Resource selection and the relationship of land cover and harvest regulations with survival and harvest of male wild turkeys in south-central Tennessee

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DEDICATION

I dedicate this thesis firstly to my lovely wife, Hannah, who has supported me in every step of my graduate school journey. Her daily encouragement was crucial in keeping me on track and giving me perseverance to accomplish this feat. I am so very grateful that she put up with me ranting weekly about wild turkeys and complaining that my “models” were not running correctly.

I also want to dedicate this thesis to those in my life that gave me a passion for wildlife and science. My father and mother, John and Melodie Smith, taught me how to work hard and gave me an enduring appreciation of hunting, hiking, and perpetually spending time in God’s creation.

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PREFACE

We studied wild turkey survival, harvest, and resource selection from 2017–2021 in south-central Tennessee. Chapter 1 contains background information and history of wild turkey conservation and their status in the United States and Tennessee, specifically. Chapter 2 describes the analysis of resource selection, the relationship of land cover with survival, and effect of regulation changes on harvest of juvenile male wild turkeys. Chapter 3 describes the analysis of land cover and predator relationships with mortality and harvest as well as the factors that affect reported harvest for adult male wild turkeys. Chapter 4 expands upon and synthesizes the management implications from Chapter 2 and 3. Chapter 2 and Chapter 3 are formatted as stand-alone chapters prepared for publication in the Journal of Wildlife Management.

ABSTRACT

Documenting male wild turkey (*Meleagris gallopavo*) resource selection, survival, and harvest is crucial to effectively manage this economically important game bird. To better understand how landscape composition affects wild turkey resource selection and what factors may influence survival and harvest of male wild turkeys, we attached backpack VHF radio transmitters to males (n = 265) in five counties of south-central Tennessee, 2017–2021. We modeled resource selection for juvenile males and estimated annual survival and harvest for both age classes. We calculated landscape metrics associated with each turkey's home range, and we calculated a predator index using remote-sensing cameras to determine if landscape characteristics and the relative abundance of predators were related to mortality. We also analyzed hunter harvest data to determine how various factors were related to male harvest. Juvenile male wild turkeys selected smaller row crop fields, less mature forest interior, and lower proportions of developed land and young forest. Hay/pasture interior was positively related to juvenile survival, but the effect was reduced as the interspersion of young forest increased. Adult non-harvest mortality increased with greater interspersion of developed land and forest and a greater predator index. Annual survival was 0.73 (95% CI = 0.65–0.81) for juveniles and 0.38 (95% CI = 0.32–0.45) for adults with an annual spring harvest rate of 0.03 (95% CI = 0.00–0.06) for juveniles and 0.32 (95% CI = 0.27–0.38) for adults. A decrease in the bag limit from 4 to 2 reduced the proportion of juveniles in the harvest but did not decrease adult harvest. The 14-day spring season delay had no effect on reported harvest. Increased fecundity two years prior to a given year was related to an increase in adult turkey harvest in that year. These results suggest woody cover may reduce survival because of increased ambush opportunities for turkey predators, whereas increased proportions of cover types that provide enhanced visibility and food resources benefit wild

turkey survival. Meso-mammalian predator management and regulatory restrictions on the harvest of juvenile males could increase the adult male population and likely increase turkey hunter satisfaction.

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CHAPTER 1. INTRODUCTION

The wild turkey (*Meleagris gallopavo*) is considered one of the greatest conservation success stories in North America (Dickson 1992). In the 1940s wild turkeys were only found in 18 of the 39 states of its original range, but extensive restoration efforts led to turkeys being present in all states except Alaska by the 1990s (Mosby and Handley 1943, Kennamer et al. 1992). The turkey population in Tennessee experienced similar patterns as populations in the rest of the country, expanding from 18 of 95 counties with wild turkeys present in 1952 to all 95 counties in 1990 following extensive restoration efforts (Shields et al. 2020). However, wild turkey harvest, population estimates, and productivity have declined since 2000 in many states, especially in the Southeast (Tapley et al. 2011, Byrne et al. 2015, Chamberlain et al. 2022). In Tennessee, there have been concerns as a result of declines in harvest in select counties as well as reductions in poult-per-female ratio and relative abundance that have mirrored recent southeastern trends post restoration (Byrne et al. 2015, United States Geological Survey 2022, Shields 2024).

Various factors have been hypothesized as the impetus for these declines. Developed land in the southeastern US has increased by more than 3 million hectares from 1997 to 2017, an increase of 26% over that period (United States Department of Agriculture 2020). A negative effect of developed land on wild turkeys may be explained by greater densities of meso-mammalian predators, less food and cover, and a greater probability of car collisions (Theobald et al. 1997, Frey and Conover 2006, Raiter et al. 2018, Dickie et al. 2020). Increases in wild turkey predators, specifically coyotes (*Canis latrans*) and bobcats (*Lynx rufus*), in the Southeast since the 1980s also may be contributing to the perceived decline (Roberts and Crimmins 2010, Casalena et al. 2015, Bragina et al. 2019, Wang et al. 2023). The effect of increased urbanization and predator populations likely has been further amplified by the increase of forest cover in the southeastern US that promotes lower visibility of predators for wild turkeys (Kunkel 1997, Whittingham and Evans 2004, Pollentier et al. 2014*b*, Ivey et al. 2024, Wang et al. 2025). More liberal hunting regulations post restoration (increased bag limits and fall harvest of adult females) also have been hypothesized to contribute to wild turkey declines (Tapley et al. 2000, Norman et al. 2004, Londe et al. 2023). These factors can affect both sexes of the wild turkey population, but understanding impacts specifically on male turkeys has many implications for hunters and the agencies that manage this economically important game bird.

More than 90,000 individuals each year are licensed to hunt wild turkey in Tennessee, contributing nearly $200 million to the economy annually by reporting a harvest of over 30,000 birds statewide each spring (Harris 2010, Shields 2024, Tennessee Wildlife Resources Agency [TWRA] 2026, CPI Inflation Calculator 2026; Figure A.1). Additionally, hunter license purchases contribute substantial funds to support TWRA management and administration. In addition to the economic benefits, the abundance of adult male wild turkeys on the landscape can be directly related to the quality of the hunting experience, as gobbling activity is considered the greatest determinant of hunter satisfaction (Swanson et al. 2005, Gruntorad et al. 2020, Quehl et al. 2024*b*). For these reasons, focusing research on adult male wild turkeys is crucial for sustaining wild turkey populations and increasing the recruitment, retention, and reactivation of hunters. Although juvenile males only compose about 10% of the male spring harvest in Tennessee, they are the direct source population for all harvestable adult male wild turkeys (Shields 2021, 2023, 2024, 2025). Therefore, researching male turkeys of all age classes is important for understanding wild turkey ecology and sustaining harvest of this popular game bird.

Calculating survival and harvest rates of wild turkeys is important for tracking and understanding turkey population trends across the range (Norman et al. 2022, Wightman et al.

2024). However, many studies report rates without further analysis into what exact factors are influencing survival and harvest (Hubbard and Vangilder 2007, Wright and Vangilder 2007, Diefenbach and Vreeland 2010, Wightman et al. 2024). Linking landscape characteristics to survival and harvest can inform landowners and land managers how to better manage a property to support sustainable turkey populations. Understanding how changes in hunting regulations impact turkey harvest also can be a tool for managing populations (Kurzejeski and Vangilder 1992, Wam et al. 2012, Watkins et al. 2018, Gruntorad and Chizinski 2021). Resource selection is another metric that can be used to inform management by examining the relative importance of landcover resources, and when studied in conjunction with survival, the quality of these resources for wild turkeys can be identified (Manly et al. 2004, Aldridge and Boyce 2008, Jenkins et al. 2019).

This study was initiated because of a lack of research on the effects of landcover characteristics and regulation changes on male wild turkeys and a specific lack of focus on the juvenile male age class. This project seeks to investigate factors related to juvenile male wild turkey survival, resources selected by this age class, and whether reductions in the bag limit and delay of the spring season impacted harvest of juvenile male turkeys. This project also seeks to investigate the factors related to adult male wild turkey harvest and predation-based mortality and whether harvest regulations, effort, fecundity, or weather impacted harvest of this age class. This study was designed to calculate survival and harvest rates for both juvenile and adult male wild turkeys to compare Tennessee to other states in the Southeast. This project aims to give state agencies and land managers the information that they need to manage the landscape to increase male turkey use and survival and to aid in the future regulatory framework used for managing harvest.

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APPENDIX A

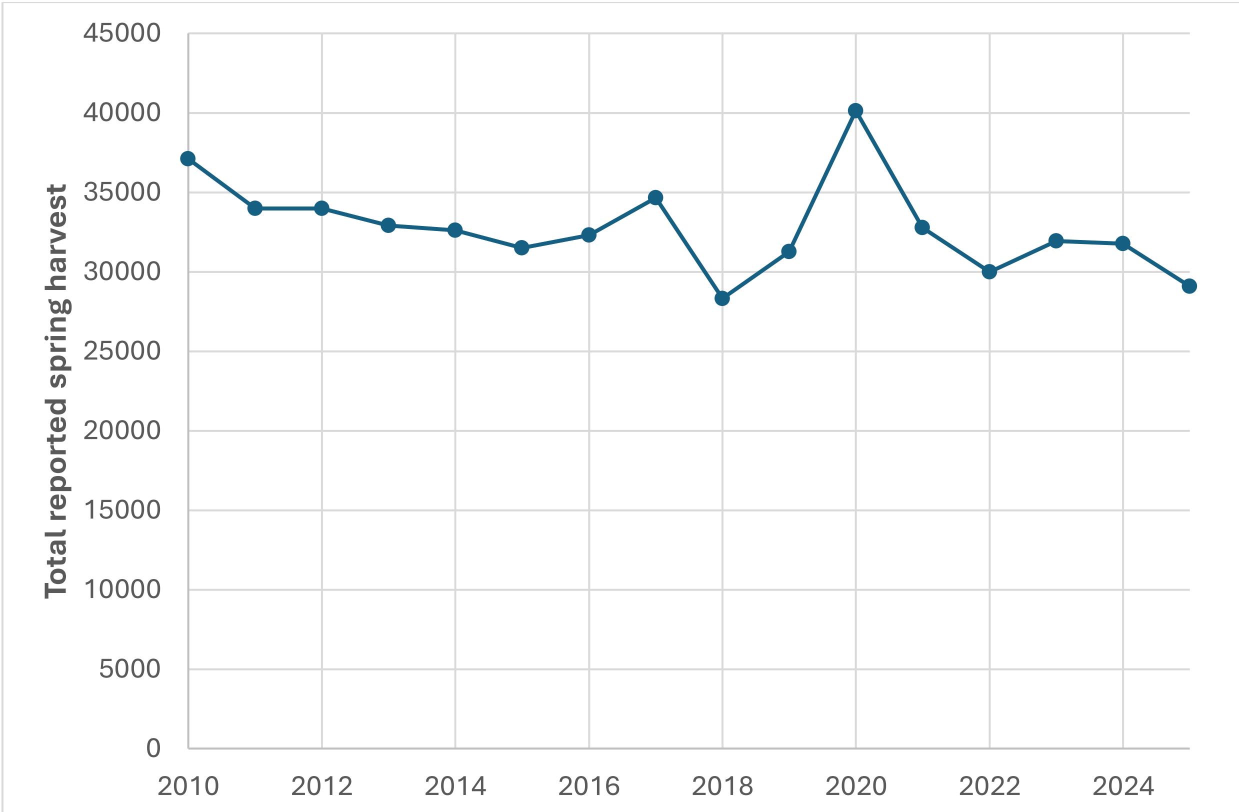

Figure A.1. Total reported spring wild turkey harvest in Tennessee, 2010–2025

CHAPTER 2. ANNUAL SURVIVAL AND RESOURCE SELECTION OF JUVENILE MALE WILD TURKEYS IN SOUTH-CENTRAL TENNESSEE

ABSTRACT

Documenting juvenile male wild turkey (*Meleagris gallopavo*) resource selection, survival, and harvest is important to better manage this life stage, which is the direct source for harvestable adult male wild turkeys. To better understand the relationship between resource selection and survival and how harvest impacts survival, we trapped and attached backpack VHF radio transmitters to juvenile males (n = 136) in five counties of south-central Tennessee, 2017–2021. We modeled resource selection using case-controlled conditional logistic regression and we fit known-fate survival models to the radio-tag data to estimate annual survival. We calculated landscape metrics and land cover proportions associated with each turkey's home range using the National Land Cover Database to determine if landscape characteristics were related to resource selection and annual survival. We also analyzed state-reported hunter harvest data using a generalized linear model to determine if changing harvest regulations affected harvest of juvenile males during the spring hunting season. Juvenile male wild turkeys selected smaller row crop fields, less mature forest interior, and lower proportions of developed land and young forest. Mean annual survival was 0.73 (95% CI = 0.65–0.81) with a harvest rate of 0.03 (95% CI = 0.01–0.09). Survival increased as hay/pasture interior increased in the home range, but this positive effect was reduced as the interspersion of young forest increased in the home range. Survival decreased as the interspersion of developed land increased in the home range, but this negative effect was reduced as the percentage of deciduous forest increased in the home range. The proportion of juvenile males in the spring harvest decreased from 14.1% to 13.7% to 8.9% as the male bag limit decreased from 4 to 3 to 2. Our results suggest that young forest, often associated with dense woody cover, may reduce survival as a result of increased ambush opportunities for wild turkey predators, whereas increased proportions of cover types that

provide enhanced visibility and food resources benefit juvenile male wild turkey survival. Given a relatively robust survival rate, regulatory restrictions on the harvest of juvenile male turkeys could increase the adult male population and likely increase turkey hunter satisfaction.

KEYWORDS harvest, juvenile male, *Meleagris gallopavo*, mortality, resource selection, survival, Tennessee, wild turkey

Eastern wild turkey (*Meleagris gallopavo silvestris*) population and harvest estimates have declined in much of the eastern United States since 2000 (Tapley et al. 2011, Chamberlain et al. 2022). Concomitant with population declines, wild turkey productivity in these states has declined (Byrne et al. 2015). In Tennessee, the poult-per-female ratio has declined since 2000 (Shields 2024). A recent (2018–2022) study reported poult survival estimates lower than many historical estimates (Quehl et al. 2025), and the relative abundance of wild turkeys has decreased by 43% from 2013 to 2022 according to the North American Breeding Bird Survey (United States Geological Survey 2022). Reported wild turkey harvest has declined in some counties of south-central Tennessee since 2005 (Buehler and Harper 2024), but the statewide harvest has remained relatively stable (Shields 2024). Collectively, these data suggest turkey harvest does not correlate precisely with turkey population trends, and therefore, these potential declines merit further investigation (United States Geological Survey 2022, Shields 2024).

Exact causes of declining wild turkey populations and harvest are unclear, but declines likely are influenced by various factors (Casalena et al. 2015, Chamberlain et al. 2022, Londe et al. 2023). Land-use change leading to an increase of human-developed areas has resulted in an increase of more than 3 million hectares of developed land in the southeastern US from 1997 to 2017, which is a 26% increase over that twenty-year period (United States Department of Agriculture 2020, World Population Review 2025). An increase in meso-mammalian predator abundance in the southeastern US since the 1980s is another factor likely contributing to declining wild turkey populations (Roberts and Crimmins 2010, Casalena et al. 2015, Bragina et al. 2019, Wang et al. 2023). The effect of increased predator populations on wild turkeys likely has been amplified by concomitant increased urbanization and increased forest cover across the southeastern US, which favors meso-mammalian predators and provides less visibility for wild turkeys (Kunkel 1997, Whittingham and Evans 2004, Pollentier et al. 2014, Ivey et al. 2024, Wang et al. 2025). Lastly, it has been hypothesized that more liberal hunting regulations following successful restoration in the 1970s and 1980s, such as increased bag limits, fall harvest of adult female turkeys, and lengthened spring hunting seasons that start too early and disrupt breeding may be contributing to population declines (Tapley et al. 2000, Norman et al. 2004, Londe et al. 2023, Quehl et al. 2024*a*). However, more conservative harvest frameworks, such as reducing bag limits and delaying the spring hunting season by as much as 14 days so as not to interfere with reproduction, had no effect on wild turkey productivity in Tennessee (Quehl et al. 2024*a*). Survival of juvenile males represents an important linkage between wild turkey reproduction and the availability of adult males for harvest (Palumbo et al. 2014). Obtaining survival and harvest estimates of juvenile male turkeys is important because these estimates enable state wildlife agencies to manage this cohort and influence subsequent availability of adult males (Humberg et al. 2009, Israelsen et al. 2020, Wightman et al. 2024).

In theory, juvenile male turkey survival and harvest rates should be inversely related unless harvest is strictly compensatory (Moore et al. 2008, Sandercock et al. 2011). Research from Georgia, Kentucky, Louisiana, Mississippi, Missouri, North Carolina, South Carolina, Virginia, and West Virginia indicates a generally inverse relationship: as juvenile male harvest increases, the survival rate generally decreases (Hubbard and Vangilder 2007, Wright and Vangilder 2007, Norman et al. 2022, Wightman et al. 2024). Given recent changes in bag limits in Tennessee, documentation of harvest and survival rates of juvenile male turkeys is warranted to further advance science-based harvest regulations.

Beyond just survival rates, little is known about resource selection and factors influencing survival for the juvenile male life stage. Investigation of home range and factors affecting resource selection and survival would provide wildlife managers with better information on how habitat management might help improve survival during this life stage. We conducted a radio telemetry study over 5 years in 5 counties of south-central Tennessee to study resource selection and how juvenile male survival is influenced by land cover type and other landscape metrics. Our research objectives were to (1) document juvenile male wild turkey resource selection of specific land cover types, (2) determine if juvenile male wild turkey survival is related to land cover type and other landscape metrics, and (3) document juvenile male wild turkey harvest proportions for our focal counties and determine if a delayed hunting season start-date and changes in bag limit affected juvenile male harvest. We hypothesized juvenile male wild turkeys would select land cover types that may provide greater potential for food and cover and less potential for predation risk, and juvenile males would avoid land cover types less likely to provide sufficient food and cover resources. We predicted turkeys would select smaller land cover patches if those patches were of land cover types that provide food or cover because such areas could provide better proximity to food and cover. We predicted juvenile male wild turkey survival would be positively related to home ranges with a greater Interspersion and Juxtaposition Index (IJI) of row crop and hay/pasture because of increased visibility and potential food resource availability. We further predicted survival would be negatively related to the proportion of developed land and land cover types with limited visibility for turkeys which may cause increased predation risk. Finally, we hypothesized the wild turkey hunting season framework would affect juvenile wild turkey harvest. We predicted a lower proportion of juvenile males would be harvested with a lower bag limit because more hunters would choose not to harvest a juvenile bird so as not to fill their seasonal bag limit before they harvest an adult. However, we predicted a 14-day delay in the spring hunting season would not affect juvenile male harvest.

STUDY AREA

We conducted research in Bedford, Giles, Lawrence, Maury, and Wayne counties in south-central Tennessee, 2017–2021. These five counties were chosen as a result of growing concern for wild turkey population declines in south-central Tennessee. These declines were supported by a 60% reduction in harvest from 2005–2015 in Giles, Lawrence, and Wayne county in conjunction with poult-per-female declines across the state (Buehler and Harper 2024, Quehl et al. 2024*a*, Shields 2024). We established north and south study sites within each county for trapping and monitoring turkeys (Figure B.1). Two of the ten study sites were on public land: Tie Camp WMA (1325 ha) in Wayne County and Yanahli WMA (5200 ha) in Maury County, whereas the remaining eight sites were on private land. Private land sites totaled >29,000 ha and included >380 individual landowners. The study sites were comprised of mature deciduous and coniferous forest, young forest (<10 years old), early successional plant communities, hayfields, pastures, row crops, and human-developed areas. The primary soil types for the study area were Bodine cherty silt loam and gravelly silt, Gladeville rock outcrop, Ashwood, Brandon silt loam, Biffle gravelly silt loam, and Frankstone cherty silt loam (United States Department of Agriculture 2019). Since 1989, the spring wild turkey season opened on the Saturday closest to April 1 (including 29–31 March), was six weeks in duration, and had a bag limit of four bearded turkeys. In 2021–2022, the spring turkey season was delayed 14 days (thereby shortened by two weeks) in Giles, Lawrence, and Wayne counties, and the bag limit was reduced to three bearded birds, of which only one could be a juvenile male. In 2023–2024, the season was delayed 14 days statewide but lengthened at the end of the season by two weeks for a total duration of six weeks, and the bag limit was reduced to two bearded birds, of which only one could be a juvenile male. Additionally, Giles, Lawrence, and Wayne counties were closed to fall turkey hunting starting in 2014, whereas Bedford and Maury counties have had a fall hunting season with a bag limit of one bearded turkey per county since 2016.

METHODS

Field methods

We trapped wild turkeys during winter, 2017–2021, using box-set rocket nets (Delahunt et al. 2011). We baited trap sites with shelled corn every other day to attract turkey flocks. We used trail cameras to monitor wild turkey visitation patterns, sex and age ratios, and flock sizes (Moultrie: Model MCG13202, Birmingham, Alabama, USA). We established a goal of maintaining a sample of 100 male turkeys each year across all five focal counties (10 males at each of the 10 sites, 5 adults and 5 juveniles). We excluded adult males from this study because we were interested in research questions specific to the juvenile cohort. We aged turkeys as juveniles based on uneven tail fan margins, primary feather barring, and leg spur length according to Pelham and Dickson (1992). We fitted each trapped juvenile male with a $75 reward aluminum leg band (National Band and Tag Company: style 1242FR8A, Newport, Kentucky, USA), assuming such a reward would encourage a greater return rate (Diefenbach et al. 2000). We attached an Advanced Telemetry Systems VHF transmitter to each juvenile male (Series A1500, ~80 grams, life expectancy of 5.7 years, Advanced Telemetry Systems, Isanti, MN). We recorded each male's weight, beard length, spur length, tarsus length, and body condition score (scoring the amount of fat and muscle found on the bird's keel; Gregory and Robins 1998). After processing, we immediately released each turkey at the trap site (University of Tennessee IACUC protocol #0561-0720). We radio-tracked wild turkeys weekly to monitor their mortality status and triangulated their location during each tracking event. When a turkey entered mortality mode as a result of a lack of activity for more than 8 hours, we retrieved the transmitter and investigated the cause and location of death as quickly as possible. We were able to investigate the majority of turkey mortalities within 2 days after mortality unless we were unable to obtain permission to access private property. Cause of mortality was determined as accurately as possible given the condition of the carcass and sign surrounding it. Great-horned owl (*Bubo virginianus*) predation was characterized by the removal of the head and breast muscle (Thogmartin and Schaeffer 2000, Little et al. 2016). Bobcat (*Lynx rufus*) predation was characterized by a cached or covered carcass (Thogmartin and Schaeffer 2000). Coyote (*Canis latrans*) predation was characterized by broken bones, scattering of remains, and presence of scat and hair (Thogmartin and Schaeffer 2000). If mammalian predation was apparent but we did not have enough evidence to attribute the mortality to a specific species, we categorized the mortality as an unknown mammal. Poaching was identified by cleanly cut transmitter straps with no evidence of predation. All harvested birds were reported via reward leg bands, and we identified any other mortalities as unknown. We acknowledge that the landcover classification associated with mortality may not be completely accurate as some carcasses could have been moved from the site of mortality to a different cover type.

Home range analysis

We calculated triangulated locations in the program LOAS using three intersecting bearings defined during VHF transmitter tracking events (Sallee 2005). We tracked turkeys weekly on average during all months of the year. We calculated our location error in LOAS using a 95% confidence interval ellipse polygon. We removed all locations from home range analysis whose error polygon was >10 ha, which resulted in a mean error of 1.4 ha/location. We created 90% minimum convex polygon (MCP) annual home ranges for turkeys using the adehabitatHR (version 0.4.22) package (Calenge 2006) in R version 4.4.1 (R Core Team 2026). We created home ranges for turkeys with ≥20 locations from their capture date to 12 months after capture. Any individuals that had <20 locations or did not survive for the entire year were excluded from the home range analysis. Even though this number of locations per bird may not capture the full extent of the home range of each turkey, a minimum of 20 locations was necessary to provide a large enough sample size for calculation. Furthermore, we only used the average home range size across all years of the study (2017–2021) in subsequent survival analysis, so basing the home range analysis on a minimum of 20 locations had little if any influence on our results.

Resource selection analysis

We analyzed resource selection of juvenile male wild turkeys using case-controlled conditional logistic regression with a use vs. availability framework to investigate which land cover types and other landscape metrics turkeys were selecting compared to what was available on the landscape (Johnson 1980, Manly et al. 2004). Locations of all turkeys were used regardless of how long they lived. This approach may not represent even distribution of selection over the year because approximately 25% of individuals died before completing the juvenile life stage. We also acknowledge analyzing resource selection using locations derived from VHF telemetry instead of GPS (Global Positioning System) limited the frequency of obtaining locations to approximately weekly per individual rather than multiple daily locations and introduced greater location error, which can limit our inference and potentially introduce some bias because there are fewer locations for each individual (Hebblewhite and Haydon 2010, Latham et al. 2015). However, because VHF radio-tags are less expensive per unit and have much greater battery life compared to GPS units, our sample size of individual turkeys and the length of our study were greater than many GPS studies, enabling more robust population-level inferences about selection (Hebblewhite and Haydon 2010, Latham et al. 2015). We buffered each triangulated location by 138 m to create a 6-ha polygon around each used location using ArcGIS Pro version 3.3.1 (Environmental Systems Research Institute, Redlands, California, USA). We obtained an average of ~30 locations per turkey for resource selection. We chose a 6-ha buffer size because it enabled us to assess the composition of the land cover types they were using as opposed to the cover type they were located in during the time of the location. The buffer size also was larger than the average location error (1.4 ha; Erickson et al. 1998), which allowed the location estimate and its error buffer to shift in any direction by a radius length of the error polygon while remaining inside the 6-ha buffer. Six ha was slightly larger than the average size of all cover type patches (5.5 ha), which enabled our scale of selection to match the landscape and facilitate inference at a patch scale. We clipped the land cover types for each polygon from the U.S. Geological Survey (USGS) National Land Cover Database (NLCD) 30-meter spatial resolution Landsat land cover map of North America using ArcGIS Pro (United States Geological Survey 2024). We then calculated landscape metrics associated with individual locations for the resource selection analyses using the landscapemetrics (version 2.2.1) package (Hesselbarth et al. 2019) in R version 4.4.1 (R Core Team 2026). These metrics included forest edge density, mean patch size, percentage of land cover by type, and percentage cover type interior. We created available, non-overlapping polygons identical in size to the associated use polygons by randomly locating 10 points 0–3000 m from the center of each use polygon in accordance with the design III protocol reported in Manly et al. (2004). We defined the area for placement of these available buffers based on a distance of 3000 m, which represented the average daily travel distance for a male turkey from other recent studies (Gross 2015, Chamberlain et al. 2018, Cohen et al. 2022). We chose to use a 3000 m extent instead of our average home range size to better characterize what was available to the turkeys because home ranges may underestimate what was truly available (Walter et al. 2015, Niedzielski and Bowman 2016, Skupien et al. 2016, Cohen et al. 2018). Although we cannot know what was truly available for each individual turkey, this approach defined a broad landscape that individual turkeys could use given their daily movements. We then calculated the same four landscape metrics for each available polygon that were applied to the used polygons and averaged the values for the ten random polygons associated with a single-use polygon (Figure B.2).

We calculated landscape metrics based on composition and structural features we thought could be important to turkey resource selection based on the hypotheses we were testing and based on the literature. Forest edge density was defined as the meters of forest edge (where forest and another cover type meet) per ha within a 6-ha buffer (McGarigal 1995). Mean patch size was defined as the sum of the areas of all patches of a given patch type within a given buffer, divided by the number of patches of the same type. Since calculating patch size using a 6-ha buffer would truncate some patches that extended beyond the boundaries of the buffer, we included all patches that were fully within or intersected the buffer. Thus, we considered the full size of all patches even if they extended beyond the limits of the 6-ha buffer. The percentage of land cover by type was calculated by dividing the number of pixels of a given land cover type by the total number of pixels within the 6-ha buffer. The percentage of cover type interior was defined as the square meters of a particular land cover type that were ≥120 m in all four cardinal directions from a patch edge divided by the total buffer area (McGarigal 1995). We chose 120 m to align with previous studies and to represent a maximum preferred foraging distance from edges

(Schroeder 1985, Glennon and Porter 1999, Fleming and Porter 2000). The formula used to calculate cover type interior was:

$$CPLAND = \left(\frac{\sum_{j=1}^{n} a_{ij}^{core}}{A}\right) * 100 \qquad (1)$$

Where a_{ij}^{core} was the cover type interior in square meters, and A was the total landscape area in square meters. When calculating cover type interior for a 6-ha buffer, we applied an additional 120 m outside the buffer to ensure the boundary of the 6-ha buffer was not artificially considered an edge. This method enabled proper identification of the entire 6-ha buffer as eligible for cover type interior consideration (Figure B.3). All landscape metrics were calculated from land cover types defined by the NLCD map which included open water, coniferous forest, deciduous forest, young forest, early succession, hay/pasture, row crop, woody wetland, emergent herbaceous wetland, developed land, and barren land. We combined open water, wetland, and barren land into a singular land cover type called nonhabitat. The developed land cover type was represented by roads, individual residential homes, and farmsteads.

We used the proportion of the 6-ha buffer composed of deciduous forest, coniferous forest, all mature forest, young forest, early succession, hay/pasture, row crop, developed land, and nonhabitat as covariates in the resource selection analysis (Table B.1). All forested and open land cover proportion covariates were included in the analysis to determine if turkeys selected different cover types or if they behaved as generalists, as documented in previous studies (Hurst 1992, Marable et al. 2012, Pollentier et al. 2017). Developed land and nonhabitat proportions were included to determine if turkeys would avoid areas that provided little food and cover and possibly presented a mortality risk. For edge density, we only calculated this metric for mature forest land cover (combined deciduous and coniferous forest) to specifically test if turkeys selected areas directly adjacent to forest edges to be closer in proximity to different cover types that provide food and cover (Davis et al. 2018, Pollentier et al. 2021; Table B.1) To determine if turkeys were avoiding larger patches of discrete and managed land cover, we calculated the mean patch size for row crop and coniferous cover types. We used a subset of the data for this particular landscape metric by ignoring any instances when a turkey did not use one of these particular land cover types because we were interested in the mean patch size only when a turkey specifically used a row crop or coniferous patch. The patch size metric was specifically chosen for row crop and coniferous cover types because they are most often spatially discrete and isolated, so that patch size reflected the true spatial extent of that cover type. Mature forest and hay/pasture were the most abundant cover types in the landscape matrix of south-central Tennessee (Rodewald 2003, Driscoll et al. 2013). We used the cover type interior metric rather than patch size for mature forest and hay/pasture because these cover types are frequently connected by narrow corridors resulting in patches perforated by edges resulting in an inflated patch size that does not properly represent a larger uninterrupted management unit (Hargis et al. 1998; Table B.1). Land cover metrics related to size or interior structure were not calculated for cover types such as early succession, young forest, nonhabitat, and developed land since these cover types were consistently represented by small patches and attempting to calculate size or interior would be uninformative. We evaluated multicollinearity among continuous covariates using VIF (variance inflation factor) from the car (version 3.1-3) package (Fox and Weisberg 2019) in R version 4.4.1 by using a threshold of 5 and removing any models that exceeded that threshold (Shrestha 2020, R Core Team 2026). We standardized all continuous covariates by z-transforming each to enable direct comparison of beta coefficient effect sizes (Quinn and Keough 2002, Zuur et al. 2007). We created univariate and multi-covariate case-controlled conditional logistic regression models clustered by turkey ID to calculate robust standard errors using the clogit function from the survival (version 3.8-3) package (Therneau et al. 2024) in R version 4.4.1 (Johnson et al. 2022, R Core Team 2026). To develop the model set, we first tested covariates univariately to examine individual effects on selection and to inform subsequent modeling (Bursac et al. 2008). We developed a confirmatory model set of univariate and multivariate models that tested our specific hypotheses, based on the literature or developed from our field experience. We used the Akaike's Information Criterion corrected for small sample size (AICc) on these models to select the best-supported model for the resource selection analysis. We considered all models with ΔAICc < 2 as part of our family of best models and interpreted results from these models. The patch size covariate models could not be directly compared with the other resource selection models because patch size models only included data if a given patch type (e.g., row crop and coniferous forest) was present, leading to differential sample sizes. Instead, we evaluated the univariate patch size models independently and determined the results based on effect size and beta coefficient confidence intervals. We considered covariates from both analyses influential if their beta coefficient estimates had 85% confidence intervals that did not include zero, following principles of AIC model selection and multi-model inference (Arnold 2010, Tan and Tan 2010). We emphasize that confidence intervals were used as heuristic filters rather than as formal hypothesis tests. This inferential approach does not imply statistical significance in the traditional null hypothesis-testing framework but rather is interpreted as indicating support for an effect given the data and model structure (Burnham and Anderson 2002).

Survival analysis

From 2017–2021, we monitored the weekly mortality status of each of the juvenile male wild turkeys until they either died, were recruited to the adult age class, or we lost contact with their radio signal. We recorded the land cover type at the mortality site for each carcass and the apparent cause of death for each mortality. We considered juvenile turkeys to be any male that was in juvenile plumage at capture during their first winter until March 23rd of the subsequent year. March 23rd was arbitrarily chosen as the end point of this age class because March 23rd was the earliest opening date of the spring youth turkey hunt in any year of the study. We acknowledge that our classification of juveniles includes a time period after each turkey's second autumn when juveniles transition to the adult age class according to feather molting patterns (Pelham and Dickson 1992). However, since they are not harvested as an adult male until at least the following March, extending the length of this age class helps answer crucial questions about turkey survival before the probability of mortality from harvest substantially increases and will better describe annual survival by including all months of the year. We estimated survival using the Kaplan-Meier known-fate survival model calculated using the RMark (version 3.0.0) package (Laake 2013) in R version 4.4.1 (Kaplan and Meier 1958, Pollock et al. 1989, White and Burnham 1999, R Core Team 2026). We used a staggered-entry design to allow for additional turkeys to be added to the analysis after the study had begun (Pollock et al. 1989). We removed individuals from the study with mortalities within 14 days of capture to avoid including turkeys that may have experienced capture myopathy (Nicholson et al. 2000). We right-censored all turkeys that we could not detect with telemetry after the last successful tracking occasion. Tracking occasions were defined as biweekly because the average time in between telemetry locating attempts was one week. The biweekly survival estimate, averaged across the entire year, was raised to the 26th power to calculate annual survival estimates. The harvest rate was calculated by right-censoring all non-harvest mortalities and subtracting the resulting survival rate from 1 (Norman et al. 2022). We included multiple covariates in the survival model to evaluate the influence of different land cover types and landscape metrics on juvenile male wild turkey survival. All birds whose mortality was attributed to hunter harvest were removed from models investigating the influence of landscape metrics on survival to isolate the effect of landscape on non-harvest mortality. The first set of models included the discrete factor variables year and county. The second set of models included landscape covariates calculated using the landscapemetrics (version 2.2.1) package (Hesselbarth et al. 2019) in R version 4.4.1 for all turkeys by using identical circular polygons equal to the mean annual home range size of all turkeys that lived 12 months or more, placed over the centroid of each turkey's estimated locations (R Core Team 2026). We chose to use average home range polygons to extract each turkey's landscape covariates since only a subset of turkeys that had lived at least 12 months would have a MCP home range produced. To justify the use of average home ranges, we created a generalized linear mixed-effects model with a gaussian error distribution using the glmmTMB function from the glmmTMB (version 1.1.14) package (Brooks et al. 2017) in R version 4.4.1 to show that there was no significant difference between the landscape covariates for MCP and average home ranges ($Z = -0.05$, $P = 0.96$; R Core Team 2026). We used different buffer sizes for the survival analysis compared to resource selection because we wanted to model survival at the home-range scale to understand how used land cover influenced survival, whereas a 3000 m buffer in the resource selection analysis represented the full extent of available land cover. We included the proportion of the home range comprised of deciduous forest, coniferous forest, all mature forest combined, young forest, early succession, hay/pasture, row crop, developed land, and nonhabitat in the survival models to determine how different amounts of land cover were related to survival probability as it related to visibility or predator abundance (Miller and Leopold 1992, Whittingham and Evans 2004). Mature forest edge density was included to determine if the total length of woody edges was related to survival by increasing predator success as a result of decreased visibility along forest edges (Šálek et al. 2010, Wang et al. 2025; Table B.1). We included the percentage of mature forest and hay/pasture interior, consistent with the resource selection analysis, to determine if large uninterrupted areas of open or forested cover would be related to survival by affecting visibility for wild turkeys (Table B.1). Lastly, we included the interspersion and juxtaposition index (IJI) of deciduous forest, coniferous forest, young forest, early succession, hay/pasture, row crop, nonhabitat, and developed land to determine if the impact of land cover was more dependent on the configuration and distribution in the home range instead of the total amount present (Mimet et al. 2016, Gengler et al. 2024; Table B.1). IJI was defined as the observed interspersion of land cover type classes over the maximum possible interspersion for the given number of land cover type classes (McGarigal 1995). The equation used to calculate IJI for each land cover type was:

$$IJI = \frac{-\sum_{k=1}^{m}\left[\left(\frac{e_{ik}}{\sum_{k=1}^{m} e_{ik}}\right) ln\left(\frac{e_{ik}}{\sum_{k=1}^{m} e_{ik}}\right)\right]}{ln(m-1)} * 100 \qquad (2)$$

Where e_{ik} are the unique adjacencies of a particular land cover type class in meters, and m is the number of all classes present (McGarigal 1995). IJI values are defined between 0 and 100 with a lower value indicating a landscape with land cover types that are more poorly interspersed (McGarigal 1995). We z-transformed all covariates and tested them for multicollinearity using VIF, similar to the resource selection analysis. We developed a model set for evaluation by first testing covariates univariately to examine individual effects on survival and to inform subsequent modeling (Bursac et al. 2008). We then created multivariate models by including covariates that tested our specific hypotheses based on the literature or developed from field experience. Beta coefficient estimates were obtained from all models with ΔAICc < 2. We used an 85% confidence interval threshold for beta coefficients to determine influential covariates similar to the resource selection analysis (Arnold 2010, Tan and Tan 2010). However, to remain consistent and comparable to the literature, we report estimates of means—annual survival, annual harvest, and home range size—with 95% confidence intervals and associated standard errors.

Harvest analysis

Because radio-tracking and trapping of juvenile male wild turkeys ceased in 2022, we were unable to analyze the effect that changes in harvest regulations after 2021 had on our survival or harvest. We obtained spring harvest data from the Tennessee Wildlife Resources Agency (R. Shields, unpublished data) for Bedford, Giles, Lawrence, Maury, and Wayne counties for 2017–2024 to assess the impact of regulation changes on juvenile male turkeys. We calculated the total reported spring season harvest of juvenile and adult males from the annual harvest data by county and year. We then used the glm function with a binomial error distribution and logit link function in the R version 4.4.1 stats package to create generalized linear models to determine if the proportion of juvenile male turkeys in the harvest differed by county and year (R Core Team 2026). We created two sets of models, one with year and county variables and one with harvest regulation variables (spring delay, bag limit). We used AICc to select the best-fitted model for each model set. We used the Anova function from the car package to perform likelihood ratio chi-square tests to detect the overall effect of year and county variables on harvest proportion. We used the summary function from the stats package to perform Wald z-tests to detect the individual effect of the changes in bag limit and 14-day delay. We used the plot function from the stats package to inspect the residuals vs fitted values, verify that the deviance residuals were normally distributed, and identify influential observations. For this analysis, we set $\alpha = 0.05$ for interpretation of results and used odds ratios (OR) to explain the effect size.

RESULTS

Trapping and home ranges

We trapped and banded 173 juvenile male turkeys from 2017–2021 across all five focal counties. We radio-tagged 148 juvenile male turkeys with backpack VHF transmitters. We included 136 turkeys in the resource selection and survival analysis (Wayne = 12, Lawrence = 30, Maury = 40, Bedford = 20, Giles = 34). Of the 136 turkeys monitored, 74 produced enough locations to create MCP home ranges. The mean number of locations/home range was 44.1 (± 1.7 SE, 95% CI = 40.8, 47.4). We excluded 431 locations from the resource selection analysis because they had error polygons greater than 10 ha. The mean annual home range size was 352.3 ha (±27.3 SE, 95% CI = 297.8, 404.7).

Resource selection

The monitoring protocol produced 3,860 used locations with a mean of 28.4 locations per individual (±1.4 SE; 95% CI = 25.8–31.2) and 38,600 random available locations (10 random locations for each used location). We created 14 univariate and 41 multivariate models. The best-supported model contained the proportion of developed land, young forest, and mature forest interior (Table B.2). All other models in the model set had $\Delta AICc > 2$ and therefore were not considered. Based on the best-supported model, turkeys selected lower proportions of developed land ($\beta = -0.43$; 85% CI = [-0.55, -0.31]; Figure B.4), such that a 10% increase in developed land decreased the odds of use by 48%. Turkeys selected lower proportions of mature forest interior ($\beta = -0.35$; 85% CI = [-0.42, -0.28]; Figure B.4), such that a 10% increase in mature forest interior decreased the odds of use by 41%. Turkeys selected lower proportions of young forest (β = -0.20; 85% CI = [-0.37, -0.04]), such that a 10% increase in young forest decreased the odds of use by 50%. Out of the two separate models using the mean patch size data sets, only the model containing the mean patch size of row crop showed a relationship with selection (β = -0.31; 85% CI = [-0.55, -0.07]; Figure B.4), indicating turkeys were selecting smaller patches of row crops. For each 50-ha increase of the mean patch size of row crops, the odds of use decreased by 37%.

Survival

We documented 32 total mortalities of 136 juvenile males monitored (24%). Apparent causes of mortality included coyote (*Canis latrans*; n = 15, 47% of mortalities), bobcat (*Lynx* rufus; n = 1, 3%), unknown mammal (n = 7, 22%), harvest (n = 3, 9%), poaching (n = 2, 6%), and unknown (n = 4, 13%). In total, 23 of 32 mortalities (72%) were attributed to predators. Of the 29 non-harvest mortalities, 28 carcass recovery locations were identified as follows: deciduous forest (n = 25, 89%) and hay/pasture (n = 3, 11%) cover types. We were unable to gain access to the private land location of one of the mortalities.

The full set of models included continuous landscape covariates (1 null, 25 univariate, 32 multivariate models; Table B.3). Of the models we fit, two had ΔAICc < 2 and were considered part of the family of best models. The best-supported model included hay/pasture interior, the interspersion of young forest, and the interaction between these covariates. This model demonstrated that an increase in hay/pasture interior increased survival (β = 1.47; 85% CI = [0.41, 2.54]) and an increase in the interspersion of young forest reduced survival (β = -0.82; 85% CI = [-1.37, -0.27]). However, the significant negative interaction between these two covariates (β = -1.61; 85% CI = [-2.77, -0.46]; Figure B.5) indicated that the positive effect that hay/pasture interior had on survival was reduced as the interspersion of young forest increased.

The second best-supported model included the proportion of deciduous forest, the interspersion of developed land, and the interaction between these covariates. This model demonstrated that an increase in the proportion of deciduous forest (β = -0.44; 85% CI = [-0.81, -0.07]) and an increase in the interspersion of developed land (β = -0.52; 85% CI = [-0.87, -0.16] reduced survival. However, the significant positive interaction between these two covariates indicated that the negative effect of developed land on survival was reduced as deciduous forest increased (β = 0.43; 85% CI = [0.10, 0.76]; Figure B.5). When fitting models with year and county, the model including county provided the best fit. Annual survival rates varied by county, as survival rates in Bedford and Maury counties were greater than survival rates in Giles, Lawrence, and Wayne counties (Table B.4). The null model with no covariates yielded an average annual survival rate of 0.73 (±0.04 SE; 95% CI = 0.65–0.81) with a harvest rate of 0.03 (±0.02 SE; 95% CI = 0.01–0.09).

Harvest

The proportion of juvenile male turkeys in the spring harvest averaged across all years and counties combined was 13.1% (SE = 0.7), with n = 40 year/county combinations. The proportions varied by year (LR χ^2 = 200.38; df = 7; $P \leq 0.001$; Figure B.6) as follows: 16.4% (2017), 10.0% (2018), 14.2% (2019), 15.2% (2020), 11.3% (2021), 16.4% (2022), 10.2% (2023), 7.7% (2024). The proportions varied by county (LR χ^2 = 72.06; df = 4; $P \leq 0.001$) as follows: 11.8% (Bedford), 14.5% (Giles), 17.0% (Lawrence), 12.2% (Maury), 10.0% (Wayne; Figure B.6). When comparing the effect of harvest regulations on the proportion of juvenile males in the harvest, the additive model containing the 14-day spring delay and bag limit as covariates had the lowest AICc score. In the five focal counties, the proportion of spring harvest comprised of juvenile male wild turkeys was 14.1% during a four-bird bag limit, 13.7% during a three-bird bag limit, and 8.9% during a two-bird bag limit. The proportion of juveniles did not differ when the bag limit was reduced from four birds to three birds (OR = 0.92; Z = -1.47; $P = 0.141$), but the odds of juvenile harvest decreased by 43% when the bag limit was reduced from three birds to two birds (OR = 0.57; Z = -7.34; $P \leq 0.001$), and the odds also decreased by 48% between the four-bird and two-bird bag limits (OR = 0.52; Z = -6.74; $P \leq 0.001$). There was a negligible increase in the proportion of juvenile males harvested during the 14-day spring delay (OR = 1.14, Z = 1.62; $P = 0.105$).

DISCUSSION

We documented previously unreported relationships of land cover on resource selection and annual survival as well as relationships of hunting regulations on juvenile male turkey harvest, which should better inform wild turkey management in Tennessee. Larger patch sizes of row crop, greater mature forest interior, and greater proportions of developed land along with young forest in the home range all were avoided by juvenile males. Survival increased as hay/pasture interior increased, but this positive effect was reduced as the interspersion of young forest increased in the home range. Survival decreased as the interspersion of developed land increased, but this negative effect was reduced as the percentage of deciduous forest increased in the home range. These results may be used to adjust land management for wild turkeys whereby the patch size of forests and fields may be reduced to increase interspersion and increased frequency of disturbance could reduce patches of dense woody cover that can conceal mammalian predators from turkeys (Gorini et al. 2012, Turner et al. 2024, Bones et al. 2026). Overall, survival of juvenile male turkeys was lower than survival in most contemporary studies in the southeastern US, but juvenile male survival could increase if regulations did not allow harvest of this age class. The proportion of juvenile males in the harvest approached 20% in some counties.

Approximately 3 out of every 4 juvenile males not harvested during their first spring season could be passed on to increase the adult male population available for hunting the following spring.

The selection for smaller row crop patches by juvenile male wild turkeys supported our first hypothesis and aligns with the fact that smaller fields when dispersed across the landscape provide better interspersion of food and cover (Porter 1992, Glennon and Porter 1999, Kollmann and Schneider 1999, Davis et al. 2018). Wild turkeys use hardwood forests for mast, thermal cover during summer, and roosting (Porter 1992, Steffen et al. 2002). Adjacent cropland provides additional food resources, such as invertebrates, agricultural foods, and seeds, especially when cropland patches are smaller (Hurst and Dickson 1992, Paisley et al. 1996, Otieno and Frenette 2017, Pollentier et al. 2021). The selection of less forest interior also indicated selection for a more interspersed landscape with smaller forest and field patches.

The lack of importance or selection of forest edge was noteworthy. Amount of edge historically has been considered desirable for various gamebird species (Leopold 1933, Glennon and Porter 1999, Matlack and Litvaitis 1999, Schindler et al. 2020). However, our results highlight how edge, defined by Bolen and Robinson (2002) as the interface of two or more vegetation types or seral stages, is not necessarily a determinant of selection or positively related to survival by juvenile male wild turkeys. Instead, resource availability, size and shape of the adjacent vegetation type patches, and the structure of vegetation, which affects visibility, should be considered reasons for selection as has been demonstrated with other game species such as northern bobwhite (*Colinus virginianus*; Guthery and Bingham 1992, Guthery 1997, Guthery et al. 2001). For example, the edge created by young forest adjoining mature forest could potentially present increased predation risk with reduced visibility for wild turkeys in relatively dense young forest cover. Thus, juvenile male turkeys would not select such areas with young forest-mature forest edges.

The avoidance of developed land where mortality risk is greater and food resources are presumably less abundant was expected and supported our hypothesis. Urbanization is one of several factors that has led to declining wild turkey populations in some areas, including loss of habitat, collisions with vehicles, and increased predation pressure (Stark et al. 2020, United States Department of Agriculture 2020, Cohen et al. 2022, Dawson et al. 2025). When developed land was poorly interspersed in the home range, deciduous forest caused reductions in survival perhaps as a result of greater predator densities and less visibility (Lagory 1986, Grossman et al. 2008, Gorini et al. 2012, Jones et al. 2022; Figure B.5). However, deciduous forest mitigated the negative effect of high interspersion of developed land on survival, most likely as a result of increased usable space with food and cover (Collins et al. 2026; Figure B.5). The south-central Tennessee region has developed rapidly over the past 30 years, expanding outward from the Nashville metro area (World Population Review 2025). If the resource selection and survival relationships for juvenile males apply similarly to other wild turkey cohorts, the regional increase in human development likely is contributing to the overall decline of regional wild turkey populations. Increased wild turkey conflicts with humans in recent years has caused some to speculate that turkeys can thrive in developed areas, and it is even proposed that some wildlife species will take refuge near human habitation which could act as a shield to discourage predators (Gregonis et al. 2010, Gaynor et al. 2025). However, it is important to note that this Human Shield Hypothesis is highly context-dependent, and it has even been demonstrated that two of the most common wild turkey predators—the coyote and bobcat—will actively seek out developed areas rather than avoid them because of human interactions (Prugh et al. 2023).

Although our study sites were predominantly exurban or rural, wild turkeys that inhabit more urbanized areas still select areas farther from roads and closer to herbaceous cover according to other studies (Collins et al. 2026).

The benefit of hay/pasture interior on juvenile male survival was greatly reduced as the interspersion of young forest cover reached greater levels, which supports our hypothesis related to increased predation risk where visibility for turkeys is limited (Figure B.5; Whittingham and Evans 2004, Gorini et al. 2012). Meso-mammalian predators were responsible for most of the wild turkey mortalities we documented, and several species including coyote, bobcat, and foxes may concentrate their search for prey along the edge of forest cover, which is consistent with the negative effect young forest cover had on survival (Sunquist and Sunquist 1989, Grossman et al. 2008, Deuel et al. 2017, Jones et al. 2022, Webster et al. 2022). The negative effect of deciduous forest on survival when the interspersion of developed land was lower also was supported by the large majority (89%) of the mortalities we recorded in deciduous forest when only 57% of the landscape was deciduous forest. These results further support the concept of visibility being a key limiting factor in wild turkey survival, especially as we documented greater survival in home ranges with a greater percentage of hay/pasture interior when the interspersion of young forest was low. It is important to note, however, that any benefit of hayfields for juvenile males can have reduced benefit for female turkeys when haying occurs during incubation and leads to nest destruction or abandonment (Tyl et al. 2023).

Our estimated annual survival rate of juvenile male wild turkeys was greater than that reported in West Virginia, Mississippi, the North Carolina Piedmont, the Missouri Ozarks South Study Area, the Missouri Ozarks Peck Ranch Conservation Area, and Kentucky (Wieme 2001, Hubbard and Vangilder 2007, Wright and Vangilder 2007, Norman et al. 2022, Wightman et al.

2024; Table B.5). However, most of these rates were from studies that occurred more than 20 years ago, and many of the spring harvest rates were much greater than ours (Table B.5). Reported juvenile male survival rates for more contemporary studies in coastal North Carolina, Virginia, Webb South Carolina, East Louisiana, North Carolina mountains, West Louisiana, and Georgia all were greater than ours, even with some of those locations having a greater spring harvest of juvenile males (Table B.5; Norman et al. 2022, Wightman et al. 2024). Most of the juvenile male mortalities from our study (72%) were attributed to predators compared to 28%, 33%, and 53% of mortalities in three other southeastern studies (Wieme 2001, Wright and Vangilder 2007, Wightman et al. 2024). Importantly, the predator mortality proportions from Mississippi (28%) and Kentucky (33%) were from studies that occurred more than two decades ago when predator populations were likely much lower and harvest rates for juvenile males were greater (Roberts and Crimmins 2010, Hody and Kays 2018). We documented >7 times greater predation-based mortality than harvest-based mortality (7:1 ratio), which is much greater than ratios reported from several recent southeastern studies (1.5:1- Webb SC, 4:1- Coastal NC, 1:5- GA, 1:7- West LA; Wightman et al. 2024).

The decline in the proportion of juvenile males harvested after a two-bird bag limit was implemented likely was caused by changes in hunter behavior, in support of our hypothesis. Even though most hunters in the focal counties harvested 0 or 1 bird per season, the lower bag limit likely caused hunters to pass on harvesting a juvenile male bird over fear of filling their seasonal bag limit before they could harvest a second adult male turkey (Quehl et al. 2024*b*). Although the proportion of juvenile males in the harvest approached significance during the 14-day spring delay ($P = 0.105$), the behavioral shift of hunters was more strongly influenced by bag limit than the timing of the hunting season. We were not able to account for changes in reproduction or hunter effort when analyzing the effects of harvest regulations, and we acknowledge those factors also may influence juvenile harvest.

Rather than relying on modest effects of a lower bag limit on reducing the number of juvenile male birds harvested, eliminating the harvest of juvenile males directly could have a great effect on the availability of adult males. For example, when Mississippi eliminated juvenile male harvest, the average proportion of ≥3-year-old males in the harvest increased from 41% to 48% and gobbles heard per hour also increased (Leopold and Cummins 2016). The harvest of adult male wild turkeys/100 hours in Arkansas increased by nearly 2.5x in 2014, the year after a new regulation prohibited harvest of juveniles, and adult harvest has remained 1.5x greater than the 7-year pre-ban average (Moscicki 2024). Our survival results demonstrate that eliminating juvenile males from harvest could pass 3 of 4 juvenile males to the adult age class, which would be available for harvest the following spring. Although the harvest rate of radio-tagged juvenile males on our study sites was low, juvenile males represented ≥10% of the total spring harvest in our study site counties and approached 20% in some years (Figure B.6). Therefore, limiting harvest of juveniles likely would lead to increased numbers of adult males available for harvest and ultimately increase hunter satisfaction.

MANAGEMENT IMPLICATIONS

A relatively robust survival rate among juvenile males allows predictable recruitment of adult males into the population the following spring. Any management that would increase juvenile male survival could translate directly to an increase in adult male availability for the hunting season, which would translate to increased hunter satisfaction since multiple wild turkey hunter surveys have identified gobbling activity as the primary determinant of hunter satisfaction. Considering the survival rate for juvenile males, regulations and voluntary action of hunters that limit harvest of juvenile males to youth hunters only, similar to regulations implemented in Arkansas, Mississippi, South Carolina, and Louisiana, could lead to increased adult males available for harvest. Mammalian predators were responsible for a majority of wild turkey mortality. Implementing some level of control on these predators may be warranted in situations where mortality is relatively high and limiting fecundity and recruitment into the population. Habitat management also can influence wild turkey survival. Practices that provide more open structure and increase wild turkey visibility, and decrease large homogenous patches of land cover, such as forest clearing and thinning coupled with frequent prescribed fire, should be conducted on properties where the objective is to increase the wild turkey population.

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APPENDIX B

Table B.1. Description of landscape metrics calculated for juvenile male wild turkey resource selection and known-fate survival analysis, south-central Tennessee, 2017–2021.

Metric name	Land cover types used	Definition
---------------------------------------*Resource selection analysis*---------------------------------------		
Land cover proportion	Deciduous forest, coniferous forest, mature forest, young forest, early succession, hay/pasture, row crop, developed, non-habitat	% of the 6-ha buffer composed of a given land cover type
Patch size	Coniferous forest, row crop	Mean patch size (ha) of the given land cover type that is within or intersected by the 6-ha buffer
Cover type interior	Mature forest, hay/pasture	% of the 6-ha buffer for a given land cover type that is ≥ 120 m from a land cover edge
Edge density	Mature forest	m/ha of edge for a given land cover type within the 6-ha buffer
--*Survival analysis*--		
Land cover proportion	Deciduous forest, coniferous forest, mature forest, young forest, early succession, hay/pasture, row crop, developed, non-habitat	% of the home range composed of a given land cover type
Cover type interior	Mature forest, hay/pasture	% of the home range for a given land cover type that is ≥ 120 m from a land cover edge
Edge density	Mature forest	m/ha of edge for a given land cover type within the home range
Interspersion and juxtaposition index	Deciduous forest, coniferous forest, mature forest, young forest, early succession, hay/pasture, row crop, developed, non-habitat	Interspersion and juxtaposition of a given land cover type within the home range

Table B.2. Top 10 models for AICc model selection results for juvenile male wild turkey case-controlled resource selection analysis, south-central Tennessee, 2017–2021. We present the number of parameters (K), Corrected Akaike Information Criterion (AICc), the difference between model rankings (ΔAICc), AICc model weight (AICcWt), and model deviance (Dev).

Model	K	AICc	ΔAICc	AICcWt	Dev
DEV + YGF + FORINT	3	4934.89	0.00	1.00	4928.89
DEV + FORINT + HAY	3	4950.98	16.09	0.00	4944.98
DEV + FORINT	2	4972.65	37.76	0.00	4968.65
DEV + EDGE + FORINT	3	4973.40	38.51	0.00	4967.40
DEC + DEV + YGF + HAY	4	4990.02	55.13	0.00	4982.02
DEC + DEV + HAY	3	5005.42	70.53	0.00	4999.41
FOR + DEV	2	5025.64	90.75	0.00	5021.64
DEC + DEV + EDGE	3	5062.69	127.80	0.00	5056.69
DEC + DEV + YGF	3	5064.59	129.70	0.00	5058.59
DEV + YGF + EDGE	3	5067.89	133.00	0.00	5061.88

[a] DEC = proportion of deciduous forest, YGF = proportion of young forest, DEV = proportion of developed land, EDGE = forest edge density, FORINT = proportion of forest interior, HAYINT = proportion of hay/pasture interior, HAY = proportion of hay/pasture

Table B.3. AICc model selection results for juvenile male wild turkey known-fate survival analysis, south-central Tennessee, 2017–2021. Only models that outperformed the NULL (no covariates) are shown. We present the number of parameters (K), Corrected Akaike Information Criterion (AICc), the difference between model rankings (ΔAICc), AICc model weight (AICcWt), and model deviance (Dev).

Model[a]	K	AICc	ΔAICc	AICcWt	Dev
HAYINT * IJIYGF	4	316.64	0.00	0.16	308.62
DEC * IJIDEV	4	317.34	0.70	0.11	309.33
IJIDEV	2	318.89	2.25	0.05	314.88
HAYINT	2	318.99	2.36	0.05	314.99
DEC + IJIDEV	3	319.30	2.66	0.04	313.29
HAYINT * IJIDEV	4	319.36	2.72	0.04	311.34
NON + IJIDEV	3	319.60	2.96	0.04	313.59
DEC + IJIYGF	3	319.87	3.23	0.03	313.86
HAYINT + IJIYGF	3	320.07	3.43	0.03	314.06
DEC + HAYINT + IJIDEV	4	320.11	3.47	0.03	312.10
DEC + IJIDEV + NON	4	320.14	3.50	0.03	312.13
IJIYGF + IJIDEV	3	320.21	3.58	0.03	314.21
HAYINT + ES	3	320.32	3.69	0.03	314.32
DEC + HAYINT	3	320.74	4.10	0.02	314.73
HAYINT + FOR	3	320.76	4.12	0.02	314.75
IJIDEV + FOR	3	320.83	4.19	0.02	314.82
NON * IJIDEV	4	321.59	4.95	0.01	313.57
DEC + IJIYGF + ES	4	321.65	5.01	0.01	313.63
IJIYGF * IJIDEV	4	321.68	5.05	0.01	313.67
DEC * IJIYGF	4	321.83	5.19	0.01	313.82
HAYINT + ES + IJIYGF	4	321.86	5.22	0.01	313.84
DEC + HAYINT + ES	4	321.86	5.23	0.01	313.85
DEC	2	321.96	5.32	0.01	317.95
HAYINT * DEC	4	322.20	5.57	0.01	314.19
DEC + ES	3	322.29	5.66	0.01	316.28
DEC + HAYINT + ES + IJIYGF	5	322.37	5.74	0.01	312.35
NULL	1	322.45	5.81	0.01	320.45

[a] DEC = proportion of deciduous forest, ES = proportion of early succession, NON = proportion of nonhabitat, FOR = proportion of all mature forest, IJIYGF = interspersion and juxtaposition index of young forest, IJIDEV = interspersion and juxtaposition index of developed areas, HAYINT = proportion of hay/pasture interior

Table B.4. Juvenile male wild turkey known-fate annual survival estimates by county, south-central Tennessee, 2017–2021. We present survival rates (S), standard errors (SE), 95% confidence intervals (95% CI), and sample sizes (*n*).

County	S	SE	95% CI	*n*
Bedford	0.95	0.05	0.67 – 0.99	20
Maury	0.83	0.06	0.67 – 0.92	40
Wayne	0.71	0.14	0.35 – 0.90	12
Lawrence	0.70	0.09	0.49 – 0.84	30
Giles	0.54	0.09	0.37 – 0.70	34

Table B.5. Juvenile male wild turkey survival estimates, harvest estimates, and year of study for various southeastern US studies.

Study area	Survival estimate	Spring harvest Estimate (%)	Years of study	Citation
Georgia	0.96	1	2017-2022	Wightman et al. 2024
West LA[a]	0.95	5	2013-2022	Wightman et al. 2024
Mountains NC[b]	0.94	0	2020-2022	Wightman et al. 2024
East LA[a]	0.90	0	2020-2022	Wightman et al. 2024
Webb SC[c]	0.82	5	2014-2018	Wightman et al. 2024
Virginia	0.77	4	2004-2007	Norman et al. 2022
Coastal NC[b]	0.74	5	2020-2022	Wightman et al. 2024
South-central TN[d]	0.73	3	2017-2021	Our study
West Virginia	0.70	9	2004-2007	Norman et al. 2022
Mississippi	0.69	0	1998-2000	Wieme 2001
Piedmont NC[b]	0.65	0	2020-2022	Wightman et al. 2024
Missouri Ozarks SSA[e]	0.60	11	1989-2000	Hubbard and Vangilder 2007
Missouri Ozarks PRCA[f]	0.57	16	1989-2000	Hubbard and Vangilder 2007
West KY[g]	0.53	N/A	1995-2002	Wright and Vangilder 2007

[a] Louisiana
[b] North Carolina
[c] South Carolina
[d] Tennessee
[e] South Study Area
[f] Peck Ranch Conservation Area
[g] Kentucky

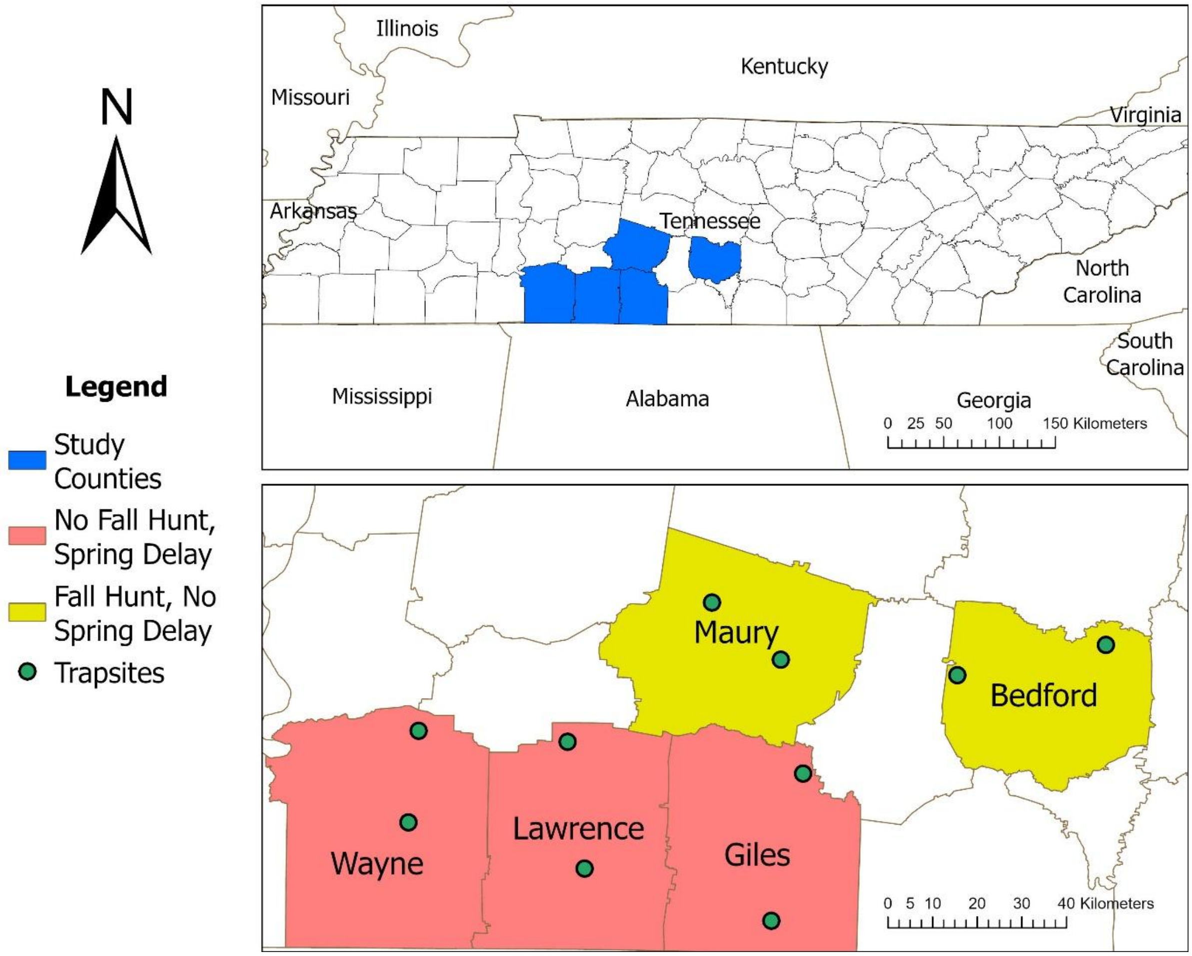

Figure B.1. Our five study counties and associated northern and southern trap sites for juvenile male wild turkeys, south-central Tennessee, 2017–2021. Counties are separated by whether an experimental 14-day spring hunting season delay was present in the county from 2021–2022 and whether the county had a fall turkey season.

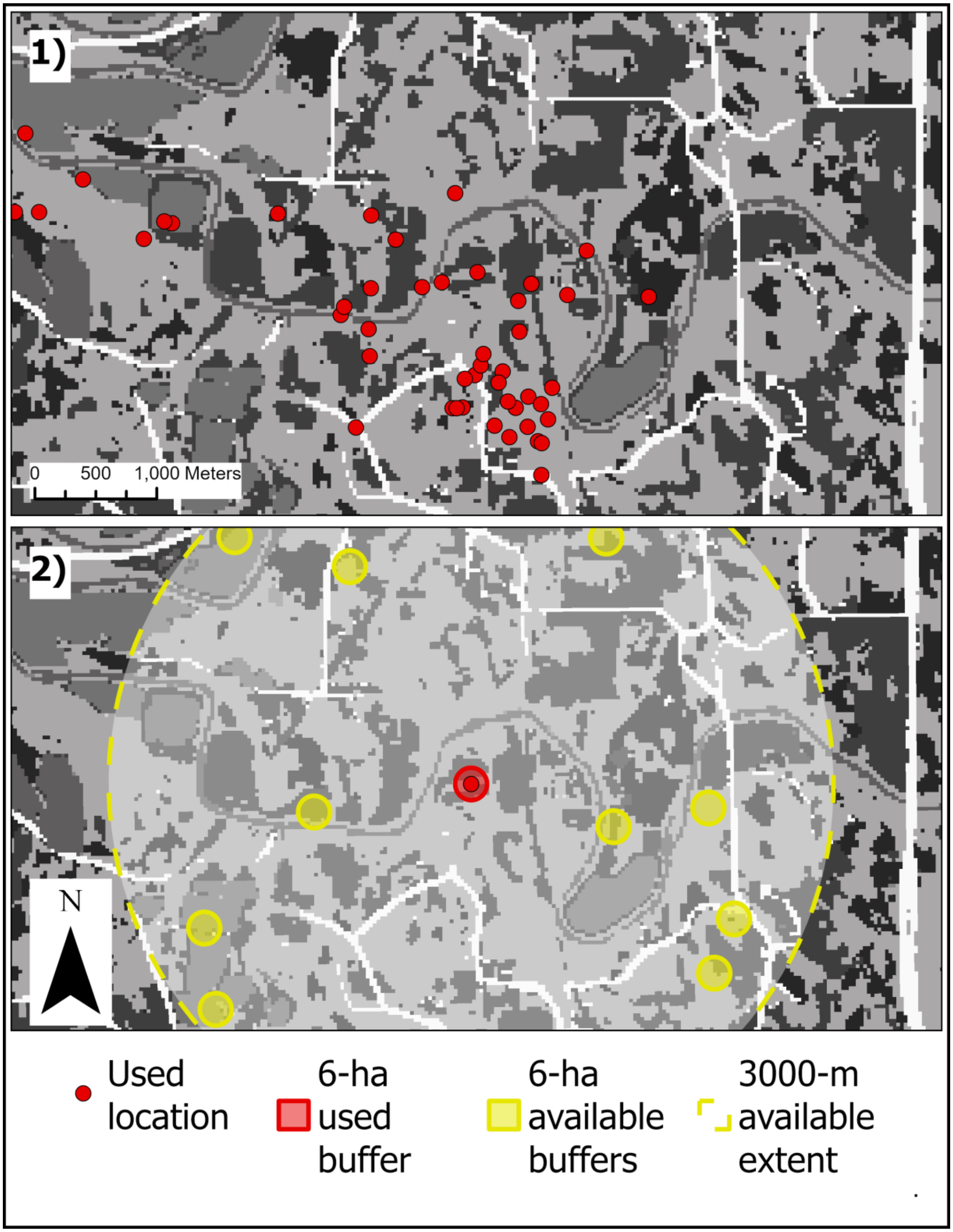

Figure B.2. Resource selection framework for juvenile male wild turkeys in south-central Tennessee, 2017–2021. 1) Example of used annual locations for one turkey. 2) Example of one used location and its 6-ha buffer surrounded by its associated 10 random available buffers within the 3000-m extent.

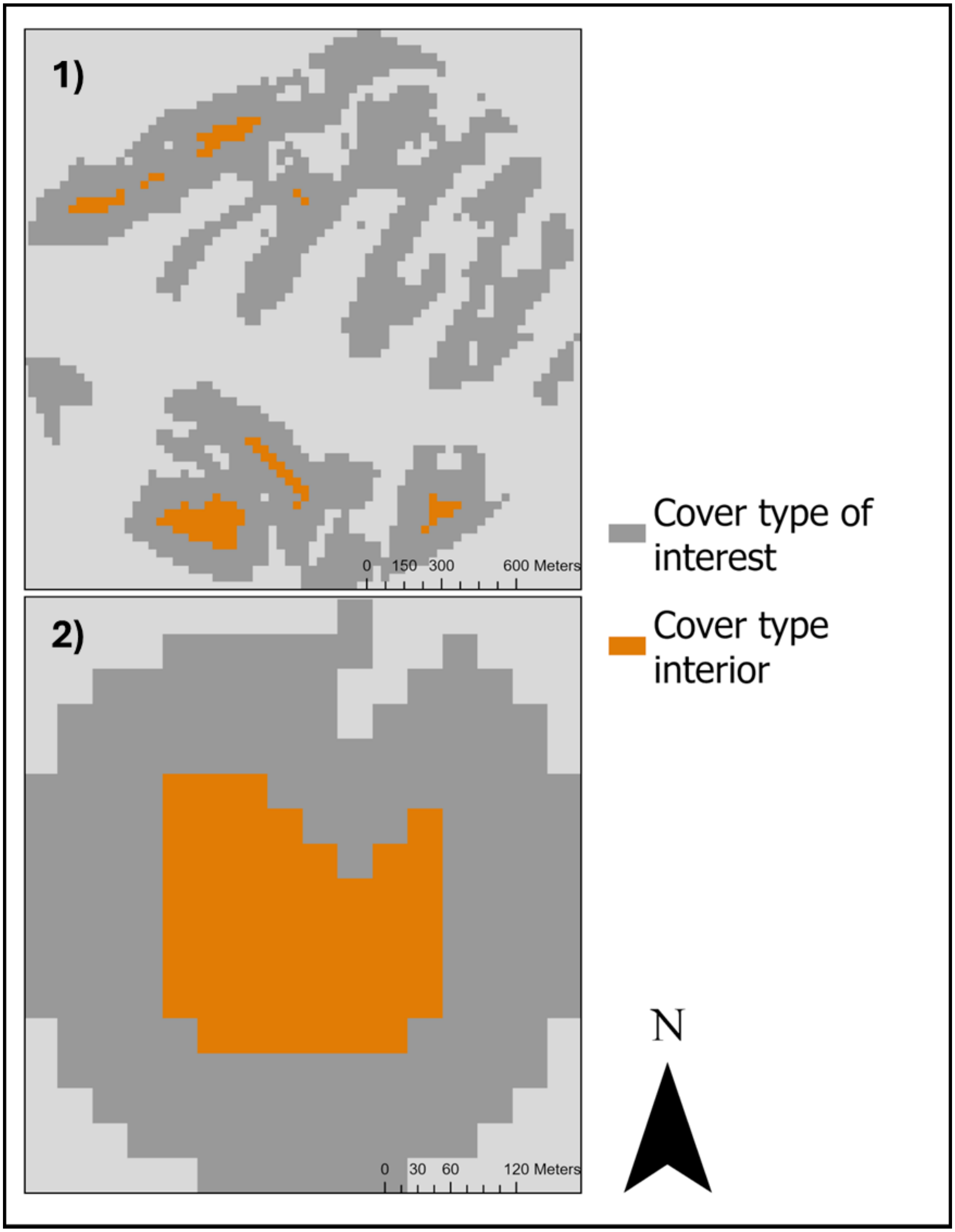

Figure B.3. Cover type interior metric for juvenile male wild turkey resource selection in south-central Tennessee, 2017–2021. 1) An example of what is considered to be cover type interior for a given cover type within the home range for known-fate survival analysis. 2) An example of what is considered to be cover type interior for a given cover type within a 6-ha buffer for case-controlled resource selection.

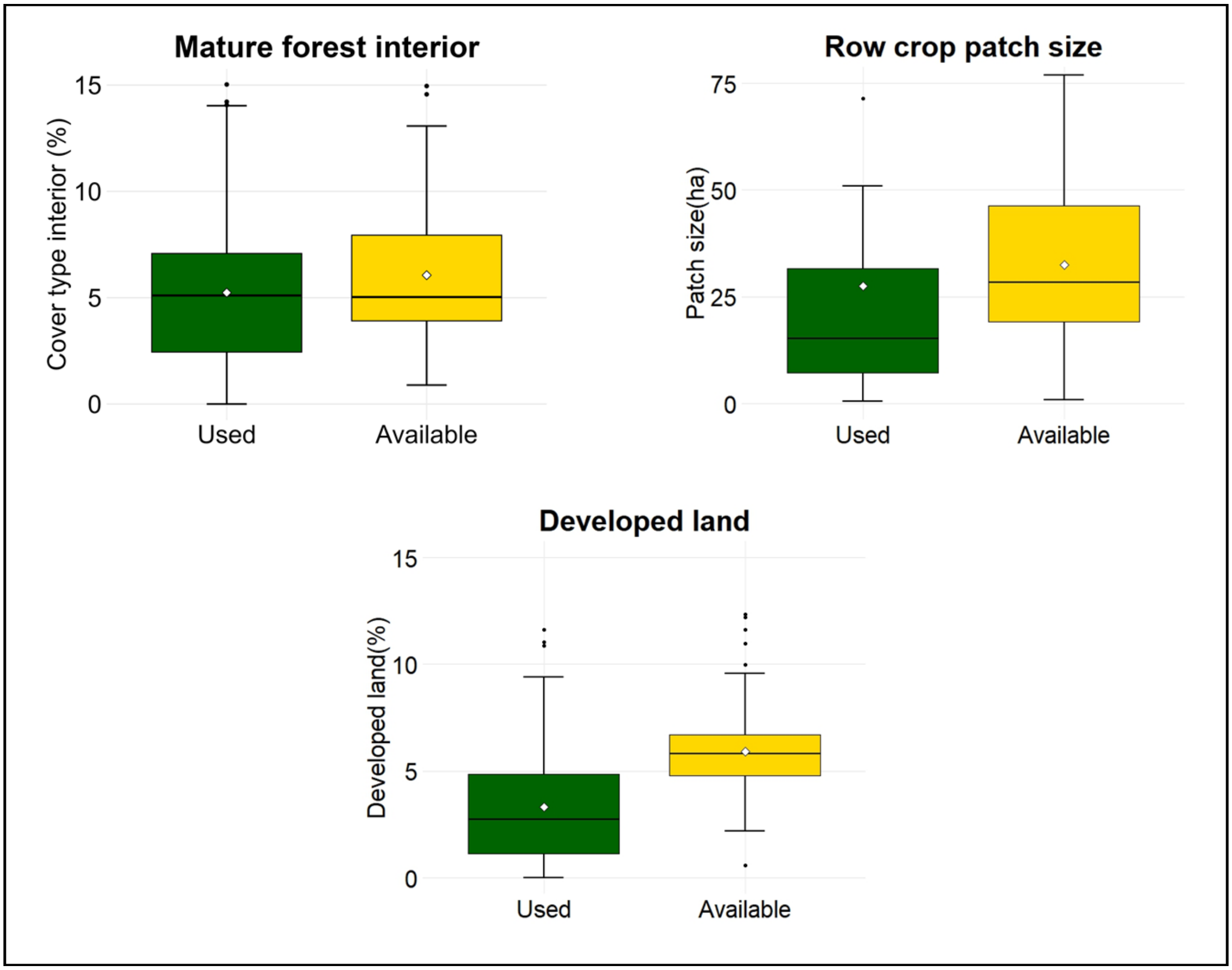

Figure B.4. Comparison of used vs available mature forest interior, row crop patch size and the proportion of developed land associated with location buffers for juvenile male wild turkeys, south-central Tennessee, 2017–2021. Box plots display the interquartile range, median, mean, upper and lower whisker limits, and outliers.

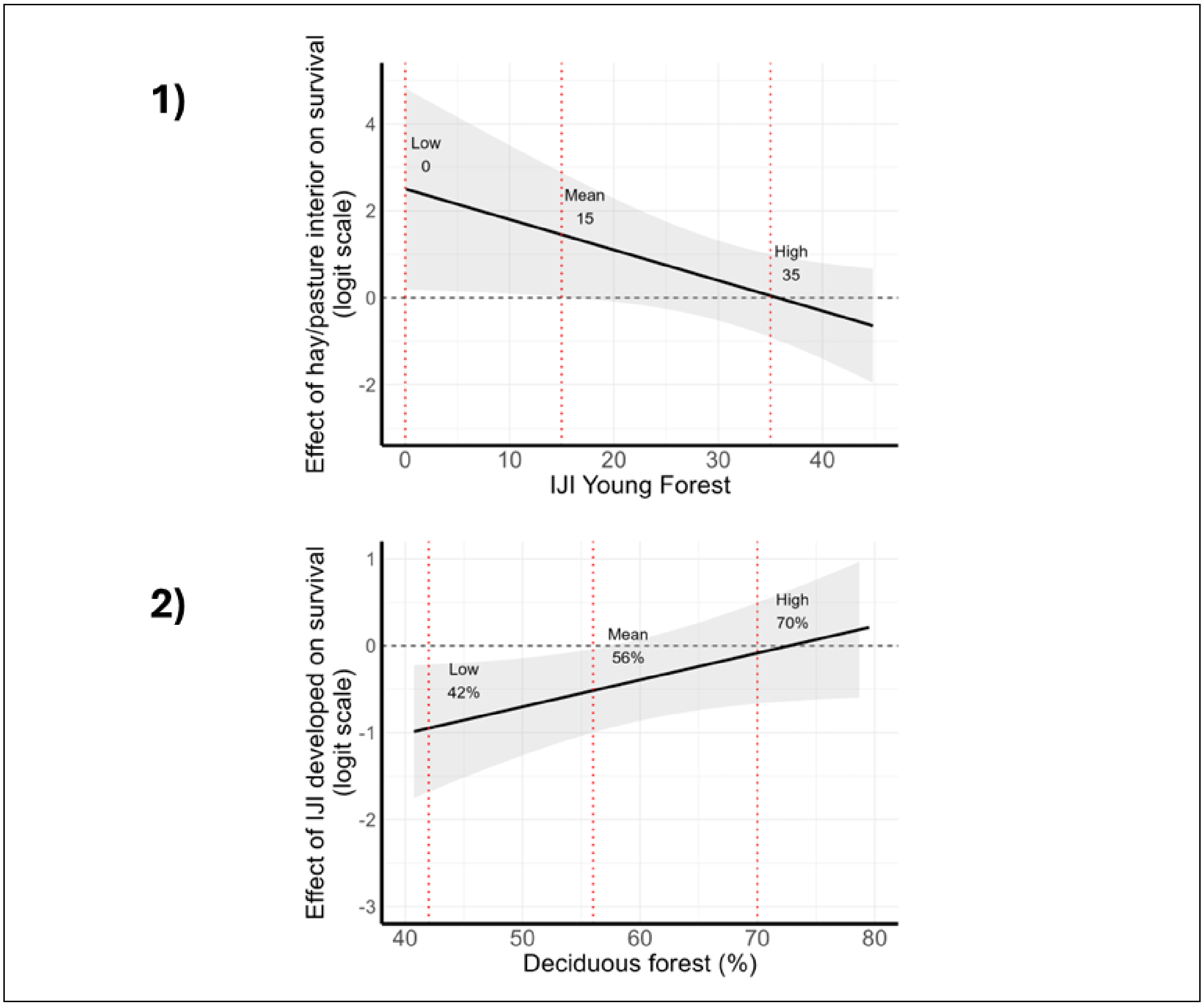

Figure B.5. Conditional effect plots for juvenile male wild turkey annual survival, south-central Tennessee, 2017–2021. 1) The effect of hay/pasture interior on annual survival across a range of young forest interspersion and juxtaposition index (IJI) values. 2) The effect of the IJI of developed land on annual survival across a range of deciduous forest proportions. The three levels indicated by the red dotted lines were determined by setting the middle level to the mean of that covariate and calculating 1 standard deviation below and above the mean for the low and high levels respectively. The shaded area represents the 95% confidence interval around the conditional effect. Values above the horizontal dotted line represent conditions that increase survival while those below the line represent conditions that decrease survival.

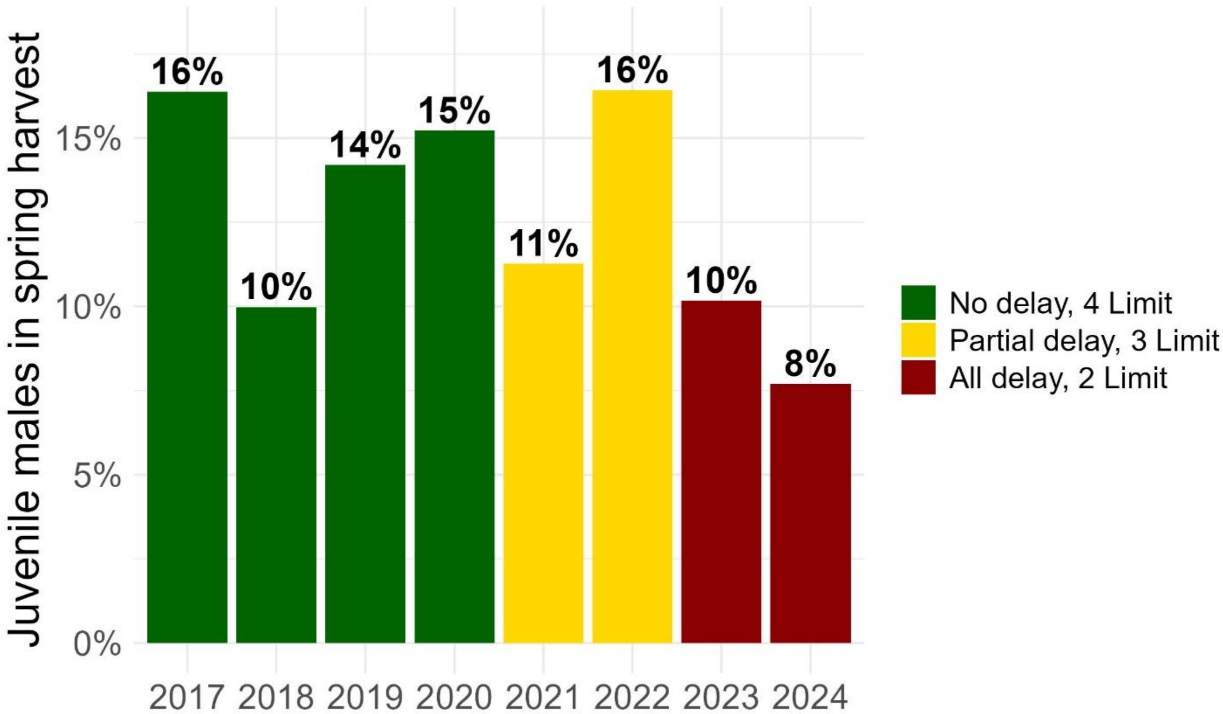

Figure B.6. The proportion of the spring wild turkey harvest in Bedford, Giles, Lawrence, Maury, and Wayne counties that was made up of juvenile male turkeys, south-central Tennessee, 2017–2021. No delay, 4 Limit = no 14-day spring hunting season delay in any of the five counties and a 4-bird bag limit, partial delay = 14-day spring hunting season delay in Giles, Lawrence, and Wayne Counties and a 3-bird bag limit, all delay = 14-day spring hunting season delay in all five counties and a 2-bird bag limit.

CHAPTER 3. THE RELATIONSHIP OF LAND COVER AND PREDATOR INDICES WITH SURVIVAL AND HARVEST OF ADULT MALE WILD TURKEYS IN SOUTH-CENTRAL TENNESSEE

ABSTRACT

Documenting survival, harvest, and cause-specific mortality for adult male wild turkeys is important for monitoring populations and comparing vital rates between regions. Furthermore, understanding how mortality varies with landscape composition and structure is critical for management. To better understand the relationship that land cover, predators, and harvest regulations have with adult male wild turkey mortality, we trapped, banded, and attached backpack VHF radio transmitters to adult males (n = 188) in five counties of south-central Tennessee, 2017–2021. We modeled mortality in relation to land cover and predators using Cox proportional hazards models, and we calculated cause-specific mortality using a cumulative incidence function. We calculated all land cover covariates using the National Land Cover Database to obtain landscape metrics associated with each turkey's home range. We calculated a predator index using photos from remote-sensing cameras at each study site. We calculated annual survival using the Barker joint live and dead model to incorporate band recovery and telemetry data. We also analyzed reported county-wide hunter harvest data using a generalized linear mixed model to determine if changes in harvest regulations, juvenile male harvest, hunter effort, and fecundity affected total adult male harvest. Mean annual survival was 0.38 (95% CI = 0.32–0.45). Adult male wild turkey non-harvest mortality increased with greater interspersion of developed land, greater interspersion of mature forest during the growing season, and a greater predator index. Mean annual spring harvest rate was 0.32 (95% CI = 0.27–0.38) and non-harvest mortality was 0.25 (95% CI = 0.19–0.30). A 14-day delay in the spring hunting season had no effect on harvest, and a reduction in the bag limit from 4 to 3 to 2 did not reduce county-wide adult male wild turkey harvest. However, increased fecundity was related to increased turkey harvest two years later. The number of predators detected on camera was proportional to turkey

mortality and woody cover likely increased ambush opportunities for these predators. Human development also may negatively impact wild turkeys by attracting greater numbers of predators and exposing turkeys to other risks, such as vehicle collisions. Reducing dense forest cover and implementing predator removal when predator abundance and predation mortality is high may increase adult male wild turkey survival. Other management actions, such as those that increase wild turkey productivity, have the potential to increase wild turkey harvest, whereas harvest regulations implemented without scientific support may reduce hunter participation and have no measurable effect on harvest.

KEYWORDS harvest, hunting regulations, adult male, *Meleagris gallopavo*, mortality, survival, Tennessee, wild turkey

Declines in harvest and population estimates of wild turkeys (*Meleagris gallopavo*) have been observed in many southeastern states in the last decade following the successful restoration of the species throughout the United States (US) from the 1950s to late 1990s (Tapley et al. 2005, 2011, Chamberlain et al. 2022). Poult-per-female ratios have declined throughout much of the Southeast, including Tennessee, which has reported a downward trend for the past couple decades (Byrne et al. 2015, Shields 2024). Historically low poult survival was reported in a recent Tennessee study (Quehl et al. 2025) as well as a 43% decline in relative abundance from 2013 to 2022, based on North American Breeding Bird Survey data (United States Geological Survey 2022). Declining harvest in some counties in south-central Tennessee since 2005 has generated additional concern since the harvest of this economically important game bird contributes nearly $200 million to the state economy annually (Harris 2010, CPI Inflation Calculator 2026). Collectively, these factors highlight the need to investigate the causes of wild turkey declines in the southeastern US where these declines have been more pronounced.

Wild turkey population declines are not likely caused by a single factor, but instead may be driven by various mechanisms operating concurrently, such as land-use change, increased predation pressure, and liberal harvest regulations (Casalena et al. 2015, Chamberlain et al. 2022, Londe et al. 2023). Land-use change in the US is exemplified in the Southeast where more than 3 million ha were developed from 1997 to 2017, an increase of 26% over that twenty-year period (United States Department of Agriculture 2020). Meso-mammalian predators have increased in the Southeast since the 1980s. Bobcat (*Lynx rufus*) populations have increased and coyotes (*Canis latrans*) have expanded their range eastward such that they now are ubiquitous throughout the region (Roberts and Crimmins 2010, Casalena et al. 2015, Bragina et al. 2019, Wang et al. 2023). The effect of predators has been compounded as urbanization and forest cover have increased in the southeastern US, favoring meso-mammalian predators and increasing vulnerability for wild turkeys (Kunkel 1997, Whittingham and Evans 2004, Pollentier et al. 2014*b*, Ivey et al. 2024, Wang et al. 2025). Harvest frameworks have been hypothesized to contribute to wild turkey population declines. Fall harvest of female wild turkeys, which was allowed in some states including Tennessee following restoration, relatively liberal bag limits, and spring hunting seasons starting relatively early in the breeding season of wild turkeys have been hypothesized as possible contributors to wild turkey population declines (Norman et al. 2004, Chamberlain et al. 2012, Isabelle et al. 2018). Peer-reviewed studies documenting the effect of these hunting regulations are severely lacking, with the exception of a recent study in Tennessee that reported a 14-day delay in the opening date of the spring season had no effect on wild turkey productivity over a 6-year period (Isabelle et al. 2018, Quehl et al. 2024*a*). Although many factors may be contributing to wild turkey population declines, the influence of these hypothesized mechanisms must be examined to provide critical information needed to make science-based regulatory decisions and habitat and population management recommendations.

Wild turkey research has focused on females because of their important role in reproduction, recruitment, and population growth, but research on adult males is crucial to sustain hunter satisfaction and the economic benefit of wild turkey hunting (Pollentier et al. 2014*a*, Lashley et al. 2025, Tyl et al. 2025). More than 90,000 hunters are licensed to hunt wild turkeys annually in Tennessee, contributing substantial funds to the state agency budget in license sales and to the economy via travel and equipment expenses and sales tax (Harris 2010, Shields 2024). The number of adult male wild turkeys on the landscape also directly translates to hunting quality, because gobbling activity is strongly linked to hunter satisfaction (Swanson et al. 2005, Gruntorad et al. 2020, Quehl et al. 2024*b*). Therefore, increasing our understanding of male mortality and harvest is essential for better management of this important gamebird.

Estimation of annual survival and harvest rates for adult male wild turkeys is necessary information for monitoring trends and to compare vital rates between states or regions (Norman et al. 2022, Wightman et al. 2024). However, many studies provide such rates without further analysis to determine what factors are contributing to changes in these rates in their study system (Hubbard and Vangilder 2007, Wright and Vangilder 2007, Diefenbach and Vreeland 2010, Wightman et al. 2024). Understanding how these vital rates are linked to potential causal factors can further our understanding of why wild turkey populations or harvest rates are changing over time. The impact of hunting regulations is an important research topic since regulations are important management tools and may encourage or restrict hunter participation (Kurzejeski and Vangilder 1992, Wam et al. 2012, Watkins et al. 2018, Gruntorad and Chizinski 2021). Recruiting, retaining and reactivating hunters has become an increasingly important consideration in state regulation setting because the overall number of hunters is declining nationwide (Aiken 2019, U.S. Fish & Wildlife Service 2023). Researching the effects of landscape, predators, and regulations on adult male wild turkeys can identify management opportunities to increase wild turkey populations and increase hunter participation and satisfaction.

We radio-tagged, banded, and monitored wild turkeys over 5 years in 5 counties of south-central Tennessee to study how adult male wild turkey mortality rates and harvest are related to land cover, predators, and harvest regulations. Our research objectives were to (1) calculate the annual survival and harvest rate for adult male wild turkeys and determine to what extent specific types of mortality are related to land cover type, landscape metrics, and a predator index and (2) determine to what extent the harvest of adult male wild turkeys in Tennessee is linked to reproduction, juvenile male harvest, hunter effort, weather, and changes in harvest regulations. We hypothesized non-harvest mortality would be related to the abundance and visibility of wild turkey predators. We also hypothesized non-harvest mortality would be related to human development. We predicted greater non-harvest mortality in home ranges with more developed land and a greater predator index because of increased densities of predators and less food and cover for turkeys. We predicted greater non-harvest mortality for turkeys with home ranges that contained a greater amount or increased interspersion of low visibility land cover types, such as young forest, because of increased ambush opportunities for predators. We predicted greater non-harvest mortality for turkeys with home ranges that had a greater density of roads as a result of increased poaching from roads, more vehicle collisions, and denser edges that reduce visibility. Lastly, we hypothesized adult male wild turkey reported harvest would be related to harvest of juvenile males, reproduction, and hunter effort, and would not be related to changes in harvest regulations or the amount of rainfall during the season. We predicted greater adult male wild turkey harvest in a given year would be related to decreased juvenile male harvest one year prior since juveniles are the direct source population for adult male wild turkeys. We predicted greater adult male wild turkey harvest would be related to greater reproduction two years prior and greater hunter effort.

STUDY AREA

We conducted research in Bedford, Giles, Lawrence, Maury, and Wayne counties in south-central Tennessee, 2017–2021. These five counties were chosen because of growing concern for wild turkey population declines in south-central Tennessee. These declines were supported by a

60% reduction in harvest from 2005–2015 in Giles, Lawrence, and Wayne county in conjunction with poult-per-female declines across the state (Buehler and Harper 2024, Quehl et al. 2024*a*, Shields 2024). We established north and south study sites within each county for trapping and monitoring turkeys (Figure C.1). Two of the ten study sites were on public land: Tie Camp WMA (1325 ha) in Wayne County and Yanahli WMA (5200 ha) in Maury County, whereas the remaining eight sites were on private land. Private land sites totaled >29,000 ha and included >380 individual landowners. The study sites were comprised of mature deciduous and coniferous forest, young forest (<10 years old), early successional plant communities, hayfields, pastures, row crops, and human-developed areas. The primary soil types for the study area were Bodine cherty silt loam and gravelly silt, Gladeville rock outcrop, Ashwood, Brandon silt loam, Biffle gravelly silt loam, and Frankstone cherty silt loam (United States Department of Agriculture 2019). Since 1989, the spring wild turkey season opened on the Saturday closest to April 1 (including 29–31 March), was six weeks in duration, and had a bag limit of four bearded turkeys. In 2021–2022, the spring turkey season was delayed 14 days (thereby shortened by two weeks) in Giles, Lawrence, and Wayne counties, and the bag limit was reduced to three bearded birds, of which only one could be a juvenile male. In 2023–2024, the season was delayed 14 days statewide but lengthened at the end of the season by two weeks for a total duration of six weeks, and the bag limit was reduced to two bearded birds, of which only one could be a juvenile male. Additionally, Giles, Lawrence, and Wayne counties were closed to fall turkey hunting starting in 2014, whereas Bedford and Maury counties have had a fall hunting season with a bag limit of one bearded turkey per county since 2016.

METHODS

Field methods

We trapped wild turkeys during winter, 2017–2021, using box-set rocket nets (Delahunt et al. 2011). We baited trap sites with shelled corn every other day to attract turkey flocks. We used remote cameras to monitor wild turkey visitation patterns, sex and age ratios, and flock sizes (Moultrie: Model MCG13202, Birmingham, Alabama, USA). We established a goal of maintaining a sample of 100 male turkeys each year across all five focal counties (10 males at each of the 10 sites, 5 adults and 5 juveniles). We aged turkeys based on tail fan margins, primary feather barring, and leg spur length according to Pelham and Dickson (1992). We fitted each trapped male with a $75 reward aluminum leg band (National Band and Tag Company: style 1242FR8A, Newport, Kentucky, USA), assuming such a reward would give a near 100% return rate (Diefenbach et al. 2000). We attached an Advanced Telemetry Systems VHF backpack transmitter to each male (Series A1500, ~80 grams, life expectancy of 5.7 years, Advanced Telemetry Systems, Isanti, MN). We recorded each male's weight, beard length, spur length, tarsus length, and body condition score (scoring the amount of fat and muscle found on the bird's keel; Gregory and Robins 1998). After processing, we immediately released each turkey at the trap site (University of Tennessee IACUC protocol #0561-0720). We radio-tracked wild turkeys weekly to monitor their mortality status and to triangulate their location during each tracking event. When a turkey entered mortality mode because of a lack of activity for more than 8 hours, we retrieved the transmitter and investigated the cause and location of death. We right-censored all turkeys that we could not detect with telemetry after the last successful tracking occasion. We investigated the majority of turkey mortalities within 2 days after mortality unless we were unable to obtain access to private property. We removed individuals that died within 14 days of capture from analysis to avoid including turkeys that may have experienced capture myopathy (Nicholson et al. 2000). Cause of all mortalities was determined based on the condition of the carcass and field sign surrounding it. We characterized bobcat predation by a cached or covered carcass (Thogmartin and Schaeffer 2000). We characterized coyote predation by broken bones, scattering of remains, and presence of coyote scat and hair (Thogmartin and Schaeffer 2000). We characterized great horned owl (*Bubo virginianus*) predation by the removal of the head and breast muscle (Thogmartin and Schaeffer 2000, Little et al. 2016). If mammalian predation was apparent but we did not have enough evidence to attribute the mortality to a given species, we categorized the mortality as an unknown mammal. Poaching was identified by removal of the transmitter with no evidence of predation. All harvested birds were reported via reward leg bands, and we identified any other mortalities as unknown. The land cover type associated with each predation mortality was recorded for the turkeys where the kill site was positively identified from feather remains and evidence of a predation event.

Home range analysis

We calculated locations via triangulation in the program LOAS using three intersecting bearings obtained during VHF transmitter tracking events (Sallee 2005). We attempted to triangulate turkeys weekly during all months of the year, but there were some instances in which an accurate triangulation could not be obtained as a result of topography or property access. We calculated our location error in LOAS using a 95% confidence interval ellipse polygon. We removed all locations from home range analysis whose error polygon was >10 ha, which resulted in a mean error polygon of 1.4 ha/location. We created 90% minimum convex polygon (MCP) annual home ranges for turkeys using the adehabitatHR (version 0.4.22) package (Calenge 2006) in R version 4.4.1 (R Core Team 2026). We created home ranges for turkeys with ≥20 locations from their capture date to the end of the year. Any individuals that had <20 locations or did not survive for the entire year were excluded from the home range analysis.

Land cover variables and predator index

To study the relationship of land cover and predators with adult male wild turkey mortality, multiple metrics were calculated that were associated with each wild turkey. All landscape covariates were calculated from the 2017–2021 United States Geological Survey (USGS) National Land Cover Database (NLCD) 30-meter spatial resolution Landsat land cover maps of North America using ArcGIS Pro 3.3.1 (Environmental Systems Research Institute, Redlands, California, USA; United States Geological Survey 2024). Landscape covariates were calculated using the landscapemetrics (version 2.2.1) package (Hesselbarth et al. 2019) in R version 4.4.1 (R Core Team 2026) for all turkeys by using identical circular polygons equal to the mean annual home range size of all turkeys that lived until the end of the year placed over the centroid of each turkey's estimated locations. We chose to use average home range polygons to clip the NLCD map for each turkey because only a subset of turkeys that lived until the end of the year had an individual MCP home range produced. To justify the use of average rather than actual home ranges, we created a generalized linear mixed-effects model with a gaussian error distribution using the glmmTMB function from the glmmTMB (version 1.1.14) package (Brooks et al. 2017) in R version 4.4.1. There was no difference between the landscape covariates for actual MCP and average home ranges ($Z = -0.05$, $P = 0.96$; R Core Team 2026). Using the clipped average home range polygons, we calculated the percentage of land cover by type, forest edge density, percentage cover type interior, and an interspersion and juxtaposition index (IJI). The percentage of land cover metric was calculated by dividing the number of pixels of a given land cover type by the total number of pixels within the home range. Forest edge density was defined as the meters of forest edge (where forest cover and another cover type met) per hectare within the home range (McGarigal 1995). The percentage of cover type interior was defined as the square meters of land cover of a particular land cover type that was ≥120 m in all four cardinal directions from a patch edge divided by the home range area (McGarigal 1995). The 120-m distance was chosen to align with previous studies and to represent a maximum preferred foraging distance from edges (Schroeder 1985, Glennon and Porter 1999, Fleming and Porter 2000). The formula used to calculate cover type interior was:

$$CPLAND = \left(\frac{\sum_{j=1}^{n} a_{ij}^{core}}{A}\right) * 100 \qquad (1)$$

Where a_{ij}^{core} was the cover type interior in square meters, and A was the total home range area in square meters. IJI was defined as the observed interspersion of land cover type classes over the maximum possible interspersion for the given number of land cover type classes (McGarigal 1995). The equation used to calculate IJI for each land cover type was:

$$IJI = \frac{-\sum_{k=1}^{m}\left[\left(\frac{e_{ik}}{\sum_{k=1}^{m} e_{ik}}\right) ln\left(\frac{e_{ik}}{\sum_{k=1}^{m} e_{ik}}\right)\right]}{ln(m-1)} * 100 \qquad (2)$$

Where e_{ik} are the unique adjacencies of a particular land cover type class in meters, and m is the number of all classes present (McGarigal 1995). IJI values range between 0 and 100 with a lower value indicating a home range with land cover types with less interspersion (McGarigal 1995). All landscape metrics were calculated from land cover types defined by the NLCD map which included open water, coniferous forest, deciduous forest, young forest, early successional vegetation, hay/pasture, row crop, woody wetland, emergent herbaceous wetland, developed land, and barren land. We combined open water, wetland, and barren land into a singular cover type called nonhabitat. The developed land cover type consisted mainly of roads, individual residential homes, and farmsteads. Average distance to roads was calculated using the Tennessee Department of Transportation (TDOT) road segment layer in ArcGIS Pro 3.3.1 (2022). We created three separate layers from the original layer (low-traffic roads, high-traffic roads, and all roads) to examine the effects of roads at different traffic levels. Low-traffic roads consisted of roads defined with the functional class of urban local, rural local, or access roads. Access roads on our study sites included private roads, logging roads, and field roads that were intended for vehicles. We hand traced these access roads in ArcGIS Pro 3.3.1 using available satellite imagery and ground-verification of the road system because the TDOT layer does not define this type of low-traffic road. High-traffic roads consisted of all roads minus those defined in the low-traffic road category. The average distance to road metric was calculated by placing average home range polygons over the road layers in ArcGIS Pro 3.3.1 and using the Create Spatial Sampling Locations tool to place 100 evenly spaced points across the home range. The mean distance from all points to their nearest road was then calculated for each road layer to obtain an estimate of exposure to roads for each turkey's home range. We calculated a predator index for our study sites by using photos captured using remote-sensing cameras deployed at each turkey trap site. We deployed cameras from January–March, 0.5–0.75 m aboveground and captured photos on a 30-sec delay. We uploaded all photos to Wildlife Insights and photos flagged to contain coyotes, bobcats, or foxes (*Vulpes vulpes* and *Urocyon cinereoargenteus*) were validated and the number of these predators were summed at each site in a given year (Wildlife Insights 2026). Photos of the same species < 5 min apart from each other were counted as one individual to avoid double counting the same individual. The total number of each species detected at a site was divided by the total number of trap nights that the camera was deployed to calculate a predator index for that site for a given year.

Cox proportional hazards mortality analysis

We calculated cause-specific hazard ratios to understand the relationship that different variables had with adult wild turkey mortality by fitting time-to-event Cox proportional hazards models with staggered entry and right censoring to our adult male wild turkey survival data obtained during weekly tracking events using the survival (version 3.8-3) package (Therneau et al. 2024) in R version 4.4.1 (Therneau and Grambsch 2000, R Core Team 2026). Time-to-event data were compiled on an annual cycle, starting at capture for adults or on January 1st of their second winter if captured as a juvenile, and ending at death or the end of the year. Structuring the models in this way enabled the calculation of annual mortality rates rather than total mortality for the duration of the study. Models were clustered by the unique turkey ID number to calculate robust standard errors and to account for lack of independence (Therneau and Li 1998, Virgili et al. 2007). Causes of mortality were grouped into three categories: harvest, predation, and other sources. The other sources of mortality consisted of poaching, car collisions, and unknown mortality. We created three separate model sets for harvest, predation, and non-harvest mortality (combination of predation and other mortality) to study relationships specific to these mortality sources. Our model sets were intended to investigate the separate relationships of covariates with harvest, predation, and non-harvest mortality. We included percentages of all land cover types to determine if the amount of a given land cover type in the home range was related to survival (Miller and Leopold 1992, Whittingham and Evans 2004). We included IJI covariates in models for all land cover types to determine if the relationship of land cover with survival was more dependent on the configuration and distribution in the home range instead of the total amount present (Mimet et al. 2016, Gengler et al. 2024). We included row crop, mature forest, and hay/pasture interior covariates in models to determine if large uninterrupted areas of open or forested cover would be related to survival by affecting visibility for wild turkeys. We included mature forest edge density in models to determine if the total length of woody edges was related to survival by increasing predator success as a result of decreased visibility along forest edges (Šálek et al. 2010, Wang et al. 2025.) We included the average distance to all three road types (low-traffic, high-traffic, and all roads) in the models to determine if the presence of roads increased mortality, potentially as a result of increased dense edge with limited visibility and increased poaching from vehicles. We included the predator index in the models to determine if an increase in predators at a site would increase mortality. We included an intercept-only (null) model in each model set as a baseline for evaluating model fit. We were unable to calculate a predator index for all sites and years because some cameras malfunctioned, so our predation mortality models were all run on a reduced data set. We standardized all continuous covariates by z-transforming each to enable direct comparison of beta coefficient effect sizes (Quinn and Keough 2002, Zuur et al. 2007). We created univariate and multi-covariate models to address our hypotheses. First, covariates were tested univariately to examine individual effects on mortality and to inform subsequent modeling (Bursac et al. 2008). Multivariate models were then created by including covariates that would test our specific hypotheses based on the literature or developed from field experience. We evaluated multicollinearity among continuous covariates using the vif function from the car (version 3.1-3) package (Fox and Weisberg 2019) in R version 4.4.1 (R Core Team 2026) by using a threshold of 5 and removing any models that exceeded that threshold (Shrestha 2020). We tested the models to check that the proportional hazards assumption was met by using the cox.zph function in the survival package in R version

4.4.1 (R Core Team 2026). If P was <0.05 (indicating the effect of a covariate varied across time), we refit models containing that covariate using the survSplit function in the survival package to explore time-varying covariates (Zhang et al. 2018). We split the year into winter months (January–March and November–December) and growing season months (April–October) when fitting time-varying covariates. We chose these time periods because the effect of land cover and predators on mortality likely differ based on growing season as denser vegetation could differentially affect mortality for adult male wild turkeys. For model selection, we used Akaike's Information Criterion corrected for small sample size (AICc). We considered all models with ΔAICc < 2 to have substantial support, and we evaluated each model with respect to our research hypotheses (Burnham and Anderson 2002). We used Wald z-tests to evaluate the influence of the covariates on each type of mortality. We considered covariates from both analyses influential if their beta coefficient estimates had 85% confidence intervals that did not include zero, following principles of AIC model selection and multi-model inference (Arnold 2010, Tan and Tan 2010). We used confidence intervals as heuristic filters rather than as formal hypothesis tests. This inferential approach does not imply statistical significance in the traditional null hypothesis-testing framework but rather is interpreted as indicating support for an effect given the data and model structure (Burnham and Anderson 2002). We then tested if there were differences in harvest, predation, and non-harvest rates between counties, sites, and years by using the Anova function from the car (version 3.1-3) package in R version 4.4.1 (R Core Team 2026) to perform Wald chi-square tests (Fox and Weisberg 2019) based on an α = 0.05 decision criterion. Cause-specific mortality rates were calculated using the cumulative incidence function estimator from the mstate (version 0.3.3) package (Wreede et al. 2011) in R version 4.4.1 (Heisey and Patterson 2006, R Core Team 2026). However, to remain consistent and comparable to the literature, we report estimates of means—annual survival, annual harvest, and home range size—with 95% confidence intervals and associated standard errors.

Barker model survival and harvest analysis

We estimated annual survival using the Barker joint live and dead encounters and resighting model in the RMark (version 3.0.0) package (Laake 2013) in R version 4.4.1 (Barker 1997, 1999, Kendall et al. 2013, R Core Team 2026). The model consists of the following 7 parameters: survival probability (S_i)—the probability that a turkey alive at i was alive at i+1, capture probability (p_i)—the probability a turkey at risk of capture at i was captured at i, reporting rate (r_i)—the probability a turkey that died in between i and i+1 was found dead and reported, resight probability (R_i)—the probability a turkey that survived from i to i+1 is resighted alive between i and i+1, resight before death (R'_i)—the probability a turkey that died in between i and i+1 without being found dead was resighted before death from i to i+1, site fidelity (F_i)—the probability a turkey at risk of capture at i is at risk of capture at i+1, temporary emigrants (F'_i)—the probability a turkey not at risk of capture at i is at risk of capture at i+1. The index i refers to the time interval, which was defined in our model as one year. We chose to use the Barker model as a way of increasing the accuracy of our survival and harvest rate estimates since the model incorporates recapture, band/tag recoveries, and resighting occasions (Grisham et al. 2008, Kendall et al. 2013, Wightman et al. 2024). The two parameters of interest for our study were survival (S_i) and reporting rate (r_i), whereas the five remaining parameters were treated as nuisance parameters; they were needed for parameterization of the model but were not of primary interest. We constructed two model sets: one was parameterized to calculate annual survival and the other for annual harvest rate. For the survival model candidate set we modeled p_i as intercept only (.). We modeled r_i with a covariate for tagging method (transmitter with band or band only) since the recovery rate of dead turkeys with transmitters is assumed to be greater than recovery rates for band-only birds. We modeled R_i and R'_i so that turkeys with transmitters had estimable resighting probabilities, whereas band-only birds were constrained at zero since we never resighted a band-only bird for the duration of the study. We fixed $F_i = F'_i = 1$ in accordance with Cooch and White (2026) since there was no evidence of permanent emigration from study sites for the duration of our study (Barker and White 2001, Wightman et al. 2024). With the aforementioned parameters modeled as specified, we fit a candidate set where S_i was modeled as intercept-only (.) as well as by the individual, additive, and interaction effects of year, county, and site. For our second candidate set used to calculate harvest rate, all parameters were modeled identical to the first set except for r_i and R'_i. We fixed R'_i at zero in our second candidate set because natural mortalities were censored to calculate the reporting rate and survival of harvested birds alone which was used to obtain the annual harvest rate (Wightman et al. 2024). We then fit a candidate set where S_i and r_i were modeled using the same methods and covariates (year, county, site) as the first candidate set. Survival and reporting rates from the second candidate set could be used in the Seber reporting rate equation ($f = r(1\text{-}S)$) to calculate an index for harvest rate based on the assumption that our reward bands encouraged a near 100% reporting rate (Diefenbach et al. 2000, Sedinger et al. 2010, Wightman et al. 2024). For each candidate set, we calculated the overdispersion parameter $\hat{c}$ for the most general model in the set using the median $\hat{c}$ estimation available using Program MARK (Cooch and White 2026). For a $\hat{c} > 1$ we used the quasi-likelihood modified Akaike's Information Criterion adjusted for small sample size (QAICc) for model selection (Burnham and Anderson 2002). All models with ΔQAICc < 2 were considered to have substantial support and we evaluated each model with respect to our research hypotheses (Burnham and Anderson 2002).

We performed a simple linear regression of county- and year-specific harvest rates against survival rates to determine if harvest was additive and we tested if the slope was equal to zero using a F-test in R version 4.4.1 (R Core Team 2026). We expected that if harvest was compensatory, survival rates would remain relatively consistent regardless of increasing harvest rates (Robinson et al. 2014, Wightman et al. 2024). As noted by Riecke et al. (2022) and Wightman et al. (2024), this method for determining additive mortality does not take into account possible density-dependent effects or environmental factors that may influence survival and harvest.

Reported harvest analysis

We were unable to analyze the effect that changes in harvest regulations in 2021 and 2023 had on our survival or harvest because radio-tracking and trapping of adult male wild turkeys ceased in 2022. For this reason, we obtained reported spring harvest data from the Tennessee Wildlife Resources Agency (R. Shields, unpublished data) for Bedford, Giles, Lawrence, Maury, and Wayne counties for 2017–2025 to assess the impact that regulation changes, fecundity, hunter effort, rainfall, and juvenile male harvest had on adult male turkey harvest each year. We calculated the total reported spring season harvest of juvenile and adult males from the annual harvest data by county and year for 2017–2025. We calculated fecundity from monitoring a sample of >100 radio-tagged females per year by multiplying the proportion of females to nest, nest success, clutch size, hatch rate, and poult survival together for initial and renesting females and added the two values together. Fecundity equaled the estimated number of male poults alive at 28 days post-hatching for each female, assuming a 50:50 sex ratio at hatching. All female reproduction and hunter effort data were obtained from a companion study we implemented at the same study sites, 2017–2022, focusing on female wild turkey survival, reproduction, and hunter attitudes (L. Phillips, unpublished data; Phillips et al. 2024, Quehl et al. 2024*b*). We obtained average turkey hunting hours/hunter for each county and year, 2017–2022, from 1,893 survey responses from hunters in our focal counties. We wanted to test if rainfall, which has been shown to reduce harvest of white-tailed deer (*Odocoileus virginianus*), had a similar effect on turkey harvest by reducing hunter effort or success (Hansen et al. 1986, Elliott and Harms 2023). We obtained rainfall data from the National Oceanic and Atmospheric Administration (NOAA) National Climatic Data Center (2026). We chose data from a single weather station for each of our five focal counties. Each chosen station was located centrally for that county to represent rainfall for the county more accurately. For each county, we counted the number of days during the spring hunting season that exceeded 2.5 cm of rainfall, a common threshold for when recreational activities like hunting can be affected and effort reduced (Paudyal et al. 2019, Elliott and Harms 2023). We created a second covariate counting the number of days during the hunting season that had rainfall >1.3 cm to determine if less daily rainfall could still potentially decrease hunter effort and reduce harvest. Two variables representing harvest regulations were created for analysis: bag limit (four-, three-, and two-bird limits) and spring season delay (14-d delayed and no delay). We standardized all continuous covariates by z-transforming and evaluated multicollinearity using VIF (Quinn and Keough 2002, Zuur et al. 2007, Shrestha 2020). We created generalized linear mixed models with a negative binomial error distribution using the glmmTMB function in the glmmTMB (version 1.1.14) package in R version 4.4.1 (R Core Team 2026) to assess the relationship between our covariates and adult male wild turkey spring harvest numbers (Brooks et al. 2017). We first created models including the following predictors for 2017–2025: one-year prior juvenile male wild turkey harvest, one-year prior adult male wild turkey harvest, rainfall days >2.5 cm, rainfall days >1.3 cm, bag limit, and 14-d delay. We included an intercept-only (null) model in the model set to evaluate the support of a model with no covariates. All models with ΔAICc < 2 were considered to have substantial support and we evaluated each model with respect to our research hypotheses (Burnham and Anderson 2002). The remaining covariates of hunter effort (2017–2022) and fecundity two years prior (2019–2024) were not available for the full 2017–2025 time period. Therefore, we evaluated these covariates individually without using model selection in univariate models to detect any effect on adult male wild turkey harvest. We fit all models with a binary dummy-coded covariate for the year 2020 to represent the temporary increase in harvest that occurred during the COVID-19 pandemic (Phillips et al. 2024). County was treated as a random effect in all models to account for varying baseline levels of harvest between counties. We used the emmeans (version 1.11.2-8) package in R version 4.4.1 (R Core Team 2026) to perform one-tailed Wald z-tests to test the specific hypotheses for the effects of the covariates on adult male wild turkey harvest (Piaskowski and Lenth 2026). For this analysis, we based conclusions on an $\alpha = 0.05$ decision criterion and further interpreted the results using standardized beta coefficients to explain the effect size. We used the simulateResiduals and plot functions from the DHARMa (version 0.4.7) package in R version 4.4.1 (R Core Team 2026) to test for outliers, test for overdispersion, examine residual-versus-predicted plots, examine quantile-quantile (QQ) plots, and perform the Kolmogorov-Smirnov test for goodness of fit (Hartig 2024).

Fecundity analysis

To assess if harvest on our study sites was removing enough male wild turkeys to reduce the number of females that are successfully bred, we created generalized linear mixed models with a gaussian error distribution using the glmmTMB function in the glmmTMB (version 1.1.14) package in R version 4.4.1 (Exum et al. 1987, Healy and Powell 2001, Brooks et al. 2017, R

Core Team 2026). The fecundity response variable was calculated identically to fecundity in the reported harvest analysis. We used our site-wide harvest rates for each year calculated using the Barker model as the predictor variable in our model. If the model demonstrated a significant decrease in fecundity estimates as harvest rates increased for a given year, this would support the hypothesis that harvest was reducing the fecundity measured for that breeding season. We used the emmeans (version 1.11.2-8) package in R version 4.4.1 (R Core Team 2026) to perform one-tailed Wald z-tests to specifically test if fecundity decreased when harvest increased (Piaskowski and Lenth 2026). For this analysis, we based conclusions on an $\alpha = 0.05$ decision criterion and further interpreted the results using standardized beta coefficients to explain the effect size. We used the simulateResiduals and plot functions from the DHARMa (version 0.4.7) package in R version 4.4.1 (R Core Team 2026) to test for outliers, test for overdispersion, examine residual-versus-predicted plots, examine quantile-quantile (QQ) plots, and perform the Kolmogorov-Smirnov test for goodness of fit (Hartig 2024).

RESULTS

Trapping and home ranges

We trapped and banded 299 male wild turkeys (126 adults and 173 juveniles) from 2017–2021 across the five focal counties. We radio-tagged 265 male wild turkeys (116 adults and 149 juveniles) with backpack transmitters. We included 188 radio-tagged adult male turkeys in the Cox model mortality analysis (111 captured as adults and 77 trapped as juveniles and survived to the adult age class) and 204 (16 with leg bands only) adult male turkeys in the Barker survival analysis. We analyzed 204 of the 299 total birds captured for adult survival and mortality because 17 birds died within the 2-week capture myopathy threshold, 31 died as juveniles and did not reach the adult age class, 3 lost radio signal as juveniles, 20 were band-only juveniles that were never harvested or recaptured, and 24 were recruited to the adult age class after tracking was finished. Of the 188 radio-tagged adult male turkeys monitored, 73 lived for at least one year and we were able to obtain ≥20 locations to create MCP annual home ranges. The mean number of locations/home range was 39.4 (±1.2 SE, 95% CI = 37.0, 41.8). The mean annual home range size for adult male turkeys was 223.6 ha (±19.2 SE, 95% CI = 186.1, 261.2).

Cox model mortality

We documented 142 mortalities of 188 radio-tagged adult male turkeys monitored. Apparent cause of mortality included coyote (n = 18, 12% of mortalities), great horned owl (n = 5, 3%), unknown mammal (n = 11, 8%), legal harvest (n = 83, 58%), poaching (n = 1, 1%), harvest wounding (n = 1, 1%), and unknown (n = 18, 13%; birds that we were unable to gain permission to check on private property). In total, 38 of 142 mortalities (27%) were attributed to predators. Of the 38 predator mortalities, 30 kill sites could be identified as follows: coniferous forest (n = 1, 3%), deciduous forest (n = 18, 60%), hay/pasture (n = 8, 27%), row crop (n = 1, 3%), and young forest (n = 2, 7%) cover types.

We created 56 models in our candidate set (1 null, 25 univariate, and 30 multivariate; Table C.1) to model covariate relationships with harvest mortality. Of the models we fit, two had ΔAICc < 2 and were considered part of the family of best models. The best-supported model included the percentage of developed land and the percentage of hay/pasture interior. This model demonstrated that an increase in developed land and an increase in the hay/pasture interior decreased the risk of harvest mortality (developed: hazard ratio [HR] = 0.72; 85% CI = [-0.50, -0.15], hay/pasture interior: HR = 0.68; 85% CI = [-0.60, -0.16]; Figure C.2). The second best-supported model (ΔAICc = 1.25) demonstrated the same relationships as our top model with an additional covariate (row crop interior). Row crop interior did not have a significant effect (85% CI = [-0.04, 0.19]).

We created 63 models in our candidate set (1 null, 25 univariate, and 37 multivariate; Table C.2) to model covariate relationships with non-harvest mortality. Of the models we fit, three had ΔAICc < 2 and represented the top-ranked models. The best-supported model included the IJI of developed land and the IJI of mature forest. This model demonstrated that an increase in the IJI of developed land and an increase in the IJI of mature forest during the growing season increased the risk of non-harvest mortality (IJI developed: HR = 1.41; 85% CI = [0.10, 0.58], IJI mature forest (growing season): HR = 1.55; 85% CI = [0.13, 0.73]; Figure C.3). The second best-supported model (ΔAICc = 0.27) added the distance to all roads covariate which demonstrated that increased distance to roads increased non-harvest mortality (HR = 1.19; 85% CI = [0.02, 0.36]). However, this covariate's effect size was substantially smaller than the other covariates, indicating a weak relationship.

We created 36 models in our candidate set (1 null, 26 univariate, and 9 multivariate; Table C.3) to model covariate relationships with predation mortality. Of the models we fit, four had ΔAICc < 2 and were considered part of the family of best-supported models. All four models contained the predator index covariate and the other two significant covariates (IJI of deciduous and distance to all roads) had the same demonstrated effect on predation mortality as that demonstrated in our non-harvest mortality model set. We therefore chose to interpret the univariate predator index model alone. The model demonstrated that an increase in the predator index caused a subsequent increase in risk of predation mortality (HR = 1.56; 85% CI = [0.21, 0.67]; Figure C.4). Specifically, the model predicted that a property with a predator index of 0.1

(1 predator on camera/10 days) would lose 15% of its adult male wild turkeys to predation annually, whereas a property with an index of 1 (1 predator/day) would lose 20%.

Wald chi-square tests indicated that harvest rate differed by county, site, and year (county: $\chi^2 = 9.27$; df = 4; P = 0.05, site: $\chi^2 = 26.04$; df = 9; P ≤ 0.01, year: $\chi^2 = 10.67$; df = 4; P = 0.03; Table C.4). The overall adult male spring harvest rate pooled across all sites and years was 0.32 (±0.03 SE; 95% CI = 0.27–0.38). The Wald chi-square tests did not detect any differences in predation for county, site, or year (P > 0.05). The overall annual predation rate pooled across all sites and years was 0.16 (±0.02 SE; 95% CI = 0.11–0.21). Wald chi-square tests indicated non-harvest rate differed by site ($\chi^2 = 17.25$; df = 9; P = 0.04) but not by county or year (P > 0.05). The overall annual non-harvest rate pooled across all sites and years was 0.25 (±0.03 SE; 95% CI = 0.19–0.30).

Barker model survival and harvest

We created 9 models to determine the annual survival rate of adult male wild turkeys (Table C.5). Our two best-supported models were a year-specific survival model (ΔQAICc = 0.00) and a constant survival (null) model (ΔQAICc = 0.98). Annual survival by year was: 2017 (0.44 ±0.08 SE; 95% CI = 0.29–0.59), 2018 (0.31 ±0.06 SE; 95% CI = 0.21–0.43), 2019 (0.46 ±0.07 SE; 95% CI = 0.32–0.60), 2020 (0.25 ±0.06 SE; 95% CI = 0.15–0.39), 2021 (0.52 ±0.07 SE; 95% CI = 0.39–0.66; Table C.4). Annual survival rate pooled across all sites and years was 0.38 (±0.03 SE; 95% CI = 0.32–0.45).

We created 20 models to estimate the annual harvest rate of adult male wild turkeys (Table C.6). Three models were within ΔQAICc < 2 and were considered to have substantial support: a year-specific survival/constant reporting rate model S(year) r(.) (ΔQAICc = 0.00), a year-specific survival/year-specific reporting rate model S(year) r(year) (ΔQAICc = 1.03), and a constant survival/constant reporting rate model S(.) r(.) (ΔQAICc = 1.46). We chose the second model (year-specific survival and year-specific reporting) for harvest rate calculations because we believe it best represented the variation in the data. The third-ranked constant survival and constant reporting rate model was used to calculate harvest rate over all years. Annual harvest rate by year was as follows: 2017 (0.20 ±0.06 SE; 95% CI = 0.07–0.33), 2018 (0.41 ±0.06 SE; 95% CI = 0.29–0.53), 2019 (0.17 ±0.05 SE; 95% CI = 0.07–0.28), 2020 (0.41 ±0.06 SE; 95% CI = 0.28–0.53), 2021 (0.29 ±0.06 SE; 95% CI = 0.17–0.42; Table C.4). Annual harvest rate pooled across all sites and years was 0.31 (±0.03 SE; 95% CI = 0.25–0.36). Harvest and survival were negatively related based on linear regression (slope = -1.66; $F_{1,23}$ = 155.23; $P \leq 0.01$; Figure C.5).

Effects on reported harvest

We created 9 models (1 null, 4 univariate, and 4 multivariate; Table C.7) to model covariate relationships with reported county-wide spring adult male harvest from 2017–2025. The best-supported model included the bag limit covariate. Based on this model, a decrease in the bag limit did not reduce adult male wild turkey harvest based on left-tailed tests (Bag limit 4 vs. 3: β = -0.02, Z = -0.32, P = 0.61; bag limit 4 vs. 2: β = 0.11, Z = 2.04, P = 1.00). The second-ranked model demonstrated the same relationship with bag limit and indicated that increased rainfall did not decrease harvest (β = 0.03, Z = 1.05, P = 0.85). All models with the previous year juvenile male harvest and adult male harvest covariates showed that residuals increased with fitted values. Because covariate transformations and modeling for nonlinearity did not improve model fit, and all other diagnostics were acceptable (uniformity, overdispersion, and QQ plots), this trend in residuals likely indicated unmodeled variation that was not captured by these harvest covariates we had available from 2017–2025. However, adding and removing these harvest covariates from our top models still demonstrated that adult harvest did not decrease as the bag limit was reduced. Univariate models of these harvest covariates demonstrated that an increase in juvenile and adult harvest in a given year did not reduce adult harvest the following year (Juvenile harvest: $\beta = 0.08$, $Z = 2.06$, $P = 0.98$; adult harvest: $\beta = 0.13$, $Z = 1.96$, $P = 0.97$). Models that incorporated the 14-d delay were not supported ($\Delta AICc > 8$; Table C.7), indicating the 14-d delay had no measurable influence on reported harvest. The univariate fecundity model (2017–2022) demonstrated an increase in adult male harvest as fecundity two years prior increased ($\beta = 0.06$; $Z = 2.08$; $P = 0.02$; Figure C.6). The univariate hunter effort model did not demonstrate an increase in harvest as average hunter effort increased ($\beta = 0.02$; $Z = 0.89$; $P = 0.19$). The univariate model investigating harvest effects on fecundity demonstrated that an increase in harvest did not correlate with reduced fecundity in a given year, based on left-tailed tests ($\beta = -0.05$, $t = -0.78$, $df = 22$, $P = 0.22$).

DISCUSSION

We identified relationships between land cover and different adult male wild turkey mortality sources, reported harvest and survival rates for south-central Tennessee, and demonstrated factors that affected spring harvest numbers. The interspersion and juxtaposition of developed land and mature forest cover were related to greater non-harvest mortality, and a greater predator index was related to greater predation mortality as we hypothesized. Survival of adult male wild turkeys was lower and harvest was greater at our study sites than that reported in most contemporary studies in the Southeast (Table C.8). Shifting the spring hunting season two weeks later had no effect on harvest and a reduction in the bag limit from four to two did not reduce harvest, but harvest was greater when fecundity from two years prior increased as we hypothesized. These results indicate that management effort may be better spent on increased habitat management and reducing the meso-mammalian predator community to increase adult male wild turkey survival, fecundity, and recruitment than shifting hunting season dates and bag limits.

The increase in predation mortality with a greater predator index was expected and supported our hypothesis. A greater abundance of coyotes, bobcats, and foxes on the landscape logically will lead to greater predation rates on wild turkeys (Prugh et al. 2009, Ritchie and Johnson 2009, Ripple et al. 2013, Gordon et al. 2015, Wang et al. 2023). Although adult male wild turkeys are considered less vulnerable to predation than female or juvenile turkeys because of their size, a single remote camera at each of our ~1,200-ha sites during winter adequately predicted annual predation mortality (Miller and Leopold 1992, Hughes et al. 2005). More intensive camera monitoring in terms of number of cameras/site and duration of operation may yield even stronger relationships with turkey survival.

Increased mortality as developed land was more interspersed in the home range supported our hypothesis and may reflect a greater density of predator populations near human development. Supplemental food via feeders, refuse, and pet food attracts coyotes and foxes as well as prey for those animals and bobcats (Sorace 2002, Newsome et al. 2014, Stark et al. 2020). Developed areas commonly have increased edge juxtaposing dense woody vegetation with small openings as well as more vehicular traffic, both of which introduce risk of mortality (Urban et al. 1987, Ng et al. 2008, Dawson et al. 2025). Tennessee had an increase in developed land of 21% (219,308 ha) from 1997–2017 and 7 of the 10 fastest-growing cities in Tennessee from 2020–2023 surrounded Nashville and its metro area (United States Department of Agriculture 2020, Tennessee State Date Center 2024). Such increases in developed land may be causing significant negative impacts on wild turkey survival, and as more wild turkey habitat is removed from the landscape, increases in predation pressure may lead to continued wild turkey declines across the state. In contrast, more developed land may reduce harvest mortality as a result of less hunter access and less hunting opportunity within a distance of residential areas; and there typically are fewer turkeys in areas near development (Poudyal et al. 2008, Burke et al. 2019, Sibiya et al. 2025). The slight decrease in non-harvest mortality as the distance to all roads decreased may be related to turkey predators avoiding high-traffic road systems, but the effect size was small indicating that any possible relationship was not substantial (Grinder and Krausman 2001, Kautz et al. 2021, Youngmann et al. 2022).

The increase of non-harvest mortality during the growing season as the interspersion of mature forest increased supported our hypothesis and suggests a relationship with predators and their visibility to wild turkeys. Many wild turkey predators, such as coyotes, bobcats, foxes, and great horned owls, hunt in and around the edges of forest cover where their presence is concealed and visibility may be limited (Grossman et al. 2008, Gorini et al. 2012, Deuel et al. 2017, Jones et al. 2022, Webster et al. 2022). Visual obstruction can provide a hiding place for prey or it can provide ambush sites for predators (Elton 1939, Metcalfe 1984, Lazarus and Symonds 1992). For large-bodied Galliformes, such as wild turkeys, visual obstruction from dense vegetation will almost always benefit the approaching predator because turkeys rely heavily on sight to detect predators and escape by running or taking flight (Blankenship 1992, Miller and Leopold 1992). Therefore, wild turkeys may be ambushed more frequently in forested cover types during the growing season when vegetation is dense and visibility more restricted.

The reduction in the bag limit from four to three to two bearded wild turkeys per hunter during the spring season adopted by the Tennessee Fish and Wildlife Commission was an effort to address concerns about overall wild turkey population declines. This regulatory action, however, had no effect on the total county harvest in our five study counties. This outcome was predictable given our knowledge of hunters in south-central Tennessee. From 2017–2022, only 50% of hunters in our focal counties killed a male turkey, and only 9.6% of hunters in our focal counties harvested ≥ 3 turkeys, indicating how reducing the bag limit was unlikely to have an influence on total harvest because >90% of hunters never harvested > 2 birds (Quehl et al. 2024*b*). Our harvest-survival relationship (Figure C.5) suggested harvest mortality may be additive, similar to other studies (Moore et al. 2008, Wightman et al. 2024). Thus, if reduction in the bag limit actually led to reduced harvest, then adult male turkey abundance could increase in subsequent years because of an increase in survival. However, our results emphasize the need to examine any actual benefit of regulation change, because when regulations are ineffective, they can frustrate hunters and discourage continued participation (Wam et al. 2012, Watkins et al. 2018, Gruntorad and Chizinski 2021, Quehl et al. 2024*b*).

Our hypothesis proposing that harvest of adult male wild turkeys would be related to reproduction from two years prior was supported by our results and highlights the reproduction linkage to population growth (Roberts and Porter 1995, Londe et al. 2023, Lashley et al. 2025, Tyl et al. 2025). Poult survival and nest success are critical vital rates on wild turkey populations and subsequently harvest (Vangilder and Kurzejeski 1995, Londe et al. 2023, Lashley et al. 2025, Tyl et al. 2025). Reduced productivity and subsequent harvest is particularly concerning because Tennessee and other states in the Southeast have recorded a steady decline in poult-per-female ratios over the past few decades (Byrne et al. 2015, Shields 2024). We also hypothesized there would be no relationship between rainfall and harvest, because the wild turkey season in Tennessee is six weeks long, and we surmised that there would be ample opportunity for harvest despite some days during the season with heavy rain. This result directly challenges the notion that low wild turkey harvest in certain years is driven primarily by poor weather.

Our annual survival rate (0.38) for adult male wild turkeys was lower than most contemporary southeastern radio-telemetry studies (Ielmini et al. 1992, Hubbard and Vangilder 2007, Wright and Vangilder 2007, Norman et al. 2022, Wightman et al. 2024; Table C.8). In addition, our annual harvest rate (0.32) was greater than harvest rates reported from the same studies (Table C.8). Stevens et al. (2017) reported a spring harvest ≤ 30% helped maintain high turkey populations, and Vangilder and Kurzejeski (1995) demonstrated that when spring harvest was 30%, the percentage of adult males in the harvest dropped below 70%. However, Vangilder and Kurzejeski (1995) modeled harvest with juvenile male survival ≈ 45%, which is substantially less than juvenile male survival in our study (0.73; see Chapter 2). Increased productivity could offset reduced juvenile male survival. Hen success reported by Vangilder and Kurzejeski (1995) was 40.1%, whereas it was 28.6% in our study (Johnson et al. 2022). Therefore, our greater juvenile male survival but lower female success than Vangilder and Kurzejeski (1995) make it difficult to determine if a harvest rate of 0.32 will negatively affect future harvest. Presumably, if the harvest rate in the counties of our study was too high, we would see years with greater adult male harvest rates followed by years with decreased harvest rates as enough adult males would be removed from the landscape to reduce harvest the following year (Vangilder and Kurzejeski 1995, Stevens et al. 2017). However, the reported harvest analysis demonstrated that greater adult harvest in one year did not cause adult harvest to decline the following year. We also demonstrated that harvest rates at our sites were not great enough to affect fecundity of our female turkeys by reducing the number of females successfully bred. As long as the juvenile male harvest rate does not reduce survival to near 50% as was reported in Vangilder and Kurzejeski (1995) and female productivity remains constant or increases, the annual harvest rate we documented likely is sustainable. Recruitment of juvenile male wild turkeys into the adult age class could be increased by eliminating juvenile male harvest. Our results also demonstrate that logically any management actions that can increase wild turkey fecundity have the potential to increase harvest and promote population growth (Vangilder and Kurzejeski 1995, Stevens et al. 2017, Londe et al. 2023, Lashley et al. 2025, Tyl et al. 2025).

MANAGEMENT IMPLICATIONS

Our data indicate increased forest cover, a greater abundance of wild turkey predators on the landscape, and increased development can have a negative impact on survival of adult male wild turkeys. However, our data showed that harvest rate was the single most important factor determining annual survival for adult male wild turkeys. In spite of this result, regulatory restrictions, including a delayed spring hunting season and a reduction in the bag limit had no effect on wild turkey harvest. Implementing practices such as forest thinning coupled with frequent prescribed fire can increase visibility for wild turkeys and should be considered on properties where increasing adult male survival is desired. Remote cameras can be used to assess the abundance of wild turkey predators on a property and help determine if some level of predator management may be necessary to help increase wild turkey survival and abundance. Thinning and burning to increase open structure and reduce predator abundance can have additional benefits for providing cover for nesting and brood-rearing females, which could impact future adult male wild turkey harvest. The continued loss of wild turkey habitat as a result of human development should be considered when monitoring wild turkey populations in the future, as less available wild turkey habitat on the landscape may be associated with increased predation pressure, population declines or lower harvest numbers in the future. Finally, it is important for state agencies to evaluate regulations related to timing of the turkey season and bag limits. If delays in the season are not increasing harvest or female productivity, then season opening dates could be based on hunter preference for earlier seasons to increase hunter satisfaction. In addition, setting bag limits should be viewed more as a tool to distribute the harvest among hunters than to protect the adult male population. However, we acknowledge bag limit reductions may be necessary on public lands to provide harvest opportunities for more hunters. Eliminating the harvest of juvenile male wild turkeys except for youth seasons also would lead to increased adult males available for spring harvest as a result of increased recruitment.

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APPENDIX C

Table C.1. Top ten models for AICc model selection results for the effect of covariates on adult male wild turkey harvest mortality, south-central Tennessee, 2017–2021. We present the number of parameters (K), Corrected Akaike Information Criterion (AICc), the difference between model rankings (ΔAICc), AICc model weight (AICcWt), and model deviance (Dev).

Model[a]	K	AICc	ΔAICc	AICcWt	Dev
DEV + HAYINT	2	808.66	0.00	0.43	804.61
DEV + HAYINT + ROWINT	3	809.91	1.25	0.23	803.81
ROAD * HAYINT	3	812.91	4.26	0.05	806.82
ROAD + HAYINT	2	814.13	5.47	0.03	810.08
DEV	1	814.38	5.72	0.02	812.36
HAYINT	1	814.38	5.72	0.02	812.36
FOR + DEV	2	814.64	5.98	0.02	810.59
FOR + ROW	2	814.86	6.20	0.02	810.81
ROWINT + HAYINT	2	814.90	6.24	0.02	810.85
ROAD + ROWINT + HAYINT	3	815.49	6.83	0.01	809.39

[a] ROWINT = proportion of row crop interior, HAYINT = proportion of hay/pasture interior, ROW = proportion of row crop, DEV = proportion of developed land, FOR = proportion of all mature forest, ROAD = average distance to roads

Table C.2. Top ten models for AICc model selection results for the effect of covariates on adult male wild turkey non-harvest mortality, south-central Tennessee, 2017–2021. We present the number of parameters (K), Corrected Akaike Information Criterion (AICc), the difference between model rankings (ΔAICc), AICc model weight (AICcWt), and model deviance (Dev).

Model[a]	K	AICc	AICc	AICcWt	Dev
IJIDEV + IJIFOR	3	576.04	0.00	0.21	570.00
IJIDEV + IJIFOR + ROAD	4	576.31	0.27	0.18	568.24
ROAD + IJIFOR	3	578.08	2.04	0.07	572.03
IJIDEC + IJIDEV + IJIHAY + IJIYGF	5	578.26	2.22	0.07	568.16
IJIFOR	2	578.28	2.24	0.07	574.26
IJIDEV	1	578.96	2.92	0.05	576.95
ROW + IJIDEV	2	579.41	3.37	0.04	575.39
IJIDEV + ROAD	2	579.69	3.65	0.03	575.67
IJIDEV + FORINT	2	579.70	3.66	0.03	575.68
IJIDEV + FORINT + HAYINT	4	580.08	4.04	0.03	572.01

[a] FORINT = proportion of forest interior, HAYINT = proportion of hay/pasture interior, ROW = proportion of row crop, IJIDEC = interspersion and juxtaposition index of deciduous forest, IJIDEV = interspersion and juxtaposition index of developed land, IJIFOR = interspersion and juxtaposition index of all mature forest, IJIHAY = interspersion and juxtaposition index of hay/pasture, IJIYGF = interspersion and juxtaposition index of young forest, ROAD = average distance to roads

Table C.3. Top ten models for AICc model selection results for the effect of covariates on adult male wild turkey predation mortality, south-central Tennessee, 2017–2021. We present the number of parameters (K), Corrected Akaike Information Criterion (AICc), the difference between model rankings (ΔAICc), AICc model weight (AICcWt), and model deviance (Dev).

Model[a]	K	AICc	ΔAICc	AICcWt	Dev
PRED + IJIDEC	3	145.85	0.00	0.17	139.76
PRED	1	146.09	0.23	0.15	144.07
PRED + ROAD	2	146.38	0.52	0.13	142.33
PRED + DEV	2	147.46	1.61	0.07	143.42
ROAD	1	148.26	2.41	0.05	146.25
ROAD + IJIDEC	3	148.34	2.49	0.05	142.25
CON	1	149.14	3.29	0.03	147.12
DEV	1	149.80	3.95	0.02	147.78
IJIES	2	149.88	4.03	0.02	145.84
ROAD + HAYINT	3	149.92	4.07	0.02	143.83

[a] CON = proportion of coniferous forest, HAYINT = proportion of hay/pasture interior, ROW = proportion of row crop, DEV = proportion of developed land, IJIDEC = interspersion and juxtaposition index of deciduous forest, IJIES = interspersion and juxtaposition index of early succession, PRED = site level predator index, ROAD = average distance to roads

Table C.4. Annual survival and harvest rate by year from both the Barker joint live and dead encounters and resighting model and Cox proportional hazards model for adult male wild turkeys, south-central Tennessee, 2017–2021.

Year	Barker survival rate	Barker harvest rate	Cox harvest rate
2017	0.44 (0.29–0.59)	0.20 (0.07–0.33)	0.20 (0.08–0.31)
2018	0.31 (0.21–0.43)	0.41 (0.29–0.53)	0.43 (0.31–0.54)
2019	0.46 (0.32–0.60)	0.17 (0.07–0.28)	0.21 (0.08–0.34)
2020	0.25 (0.15–0.39)	0.41 (0.28–0.53)	0.41 (0.28–0.54)
2021	0.52 (0.39–0.66)	0.29 (0.17–0.42)	0.31 (0.20–0.42)

Table C.5. QAICc model selection results for Barker joint live and dead model with parameters survival (S), recapture (p), reporting (r), resight (R), resight before death (R'), site fidelity (F), and temporary emigration (F') for adult male wild turkey survival calculations, south-central Tennessee, 2017–2021. We present the number of parameters (K), Corrected Quasi-Akaike Information Criterion (QAICc), the difference between model rankings (ΔQAICc), model weight (QAICcWt), and model deviance (QDev).

Model[a]	K	QAICc	ΔQAICc	QAICcWt	QDev
S(YR), p(.), r(MK2), R(MK1), R'(MK1), F(1), F'(1)	10	472.89	0.00	0.55	206.10
S(1), p(.), r(MK2), R(MK1), R'(MK1), F(1), F'(1)	6	473.87	0.98	0.34	215.42
S(CNTY + YR), p(.), r(MK2), R(MK1), R'(MK1), F(1), F'(1)	14	477.01	4.12	0.07	201.72
S(CNTY), p(.), r(MK2), R(MK1), R'(MK1), F(1), F'(1)	10	478.19	5.31	0.04	211.41
S(SITE + YR), p(.), r(MK2), R(MK1), R'(MK1), F(1), F'(1)	19	483.83	10.95	0.00	197.68
S(SITE), p(.), r(MK2), R(MK1), R'(MK1), F(1), F'(1)	15	484.33	11.45	0.00	206.89
S(.), p(.), r(.), R(.), R'(.), F(.), F'(.)	5	503.12	30.23	0.00	246.73
S(YR), p(YR), r(YR), R(YR), R'(YR), F(YR), F'(YR)	18	506.14	33.25	0.00	222.18
S(CNTY * YR), p(.), r(MK2), R(MK1), R'(MK1), F(1), F'(1)	30	506.84	33.96	0.00	195.79

[a] CNTY = county, MK1 = mark type (transmitter estimated and band only fixed to 0), MK2 = mark type (transmitter and band estimated), SITE = study site, YR = year, (.) = constant, (1) = fixed at 1

Table C.6. Top ten models for QAICc model selection results for Barker joint live and dead model with parameters survival (S), recapture (p), reporting (r), resight (R), resight before death (R'), site fidelity (F), and temporary emigration (F') for adult male wild turkey harvest calculations, south-central Tennessee, 2017–2021. We present the number of parameters (K), Corrected Quasi-Akaike Information Criterion (QAICc), the difference between model rankings (ΔQAICc), model weight (QAICcWt), and model deviance (QDev).

Model[a]	K	QAICc	ΔQAICc	QAICcWt	QDev
S(YR), p(.), r(.), R(MK1), R'(0), F(1), F'(1)	8	587.59	0.00	0.33	272.10
S(YR), p(.), r(YR), R(MK1), R'(0), F(1), F'(1)	12	588.61	1.03	0.20	264.71
S(.), p(.), r(.), R(MK1), R'(0), F(1), F'(1)	4	589.04	1.46	0.16	281.81
S(YR), p(.), r(CNTY), R(MK1), R'(0), F(1), F'(1)	12	589.96	2.38	0.10	266.05
S(CNTY + YR), p(.), r(.), R(MK1), R'(0), F(1), F'(1)	12	591.64	4.05	0.04	267.73
S(YR), p(.), r(SITE), R(MK1), R'(0), F(1), F'(1)	17	591.77	4.18	0.04	257.10
S(.), p(.), r(.), R(.), R'(.), F(.), F'(.)	5	592.37	4.79	0.03	283.09
S(CNTY + YR), p(.), r(YR), R(MK1), R'(0), F(1), F'(1)	16	592.69	5.10	0.03	260.19
S(CNTY), p(.), r(.), R(MK1), R'(0), F(1), F'(1)	8	593.42	5.84	0.02	277.93
S(CNTY + YR), p(.), r(CNTY), R(MK1), R'(0), F(1), F'(1)	16	594.02	6.43	0.01	261.52

[a] CNTY = county, MK1 = mark type (transmitter estimated and band only fixed to 0), SITE = study site, YR = year, (.) = constant, (1) = fixed at 1, (0) = fixed at 0

Table C.7. AICc model selection results for the effect of covariates on adult male wild turkey reported harvest, south-central Tennessee, 2017–2025. We present the number of parameters (K), Corrected Akaike Information Criterion (AICc), the difference between model rankings (ΔAICc), AICc model weight (AICcWt), and model deviance (Dev).

Model[a]	K	AICc	ΔAICc	AICcWt	Dev
BAG	6	536.88	0.00	0.35	522.67
RAINH + BAG	7	538.61	1.73	0.15	521.58
RAINM	5	538.88	2.00	0.13	527.34
RAINM + BAG	7	539.33	2.46	0.10	522.31
RAINH	5	539.51	2.64	0.09	527.97
DLY	5	539.60	2.72	0.09	528.06
RAINM + DLY	6	541.08	4.20	0.04	526.87
RAINH + DLY	6	541.27	4.39	0.04	527.06
NULL	2	618.55	81.67	0.00	614.26

[a] BAG = bag limit, DLY = spring hunting season delay, RAINH = days during hunting season with > 2.54 cm rainfall, RAINM = days during hunting season with > 1.27 cm rainfall, NULL = no covariates

Table C.8. Adult male wild turkey survival estimates, harvest estimates, and year of study for various southeastern US studies.

Study Area	Survival Estimate	Spring Harvest (%)	Years of Study	Citation
Virginia	0.65	27	2004–2007	Norman et al. 2022
West Virginia	0.62	23	2004–2007	Norman et al. 2022
East LA[a]	0.48	19	2020–2022	Wightman et al. 2024
Mountain NC[b]	0.46	20	2020–2022	Wightman et al. 2024
West LA[a]	0.41	22	2013–2022	Wightman et al. 2024
Georgia	0.39	23	2017–2022	Wightman et al. 2024
Webb SC[c]	0.38	23	2014–2018	Wightman et al. 2024
Piedmont NC[b]	0.38	23	2020–2022	Wightman et al. 2024
South-central TN[d]	0.38	32	2017-2021	Our study
Missouri Ozarks SSA[e]	0.37	N/A	1989–2000	Hubbard and Vangilder 2005
Coastal NC[b]	0.37	24	2020–2022	Wightman et al. 2024
Missouri Ozarks PRCA[f]	0.37	N/A	1989–2000	Hubbard and Vangilder 2005
Georgia	0.36	N/A	1989–1990	Ielmini et al. 1992
West KY[g]	0.28	61	1995–2002	Wright and Vangilder 2007

[a]Louisiana
[b]North Carolina
[c]South Carolina
[d]Tennessee
[e]South Study Area
[f]Peck Ranch Conservation Area
[g]Kentucky

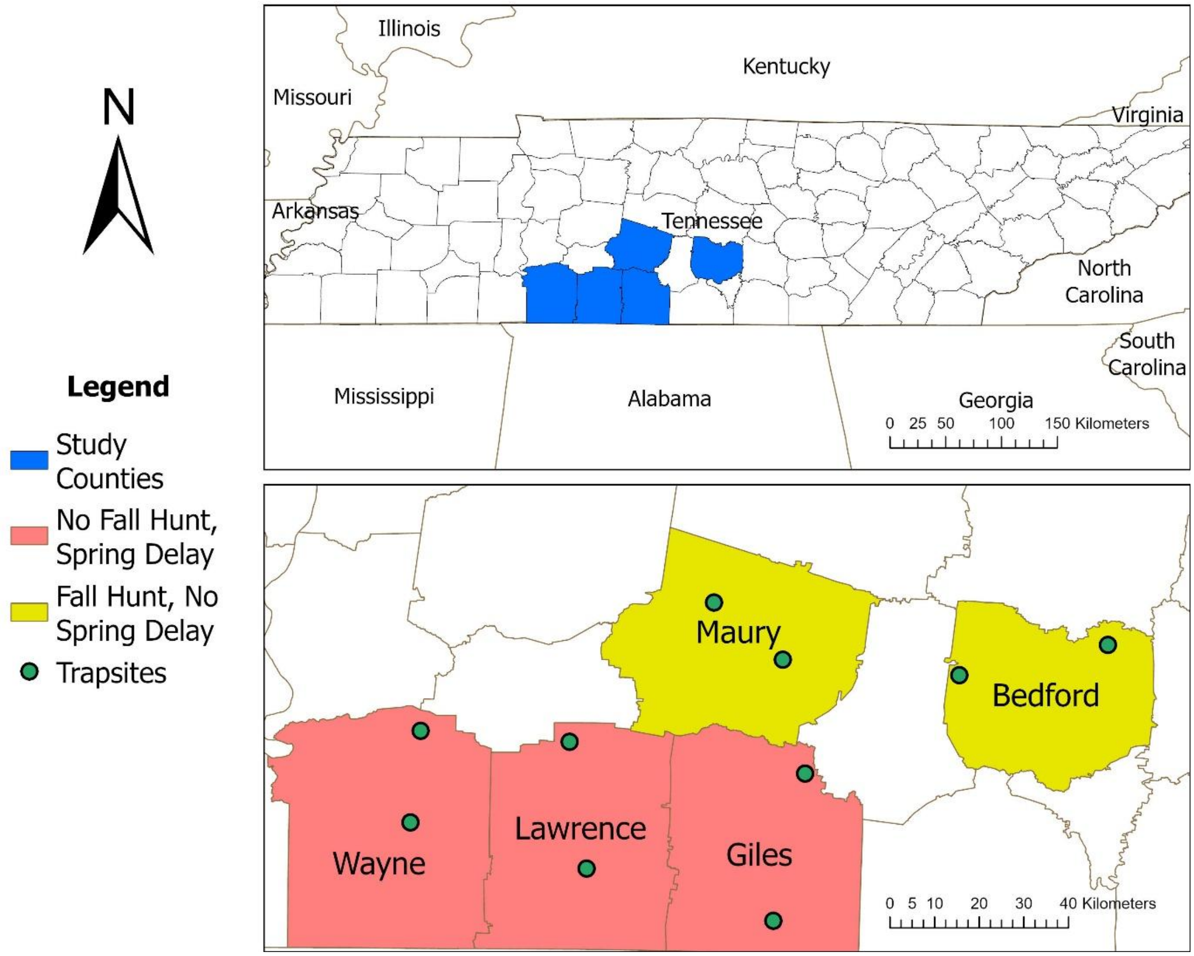

Figure C.1. Our five study counties and associated northern and southern trap sites for adult male wild turkeys, south-central Tennessee, 2017–2021. Counties are separated by whether an experimental 14-day spring hunting season delay was present in the county from 2021–2022 and whether the county had a fall turkey season.

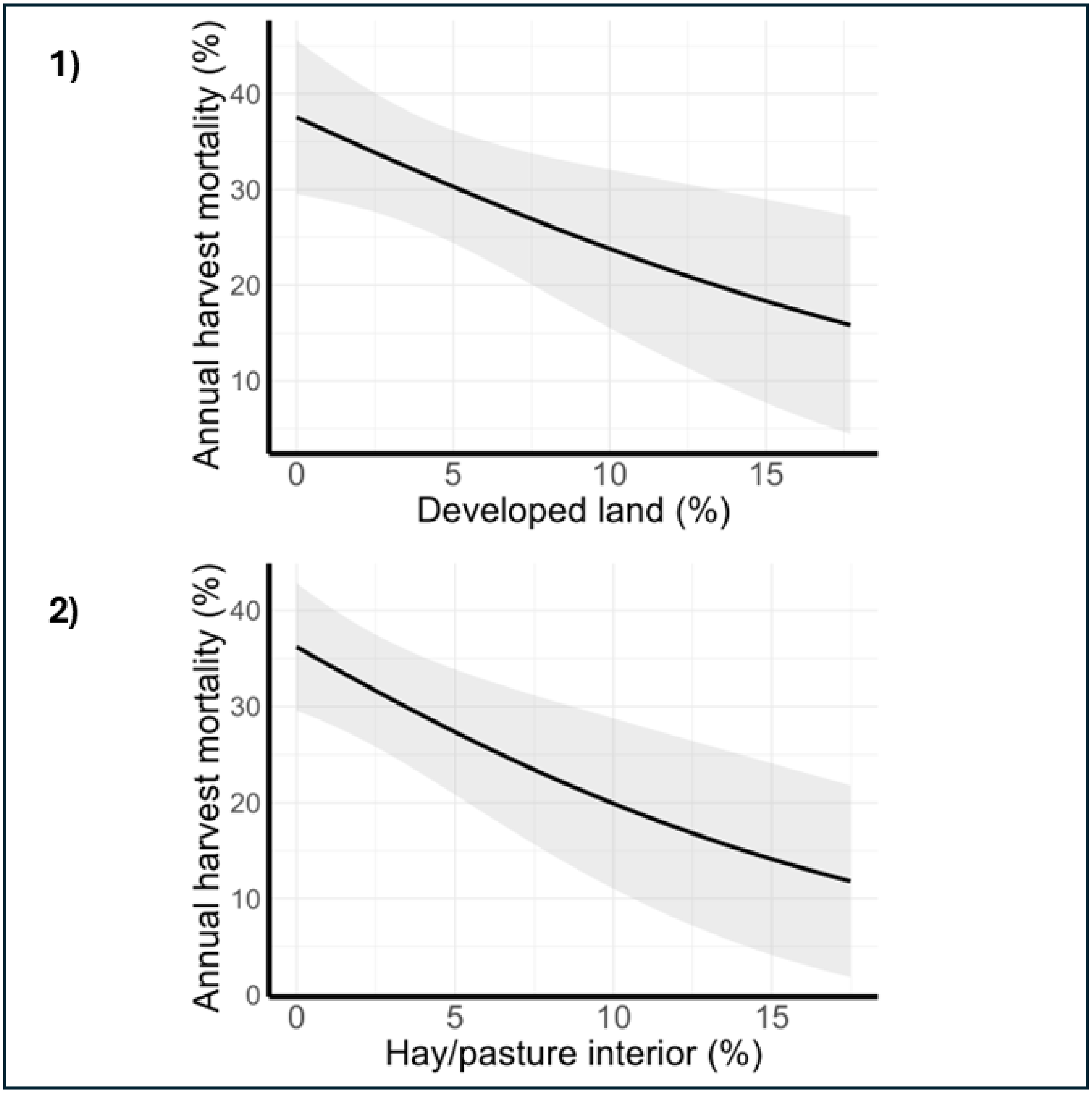

Figure C.2. Predicted annual harvest mortality for adult male wild turkeys, south-central Tennessee, 2017–2021. 1) The relationship between the percentage of developed land in the wild turkey home range and harvest mortality. 2) The relationship between the percentage of hay/pasture interior in the wild turkey home range and harvest mortality. The shaded area represents the 95% confidence interval around the prediction.

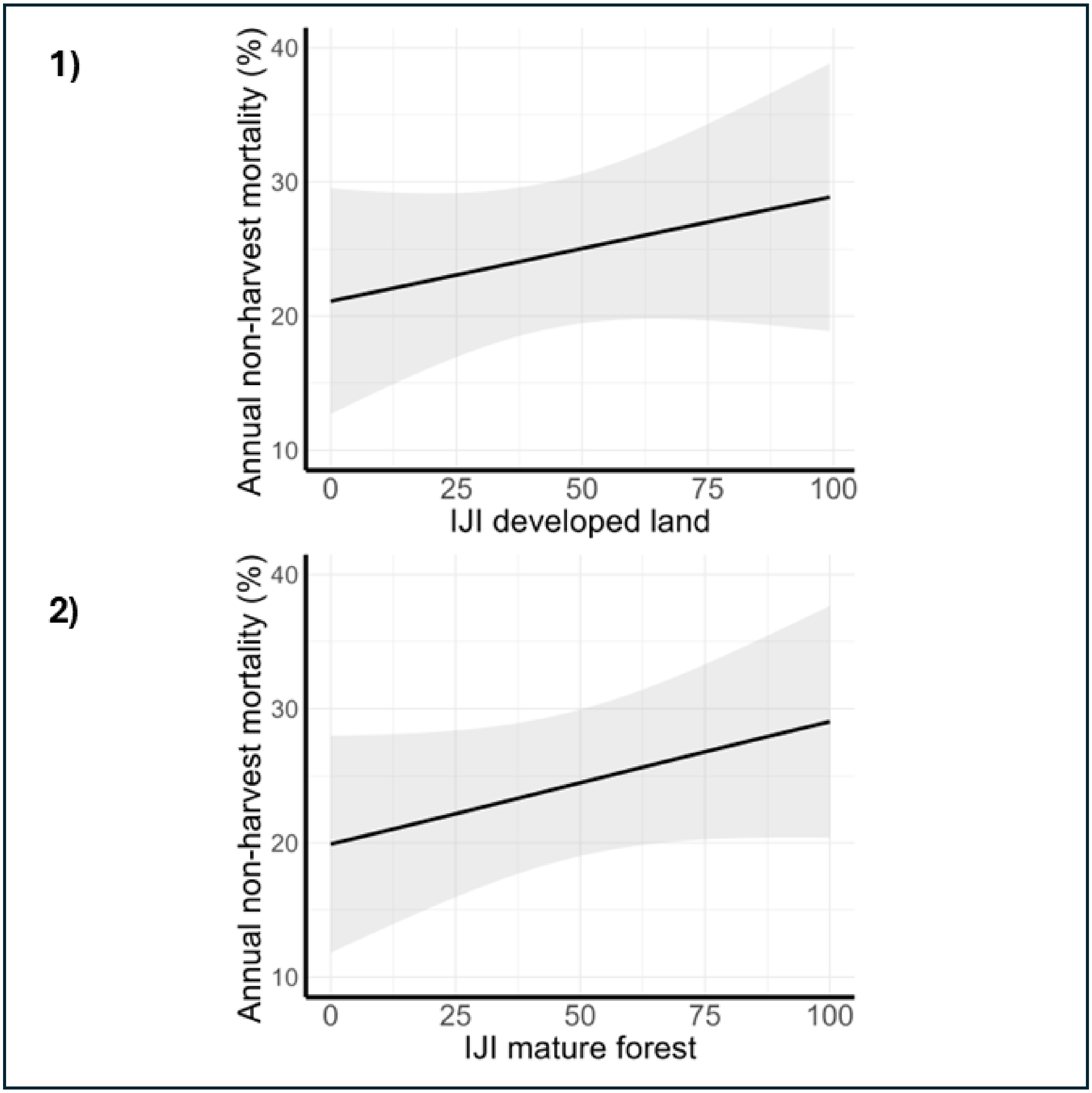

Figure C.3. Predicted annual non-harvest mortality for adult male wild turkeys, south-central Tennessee, 2017–2021. 1) The relationship between the interspersion and juxtaposition index (IJI) of developed land in the wild turkey home range and non-harvest mortality. 2) The relationship between the IJI of mature forest in the wild turkey home range and non-harvest mortality. The shaded area represents the 95% confidence interval around the prediction.

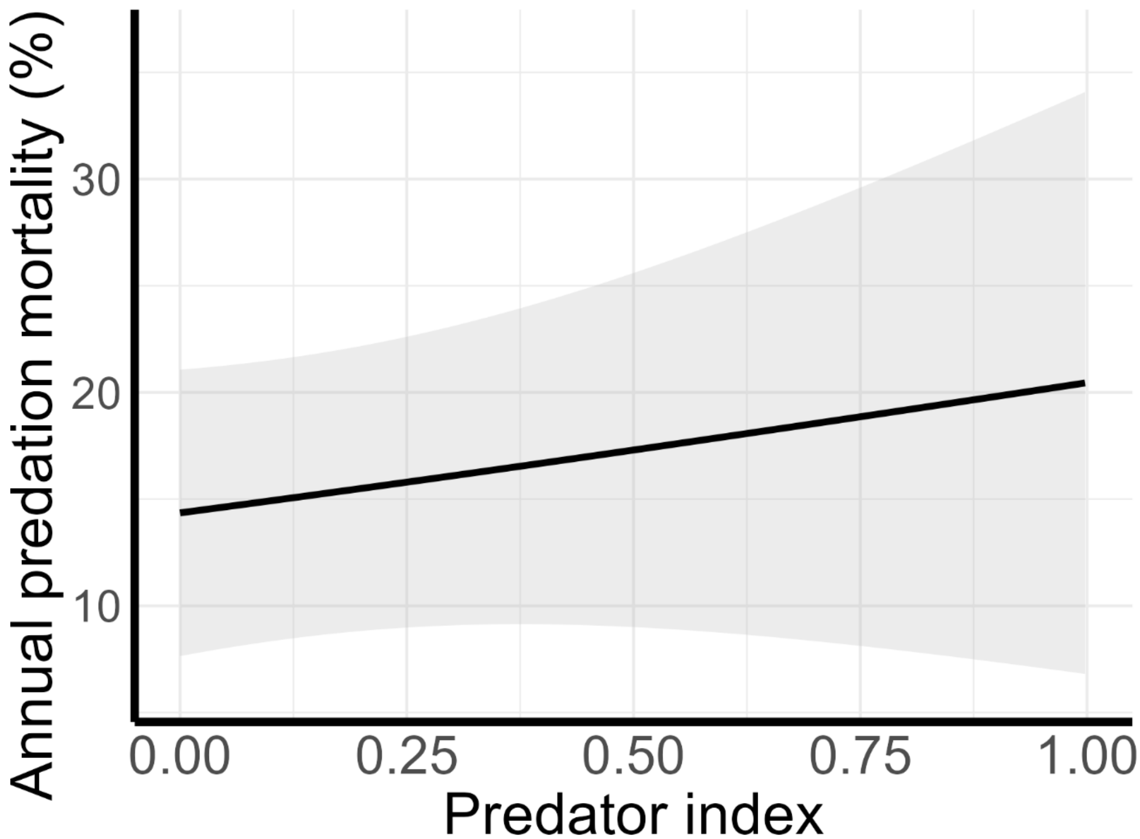

Figure C.4. Predicted annual predation mortality for adult male wild turkeys, south-central Tennessee, 2017–2021. The relationship between the predator index on a given site and predation mortality. The shaded area represents the 95% confidence interval around the prediction.

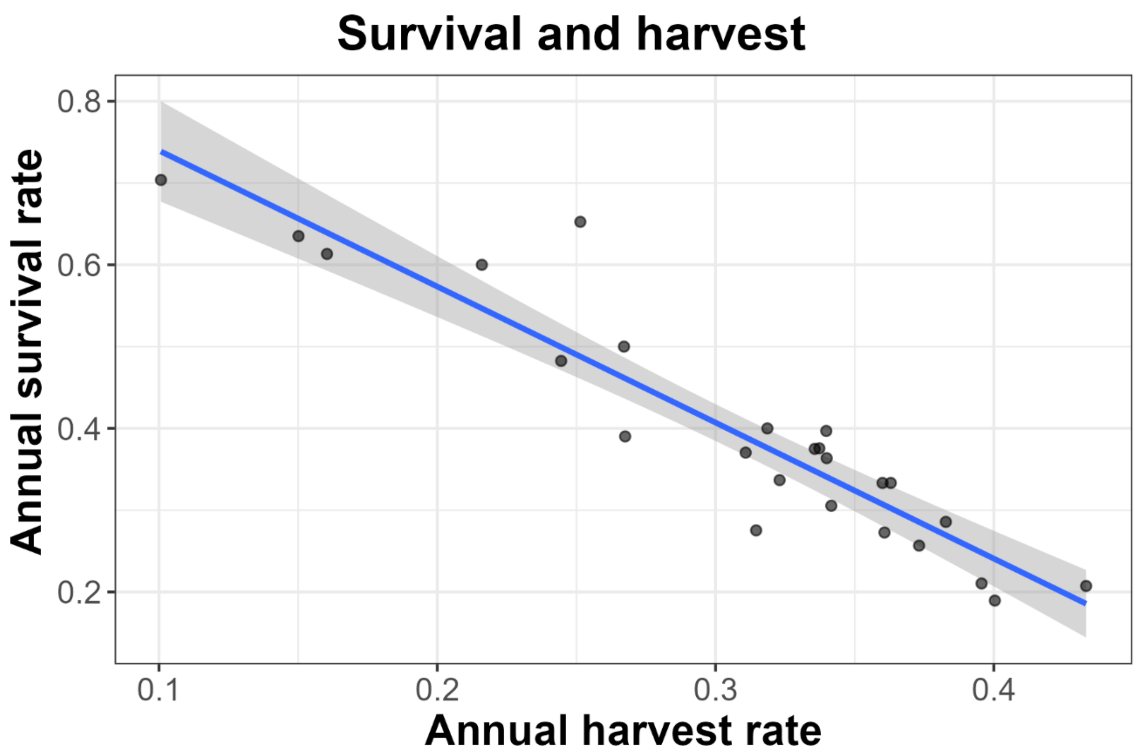

Figure C.5. The regression of the relationship between annual survival rate and annual harvest rate calculated using the Barker joint live and dead encounters and resighting model at each study site for adult male wild turkeys, south-central Tennessee, 2017–2021.

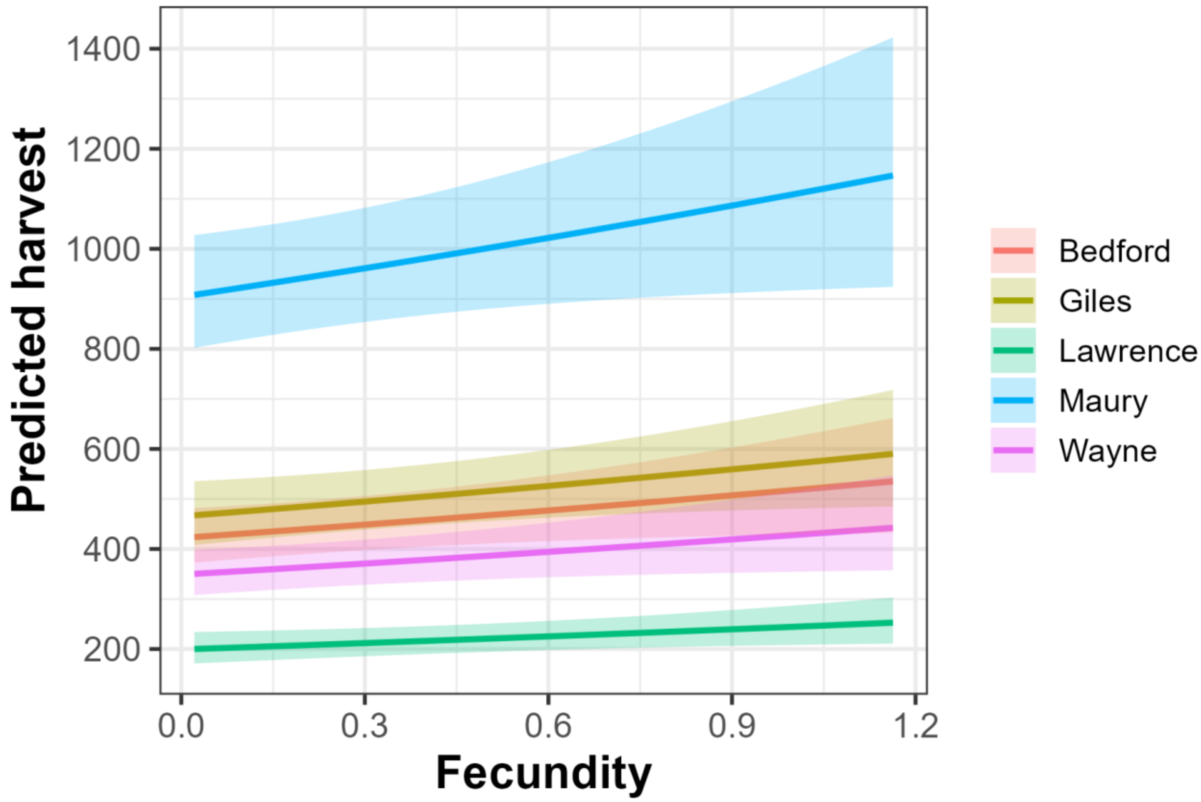

Figure C.6. The predicted relationship between fecundity (expected number of male poults with each hen at 28 days) from two years prior and spring harvest of adult male wild turkeys, south-central Tennessee, 2017–2022. Predicted county-wide harvest is shown with each county assigned its own random effect intercept representing the different baseline harvest numbers for each county. Shaded areas represent 95% confidence intervals around predictions.

CHAPTER 4. MANAGEMENT IMPLICATIONS FOR WILD TURKEYS IN TENNESSEE

1. **Key Finding: Cover types that reduce visibility, such as young forest and other woody thickets, can reduce wild turkey survival by increasing predator success.**

   Male wild turkeys in our study had greater mortality as young and mature forest cover types were more abundant and more interspersed around the home range. Wild turkeys are generalists, using forest for thermal cover, hard mast, and roosting while also using open fields for foraging on invertebrates, seeds, and agricultural crops. However, in Tennessee, forest cover is not limited, as 52% of the state is comprised of forest cover whereas in other states where turkeys persist, such as Kansas, forests compose 5% of the state. Therefore, thinning and burning some existing forest cover to reduce relatively large, homogenous patches of forest can improve visibility for wild turkeys and reduce mortality. For example, our model demonstrated that reducing deciduous forest cover from 80% to 60% on the landscape could result in 11% more juvenile male turkeys surviving each year. Promoting a matrix of forest clearing and thinning paired with frequent prescribed fire (1–3-year return interval) can reduce dense woody cover and can limit ambush opportunities for turkey predators. Our research also demonstrates how some level of predator removal may directly improve wild turkey survival on a site.

2. **Key Finding: Human development may increase wild turkey mortality by increasing predator densities associated with dense edges but might also reduce harvest-based mortality.**

   Wild turkeys in our study experienced an increase in mortality as development in the home range increased. Areas managed specifically for wild turkeys should be farther from roads and human development, where dense edges are present and predator densities often are greater because of anthropogenic food sources that attract meso-mammalian predators. Specific management actions on a property surrounded by human development may be limited. As wild turkey habitat continues to decline and is replaced by developed land, wild turkey populations and harvest likely will decline. The impact of development on turkey populations should be considered before harvest regulations are considered a cause for a population decline.

3. **Key Finding: Wild turkey harvest rates in central Tennessee are currently relatively high but appear sustainable.**

   Thirty percent annual harvest often is cited as a threshold that should not be exceeded when managing harvest rates, but this number is based on past research that is specific to certain demographic conditions. For our study sites, with 29% female reproductive success and 73% juvenile male survival, the wild turkey mean annual harvest rate of 32% did not show any evidence of negatively affecting future harvest or reproduction. However, if the poult-per-female ratio continues to decline, current harvest rates may not be sustainable as recruitment may fail to compensate for harvest. Continued monitoring of juvenile male, adult male, and female survival and fecundity are warranted to understand the role harvest is having in population regulation.

4. **Key Finding: The 14-day spring delay and bag limit change did not affect adult harvest.**

   Given that recent regulatory changes have not had any measurable effect on the harvest or reproduction of wild turkeys in Tennessee, the Tennessee Fish & Wildlife Commission may be able to increase hunter satisfaction and participation by opening the spring hunting season on the first Saturday in April, consistent with findings of Quehl et al. (2024), who reported a 14-day delay had no effect on wild turkey productivity when compared to starting the season on April 2, 3, and 4. The Commission also could consider increasing the bag limit to 3, but if an increased adult male age structure is desired, the bag limit may be left at 2. It is critical to base regulation changes on scientific evidence to help ensure confidence in such decisions and provide sound reasoning for change to the hunting public.

5. **Key Finding: Eliminating harvest of juvenile male wild turkeys could increase the harvest of adult male birds.**

   The survival rate of juvenile male wild turkeys in our study demonstrated that 3 out of every 4 juvenile males not harvested during hunting season will be recruited to the adult age class. TWRA could recommend the elimination of juvenile male harvest except for one per youth during the youth hunting season. Mississippi and Arkansas have demonstrated positive effects on adult harvest after eliminating juvenile male harvest. This regulation could have a meaningful impact on recruitment and increased opportunity to harvest adult male turkeys because some counties in Tennessee report up to 20% of their total harvest composed of juvenile males in certain years.

VITA

Christian Smith grew up near Cleveland, Ohio where he explored every creek and stretch of woods he could find. His true passion for wildlife was nurtured by his parents when he spent time at his hunting property near Parker, Pennsylvania. It was here that he gained a love for the outdoors and an appreciation for the natural world. Christian attended Hocking College and West Virginia University for his undergraduate education in Wildlife and Fisheries Resource Management. Before coming to the University of Tennessee, Christian worked in South Dakota on Canada goose control, in North Carolina on breeding bird surveys, in Pennsylvania on duck and songbird banding, and in West Virginia on a wild turkey project. After graduation, Christian plans to start a job as a wildlife biologist for a state or federal agency.