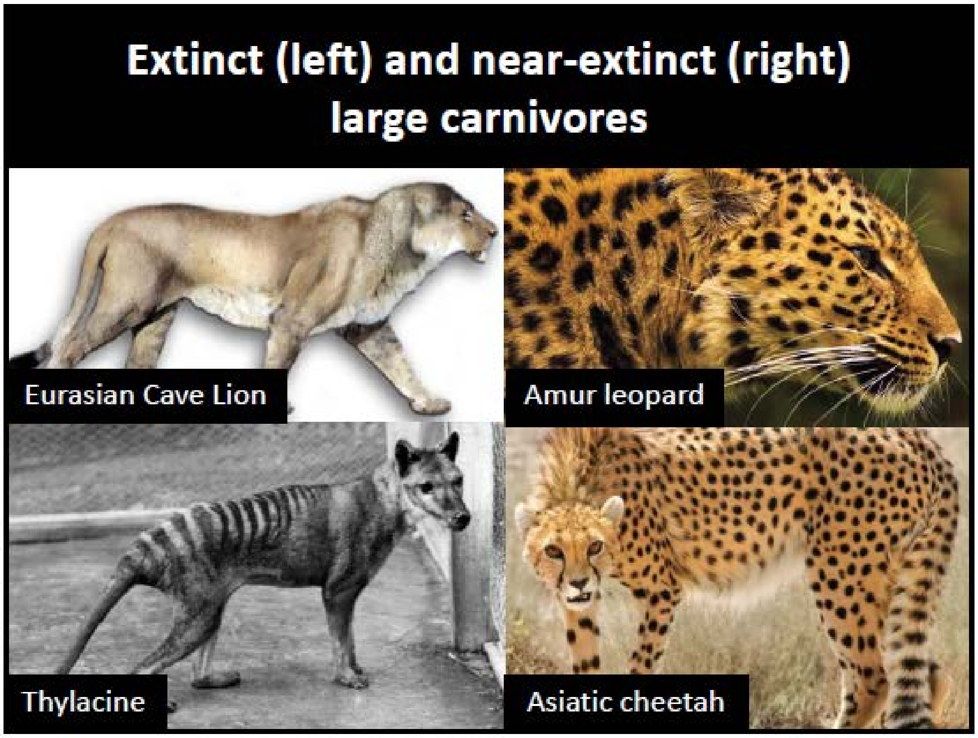
7. In the USA, state wildlife agencies are primarily funded by excise taxes on hunting and fishing gear (i.e., firearms and ammunition) and through the purchase of recreational hunting and fishing licenses. However, interest in sport hunting has declined dramatically within the millennial generation, as this cohort tends to prefer the non-consumptive use of wildlife (e.g., bird watching). Given your recent discussion on the potential application of harvest as a conservation tool, what are the

implications of these broader hunter participation trends for wildlife research and management?

i. *Answer: Unless state wildlife management agencies invest in educating the public across the socio-political spectrum on the benefits of harvest, thereby increasing participation and revenue, then state agencies will need to seek different funding opportunities (e.g., excise taxes on non-consumptive equipment) or reduce management and research budgets accordingly.*

Appendix 5.4. Accompanied Slides for Case Study

Why does hunting impact
human-bear conflict in New Jersey?



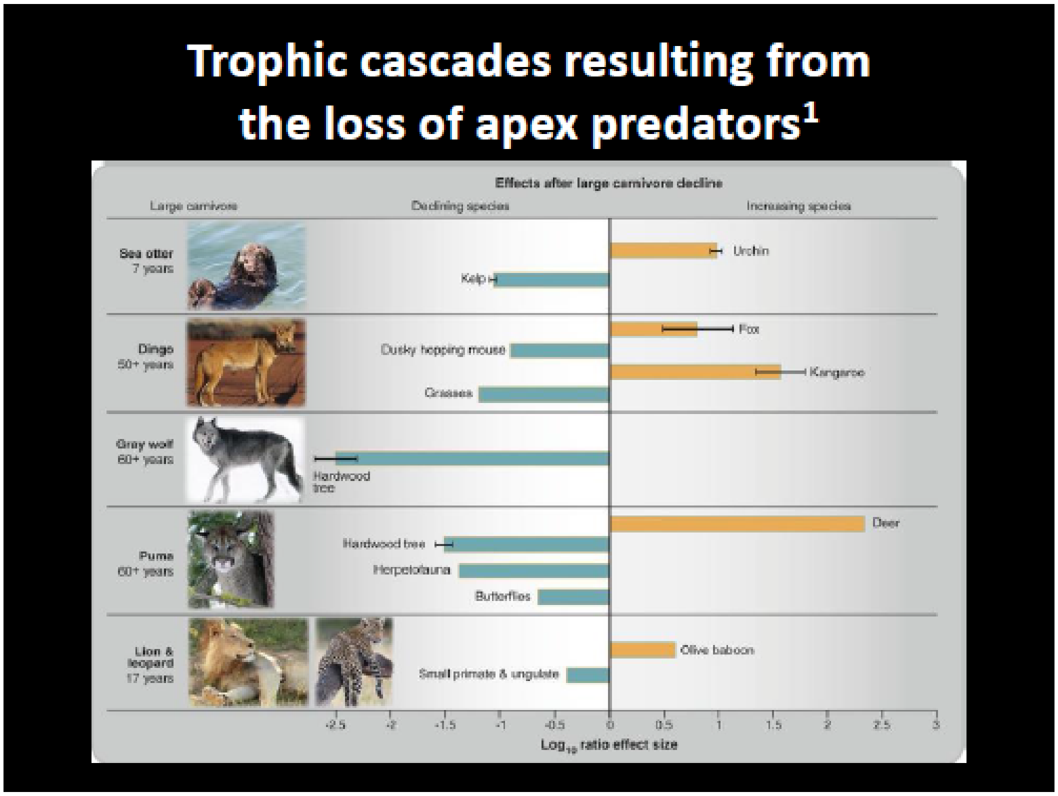
Extinct (left) and near-extinct (right)
large carnivores

Eurasian Cave Lion

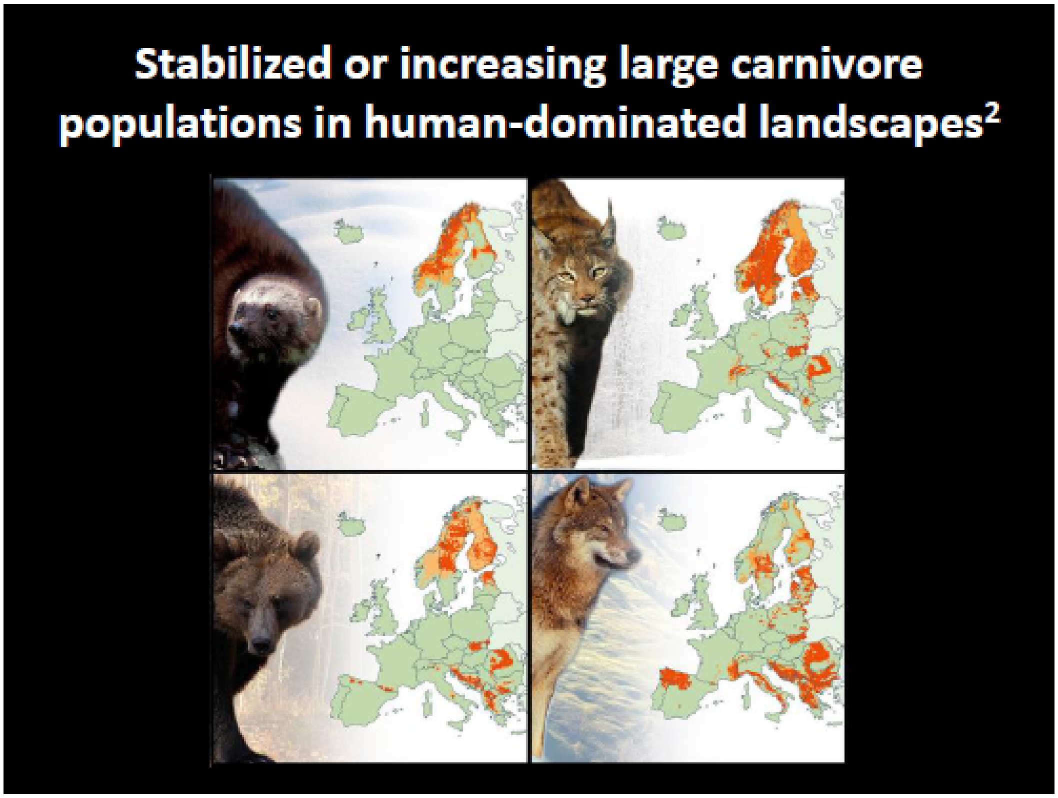
Amur leopard

Thylacine

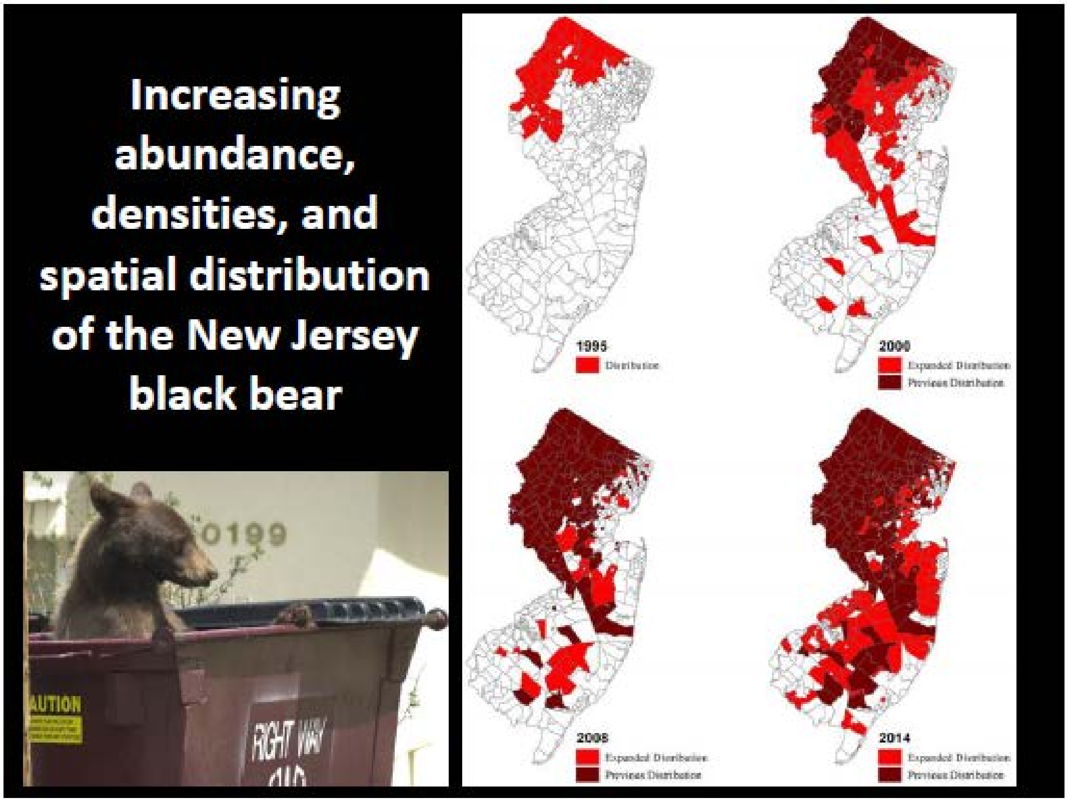
Asiatic cheetah



Trophic cascades resulting from the loss of apex predators[1]



Stabilized or increasing large carnivore populations in human-dominated landscapes[2]



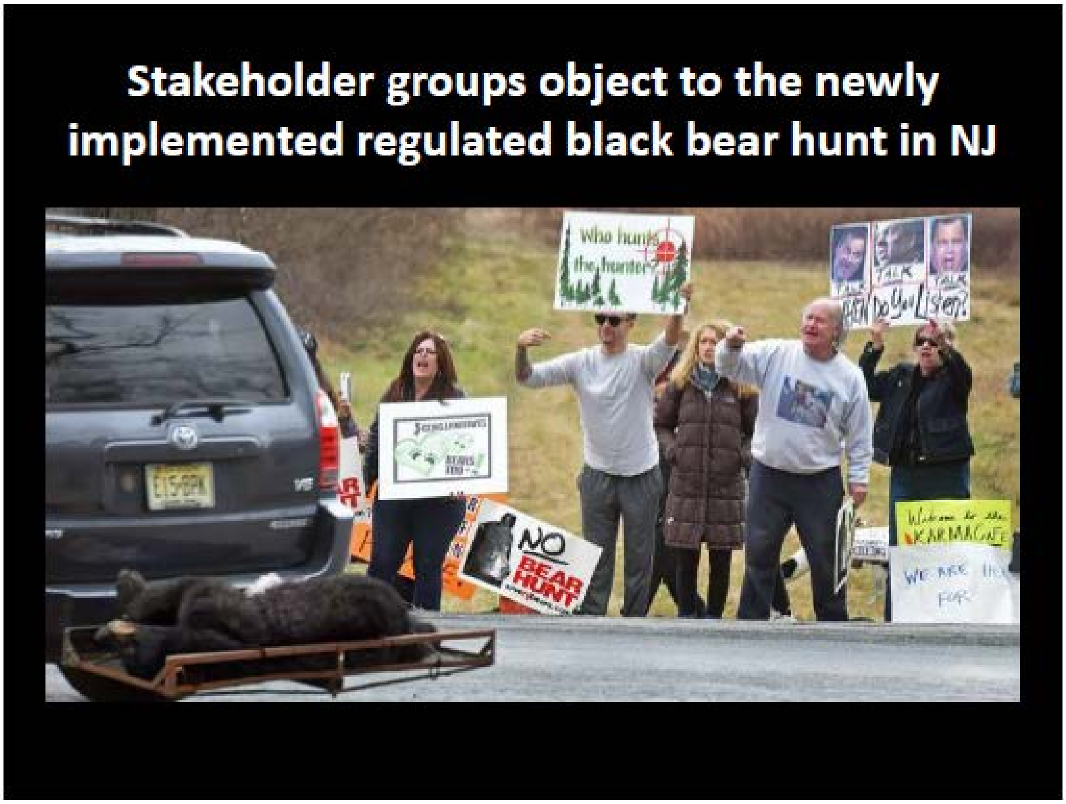
Increasing abundance, densities, and spatial distribution of the New Jersey black bear

- Increasing human densities & housing units
- Shift in settlement patterns AWAY from urban centers

>9 million USD spent by NJFDFW responding to >26,500 human-bear incidents

New Jersey hiker mauled by black bear, body found in woods with animal nearby

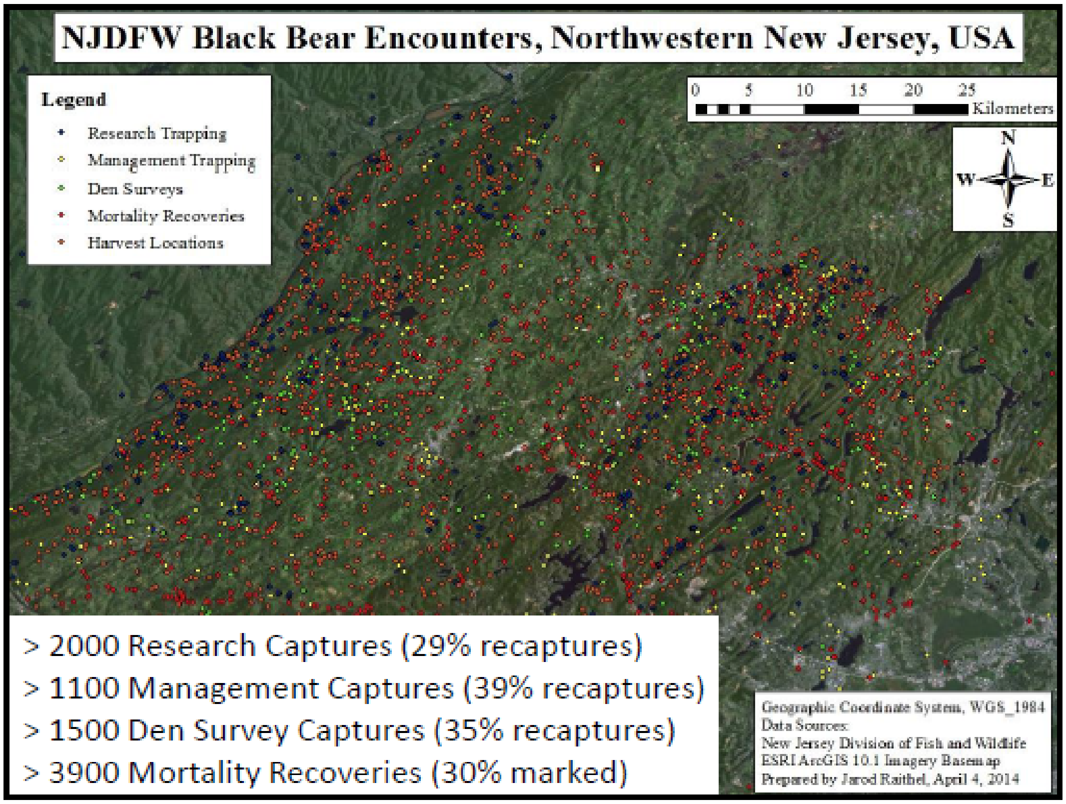
By Ray Sanchez, CNN
Updated 9:57 AM ET, Wed September 24, 2014



Stakeholder groups object to the newly implemented regulated black bear hunt in NJ

Are harvest & lethal control effective in reducing human-bear conflicts?

1. Estimate sources and rates of mortality
2. Understand how sex, age and behavior relate to mortality risk
3. Determine if a relationship exists between increasing control and harvest and reductions in conflicts



NJDFW Black Bear Encounters, Northwestern New Jersey, USA

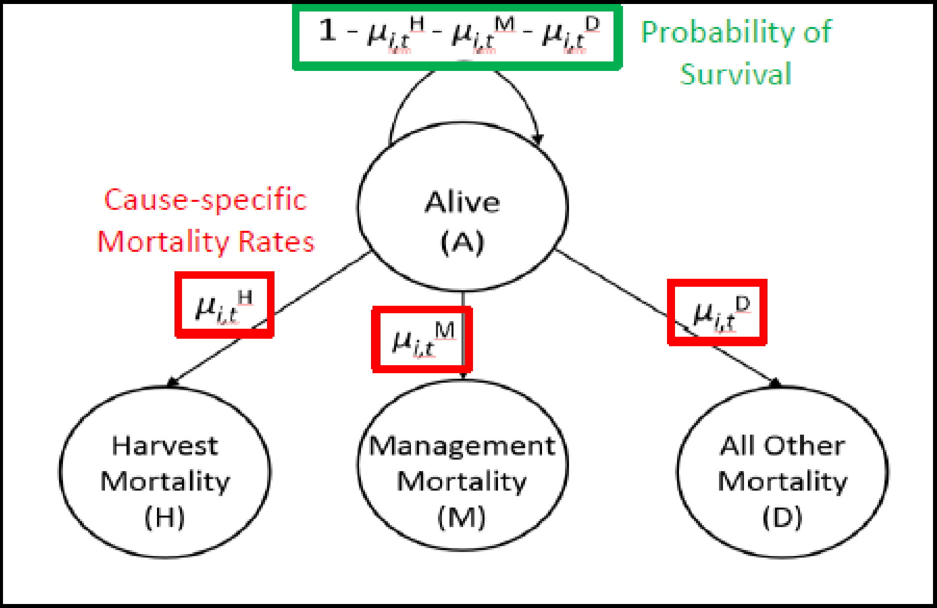
Legend

- ● Research Trapping
- ● Management Trapping
- ● Den Surveys
- ● Mortality Recoveries
- ● Harvest Locations

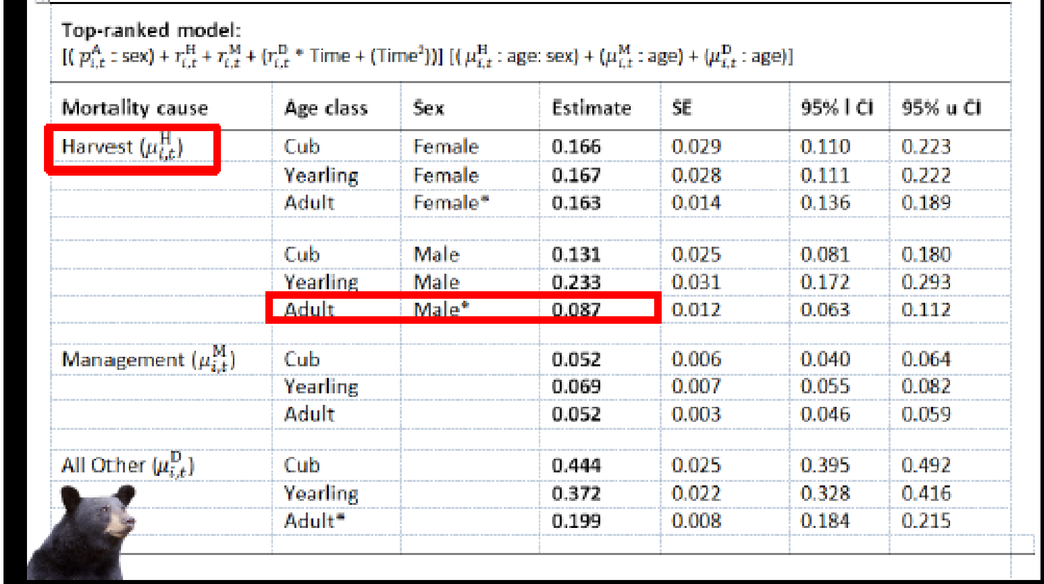
\> 2000 Research Captures (29% recaptures)
\> 1100 Management Captures (39% recaptures)
\> 1500 Den Survey Captures (35% recaptures)
\> 3900 Mortality Recoveries (30% marked)

Geographic Coordinate System, WGS_1984
Data Sources:
New Jersey Division of Fish and Wildlife
ESRI ArcGIS 10.1 Imagery Basemap
Prepared by Jarod Raithel, April 4, 2014

Capture-Mark-Recapture (CMR) Modeling Method

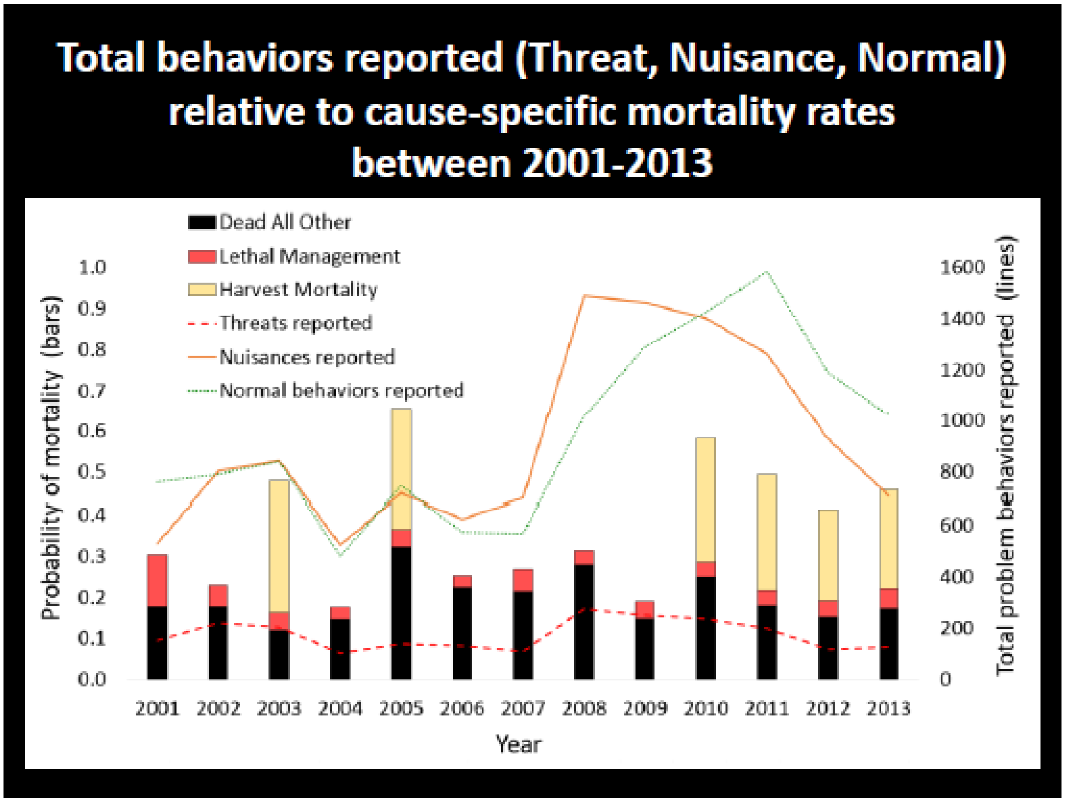


0.087 = probability that an adult male black bear will die from harvest in NJ during study period

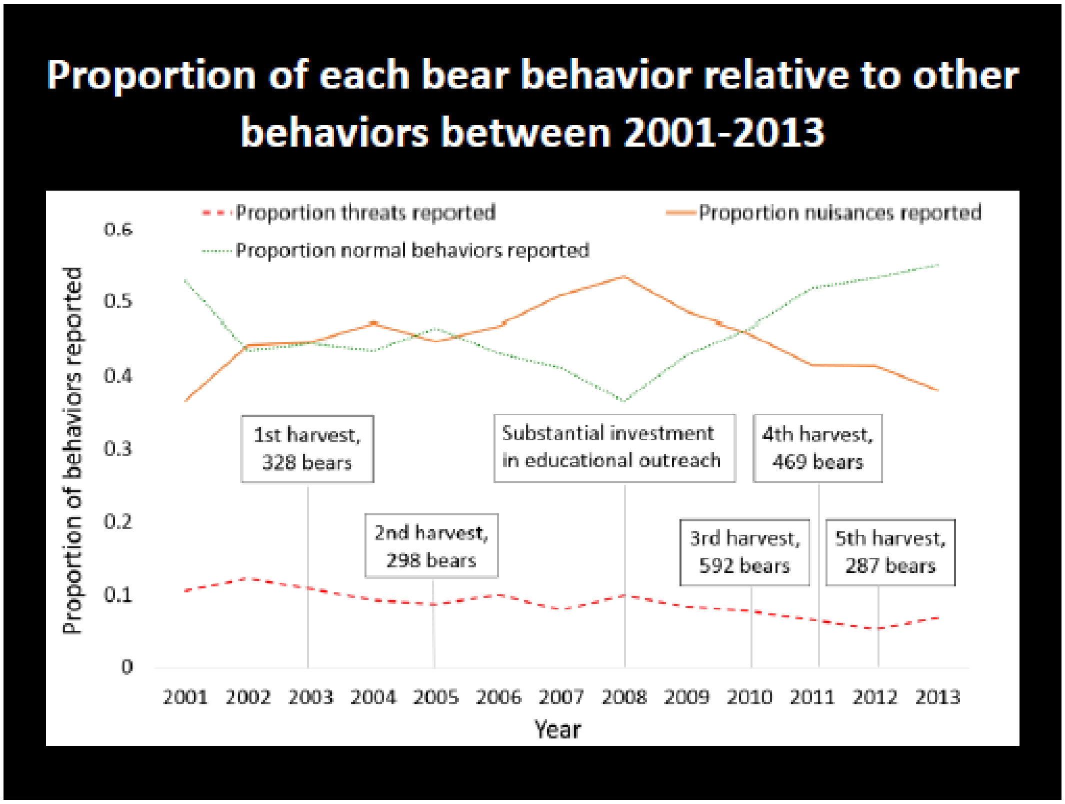


Top-ranked model:
[($p_{i,t}^{A}$: sex) + $r_{i,t}^{H}$ + $r_{i,t}^{M}$ + ($r_{i,t}^{D}$ * Time + (Time²))] [($\mu_{i,t}^{H}$: age: sex) + ($\mu_{i,t}^{M}$: age) + ($\mu_{i,t}^{D}$: age)]

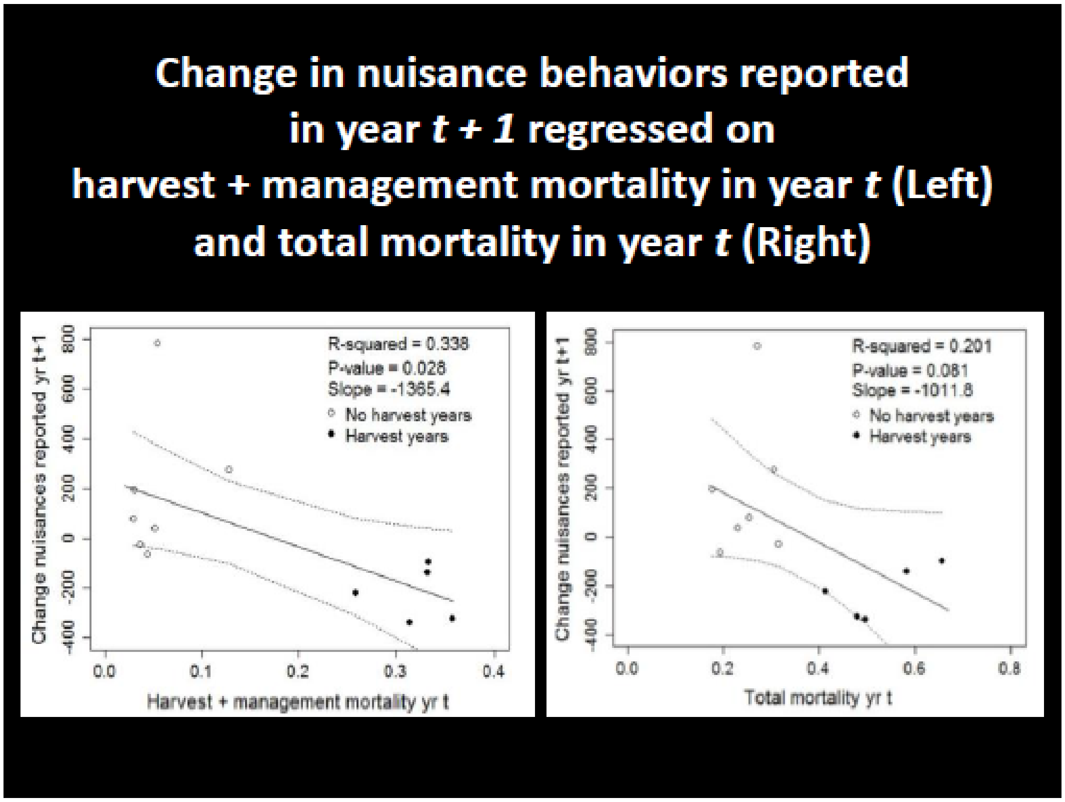
Mortality cause	Age class	Sex	Estimate	SE	95% l CI	95% u CI
Harvest ($\mu_{i,t}^{H}$)	Cub	Female	**0.166**	0.029	0.110	0.223
	Yearling	Female	**0.167**	0.028	0.111	0.222
	Adult	Female*	**0.163**	0.014	0.136	0.189
	Cub	Male	**0.131**	0.025	0.081	0.180
	Yearling	Male	**0.233**	0.031	0.172	0.293
	Adult	Male*	**0.087**	0.012	0.063	0.112
Management ($\mu_{i,t}^{M}$)	Cub		**0.052**	0.006	0.040	0.064
	Yearling		**0.069**	0.007	0.055	0.082
	Adult		**0.052**	0.003	0.046	0.059
All Other ($\mu_{i,t}^{D}$)	Cub		**0.444**	0.025	0.395	0.492
	Yearling		**0.372**	0.022	0.328	0.416
	Adult*		**0.199**	0.008	0.184	0.215



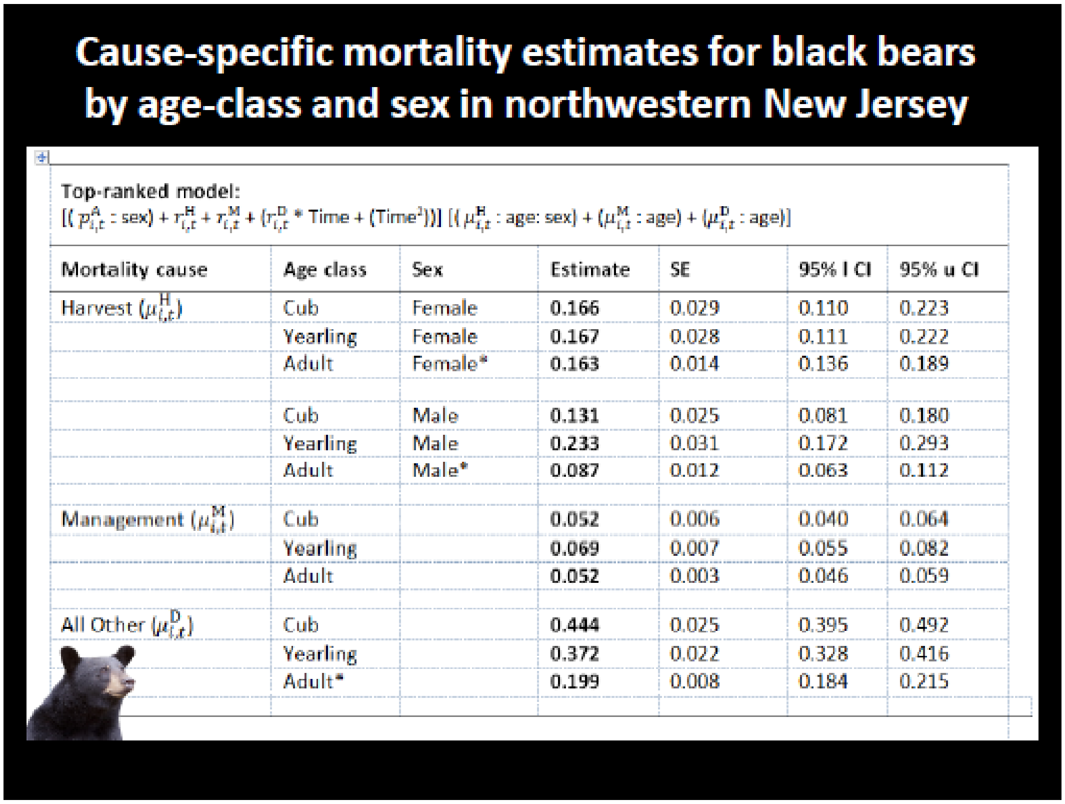
Total behaviors reported (Threat, Nuisance, Normal) relative to cause-specific mortality rates between 2001-2013



Proportion of each bear behavior relative to other behaviors between 2001-2013



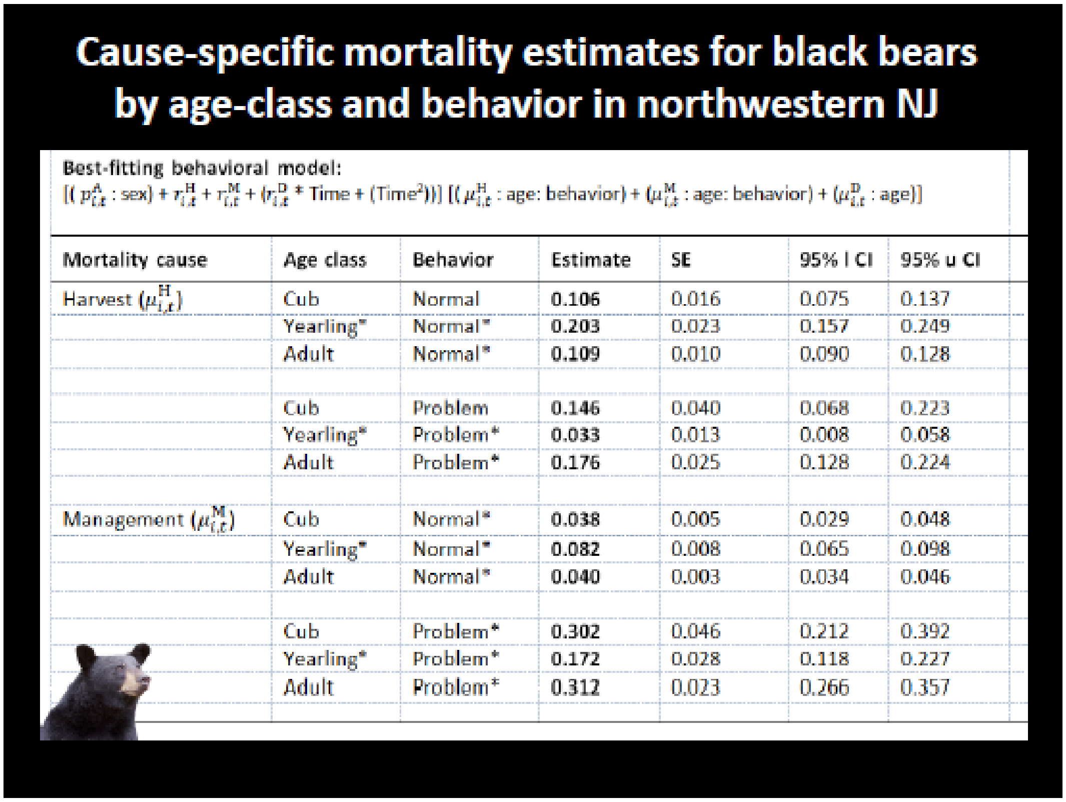
Change in nuisance behaviors reported in year *t + 1* regressed on harvest + management mortality in year *t* (Left) and total mortality in year *t* (Right)



Cause-specific mortality estimates for black bears by age-class and sex in northwestern New Jersey

Top-ranked model:
$[(p^{A}_{i,t} : \text{sex}) + r^{H}_{i,t} + r^{M}_{i,t} + (r^{D}_{i,t} * \text{Time} + (\text{Time}^{2}))]\ [(\mu^{H}_{i,t} : \text{age: sex}) + (\mu^{M}_{i,t} : \text{age}) + (\mu^{D}_{i,t} : \text{age})]$

Mortality cause	Age class	Sex	Estimate	SE	95% l CI	95% u CI
Harvest ($\mu^{H}_{i,t}$)	Cub	Female	**0.166**	0.029	0.110	0.223
	Yearling	Female	**0.167**	0.028	0.111	0.222
	Adult	Female*	**0.163**	0.014	0.136	0.189
	Cub	Male	**0.131**	0.025	0.081	0.180
	Yearling	Male	**0.233**	0.031	0.172	0.293
	Adult	Male*	**0.087**	0.012	0.063	0.112
Management ($\mu^{M}_{i,t}$)	Cub		**0.052**	0.006	0.040	0.064
	Yearling		**0.069**	0.007	0.055	0.082
	Adult		**0.052**	0.003	0.046	0.059
All Other ($\mu^{D}_{i,t}$)	Cub		**0.444**	0.025	0.395	0.492
	Yearling		**0.372**	0.022	0.328	0.416
	Adult*		**0.199**	0.008	0.184	0.215



Cause-specific mortality estimates for black bears by age-class and behavior in northwestern NJ

Best-fitting behavioral model:

$[(p^{A}_{i,t} : \text{sex}) + r^{H}_{i,t} + r^{M}_{i,t} + (r^{D}_{i,t} * \text{Time} + (\text{Time}^{2}))]\ [(\mu^{H}_{i,t} : \text{age: behavior}) + (\mu^{M}_{i,t} : \text{age: behavior}) + (\mu^{D}_{i,t} : \text{age})]$

Mortality cause	Age class	Behavior	Estimate	SE	95% l CI	95% u CI
Harvest ($\mu^{H}_{i,t}$)	Cub	Normal	**0.106**	0.016	0.075	0.137
	Yearling*	Normal*	**0.203**	0.023	0.157	0.249
	Adult	Normal*	**0.109**	0.010	0.090	0.128
	Cub	Problem	**0.146**	0.040	0.068	0.223
	Yearling*	Problem*	**0.033**	0.013	0.008	0.058
	Adult	Problem*	**0.176**	0.025	0.128	0.224
Management ($\mu^{M}_{i,t}$)	Cub	Normal*	**0.038**	0.005	0.029	0.048
	Yearling*	Normal*	**0.082**	0.008	0.065	0.098
	Adult	Normal*	**0.040**	0.003	0.034	0.046
	Cub	Problem*	**0.302**	0.046	0.212	0.392
	Yearling*	Problem*	**0.172**	0.028	0.118	0.227
	Adult	Problem*	**0.312**	0.023	0.266	0.357

Research and Management Conclusions

- Given the disproportionate take of adult females, harvest represents a pragmatic approach to meet cultural carrying capacity and safely curb population growth
- Given the disproportionate removal of nuisance bears, harvest coupled with lethal management can result in a reduction in human-bear conflicts

Questions, concerns, comments?

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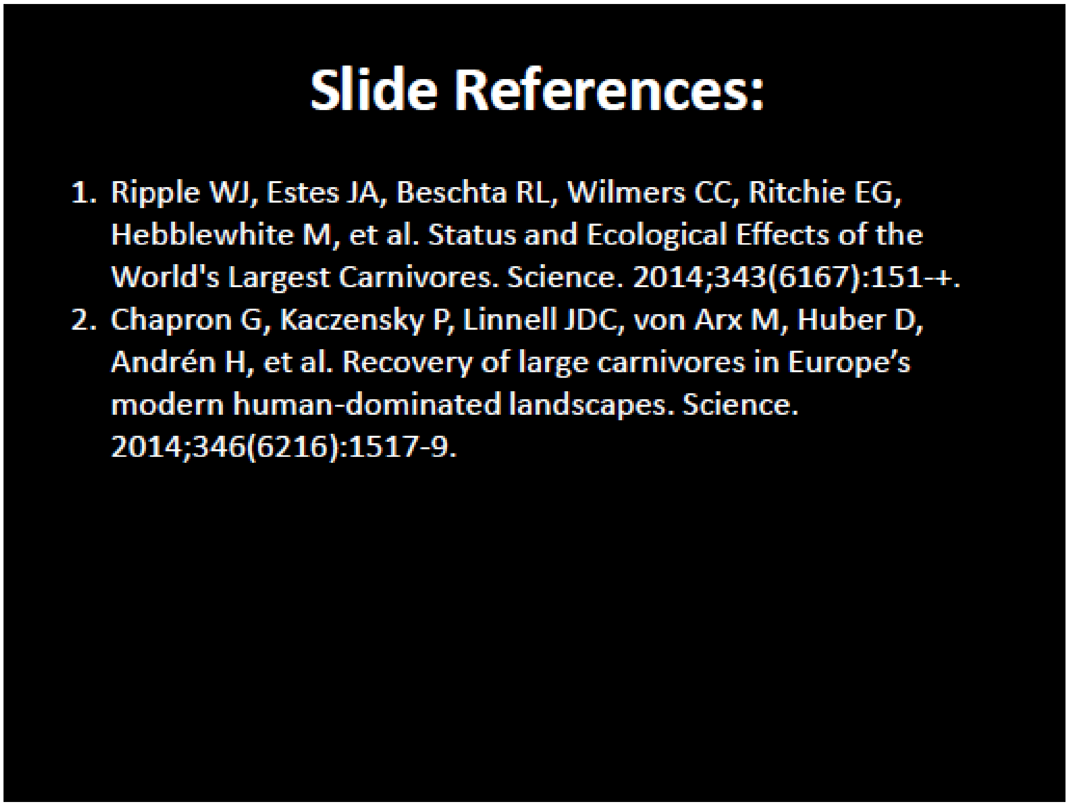

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2. Chapron G, Kaczensky P, Linnell JDC, von Arx M, Huber D, Andrén H, et al. Recovery of large carnivores in Europe’s modern human-dominated landscapes. Science. 2014;346(6216):1517-9.

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Utah State University, Logan, Utah, USA; Dept. of Wildland Resources and Ecology Center.
Advisor: Dr. Lise M. Aubry

- Awarded Presidential Doctoral Research Fellowship (https://rgs.usu.edu/pdrf/)
- My dissertation research is focused on integrating black bear behavior, spatial ecology, and population dynamics to inform bear conservation and management in northwestern New Jersey, USA.
- Skills: teaching and mentoring of students, large database management, programming and statistical analyses (e.g. mark-recapture models, matrix population models, resource selection models), community and professional presentations, grant proposal and publication writing.
- Software: Canvas, Camtasia, Microsoft Access, R, ArcGIS.

M.S. in Wildlife Biology, G.P.A. 4.0. May 2002 – May 2005

The University of Montana, Missoula, Montana, USA; College of Forestry and Conservation
Advisor: Dr. Daniel H. Pletscher

- Awarded Boone and Crockett Research Assistantship
- My thesis research measured recruitment and cause-specific mortality of calf elk in response to manipulated cougar densities in the Garnet Mountain Range of west-central Montana, USA.
- Skills: ground and helicopter elk captures, ground and fixed-wing aircraft radio-tracking, necropsies, ungulate classification flights, technician and volunteer supervision, community and professional presentations, grant proposal and publication writing.
- Software: Program MARK, ArcGIS, MATLAB

B.S. in Wildlife & Fisheries Sciences, Cons. Biology, G.P.A 3.89. Jan 1998 – Dec 2001

Texas A&M University, College Station, Texas, USA; Department of Wildlife and Fisheries Sciences. *Magna Cum Laude* Honors.

PUBLICATIONS (http://jarodraithel.wixsite.com/home/publications)

- **Raithel, J. D.** 2005. Impact of calf survival on elk population dynamics in west-central Montana. M.S. Thesis, The University of Montana, Missoula. 118 pp. Citations = 11.
- **Raithel, J. D.,** M. J. Kauffman, and D. H. Pletscher. 2007. Impact of spatial and temporal variation in calf survival on the growth of elk populations. Journal of Wildlife Management, 71(3), 795 – 803. IF = 1.522, Citations = 102.
- Griffin, K. A., M. Hebblewhite, H. S. Robinson, P. Zager, S. M. Barber-Meyer, D. Christianson, S. Creel, N. C. Harris, M. A. Hurley, D. H. Jackson, B. K. Johnson, W. L. Myers, **J. D. Raithel**, M. Schlegel, B. L. Smith, C. White, and P. J.White. 2011. Neonatal mortality of elk driven by climate, predator phenology and predator community composition. Journal of Animal Ecology, 80(6), 1246 – 1257. IF = 4.937, Citations = 70.
- **Raithel, J. D.,** M. J. Reynolds-Hogland, D. N. Koons, P. C. Carr, and L. M. Aubry. 2016. Recreational harvest and incident-response protocols reduce human-carnivore conflicts in an anthropogenic landscape. Journal of Applied Ecology, DOI: 10.1111/1365-2664.12830, IF = 5.196. Associated Popular Press:
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TEACHING EXPERIENCE (http://jarodraithel.wixsite.com/home/teaching)

***** Received above the 10th percentile in student evaluations as assessed by the IDEA student surveys across all USU courses, in all University courses listed below. (http://jarodraithel.wixsite.com/home/documentation)

Undergraduate Courses:

Instructor of Record. Utah State University. Spring 2016
- Plant and Animal Populations (WILD 3810)

Co-Instructor. Utah State University. Fall 2015
- Monitoring and Assessment in Environmental Resource Management (WILD 4750)

Graduate Courses:

Graduate Teaching Assistant. Utah State University. Fall 2014
- Population State Variables (WILD 6401)
- Demographic Vital Rates (WILD 6402)
- Dynamics of Structured Populations (WILD 6403)

Public Secondary Math & Science:

Advanced Placement Biology Teacher. Aug 2010 – May 2013
- Tivy High School, Kerrville, Texas.
- AP Biology, Introduction to Biology I and II, and Integrated Physics and Chemistry.

STEM Environmental Studies Teacher. July 2009 – May 2010
- Kihei Charter High School, Maui, Hawaii.
- Environmental Science Fieldwork, Human Anatomy and Physiology, Hawaiian Flora and Fauna, Biology, Physics, Math Projects.

Biology Teacher. Jan 2007 – May 2008
- Carroll High School, Corpus Christi, Texas.
- Introduction to Biology I and II honors

Life Science Teacher. Aug 2005 – Dec 2006
- E. Merle Smith Junior High, Sinton, Texas.
- Life Science, Genetics and Ecology differentiated for special needs and honors students.

* Received highest categorical rating “Exceeds Expectations,” on Professional Development and Appraisal System evaluations every year I taught in Texas. 2005-08 and 2010-13 (evaluations available here: http://jarodraithel.wixsite.com/home/documentation)

OUTREACH

SCUBA Interpretive Presenter. May 2007 – June 2008
- Texas State Aquarium, Corpus Christi, Texas.
- Conducted underwater feeding shows in the coral reef exhibit for the public and cleaned exhibits using SCUBA, maintained SCUBA equipment, and assisted with animal medical care.

Children’s Instructor. Jan 2000 – Mar 2000

- Brazos Valley Natural History Museum, Bryan, Texas.
- Developed lesson plans and presented materials for spring and winter natural history education mini-camps for children aged 7 – 12.

Tour Guide, Animal Keeper, Counselor. Jan 1996 – Aug 1996

- Austin Zoo, Austin, Texas.
- Presented educational, interactive tours to children and developed curriculum for summer camps.

RESEARCH EXPERIENCE (http://jarodraithel.wixsite.com/home/research)

Team Scientist, Divemaster. July 2008 – Apr 2009

- Reef Check Thailand and Earthwatch Institute, Koh Ra Ecolodge, Phang Nga Province, Thailand
- Trained and supervised Earthwatch volunteers in collecting coral reef transect data using SCUBA at numerous sites along the Andaman coast following post-tsunami fishing fleet modernization.

Black Bear Repatriation Field Crew Leader. Feb 2005 – May 2005

- Louisiana State University, Tensas National Wildlife Refuge, Red River and Three Rivers Wildlife Management Areas, Louisiana.
- Captured and translocated federally threatened Louisiana subspecies of black bear to reestablish populations across the state and foster genetic connectivity.
- Capturing sows and cubs involved climbing massive bald cypress trees and anesthetizing bears in tree dens.

Radio-Tracking Technician. Jan 2002 – Apr 2002

- United States Geological Survey and Ducks Unlimited, Klamath, Summer Lake, and Lake Malheur National Wildlife Refuges, California & Oregon.
- Radio-tracked northern pintail ducks across expansive areas of northeastern California and eastern Oregon to document habitat use and mortality during spring migration.

Habitat Technician. June 2001 – Aug 2001

- Alaska Department of Fish and Game, Thorne Bay, Prince of Wales Island, Alaska.
- Free-darted and radio-collared blacktail deer and conducted vegetation transects for spatially-explicit habitat suitability index model that evaluates predator-prey dynamics between Sitka blacktail deer and the Alexander Archipelago gray wolf.

Paleo-ornithological Research Intern. Summer 2000

- Smithsonian Institution National Museum of Natural History, Washington, D.C.
- Selected from highly competitive global pool of applicants to participate in undergraduate Research Training Program.
- Conducted morphometric analysis evaluating the presence and extent of sexual size dimorphism within the moa-nalos, two extinct species of massive, flightless Hawaiian ducks.

Ornithology Field Intern. June 1999 – Aug 1999

- United States Forest Service, Stanislaus National Forest, California. .
- Surveyed for California spotted owl, northern goshawk, mountain yellow-legged frog, willow flycatcher, and great gray owl.
- Designed and presented weekly children's educational program on tracking Sierra Nevada mammals for local children and those visiting campgrounds.

PRESENTATIONS (http://jarodraithel.wixsite.com/home/publications)

- "Recreational Harvest and Incident-Response Management Reduces Human-Carnivore Conflicts in an Anthropogenic Landscape." Ecological Society of America, Fort Lauderdale, FL, August 2016.

- "Recreational Harvest and Incident-Response Management Reduces Human-Bear Conflicts in an Anthropogenic Landscape." 24th International Conference on Bear Research and Management, Anchorage, AK, June 2016.

- "Monitoring Wildlife Populations." Guest Lecture for Dr. Kari Veblen's WILD 4750 course, November 2015.

- "What I Wished I Knew – The Importance of Your Graduate Cohort." Presenter at Graduate Training Series Seminar (GrTS), September 2015.

- "What exactly is an 'urban bear'? Characterizing black bear space use across an anthropogenic landscape gradient." Student Research Symposium, Research Week, Utah State University, Logan, UT, April 2015.

- "Infinity in the Palm of your Hand." TEDxUSU Event, Utah State University, Logan, UT, October 2014: http://tedx.usu.edu/event/jarod-raithel/

- "Revisiting Perturbation Methods: Life Stage Simulation Analysis." Guest Lecture for Dr. David Koons' WILD 6403 course, November 2014.

- "Integrating Black Bear Behavior, Spatial Ecology & Population Dynamics Across An Anthropogenic Landscape Gradient." WILD Department's Pre-Project Symposium, USU, April 2014.

- “Science and Storytelling: Wonder, Humility, Gratitude and Interconnectedness.” USU’s Ignite! Seminar Series, Utah State University, Logan, UT, April 2014: http://www.youtube.com/watch?v=V6skjb1L1EU

- “Choosing to Become a Wildlife Biologist.” Presenter at “Career Day” at Cedar Ridge Middle School, North Logan. Gave four presentations to 7th graders, October 2013.

- “Declining recruitment and cause-specific mortality of calf elk in the Garnet Mountains of west-central Montana.” The Wildlife Society 11th Annual Conference, Calgary, Alberta, Canada, September 2004.

- “Cause-specific mortality of calf elk in the Garnet Mountain Range, Montana.” The Wildlife Society Montana State Chapter Annual Conference, Bozeman, Montana, February 2004.

- “Sexual dimorphism in the moa-nalos, extinct, flightless Hawaiian waterfowl.” American Ornithologists’ Union in conjunction with the Society of Canadian Ornithologists and the British Ornithologists’ Union, St. John’s, Newfoundland, Canada, August 2000.

GRANTS, HONORS, AWARDS (http://jarodraithel.wixsite.com/home/awards)

- Awarded $4,000 USU Graduate Enhancement Award. 2017

- Awarded the Robins Award for USU Graduate Student Teacher of the Year, an institution-wide competition. Spring 2016

- Awarded Graduate Instructor of the Year for the Quinney College of Natural Resources. Spring 2016

- Awarded $80,000 by the Presidential Doctoral Research Fellowship. One fellowship given annually to applicant within each USU college. 2013 – 2017

- Awarded $28,000 by Bear Trust International (co-PI). August 2013

- Awarded $1,500 total in student travel grants from USU Research and Graduate School, Wildland Resources Department, Ecology Center and Quinney College of Natural Resources. 2013 - 2016

- Awarded $20,000 by The Rocky Mountain Elk Foundation (Grant author and co-PI). January 2004

- Awarded $20,000 by Safari Club International Foundation (Grant author and co-PI). November 2003

- Awarded $43,500 by the Montana Department of Fish, Wildlife and Parks. May 2002

- Awarded 1st place undergraduate oral presentation at Texas A&M Wildlife and Fisheries Student Research Symposium. Spring 2001

- Selected for W.B. Davis Scholars Program, awarded to top 1% of seniors within Texas A&M Wildlife and Fisheries Program. 2001-2002

- Placed in top 10 out of approximately 1200 Texas A&M students in competitive public speaking forum. Fall 1998

SERVICE (http://jarodraithel.wixsite.com/home/service)

Mentorship

- Mentored two students as a Science Fair Coach that placed 1st and 2nd in Zoology in the statewide Texas Science Fair. Spring 2013

- Mentored three undergraduate students for Project TRAIN (Teaching and Research for American Indians) at University of Montana. 2003 – 2004

University Contributions

- Facilitator of Large Mammal Ecology Working Group. Weekly discussion group composed of faculty and graduate students from WILD and BIO departments. Fall 2013

- Participated in reviewing and scoring grant proposals for the Undergraduate Research & Creative Opportunities (URCO) program. November 2015 and February 2016

- Participated in student round-table discussion with visiting National Public Radio correspondent Joe Palca. November 2014

- Facilitated discussion about my experiences in communicating science to broader audiences to Undergraduate Honors Social. November 2014

Press coverage

- Utah Public Radio: http://upr.org/post/people-and-bears-hunting-coexistence

- The British Ecological Society: http://www.britishecologicalsociety.org/press-release-controversial-black- bear-hunt-disproportionately-harvests-nuisance-bears-reduces-human-bear-conflicts/

- Phys Org: http://phys.org/news/2016-11-harvest-nuisance-black-jersey-human-bear.html

- NJ Herald: http://www.njherald.com/20161129/study-hunting-key-to-sustaining-bear-population

- Promoted TEDx 2014 during interview with Kerry Bringhurst, Utah Public Radio. October 2014

- Completed video interview describing my black bear research with videographer Mason Johnson as part of promotional series produced by USU Office of Research and Graduate Studies. May 2014

- Hosted social at our home for visiting professor, Dr. Gary Alt, Biologist and Principal Scientist at Normandeau Associates, Environmental Consultants. February 2014

Reviewed Manuscripts for:

- Biological Conservation, Journal of Animal Ecology, Journal of Mammalogy, Journal of Raptor Research

Memberships in Professional Organizations:

- International Association for Bear Research and Management
- Ecological Society of America
- The Wildlife Society

CERTIFICATIONS

- Hawaii Highly-Qualified Educator: Science Grades 5 – 12 and Math Grades 5 – 9

- Texas State Board of Educators: General Composite Science Grades 8 – 12 and 4 – 8; Math Grades 4 – 8; and Pedagogy and Professional Responsibilities Early Child – Grade 12