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WINTER SURVIVAL AND RESOURCE USE OF TRANSLOCATED NORTHERN BOBWHITE IN THE MID-ATLANTIC UNITED STATES

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ABSTRACT

Northern bobwhite (*Colinus virginianus*; hereafter, bobwhite) in the Mid-Atlantic United States have been experiencing precipitous population declines due to a combination of habitat deterioration, urban and suburban sprawl, change in forest management regimes, and farming practices. In recent years, restocking of bobwhite through translocation efforts has gained interest to rebuild local populations. However, empirical studies are warranted to understand the limitations of translocation as it relates to its potential use for long-term population recovery and persistence in this region. Further, few studies nation-wide have evaluated resource use and survival during the non-breeding season on translocated sites. As such, we translocated 360 bobwhites from source populations from southern latitudes during March–April 2015–2017 into 2 different landscape types similar to the source population vegetative communities (agricultural cropland dominated in Maryland, USA versus pine forest dominated in New Jersey, USA) and tested the effects of habitat fragmentation on survival and habitat use during the subsequent non-breeding season. We found habitat fragmentation negatively affecting survival and resource use among translocated bobwhite on fragmented cropland-dominated sites as compared to larger unfragmented forested sites. Survival was lower on cropland sites compared to forested sites such that bobwhite in cropland-dominated landscapes were >125 times less likely to survive the winter than those on forested sites. In our examination of resource use, bobwhite in structurally complex forested sites used cut pine, early-successional woody, early-successional herbaceous, and thinned pine more than what was available on the landscape. On the cropland sites bobwhite used food plots, early-successional woody, and mixed woods more than what was available on the landscape and only food plots at the home range scale. While larger unfragmented forested bobwhite habitat ultimately provided a more successful translocation landscape, birds still had large home ranges and relatively low survival. Therefore, proper pine management may be necessary to optimize habitat availability during the non-breeding season. Our findings provide rare information on demographic resiliency and resource use for translocated bobwhite during the non-breeding season. Further, this research provides valuable information to improve future translocation efforts in the Mid-Atlantic.

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Northern bobwhite (*Colinus virginianus*; hereafter, bobwhite) are one of the most studied and managed game birds in the world (Brennan et al. 2014), yet their numbers continue to decrease throughout much of their distribution. Bobwhite populations have been experiencing precipitous distribution-wide declines and distribution contractions for more than 50 years (Sauer et al. 2017). The rapid declines in bobwhite populations are largely attributed to loss of habitat to urban and suburban sprawl, changes in farming techniques with increased mechanization and clean farming practices, removal of prescribed fire, and habitat fragmentation (Brennan 1991, Church and Taylor 1992, Roseberry and Sudkamp 1998, Peterson et al. 2002, Williams et al. 2004). The most dramatic declines can be observed in the Mid-Atlantic United States. For example, bobwhite populations declined at a rate of 11.2%/year in New Jersey and 9.5%/year in Maryland between 1966–2017 compared to the average distribution-wide decline of 3.4% per year (Sauer et al. 2017). Furthermore, an accelerated decline since the 1980s has led to the functional extirpation of wild populations in New Jersey (Chanda and Herrighty 2011) and Pennsylvania (McKenzie et al. 2015, Martin et al. 2017). This observation supports the "abundant center" hypothesis (Andrewartha and Birch 1954, Hengeveld and Haeck 1982) that as distribution-wide populations decline, peripheral populations are more likely to go extinct and geographic distribution will contract (Goel and Richter-Dyn 1974, Tracy and George 1992, Mehlman 1997, Vucetich et al. 2000), often due to density-independent stochasticity rather than density-dependent maintenance (Williams et al. 2003). This concept heightens the concern for the bobwhite populations in the Mid-Atlantic given that this region is currently in the northern extent of their distribution.

New Jersey is currently ranked as the state with the greatest human population density, averaging more than 386 people/km² (U.S. Census Bureau 2010). Because of the continued urban and suburban expansion, there is amplified competition between creating more human infrastructure and setting aside land for wildlife conservation. Because bobwhite often require multiple landscape types and are a relatively low-mobility species (Terhune et al. 2010, Scott et al. 2013), this increased habitat fragmentation makes bobwhite more vulnerable to predation (Rollins and Carroll 2001, Lyons et al. 2009). For example, Duren et al. (2011) found bobwhite populations in the Mid-Atlantic were most negatively affected by the cohesion of human development within 2.5 km of their potential occupied sites. New Jersey land use has transitioned from predominantly crop and livestock agricultural lands and forests to urban and suburban sprawl with nearly 5,665 ha converted each year (Lohr 2009, Chanda and Herrighty 2011).

In addition to human population expansion, changes in agricultural and forestry practices in New Jersey and elsewhere in the United States have amplified the loss of sufficient habitat. Traditionally, production agricultural land possessed numerous field borders and hedgerows between crops which served as escape or thermal cover for adults and optimal brood cover for bobwhite chick rearing as well as additional insect and seed food sources (Klimstra 1982, Moorman et al. 2013). However, as farming technology advanced and the average farm size increased, bobwhite habitat has been reduced in the landscape (Klimstra 1982, Brennan 1991). Forest management in New Jersey and the rest of the eastern United States has changed due to a reduction in both prescribed fire and forest thinning (Brennan 1991), increasing canopy closure and thus reducing sunlight to the forest floor for early successional plants. Maintaining pine (*Pinus* spp.) canopy closure of <50% paired with prescribed burning is essential in habitat management (Brennan 1999) and is one of the most effective and cost-efficient management tools for enhancing the growth of bobwhite plant foods, increasing biomass of insects for broods, and removal of thick understory growth (Stoddard 1931, Hurst 1972).

Decreasing trends in available habitat and increased fragmentation across the landscape reduce local populations of wildlife, limiting opportunities for colonization and population growth following habitat restoration. Because local sources of bobwhite are not available in the Mid-Atlantic region, long-distance translocations would be required for recolonization. Translocation of bobwhite has demonstrated promise for restoring bobwhite populations in the Southeast (Terhune 2008; Terhune et al. 2006a, 2010), but few studies have tested its efficacy in the Mid-Atlantic. Translocation can mitigate the behavioral and genetic problems associated with captive breeding programs, thus producing comparable survival rates, nest production, and nest survival to resident bobwhites (Terhune 2008, Terhune et al. 2006a). However, this outcome can be dependent on a few factors, including the number of birds translocated from one or multiple donor source populations and whether there are conspecifics present at the release site. Specifically, for social species such as bobwhite, the addition of conspecifics to the landscape may help to change translocated individuals' perceptual error of poor habitat by translocated individuals by signaling to dispersers that the habitat is suitable (Bayard and Elphick 2012, Andrews et al. 2015). However, translocated birds may also perceive habitat as inferior and disperse—leaving the site no better off than before translocation (Martin et al. 2017). Research by Terhune (2006a) indicated that translocating wild bobwhite prior to the breeding season can enhance the adult breeding population and subsequent fall population, making it a viable option for recolonizing bobwhite populations. Furthermore, reported numbers of coveys detected on release sites have more than doubled after translocation efforts in Georgia, USA (Terhune 2006a, Sisson et al. 2017). Despite these successes, long-distance translocations present additional challenges (e.g., extended holding times, increased stress, climate variability between source site and release site) for bobwhite, potentially hindering demographic performance and resource use post-release (Coppola et al. 2021).

The Northern Bobwhite Conservation Initiative identified regional focal areas for bobwhite habitat improvement and restoration efforts. Priority areas in the Mid-Atlantic included the southwestern agricultural lands and the south-central Pine Barrens of New Jersey and much of the western Delmarva Peninsula of Delaware and Maryland (Delmarva; Figure 1).

Because these focal areas historically had bobwhite populations and currently have areas of focused habitat management (Delmarva and southwestern New Jersey) or have large contiguous tracts of managed land (New Jersey Pine Barrens), these sites are ideal candidates for bobwhite translocation. However, empirical studies are warranted to understand the limitations of translocation as it relates to its potential use for long-term population recovery and persistence in this region, especially given the limited wild bobwhite resource.

Translocation research typically focuses on immediate demographic responses during the initial breeding season as negative effects are most severe during the first few months post-translocation (Ewen et al. 2010, Terhune et al. 2010, Coppola et al. 2021). Few studies, however, have evaluated resource use and survival during the non-breeding season on translocated sites. As such, we translocated bobwhite into 2 different landscape types (agricultural cropland dominated versus forest dominated) and tested the effects of habitat fragmentation during the non-breeding season. We predicted that no difference in survival would be observed between the cropland and forested sites for translocated birds given that the lack of site familiarity was equal among translocated birds and given equal time-opportunity to assimilate to their new surroundings. That said, we expected differential resource use would be observed among forested and cropland sites, whereby bobwhite would adaptively use resources beneficial to fitness.

STUDY AREA

We conducted this research during 2015–2017 on 3 privately owned properties within the Mid-Atlantic region of the United States, near the northeastern periphery of the current bobwhite distribution (Brennan et al. 2014): Pine Island Cranberry Company (Pine Island) in Burlington County, New Jersey; Turner's Creek Farm (Turner's Creek) in Kent County, Maryland; and Chino Farm in Queen Anne's County, Maryland (Figure 1). Releases occurred at 2 distinct sites on Pine Island, Home Farm and Sim Place, which were separated (~6 km) by Penn State Forest; therefore, 4 release sites (Home Farm, Sim Place, Chino Farm, and Turner's Creek) between the 2 states were designated for this study. All properties were within the Atlantic Coastal Plain physiographic region, though they differed in landscape composition, predator community, and existing bobwhite population prior to translocation. There were no conspecifics present on Home Farm, Sim Place, or Turner's Creek before the translocations described herein. Chino Farm had a small but recovering population at the time of initial release. We justify comparing these sites in this study because of their relative proximity, similarity in climate and local weather events, and the simultaneous translocations occurring among them.

Pine Island (~6,800 ha) is a working cranberry farm situated near the geographic center of the New Jersey Pinelands National Reserve (~39.740°N, –74.500°W; Figure

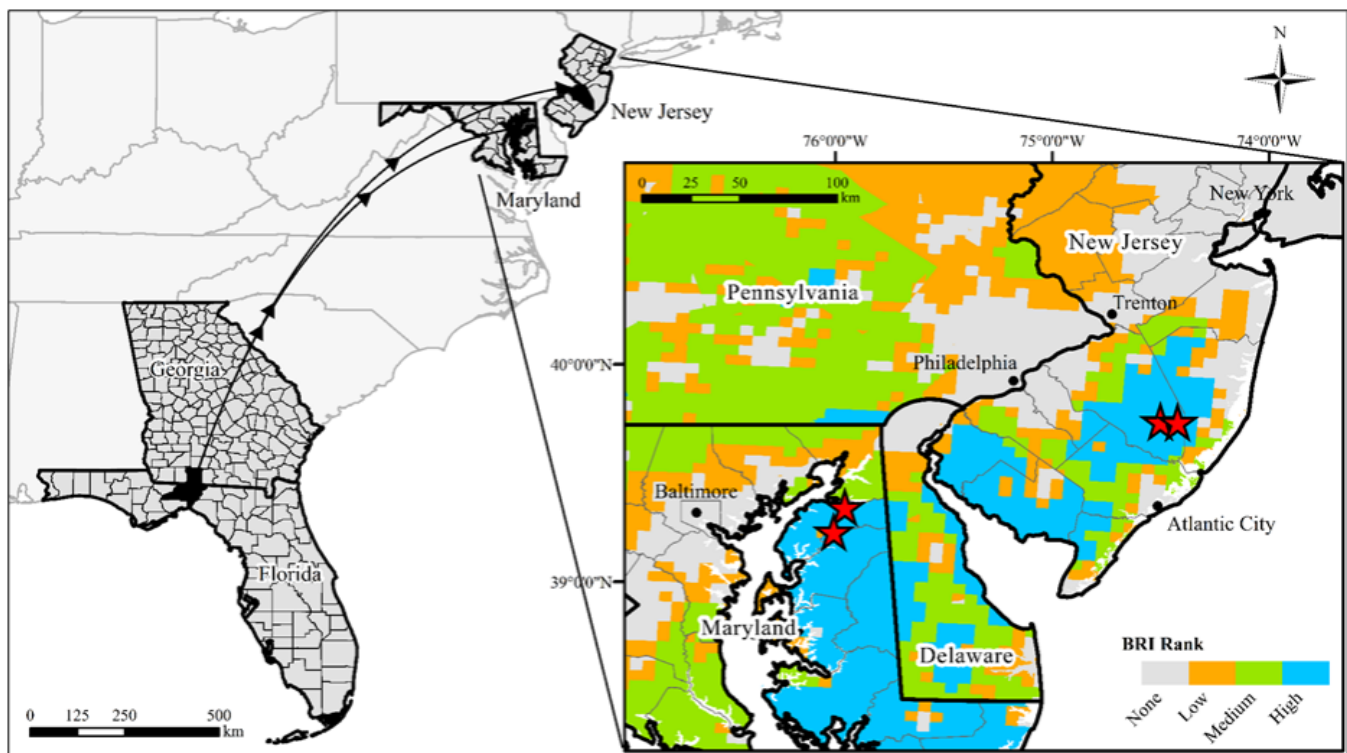Fig. 1. Translocation sites (starred) in Kent and Queen Anne's counties, Maryland, USA and Burlington County, New Jersey, USA, overlaid on Bobwhite Ranking Information 2.0 classifications (BRI Rank; Palmer et al. 2011), 2015–2017. The source sites for translocated bobwhites were in Leon and Jefferson counties, Florida, USA and Thomas County, Georgia, USA; translocation and source site counties are black-shaded.

1), a 445,000-ha forest characterized by pine (*Pinus* spp.)-oak (*Quercus* spp.) and pine and shrub uplands transected by Atlantic white cedar (*Chamaecyparis thyoides*) swamps and emergent wetlands (Forman 1998). The landscape within a 15-km dissolved buffer around Home Farm and Sim Place was 68% forested, 24% wetland, 5% urban or suburban, 1% agriculture, 1% barren land, and 1% open water (New Jersey Department of Environmental Protection 2015). The mean temperature in southern New Jersey ranged from 0.17° C in January to 24.28° C in July, with an average annual precipitation of 114.43 cm (Office of the New Jersey State Climatologist 2019). The topography was principally low relief, gently rolling hills composed predominantly of acidic sandy soils (Rhodehamel 1998). The dominant canopy species included pitch pine (*P. rigida*) and shortleaf pine (*P. echinata*), with scattered black oak (*Q. velutina*), white oak (*Q. alba*), chestnut oak (*Q. montana*), post oak (*Q. stellata*), and scarlet oak (*Q. coccinea*). The understory was generally composed of scrub oaks, including dwarf oak (*Q. prinoides*), scrub (bear) oak (*Q. ilicifolia*), and chinquapin oak (*Q. muehlenbergii*); ericaceous shrubs, including northern highbush blueberry (*Vaccinium corymbosum*), lowbush blueberry (*V. angustifolium*), and black huckleberry (*Gaylussacia baccata*); and Pine Barren golden heather (*Hudsonia ericoides*). In areas where fire or mechanical treatment has been implemented, native herbaceous groundcover, including little bluestem (*Schizachyrium scoparium*), switchgrass (*Panicum virgatum*), broomsedge *(Andropogon virginicus)*, and bracken fern (*Pteridium aquilinum*) had become established along with native forbs, including partridge pea (*Chamaecrista fasciculata*), tick-trefoils (*Desmodium* spp.), and bush-clover species (*Lespedeza* spp.). Some open areas contained bare substrate (coastal plain sands) and patches of juniper moss (*Polytrichum juniperinum*) and lichens (predominantly *Cladonia* spp.).

Forestry management had historically been limited on Pine Island, and when implemented it was principally intended for watershed management related to cranberry production. In 2005, managers in New Jersey approved a forestry stewardship plan for Pine Island that included prescriptions for timber thinning, burning, and roller-drum chopping (hereafter, roller chopping) in the upland portions of the property to support forest and watershed health. Between 2006–2012, dispersed retention cuts with planted pine regeneration occurred on 4 adjacent tracts (~50 ha each) within the Home Farm study area. Consequently, during the study period these tracts were largely dense young pine thickets, with scattered small patches of open shrubland. Timber thinning occurred across 155 ha of Home Farm to a basal area of 14–16 m²/ha. The surrounding area was mature even-aged forests of uncut (≥18 m²/ha) mixed species pine (*Pinus rigida* and *P. echinata*). Although uncut, these other forest stands were being managed with periodic prescribed burns as part of the property's Forest Stewardship Plan activities. Home Farm was fringed by agricultural fields (blueberry production), lakes, and impounded cranberry bogs. The Sim Place release site was centered on a 15-ha grassland dominated by warm-season grass species and surrounded by uncut, yet regularly burned, mature pine stands. These stands had an understory dominated by heath species—lowbush blueberry (*V. angustifolium*) and huckleberry (*G. baccata*) — and bear oak to the north and expansive cranberry bogs in the remaining 3 directions. A network of narrow canals (<10 m wide) transected the Sim Place study area, with the most centrally located canal running parallel to an abandoned bare ground airstrip. The perimeters of the canal system and airstrip areas were composed of native warm-season grasses (*Panicum virgatum, Schizachyrium scoparium, Andropogon virginicus*), greenbrier (*Smilax* spp.), winged-sumac (*Rhus copallinum*), running blackberry (*Rubus hispidus*), and native forbs, including partridge pea, tick-trefoils and bush-clovers, Additionally, both Home Farm and Sim Place had various sized (~1–8 ha) exposed substrate sandpits used for cranberry bog maintenance. These sandpits also contained a mosaic ("patches") of herbaceous growth as well as brush piles of discarded woody material from forestry activities or piles of discarded cranberry vines from bog restorations.

The assemblage of bobwhite predators at Pine Island was similar among sites. Common raptors included Cooper's hawk (*Accipiter cooperii*), sharp-shinned hawk (*A. striatus*), red-tailed hawk (*Buteo jamaicensis*), red-shouldered hawk (*Buteo lineatus*), northern harrier (*Circus cyaneus*), barred owl (*Strix varia*), and great horned owl (*Bubo virginianus*). Common mammalian predators or nest predators included coyote (*Canis latrans*), Virginia opossum (*Didelphis marsupialis*), striped skunk (*Mephitis mephitis*), raccoon (*Procyon lotor*), long-tailed weasel (*Mustela frenata*), gray fox (*Urocyon cinereoargenteus*), and red fox (*Vulpes fulva*). Snake species included northern black racer (*Coluber constrictor*), corn snake (*Pantherophis guttata*), black rat snake (*P. obsoletus*), timber rattlesnake (*Crotalus horridus*), and northern pine snake (*Pituophis melanoleucus*).

Chino Farm is 2,200 ha of mixed grassland and rotational cropland near Chestertown, Maryland (~39.230°N, −76.010°W) and Turner's Creek is a 304-ha rotational crop farm with native grassland buffers near Kennedyville, Maryland (~ 39.345° N, −75.955° W; Figure 1). The landscape within a 15-km dissolved buffer around Chino Farm and Turner's Creek was 58% agriculture, 19% forested, 14% open water, 8% urban or suburban, and 1% wetland (Maryland Department of Planning 2010). Crop fields over this landscape were planted primarily with corn, soybeans, and wheat. Forest canopy species were predominantly oak, hickory (*Carya* spp.), tulip poplar (*Liriodendron tulipifera*), Virginia pine (*Pinus virginiana*), and loblolly pine (*P. taeda*). Common understory and edge species were multiflora rose (*Rosa multiflora*), greenbrier, and Russian olive (*Elaeagnus angustifolia*). Common grasses included broomsedge, big bluestem (*Andropogon gerardi*), and orchard grass (*Dactylis glomerata*). Common forbs consisted of goldenrods (*Solidago* spp.), ragweed (*Ambrosia* spp.), and black-eyed Susan (*Rudbeckia hirta*). Raptor and mammalian species at these 2 sites were generally the same as those found at Pine Island; however, northern pine snakes and timber rattlesnakes were notably absent.

SOURCE SITES

Given the decline of bobwhite throughout its distribution and extirpation from many sites in the Mid-Atlantic region, source populations for translocation are not available. Therefore, long-distance translocation is required as the only viable option. Roberts (2018) indicated that long-distance translocation of bobwhite was feasible as translocated bobwhite had similar survival rates to resident bobwhite. We used private properties in Leon and Jefferson counties, Florida, USA and Thomas County, Georgia, as source sites for wild bobwhite. Two of the properties, Tall Timbers and Dixie Plantation, were owned and operated by Tall Timbers Research Station and Land Conservancy. These forested properties lay within the Red Hills geomorphic region, near the southwesternmost extent of the Atlantic Coastal Plain physiographic province. Landowners of each property had implemented intensive wild quail management for several decades, resulting in relatively high-density bobwhite populations (e.g., >2.5 birds/ha; Sisson et al. 2012, 2017). Management programs typically emphasized frequent fire application (<3-year fire return interval) and low timber density (2–15 m²/ha) to promote and sustain early-seral stage groundcover vegetation communities (Palmer and Sisson 2017).

These properties were principally old field pine forests, characteristic of the Red Hills landscape. Common canopy species included longleaf pine (*Pinus palustris*), shortleaf pine, slash pine (*P. elliottii*), and loblolly pine, with scattered southern live oak (*Quercus virginiana*) and turkey oak (*Q. laevis*). Mid-story species included black cherry (*Prunus serotina*), scrub oak (*Quercus* spp.), sassafras (*Sassafras albidum*), common persimmon (*Diospyros virginiana*), and gallberry (*Ilex glabra*). Understory vegetation was predominantly broomsedge and other warm season grasses (*Andropogon* spp.), bracken fern, winged-sumac, blackberry (*Rubus* spp.), goldenrod, and partridge pea.

As a result of historical intensive quail management, these properties contained adequate habitat to sustain a variety of wildlife populations, including those of bobwhite predators. Common raptors included Cooper's hawk, sharp-shinned hawk, red-tailed hawk, red-shouldered hawk, northern harrier, barred owl, and great horned owl. Common mammalian predators included armadillo (*Dasypus novemcinctus*), bobcat (*Lynx rufus*), coyote, raccoon, gray fox, Virginia opossum, and red fox. Snake species included black racer, corn snake, gray rat snake (*Pantherophis spiloides*), eastern rat snake (*Antherophis alleghaniensis*), eastern diamondback (*Crotalus adamanteus*), coachwhip (*Masticophis flagellum*), and pine snake.

METHODS

As part of a larger translocation project, the New Jersey Audubon Society, University of Delaware, and Tall Timbers Research Station conducted 3 years of bobwhite translocation in late March and early April, prior to bobwhite breeding season, 2015–2017. We translocated bobwhites ($n \geq 120$) to Home Farm, Sim Place, Turner's Creek, and Chino Farm for 3 consecutive years (2015–2017), but we removed Chino Farm as a translocation site after year 1 because of logistical constraints. We focused capture timing to the period immediately preceding covey break-up in the South (Terhune et al. 2006b), as determined from field observations of bobwhite behavior on source sites, so that larger groups of bobwhites could be captured. By increasing capture efficiency, we thereby attempted to reduce the number of translocation trips within a given year. Additionally, this prebreeding period avoids interruption of reproductive behavior (e.g., nesting) and capitalizes on the high reproductive potential of bobwhite (Terhune et al. 2010, Sisson et al. 2012).

We captured bobwhites on source sites using baited (e.g., grain sorghum and cracked corn) standard walk-in funnel traps (Stoddard 1931). We placed traps in areas of dense cover and sheltered with brush to limit stress on captured birds and to conceal traps from predators. We attached unique aluminum leg bands (National Band & Tag Company, Newport, KY, USA) to all captured bobwhites and classified them by age (adult, juvenile) and sex (male, female, based on plumage) (Rosene 1969).

For each site and year, we translocated radio-collared bobwhites at an approximate 1:1 sex ratio. All captured birds for translocation were fitted with a radio-collar and leg band after capture at the source site. Radio-collars were 6.0–7.0 g very high frequency (VHF) pendant-style transmitters (Holohil Systems, Carp, Ontario, Canada) and were affixed to bobwhites weighing ≥132 g (transmitter ≤5% of body mass). This weight limit and transmitter style are common in contemporary bobwhite translocation research (Terhune et al. 2010, Scott et al. 2013, Downey et al. 2017) and do not influence physiology (in captive birds; Hernández et al. 2004) or survival (in wild birds; Palmer and Wellendorf 2007, Terhune et al. 2007) although capture and handling can negatively impact translocation (Abbott et al. 2005). Birds were health screened before being placed in transport boxes. Afterwards, we placed captured bobwhites into transport boxes in groups that did not necessarily reflect covey membership at the time of capture. In some instances, we combined partial coveys from different traps to form complete coveys (8–12 bobwhites) as capture success dictated. We provided a small amount of feed (cracked corn and milo) in transport boxes for consumption during transit. No water was provided in transit. We spread supplemental feed at release sites immediately before and after release to mitigate any stress placed on birds during transport and limit any immediate dispersal from release sites. Supplemental feed continued at the sites through the seasons, except during the summer months.

Translocations occurred over a 36-hour (2 night) period via motor vehicle. The straight-line translocation distance was 1,340 km to Pine Island (~1,600 km driven path) and 1,220 km to Chino Farm and Turner's Creek (~1,450 km driven path). Over the 3 years of the study only 2 birds died as a function of the radio antennae getting stuck in the seam of

the door, and no birds died of stress. We released bobwhites at centralized core locations on translocation properties using a hard release methodology (Martin et al. 2017). In 2016, low trapping success forced multiple translocations over 8 days (1–8 Apr) and we held 1 covey an additional night so we could translocate it with additional bobwhites captured the following day. In all other years, translocations occurred in ≤3 days.

To maintain sample size of radio-collared birds at the translocation sites during the winter, we trapped bobwhites with walk-in funnel traps baited with cracked corn and milo (Stoddard 1931). Because the Pine Island and Turner's Creek sites had no existing conspecifics, any bobwhite encountered after translocation was considered offspring from translocated birds. Chino Farms did have conspecifics so birds without tagging could have been from wild conspecific stock or offspring of translocated birds. All captured bobwhites were weighed, and classified by age and sex. If they were untagged, we fitted them with a unique aluminum leg band, and a pendant-style radio-collar following the protocol described earlier. If juveniles were caught that did not meet the weight requirement to be radio-collared, they were fitted with a unique aluminum leg band only. During the study we caught only 2 new birds on the New Jersey site and no birds in Maryland; thus, we assume that their impact was minimal in the analysis of the project. All trapping, handling, and marking procedures followed American Ornithologists' Union Report of Committee on the Use of Wild Birds in Research (American Ornithologists' Union 1999) and our protocol was approved by the University of Delaware Institutional Animal Care and Use Committee (AUP 1278).

We located individuals using a telemetry receiver (Model R4000, Advanced Telemetry Systems, Isanti, MN, USA) and a three-element Yagi antenna (Advanced Telemetry Systems) via the homing method (White and Garrott 1990) 3–5 times/week to estimate winter survival and habitat use October–March 2015–2018. We approached individuals within 25–50 m to minimize error in habitat classification. We marked individual locations using Avenza PDF Maps (Avenza Systems, Toronto, Ontario, Canada) for iOS mobile devices during radio-tracking while in the field. We determined the cause of mortality (avian, mammal, or unknown) by the condition of the transmitter and evidence at kill site (Dumke and Pils 1973, Curtis et al. 1988).

Individuals were allowed to adjust to radio-collars for 7 days before being included in survival analysis to reduce radio-collar bias (Tsai et al. 1999). We censored bobwhite because of unknown fate, collar loss, or survival beyond the end of the study season. If exact dates of collar loss or disappearance were unknown, we used the midpoint between the last unique location and the day the collar stopped moving or was not locatable as the censor date.

Survival Analysis

We estimated survival rates of bobwhites in relation to temporal and biological (group) effects using the known-fate data type in Program MARK (White and Burnham 1999). The known-fate model employs a binomial likelihood and permits incorporation of individual covariates (e.g., sex), temporal effects (e.g., year, linear and quadratic time trends, season, week), and groups (e.g., region, site) to evaluate their effect on survival. When the covariates were biologically relevant, we constructed *a priori* candidate models incorporating additive effects and interactions using a logit-link function. We computed weekly survival rates (WSR) and we specified the appropriate interval length in Program MARK to yield accurate estimates of survival and precision.

We used an information-theoretic approach (Akaike 1973, Guiasu 1977, Anderson et al. 2000, Burnham and Anderson 2002) to sequentially evaluate 2 sets of candidate models testing explicit hypotheses. The first set of models evaluated temporal (i.e., year, season, month, week, linear and quadratic time trends) effects, and we used the top-ranked, most parsimonious model as the baseline model for the second sequential model set evaluating region, site, and sex effects on survival. We used Akaike's Information Criterion adjusted for small sample bias (AIC$_c$; Akaike 1973, Wedderburn 1974, Burnham and Anderson 2002) to compare candidate models, and we considered the model with the lowest AIC$_c$ to be the best approximating model, given the data. We assessed model fit (using evaluation of residual plots and ĉ) derived from the most general model. We assessed the relative plausibility of each model in the set of candidate models using Akaike weights (W_i; Anderson et al. 2000, Burnham and Anderson 2002), where the best approximating model in the candidate set has the greatest Akaike weight. We used model averaging (Akaike 1973, 1974; Burnham and Anderson 2002) to obtain weekly survival rates (WSR). To provide additional inferential power and to allow direct comparison of covariates, we evaluated and report beta coefficients, their standard errors (SEs), and 85% confidence intervals (CIs) for variables of interest (e.g., group; Arnold 2010).

Home Range Analysis

We used adehabitat in R (Calenge 2006) with a least-squares cross-validation as a smoothing parameter to calculate a 50% and 95% adaptive kernel home range (Worton 1989). We defined coveys as unique groups of bobwhites that remained together for >3 consecutive tracking days (Lohr et al. 2011). Coveys with <20 locations were excluded from analysis. This 20-telemetry location requirement to reduce estimation bias is lower than many other studies (Terhune et al. 2010); however, due to a small sample size, we felt it was necessary to get informative results from the model (Haines et al. 2009).

Resource Use Analysis

Second- and third-order habitat analysis was conducted for both the New Jersey and Maryland regions. Land cover classification was digitized for each site in ArcGIS version 10 (Esri Inc., Redlands, CA, USA) over spatial resolution (0.305 m) orthophotographs for the years of study (2015–2018). Due to the fundamental differences between the pine systems in the New Jersey sites and the agricultural landscape in Maryland sites, land cover categories were identified separately for the 2 regions.

The following 7 land cover categories were defined for the New Jersey sites: 1) pine woods (PNW), defined as mature closed-canopy pine-dominated upland within which no mechanical timber operations occurred with a basal area of >18 m²/ha; 2) thinned pine (PNT), defined as partially open-canopy pine-dominated upland within which mechanical timber thinning and surface scarification occurred with a basal area of ~14–16 m²/ha; 3) cut pine (PNC), defined as open-canopy pine-dominated upland within which clear cutting or dispersed retention cutting and roller chopping occurred 10–12 years preceding study period, as evidenced by mid-seral stage vegetation communities; 4) early-successional woody (ESW), defined as areas dominated by woody (e.g., shrub, scrub oak, pine saplings) vegetation and lacking any canopy; 5) early-successional herbaceous (ESH), defined as areas characteristically dominated by early stage herbaceous vegetation, including warm-season grasses and forbs; 6) wooded wetland (WTW), defined as closed-canopy wetlands dominated by hardwood trees; and 7) cedar woods (CDW), defined as wetlands dominated by Atlantic white cedar.

The following 6 land cover categories were defined for the Maryland sites: 1) cropland (CPS), defined as areas within which row crops were being cultivated; 2) cool-season grass drains (CSG), defined as areas covered by sod-forming cool-season grasses for erosion control; 3) early-successional herbaceous (see preceding paragraph); 4) early-successional woody; 5) food and cover plantings (FCP), defined as wildlife plantings meant to provide food and cover (e.g., bicolor lespedeza [*Lespedeza bicolor*]); and 6) mixed woods (MXW), defined as mature mixed-species woodlots, drains, and riparian areas bordering agricultural fields or other open land cover. For all sites, open water (including cranberry bogs) and manmade structures were classified as non-usable space.

On the New Jersey site only, we further evaluated telemetry locations for more in-depth microhabitat use assessment of pine management. Technicians in Maryland were not able to collect this information, so microhabitat analysis between pineland and agricultural management was not possible. We randomly selected 30 points/release site/year (n = 180) and collected 1) percentage of bare ground, litter, grasses, forbs, and woody debris in a 1-m² plot (Daubenmire 1959); 2) visual obstruction of location at 0.25 m and 1 m in height using a Nudds board (Nudds 1977); 3) basal density via a Jim-Gem® factor 10 prism (Doggett and Locher 2018); and 4) percentage of canopy closure via a convex spherical densitometer (Forestry Suppliers, Inc., Jackson, MS, USA). We further collected two randomized points (through randomized azimuths and paces taken from the telemetry location) to compare habitat preferences with availability. If the randomized point occurred within non-habitat (e.g., open-water, roadways), a new random point was selected from the point originally used.

We considered available resources at the second-order home range of the individual to be all the usable space within a 95% minimum convex polygon (MCP) of telemetry locations at each study site. Available resources at the third-order used within the home range were considered all usable space within the 95% adaptive kernal home range for each individual. To assess available use relative to telemetry locations of individuals, 5 random points were generated for each individual location within the second- and third-order available habitat. This provided a 5:1 available-to-use prevalence ratio for both second- and third-order analyses (Phillips et al. 2009). Fourth-order site-specific resource selection analysis compared field vegetation data at individual use points to those collected at randomized non-use points. This provided a 1:1 prevalence ratio for fourth-order analyses.

Mixed logistic regression models were used to estimate probability of use at the second, third, and fourth order. Land cover covariates were used at the second- and third-order analysis and vegetation covariates were used for the fourth-order analysis. All continuous covariates were centered on a mean of 0 and standard deviation of 1 to improve model convergence. We used Pearson's correlation tests to assess the degree of collinearity of model parameters and did not fit models when r > 0.7 (Coppola 2021). For second- and third-order analyses, we specified vague, normal priors with a mean of 0 and precision of 0.001 for all fixed effects (Royle and Dorazio 2008, Kéry and Shaub 2012). Three habitat variables—litter cover, canopy coverage, and visual obstruction at 2 m—all showed high correlation; thus, we removed litter and canopy cover and retained visual obstruction as the predicting variable in the fourth-order analyses.

R2Jags program in R (Su and Yajima 2015) was used to estimate posterior distributions using Markov chain Monte Carlo methods with 3 independent chains to assess habitat selection. Each chain ran 25,000 iterations, discarding the first 10,000 and saving every fifth iteration thereafter. We assessed convergence via visual inspection of trace plots and defined adequate convergence as Gelman–Rubin convergence statistics < 1.1 (Gelman et al. 2014). Regression coefficients whose 85% credible intervals (CrI) overlapped 0 were interpreted as indicating equal selection with availability.

RESULTS

In New Jersey, we followed 53 radio-tagged bobwhites between 2015–2017: 20 (10 male, 10 female) individuals in 5 coveys in 2015–2016, 15 (8 male, 7 female) individuals in 6 coveys in 2016–2017, and 18 (11 male, 7 female) individuals in 5 coveys in 2017–2018. Of those birds, only 2 were new captures (fall 2015) and all the remaining birds over the 3 years were from the original translocation population. In Maryland, we followed 18 radio-tagged bobwhites between 2015–2017: 4 (0 male, 4 female) individuals in 3 coveys in 2015, 4 (2 male, 2 female) in 2 coveys in 2016, and 10 (7 male, 3 female) individuals in 4 coveys in 2017. Because only 2 of the 71 birds were locally caught, we did not have the sample size to conduct any analyses that would compare survival and habitat use between translocated and local birds.

Survival

The most parsimonious model from the first sequential model set evaluating temporal effects included a quadratic time trend (W_i = 0.85; Table 1). This top-ranked model was >10 times more likely to explain temporal variation in survival than the second-ranked model, which included month (ΔAIC_c = 4.70, W_i = 0.08; Table 1). Although receiving no support, models including season (ΔAIC_c = 23.74, W_i = 0.00) indicated the season was biologically meaningful such that non-breeding survival was lower (β_{winter} = -0.512, CI = -0.871 to -0.065) during the winter period than the fall (Figure 2). Models incorporating year effects did not merit support based on Akaike weights and ΔAIC_c values > 10 (Table 1).

The most parsimonious model from the second sequential model set effects included region (AIC_c = 485.58, W_i = 0.62), which was >2.5 times more likely to explain variation in survival than individual site effects (ΔAIC_c = 1.93; W_i = 0.00; Table 2). Region and site effects were lower in cropland sites in Maryland (β_{region} = -0.815, CI = -1.263 to -0.367]) compared to forested sites in New Jersey (Figure 3a, 3b). Survival was lower during the winter in both Maryland and New Jersey, and individuals in cropland sites in Maryland were 125 times less likely to survive the winter than individuals in forested sites in New Jersey (Figure 4). Although models including sex received moderate support (ΔAIC_c = 4.48, W_i = 0.07; Table 2), its biological relevance was low (β_{female} = -0.353, CI = -0.721 to 0.261).

Table 1. Akaike's Information Criterion, adjusted for small sample (AIC_c), ranking temporal effects models for northern bobwhite (*Colinus virginianus*) survival (S) on 4 study areas in New Jersey, USA and Maryland, USA, October–March 2015–2018.

Model	AIC_c	Δ AIC_c	W_i	Model likelihood	k	Deviance
S (T + TT)[a]	489.94	0.00	0.85	1.00	3	483.92
S (Month)[b]	494.64	4.70	0.08	0.10	6	482.56
S (T)	494.89	4.94	0.07	0.08	2	490.88
S (Season)[c]	513.69	23.74	0.00	0.00	2	509.68
S (.)	515.87	25.93	0.00	0.00	1	513.87
S (Year)	518.28	28.33	0.00	0.00	3	512.25

[a] T and TT represent linear and quadratic time trends, respectively.
[b] Month evaluating individual monthly effects on survival.
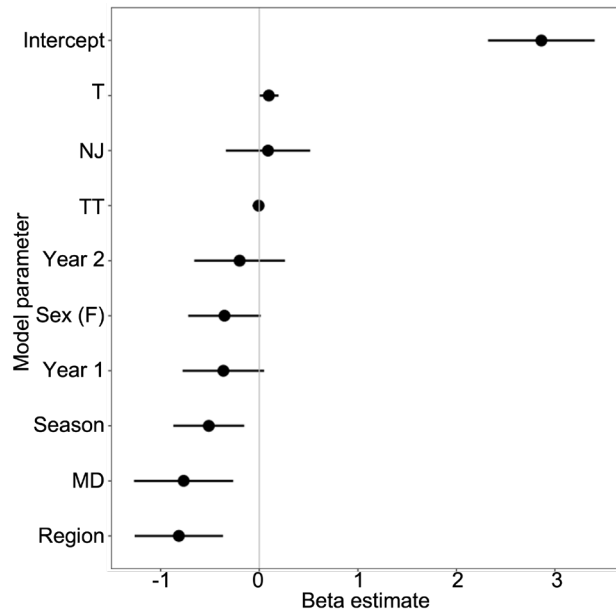[c] Season comparing fall and winter effects on survival.

Fig. 2. Model average parameter estimates and 85% confidence limits for northern bobwhite survival during the fall and winter following spring translocation during 2015–2017 in the Eastern Shore of Maryland (MD), USA and Pine Island, New Jersey (NJ), USA. Models were developed *a priori* to evaluate temporal (T = linear time trend; TT = quadratic time trend; Season = Winter season compared to fall; year 1 = 2016 and year 2 = 2017 compared to 2015 as the baseline) and group (Region = cropland sites in MD compared to pineland sites in New Jersey; Sites = Sim Place and Home Farm in New Jersey and 2 sites in Maryland) effects on bobwhite survival.
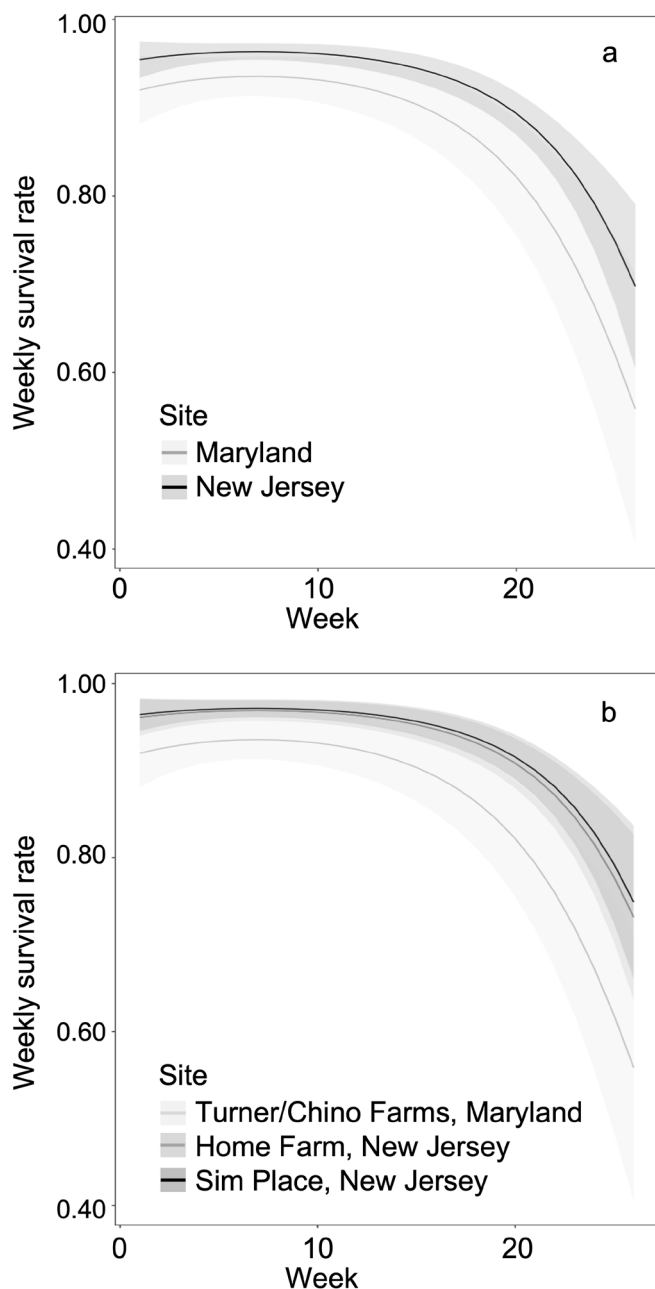
Table 2. Akaike's Information Criterion, adjusted for small sample (AIC_c), ranking treatment group effects models for northern bobwhite (*Colinus virginianus*) survival (S) on 4 study areas in New Jersey, USA and Maryland, USA, October–March 2015–2018.

Model	AIC_c	Δ AIC_c	W_i	Model likelihood	k	Deviance
S (T + TT + Region)[a]	485.58	0.00	0.62	1.00	4	477.54
S (T + TT + Site)[b]	487.51	1.93	0.24	0.38	5	477.45
S (T + TT)	489.94	4.36	0.07	0.11	3	483.92
S (T + TT + Sex)	490.06	4.48	0.07	0.11	4	482.03
S (T)	494.89	9.31	0.01	0.01	2	490.88
S (Season + Region)[b, c]	513.15	27.57	0.00	0.00	3	507.13
S (Season)[c]	513.69	28.11	0.00	0.00	2	509.68

[a] Region comparing cropland sites in Maryland to forested sites in New Jersey.
[b] Site comparing individual study sites in Maryland and New Jersey.
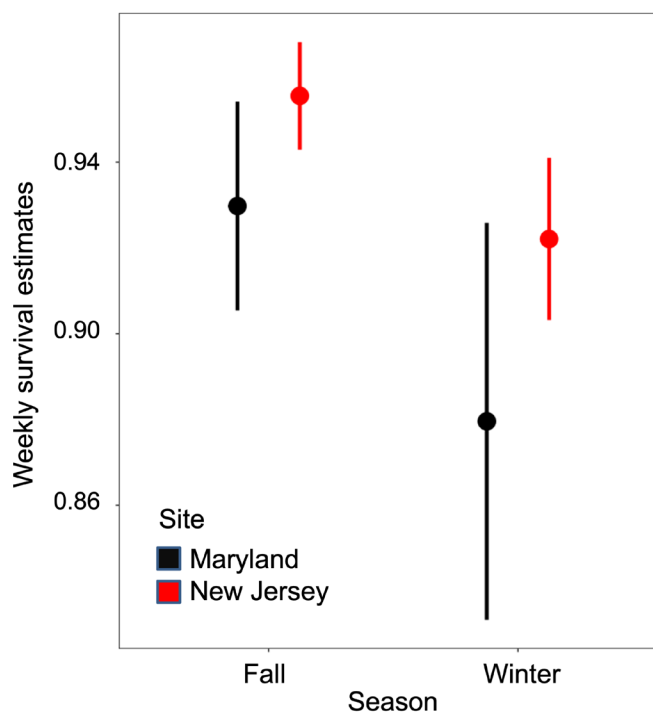[c] Season comparing fall weeks to winter weeks.

Fig. 4. Northern bobwhite weekly survival with associated 85% confidence limits during the fall and winter season, October–March 2015–2017, in Maryland and New Jersey, USA study sites.

were 81.01 ha (SE ±15.85) at Home Farm, 41.39 (SE ±5.09) at Sims Place, 22.74 (SE ±2.00) at Turner's Creek, and 10.31 (SE ±2.15) at Chino Farms. The 50% and 95% kernel densities, respectively, were 11.41 (SE ±1.55) and 54.32 (SE ±12.62) at Home Farm, 6.97 (SE ±0.53) and 31.52 (SE ±9.52) at Sim Place, 4.52 (SE ±0.59) and 20.44 (SE ±2.41) at Turner Creek, and 4.52 (SE ±0.59) and 20.44 (SE ±2.41) at Chino Farm.

RESOURCE USE

At the second-order scale for the Maryland sites, translocated bobwhites showed positive selection and probability of use for food and cover planting (β_{FCP} = 1.551, CrI = 0.973–2.141), early-successional woody (β_{ESW} = 0.711, CrI = 0.159–1.236), and mixed woods (β_{MXW} = 0.642, CrI = 0.123–1.136) land cover types, and negative selection for cool-season grass (β_{CSG} = -0.642, CrI = -1.356 to -0.012), cropland (β_{CPS} = -0.713, CrI = -1.170 to -0.251), and early-successional herbaceous (β_{ESH} = -1.535, CrI = -1.865 to -1.222) cover types (Figure 5a, 5b). At the third-order scale for the Maryland sites, translocated bobwhites showed positive selection and probability of use for food and cover planting (β_{FCP} = 1.187, CrI = 0.629–1.737) land cover types; equal selection for early-successional woody (β_{ESW} = 0.379, CrI = -0.157 to 0.913) and mixed woods (β_{MXW} = 0.066, CrI = -0.429–0.536) land cover types; and negative selection for cool-season grass (β_{CSG} = -0.818, CrI = -1.516 to -0.182), cropland (β_{CPS} = -0.573, CrI

Fig. 3. Northern bobwhite weekly survival with associated 85% confidence limits for the over-winter season, October–March 2015–2017, derived for sites pooled by a) region (Maryland, USA and New Jersey, USA) using a linear and quadratic time trend model and b) site (TCF = Turner's Creek Farm in Maryland; Home Place and Sim Place in New Jersey).

HOME RANGE

Combining coveys across years, and removing any coveys with <20 telemetry locations, provided 54 covey locations on Home Farm, 108 covey locations on Sim Place, 53 covey locations on Turner's Creek, and 11 covey locations on Chino for analysis (note: Chino had bobwhites for winter analysis only in study year 1; Table 3). The 95% MCP home ranges

Table 3. Mean home ranges (ha, 95% Minimum Convex Polygon [MCP]; kernel: 50% and 95%) for translocated, radio-marked northern bobwhites (*Colinus virginianus*) in 4 study areas in New Jersey (NJ), USA and Maryland (MD), USA, October–March 2015–2018.

| Site (state)	Year	n	Locations, $\bar{x} \pm$ SD	95% MCP ha (SE)	50% Kernel ha (SE)	95% Kernel ha (SE)
Home Farm (NJ)	2015	24	47 ± 9	26.50 (3.72)	4.65 (0.96)	23.48 (19.96)
	2016	16	59 ± 16	151.32 (45.24)	20.05 (3.65)	92.01 (9.97)
	2017	14	57 ± 18	94.09 (19.24)	13.11 (2.09)	64.11 (16.86)
	Pooled	54	53 ± 15	81.01 (15.85)	11.41 (1.55)	54.32 (12.62)
Sim Place (NJ)	2015	15	44 ± 7	21.05 (3.06)	4.20 (0.47)	18.49 (16.43)
	2016	17	55 ± 14	84.34 (20.28)	12.68 (1.99)	56.66 (5.47)
	2017	22	58 ± 16	57.45 (14.83)	9.30 (1.28)	41.83 (12.93)
	2018	54	47 ± 14	26.97 (14.83)	4.98 (0.37)	23.02 (11.62)
	Pooled	108	50 ± 15	41.39 (5.09)	6.97 (0.53)	31.52 (9.52)
Turner's Creek Farm (MD)	2015	8	43 ± 16	18.40 (4.86)	1.77 (0.45)	9.53 (2.27)
	2016	9	43 ± 16	29.59 (3.91)	6.67 (1.75)	29.32 (7.08)
	2017	28	37 ± 8	22.81 (3.07)	4.98 (0.88)	22.04 (3.51)
	2018	8	53 ± 22	19.14 (3.85)	3.27 (0.74)	15.76 (3.92)
	Pooled	53	41 ± 14	22.74 (2.00)	4.52 (0.59)	20.44 (2.41)
Chino (MD)	2015	11	52 ± 15	10.31 (2.15)	2.20 (0.63)	10.66 (3.09)
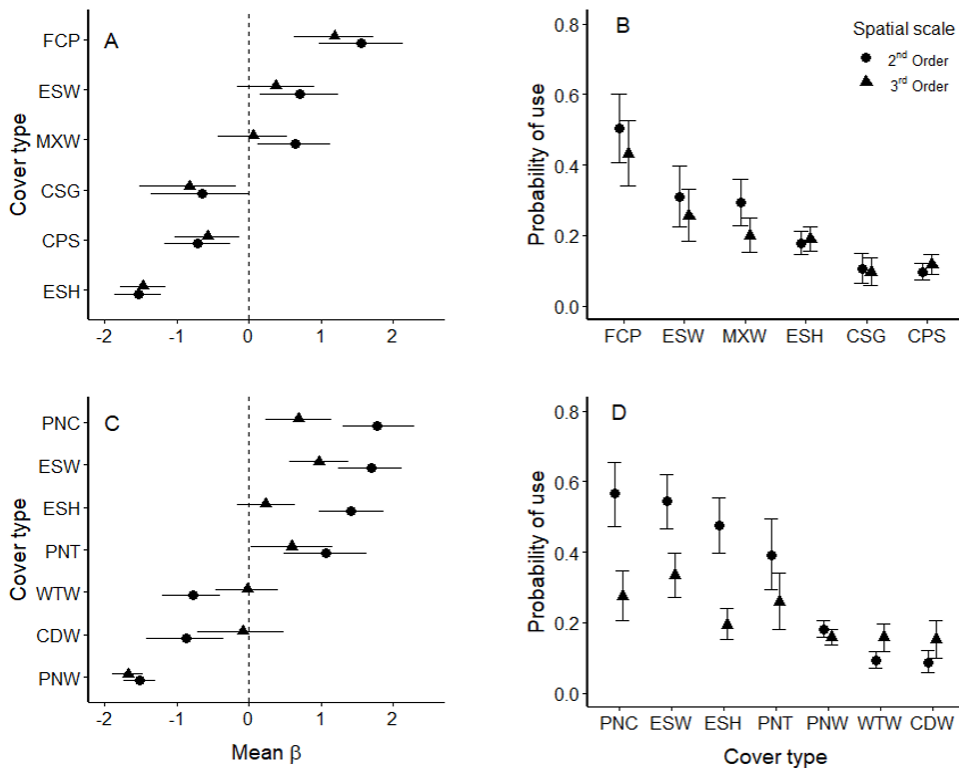
Fig. 5. Second- and third-order analyses for probability of use of various cover types (CDW: cedar woods; CPS: cropland; CSG: cool season grass; ESH: early-successional herbaceous; ESW: early-successional woody; FCP: food and cover plantings; MXW: mixed woods; PNC: cut pine; PNT: thinned pine; PNW: pine woods; WTW: wooded wetland) for translocated northern bobwhites in the Eastern Shore of Maryland, USA (A, B) and Pine Island, New Jersey, USA (C, D), during winter season (Oct–Mar 2015–2017).

= -1.036 to -0.125), and early-successional herbaceous (β_{ESH} = -1.468, CrI = -1.788 to -1.160) cover types (Figure 5a, 5b).

At the second-order scale for the New Jersey sites, translocated bobwhites had access to all vegetative types and showed positive selection and probability of use for cut pine (β_{PNC} = 1.783, CrI = 1.296–2.290), early-successional woody (β_{ESW} = 1.692, CrI = 1.242–2.130), early-successional herbaceous (β_{ESH} = 1.413, CrI = 0.966–1.871), and thinned pine (β_{PNT} = 1.063, CrI = 0.492–1.630) and negative selection for wooded wetland (β_{WTW} = -0.783, CrI = -1.199 to -0.392), cedar woods (β_{CDW} = -0.869, CrI = -1.420 to -0.350), and pine woods (β_{PNW} = -1.514, CrI = -1.731 to -1.297) land cover types (Fig. 5c, 5d). At the third-order scale for the New Jersey sites, translocated bobwhites showed positive selection and probability of use for cut pine (β_{PNC} = 0.689, CrI = 0.239–1.150), early-successional woody (β_{ESW} = 0.977, CrI = 0.565–1.379), and thinned pine (β_{PNT} = 0.560, CrI = 0.036–1.168); equal selection for early-successional herbaceous (β_{ESH} = 0.235, CrI = -0.168–0.641), wooded wetland (β_{WTW} = -0.016, CrI = -0.459 to 0.402), and cedar woods (β_{CDW} = -0.081, CrI = -0.711 to 0.482); and negative selection for pine woods (β_{PNW} = -1.678, CrI = -1.901 to -1.462) land cover types (Figure 5c, 5d).

Of the 6 vegetation variables considered as predictors of fourth-order habitat selection at the New Jersey sites, the most influential variables were grass cover, which showed a strong positive effect (β_{grass} = 0.643, CrI = -0.302 to 1.583), and visual obstruction at 1 m, which showed a strong negative effect (β_{vor1} = -0.235, 85% CrI = -0.611 to -0.138). The next most influential variables were woody groundcover (β_{woody} = 0.478, CrI = -0.048 to 1.01) and visual obstruction at 0.25

m (β_{vor025} = 0.294, CrI = -0.366 to -0.948. Probability of use decreased linearly with horizontal visual obstruction at 1 m (Figure 6). In contrast, probability of use for horizontal visual obstruction at 0.25 m, percent grass cover, and percent woody cover all increased linearly (Figure 6); however, it is important to recognize woody groundcover and visual obstruction at 0.25 m both overlapped with zero, so it is likely their selection effects were nominal.

DISCUSSION

We found that survival and resource use differed among translocated bobwhite on cropland-dominated (fragmented) sites compared to forested sites. Survival was lower on cropland sites compared to forested sites such that bobwhite in cropland-dominated landscapes were >125 times less likely to survive the winter than those on forested sites. We observed differential resource availability among sites and use among bobwhites translocated to forested sites and cropland sites during the non-breeding season that were not as evident during the breeding season (Coppola et al. 2021).

Non-breeding season survival was low on all study sites, but especially on cropland-dominated sites. Lohr (2009) found wild bobwhite in New Jersey had a daily survival rate of 0.9934 and a cumulative non-breeding season survival rate of 0.30. Population models for bobwhite in the Mid-Atlantic predicted that bobwhite populations need a daily survival rate of 0.9968, which equates to an average weekly survival rate of 0.9778 (i.e., non-breeding, overwinter survival rate of 0.556) to maintain a stable population (Williams et al. 2012). Non-breeding season period survival was lower on both sites (New Jersey = 26.2% and Maryland 11.1%) than the predicted winter survival rate required for population stability in the Mid-Atlantic states. Furthermore, average weekly survival rates in our study were lower (WSRNJ = 0.9498 and WSRMD = 0.9189) for translocated bobwhites compared to average weekly survival (0.9728) of their origination (source) sites (Terhune et al. 2007, Sisson et al. 2017). Although our sample size was low, especially on the Maryland sites, these winter survival rates are cause for concern. However, they underscore the importance of resource availability needed in the Mid-Atlantic states to overcome challenges in the states in the northern periphery of bobwhite distribution during the non-breeding season and the opportunity for improvement of habitat conditions on our study sites.

Harsh winters are known to negatively impact wildlife and bobwhite survival (Janke et al. 2015, McLaughlin et al. 2019). We found that weekly survival rates on both cropland and forested lands were similar during the fall to other bobwhite studies (Terhune et al. 2007, 2010; Sisson et al. 2017) but much lower during the winter months (see Figure 4). However, bobwhite translocated to cropland sites experienced extremely low winter weekly survival rates, even compared to forested sites in this study. This is likely a result of a habitat pinch point on cropland landscapes where during the non-breeding season much of the available cover
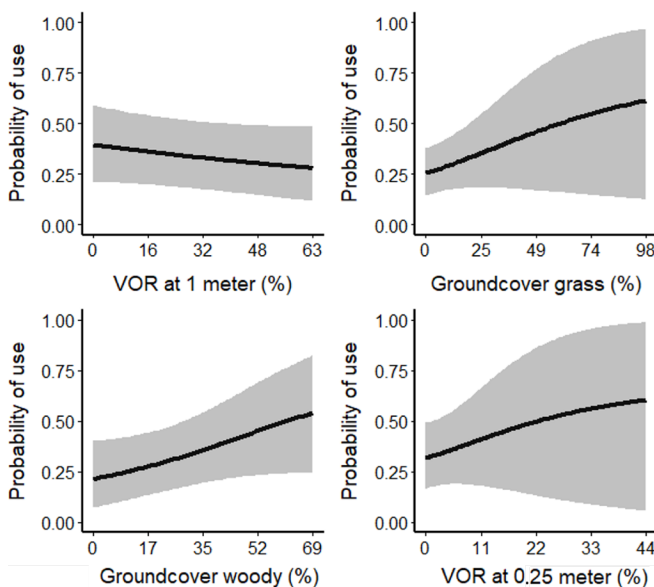

Fig. 6. Fourth-order analysis of probability of use as a function of visual obstruction (VOR) at a height of 1 m, percentage of groundcover in grass, percentage of groundcover in woody vegetation, and visual obstruction at 0.25 m for translocated northern bobwhites in Pine Island, New Jersey during winter season (October–March 2015–2017).

(~70%) present during the breeding season is removed due to late-summer and early-fall annual harvest of crops. As such, birds are forced to use resources in a more concentrated area, potentially increasing predation risk and reducing food availability. While it is well established that broad-scale patterns of abiotic and biotic conditions affect organisms' distributions and population fluctuations, discrete events may be important drivers of space use, survival, and persistence. Discrete extreme climatic events can constrain populations and space use at fine scales beyond what is typically measured in ecological studies. Recently, a growing body of literature has identified thermal stress as a potential mechanism in determining space use and survival (Tanner et al. 2016). Although not directly assessed in this study, snow events could have contributed to reduced non-breeding season survival during the winter months and could elucidate our findings (Janke et al. 2015, 2017). Multiple snow events did occur on our study sites, but the magnitude of their impact is uncertain. That said, nutritional supplementation is known to improve survival during harsh winter weather and potentially help to mitigate but not eliminate habitat deficiencies (McLaughlin et al. 2019). Future translocation research should evaluate the interactive relationships between thermal cover and necessary food resource availability during the non-breeding season to support necessary survival.

Differential resource use by bobwhite on cropland compared to forested sites was evident at multiple spatial scales. On the forested sites, bobwhite used cut pine, early-successional woody, early-successional herbaceous, and thinned pine more than what was available on the landscape, but at the home range scale bobwhite only used cut pine, thinned pine, and early-successional woody higher than their availability. On the cropland sites, bobwhite used food plots, early-successional woody, and mixed woods more than what was available on the landscape and only food plots at the home range scale. Bobwhite on both landscapes used early-successional woody resources at all spatial scales, which has been shown to directly impact non-breeding season survival in other studies in northern U.S. states (Janke et al. 2015, 2017). This use of the woody resources and higher survival at the New Jersey forested site appears to run counter to some research that suggests that, although bobwhites generally select areas with greater understory cover, they avoided uplands when pine or hardwood basal area exceeded 20 m²/ha or 12 m²/ha, respectively; a plausible explanation is the association of high basal area with increased shading and subsequent loss of understory cover (Kroeger et al. 2020). However, in the New Jersey Pinelands ecosystem, the forest contains a high basal area with an intact understory consisting of low woody shrubs that provide not only winter thermal cover but also escape cover and forage opportunities from both soft and hard mast through the seasons. On cropland sites, however, bobwhite used food and cover plots most frequently and much higher than expected at both the landscape and home range scales. Food and cover plots and much of the early-successional woody cover were established several years before translocation. Considering bobwhite's high use of artificial plantings, other limiting resources such as forested lands (e.g., pinewoods,

thinned pine) potentially indicate insufficient availability of resources conducive to adequate non-breeding season survival on these sites. In contrast, bobwhite on the forested landscapes in New Jersey had abundant, non-fragmented native resources available but still experienced large home ranges and relatively low survival. This may indicate that the resources are still limiting or are available, but habitat management modifications are needed to improve survival during the non-breeding season. For example, the high use of cut pine, early-successional cover (woody and herbaceous), and thinned pine may reveal a limitation of those resources within the forested landscape. The presence or absence of woody understory cover has been demonstrated repeatedly to be a stronger influence on bobwhite habitat selection than herbaceous cover during the non-breeding season, and managers should provide woody understory cover on the landscape (Cram et al. 2002, Lusk et al. 2006, Janke et al. 2013, Brooke et al. 2015, Rosche et al. 2019, Kroeger et al. 2020). Because forest management is not common in the New Jersey pine barrens, implementation of an intentional timber thinning and prescribed fire plan could improve all these habitat conditions (i.e., increase preferred fourth-order habitat availability and reduce home range size) on the New Jersey sites.

Resource availability is a precursor to demographic performance and a necessity for translocation success (Martin et al. 2017). Most translocation studies, to date, have focused on survival during the breeding season and did not assess non-breeding season survival and resource use. Our study demonstrates that non-breeding season survival can limit translocation success and subsequent population growth, which could elucidate why some studies show short-term success during the breeding season but do not find long-term population growth and stability (Scott et al. 2013, Downey et al. 2017). Habitat deficiencies and stress incurred by translocated bobwhite will undoubtedly influence reproductive effort and fall recruitment immediately after translocation (Coppola et al. 2021). Fewer individuals are expected to survive until fall, and insufficient non-breeding season habitat conditions will result in even fewer individuals making it through the winter. Sandercock et al. (2008) indicated the importance of adult survival, particularly non-breeding season survival, to bobwhite population stability. Whereas translocations in the southeastern United States have had widespread success, translocation in the northeastern United States poses additional constraints (e.g., limited local source populations) and challenges to population stability and growth.

MANAGEMENT IMPLICATIONS

Our findings provide rare information on demographic resiliency and resource use for translocated bobwhite during the non-breeding season. We recommend that translocation preferentially focus on forested landscapes as a priority over fragmented, cropland landscapes unless sites could potentially meet the minimum of 1,500 acres (607 ha) of year-round

available quail habitat (Palmer et al. 2011). Further, in New Jersey, our research supports the New Jersey Bobwhite Action Plan recommendation that the Pinelands probably represents the greatest potential growth area for wild bobwhite within their former distribution in New Jersey. In addition, intentional management plans employing appropriate management activities such as prescribed fire and timber thinning on an annual basis prior to, during, and after translocation, and at a spatially relevant scale, will improve breeding and non-breeding season habitat conditions and the overall likelihood of success.

ACKNOWLEDGMENTS

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