SOUTH FLORIDA DEER STUDY

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Contract No. 14003 — South Florida Deer Project
Final Report to the Florida Fish and Wildlife Conservation Commission
Warnell School of Forestry and Natural Resources

University of Georgia
Athens, GA 30602

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Final Report

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EXECUTIVE SUMMARY

White-tailed deer (*Odocoileus virginianus*) are one of the most valued game species in Florida. In South Florida, deer are also the most important prey species of the Florida panther (*Puma concolor coryi*). In recent years, aerial survey and harvest data suggested that deer populations have declined in some regions of South Florida where environmental conditions and the predator community have changed substantially since the 1990's. In addition, essential deer demographic data were outdated and while aerial surveys provide valuable population trend data, determining accurate density and abundance estimates necessitated a better monitoring method. These concerns and data needs prompted a study from January 2015 to January 2019 in the Florida Panther National Wildlife Refuge (FPNWR), and the Bear Island (BI) Unit and a northern section of the Addition Unit, commonly referred to as the North Addition Land (AL) of the Big Cypress National Preserve (BCNP). The two main objectives were to (1) assess the effects of hydrology, hunting, and predation on population dynamics of white-tailed deer and (2) develop a camera trapping study for a large-scale investigation and monitoring of deer populations in South Florida.

We captured 294 deer and fitted 263 (172 females and 91 males) with GPS collars. We used telemetry data to evaluate cause-specific mortality, annual and seasonal survival, and factors influencing mortality risk for adult deer. In addition, we used telemetry data to quantify home range area, resource selection, movement, and activity patterns across hydrological and biological seasons. We deployed 180 trail cameras without bait or attractants across the study area in three grids of 60 (40 on-trail and 20 off-trail) cameras and developed camera-based methods to investigate spatiotemporal trends in detection rates for deer and their predators. We combined camera and telemetry data to study spatial and temporal variation in the density of

adult deer, and we uniquely identified fawns using their spot patterns to estimate fawn density and recruitment. We leveraged this information to evaluate multiple trail camera design scenarios and to provide recommendations for future investigation and monitoring of deer populations.

The first objective of our study was to examine the effects of hydrology, hunting, and predation on population dynamics of white-tailed deer. We found evidence that hydrology can influence deer population dynamics through at least two mechanisms. First, we documented a negative effect of water level on adult deer survival, an effect that was stronger for females than males. Although no deer died directly from drowning, water levels influenced deer movement by increasing the use of edges and roads during high water, which may have increased susceptibility to predation. Second, fawn recruitment in 2016, a year with severe flooding during the fawning season, was approximately one third of the recruitment estimate of the 2015 fawning season when conditions were much drier. Hunter harvest had a negligible effect on the deer population as only 1 of 263 deer with an active GPS collar was legally harvested during the observation period, although 2 deer appeared to have been killed illegally. However, 22% of collared deer were on FPNWR where hunting is not allowed, and of the remaining collared deer in BCNP, approximately 33% were males and therefore potentially eligible for harvest depending on antler characteristics. We investigated 134 deer mortalities, and the leading cause of mortality across all years was predation (82%), followed by unknown cause of death (10%) and disease (3%), research induced (2%), hunting (1%) and poaching (1%). Florida panther predation accounted for 72% of mortalities (*n* = 96), followed by bobcat (5%), unidentified predator (3%), black bear (1%), and alligator (1%).

We observed sex-specific seasonal variation in deer survival probability. Female survival probability was greatest during the rut (July-September), and lowest during the fawning season (January-March). Male survival probability peaked during fawning and rearing (April-June) seasons, was intermediate during the post-rut (October-December), and was lowest during the rut. Male survival probability was generally lower than female survival, except during the fawning season. Deer survival probability was negatively correlated with water depth and female survival was more strongly negatively affected by water depth. We observed an increasing temporal trend in survival that may have been the result of wetter conditions during the first half of the study or a change in predation pressure. Because there is no physical boundary between BI and AL and many collared deer in BCNP used both units, we combined deer in BI and AL to compare annual survival on BCNP and FPNWR. Average annual male survival probabilities in BCNP and FPNWR were 0.59 (95% CI: 0.50-0.68) and 0.69 (0.57-0.80), respectively; average annual female survival probabilities in BCNP and FPNWR were 0.70 (0.64-0.75) and 0.78 (0.68-0.85), respectively. Annual male survival probabilities in BCNP were 0.44 (0.31-0.56), 0.53 (0.42-0.63), 0.65 (0.54-0.75), 0.76 (0.64-0.85), in 2015, 2016, 2017, and 2018, respectively. In FPNWR, during the same time period, annual male survival probabilities were 0.56 (0.38-0.73), 0.64 (0.50-0.78), 0.74 (0.63-0.84), 0.83 (0.73-0.90). For females, annual survival probabilities in BCNP were 0.60 (0.51-0.69), 0.60 (0.52-0.67), 0.75 (0.67-0.81), 0.84 (0.76-0.90), in 2015, 2016, 2017, and 2018, respectively, while in FPNWR, female annual survival probabilities were 0.70 (0.56-0.82), 0.70 (0.57-0.81), 0.82 (0.73-0.89), 0.88 (0.82-0.94).

We examined space use by estimating annual and seasonal home range size. Annual and seasonal home ranges were larger for males than females during all seasons. Average home range size was 5.36 ± 0.85 (mean ± 95% CI) km^2 for males and 1.42 ± 0.19 km^2 for females. Across sites and seasons, female home ranges remained stable, suggesting that females were able to meet their space use requirements within a similar area throughout the year, in spite of variation in energetic requirements and resource availability. In contrast, we found that male seasonal home ranges were larger in the wet season (May-October) than the dry season (November-April), a result primarily driven by increases in space use during the rut.

Water levels influenced resource selection during the wet and dry seasons for males and females. During the dry season, males selected marshes, and as water levels increased, males selected areas closer to forest edges, roads, and flatwoods, and avoided swamps. Males also selected for marshes during the wet season, and as water levels increased, males selected areas nearer to roads, and avoided swamps. During the dry season females selected prairies, and as water levels increased, females selected forested edge and roads, and avoided swamps and marshes. During the wet season females selected prairies and as water levels increased females selected forested edges and roads, and avoided swamps, marshes, and hardwood hammocks. Males selected marshes and females selected prairies in both seasons, and increasing water reduced use of low elevation cover types such as swamps and marshes and increased use of roads and forest edges. The increased use of roads and forest edges by deer when water levels were high may explain the relationship between rising water levels and increased mortality risk as roads and forest edges were frequently used by Florida panthers. Spatial variation in Florida panther and human activity at the daily scale had limited effects on deer resource selection. During the wet season, increasing water levels reduced deer use of low elevation cover types such as swamps and marshes and increased use of roads and forest edges, except in cases where panther activity was high, when males and females selected areas with higher water levels. This suggests that as water levels rise in the wet season deer increase use of risky upland areas where forage is available and movement is not hindered by water. However, when risk was high due to panther activity, deer temporarily moved back into the low-quality high-water areas. Human activity had no effect on resource selection for males or females during the dry season. However, with increasing human activity females avoided roads during the wet season.

We evaluated the effects of cover type (hardwood hammock, marsh, pine flatwoods, prairie, and swamp), linear features (roads and forest edges) and water level on two attributes of movement: speed (i.e., step length) and path tortuosity (i.e., turn angles). During both seasons, males increased speed with increasing water levels and near hardwood hammocks, and decreased speed near marshes. During the dry season males increased speed near prairies and increased path tortuosity near prairies and flatwoods. During the wet season, males increased speed near swamps and roads, increased path tortuosity with increasing water levels and near roads, and decreased path tortuosity near marshes. During both seasons, females increased speed with increasing water level and near forest edges, and decreased speed near marshes. During the wet season, females decreased speed near roads and increased path tortuosity near marshes and flatwoods, and with increasing water levels. For both sexes in both seasons, speed increased with water levels which may explain decreased survival probability with increasing water levels.

We examined the effects of spatial variation in panther predation risk on sex-specific activity patterns of deer across seasons using camera data. Panthers were most frequently detected on trails and at night. In low-risk, off-trail areas, the sexes only differed in temporal overlap with panthers during the fawning season when female-panther overlap was greater. However, the sexes differed in overlap with panthers during all seasons in high-risk, on-trail areas where female deer overlapped with panthers more during the fawning season, and male deer overlap with panthers was greater during the rearing, pre-rut, rut, and post-rut seasons. Sex-specific seasonal variation in activity overlap with panthers appears to be driven by risky behaviors associated with reproduction (fawning for females and rutting for males) and may explain the seasonal variation in sex-specific survival.

We evaluated spatiotemporal trends in important deer predators on our study sites. Panther detection rates tended to be higher on BCNP than on FPNWR, although 2017 detections were similar among management areas. Black bear detection rates were similar at BI and FPNWR, and lower at AL. Bobcat detection rates were higher at BI than at either AL or FPNWR. Coyotes were detected infrequently (<1% of the camera days) over the course of the study, but coyote detection rates were highest during the fawning and fawn rearing seasons. Alligators were also rarely detected, but showed no temporal pattern in detections.

We did not observe strong temporal trends in deer detection rates during the camera study spanning January 2015 - December 2017. At AL, adult male deer detections rates decreased slightly over time, but no trend was evident for adult females. Adult female detection rates decreased slightly at BI. On FPNWR, detection was relatively stable for adult males and females. We developed a framework for extending existing unmarked spatial capture-recapture (SCR) methods to model deer density by combining camera and telemetry data from January 2015 - December 2017. We estimated adult male and female densities for 78, 2-week intervals during 2015-2017. We did not observe a significant decline in deer density on any site during the study. We estimated mean adult female densities for the 3-year period to be greatest on FPNWR (4.57 females/km^2; range: 1.61-7.97) followed by BI (3.54 females/km^2; range: 1.74-7.28) and AL (1.46 females/km^2; range: 0.56-2.63). For the 3-year period, we estimated mean male densities to be greatest on AL (2.03 males/km^2; range: 0.38-7.59) followed by FP (1.89 males/km^2; range: 0.45-4.39) and BI (1.75 males/km^2; range: 0.12-7.35). Using the ratio of density estimates, we calculated the mean adult deer sex ratio to be 58% males on AL, 33% males on BI, and 29%

males on FP.

We examined fawn recruitment using a novel SCR model applied to encounter histories of fawns uniquely identified using their spot patterns. We uniquely identified 145 fawns (BI = 57, AL = 33, FP = 55) in 2015, a relative dry fawning season, and 124 fawns (BI = 31, AL = 29, FP = 64) in 2016, an extremely wet fawning season. We estimated that 127 of the 267 fawns (42.5%) born in the 2015 fawning season survived to the recruitment age. While in 2016, we estimated only 36 of 234 (13.2%) fawns survived to recruitment. Density of fawn birth locations was greatest in the hardwood hammocks, followed by flatwoods, cypress, and open canopy habitats.

Antler casting dates can improve understanding of reproductive chronology and inform hunting regulations. To estimate the timing of antler casting, we used the trail camera data to categorize all male deer detections separated by a minimum of 5 minutes where the state of antler growth could clearly be identified as antlered (*n* = 9,046) or antlerless (*n* = 2,702). Annual antler cycles of growth, mineralization, and casting were closely timed with reproductive seasons. The proportion of antlerless males detected peaked on 27 January at FPNWR, 5 February at BI, and 15 January at AL.

We conducted a simulation study to identify the optimal design for camera-based monitoring of white-tailed deer populations in the study area. We developed 10 potential designs for each of the three study sites using simulated data based on the camera study. Our results demonstrated that the current design resulted in relatively low bias and high precision, however, little is gained with the labor-intensive step of deploying a portion of cameras off trails. The best design for estimating abundance and density involved 60 on-trail cameras at each of the three study sites. However, when balancing the costs of each design with the accuracy of the estimates, the optimal design involved 40 on-trail cameras at each of the three sites. Cameras should be spaced 400-600 m apart, equivalent to 1 camera per 16-36 ha. Cameras can be operated continuously throughout the year, or they can be deployed seasonally to reduce costs. For seasonal monitoring, we recommend that cameras be deployed during May and June for an 8-week survey when movement and associated detection probability parameters are most consistent. We recommend using our mean GPS-collar derived detection probability parameter estimates when modeling future camera data within an unmarked SCR framework. Sampling for eight weeks would yield four (2 week) closed population density estimates, which could be averaged to mitigate temporal variation during the survey. Male deer had greater variation in our estimates of the detection probability parameters because their movement is less consistent and our sample size for males was smaller than for females. Precision in male density estimates could be improved by incorporating individual capture histories by uniquely identifying males based on the antler morphology. However, if this approach is employed the survey should be shifted or extended to later in the antler cycle when males are more easily distinguished to create more robust capture histories. Similarly, fawns can be individually identified by their spot patterns and capture histories could be used to enhance estimates of fawn density. However, due to rapid mortality of fawns, we recommend dividing the 8-week survey into eight 1-week periods, to meet the assumptions of closed population models. We offer a more robust approach to estimating fawn survival and density within this report, but the method requires camera surveys be conducted from December-August and is not recommended as a long-term monitoring strategy. However, this approach could be used periodically or if concerns about fawn recruitment arise. Data on uniquely identified individuals is much more labor intensive to generate, but will yield more precise estimates of abundance for males and fawns. Relative to aerial surveys, which have been the primary method for monitoring deer population trends in South Florida, camera surveys cost more in the first year due to initial purchase of equipment than aerial surveys, but over a 5-year monitoring period, all evaluated camera survey designs were less expensive than annual aerial surveys. Furthermore, aerial surveys are not effective in closed-canopy habitats where deer cannot be seen from the air, and do not provide information available from camera surveys, such as distributions of other wildlife species or drivers of spatial variation in deer density and demographic parameters.

Our research has several important implications for managers seeking to maintain viable populations of white-tailed deer and their predators in South Florida. First, hunter harvest had minimal impact on the deer population in our study area suggesting that the current harvest regulations are appropriate. Second, deer survival and recruitment were negatively correlated with water levels, and our research supports conclusions from previous research that suggested that deer populations are negatively affected by water depths greater than 0.5 m. Water management decisions in South Florida integrate multiple objectives including wildlife management and our results provide empirical evidence that increasing water levels are likely to negatively affect the deer population and the predators they support. Finally, our results clearly show that predation by panthers is much greater now than it was when the last deer survival studies were conducted in the 1990's, and annual survival of adult female deer was lower during our study, especially during the first two years, than in other parts of the Southeast and previous studies in BCNP. Deer fecundity is also lower in South Florida than in other parts of the species' range, suggesting that managers should continue to monitor deer populations and be prepared to enact habitat management, such as mechanical removal of cabbage palm from pine uplands, and chemical control of invasive species to improve habitat quality, deer nutritional condition, and productivity. Although we found no evidence of rapid population declines during our study, three years of camera data is not enough to evaluate long-term trends, and additional monitoring is needed to determine if the deer population can maintain viability in the presence of increased predation rates. Given the relatively low survival rates we report for some years, this population has the potential to decline over broader time scales than our study. Continued close monitoring of the deer population is essential to detect any potential population declines and trigger management responses. The optimal camera design that we identified would allow for such monitoring. Furthermore, the substantial variability in survival rates suggest long-term monitoring and research will be required to disentangle the drivers of population dynamics in this unique predator-prey system.

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INTRODUCTION

White-tailed deer (*Odocoileus virginianus*) play important economic, cultural, and ecological roles in South Florida. Annual hunter expenditures in Florida exceeds $700M, with deer being the most popular game species in the state (U.S. Dept. of the Interior et al. 2011). Ecologically, deer are the dominant herbivore in South Florida, and they are the primary prey of the endangered Florida panther (*Puma concolor coryi*; Maehr et al. 1990a, Fleming et al. 1994, Caudill et al. 2019). Aerial surveys and hunter harvest data suggest that since 2000, white-tailed deer populations have declined in some units of the Big Cypress National Preserve (Garrison et al. 2011). While complete population crashes have occurred only in the southernmost units of the Preserve, harvest and survey data indicate variable population trends in other units (Florida Fish and Wildlife Commission 2019). Although these declines have coincided with changes in hydrology, habitat, and predators, the extent to which these variables have affected the deer population was unknown, raising questions about the long-term viability of deer in South Florida.

White-tailed deer were nearly extirpated from South Florida in the early part of the 20th century before making a slow recovery in the second half of the century. Prior to 1900, market and subsistence hunting were the primary threats to the species, and unregulated hunting of deer likely contributed to the decline of the Florida panther (Schortemeyer et al. 1991). Hunting regulations introduced in the early 20th century benefited deer populations, but in 1933, the New World screwworm fly (*Cochliomyia hominivorax*) arrived in South Florida and increased mortality rates (Dove and Parman 1935). In 1939, deer eradication programs were initiated in an effort to control cattle fever tick (*Boophilus spp.*) infestation. More than 9,000 deer were culled between 1939 and 1943, most of which were killed in Collier County (Davis 1943, Game and

Freshwater Fish Commission 1946, Alvarez 1993). During this period, major human encroachment also began with the construction of roads and railroads that facilitated commercial logging and increased human development (Schortemeyer et al. 1991). Similar to many southeastern states, deer populations in South Florida reached their lowest point in the 1940's (FWC 2007). In response, several management actions and regulations were implemented. The Florida Game and Freshwater Fish Commission (FWC) translocated deer from within the state, from out-of-state populations, and from Florida game farms to areas with low deer populations (Blackard 1971, Schaefer and Main 2014). Stricter harvest regulations were established, including defined seasons and "buck only" harvest. Subsequent screwworm eradication in 1958, as well as increased efforts to protect habitat, improve habitat quality, restock deer, and enforce harvest regulations resulted in the successful recovery of deer herds in South Florida.

Previous research on white-tailed deer ecology in South Florida was conducted before major changes in the predator community that occurred over the past 25 years. The primary change in the predator community has been the increase in the panther population following a genetic rescue effort. After decades of persecution, by the 1920's panthers only existed in Central and South Florida. In 1967, they were listed as an Endangered Species, yet the population continued to decline (Onorato et al. 2010). In the early 1990's, less than 30 individuals remained, and their range was limited to a few areas in South Florida (Onorato et al. 2010). In 1995, eight female cougars from Texas were introduced, five of which successfully reproduced (Onorato et al. 2010). The population has been steadily increasing since 1995, with recent estimates exceeding 200 individuals (Sollmann et al. 2013, McClintock et al. 2015, FWC 2017a). During the low point, prior to genetic rescue efforts, panther predation of deer was rare, and bobcats (*Lynx rufus*) were the primary predators of fawns and adult deer (Boulay 1992, Zultowsky 1992). Although it was assumed that panther predation on deer increased as the panther population grew, prior to this study, no deer survival research had been conducted to quantify mortality rates since 1995.

Another change that has occurred in this region is a major hydrological restoration effort. In 2000, Congress passed the 30-year Comprehensive Everglades Restoration Plan (CERP) to restore, protect and preserve the greater Everglades system, and water levels in many areas are expected to rise as a result (Sklar et al. 2005). Increases in water depth can affect deer populations by reducing survival and recruitment and by forcing deer to use lower quality habitat (Loveless 1959b, Labisky et al. 1999, MacDonald-Beyers and Labisky 2005). Even without additional increases in water depth, white-tailed deer populations in South Florida are characterized by low population density, low productivity, and smaller body size relative to other populations in North America, primarily due to seasonal flooding, climatic stress, and shallow, nutrient poor soils that contribute to the low nutritional value of available forage (Harlow and Jones 1965, Richter and Labisky 1985, Boulay 1992, Fleming et al. 1994, Labisky et al. 1995, Heffelfinger 2011). Previous research indicated that water depths greater than 50 cm were detrimental to deer populations (Loveless 1959b, MacDonald-Beyers and Labisky 2005), but these recommendations were made prior to the recent increases in the panther population, and the extent to which water levels and panther predation interact is unknown.

In addition to panther predation, another source of deer mortality is hunter harvest. Concerns over excessive harvest in the 1980’s and 1990’s led to several regulation changes designed to limit access, reduce overall pressure and harvest, and to better protect fawns and female deer (Schortemeyer et al. 1991). Antlerless harvest was prohibited in the private properties outside of BCNP, extending “buck only” harvest regulations more widely throughout the region. In the last decade, additional antler regulations have been enacted to protect yearling males throughout BCNP and to limit or exclude harvest in the areas with most significant deer population declines. The potential impact of these regulations and hunter harvest in this changed landscape has not been investigated since the early 1990's.

In addition to changes in panther abundance, hydrological conditions, and hunting regulations, several other important changes have occurred since the last studies on deer ecology were conducted in South Florida. For example, there is evidence that populations of other large predators in the area have changed. Black bear (*Ursus americanus*) populations have increased, whereas bobcat (*Lynx rufus*) abundance may have declined in response to increases in the panther population (Roberts and Crimmins. 2010, Telesco 2012, Humm et al. 2017). Additionally, coyotes (*Canis latrans*) colonized South Florida concurrent with panther restoration (Bragina et al. 2019). Invasive Burmese pythons (*Python bivittatus*) have become established in parts of South Florida and have caused widespread declines in the mesomammal community (McCleery et al. 2015). Prey populations have also changed in recent years. Wild pigs (*Sus scrofa*) were once the primary prey of panthers (Maehr et al. 1990a), but their populations appear to have declined rapidly in recent years (Caudill et al. 2019). The reason for the decline in the wild pig population is unknown, but panther predation, changes in hydrology, and a reduction in pig translocations following the reduction of hunting camps in BCNP likely played a role. Finally, the frequency and intensity of wildfire and prescribed fire have changed in recent years as managers are attempting to restore the historical fire regime after many decades of fire suppression (Day et al. 2015, Cherry et al. 2018).

Given concerns about deer population declines, and the important cultural, economic, and ecological roles that deer play in the rapidly changing South Florida ecosystem, reliable techniques are needed to monitor deer population dynamics and to guide management decisions. The most commonly employed technique used to monitor white-tailed deer in South Florida are aerial line transect distance sampling surveys (Buckland et al. 2001). Aerial surveys make it possible to cover large regions that are difficult to access, but aerial surveys are dangerous and expensive, and the precision and accuracy of estimates are often highly variable and dependent on time of day, flying altitude, observer skill, and population density within an area (Graves et al. 1972, Rice and Harder, 1977, Beasom 1979, Beasom et al. 1986, Caughley 1997, Dunn et al. 2002, DeYoung 2011). Moreover, detection probability can be zero in closed-canopy forested habitats, which prevents inferences about population density and trends in these areas (Dunn et al. 2002, Potvin et al. 2004, DeYoung 2011). Camera traps are increasingly being used to monitor wildlife populations because they are safer and less expensive than aerial surveys, they are relatively non-invasive, and they can be used to monitor entire wildlife communities (O'Connell et al. 2011). However, only recently have statistical methods been developed to estimate abundance and density of unmarked animals detected using cameras, and little work has been conducted on the effectiveness of these techniques for monitoring wildlife populations in South Florida (Chandler and Royle 2013, Sollmann et al. 2013).

RESEARCH OBJECTIVES

Motivated by concerns about deer population declines and changes in the South Florida ecosystem, this research was initiated in January 2015 to (1) understand the effects of hydrology, hunting, and predation on white-tailed deer population dynamics, and (2) develop a camera trapping methodology for large-scale investigation and monitoring of white-tailed deer populations in South Florida. We collected GPS telemetry data to evaluate cause-specific mortality, annual and seasonal survival, and factors influencing mortality risk for adult deer. We used telemetry data collected on 263 deer to quantify home range area, resource selection, movement, and activity patterns across hydrological and biological seasons. We developed camera-based methods to investigate spatiotemporal trends in detection rates for deer and their predators. We used data from 180 camera traps and telemetry data to study spatial and temporal variation in the density of adult female and male deer, and to quantify fawn density and recruitment. We leveraged this information to evaluate multiple trail camera design scenarios to provide recommendations for future investigation and monitoring of deer populations.

STUDY AREA

The study occurred within the Big Cypress Basin, which is characterized by hot, wet summers (May–October), and cool, dry winters (November–April; Hela 1952, Duever et al. 1986). This region consists of a mosaic of vegetative community types influenced by seasonal fluctuations in hydrology (McPherson 1974). A secondary influence on the ecosystem is fire, which controls the structure and composition of plant communities (Duever et al. 1986). Within the Big Cypress Basin, primary vegetation communities included pine (*Pinus spp.*) forests, cypress (*Taxodium spp.*) forests, and freshwater marshes interspersed with hammock forests and wet prairies (McPherson 1974, Duever et al. 1986). Pine and hammock forests are found at higher elevations than the surrounding cypress forests, marshes, and wet prairies (McPherson 1974). Pine forests are typically drier than the other vegetation communities, but they become inundated following heavy rainfall and can persist in a flooded state for several months during the wet season. Pine forests often occur on mineral soils and are dominated by slash pine (*P. elliottii*) with an understory of cabbage palm (*Sabal palmetto*), saw palmetto (*Serenoa repens*), and evergreen shrubs. Islands of hammock forests, which often occur on bedrock outcrops in wetlands, are comprised of hardwoods, palms, ferns, and shrubs (McPherson 1974, Duever et al. 1986).

Cypress forests occur at low elevations and they are often inundated for most of the year. These forests have organic soils on top of limestone bedrock and can vary in composition from large, open stands of bald cypress (*T. distichum*) to mixed swamps with dense tangles of trees, vines, shrubs, and epiphytes (Duever et al. 1986). Wet prairies and marshes are seasonally inundated communities dominated by emergent vegetation. Marshes have deeper standing water, but occasionally dry out. These areas consist of organic soils dominated by sawgrass (*Cladium spp.*) and rushes (*Juncus spp.*) with alligator flag (*Thalia geniculata*) in deeper depressions (McPherson 1974, Duever et al. 1986).

Within the Big Cypress Basin, sampling occurred in the BCNP and the adjacent Florida Panther National Wildlife Refuge (FPNWR; Fig. 1). The study area was located north of I-75, and in BCNP, research activities were conducted in the Bear Island (BI) and North Addition Lands (AL) management units. These study sites were selected because they occurred within the range of the Florida panther, and because they spanned gradients in hydrology and hunting pressure. BI contained a network of off-road vehicle (ORV) trails for public use by permit and permitted licensed hunting. Public access was allowed on AL, but ORVs were prohibited and fewer hunting permits were issued. The majority of FPNWR was restricted from public access, although it contained a network of ORV trails to facilitate management practices. Hunting was prohibited in FPNWR.

STUDY DESIGN AND FIELD METHODS

We studied deer population dynamics using GPS telemetry and camera traps. Telemetry was used to study cause-specific mortality rates, space use, resource selection, and movement behavior. Camera traps provided information about spatial and temporal variation in abundance and distribution as well as information on activity patterns, recruitment and reproductive phenology.

Capture Procedures

During 2015-2017, we captured 294 deer in BCNP and FPNWR using three methods: net-gunning (Barrett et al. 1982) from helicopters, chemical immobilization via darting, and rocket netting (Hawkins et al. 1968; Appendices B, C; Table 1). Deer captured via net-gunning and rocket netting were blindfolded and physically restrained during morphometric data collection and processing, and were released within 20 min of capture. Deer captured via darting were administered a mixture of xylazine-hydrochloride (2.2 mg/kg body weight; Congaree Veterinary Pharmacy, Cayce, SC, USA) and Telazol (4.4 mg/kg body weight; Congaree Veterinary Pharmacy, Cayce, SC, USA; Kreeger et al. 2002). Once immobilized, deer were blindfolded and placed in the sternal position, and received an application of ophthalmic ointment. Heart rate, body temperature, and respiration were monitored every five min until 90 min post-darting when we reversed xylazine-hydrochloride with Tolazoline (1.4 mg/kg body weight; Kreeger et al. 2002). We captured deer following protocols accepted by University of Georgia IACUC permit A2014 07-009-Y3-A1.

For each captured deer, we recorded spatial coordinates, observer names, capture method, time of capture, sex of deer, time of release, and the following morphological measurements: estimated age based on dentition, estimated weight, total length, tail length, ear length, hindfoot length, chest length, neck length, body condition score, pelage score, ectoparasite score, and tail fat score. Each captured deer was assigned a unique identification number and ear-tagged in both ears. All captured adult deer (≥1 yr, 172 females, 91 males) were fit with Iridium ATS (Advanced Telemetry Systems, Isanti, MN, USA) Model G2110E GPS collars programmed to record a location every 3-4 h on a rotating schedule such that each hour of the day was represented every four days.

Telemetry Monitoring

From January 2015 - December 2018, we monitored the daily transmissions and locations for all GPS-collared individuals. In a small number of cases when GPS collars failed, we transitioned to monitoring the individual using radio-telemetry and triangulating the deer's location from two or three bearings.

Mortality Investigations

Mortality investigations were initiated if a collar transmitted a mortality signal, if the telemetry data indicated missing locations for a collar (indicating deer was cached and collar was unable to connect to satellite), or if locations were clustered (indicating deer was not moving as expected based on typical movement patterns). At each mortality site, we recorded GPS location, date, time, and we took detailed field notes and photographs. We assessed the cause of mortality based on kill site and feeding site evidence (e.g. predator tracks, scat, feeding behavior, signs of struggle, bite wounds). Predation events were deemed "confirmed" if the kill site could be positively identified and sign of a single predator species was present at the kill site, or if bite wounds with associated subcutaneous hemorrhaging were observed and were consistent with observed predator sign. If neither of these conditions could be met, but all other field evidence indicated predation was likely, the predation was deemed "probable," as we could not definitively rule out the possibility of scavenging. In predation cases, disturbance to and around the carcass was minimized. When the cause of death was unknown based on field sign and no clear evidence of predation or feeding was present, carcasses were necropsied in the field or when feasible, remains were collected and transported to FWC's Wildlife Research Lab in Gainesville, FL for full necropsy.

Trail Camera Monitoring

We deployed 180 motion-triggered white-flash cameras (HCO Outdoor Products, model SG565FV, Norcross, GA, USA) from January 2015 - December 2017 (Figs. 1, 2, 3, 4). At each of the three study sites, we placed 60 cameras within a 29 km^2 rectangular region. The rectangular regions were separated by at least 13 km. At each site, 40 cameras were placed on ORV trails and the remaining 20 cameras were placed approximately 250 m from the trail (Figs. 2, 3, 4). We selected on-trail camera trap locations by using ArcGIS 10.2 (Environmental Systems Research Institute [ESRI], Redlands, CA, USA) to overlay a 700 m grid over the study area and placing cameras on the closest trail to each grid point. To increase the probability of detecting animals at off-trail camera locations, we deployed cameras on the most well-defined wildlife trail or habitat edge within 50 m of the selected point location, while maintaining a distance of approximately 250 m from the closest trail. We positioned cameras approximately 0.30 m above the ground, oriented either north or south, and adjusted height according to surface water levels to avoid inundation. No bait or attractants were used at camera sites. Vegetation was cleared as needed to minimize false-triggering of cameras. We visited each camera on 30-day intervals for data retrieval and camera maintenance. In the event of a wildfire or prescribed burn, we preemptively removed cameras when access was available and replaced cameras once the area was safe.

Data Processing

Telemetry Monitoring Data

Initial raw data counted 732,813 location records for 263 deer. We excluded GPS telemetry data recorded within two weeks after capture to avoid the influence of atypical behaviors related to capture. We discarded GPS locations recorded after the mortality date, and we removed 6 outlier locations that were clearly location errors because they were >5 km from locations recorded immediate before and after. A set of collars experienced a software bug that caused a temporal drift in data uploaded to the Iridium network. For affected collars that we were able to recover, we corrected the errors by downloading the data that was logged on the collar. For unrecovered collars, we excluded locations acquired after the time-bug occurred. Finally, we corrected dates for a set of collars that improperly recorded Julian dates in the years following the 2016 leap year. After processing, we had a total of 590,533 location records for 241 individuals.

Trail Camera Monitoring Data

All trail camera photos were downloaded from the onboard memory cards and were processed using Media Pro (Phase One, Version 2.1.0.161, Copenhagen, Denmark). We tagged each photograph with keywords describing the vertebrates in the photograph and the environmental conditions (Appendix B). Each photograph received tags for study area and camera ID. We recorded the presence of fire, humans, and vehicles. We recorded the taxonomic class of all vertebrates in each photograph. Detections of all mammals, Wild Turkey (*Meleagris gallopavo*), and American alligator (*Alligator mississippiensis*) were classified at the species level. White-tailed deer camera photographs underwent further processing to identify group size, the presence of spots on fawns, and the sex of adult deer. When we encountered a deer marked with ear tags or a GPS-collar in the photos, we tagged the photograph with the deer’s individual identification number.

Remotely Sensed Data and Hydrology Data

We derived habitat data from the Florida Natural Areas Inventory (FNAI) land cover data (10x10 m resolution, FNAI, 2016). We reclassified habitat data into ten primary cover types using FNAI, version 3.2 site-level land cover data (Table D1). Once cover types were reclassified, Euclidean distance to each cover type from each raster cell was calculated to provide a continuous distance surface for each land cover type. Elevation data used in this study was derived from an ASTER digital elevation model (30x30 m resolution, ASTER GDEM 2).

We derived a surface water index from hydrologic well data (DBHYDRO 2019) and Florida Natural Areas Inventory (FNAI) land cover data (10x10 m resolution, FNAI, 2016). Hydrologic well data used in this analysis is the average surface water (ft) derived from DBHYDRO surface water data (DBHYDRO 2019, Datum = NGVD29, Fig. 5). We used vegetation type to inform the spatial extent of surface water inundation. The surface water needed to inundate specific land cover types was estimated throughout our study area (Robert Sobczak, *unpublished data*, Fig. 6). We classified land cover types from Florida Natural Areas Inventory data into the cover types needed to estimate surface water levels across the study area (Table D2). We created raster layers of surface water depth estimates across the study area for each day from 1 January 2015 to 1 March 2019. We then used surface water depth raster to create surface water index (SWI) raster layers. Specifically, SWI was calculated by subtracting the surface water depth raster layers from the vegetation threshold (Fig. 6b) raster such that higher values indicate more flooding and lower values represent drier conditions. SWI allowed us to describe spatial variation in surface water and habitat-specific inundation. For example, SWI captures trends associated with habitat-specific inundation such that surface water tends to be deeper in cypress and marshes than in pine uplands and hardwood hammocks during flood events.

OBJECTIVE 1

Survival Analysis

Survival Analysis – Introduction

Understanding the effects of predation, hydrology, and hunting on deer survival is essential to managers seeking to maintain viable deer and panther populations. Previous studies on deer survival occurred before the 1995 Florida panther genetic rescue program, and prior to this study, minimal information was available regarding the panther predation rates on white-tailed deer. We evaluated the following hypotheses and predictions. We predicted that survival rates would be lower than previously reported due to the growth of the panther population. We expected that survival would be negatively correlated with water depth because of shifts in habitat selection due to high water. We hypothesized that female survival would be greater than male survival, especially during the breeding season, because males make risky decisions when seeking mates. We expected that female survival would also vary seasonally, with increased mortality rates during fawn rearing when females may exhibit risky behaviors to meet increased nutritional demands. We also hypothesized that deer survival would exhibit a temporal trend in relation to unobserved changes in the predator community.

Survival Analysis – Methods

To understand the factors influencing deer survival in South Florida, we collected known-fate survival data on deer that were captured and monitored between January 2015 and December 2018 in FPNWR and BCNP. Deer were captured across gradients in hydrology and hunting pressure. Capture, marking, and collaring methods are described in the Capture Procedures section above. When collars transmitted a mortality signal or when we detected unusually low movement, we conducted a mortality investigation within 24 h. We determined the cause of death via field investigation as described in the Mortality Investigations section.

We assessed the factors influencing deer mortality rates using known-fate survival models (Cox and Oakes 1984). The exact time of mortality could not be determined as is required for standard continuous time survival models, so we used a discrete-time survival model with daily hazard rates. For each individual, the date of mortality t_i was modeled as a categorical random variable with probability distribution:

$$p(t_i) = \Pr(t_i = j) = \left\{ \prod_{j=f_i}^{t_i - 1} 1 - \lambda_{i,j} \right\} \lambda_{i,t_i}$$

where $\lambda_{i,j}$ is the hazard rate defined as the probability of dying during day *j*. Because the individual is known to be alive on the day of capture (f_i), the hazard rate was set to zero on and before f_i. For subsequent time periods, the hazard can be interpreted as daily mortality risk, with daily survival probability being $\varphi_{i,j} = 1 - \lambda_{i,j}$. Therefore, the equation above can be interpreted as the probability of surviving from the capture date until day t_i, and then dying on day t_i. Right-censoring occurs when an individual survives beyond the last date that the individual was monitored (J_i). The probability of such an event is given by $\Pr(t_i > J_i) = \prod_{j=f_i}^{J_i} 1 - \lambda_{i,j}$.

We modeled daily survival probability as a linear combination of covariates on the logit scale. Covariates considered to potentially affect deer survival included hydrological conditions (as measured by water depth and time since the last dry day), sex, behavioral season (rut, post-rut, fawning, and rearing), management area (BCNP and FPNWR), and time (i.e. a simple trend effect).

To estimate the importance of panther predation pressure, we conducted two analyses. In the first analysis, all causes of mortality were treated equally, and observations were censored when a deer survived to the end of the study or when its collar failed. In the second analysis, we focused exclusively on panther predation. Observations of deer that died by other causes were censored on the mortality date. This is equivalent to a two-stage competing risks survival analysis.

To assess the effect of hydrology, we obtained data from the Everglades Depth Estimation Network (EDEN) database through the Explore and View EDEN (EVE) web application. We downloaded daily median water levels for water gauges BCA1, BCA12, BCA17 and BCA18 for the period of our study, and corrected these values to obtain actual water depth (i.e. water height in relation to the ground). Recorded values ranged from -56 cm to 93 cm (Fig. 7). Additional details are described in the Data Processing section.

Biological seasons were categorized as follows: rut (July-September), post-rut (October-December), fawning (January-March), and rearing (April-June). We modeled seasonal effects on survival using two approaches. First, we considered biological season to be a categorical variable with the four levels described above. We also modeled seasonal variation using a continuous covariate defined as the number of days since peak fawning (15 February; Chandler et al. 2018), ranging from 0 to 182 (where 0 corresponds to peak fawning). We considered possible interactions between sex and season (both for continuous and categorical variables), sex and water depth, and season and time trend.

We implemented our models in a Bayesian hierarchical framework using the statistical computing software R (R Core Team 2019), using the package *rjags* (Plummer 2018) to call the program JAGS (Plummer 2003) to perform Markov chain Monte Carlo (MCMC) computations.

Model selection was done using the Watanabe-Akaike information criterion (WAIC; Watanabe 2010, Hooten and Hobbs 2015, Vehtari et al. 2016), and selecting the model with the lowest WAIC.

Survival Analysis – Results and Discussion

We analyzed survival data on 241 individual collared deer (156 females and 85 males). Out of the 293 initially captured individuals, we collared 263. Out of the 293 initially captured individuals, we collared 263. We excluded GPS telemetry data recorded within two weeks after capture to avoid the influence of atypical behaviors related to capture and therefore excluded 22 deer for which we had less than two weeks of GPS data, leading to a final sample size of 241 deer for this survival analysis. Of these, 188 individuals were captured in BCNP and 53 in FPNWR. The dataset consisted of 118,254 observation-days. Of the 241 individuals, 107 survived to the end of the observation period, and 134 died (Table 2). Causes of death included: predation (panther, bobcat, bear, and alligator), pathology, poaching and hunting. The most common cause of death was predation by Florida panthers (*n* = 96, 71.6%), followed by other predation (e.g. bobcat, bear, alligator, *n* = 14, 10.4%; Table 2). We only recorded one case of legal hunting harvest, and two poaching cases (one confirmed and one suspected) for deer included in the survival analysis. One other deer was legally harvested during the study, but the harvest occurred several months after its collar failed and monitoring ceased.

Previous research on deer survival in the hunted portion of BCNP and in the Everglades National Park (ENP) where hunting is prohibited, conducted prior to the 1995 Florida panther genetic rescue program, attributed 50% of adult male mortality to legal and illegal harvest and 17% to bobcat predation (Labisky et al. 1995). Labisky et al. (1995) reported only a single case of Florida panther predation (equating to 3% of recorded deaths, equal to the amount of alligator predation). In contrast, results from our survey clearly show that panthers were responsible for most of the observed mortalities. In their 3-year study in BCNP and ENP, Labisky et al. (1995) reported that, although hunting (both legal and illegal) was the principal cause of death for males in BCNP, bobcat predation was also an important factor regulating the growth and abundance of the deer population. Bobcats impacted the deer population through predation on fawns and adults, representing 17% of recorded deaths (6 cases out of 36 adult mortalities). In a study conducted between 1986 and 1993, Land et al. (1993) reported that bobcats were responsible for 11 of 23 recorded deaths in adult females in BCNP. In our study, conducted after the Florida panther population increased substantially, bobcat predation on adult deer was much lower, and Florida panther predation represented 82.8% (96 of 116) of the mortalities that could be assigned causes of death.

The effects of hydrology, sex, season, management area, and time on deer survival were similar for the global analysis (including all causes of mortality), and the second analysis focusing exclusively on panther predation (Tables 3, 4). Both analyses included the same set of four supported models within 2 WAIC units of the best model (Tables 3, 4). The ranking of the supported models varied slightly between the two analyses. The top models indicated that variation in daily survival probability was explained by sex, water depth, number of days to peak fawning, time, and an interaction between sex and the number of days to peak fawning. The most supported models differed primarily in whether management area and an interaction between sex and water depth were included. Because supported models (in both analyses) had similar WAIC scores, we present results from the model that including both management area and the interaction between water depth and sex to illustrate the effects of these variables on survival.

Female survival probability was strongly negatively correlated with water depth (Tables 5, 6; Fig. 8), while male survival did not appear to be significantly impacted by the water depth observed in our survey. This difference in sex-specific survival probability response to water depth was even more important when considering panther-only mortality (Table 6; Fig. 8). The impact of water depth on deer survival is well-documented, but previous research occurred before the panther population rebounded following genetic rescue in 1995. Several studies have demonstrated that prolonged flooding caused by episodic tropical storms can increase adult deer mortality rates (Brunett 1958, Loveless 1959b, Lampton 1982, Langenau et al. 1984, Kushlan 1989, MacDonald 1997, Labisky et al. 1999, MacDonald-Beyers and Labisky 2005). Flooding can also reduce fawn production and survival (Fleming et al. 1994, Labisky et al. 1995). Flooding can affect deer survival by reducing mobility (MacDonald-Beyers and Labisky 2005) and availability of escape habitat (Fleming et al. 1994), which might explain the effect of water depth on female deer survival in our study. Previous studies have suggested that water depths exceeding 0.5 m are detrimental to deer populations (Loveless 1959b, Labisky et al. 1999, MacDonald-Beyers and Labisky 2005). Our study reinforces the results of the impact of water depth on female deer survival, although our analysis suggests that the relationship between survival probability and water depth is continuous. Female survival was decreased by up to 9% when water level reached 0.5 m, and up to 22% at 1 m (Table 6; Fig. 8).

Unlike in the 3-year survey in southern units of BCNP conducted by Labisky et al. (1995) where they reported that hunting (both legal and illegal) was the principal cause of death for males, hunting did not prove to be an important source of mortality in our study. This is likely explained by hunting regulations that prohibited or reduced access to many parts of the study area. For example, harvest was restricted to males with at least three antler points on BCNP, and no hunting was allowed on FPNWR.

Female survival probability was greatest during the rut, and lowest during the fawning season. Male survival probability was generally lower than female survival, except during the fawning season. Male survival probability peaked during fawning and rearing seasons, and was lowest during the rut (Fig. 9). We hypothesize that this strong negative covariance between male and female survival rates among seasons is the result of interactions between predator-prey dynamics and deer life history strategies. Differential survival between sexes of white-tailed deer is well documented (e.g. Nelson and Mech 1986a; Nixon et al. 1991, 2001; Demarais et al. 2000; Lopez et al. 2003), and sex-specific predation vulnerability can vary depending on biological seasons (Beier and McCullough 1990). Variations in survival as a function of sex and season can be related to differences in social behavior for males and females, in particular reproductive behavior. For example, during the rut, males have greater seasonal movements, often move into less familiar areas, and express high levels of male-male aggression (Loison et al. 1999, Demarais et al. 2000), which might explain the decrease in male survival probability we observed during this biological season. Our results clearly show that female survival probability dropped during the fawning season, possibly in response to an increase in predation because of the presence of a fawn, and the increased need to forage.

Management area was included in the set of supported models, but the difference between survival in FPNWR and BCNP was small, and the 95% Credible Interval (CrI) included zero (Tables 5, 6; Figs. 10, 11). This suggests that the different management practices employed in the two areas do not have a strong impact on adult survival, and most of the differences in survival between the areas can be attributed to the other factors included in the model. However, it is important to note that we did observe higher rates of recruitment and density in FPNWR than in BCNP (See sections: Abundance and Density and Fawn Recruitment and Phenology).

We observed an increasing temporal trend in survival (Tables 5, 6; Figs. 9, 10). Yearly male survival probabilities in BCNP were 0.44 (95% CI: 0.31-0.56), 0.53 (0.42-0.63), 0.65 (0.54-0.75), 0.76 (0.64-0.85), in 2015, 2016, 2017, and 2018, respectively. In FPNWR, during the same time period, yearly male survival probabilities were 0.56 (0.38-0.73), 0.64 (0.50-0.78), 0.74 (0.63-0.84), 0.83 (0.73-0.900). For females, survival probabilities in BCNP were 0.60 (0.51-0.69), 0.60 (0.52-0.67), 0.75 (0.67-0.81), 0.84 (0.76-0.90), in 2015, 2016, 2017, and 2018, respectively, while in FPNWR, female survival probabilities were 0.70 (0.56-0.82), 0.70 (0.57-0.81), 0.82 (0.73-0.89), 0.88 (0.82-0.94).

Previous studies conducted in South Florida, prior to the start of the 1995 Florida panther genetic rescue program, reported higher annual survival rates. In a study in the BCNP and the Everglades National Park (ENP), Labisky et al. (1995) reported mean annual survival rates between 1988 and 1992, pooled across both locations, equal to 0.67 and 0.81 for adult males and adult females, respectively. In their study, annual survival rates of adult males in the hunted BCNP population (mean annual survival rate: 0.43) were lower than those reported in the nonhunted ENP population (0.84). For females, Labisky et al. (1995) found that survival rates were similar in BCNP (0.87) and ENP (0.74). Male annual survival rates ranged from 0.22-0.62 in the hunted BCNP population, and from 0.75-0.91 in the nonhunted ENP population (Labisky et al. 1995). Female annual survival rates ranged from 0.81-0.93 in the BCNP, and from 0.45-1.00 in the ENP (Labisky et al. 1995). In separate findings, mean adult female survival rate in the hunted Bear Island Unit of BCNP between 1987 and 1991 was estimated to be 0.81 (Land 1991), while hunted population in northern Florida experienced mean annual survival rates of 0.90 and

0.56, for adult females and males respectively (Labisky et al. 1991). In comparison, female survival rates were lower during our study (in particular during the first two years) than in what appears in previous research.

After an initial drop during the first year of our survey, the survivorship curve leveled off during the second half of our study (Fig. 11). The cause of the increasing temporal trend in survival probability is unknown, but it may have been the result of wetter conditions during the first half of the study (Fig. 7), or a change in predation pressure.

Our results are largely consistent with our initial hypotheses, and they indicate that patterns of deer survival have changed greatly in South Florida since the last studies were conducted before the panther restoration and before changes in hunting regulations. Results from our survey show that most mortalities were due to predation by Florida panthers and that deer survival probability was negatively correlated with water depth. Negative effects of water depth on survival were more prominent in females than males. Hunting did not prove to be an important source of mortality during our study.

Space Use – Utilization Distributions and Home Ranges

Space Use – Introduction

White-tailed deer seasonal and lifetime space use varies across their range and is influenced by numerous factors, including climate, forage quality, predation risk, and population density (Burt 1943, Lewis and Murry 1993, Kilpatrick et al. 2001). However, deer in the Big Cypress Basin face a unique combination of abiotic factors from frequent disturbance regimes. Much of the landscape is inundated during the wet season, and both prescribed burns and wildfires occur regularly (Hela 1952, Duever et al. 1986, Day et al. 2015, Cherry et al. 2018). Deer in South Florida have a diffuse breeding chronology more similar to ungulates in semi-tropical environments, with longer breeding and fawning seasons (Loveless 1959b, Fleming et al. 1994). Deer must also balance the threat of predation from their primary predator, the Florida panther, with seasonal resource needs (Crawford et al. 2019). This unique combination of disturbance regimes, breeding chronology, and predation has the potential to affect white-tailed deer seasonal and lifetime space use.

Frequent disturbance regimes influence seasonal spatial ecology and reproductive chronology of deer in the Big Cypress Basin. Fawning coincides with the peak of the dry season, which is believed to be an adaptation to the hydrological seasons, as females experience lower reproductive success during times of high water (Land 1991, Land et at. 1993, MacDonald-Beyers and Labisky 2005). Seasonal fluctuations in surface water depth alter deer movement, space use, and resource availability (Fleming et al. 1994). Fire can alter both vegetation structure and forage quality, positively influencing forage quality while reducing concealment cover (Lay 1967, Halls 1970). Reductions in concealment cover may improve the ability of prey to detect and evade predators by opening sight lines. In response to a wildfire in the BCNP, deer with home ranges overlapping the area affected by wildfire increased their space use post-fire to incorporate more burned area within their home ranges (Cherry et al. 2018).

Home range size may also fluctuate through biological seasons as sex-specific resource requirements change. For females, energetic demands increase throughout gestation and peak during lactation, but females must also balance offspring safety with the high energetic demands of reproduction. Males accumulate energy stores during the pre-rut phase to increase reproductive success and may increase home range size during the rut to increase access to resources or mating opportunities (Main et al. 1996).

Previous studies in the Big Cypress Basin have documented seasonal, annual, and lifetime space use. In BI, Land et al. (1993) reported an average home range size for adult females of 1.95 km^2 (0.24–2.24 km^2), which remained stable between fall, spring and summer seasons. A study conducted in the Stairstep Unit of the BCNP reported mean annual adult male home ranges of 7.00 km^2 and mean adult female home ranges of 5.40 km^2. Home ranges were also calculated for both hydrological and biological seasons. Hydrological seasons were defined by Labisky et al. (1995) as wet (May – October) and dry (November – April), while biological seasons were defined as lactation (April – June), rut (July – September), gestation (October – December), and parturition (January – April). Both females and males showed no difference in mean home range size between wet and dry seasons, and males showed no difference across biological seasons. Females, however, demonstrated differences across biological seasons. Mean home ranges during gestation were greater than home ranges during the other three seasons, and home ranges during parturition were greater than home ranges during lactation (Labisky et al. 1995).

In order to understand both seasonal and annual space requirements of deer in the Big Cypress Basin, we estimated cumulative and seasonal home ranges (i.e., utilization distributions). Cumulative home ranges were created using all GPS location information collected during the study period, 1 January 2015 – 31 December 2018. Seasonal home ranges were created for both hydrological seasons and biological seasons. We defined hydrological seasons as wet (May – October) and dry (November – April), and biological seasons as fawning (January – March), fawn-rearing (April – June), rut (July – September), and post-rut (October – December). For males, the fawn-rearing season (henceforth, rearing) is a time for antler growth and accumulation of body mass in preparation for the rut. For females, gestation occurs during the post-rut season. Males and females incur their greatest energetic expenditures related to reproduction during the rut and rearing seasons, respectively (Main et al. 1996b). During the rut, male activity rates peak as they actively seek out breeding opportunities to maximize their reproductive success (Nelson and Mech 1981, Webb et al. 2010, Crawford et al. 2019). Conversely, female reproductive success depends on the survival of offspring, which female ungulates maximize by utilizing smaller areas that provide forage for themselves and hiding cover for their fawns (Main and Coblentz 1996, Main et al. 1996, Bongi et al. 2008, Cherry et al. 2017). We predicted home ranges for males would be larger than for females, regardless of hydrological season. We expected that males would have larger home ranges in the wet season, which includes the rut, than during the dry season. We predicted that females would have smaller home ranges during the dry season, when they need to balance both energetic demands and offspring safety in a predator-rich system. Across biological seasons, we predicted that males would have the largest home ranges during the rut in order to maximize mating opportunities. For females, we predicted that relative to other seasons female home ranges would be smaller during fawning and rearing, when females need to balance energy requirements with offspring safety.

Space Use – Methods

We used dynamic Brownian Bridge Movement Models (dBBMM) to estimate cumulative and seasonal utilization distributions (UDs). Conceptually, UDs are 3-dimensional probability distributions which provide the probability density that an animal is found at a given point in space. UDs were reported as the 2-dimensional area included within the 95% isopleth for all qualifying individual-season combinations (Horne et al. 2007, Kranstauber et al. 2012). Dynamic BBMMs are well suited to GPS collar data because they can accommodate fine scale location data and account for serial autocorrelation (Horne et al. 2007). These models calculate UDs using dynamic movement paths between successive points rather than the locations themselves. Because dBBMMs incorporate serial location data and the potential pathways linking two successive GPS locations, areas of frequent use in UDs are more likely to be connected by actual movement corridors used by individuals (Kranstauber et al. 2012, Byrne et al. 2014).

To estimate cumulative home ranges, we fit dBBMMs with a subset of deer that were monitored for at least one year and collected 80% of their scheduled GPS locations during 1 January 2015 – 31 December 2018. We used a window size of 21 steps, a margin size of 5 steps, and an 18 m location error for all deer, as visual inspection indicated these settings were sufficient to identify relevant changes in behavior. We used a linear regression model to estimate the effects of sex, location, and their interactions with study area (FPNWR or BCNP) on home range area.

To estimate seasonal home ranges, we fit dBBMMs with a subset of deer that deer survived 80% of the defined season and collected 80% of their scheduled GPS locations. We modeled the effects of sex, season, study site location, and their interaction on the seasonal home range area using a generalized linear mixed model. These models can include both fixed and random effects and are generalizable across a range of distributions (Bolker et al. 2009). We treated the individual deer as random effect on the y-intercept term to account for variation among individual deer.

Space Use – Results and Discussion

We collected 590,533 GPS locations for all deer (174,159 locations for males; 416,374 locations for females) during 1 January 2015 – 31 December 2018. We estimated cumulative home ranges for 114 deer (32 males, Fig. E1; 79 females, Fig. E2). We observed an interactive effect of sex and location ($p = 0.002$, $\beta = -0.98$, se = 0.31) on home range size, with males maintaining larger home ranges (5.36 ± 0.85 km^2) than females (1.42 ± 0.19 km^2). However, male home ranges were smaller on FPNWR (3.88 ± 0.96 km^2) than BCNP (6.25 ± 1.08 km^2), while female home ranges were similar across sites (FPNWR = 1.24 ± 0.26 km^2 and BCNP = 1.47 ± 0.24 km^2, Figs. E3, E4, E5).

For the two hydrological seasons, we calculated 381 seasonal home ranges (93 males; 288 females) for 153 individual deer. In both the FPNWR and the BCNP, male seasonal home ranges during the wet season were 1.8 times larger than during the dry season, while females showed no difference in seasonal home range areas between hydrologic seasons (Fig. 12, Table 7). Male hydrological seasonal home ranges were 2.8 to 4.4 times larger than female home ranges across both seasons and sites. For the four biological seasons, we calculated 872 seasonal home ranges (247 males, 625 females) for 188 individual deer. Mean seasonal home range area for males in both the FPNWR and the BCNP did not differ between the fawning, rearing, and post-rut seasons, however the mean seasonal home range during the rut was 1.6 to 2.4 times larger than during other seasons. Females did not show any significant differences between fawning, rearing, rut, or post-rut seasons in either FPNWR or BCNP (Fig. 13; Table 8). Across sites and biological seasons, male home ranges were 2.3 to 5.2 times larger than females.

Our results support previous home range estimates in our study area where seasonal home ranges for females were stable across the year (Land et al. 1993). However, our findings contrast to results from the southern portion of BCNP, where males and females showed no difference in space use across hydrological seasons, but female space use varied across biological seasons. Female home ranges were larger during gestation, and home ranges during partition were larger than home ranges during lactation (Labisky et al. 1995). These differences may have been due to the habitat differences; the southern portion of BCNP is dominated by wet prairie interspersed with hardwood tree islands (Duever et al. 1986).

In sexually dimorphic ungulate species, different space use requirements are expected between the sexes. Greater male body mass necessitates use of more space than females to meet energetic requirements (Main et al. 1996). Our results support this premise as males maintained larger cumulative and seasonal home ranges than females across all sites and seasons.

Across sites, males had significantly larger home ranges in BCNP compared to FPNWR. Such patterns in home range size may arise due to differences in forage quality or quantity across the study locations as these two factors are major determinants of vertebrate home range size (Mace et al. 1984). Relative to BCNP, FPNWR occurs at a slightly higher elevation and undergoes significant habitat management via prescribed burning and chemical and mechanical removal of invasive species. Prescribed burns are usually conducted during the late winter and early spring seasons to promote growth of herbaceous understory, benefit fire-maintained communities, and promote overall higher yields of forage (Stansky and Harlow 1981, Kilburg et al. 2014). Prescribed burning produces high-quality forage by increasing palatability and crude protein percentages which enhances deer nutritional condition and reproductive output (Stransky and Harlow 1981, Main and Richardson 2002, Cherry et al. 2017, Cherry et al. 2018). Thus, it is likely that higher quality forage due to higher elevation and effective habitat management in FPNWR explains the size differences in home ranges for males across sites. However, there was no difference in female home range area between sites suggesting other factors such as population structure (i.e., density or sex ratios) may be driving this sex-specific response.

Across seasons, male home ranges were 1.8 times larger during the wet season than the dry season, which is primarily driven by increases in space use during the rut. Males increase their home range size and movement rates during the rut to learn the distribution and breeding status of females (Beier and McCullough 1990). During the rearing season, male energetic requirements increase as they accumulate body mass and grow antlers in preparation for the rut, however males showed no difference in space use between the fawning, rearing, and post-rut seasons. The rearing season coincides with the start of the vegetative growing season, which may allow males to meet their resource requirements within a smaller area as forage quality increases. These results mirror similar findings in the Everglades National Park, where males had larger home ranges during the wet season than the dry season (Labisky et al. 1995).

Across sites and seasons, female home range areas remained stable, suggesting females are able to meet their space use requirements within the same sized area throughout the year despite seasonal variation in energy requirements and resource availability. Other studies have demonstrated a reduction in space use during fawning and rearing and an increase in space use during gestation (Nelson and Mech 1981, Main et al. 1996, Cherry et al. 2017). During the fawning and rearing seasons, females typically minimize space use to maximize offspring safety, which coincides with the start of the vegetative growing season and a subsequent increase in forage quality. However, our results do not reflect the expected reduction in space use by females. This suggests that the increase in forage quality is not adequate to allow a decrease in space use during fawning and lactation, the peak of energetic requirements for females (Oftedal 1985). Additionally, limited availability of concealment cover for fawns may restrain the ability of females to minimize space use. Fawns require adequate concealment cover and dry conditions for maximum probability of survival (Dion and Haus 2018). Rising water levels during the onset of the wet season may fragment and limit the distribution of fawn concealment cover across the landscape. Space use may remain stable during the fawning and rearing seasons to provide females with access to sufficient concealment cover for their fawns, especially in a predator rich system. Contrary to our prediction, we did not observe an increase in space use by females during gestation (post-rut season). Loveless (1959a) determined that browse in this system has high protein and nutrient content during the winter season and meets the baseline energetic requirements for gestation. The increase in forage quality during this time may allow females to meet the energetic requirements for gestation without the need to increase space use.

Resource Selection

Resource Selection – Introduction

Across their range, variation in resource availability plays a major role in determining when and where deer spend time. At higher latitudes, deer may seek refuge from winter conditions in dense evergreen thickets that provide thermal refugia, relative safety from predators, and meager forage (Messier and Barrette 1985). Conversely, deer in southwestern North America seek shelter from solar radiation during summer months (Ockenfels and Brooks 1994). In South Florida, deer experience seasonal variation in resource availability as a result of widespread seasonal flooding during summer months (Duever et al. 1986). However, drought conditions are not uncommon in winter months resulting in pronounced wet and dry periods that are highly variable in terms of timing, intensity, and duration. Thus, seasonal hydrology affects many aspects of white-tailed deer ecology, including resource selection. Annual variation in hydrology can have profound influences on deer demography and habitat selection (Richter and Labisky 1985, Miller 1993, Labisky and Boulay 1998, Labisky et al. 1999, MacDonald-Beyers and Labisky 2005). For example, Sargent (1992) and Miller (1993) found no effect of standing water on deer movement contrary to the conclusions of Flemming et al. (2005) and Loveless (1959b). However, Sargent (1992) and Miller (1993) collected data in years of relatively low precipitation with mean water levels not exceeding 0.3 m. Conversely, MacDonald-Beyers and Labisky (2005) compared years of relatively high and low standing water and documented significant shifts in habitat selection and a 100% increase in mortality rates of radio-collared deer in 1994-1995, a year of record high water during the dry season (mean water level = 0.73 m in January), as deer sought refuge from flooding on hardwood islands.

Seasonal hydrological fluctuations affect space use at a broad temporal scale, but risk of predation must be continuously assessed by prey species as they make behavioral decisions about when and where to forage. Prey species may attempt to mitigate exposure to risk via behavioral responses such as shifts in vigilance, grouping, or alterations in patch-use. Changes in patch-use often manifest in avoidance of perceived high-risk features or habitats. Mortality of white-tailed deer attributed to direct predation has been well documented in South Florida (Beier and McCullough 1990, Land 1991, Miller 1993, Labisky and Boulay 1998, Beier et al. 2003), but the behavioral effects of predators on this population remain largely unexamined. Evidence suggests the indirect effects of a predator may be predicted by the hunting mode of the predator (Lingle 2001, Lingle and Wilson 2001, Stankowich and Coss 2007). Compared to wide-ranging cursorial predators, stalk and ambush predators such as panthers utilize habitat edges or thick concealment cover in order to ambush prey. As a result, stalk and ambush predators may have greater behavioral effects on their prey because informative habitat cues (e.g., edge) may be associated with risk (Schmitz 2008). Due to the linkage between habitat and predation, such predators can provide more predictable predation cues over time (Preisser et al. 2007), and prey can alter habitat use as a function of predation (Sih 1992). Given local predator community composition and circannual variably in surface water levels, there is potential for an interactive effect of hydrology and predation risk on resource selection.

In addition to hydrology and predation pressure, human disturbance has also been shown to affect white-tailed deer behavior (Geist 1971). Such effects may indirectly influence population dynamics of the Florida panther and have important implications for prey and predator management. Hunting is a traditional deer management tool and while direct effects to deer populations are well understood (McCullough 1982, Nelson and Mech 1986a), relatively less is known regarding the effects on deer behavior. White-tailed deer may respond to human disturbance by increasing use of refuge habitat (Kammermeyer and Marchinton 1976, Pilcher and Wampler 1981), avoiding areas of high human use (Dorrance et al. 1975, Rost and Bailey 1979), and altering movement (Marshall and Whittington 1968), activity patterns (Autry 1967, Vogel 1989), and habitat selection (Swenson 1982, Kufeld et al. 1988).

We examined sex-specific resource selection of white-tailed deer in the Big Cypress Basin in the context of hydrology, predation, and human disturbance. Overall, in accordance with the reproductive-strategy hypothesis (Main et al. 1996), we hypothesized that male and female resource selection would differ due to differing reproductive requirements. In regards to hydrology, we expected a seasonal effect of water level on resource selection because greater water depth may represent mesic sites with palatable forage during the dry season and inundated, energetically expensive sites during the wet season. We also expected seasonal variation in resource selection between the sexes because the hydrological season covaries with deer reproductive chronology. Last, we predicted that use of areas associated with high risk of predation would be positively correlated with surface water levels as increased water depth decreases availability of other resources.

In regards to predation and human disturbance, we tested the hypothesis that deer would avoid both spatial features associated with relatively high risk of predation (i.e., ‘risky places hypothesis') and areas with high panther and human activity rates. We considered forest edge and off-road vehicle (ORV) trails to be associated with relatively high risk of predation by panthers as forest edges are associated with increased occurrence of panthers (Frakes et al. 2015) and ORV trails are preferred corridors of panthers (Crawford et al. 2019). We also considered ORV and hiking trails to be associated with high risk of deer-human interactions as such areas provide most recreational opportunities for humans. We hypothesized that selection for high-risk features would increase during the dry season and wet season for females and males, respectively, as a result of differential requirements for reproductive success. Lastly, we expected avoidance of areas with greater panther activity as well as a seasonal effect of panther activity on resource selection because greater water depth will likely restrict available habitat, resulting in an increase in selecting riskier, but drier areas. We hypothesized that selection for high-risk features would decrease as a function of panther activity rates for both females and males. In regards to human disturbance, we tested the hypothesis that deer would avoid spatial features associated with relatively high risk of deer-human interactions.

Resource Selection – Methods

We conducted three sets of resource selection analyses to address our hypotheses. We used a distance-based approach including five cover types (hardwood hammock, marsh, pine flatwoods, prairie, and swamp), two linear features associated with predation risk and humans (roads and edges), SWI, and panther and human activity rates. We used step-selection functions (SSFs) for all sets of resource selection analyses to evaluate the effects of cover type, hydrology, and panther and human activity rates on deer habitat selection. Step-selection functions efficiently deal with the inherent difficulty of defining availability associated with resource selection functions and resource selection probability functions by employing conditional logistic regression in a case-control design at the step level (Fortin et al. 2005). Animal trajectories may be characterized by attributes associated with sequential steps, or lines between sequential relocations, such as step length and turn angle (Turchin 1998). Step length is the linear distance between sequential relocations. Turn angles are measured as the angular change in bearing relative to the bearing of the previous step resulting in values ranging from $-\pi$ to π. By sampling step attributes from observed distributions and constructing “available” steps from those samples, availability is defined by the underlying movement process.

We generated 15 random steps per observed step. This was accomplished by 1) sampling 15 step lengths from our gamma-distributed observed step lengths; 2) sampling 15 turn angles from a uniform distribution; and 3) calculating the resulting endpoints relative to the origin of the observed step. Conditional logistic regression requires that the case (observed step) be compared to a number of controls (random steps), thus all steps originating from the same location, observed or random, were assigned a common “step identifier”, which was included as the stratum in SSF models. Because any single animal’s relocation data are inherently autocorrelated in space and time, we used the unique deer identification number as a cluster variable to account for individual variation in movement patterns that can otherwise bias population-level selection coefficients (Fortin et al. 2005, Prima et al. 2017).

For the first modeling approach, we examined sex-specific seasonal variation and included interactions between SWI and cover types and linear features to examine how hydrology influences resource selection. We quantified resource selection by fitting four sets of SSF models representing males and females during the wet and dry seasons (78 males, dry season; 70 males, wet season; 143 females, dry season; 120 females, wet season). For the remaining two modeling approaches, we evaluated seasonal variation in the effect of panther and human activity rates on deer resource selection by fitting eight sets of SSF models representing males and females during the wet and dry seasons. To create daily activity rates of panther and human, we summed respective daily detections of each species at each camera. We controlled for camera effort by excluding non-operational cameras for all days they were not functioning. We excluded detections less than 30 min apart to maintain independence among detections. To relate camera detections of panther and human to location data of deer, we only included deer that had seasonal home ranges within a 700 m buffer of the camera grid (38 males, dry season; 35 males, wet season; 70 females, dry season; 62 females, wet season).

In order to relate panther and human activity rates to deer GPS location data, we created daily continuous surfaces human and panther activity using inverse distance weighting (IDW) interpolation in the package *gstat* (Pebesma et al. 2019) in Program R (R Core Team 2019). IDW uses a weighted average of activity patterns from nearby camera locations to predict the activity patterns for each camera location within a grid comprised of user-specified areas (de Smith et al. 2015). Our user-specified areas of inference were 500 m^2 because it most closely matched the distance between cameras (i.e., 700 m and 250 m, on-trail and off-trail respectively). This interpolation process provided spatial explicit estimates human and panther activity at the daily scale.

Remotely sensed spatial data were extracted to the endpoints of all observed and random steps and included as covariates in our SSFs. For subset one, we extracted Euclidian distance to each cover type (hardwood hammock, marsh, pine flatwoods, prairie, and swamp), Euclidean distance to linear features (roads and edges), and SWI. We then created a global model that included all cover types, linear features, and SWI to identify important explanatory variables. For subset two, we extracted panther activity rates, Euclidian distance to each cover type, and SWI to each observed and random step endpoint. Next, we created a global model including all cover types, linear features, and panther activity rates to identify important explanatory variables. For subset three, we included all variables in subset two, excluding panther activity rates, and including human activity rates. Last, we created a global model that incorporated all cover types, linear features, SWI, and human activity rates to identify important explanatory variables.

For subset one, we parsed the data by sex and hydrological season (wet season: May-October; dry season: November-April), developed four sets of candidate models using various combinations of our covariates, and used corrected Akaike's Information Criterion (AICc) to identify a top model from the set of candidate models (Burnham and Anderson 2002) for each sex-season subset (Table 9). For subset two, we used the same modeling approach, but included candidate models with panther activity rates and their interaction with land cover, linear features, and SWI. For subset three, we used the same modeling approach as subset one and two but included models with human activity rates and their interaction with all other land cover types, linear features, and SWI. We used the same model selection procedure as subset one to identify the top model from eight sets of candidate models for each sex-season combination of subset one and two, respectively (Tables 10, 11). All analyses were performed in program R (Bates et al. 2014, R Core Team 2019).

Resource Selection – Results and Discussion

We used 579,682 GPS locations collected during 01 January 2015 – 31 December 2018 for resource selection analyses. For the sex-season subset analysis, model selection indicated that our global model was the top model for all four subsets (Table 9). We observed variation in selection across both sex and season. Male deer selected for marshes during both seasons, and selection for marshes was stronger during the dry season (Fig. 14). We observed significant interactive effects of SWI and cover type which differentially affected seasonal resource selection of forest edges, roads, flatwoods, prairies, and swamps. Increasing SWI decreased distance to forest edges and flatwoods during the dry season for males, but decreased distance to prairie during the wet season. As SWI increased, males were nearer to roads and further from swamps than would be expected at random during both seasons.

Females exhibited greater seasonal variation in selection than males (Tables 12, 13). During both seasons, females selected prairies with no effect of SWI on prairie selection (Fig. 15). Similarly, we observed no effect of SWI on selection of pine flatwoods, which females selected for during the wet season. SWI significantly affected female selection of marshes, swamps, hammocks, and our proxies for predation risk and human disturbance, forest edges and roads. There was no seasonal variation in selection of marshes and swamps, which were avoided by females as SWI increased. Increasing SWI increased distance to hardwood hammocks, but only during the wet season. Female deer also selected areas nearer to roads and edges as SWI increased during both seasons.

For the eight sex-season specific analyses including panther and human activity rates, model selection indicated that the global model was the top model for six of the sex-season specific analyses, while the top model for males during the dry season excluded the interaction between prairie and panther activity and human activity, respectively (Tables 10, 11). Panther activity had minimal effects on selection for both sexes. During the dry season, areas with increased panther activity rates were marginally avoided by males (Table 14). For females, panther activity rates were not significant in regards to habitat selection during the dry season (Table 15). In the wet season, males avoided prairies with increasing panther activity rates (Table 14). During the wet season females and males selected for higher water levels with increasing panther activity rates, suggesting deer use flooded cover types when panthers are active during the wet season (Table 15).

Human activity rates (hereafter human disturbance) had stronger effects on resource selection of females compared to males in the wet season. Increasing human disturbance had no effect on male habitat selection in either season (Table 16) or female habitat selection in the dry season (Table 17). During the wet season, increasing human disturbance caused avoidance of roads and pine flatwoods, and increased selection for prairie for females (Table 17).

As we hypothesized, our results indicated a pervasive effect of surface water levels on resource selection which varied with respect to sex and hydrological season. The dry season includes biological seasons such as fawning and fawn rearing and antler casting and bachelor group formation for females and males, respectively. These biological seasons represent periods when males attempt to recuperate body mass lost during the breeding season and female energetic demand increases throughout gestation. The similarity in energetic demand is reflected by the similarity in resource selection across sexes during the dry season when only hardwood hammocks and pine flatwoods were differentially selected or avoided by the sexes. Both sexes selected for marshes during the dry season; however, only female selection of marshes was influenced by SWI. The sex-specific difference in the effect of SWI on the selection of marshes may be associated with female avoidance of relatively high water during the fawn rearing season when fawns are not yet large enough to traverse flooded areas. Conversely, females showed no selection for flatwoods during the dry season when increasing water levels increased male selection for that cover type. Pine flatwoods are typically associated with a dense understory, which may serve as concealment cover for predators. According to the predation risk hypothesis for sexual segregation in ungulates, male thresholds for risk exposure should be greater than those of females due to size, especially at a time when neonates are present (Main et al. 1996). Thus, males may be willing to utilize areas perceived to be high-risk by females as a function of risk thresholds and relative importance of the season to sex-specific reproductive success.

We observed greater variation in selection between sexes during the wet season, which included the pre-rut and rut, biological seasons of marked variation in behavior. For example, selection of pine flatwoods and the effect of SWI on selection of hardwood hammocks, marshes, and prairies varied by sex during the wet season. The wet season coincides with an increase in male testosterone levels, which ultimately induces hyperphagy and mate-searching behaviors, and generally increases male thresholds for exposure to predation risk. Conversely, females are relatively free from physiological stressors associated with reproduction during this period due to cessation of lactation and relative independence of young of the year. This reduction in energetic demands and for offspring survival may explain female selection for flatwoods during the wet season. An alternative explanation for the seasonal differences in flatwoods selection by females may be that mean surface water levels were higher during that season, which would also explain the lack of an effect of SWI on flatwoods selection by either sex. Contrary to previous findings in the Florida Everglades, females in the Big Cypress Basin avoided hardwood hammocks as SWI increased. Miller (1993) and MacDonald-Beyers and Labisky (2005) documented increases in selection for hardwood hammocks during a period of prolonged inundation. The discrepancy in results may be attributed to higher availability of alternative upland habitat (i.e., flatwoods) in our study area. The differential in energetic demand for the sexes during this season and its importance to reproductive success of males may explain observed variation in selection between the sexes during the wet season. This sex- and season-specific difference in selection supports our hypothesis that requirements for reproductive success result in sexual variation in selection.

Significant effects of SWI on selection of edges and roads support our hypothesis that increasing water levels force deer nearer to edges and roads than expected at random because open cover types in the Big Cypress Basin (i.e., marshes and prairies) are characterized by slightly lower elevation than adjacent pine flatwoods and hardwood hammocks. As a result of the elevational gradient, open cover types were prone to inundation. Similarly, roads were typically elevated. Both sexes selected for areas nearer to roads as SWI increased during both seasons, but increasing SWI affected only female selection of edges during the wet season, which was likely a result of erratic male movement behavior during the rut. As SWI increased and availability of open cover types decreased, deer may have sought refuge from standing water near roads and forest edges.

We included distance to forest edges and roads as proxies for predation risk by panthers and human disturbance in our SSFs, as well as panther activity rates and human disturbance directly. Onorato et al. (2011) demonstrated strong selection for wetland forests (e.g., cypress, cypress/pine/cabbage palm, mixed wetland forest, and hardwood swamp) by panthers using Euclidean distance analyses and GPS-telemetry locations from 20 adult panthers. Additionally, prey species may associate habitat features, such as forest edges, with ambush predators and alter behavior in proximity to such features as Altendorf et al. (2001) demonstrated in a deer-puma system (Preisser et al. 2007). Distance to ORV trails was also included as covariate representing high-risk areas because concurrently collected camera trap data indicated roads as areas of greater panther activity (Crawford et al. 2019).

Our hypothesis regarding sex-specific responses to predation risk was not supported. We observed no avoidance of areas with high panther activity rates or avoidance to edges or roads with increased panther activity rates during either season. However, we did find that increasing panther activity rates caused an increase in selection for deeper water in the wet season for both sexes. We also found that in areas with greater panther activity, males selected prairie more than expected at random in the wet season. This pattern may be partially explained by selection for areas of higher water levels as marshes and prairies are characterized by slightly lower elevation relative to the surrounding cover types, and typically experience greater flooding. Selection of flooded areas in the presence of panthers may indicate that marginally flooded areas can serve as a refuge for deer, however the negative energetic and nutritional costs of using flooded areas deserves more attention.

Our hypothesis that human disturbance effects selection of spatial features associated with relatively high risk of predation was partially supported. In the wet season, increasing human disturbance caused females to avoid roads, but had no effect on the selection of edge in any season. Male habitat selection was not affected by increasing human disturbance in either season. Females avoided roads and pine flatwoods and selected for prairies with increasing human disturbance in the wet season. Thus, our hypothesis regarding differences among sexes was supported. This finding may be driven by predation risk hypothesis for sexual segregation in ungulates; specifically, male tolerance for risk exposure is predicted to be greater than those of females due to body size (Main et al. 1996). This difference in tolerance of risk by males should be greater in the wet season due to increased testosterone levels associated with breeding.

Last, it should be noted that female selection was not affected by humans in the dry season, which may be attributed to relatively high energetic demands associated with gestation and lactation. The risk of incurring a nutritional deficit may outweigh encountering a human for females. Alternatively, the implications of human interactions may vary by season since deer hunting is restricted to the wet season.

Others have reported female deer avoidance of roads during the hunting season. Kilgo et al. (1998) demonstrated that deer in Osceola National Forest, FL, preferred forested areas and avoided roads during the hunting season and posited that the avoidance behavior of roads and the selection of forest habitat by deer could increase panther hunting success by increasing prey concentrations in areas preferred by panthers. While females in our study avoided roads, they also avoided pine flatwoods and selected open prairies. Females likely selected such areas to maintain open sight lines and avoid predator concealment cover associated with pine flatwoods, which were reportedly selected by panthers in South Florida (Maehr et al. 1990b). More recent work suggests that upland forest is not as critical for panthers (Cox et al. 2006, Kautz et al. 2006, Benson et al. 2008), and Onorato et al. (2011) observed strong selection for prairies at night. However, the effect of human disturbance on selection for prairie was only observed during the wet season, and hydrology is one of the most important factors determining the presence of adult panthers (Frakes et al. 2015). Thus, when water levels are higher in the wet season, panthers likely avoid flooded areas (e.g., prairies and marshes). While the effects of deer resource selection on panther hunting efficacy is beyond the scope of this study, avoidance of roads and selection for prairies by deer may decrease deer-panther encounters. Human disturbance has the potential to decrease panther hunting success by displacing female deer away from preferred panther habitat (e.g., pine flatwoods; Benson et al. 2008, Cox et al. 2006, Kautz et al. 2006, Frakes et al. 2015) and areas heavily travelled by panther (e.g., ORV trails; Crawford et al. 2019), and increasing concentrations of deer in open areas of deeper water.

In summary, changes in surface water levels significantly affected habitat selection of both sexes, but the magnitude and direction of the effects varied across sexes and seasonally within each sex. The differences in selection are likely driven by sex-season dependent energy budgets and predation risk thresholds associated with optimizing reproductive success. For example, the dry season includes biological seasons that are pivotal to female reproductive success (i.e. fawning, fawn-rearing), and our results suggest that females avoid flood-prone habitat types during this period when fawns may be present. However, males selected for marshes more strongly during the wet season, which coincides with the breeding season, or rut. In South Florida, the rut occurs in mid- to late-August and is characterized by the greatest male activity rates (Crawford et al. 2019). Thus, male selection for marshes during the wet season may serve as a thermoregulatory mechanism enabling males to maintain high activity rates during the hottest part of the year as they seek mating opportunities. We found panther activity rates only impacted habitat selection in the wet season, when both sexes were shown to select for areas of deeper surface water. This pattern suggests deer may use flooded habitats as a refuge from predators. Last, human activity only altered female habitat selection in the wet season, which may decrease hunting success for panther in the wet season.

Movement Behavior

Movement Behavior – Introduction

Animal movement influences individual fitness (Andreassen and Ims 1998, Doherty and Driscoll 2017, Hooten et al. 2018). In systems devoid of predators, prey species may make movement decisions based solely on forage availability and patch quality. However, when moving through a landscape of risk with predators, prey must assess spatial variation in risk and optimize movement to maximize energetic intake while minimizing their probability of being eaten (Brown et al. 1999). This is one of the mechanisms by which predators can shape the spatial distribution of prey (Laundré et al. 2010). Disturbances, both environmental and anthropogenic, can interact with spatial variation in predation risk to affect exposure of prey to risk (Tucker et al. 2018).

Stochastic environmental conditions can influence animal movement and space use by altering the availability of resources, such as forage or refugia, or by impeding locomotion. In either case, prey species may subsequently experience elevated risk of predation. When prey mobility is reduced as a result of adverse environmental conditions, movement comes at a greater energetic cost (Shepard et al. 2013). Further, reduced mobility can reduce the probability of escaping an encounter with a predator that is better adapted to the conditions. For example, deer movement rates decrease with increasing snow depth, while risk of predation by wolves increases (Nelson and Mech 1986b, Gilbert et al. 2017). Conversely, adverse environmental conditions may offer prey refugia as suggested in the Resource Selection - Results and Discussion section.

Additionally, anthropogenic features, such as roads and trails, have the potential to influence animal movement and impact individual fitness. For example, Prokopenko et al. (2017) reported increases in movement rates and use of coniferous cover by elk (*Cervus canadensis*) as distance to roads decreased. Further, elk responded to roads just as they responded to natural predation risk suggesting that they perceive anthropogenic corridors as high-risk features on the landscape. Intensity, frequency, and duration of human disturbance may directly impact prey species movement decisions (Leblond et al. 2013). Alternatively, anthropogenic landscape features may indirectly influence prey movement by providing efficient movement corridors for predators. For example, James and Stuart-Smith (2000) observed avoidance of linear corridors by caribou, selection for such corridors by wolves, and reported that caribou mortality events attributed to wolf predation occurred closer to corridors than expected at random. BCNP is open to the public and hosts thousands of visitors annually for various activities including ORV trail riding, biking, hiking, and hunting. While the majority of FPNWR is not accessible to the public year-round, extensive habitat management efforts including prescribed fires, invasive plant treatments, and data collection result in frequent human disturbances. Thus, determining how human activity influences deer movement is requisite to understanding how human disturbance may affect exposure of deer to predation risk by panthers.

We evaluated sex- and season-specific movement behaviors of white-tailed deer in South Florida relative to hydrology, human disturbance, and risk of predation by Florida panthers using integrated step selection analyses (iSSAs). We hypothesized that hydrology would differentially affect movement of male and female deer both within and across seasons as a result of differing requirements for reproductive success. Additionally, we hypothesized that surface water levels would induce habitat-specific shifts in movement resulting in increased exposure to predation risk as certain habitats were predisposed to inundation. Lastly, we hypothesized that our proxy for human disturbance, distance to roads, would have limited effect on deer movement, given restricted access and relatively low hunter densities result in negligible annual deer harvest rates in the study area.

Movement Behavior – Methods

We utilized integrated step selection analyses (iSSAs) to evaluate deer movement (Avgar et al. 2016). Integrated SSAs are extensions of step selection functions (SSFs) wherein step lengths and turn angles are included as covariates in an SSF. When these step attributes are included as interactive effects with habitat type or landscape features, the resulting parameter estimates allow for inference on the effects of landscape attributes on movement behavior. Therefore, we integrated step lengths and turn angles into the top SSF model associated with subset one discussed in the Resource Selection – Methods to make inferences regarding the seasonal effects of hydrology, human disturbance, and predation risk on deer movement in South Florida.

Movement Behavior – Results and Discussion

We observed sex- and season-specific effects of spatial covariates on step length and turn angle indicating habitat- and feature-specific differences in movement behavior of white-tailed deer in South Florida (Tables 18, 19, 20, 21, 22). Step length was positively correlated with surface water levels across all sex-seasons (Figs. 16a, 17a). Increasing SWI decreased turning angles of both sexes during the wet season, but had no effect during the dry season (Figs. 16b, 17b). This suggests that, during the wet season, both sexes travel less linear paths through areas of high water than they do during the dry season. However, as suggested in the Resource Selection - Results and Discussion section, increasing tortuosity of movement trajectories in areas of high water may indicate use of inundated patches as refugia from predators. Seasonal variation in the effect of SWI on turn angles, but not step length may be explained by the magnitude and duration of wet season flooding and the subsequent necessity of animals to circumnavigate areas of relatively high water. Alternatively, the inverse relationship between forage availability and SWI may force animals to travel farther in an effort to meet energetic demands. No effect of SWI on turn angle during the dry season may be a function of low landscape resistance relative to the wet season, which is characterized by prolonged inundation.

The sexes exhibited similar movement behaviors in relation to marshes. Step lengths of both sexes increased significantly near marshes during both wet and dry seasons. However, distance to marsh only affected their turn angles during the wet season when both sexes traveled more trajectories near marshes (Figs. 16, 17). As a cover type predisposed to inundation, deer appear to move more quickly through marshes in both seasons. However, the seasonal difference in the effect of marsh on turn angles suggests that both sexes travel more linear paths near marshes during the wet season when the magnitude and duration of flooding are generally greater. During the dry season, typical mean surface water levels are unlikely to impede travel, and moist soils associated with marshes provide the highest quality available forage at that time. Forage quality in marshes may explain greater turn angles, or tortuosity, for trajectories of both sexes during the dry season. More linear paths near marshes during the wet season are likely related to decreased efficiency of locomotion, reduced forage availability, increased risk of predation by alligators, or a combination of those factors as a result of relatively high mean surface water levels and prolonged inundation.

Distance to swamps had minimal effects on movement as male step length decreased near swamps during the wet season, but no other sex-season combination exhibited an effect of swamp on step length or turn angle. Pine flatwoods had no effect on step lengths of either sex; however, we observed seasonal differences in turn angles for both sexes in relation to pine flatwoods. For males, turn angles increased near pine flatwoods during the dry season when female turn angles were unaffected. Conversely, female turn angles increased near pine flatwoods during the wet season when male turn angles were unaffected. Therefore, males moved more tortuously through pine flatwoods during the dry season while females had more tortuous trajectories in pine flatwoods during the wet season. Relatively high tortuosity in a given cover type implies relatively intensive use of that cover type while linear trajectories suggest traveling through a patch. Interestingly, the sexes exhibited similar movement behaviors with respect to pine flatwoods, but during different seasons. These results concur with our SSF results (Figs. 14, 15), which suggest selection for pine flatwoods by females and interactive effects of SWI and distance to pine flatwoods during the wet and dry seasons, respectively. Our iSSA results reinforce these findings and suggest that females may more intensively utilize pine flatwoods only during the wet season when their fawns are self-sufficient and relatively high surface water levels reduce the availability of more preferred habitats. Given the typical density of the pine flatwoods understory, male movement behaviors associated with flatwoods during the dry season may be explained by their relatively high threshold of predation risk and lack of obligation to the safety offspring.

To evaluate the effect of human disturbance on deer movement in the context of predation risk, we included distance to forest edges as a proxy for predation risk and distance to road as a metric of both human disturbance and predation risk in our iSSAs. Distance to roads did not affect turn angles of any sex-season combination. However, we observed variation in the effect of distance to roads on step length across sexes and seasons. Male and female step lengths increased near roads during the dry and wet seasons, respectively. Contrary to our hypothesis, this suggests that the sexes move more quickly in proximity to roads than would be expected at random. Prokopenko et al. (2017) reported a human “disturbance effect” of roads which manifested in faster movement of elk in proximity to roads. Further, they reported that elk responded to roads just as they responded to natural predation risk in the context of movement behavior. Our results indicate that distance to forest edges affected female step length during the wet and dry seasons, but had no effect on female turn angles or any male movement behavior. Interestingly, female step length decreased near edges during both seasons, which suggests that females move more slowly in close proximity to forest edges. While this finding in contrary to movement behaviors of elk in high risk areas reported by Prokopenko et al. (2017), the difference in behavior may be attributed to predator hunting mode. Prokopenko et al. (2017) analyzed data collected on elk in their winter range where wolves likely posed the greatest risk of predation, and in the presence of a cursorial predator, increased movement through an area of high risk may minimize the probability of encountering a predator. However, in the presence of a stalking ambush predator and absence of cursorial predators, increased movement rates through risky areas may increase detectability of prey by predators. If our observed effect of roads on deer movement is a “disturbance effect” and female behavior in relation to forest edges is an effect of predation risk, the observed increase in movement rates in proximity to roads suggests that human disturbance may influence deer movement behavior such that they are at a greater risk of predation by panthers.

Activity Patterns and Predation Risk

Activity Patterns and Predation Risk – Introduction

We examined the effects of variation in panther predation risk across space and at multiple temporal scales on sex-specific activity patterns of white-tailed deer. We tested the hypothesis that predation risk induces sex-specific differences in spatiotemporal activity patterns as determined by spatial (‘risky places hypothesis’) and temporal (‘risky times hypothesis’) variation in panther activity. High-risk scenarios were characterized by relatively high panther activity. We hypothesized that risk exposure would increase with the relative reproductive importance of each biological season to each sex. Under this hypothesis, females investing in lactation during the fawn-rearing season should increase their predisposition to risk relative to other, less demanding seasons (Oftedal 1985). Because their reproductive success is positively correlated with body mass (Townsend and Bailey 1981, DeYoung et al. 2006), males should be more risk prone than females across all biological seasons, particularly during the breeding season.

Activity Patterns and Predation Risk – Methods

We analyzed panther and adult deer detections recorded at all camera trap locations from 01 February 2015 - 31 October 2015. Because deer activity is closely linked to reproductive stage, we organized our study in the context of biological seasons of deer in the Big Cypress Basin (Richter and Labisky 1985). Camera trap data indicated a broad window of fawning across most of February and March (Chandler et al. 2018), thus we designated these months as the fawning season. This timescale (February-March) was chosen to appropriately characterize behaviors leading up to fawning, such as fawning site selection, while including the time period when the majority of fawns were born. Because most fawns on our study site were born by the end of March and nearly all males had initiated antler growth by this point, we designated April-June 2015 as the fawn-rearing and antler growth season (hereafter, rearing). Rearing is an energetically expensive time for reproductive females as fawns grow and lactation peaks (Clutton-Brock 1982, Oftedal 1985, Pekins et al. 1998). Similarly, males invest in antler development and body growth during this period because antler size and body mass are positively correlated with dominance and reproductive opportunity (Townsend and Bailey 1981). To evaluate seasonal variation in activity at a fine scale, we included an additional biological season in this analysis. Thus, we designated July as the pre-rut when males exhibit hyperphagy, increased activity, and increased antler sparring in preparation for conspecific competition. Given the relatively broad fawning window, some breeding occurred through August and September, however peak breeding behavior occurs in mid- to late-August. Thus, we designated August as the rut. This is a stressful time for males as they forage minimally and maximize mate searching behaviors. Following the rut, males enter a recuperation phase known as the post-rut (September-October). During this biological season, bred females are in the earliest stages of gestation. While the third trimester witnesses a peak in energetic demand for females, this period is partially included in the early fawning season. The majority of the period between the post-rut season and fawning seasons, or gestation, requires relatively low reproductive energetic investment. Thus, we did not consider gestation in analyses.

To maximize independence of detections, we sorted records chronologically by camera, and omitted records with the same sex, age, and species class as the previous record from analysis if the time from the previous record was <6 min. We determined this threshold by filtering the data at 1 min intervals and visually inspecting the mean difference in time between images at each thinning interval. The resulting curve indicated a rapid decrease in rate of change in the mean interval when images separated by 5 min or less were omitted. This procedure improved independence of detections by removing sequential images of lingering individuals. We then classified detections based on biological seasons and characterized each as either diurnal (between sunrise and sunset) or nocturnal (between sunset and sunrise). We used package maptools (Bivand and Lewin-Koh 2015) in Program R (R Core Team 2019) to determine daily sunrise and sunset times associated with the coordinates of the centroid of our study area.

We estimated predation risk by modeling panther activity patterns to predict when and where a deer is likely to encounter a panther. We analyzed count data of male and female deer and panthers at each camera using Poisson generalized linear mixed models (GLMM) with a log link. The response variable (y_{ijk}) was the number of detections at each camera (i = 1,...,180), during each time period (j = 1,2; for diurnal and nocturnal) and biological season (k = 1,...,5; for fawning, rearing, pre-rut, rut, and post-rut). Explanatory variables included trail (i.e. on- and off-trail), time, and biological season. We fit GLMMs for each sex of deer and a single model for both sexes of panther. We constructed four candidate models representative of specific hypotheses, used AIC model selection to identify the best candidate model for panthers and for each sex of deer (Table 23), and used the most parsimonious model of each for inference. Candidate models included various combinations of the main effects of trail, time, and biological season as well as two-way interactions of each with the others. We hypothesized that time and biological season would interact such that deer detection rates would be greater at high risk times during biological seasons of reproductive importance. Similarly, we hypothesized that trail and season would interact such that deer detection rates would be greater in high risk places during biological seasons of reproductive importance. The number of camera hours varied among scenarios due to variable season and day length (e.g., nocturnal on-trail during the fawning season) and among cameras due to camera failure, which we accounted for by using log (camera hours) as an offset in the GLMMs. As a result, the estimates can be interpreted as the number of detections per hour. Variation among cameras was modeled using camera-specific random effects. Due to difficulty of deriving standard errors from linear models including random effects, we calculated 95% confidence intervals (CI) for detection rates via parametric bootstrapping. Detection rates of male and female deer and panthers were deemed significantly different when CIs for season-specific and sex/species-specific detections rates did not include the mean rate of another season or sex/species. We conducted detection rate analyses in program R using package lme4 (Bates et al. 2015).

To test for differences in activity overlap of deer with panthers, we calculated the coefficient of overlap in activity patterns of male and female deer with panthers using non-parametric kernel density estimation of image timestamps (Ridout and Linkie 2009). We employed nonparametric bootstrapping to calculate confidence intervals for estimates of activity overlap. Sex-specific deer-panther activity overlap was estimated for every combination of trail (i.e. on, off) and biological season (i.e. fawning, rearing, pre-rut, rut, and post-rut). We identified significant differences in activity overlap using CIs in the same manner as described for detection rates. We conducted activity pattern overlap analyses in program R using package *overlap* (Ridout and Linkie 2009, R Core Team 2019).

Activity Patterns and Predation Risk – Results and Discussion

We recorded 1,058 independent detections of panthers, 1,799 independent detections of adult (i.e. ≥1 year of age) male deer, and 2,624 detections of adult female (i.e. ≥1 year of age) deer from February-October 2015. At the diel timescale, only 28% (*n* = 296) of panther detections occurred during diurnal periods. Spatially, 91% (*n* = 966) of panther detections occurred at on-trail traps. Sixty-five percent (*n* = 1,177) of male deer detections were diurnal and 65% (*n* = 1,175) occurred at on-trail traps. Seventy-one percent (*n* = 1,862) of female deer detections occurred during diurnal hours, while 60% (*n* = 1,565) of adult female deer detections occurred at on-trail traps. However, only 11% (*n* = 279) of female deer detections occurred on-trail during nocturnal hours.

The most supported model for panthers and both sexes of deer included trail *x* time, trail *x* season, and season *x* time interactions (Table 23). We observed an interactive effect of trail and time on the rate of detection of panthers. This interaction is evident in an 875% increase in detection rates from diurnal off-trail traps during the rut (0.24, 95% CI: 0.15-0.36; detections/1,000 h) to nocturnal on-trail traps in the fawning season (1.02, 95% CI: 0.80-1.24). The detection rate of panthers was greater on-trail than off-trail during both day and night across all seasons with the highest detection rates observed on-trail at night during the fawning season (1.03, 95% CI: 0.82-1.28) and on-trail at night during the rut (1.02, 95% CI: 0.80-1.24). The lowest panther detection rates occurred off-trail during diurnal hours of the pre-rut (0.23, 95%

CI: 0.13-0.34) and rut seasons (0.24, 95% CI: 0.15-0.36). The difference between diurnal and nocturnal detection rates of panthers clearly identified nocturnal hours as periods of higher predation risk to deer while the discrepancy between detections at on- and off-trail traps suggested relatively high risk of predation for deer in proximity to trails (Fig. 18). We also observed a season by time interactive effect on detection rates of panthers with diurnal and nocturnal detection rates being highest during the fawning season at both on-trail and off-trail traps. Thus, we considered deer activity in the context of spatially and temporally variable risk of predation and classified diurnal and nocturnal periods as low- and high-risk times, respectively. We considered on- and off-trail locations as areas presenting respective high and low risk. Therefore, diurnal, off-trail deer activity imposed the least risk and nocturnal, on-trail deer activity imposed the greatest risk.

For male deer, we observed significant interactive effects of trail and time as well as time and season on detection rates (Fig. 19). In high-risk areas at low-risk times, detection rates (detections/1,000 h) of males were lowest during the fawning season (0.98, 95% CI: 0.77-1.20) and peaked during the rut (3.31, 95% CI: 2.80-3.90). Detection rates were greater at low-risk times than high-risk times across all seasons. In high-risk areas at high-risk times, male activity was lowest during the fawning season (0.23, 95% CI: 0.15-0.31) and increased each season through the rut (2.09, 95% CI: 1.60-2.54), then decreased during the post-rut (0.93, 95% CI: 0.75-1.16). In low-risk areas at low-risk times, activity of males was lowest during fawning (0.90, 95% CI: 0.64-1.20), and peaked during pre-rut (2.35, 95% CI: 1.81-2.90) and rut (2.12, 95% CI: 1.64-2.76). In low-risk areas, male activity during low-risk times was greater than during high-risk times during fawning and rearing, but there was no difference during any other season. The observed increase in high-risk activity by males supports our hypothesis that deer would expose themselves to the greatest risk during periods of reproductive importance.

We observed interactive effects of trail and time, as well as season and time on detection rates of female deer (Fig. 20). Detection rates of females were greater at low-risk times across all seasons regardless of location. The greatest female detections rates occurred in high risk areas at low-risk times during the rearing (2.79, 95% CI: 2.38-3.22), pre-rut (3.23, 95% CI: 2.64-3.79), and rut (3.98, 95% CI: 3.41-4.65) seasons. However, detection rates of females at high-risk times were greater in low-risk areas through all seasons.

We observed significant effects of trail and season on the coefficient of overlap of males and females with panthers (Fig. 21). In low-risk, off-trail areas, the sexes only differed in overlap with panthers during the fawning season when female-panther overlap was greater. However, the sexes differed in overlap with panthers during all seasons in high-risk, on-trail areas where females overlapped with panthers more during the fawning season, and male-panther overlap was greater during the rearing, pre-rut, rut, and post-rut seasons. We also observed seasonal differences in overlap within the sexes. In low-risk areas, activity overlap was greater during fawning and rearing than pre-rut, rut, and post rut for both sexes, and female-panther overlap was lower during the rut than any other season. In high-risk, on-trail areas, female-panther overlap was greatest during the fawning season, while male-panther overlap was greatest during the rut. Within the sexes, we also observed effects of spatial variation in risk of predation on deer-panther overlap; female-panther overlap was lower in high-risk areas than low-risk areas during fawning and rearing, while male-panther overlap in high-risk areas was lowest during rearing and greatest during rut. While detection rates indicate no increased female risk exposure during fawning and rearing, our activity overlap results supported the hypothesis that differing requirements for reproductive success explains the behavioral differences between the sexes.

Our results provide strong evidence that risk of predation by panthers induces white-tailed deer activity patterns that are substantially different from activity patterns in other parts of their range where panthers do not occur. Activity patterns of deer vary based on geographical, physiological, and environmental factors. However, peaks in activity during crepuscular hours are ubiquitous across the species' range (Kammermeyer and Marchinton 1977, Beier and McCullough 1990). Increases in nocturnal activity of deer exposed to human hunting pressure are also well documented (Kilgo et al. 1998, Kilpatrick and Lima 1999, Webb et al. 2010, Little et al. 2015). Our results suggest that both sexes of deer displayed preference for diurnal activity and that males engaged in riskier, nocturnal activity more than females, which may be attributed to their inability to forgo activity during periods of high risk while meeting energetic requirements for maintaining reproductively competitive body mass. Conversely, female detection rates suggest a strong aversion to nocturnal activity.

Predation risk has been suggested as a driver of behavioral variation among male and female deer (Ruckstuhl and Neuhaus 2000) as sex-specific energetic demands associated with reproductive success require trade-offs between safety and energetic intake (Main et al. 1996, Ruckstuhl and Neuhaus 2000). We observed increased exposure to high-risk scenarios for male deer during seasons of high reproductive importance. However, females did not appear to increase risk exposure as predicted during the fawning and rearing seasons relative to other seasons.

Our results suggest that deer in South Florida perceive ORV trails as high-risk areas and reserve activity in those areas for low-risk times to minimize probability of encounters with panthers, which we detected disproportionately on-trail at night. Relatively high detection rates of panthers at on-trail traps suggests that ORV trails may facilitate the efficient movement of panthers across the South Florida landscape. Our results provide support for the hypothesis that predation risk shapes the spatial distribution and temporal activity patterns of prey populations (Brown et al. 1999, Laundré 2010) as well as evidence that white-tailed deer perceive spatial and temporal variability in risk and alter their behavior to mitigate exposure to that risk.

OBJECTIVE 2

Trends in the Camera Data

Trends in the Camera Data – Introduction

Effective wildlife monitoring requires data collection and analytical methods that yield accurate estimates of population state variables such as abundance and density (Nichols and Williams 2006). Collecting data at appropriate spatial and temporal scales is a challenge for large vertebrates such as deer because of the costs of surveying expansive regions. In South Florida, the challenge is especially great because much of the region is remote and difficult to access. Prior to this study, aerial line transect distance sampling was the preferred method for monitoring deer populations in South Florida. However, this method is expensive, potentially hazardous, and is not effective for surveying habitats with dense canopy cover where deer cannot be seen from the air. We therefore evaluated the efficacy of a camera-based survey method, and we used the results of our camera study to develop an optimal design for long-term monitoring of the deer population.

The primary purpose of the camera study was to assess spatial and temporal variation in deer abundance and density. We developed and applied novel methods for estimating abundance and density from camera data, and those results are presented in the next section. In this section, we summarize the raw camera data, and we describe trends in the observed detection rates of deer and their predators. We note that the raw camera data cannot be used as a reliable index of abundance because they will not be proportional to abundance if there is variation in detection probability arising from changes in activity patterns over time (Nichols and Williams 2006). For example, detections of male deer often spike during the rut, and the increased detection rate is a reflection of increased activity, not changes in abundance. Abundance must therefore be estimated by accounting for variation in detection probability, which is the focus of the Abundance and Density section.

Trends in the Camera Data – Methods

We used two summary statistics to characterize temporal and spatiotemporal trends in the camera data from 1 January 2015 - 31 December 2017. First, we calculated the daily detection rates at each of the three study areas for deer, panthers, bobcats, black bears, coyotes, and alligators. Daily detection rates were calculated by dividing the total number of detections at a site by the number of cameras that were operational on that day. In most cases, all 60 cameras were operational at each site on each day, but cameras occasionally failed, and in one instance, cameras were removed from a site that was threatened by wildfire. Daily detection rates provide a useful summary of trends in the camera data, but they can be influenced by high levels of variability caused by animals that spend long periods of time in front of a camera and are repeatedly photographed. To reduce the influence of these bursts of photographs, we calculated a second summary statistic: the daily proportion of cameras that had a least one detection of the species listed above. For deer, we calculated both summary statistics for adult males, adult females, and fawns. To visualize both spatial and temporal trends, we mapped 6-month averages of the detection rates at each camera.

Trends in the Camera Data – Results and Discussion

Over the three year camera study period, trail cameras recorded 31,798 deer photographs at AL, 39,063 deer photographs at BI, and 82,029 deer photographs at FPNWR (Table 24). AL had 7,465 male deer detections, 11,549 female deer detections, and 3,735 fawn detections. BI had 5,986 male deer detections, 14,315 female deer detections, and 5,290 fawn detections. FPNWR had 14,945 male deer detections, 35,471, female deer detections, and 9,304 fawn detections (Table 24). FPNWR had the highest average number of deer detections per camera across all age and sex groups for the three-year camera study period (Table 25). These values reflect counts of deer that could be reliably aged and sexed.

We did not observe strong temporal trends in overall deer detection rates over the course of the study (Tables 25, 26; Figs. 22, 23). At AL, deer detections decreased during 2017 compared to 2015 and 2016 for all sex and age classes. Adult female detection rates decreased slightly at BI. On FPNWR, the daily proportions of cameras with at least one detection was relatively stable for adult males and females. Adult male deer had higher detection rates during the rut (July - September; Figs. 24, 25). Adult female deer displayed no seasonal trend in detection (Figs. 26, 27). Fawn detection rates were highest during parturition (January - March) and over the following 5-6 months (Figs. 28, 29). Fawn detection rates were highest at FPNWR and lowest at AL where a slight decrease in detection rates occurred over the three years. Detailed analysis of the fawn data is provided in the Fawn Recruitment and Phenology section. All sex and age classes displayed spatial heterogeneity in detection rates within and among the three sites (Appendix F).

Panther detection rates tended to be higher on BCNP than on FPNWR, although 2017 detections were similar among management areas (Tables 27, 28). The proportion of cameras with at least one panther detection per day averaged 0.033 (0.022 - 0.040) at AL, 0.033 (0.021 - 0.047) at BI, and 0.020 (0.018 - 0.023) at FPNWR (Table 29). Panthers were detected throughout the year with no distinct seasonal correlations with white-tailed deer detection rates (Figs. 30, 31).

Black bear detection rates were similar at BI and FPNWR, and lower at AL (Tables 27, 28). The daily proportion of cameras with at least one black bear detection was 0.051 (0.045 - 0.056) at BI, 0.040 (0.033 - 0.052) at FPNWR, and 0.032 (0.019 - 0.051) at AL (Table 29). Bear detection rates were lowest during the winter months corresponding with bear reproductive cycle (Figs. 32, 33). Although males and non-reproductive female bears in Florida may remain active during the winter, parturient females typically den from late December to mid-April (Garrison et al. 2012). In South Florida, periods of low bear activity are concurrent with deer fawning season.

BI had more bobcat detections than AL and FPNWR across the three-year study (Table 27). BI had an average daily detection rate of 0.068 bobcats per camera day as opposed to 0.043 on FPNWR and 0.042 on AL (Table 28). The daily proportion of cameras with at least one bobcat detection averaged 0.047 (0.038 - 0.055) at BI, 0.031 (0.028 - 0.035) at FPNWR, and 0.031 (0.021 - 0.041) at AL (Table 29). Bobcats were detected throughout the year with no distinct seasonal trend (Figs. 34, 35).

Coyotes were detected infrequently (<1% of the camera days) over the course of the study (Tables 27, 28, 29). Coyote detection rates were highest during the fawn parturition and fawn rearing seasons (Figs. 36, 37). Alligators also exhibited infrequent camera detections (<1% of the camera days) over the course of the study (Tables 27, 28, 29). However, unlike coyotes, alligator detections showed no distinct seasonal trend coinciding with white-tailed deer detections (Figs. 38, 39).

Abundance and Density

Abundance and Density – Introduction

Estimating abundance and density of wildlife species that lack unique individual markings is a difficult challenge when conducting surveys with camera traps. For species with unique natural markings, camera data can be used with spatial capture-recapture (SCR) models—spatially-explicit extensions of traditional mark-recapture models—to make inferences about density and other population parameters (Efford 2004, Borchers and Efford 2008, Royle et al. 2013b, Appendix A). However, many species, including white-tailed deer females, lack natural markings, making it impossible to use standard SCR techniques. Recent extensions of SCR models have relaxed the requirement of individual recognition, allowing for inferences about abundance and density to be drawn from camera data on unmarked or partially marked populations (Chandler and Royle 2013, Sollmann et al. 2013). These approaches work best when ancillary information about detection probability is available, such as when a subset of the population has been marked and when telemetry data is available (Sollmann et al. 2013, Ramsey et al. 2015, Whittington et al. 2018).

Few studies have applied SCR models to data on white-tailed deer. Beaver et al. (2017) demonstrated that standard SCR models can be fitted to data on male deer that have been uniquely identified by their antler characteristics. Chandler et al. (2018) developed an SCR model to estimate white-tailed deer fawn survival and recruitment. Their model requires that all detected fawns can be uniquely identified based on spot patterns (see Fawn Recruitment and Phenology). Although these methods were shown to be highly effective, they are labor intensive because they can involve manual interpretation and cross-referencing of thousands of images. Moreover, adult female deer lack unique markings, prohibiting the use of standard SCR models.

Our aim was to evaluate SCR models of abundance and density that do not require individual encounter history data.

Abundance and Density – Methods

We developed a framework for extending existing “unmarked SCR” methods (Chandler and Royle 2013) to model white-tailed deer by combining camera and telemetry data from January 2015 - December 2017. The core of the model is a standard SCR model in which each individual is assumed to have an activity center that is uniformly distributed within the spatial region of interest, $s_i \sim \text{Unif}(\mathcal{S})$. The expected number of detections (λ_{ijk}) of individual i at camera j on occasion k is assumed to decrease with distance (d_{ij}) between the activity center and the trap: $\lambda_{ijk} = \lambda_0 \exp(-d_{ij}^2/(2\sigma^2))$. The baseline encounter rate parameter λ_0 is the expected number of detections on a single occasion for an individual when the distance between its activity center and a trap is zero (Appendix A). The spatial scale parameter σ determines the rate at which detection probability decreases with distance. If individuals were uniquely identifiable, the data would be individual-level encounter histories. However, without information about identity, the data (n_{jk}) are counts of individuals, or binary values indicating if at least one deer was detected at each camera on each occasion. Count data can be difficult to model because deer frequently stop in front of the cameras and are photographed repeated in a short time period. Discarding non-independent detections often involves making subjective decisions, and modeling the dependent detections would involve a complicated model for the lack of independence, which is a nuisance process. We therefore chose to analyze the binary data, using an occasion length of

24-h. Under the assumption that the number of independent detections of each deer is Poisson distributed, the model for the binary data becomes: $n_{jk} \sim \mathrm{Bern}(1 - \exp(-\Lambda_{jk}))$ where The Bernoulli variable z_i indicates if individual i was a member of the population. This variable is introduced as part of a data augmentation process that is frequently used in Bayesian analysis of SCR models (Chandler and Royle 2013, Royle et al. 2013b). When using data augmentation, abundance is estimated as

$$N = \sum_{i=1}^{M} z_i$$

Density is estimated by dividing N by the area of $\mathcal{S}$.

Without individual-level encounter histories, the binary camera data provide little information about the encounter rate parameters λ_0 and σ (Chandler and Royle 2013). We therefore used telemetry data from the GPS-collared individuals to estimate these two parameters. The telemetry data provide direct information about the location of an individual's activity center and the probability that it is detected by a camera. Our model for the encounter histories of the GPS-collared deer was $y_{ijk} \sim \mathrm{Bern}(1 - \exp(-\lambda_{ijk}))$. We used a simple bivariate normal model for the telemetry locations: $u_{it} \sim \mathrm{Norm}(s_i, I\sigma^2)$.

For the female deer analyses, we defined the state-space using a 1 km buffer around each camera array, while we used a 2 km buffer for the male deer analyses. We assumed population closure during 14-day periods. In the analysis of the GPS-collared deer, we modeled temporal autocorrelation in encounter rate parameters among the 78 fortnights using a Gaussian AR(1) model on the natural log scale. The parameter means over the three year time period are denoted

$$\Lambda_{jk} = \sum_{i=1}^{M} \lambda_{ijk} z_i$$

by $\bar{\lambda}_0$ and $\bar{\sigma}$. Telemetry locations during the fortnight period of 7-20 September 2017 were excluded from the analysis due to irregular deer movements relating to Hurricane Irma. We ran the model for the collared deer for 10,000 iterations after an adaptive phase of 100 iterations and a burn-in of 15,000 iterations. We fit the first stage of the model to the GPS collared deer using the package *rjags* (Plummer et al. 2018) in program R (R Core Team 2019). We report model estimates starting with the first fortnight including GPS-collared individuals within the camera sites.

Posterior distributions of λ_0 and σ obtained from the first stage of the analysis were used as prior distributions in the analysis of the binary camera data (stage two) using the model of Chandler and Royle (2013). The second stage of the analysis was conducted using a custom MCMC algorithm written in R. We ran the female deer model for 10,000 iterations after a burn-in of 1,000 iterations and ran the male deer model for 5,000 iterations after a burn-in of 1,000 iterations. Parameter estimates reported below are posterior means and 95% CIs unless specified otherwise.

Abundance and Density – Results and Discussion

Female Abundance and Density

For AL, we used detection histories and telemetry locations from 10 GPS-collared female deer to estimate the encounter rate parameters (Fig. 40). The estimate of the mean spatial scale parameter ($\bar{\sigma}$) was 369.2 (151.5-849.1; Fig. G1). The estimate of the mean baseline encounter rate parameter ($\bar{\lambda}_0$) was 0.056 (0.025-0.112; Fig. G2). We estimated an average female deer abundance across the three year study within a 1 km buffer to be 72 individuals (Fig. 41), translating to a density of 1.46 females/km^2. The lowest mean fortnight estimate was 0.56 females/km^2, while the highest fortnight estimate was 2.63 females/km^2.

For BI, we used detection histories and telemetry locations from 31 GPS-collared female deer (Fig. 42). The estimate of $\bar{\sigma}$ was 231.1 (219.8-242.2; Fig. G3). The estimate of $\bar{\lambda}_0$ was 0.062 (0.042-0.088; Fig. G4). We estimated an average female deer abundance across the three year study within a 1 km buffer to be 175 individuals (Fig. 43), translating to a density of 3.54 females/km^2. We excluded density estimates for two fortnights due to models failing to converge and camera removal during wildfires. The lowest mean fortnight estimate was 1.74 females/km^2, while the highest fortnight estimate was 7.28 females/km^2.

For FPNWR, we used detection histories and telemetry locations from 18 GPS-collared female deer (Fig. 44). The analysis of the data on GPS-collared deer resulted in an estimate of 197.5 (189.8-205.6, Fig. G5) for $\bar{\sigma}$ and 0.126 (0.070-0.217; Fig. G6) for $\bar{\lambda}_0$. We estimated an average female deer abundance across the three year study within a 1 km buffer to be 240 individuals (Fig. 45), translating to a density of 4.57 females/km^2. We excluded density estimates for five fortnights due to models failing to converge. The lowest mean fortnight estimate was 1.61 females/km^2, while the highest fortnight estimate was 7.97 females/km^2. The variation in density estimates at FPNWR are a reflection of the encounter rate parameter ($\bar{\lambda}_0$) priors incorporated in the unmarked model (stage two). Imprecise estimation of $\bar{\lambda}_0$ (Fig. G6) yielded highly variable density estimation resulting in high density estimates during periods of low encounter rates.

Male Abundance and Density

For AL, we used detection histories and telemetry locations from 7 GPS-collared male deer to estimate the encounter rate parameters (Fig. 46). The estimate of the mean spatial scale parameter ($\bar{\sigma}$) was 520.7 (471.1-575.0; Fig. G7). The estimate of the mean baseline encounter rate parameter ($\bar{\lambda}_0$) was 0.018 (0.011-0.127; Fig. G8). From 4 April 2015 - 13 January 2016, no

GPS-collared males remained on site, yielding imprecise parameter estimates for the 20 fortnight periods. We estimated an average male deer abundance across the three year study within a 2 km buffer to be 166 individuals (Fig. 47), translating to a density of 2.03 males/km^2. We excluded density estimates for two fortnights due to models failing to converge. The lowest mean fortnight estimate was 0.38 males/km^2, while the highest fortnight estimate was 7.59 males/km^2.

For BI, we used detection histories and telemetry locations from 13 GPS-collared male deer to estimate the encounter rate parameters (Fig. 48). The estimate of the mean spatial scale parameter ($\bar{\sigma}$) was 381.5 (304.7-475.5; Fig. G9). The estimate of the mean baseline encounter rate parameter ($\bar{\lambda}_0$) was 0.027 (0.014-0.047; Fig. G10). We estimated an average male deer abundance across the three year study within a 2 km buffer to be 144 individuals (Fig. 49), translating to a density of 1.75 males/km^2. We excluded density estimates for two fortnights due to models failing to converge and camera removal during wildfires. The lowest mean fortnight estimate was 0.12 males/km^2, while the highest fortnight estimate was 7.35 males/km^2.

For FPNWR, we used detection histories and telemetry locations from 12 GPS-collared male deer to estimate the encounter rate parameters (Fig. 50). The estimate of the mean spatial scale parameter ($\bar{\sigma}$) was 350.0 (319.7-382.2; Fig. G11). The estimate of the mean baseline encounter rate parameter ($\bar{\lambda}_0$) was 0.051 (0.029-0.073; Fig. G12). During December 3 - December 30, 2015, no GPS-collared males remained on site, yielding imprecise parameter estimates for the 2 fortnight periods. We estimated an average male deer abundance across the three year study within a 2 km buffer to be 162 individuals (Fig. 51), translating to a density of 1.89 males/km^2. The lowest mean fortnight estimate was 0.45 males/km^2, while the highest fortnight estimate was 4.39 males/km^2.

Since the state space (*S*) varied in model development for each sex, no comparisons can be drawn between abundance estimates for males and females. Therefore, we calculated sex ratios using the ratio of mean density estimates for each site. The adult deer sex ratio was 58% males on AL, 33% males on BI, and 29% males on FPNWR. The adult deer sex ratios reported were calculated using the mean for the 3-year period and do not reflect the temporal variation in sex-specific density estimates. Male and female density estimates were combined to produce adult deer density estimates (Fig. 52). Mean adult deer density was 3.51 deer/km^2 for AL, 5.31 deer/km^2 for BI, and 6.39 deer/km^2 for FPNWR. Historical aerial line-transect surveys yielded deer density estimates of approximately 3.5-4.0 deer/km^2 within lower BCNP (Labisky et al. 1995). During 1990-1994, spotlight surveys in western BI resulted in variable annual deer density estimates ranging from 0.973 to 6.336 deer/km^2 (Bozzo and Schortemeyer 1995). Thus, density estimates from our study fall within the range of historical estimates. Estimates from more recent aerial surveys using distance sampling methods were considerably lower; in AL, a five year (2014-2018) global density estimate was 1.20 deer/km^2 (95% CI: 0.98-1.45) and in BI 2.12 deer/km^2 (1.66-2.77). However, due to the low detection rates of deer in the closed canopy portions of the units, the estimates are known to be biased low and are used as indices of population trends, rather than actual density estimates. In addition, aerial surveys are conducted throughout the whole unit, whereas camera grids covered a smaller proportion of the unit. Therefore, direct comparison with camera-based density estimates should be viewed with those caveats in mind.

We detected spatial variation in density across our study area. Female density estimates followed a longitudinal gradient, being highest in the drier, western portion of the study area in FPNWR and lowest in the wetter eastern region in AL. The estimates of the baseline encounter rates (λ_0) for both sexes followed a similar gradient, while the estimates of the spatial scale parameter (σ) displayed a reverse gradient, being highest in AL and lowest in FPNWR. Male density estimates remained relatively stable across sites.

Several options exist for obtaining more precise density estimates. Data on unmarked individuals will always yield estimates that are less than precise than data on marked individuals. If resources are available to uniquely identify males and spotted fawns, these data could be analyzed with standard spatial capture-recapture models. Modeling temporal autocorrelation in density estimates could yield more precise estimates. Additionally, transitioning the SCR model from the closed population model of 78-fortnight periods to a 3-year open population model would allow for the incorporation of demographic processes such as survival and recruitment (Gardner et al. 2010, Chandler and Clark 2014, Gardner et al. 2018).

Highly dynamic systems often require long-term data to understand population dynamics and conservation status of species in order to make informed management decisions. While our study occurred continuously for three years, >10 years of monitoring is often required to detect population trends at high statistical power (Gerrodette 1987, Dixon et al. 1998, White 2018). Our estimates indicated that density varied over time at each of the three study areas, but most of the variation was seasonal, and there was no evidence of persistent population declines. Gradual trends in density could be detected by applying our camera trapping methodology and statistical modeling framework to monitoring data collected over longer time periods. Design recommendations are provided in the Optimal Monitoring section.

Fawn Recruitment and Phenology

Fawn Recruitment and Phenology – Introduction

Fawn survival and recruitment are critical processes influencing deer population dynamics, but reliable estimates are difficult to obtain because of the challenges associated with capturing and monitoring fawns over sufficient time scales. Although much published information is available about adult deer survival rates, studies focused on juvenile survival and recruitment are often highly variable or limited by small sample sizes. Previous studies focused on white-tailed deer fawn survival have documented a wide range of mortality rates, from 9.3% to 90% (DeYoung 2011). Numerous environmental variables, including the quality and quantity of vegetation, availability of concealment cover, previous fire history of an area, the suite of predator species present, and intensity of human disturbance, can influence both fawn production and fawn survival across the landscape (Taber and Dasmann 1957, Loveless 1959a, Nelson and Woolf 1987, Berger 2007, Tollefson et al. 2011, Shuman et al. 2017).

South Florida deer exhibit lower productivity than many other deer populations in temperate North America, due primarily to a low fecundity rate, estimated between 1.18-1.26 fetuses/pregnant female (Richter and Labisky 1985, McCown 1991, Fleming et al. 1994). Fawn survival in South Florida can be highly variable due to fluctuations in local environmental conditions. A previous BCNP study recorded annual fawn survival rates ranging from 0% to 42.6% (Labisky et. al 1995). The Florida Fish and Wildlife Conservation Commission’s annual aerial recruitment surveys over the Everglades Wildlife Management Area and surrounding state lands documented fawn-to-doe ratios averaging 35.44% from 2007-2017 (Ward, M. R., Florida Fish and Wildlife Conservation Commission, unpublished data). However, aerial surveys are often biased by detectability (Caughley 1975), and they are best used to track annual trends in fawn recruitment rather than provide precise estimates of fawn survival.

We examined white-tailed deer fawn production and survival to recruitment using a novel and non-invasive SCR model (Chandler et al. 2018) applied to encounter histories of fawns uniquely identified using their spot patterns. This model was developed to build upon recent SCR advancements that allow estimation of survival and recruitment while accounting for individual variation in detection probability (Gardner et al. 2010, Royle et al. 2013a). We used data from 180 passive cameras to estimate (1) fawn productivity in the study area, (2) effects of factors affecting spatial variation in birth locations, and (3) fawn survival rate to the recruitment age. We investigated the effects of vegetation type and fire history on the spatial variation in the density of fawn birth locations. We also evaluated the effect of birth location and fawn age on survival rate.

Fawn Recruitment and Phenology – Methods

We utilized photos of spotted fawns taken by 180 cameras over two fawning seasons, spanning from 1 December 2014 to 1 October 2015 (hereafter the 2015 fawning season) and from 1 December 2015 to 1 October 2016 (the 2016 fawning season). We uniquely identified fawns in each photo using their spots, which are distinctive from birth until approximately six months old (Fig. 53). As described in Chandler et al. (2018), we identified unique spot patterns from each side of the fawn, as well as photos from directly behind or in front of the fawn, to confirm a complete individual fawn ID. We removed non-independent fawn detections using a 1-hour detection threshold of the same individual at the same camera. We then created spatially-referenced capture histories for each individual indicating all detections of each fawn at each camera (Fig. 54). We developed a binary camera operational status matrix for both fawning seasons to account for camera malfunctions or non-operational occasions.

We estimated a birth date range for each fawn based on morphological characteristics in the photo detections (Fig. 54). We considered body size of the fawn relative to the dam, head size and shape, brightness of spots, and length of hind foot relative to body size to determine minimum and maximum birth dates. Two experienced observers independently viewed all images of each fawn and created a range of possible birth dates by estimating the minimum and maximum age of the fawn in each photo. Fawns detected very young (<10 days) and surviving many months provided a baseline for determining the birth date ranges of fawns that were detected less frequently. Discrepancies between the independent age estimates were resolved by combining or averaging the estimates from the two observers to create a conservative birth date range.

We modeled the number of fawns born, spatial and temporal variation in the density of fawn birth locations, fawn survival to recruitment, and detection probability using the model described in Chandler et al. (2018). This model is a type of open population SCR model in which birth locations are modeled as outcomes of a spatial point process (Borchers and Efford 2008, Gardner et al. 2010). A birth location is defined as the estimated area where the fawn was born, which cannot be precisely determined using camera data. Because our methods did not include capturing neonates at the birth site, we inferred birth locations by using the locations and the ages of the fawns detected by our cameras. The older a fawn was at first detection, and the fewer photos of a fawn in the dataset, the more uncertainty was present around the estimated birth location. Spatial variation in the density of birth locations was modeled using spatially-referenced covariates. Individual lifetimes were modeled using a failure-time approach

(Chandler et al. 2018). We modified the model presented in Chandler et al. (2018) to include covariates of the hazard rate, which was defined as the daily mortality rate. Detection probability was modeled as a function of the individual's age and the distance between its birth location and the location of the camera trap, while accounting for the fact that the birth location is unknown and must be estimated. As discussed in Chandler et al. (2018), the probability of detecting a fawn far from its birth location is assumed to increase with age, as the fawn becomes more mobile. Additional detection parameters can also be modeled using trap-specific temporally-varying covariates. To account for the fact that fawns could not be individually identified after they lost their spots, we set the detection probability to zero for individuals greater than 180 days old.

To define the spatial region that included the population of interest, we created an 800 m buffer around each trail camera, which resulted in a cumulative 10,941 ha region around the three camera grids. This buffer was chosen to be large enough to include the birth locations of all fawns that could have been detected by our cameras. This was determined by estimating the detection parameters with a larger buffer and reducing it to meet the requirement that fawns born near the edge of the spatial region had a negligible detection probability.

We examined the effect of vegetation type and fire history on the number of fawns born and spatial variation in birth location density. We classified vegetation type and cover using FNAI GIS raster site data (FWC/FNAI 2016) cropped to our rectangular 53,333 ha complete study area and reclassified the 72 FNAI vegetation types into four mutually-exclusive categories to capture the variation in canopy closure, cover availability, forage production, and seasonal hydroperiod that may impact movement and survival of females and fawns. We grouped vegetation types as pine flatwoods, cypress forests, hardwood hammocks, and open canopy (including marsh and prairie). For the fire history on FPNWR, we used burn-unit-level records of both wildfires and prescribed burns from 1 January 1995 - 31 December 2015 to calculate fire frequency. For the two BCNP camera grids, we used ArcGIS 10.5 (ESRI, Redlands, CA, USA) to stack individual raster layers that detailed each fire within our study area between 1 January 1995 and 31 December 2015 and calculated the frequency of fire in each 30x30 m pixel in the study area.

We used Bayesian methods for statistical inference. We preformed analysis using a Gibbs sampler written in program R (R Core Team 2019). We fit one model to each fawning season dataset separately. Each model included a raster for the four vegetation types and 20-year fire history as predictors for the variation in density of fawn birth locations. We included trail status (on- or off-trail) for each camera as a covariate of fawn detection probability. We modeled survival as a function of both fire history in the fawn's home range and individual fawn age.

Fawn Recruitment and Phenology – Results and Discussion

In 2015, we utilized 6,914 images of spotted fawns and identified 145 unique individuals across the study area: 33 individuals at AL grid, 57 at BI, and 55 at FPNWR. In 2016, we utilized 5,801 images of spotted fawns and identified 124 unique individuals across the study area: 29 individuals at AL grid, 31 at BI grid, and 64 at FPNWR.

The estimated peak birth date across our study site for both the detected and undetected fawns was 24 January in 2015 and 28 January in 2016 (Fig. 54). For the 2015 fawning season, we estimated that 283 fawns (95% CI: 231-361) were born in the 10,941 ha study grid. The model predicted that the density of fawn birth locations per square km was highest in the hardwood hammocks (7.95 fawns/sq. km, 95% CI 4.43-12.64), and this estimate was over twice as high as the densities in flatwoods, cypress, or open canopy habitats (Fig. 55). For the 2016 fawning season, our model predicted that 268 (95% CI 210-357) fawns were born in the study area. Hardwood hammocks still had the highest density of fawn birth locations in 2016 (4.14 fawns/sq. km, 95% 1.05-7.60), but this was only slightly higher than the other habitat types and was followed closely by cypress (3.93 fawns/sq. km, 95% 2.38-6.17). Hammocks may provide increased cover for females and fawns for parturition and throughout the hiding stage, highlighting the importance of patches of thick concealment cover for fawn survival. Fire history over the past 20 years did not have a significant effect of the density of fawn birth locations in either 2015 or 2016.

Trail status (on- or off-trail) did not have a significant impact of fawn detection probability in 2015. However, in 2016, trail status had a significantly positive effect on fawn encounter rate in 2016, where fawns were more likely to be detected at an on-trail camera than off-trail camera. Fire history did not have a significant impact on overall fawn survival rate in either 2015 or 2016. Additionally, there were no significant changes in fawn survival rate in either 2015 or 2016 with increasing fawn age.

Fawn survival to 30 days was 83.92% in 2015 and 70.18% in 2016 (Fig. 56). We estimated that 123 of the 283 fawns (43.5%) born in the 2015 fawning season survived to the recruitment age, which we specified as 180 days (Fig. 57). For the 2016 fawning season, only 36 of 268 (13.4%) fawns survived to the recruitment age. This significant difference in fawn recruitment between 2015 and 2016 is evident from early on in both the realized survivorship curve (Fig. 56) and the fawn and recruit abundance graph (Fig. 57).

The drastic difference we detected in recruitment rates through our camera study is corroborated by annual aerial recruitment surveys that estimated a recruitment rate of 43.6% in 2015, but only 17.0% in 2016 (Ward, M. R., Florida Fish and Wildlife Conservation Commission, unpublished data). The decrease in fawn survival between the two years is likely due to the high-intensity rain and record flooding that South Florida experienced during the wet season of 2016. Mean daily water levels from 1 January to 1 April 2016 ranged from 60.5 - 85.4 cm in marsh vegetation, whereas the water levels documented at the same wells from 1 January to 1 April 2015 ranged from only 7.4 - 46.6 cm (data from EDEN). A previous BCNP study indicated that water depths above 50 cm severely impact adult deer movement, survival, and productivity (MacDonald-Beyers and Labisky 2005). Additionally, their study documented a 10-fold decrease in apparent fawn recruitment during a year of significantly high rainfall (MacDonald-Beyers and Labisky 2005). In two other studies, standing water depths over 30 cm and 45 cm during the fawning season significantly reduced fawn recruitment (Fleming et al. 1994, Garrison et al. 2011). Thus, it is likely that the 3-fold decrease in fawn survival rate between the two years was due to the severity of the wet season. The raw photo data shows a lower rate of spotted fawn detections in the 2016 fawning season compared to 2015 (Fig. 28). Additionally, our individual fawn detection histories show that fewer fawns are continuously detected throughout the length of 2016 fawning season, further supporting our finding of lower survival in 2016 compared to 2015 (Fig. 54).

The fawn survivorship curves in both years showed a relatively high survival rate through the first 30 days (83.92% and 70.18% for 2015 and 2016, respectively) compared to many other white-tailed fawn survival studies in the southeast (Fig. 56). Collared fawn survival studies typically report a steep initial decline in survival, where roughly one-third of fawns survive the first 30-60 days (Jackson and Ditchkoff 2013, Watine 2015, Shuman et al. 2017). This high mortality rate often stabilizes as fawns reach a size that reduces their risk of predation, and overall survivorship remains around one-third of the original collared sample. However, the survival rate we observed continued to decline sharply, with 55.14% and 30.75% (2015 and

2016, respectively) of fawns surviving to 120 days and 43.46% and 13.43% of fawns surviving to 180 days (Fig. 56). The unique pattern of fawn survivorship detected may be due to the structure of the predator community in South Florida. Many fawn studies in the Southeast find coyotes and bears to be the most highly efficient fawn predators within the first 30 days (Jackson and Ditchkoff 2013, Watine 2015, Shuman et al. 2017). However, coyotes exist only at low densities in South Florida, and bears are still largely inactive when fawns are born due to the early spring fawn parturition seen in South Florida. These unique environmental characteristics may allow neonate fawns to survive at a higher rate in the first 30 days than seen in comparative studies in the Southeast. However, once fawns are slightly older, bobcats and panthers likely provide an increased risk of predation that continues to reduce the fawn population, unlike in the rest of the Southeast where fawn survival stabilizes after 30-60 days and there are fewer highly efficient predators of juvenile or adult deer.

In this analysis, we demonstrated the use of a recently published non-invasive SCR method (Chandler et al. 2018) to resolve a critical lack of information regarding white-tailed deer fawn survival and recruitment in South Florida, where deer are the primary prey of the Florida panther. Additionally, we provided the first estimates of fawn survival to recruitment in BCNP since the 1990's (Land 1991, Labisky et al. 1995) and evaluated the landscape-level environmental variables that impact fawn birth location and activity rates in this region. We modeled survival and recruitment of fawns in a typical year and a high-severity flood year in South Florida. Estimates suggest that few fawns born in this region will survive to reproductive maturity in extremely high-water years, but recruitment in typical years may be sufficient to sustain the population. Due to the severity of the flood-drought cycle in South Florida, monitoring annual survival and recruitment of adults and juveniles over many years is necessary to understand the true trajectory of population growth. However, our fawn recruitment study only consisted of two years of fawning data in this highly variable ecosystem. Analysis of multiple fawning seasons of data is needed to thoroughly assess the annual variability in fawn survival and recruitment over time.

Antler Casting

Antler Casting – Introduction

Antlers are a secondary sexual characteristic of males that go through an annual cycle of growth, mineralization, and casting. This cycle is closely timed with reproductive seasons and regulated by both sexual hormones and photoperiod (Bubenik 1990). In the Everglades, antler growth begins in February and continues through July. By August, antlers have mineralized. Males shed their antlers beginning in late November and most have been shed by late January (Loveless 1959b). Antler casting dates can improve understanding the reproductive chronology in males and help inform hunting regulations and harvest information, as harvest is often regulated by antlered or antlerless deer.

Antler Casting – Methods

To estimate the timing of antler casting, we used the trail camera data to categorize all male deer independent detections (5 min threshold, $n = 11,748$) where the state of the antler growth could be clearly identified as antlered ($n = 9,046$) or antlerless ($n = 2,702$). Males were categorized as antlered if they had at least one calcified antler or significant velvet growth. Males were categorized as antlerless if they had shed their calcified antlers from the previous growth cycle and had yet to begin significant velvet growth. In order to account for camera trapping effort, the number of antlerless photos per day was represented as a proportion of the total number of male deer photos captured that day. Bar graphs of the proportion of daily antlerless deer captured were created across the year for each of the three grids to visually estimate the peak casting date across each grid.

Antler Casting – Results and Discussion

Antler casting for the FPNWR grid peaks around 27 January, antler casting for the BI grid peaks around 5 February, and antler casting for the AL grid peaks around 15 January (Fig. 58).

Optimal Monitoring Design

Optimal Monitoring Design – Introduction

Camera traps can be used to study population dynamics and to monitor populations as a component of management programs (Nichols et al. 2011, O'Connell et al. 2011). When only a small fraction of a large, heterogeneous landscape is able to be surveyed, selecting an appropriate camera trap design becomes vital in obtaining reliable estimates of abundance and distribution. Modern approaches for analyzing data from camera trap studies do not require random placement of camera traps, but the scope of inference and the precision of estimates is strongly influenced by the number and distribution of deployed cameras (Royle et al. 2013b, Ch. 10). Simulation studies can be used to assess the impacts of design decisions on parameter estimates. When combined with a cost analysis, this approach can be used to identify an optimal design to meet specific monitoring objectives.

Optimal Monitoring Design – Methods

We conducted a simulation study to identify the optimal design for camera-based monitoring of white-tailed deer populations in the study area. We developed 10 potential designs for each of the three study sites using data from the camera study and using input from agency biologists about the feasibility of operating cameras in the region. For each design, we simulated 500 datasets using an extension of the population model described in the Abundance and Density section that allowed the baseline detection rate to differ between on- and off-trail cameras. Next, we fitted the model to each simulated dataset and computed the bias and precision of abundance estimates for each design. The design resulting in the lowest root-mean-squared error (RMSE), which is a function of bias and precision, was deemed the optimal design from an estimation perspective. Financial considerations are taken into account in the following Cost Analysis section.

Evaluated Camera Trap Designs (Maps of each design are shown in Figs. H1, H2, H3).

1) Status Quo (SQ)

This design corresponds to the design used during the study, with 60 cameras (40 on-trail, 20 off-trail) at each of the three sites site.

2) Forty paired on- and off-trail cameras (P)

This design involves 40 cameras at each of the three sites, with 20 on-trail cameras and 20 paired off-trail cameras. The paired camera locations are the same as those used during the study.

3) Twenty random cameras (R20)

This design involves 20 cameras at each of the three sites. The 20 camera locations per site were randomly sampled from the existing 60 locations used in the study, allowing for both on- and off-trail.

4) Thirty random cameras (R30)

Similar to R20 but with 30 randomly selected cameras at each of the three sites.

5) Forty random cameras (R40)

Similar to R20 but with 40 randomly selected cameras at each of the three sites.

6) Twenty on-trail cameras, no off-trail cameras (T20)

This design involves 20 on-trail cameras and no off-trail cameras at each of the three sites. The 20 camera locations per site were randomly sampled from the 40 on-trail locations used in the study.

7) Thirty on-trail cameras, no off-trail cameras (T30)

Similar to T20 but with 30 cameras at each of the sites.

8) Forty on-trail cameras, no off-trail cameras (T40)

Similar to T20 but with 40 cameras at each of the three sites.

9) Fifty on-trail cameras, no off-trail cameras (T50)

Similar to T20 but with 50 cameras at each of the three sites. The 50 on-trail cameras per site include the 40 camera locations used in the study, and an additional 10 on-trail camera locations that were randomly selected with the constraint that they were spaced by a minimum of 250 m from existing cameras.

10) Sixty on-trail cameras, no off-trail cameras (T60)

Similar to T50 but with 60 cameras at each of the three sites.

Optimal Monitoring Design – Results and Discussion

Increasing the number of cameras deployed increased the average number of deer detection events (Fig. 59), defined as the detection of at least one deer at a camera during a 24 h sampling occasion. In general, all camera designs performed well with minimal bias and nominal coverage of 95% CIs (Figs. 60, 61). Precision increased with the number of camera traps, but precision was not influenced by the placement of cameras on vs. off trails. In other words, for designs with the same number of cameras, precision of abundance estimates was similar when all the cameras were on trails as when half of the cameras were off trails. Across all camera sites, the optimal design for estimating abundance (i.e., the design with the lowest RMSE) was design 10 (60 on-trail cameras, Table 30). Camera designs involving only 20 camera traps performed worst overall. However, this analysis did not include financial costs, and the optimal design for balancing precision and costs are described in the Cost Analysis section.

Spacing of cameras did not strongly influence bias or precision of estimates of abundance, a finding that contrasts with previous studies of SCR designs (Royle et al. 2013b, Ch. 10). For a fixed number of cameras, spacing influences the number of individuals that will be detected and the number of spatial recaptures (i.e., recaptures at different locations) of each individual. In most settings, an optimal SCR design is one that maximizes the number of individuals detected and the number of spatial recaptures. However, the primary role of spatial recaptures is to provide information about the encounter rate parameters (λ_0 and σ), which is relatively unimportant in this case because we used GPS telemetry data to provide direct information about the encounter rate parameters. Regardless, in future years, when telemetry data is not available, we recommend following the guidance from Chandler and Royle (2013) that camera spacing should be approximately 2σ. Thus, for females that had values of σ ranging from approximately 200 to 300, we recommend spacing cameras by 400-600 m, equivalent to 1 camera per 16-36 ha. Males have larger home ranges, but population viability is more closely linked to female abundance, and we therefore suggest designing the study to optimally monitor females.

Cameras can be operated continuously throughout the year, or they can be operated seasonally to reduce costs. For seasonal monitoring, we recommend deploying cameras during May and June for an 8-week period when movement and detection parameters are relatively stable. This timeframe also allows for the potential to uniquely identify fawns and adult males. May is the best month for collecting data on fawns because most fawns in a cohort have been born by that time, and most fawns have unique spot patterns until June. Incorporating sampling in June allows for data to be collected on males when most antlers have hardened, allowing for individuals to be uniquely identified. If data are collected continuously over two months, density of adults could be estimated during four 14-day periods within which the population could be assumed to be closed with respect to recruitment and mortality. The closure assumption is more difficult to meet for fawns because their mortality rate is higher, so density could be estimated separately for each 7-day period within May, or open population models could be used to relax the closure assumption (Chandler et al. 2018). If open population models are used, data should be collected from 1 December until 1 August to span the parturition period through the spot loss period.

Cost Analysis

Cost Analysis – Introduction

A tradeoff exists between the cost of a study design and the accuracy of estimates yielded by the design. The previous section focused on accuracy. In this section, we present the results of a cost analysis in which we calculated the monetary costs of implementing each of the design options being considered for long-term monitoring.

Cost Analysis – Methods

We conducted a cost analysis comparing aerial line transect surveys to trail camera monitoring for white-tailed deer in South Florida. Currently, FWC conducts annual aerial surveys for white-tailed deer within BCNP. Thus, for comparison purposes, we used the BI unit as an example survey unit for both monitoring techniques. We evaluated the cost for each camera monitoring design under the assumption of 14 camera days during May. The 14-dayperiod represents a balance between the goals of obtaining a large sample of detections and meeting the population closure assumption of SCR models (see Abundance and Density). A 14-day period is also comparable to the duration of a single aerial survey. Additionally, we evaluated the costs for each camera monitoring design under the assumption of seasonal monitoring during May and June.

Cost Analysis – Results and Discussion

Based on the estimated cost analysis, camera design 6 (20 on-trail cameras, no off-trail cameras) is the most cost efficient approach to monitoring deer in South Florida (Table 31). When comparing a single year of surveying, only two camera designs produced costs less than costs of aerial surveys. Both of these designs used the minimum number of trail cameras proposed. The majority of costs for each camera design survey was incurred during the initial purchase of equipment, which included: trail cameras, trail camera boxes, memory storage cards, batteries, and GPS units (Table 31). Labor required for surveying was higher for trail camera monitoring than aerial surveying. Aerial surveying required a two-person survey team for two half day flights. In contrast, trail camera surveying required both the deployment and retrieval of cameras following the survey period. Camera designs that implemented both on- and off-trail cameras resulted in higher field labor due to the extended time needed to travel to off-trail locations. Data management and analysis costs were higher for camera monitoring due to extended time needed for the photo tagging process.

When evaluating the costs of surveying on a recurring yearly basis, aerial surveys were three times more expensive to implement than trail camera monitoring. Aerial surveys required the rental of helicopter services for two flight paths, resulting in an estimated annual cost of $5,752.50 to survey BI. In contrast, trail camera monitoring only required labor since all equipment would be owned by FWC. The variation in recurring yearly costs became noticeable when investigating the potential for implementing a 5-year study protocol. Trail camera monitoring became more cost efficient than aerial surveying with a minimum estimated savings of $6,363 using the most expensive camera design.

The optimal monitoring design should achieve a balance between the objectives of maximizing the precision of estimates while minimizing costs. When comparing the camera designs across all sites, camera design 10 (60 on-trail cameras) performed best at maximizing the precision of estimates, while camera design 6 (20 on-trail cameras) best minimized the costs of surveying (Fig. 64). However, camera design 8 (40 on-trail cameras) performed best overall at meeting both objectives when equally weighting each objective.

While the cost analysis focuses on comparing 14-day camera monitoring, cameras can be operated throughout the year or seasonally depending on management objectives. Extending camera monitoring to a 2-month period would allow for 4-fortnight periods from which to estimate density. For seasonal monitoring, we recommend that cameras be deployed during May and June when males can be uniquely identified by antler morphology (June), and when fawns can be uniquely identified by their spot patterns (May). Increasing camera monitoring to a two month period incurs additional costs for data analysis, but is still below the operational costs of aerial surveying for long-term monitoring (Table 32). During May and June, the seasonal hydrology is shifting from the dry season into the wet season (Fig. 7). Depending on the timing and severity of the hydrological season, field labor cost may increase due to increased travel time caused by trail inundation.

MANAGEMENT RECOMMENDATION

Water Management

Water management decisions in South Florida address multiple objectives including wildlife management. Our results support previous research indicating that high water levels can negatively impact deer recruitment and female survival. Female survival was decreased by up to 9% when water level reached 0.5 m, and up to 22% at 1 m. Managers seeking to increase deer population viability should not only take into consideration the impact of water when setting objectives for survival, but also the linear relationship between these parameters, as any increase in water depth could affect survival. Likewise, hydrological restoration efforts aiming at the restoration and protection of water resources in Central and South Florida need to acknowledge the consequences of water management decisions on local conditions that may affect deer populations. Water management decisions in South Florida address multiple objectives including wildlife management. Our results support previous research indicating that high water levels can negatively impact deer recruitment and adult survival. Managers seeking to increase deer population viability should ensure that habitats with less than 0.5 m of standing water are interspersed throughout the region. Currently, these conditions exist where hydric pines, mesic pines, and hardwood hammocks occur in the study area. Preventing these habitats from becoming inundated during most of the year would be beneficial to the deer population.

Harvest Management

Legal harvest represented a small fraction of the 134 mortalities documented during our study. Only one out of the 241 deer with active GPS collars was legally harvested, and only one other deer was known to have been legally harvested after its GPS collar failed. These results suggest that legal hunting had a negligible impact on the deer population during the study period. The low impact of legal hunting on the deer population was the result of regulations that prohibited hunting on the FPNWR, and limited harvest to males with at least two antler points on BCNP. We found that hunter harvest rates were very low, which suggests that current harvest regulations are adequate for meeting the objectives of allowing for sustainable deer hunting opportunities without compromising panther recovery efforts. However, additional management actions (including changes to hunting regulations) may be necessary in the future if there are significant chances in deer population trends, habitat suitability, etc. We also recommend that law enforcement efforts continue to remain vigilant because we found evidence that two deer were harvested illegally during the study period.

Predator Considerations

One of the primary findings of this research was that predation by Florida panthers has increased greatly since the last deer survival studies were conducted in the 1990's. The increased predation rate is the result of panther recovery efforts, which have caused the panther population to grow from less than 40 individuals to approximately 200 individuals over a 30-year period (FWC 2017b). The predation rate and the annual mortality rates that we documented are higher than most published estimates from other regions in the Southeast, and are higher than most previous estimates from South Florida. Although we found no evidence of persistent population declines during the 3-year camera study, low adult survival rates and low fecundity rates in the region suggest that the deer population could decline in the future. Population monitoring efforts should be enacted to alert managers if persistent declines do occur. In the event that population declines are detected, we recommend that managers respond by improving deer habitat quality to increase productivity and survival rates. Although it was not the focus of our research, we suggest that habitat quality could be improved by increasing the use of prescribed fire, mechanical removal of cabbage palm from pine uplands, and chemical control of invasive species such as cogon grass (*Imperata cylindrica*), Brazilian pepper (*Schinus terebinthifolia*), melaleuca (*Melaleuca quinquenervia*), and climbing fern (*Lygodium japonicum*). Such management activities are already being implemented at FPNWR, and they will likely improve range conditions.

Monitoring Recommendations

Our results suggest that camera traps can be used to effectively monitor deer populations in South Florida. We recommend that arrays of 40-60 cameras, spaced by 400-600 m (1 camera per 16-36 ha), be deployed on trails in regions of interest. Cameras can be operated continuously throughout the year or seasonally to reduce costs. For seasonal monitoring, we recommend that cameras be deployed during May and June when movement and associated detection parameters are most consistent. May provides the best month for estimating fawn abundance because most fawns in a cohort have been born and most fawns still have their spots. Recruitment could be estimated in two ways. First, closed population SCR models could be used to estimate the number of fawns alive during eight consecutive 1-week time intervals in May and June. This would provide information about productivity that could be compared among years. Alternatively, the method of Chandler et al. (2018) could be used to estimate the total number of fawns that are born and survive to the recruitment age of 180 days. This method provides a richer perspective on recruitment, but it requires much more data. To use the method of Chandler et al. (2018), cameras should be operated from 1 December – 1 August each year, to collect data from the parturition period until the time when most spots have been lost. Incorporating sampling in June allows for data to be collected on males when most antlers have hardened, allowing for individuals to be uniquely identified. Some males are still in velvet during June, so surveys could continue into July if individual-level encounter history data on males is desired, but male movement behavior also become erratic in July, making it more difficult to fit SCR models.

Data on uniquely identifiable individuals will provide the best estimates of abundance and recruitment because these data provide direct information about individual-level detection probability; however, females cannot be uniquely identified from camera trap photographs, and uniquely identifying adult males and fawns can be very time consuming and expensive. We therefore developed methods that can be used to estimate abundance using simple data on "unmarked" individuals. These "unmarked SCR models" require binary detection data in which each observation indicates if at least one deer was detected at a camera location during a 24-hour period. We recommend estimating density during 14-day periods in May and June. The 14-day periods are short enough to assume population closure for adults, and long enough to obtain enough detection events to estimate density. By estimating density in four consecutive 14-day periods each year, managers will also be able to assess within-season variation in abundance and detection parameters.

Unmarked SCR models will not yield precise estimates of abundance without ancillary information about individual-level detection probability, but we were able to obtain direct information about detection probability using telemetry data. We recommend that our estimates of the detection probability parameters be used in the analysis of future monitoring data on unmarked individuals unless there is reason to believe that activity patterns and home range sizes change substantially over time. In that case, periodic telemetry studies could be conducted to update estimates of the detection probability parameters.

PRIORITY RESEARCH NEEDS

Several lines of research could be pursued to build upon our work and to better inform management activities. We documented that panther predation rates on deer have increased greatly since 1995, while bobcat predation and hunter harvest rates have declined. Deer survival rates were lower and more variable than in most other deer populations in the Southeast. These findings raise concerns about long-term viability of deer populations in South Florida. While we did not find evidence of persistent population declines over the 3-year camera survey period, our research was not conducted over a sufficiently long time period to understand how changes in environmental conditions, the predator community, and deer mortality rates will impact long-term deer population viability in this highly variable system.

Future research could be coordinated with monitoring efforts to collect data at scales appropriate for making long-term forecasts of deer and predator population dynamics. For example, camera data could be combined with additional data on age-specific and location-specific demographic parameters to develop population models needed for viability analysis. Spatially-explicit models could be used to forecast population dynamics in specific regions in South Florida. Ideally, such research would be coordinated with agencies and personnel involved with the management of hydrology and predators in the region. One benefit of coordinating deer research with ongoing panther research would be to answer the question of why deer survival rates increased over the four years of our study. It is possible that the increasing trend in deer survival was the result of declines in the local density of panthers during our investigation, but additional research focusing on both deer and panthers would be needed to assess this possibility and to understand if the upward trend in survival continues.

Managers of deer populations in South Florida are unable to directly control variables such as water levels and predation rates; however, managers can impact deer habitat quality through the use of prescribed fire, herbicide applications, and mechanical treatments. Research could be conducted to identify optimal vegetation management regimes for enhancing deer survival and fecundity rates. Ideally, experimental manipulations would be used to isolate the effects habitat management treatments while avoiding confounding with other factors that affect deer population parameters. This would lead to habitat management recommendations targeting deer productivity, which is likely vital rate most easily influenced by managers. Research on the effects of habitat management practices could be incorporated in a population viability analysis to identify optimal management practices for meeting panther recovery goals while sustaining healthy deer populations in South Florida.

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TABLES

Table 1. Capture data summary by sex, year, area, and collared status (*n* = 294). These data were collected from January 2015 to December 2017 in Big Cypress National Preserve (BCNP) and Florida Panther National Wildlife Refuge (FPNWR).

		2015			**2016**			**2017**			**Grand Total**
		BCNP	**FPNWR**	**Total**	**BCNP**	**FPNWR**	**Total**	**BCNP**	**FPNWR**	**Total**	
Collared											
	Female	65	17	82	52	8	60	20	10	30	172
	Male	33	5	38	25	8	33	8	12	20	91
	Total	98	22	120	77	16	93	28	22	50	263
Ear-tagged only											
	Male	2	1	3	3	2	5	4	4	8	16
	Female	5	2	7	2	1	3	4		4	14
	Fawn		1	1							1
	Total	7	4	11	5	3	8	8	4	12	31
Grand total		105	26	131	82	19	101	36	26	62	294

Table 2. Summary of the causes of white-tailed deer mortality (*n* = 241), sorted by frequency, in in Big Cypress National Preserve and Florida Panther National Wildlife Refuge between January 2015 – December 2018.

Cause of death	**Mortalities**	**Frequency (%)**	**Proportion of mortalities**
Panther	96	39.8	0.72
Bobcat	7	2.9	0.05
Pathology	4	1.7	0.03
Predation (unknown)	4	1.7	0.03
Research induced	3	1.2	0.02
Bear	2	0.8	0.01
Poaching	2	0.8	0.01
Alligator	1	0.4	0.01
Hunting	1	0.4	0.01
Unknown cause of death	14	5.8	0.10
Survived	107	44.4	

Table 3. Survival model selection results using all mortality data for adult male and female white-tailed deer (n = 241) in Big Cypress National Preserve and Florida Panther National Wildlife Refuge from January 2015 – December 2018 in South Florida (*n* = 241). Best supported models are in bold.

Model description	**WAIC**
sex * days to peak fawning + sex * water depth + time	**2055.97**
sex * days to peak fawning + management area + sex * water depth + time	**2055.98**
sex * days to peak fawning + management area + water depth + time	**2056.76**
sex * days to peak fawning + water depth + time	**2057.78**
sex * season + management area + sex * water depth + time	2062.60
sex * season + sex * water depth + time	2062.64
sex * season + water depth + time	2063.44
sex * season + management area + water depth + time	2064.14
sex * days to peak fawning + management area + # of days since last dry + time	2065.14
sex * days to peak fawning + # of days since last dry + time	2065.29
sex + season + management area + time	2068.91
sex * season + management area + time	2069.55
sex + season + time	2069.76
sex * season + time	2069.99
sex * season + management area + time + # of days since last dry	2071.39
sex * season + management area + sex * water depth	2073.40
sex * season + management area + water depth	2074.73
sex * season + sex * water depth	2076.78
sex * season + management area	2078.48
sex	2079.03
sex * season + water depth	2079.71
Intercept only	2080.17
sex * season	2082.76
sex + season	2083.04
season	2083.97

Table 4. Survival model selection results for panther only mortality for adult male and female white-tailed deer (n = 241) in Big Cypress National Preserve and Florida Panther National Wildlife Refuge from January 2015 – December 2018 in South Florida (n = 241). Best supported models are in bold.

Model description	WAIC
sex * days to peak fawning + management area + water depth + time	**1536.53**
sex * days to peak fawning + water depth + time	**1537.11**
sex * days to peak fawning + management area + sex * water depth + time	**1537.21**
sex * days to peak fawning + sex * water depth + time	**1537.49**
sex * days to peak fawning + management area + # of days since last dry + time	1539.10
sex * days to peak fawning + # of days since last dry + time	1539.36
sex * season + management area + sex * water depth + time	1542.18
sex * season + management area + water depth + time	1542.42
sex * season + water depth + time	1542.75
sex * season + sex * water depth + time	1542.86
sex * season + management area + time	1543.59
sex * season + time	1543.81
sex + season + management area + time	1544.53
sex * season + management area + time + # of days since last dry	1544.85
sex + season + time	1545.14
sex * season + management area + sex * water depth	1549.28
sex * season + management area + water depth	1549.89
sex * season + management area	1551.19
sex * season + water depth	1554.33
Intercept only	1554.77
sex * season + sex * water depth	1554.95
sex	1555.01
sex * season	1555.12
season	1555.63
sex + season	1556.50

Table 5. Estimates and 95% credible intervals (CI) for the parameters of deer survival model that included all sources of mortality in Big Cypress National Preserve and Florida Panther National Wildlife Refuge during January 2015 – December 2018 (*n* = 241). This model did not have the lowest WAIC, but it is presented to show the estimates for the management area effect and the sex by season interaction.

	Mean	**Lower CI**	**Upper CI**
Intercept (Female, peak-fawning, BCNP)	6.34	5.98	6.70
Male	-0.43	-0.79	-0.05
Management area - FPNWR	0.38	-0.07	0.88
Water depth (females)	-0.43	-0.70	-0.17
Water depth (males)	-0.09	-0.38	0.18
Days to peak fawning (females)	0.22	0.01	0.43
Days to peak fawning (males)	-0.24	-0.51	0.02
Time	0.43	0.20	0.67

Table 6. Estimates and 95% credible intervals (CI) for the parameters of white-tailed deer survival model that only included mortality attributed to panther in Big Cypress National Preserve and Florida Panther National Wildlife Refuge during January 2015 – December 2018 (*n* = 241). All other sources of mortality were treated as censoring events in this analysis. This model did not have the lowest WAIC, but it is presented to show the estimates for the management area effect and the sex by season interaction.

	Mean	**Lower CI**	**Upper CI**
Intercept (Female, peak-fawning, BCNP)	6.67	6.27	7.1
Male	-0.44	-0.88	-0.02
Management area - FPNWR	0.53	-0.06	1.2
Water depth (females)	-0.31	-0.62	-0.03
Water depth (males)	-0.05	-0.38	0.27
Days to peak fawning (females)	0.34	0.1	0.6
Days to peak fawning (males)	-0.22	-0.54	0.10
Time	0.43	0.16	0.7

Table 7. Mean seasonal home range (km^2) and 95% confidence intervals (CI) for male and female white-tailed deer during hydrological seasons (wet season, May – October; dry season, November – April) in Big Cypress National Preserve and Florida Panther National Wildlife Refuge for all deer qualifying for analyses from January 2015 to October 2018 (*n* = 153).

	Mean	**Lower CI**	**Upper CI**
Florida Panther National Wildlife Refuge			
Male - Wet	3.66	3.35	3.96
Male - Dry	2.03	1.72	2.34
Female - Wet	0.82	0.52	1.12
Female - Dry	0.72	0.42	1.03
Big Cypress National Preserve			
Male - Wet Season	4.90	4.61	5.20
Male - Dry Season	2.72	2.42	3.02
Female - Wet Season	1.10	0.82	1.39
Female - Dry Season	0.97	0.68	1.26

Table 8. Mean seasonal home range (km^2) and 95% confidence intervals (CI) for male and female white-tailed deer for the four biological seasons (fawning, January – March; fawn-rearing, April – June; rut, July – September; post-rut, October – December; n = 188) in Big Cypress National Preserve and Florida Panther National Wildlife Refuge for all deer qualifying for analyses from January 2015 – December 2018.

	Mean	**Lower CI**	**Upper CI**
Florida Panther National Wildlife Refuge			
Male - Fawning	1.65	1.05	2.25
Male - Rearing	2.11	1.51	2.71
Male - Rut	3.95	3.36	4.55
Male - Post-Rut	2.42	1.82	3.02
Female - Fawning	0.70	0.11	1.30
Female - Rearing	0.72	0.12	1.31
Female - Rut	0.76	0.17	1.35
Female - Post-Rut	0.71	0.11	1.30
Big Cypress National Preserve			
Male - Fawning	2.10	1.51	2.69
Male - Rearing	2.69	2.10	3.28
Male - Rut	5.04	4.46	5.63
Male - Post-Rut	3.09	2.50	3.68
Female - Fawning	0.90	0.32	1.48
Female - Rearing	0.92	0.34	1.50
Female - Rut	0.97	0.39	1.55
Female - Post-Rut	0.90	0.33	1.48

Table 9. Sex- and season-specific candidate step selection functions (Formula) for white-tailed deer (F wet season $n = 120$, F dry season $n = 143$; M wet season $n = 70$, M dry season $n = 78$) during January 2015 – December 2018 in Big Cypress National Preserve and Florida Panther National Wildlife Refuge and associated degrees of freedom (df), log-likelihood (logLik), Akaike information criteria (AIC_c), difference in AIC_c from top model (ΔAIC_c), and model weights (W) with respect to distance to habitat and surface water index (SWI). Top sex-season specific models are in bold.

Sex-Season	Model	Formula	df	logLik	AICc	ΔAICc	W
Female							
Wet							
	Global	**(Hammock + Swamp + Marsh + Flatwoods + Prairie + Road + Edge) * SWI + strata(StepID) + cluster(id)**	**15**	**-564867**	**1129764**	**0**	**1**
	4	(Hammock + Swamp + Marsh + Flatwoods + Prairie) * SWI + strata(StepID) + cluster(id)	11	-565027	1130077	313	0
	3	(Hammock + Swamp + Marsh + Flatwoods + Road + Edge) * SWI + strata(StepID) + cluster(id)	13	-565034	1130094	330	0
	5	(Swamp + Marsh + Flatwoods + Road + Edge) * SWI + strata(StepID) + cluster(id)	11	-565069	1130160	396	0
	6	(Swamp + Marsh + Flatwoods + Road) * SWI + strata(StepID) + cluster(id)	9	-565167	1130351	587	0
	7	(Swamp + Marsh + Flatwoods + Edge) * SWI + strata(StepID) + cluster(id)	9	-565173	1130364	600	0
	Null	1 + strata(StepID) + cluster(id)	0	-3068023	6136045	5006281	0
Dry							
	Global	**(Hammock + Swamp + Marsh + Flatwoods + Prairie + Road + Edge) * SWI + strata(StepID) + cluster(id)**	**15**	**-569200**	**1138430**	**0**	**1**
	3	(Hammock + Swamp + Marsh + Flatwoods + Road + Edge) * SWI + strata(StepID) + cluster(id)	13	-569318	1138663	233	0
	5	(Swamp + Marsh + Flatwoods + Road + Edge) * SWI + strata(StepID) +	11	-569323	1138668	238	0

		cluster(id)					
	4	(Hammock + Swamp + Marsh + Flatwoods + Prairie) * SWI + strata(StepID) + cluster(id)	11	-569392	1138806	376	0
	7	(Swamp + Marsh + Flatwoods + Edge) * SWI + strata(StepID) + cluster(id)	9	-569400	1138818	389	0
	6	(Swamp + Marsh + Flatwoods + Road) * SWI + strata(StepID) + cluster(id)	9	-569463	1138943	513	0
	Null	1 + strata(StepID) + cluster(id)	0	-3087019	6174038	5035609	0
Male							
Wet							
	Global	**(Hammock + Swamp + Marsh + Flatwoods + Prairie + Road + Edge) * SWI + strata(StepID) + cluster(id)**	**15**	**-231514**	**463058**	**0**	**1**
	3	(Hammock + Swamp + Marsh + Flatwoods + Road + Edge) * SWI + strata(StepID) + cluster(id)	13	-231563	463151	93	0
	5	(Swamp + Marsh + Flatwoods + Road + Edge) * SWI + strata(StepID) + cluster(id)	11	-231567	463155	97	0
	4	(Hammock + Swamp + Marsh + Flatwoods + Prairie) * SWI + strata(StepID) + cluster(id)	11	-231572	463166	108	0
	6	(Swamp + Marsh + Flatwoods + Road) * SWI + strata(StepID) + cluster(id)	9	-231580	463177	119	0
	7	(Swamp + Marsh + Flatwoods + Edge) * SWI + strata(StepID) + cluster(id)	9	-231633	463285	226	0
	Null	1 + strata(StepID) + cluster(id)	0	-1181364	2362727	1899669	0
Dry							
	Global	**(Hammock + Swamp + Marsh + Flatwoods + Prairie + Road + Edge) * SWI + strata(StepID) + cluster(id)**	**15**	**-237203**	**474435**	**0**	**1**
	3	(Hammock + Swamp + Marsh + Flatwoods + Road + Edge) * SWI + strata(StepID) + cluster(id)	13	-237241	474509	74	0
	5	(Swamp + Marsh + Flatwoods + Road + Edge) * SWI + strata(StepID) + cluster(id)	11	-237249	474521	86	0
	7	(Swamp + Marsh + Flatwoods + Edge) * SWI + strata(StepID) +	9	-237301	474620	185	0

	cluster(id)					
4	(Hammock + Swamp + Marsh + Flatwoods + Prairie) * SWI + strata(StepID) + cluster(id)	11	-237306	474634	199	0
6	(Swamp + Marsh + Flatwoods + Road) * SWI + strata(StepID) + cluster(id)	9	-237331	474681	246	0
Null	1 + strata(StepID) + cluster(id)	0	-1211286	2422571	1948136	0

Table 10. Sex- and season-specific candidate step selection functions (Formula) for white-tailed deer (F wet season *n* = 120, F dry season *n* = 143; M wet season *n* = 70, M dry season *n* = 78) during January 2015 - December 2018 in Big Cypress National Preserve and Florida Panther National Wildlife Refuge examining habitat selection in regards to distance to habitat, surface water (SWI), and panther activity rates (PAR) and associated degrees of freedom (df), log-likelihood (logLik), corrected AIC (AICc), and model weight (W). Top sex-season specific models are in bold.

Sex-Season Model	Formula	df	logLik	AICc	ΔAICc	W
Female						
Wet						
Global	**(Hammock + Swamp + Marsh + Flatwoods + Prairie + Road + Edge + SWI) * PAR + strata(StepID) + cluster(id)**	**17**	**-248721.44**	**497476.88**	**0.00**	**1**
4	(Hammock + Swamp + Marsh + Flatwoods + Prairie + SWI)*PAR + strata(StepID) + cluster(id)	13	-248798.52	497623.04	146.15	0
3	(Hammock + Swamp + Marsh + Flatwoods + Road + Edge + SWI)*PAR + strata(StepID) + cluster(id)	15	-248860.99	497751.99	275.11	0
5	(Swamp + Marsh + Flatwoods + Road + Edge + SWI) * PAR + strata(StepID) + cluster(id)	13	-248875.75	497777.51	300.63	0
6	(Swamp + Marsh + Flatwoods + Road + SWI) * PAR + strata(StepID) + cluster(id)	11	-248890.04	497802.09	325.21	0
12	(Marsh + Prairie + Road + Edge + SWI)*PAR + strata(StepID) + cluster(id)	11	-248923.32	497868.65	391.77	0
7	(Swamp + Marsh + Flatwoods + Edge + SWI) * PAR + strata(StepID) + cluster(id)	11	-248967.32	497956.65	479.77	0
11	(Hammock + Flatwoods + Prairie + Road + Edge + SWI)*PAR + strata(StepID) + cluster(id)	13	-249119.39	498264.78	787.90	0
13	(Swamp + Flatwoods + Road + Edge + SWI)*PAR + strata(StepID) + cluster(id)	11	-249247.08	498516.16	1039.28	0
10	(Hammock + Swamp + Prairie + Road + Edge + SWI)*PAR +	13	-249271.43	498568.86	1091.97	0

		strata(StepID) + cluster(id)					
	14	(Flatwoods + Edge + SWI)*PAR + strata(StepID) + cluster(id)	7	-249368.04	498750.08	1273.20	0
	15	(Swamp + Edge + Road + SWI)*PAR + strata(StepID) + cluster(id)	9	-249383.87	498785.75	1308.86	0
	9	(Road + Edge + SWI) * PAR + strata(StepID) + cluster(id)	7	-249413.93	498841.85	1364.97	0
	8	Hammock + Swamp + Marsh + Flatwoods + Prairie + Road + edge + SWI + PAR strata(StepID) + cluster(id)	8	-258163.13	516342.27	18865.39	0
	Null	1 + strata(StepID) + cluster(id)	0	-1329563.74	2659127.48	2161650.60	0
Dry							
	Global	**(Hammock + Swamp + Marsh + Flatwoods + Prairie + Road + Edge + SWI) * PAR + strata(StepID) + cluster(id)**	**17**	**-241493.99**	**483021.98**	**0.00**	**0.92**
	4	(Hammock + Swamp + Marsh + Flatwoods + Prairie + SWI)*PAR + strata(StepID) + cluster(id)	13	-241500.47	483026.95	4.97	0.08
	3	(Hammock + Swamp + Marsh + Flatwoods + Road + Edge + SWI)*PAR + strata(StepID) + cluster(id)	15	-241594.22	483218.45	196.47	0.00
	5	(Swamp + Marsh + Flatwoods + Road + Edge + SWI) * PAR + strata(StepID) + cluster(id)	13	-241612.19	483250.38	228.40	0.00
	7	(Swamp + Marsh + Flatwoods + Edge + SWI) * PAR + strata(StepID) + cluster(id)	11	-241615.75	483253.50	231.52	0.00
	6	(Swamp + Marsh + Flatwoods + Road + SWI) * PAR + strata(StepID) + cluster(id)	11	-241622.04	483266.08	244.10	0.00
	12	(Marsh + Prairie + Road + Edge + SWI)*PAR + strata(StepID) + cluster(id)	11	-241638.46	483298.93	276.94	0.00
	11	(Hammock + Flatwoods + Prairie + Road + Edge + SWI)*PAR + strata(StepID) + cluster(id)	13	-241774.81	483575.62	553.63	0.00
	10	(Hammock + Swamp + Prairie + Road + Edge + SWI)*PAR + strata(StepID) + cluster(id)	13	-241837.79	483701.58	679.60	0.00
	13	(Swamp + Flatwoods + Road + Edge + SWI)*PAR + strata(StepID) + cluster(id)	11	-241850.95	483723.90	701.91	0.00
	14	(Flatwoods + Edge + SWI)*PAR + strata(StepID) + cluster(id)	7	-241895.66	483805.31	783.33	0.00
	15	(Swamp + Edge + Road + SWI)*PAR + strata(StepID) +	9	-241934.57	483887.14	865.15	0.00

		cluster(id)					
	9	(Road + Edge + SWI) * PAR + strata(StepID) + cluster(id)	7	-241988.39	483990.79	968.81	0.00
	8	Hammock + Swamp + Marsh + Flatwoods + Prairie + Road + edge + SWI + PAR strata(StepID) + cluster(id)	8	-249700.82	499417.64	16395.66	0.00
	Null	1 + strata(StepID) + cluster(id)	0	-1280707.64	2561415.28	2078393.30	0.00
Male Wet							
	Global	**(Hammock + Swamp + Marsh + Flatwoods + Prairie + Road + Edge + SWI) * PAR + strata(StepID) + cluster(id)**	**17**	**-83017.62**	**166069.26**	**0.00**	**1**
	4	(Hammock + Swamp + Marsh + Flatwoods + Prairie + SWI)*PAR + strata(StepID) + cluster(id)	13	-83026.31	166078.63	9.37	0
	3	(Hammock + Swamp + Marsh + Flatwoods + Road + Edge + SWI)*PAR + strata(StepID) + cluster(id)	15	-83029.86	166089.74	20.48	0
	12	(Marsh + Prairie + Road + Edge + SWI)*PAR + strata(StepID) + cluster(id)	11	-83068.73	166159.46	90.21	0
	5	(Swamp + Marsh + Flatwoods + Road + Edge + SWI) * PAR + strata(StepID) + cluster(id)	13	-83078.59	166183.19	113.93	0
	6	(Swamp + Marsh + Flatwoods + Road + SWI) * PAR + strata(StepID) + cluster(id)	11	-83084.12	166190.26	121.00	0
	7	(Swamp + Marsh + Flatwoods + Edge + SWI) * PAR + strata(StepID) + cluster(id)	11	-83085.20	166192.41	123.15	0
	11	(Hammock + Flatwoods + Prairie + Road + Edge + SWI)*PAR + strata(StepID) + cluster(id)	13	-83130.28	166286.58	217.32	0
	10	(Hammock + Swamp + Prairie + Road + Edge + SWI)*PAR + strata(StepID) + cluster(id)	13	-83142.13	166310.28	241.02	0
	13	(Swamp + Flatwoods + Road + Edge + SWI)*PAR + strata(StepID) + cluster(id)	11	-83192.53	166407.08	337.82	0
	14	(Flatwoods + Edge + SWI)*PAR + strata(StepID) + cluster(id)	7	-83202.33	166418.66	349.40	0
	9	(Road + Edge + SWI) * PAR + strata(StepID) + cluster(id)	7	-83202.70	166419.40	350.14	0
	15	(Swamp + Edge + Road + SWI)*PAR + strata(StepID) + cluster(id)	9	-83202.36	166422.72	353.47	0

	8	Hammock + Swamp + Marsh + Flatwoods + Prairie + Road + edge + SWI + PAR strata(StepID) + cluster(id)	8	-101084.57	202185.14	36115.88	0
Dry							
	3	**(Hammock + Swamp + Marsh + Flatwoods + Road + Edge + SWI)*PAR + strata(StepID) + cluster(id)**	**15**	**-79654.92**	**159339.85**	**0.00**	**0.70**
	Global	(Hammock + Swamp + Marsh + Flatwoods + Prairie + Road + Edge + SWI) * PAR + strata(StepID) + cluster(id)	17	-79653.76	159341.54	1.69	0.30
	5	(Swamp + Marsh + Flatwoods + Road + Edge + SWI) * PAR + strata(StepID) + cluster(id)	13	-79702.77	159431.55	91.70	0.00
	4	(Hammock + Swamp + Marsh + Flatwoods + Prairie + SWI)*PAR + strata(StepID) + cluster(id)	13	-79711.31	159448.63	108.78	0.00
	10	(Hammock + Swamp + Prairie + Road + Edge + SWI)*PAR + strata(StepID) + cluster(id)	13	-79715.15	159456.32	116.47	0.00
	7	(Swamp + Marsh + Flatwoods + Edge + SWI) * PAR + strata(StepID) + cluster(id)	11	-79728.28	159478.58	138.73	0.00
	11	(Hammock + Flatwoods + Prairie + Road + Edge + SWI)*PAR + strata(StepID) + cluster(id)	13	-79755.99	159538.00	198.15	0.00
	6	(Swamp + Marsh + Flatwoods + Road + SWI) * PAR + strata(StepID) + cluster(id)	11	-79760.04	159542.10	202.25	0.00
	13	(Swamp + Flatwoods + Road + Edge + SWI)*PAR + strata(StepID) + cluster(id)	11	-79766.76	159555.53	215.68	0.00
	15	(Swamp + Edge + Road + SWI)*PAR + strata(StepID) + cluster(id)	9	-79768.92	159555.85	216.00	0.00
	12	(Marsh + Prairie + Road + Edge + SWI)*PAR + strata(StepID) + cluster(id)	11	-79770.08	159562.16	222.31	0.00
	9	(Road + Edge + SWI) * PAR + strata(StepID) + cluster(id)	7	-79817.14	159648.29	308.44	0.00
	14	(Flatwoods + Edge + SWI)*PAR + strata(StepID) + cluster(id)	7	-79838.78	159691.57	351.72	0.00
	8	Hammock + Swamp + Marsh + Flatwoods + Prairie + Road + edge + SWI + PAR strata(StepID) + cluster(id)	8	-101228.69	202473.38	43133.54	0.00
	Null	1 + strata(StepID) + cluster(id)	0	-1280707.64	2561415.28	2402075.44	0.00

Table 11. Sex- and season-specific candidate step selection functions (Formula) for white-tailed deer (F wet season *n* = 120, F dry season *n* = 143; M wet season *n* = 70, M dry season *n* = 78) during January 2015 – December 2018 in Big Cypress National Preserve and Florida Panther National Wildlife Refuge examining habitat selection in regards distance to habitat, surface water (SWI), and human activity rates (HAR) and associated degrees of freedom (df), log-likelihood (logLik), corrected AIC (AICc), and model weight (W). Top sex-season specific models are in bold.

Sex-Season Model			**Formula**	**df**	**logLik**	**AICc**	**ΔAICc**	**W**
Female								
	Wet							
		Global	**(Hammock + Swamp + Marsh + Flatwoods + Prairie + Road + Edge + SWI) * HAR + strata(StepID) + cluster(id)**	**17**	**-248782.06**	**497598.13**	**0.00**	**1**
		4	(Hammock + Swamp + Marsh + Flatwoods + Prairie + SWI)*HAR + strata(StepID) + cluster(id)	13	-248861.24	497748.49	150.36	0
		12	(Marsh + Prairie + Road + Edge + SWI)*HAR + strata(StepID) + cluster(id)	11	-248923.32	497868.65	270.52	0
		3	(Hammock + Swamp + Marsh + Flatwoods + Road + Edge + SWI)*HAR + strata(StepID) + cluster(id)	15	-248926.58	497883.17	285.04	0
		5	(Swamp + Marsh + Flatwoods + Road + Edge + SWI) * HAR + strata(StepID) + cluster(id)	13	-248940.68	497907.35	309.22	0
		6	(Swamp + Marsh + Flatwoods + Road + SWI) * HAR + strata(StepID) + cluster(id)	11	-248954.18	497930.36	332.23	0
		7	(Swamp + Marsh + Flatwoods + Edge + SWI) * HAR + strata(StepID) + cluster(id)	11	-249034.02	498090.05	491.92	0
		11	(Hammock + Flatwoods + Prairie + Road + Edge + SWI)*HAR + strata(StepID) + cluster(id)	13	-249181.37	498388.74	790.61	0
		13	(Swamp + Flatwoods + Road + Edge + SWI)*HAR + strata(StepID) + cluster(id)	11	-249247.08	498516.16	918.03	0
		10	(Hammock + Swamp + Prairie + Road + Edge + SWI)*HAR +	13	-249333.49	498692.99	1094.86	0

	strata(StepID) + cluster(id)					
14	(Flatwoods + Edge + SWI)*HAR + strata(StepID) + cluster(id)	7	-249368.04	498750.08	1151.95	0
15	(Swamp + Edge + Road + SWI)*HAR + strata(StepID) + cluster(id)	9	-249383.87	498785.75	1187.61	0
9	(Road + Edge + SWI) * HAR + strata(StepID) + cluster(id)	7	-249482.79	498979.58	1381.45	0
8	Hammock + Swamp + Marsh + Flatwoods + Prairie + Road + edge + SWI + HAR strata(StepID) + cluster(id)	8	-258163.13	516342.27	18744.14	0
Null	1 + strata(StepID) + cluster(id)	0	-1329563.74	2659127.48	2161529.35	0
Dry						
Global	**(Hammock + Swamp + Marsh + Flatwoods + Prairie + Road + Edge + SWI) * HAR + strata(StepID) + cluster(id)**	**17**	**-241478.01**	**482990.03**	**0.00**	**0.91**
4	(Hammock + Swamp + Marsh + Flatwoods + Prairie + SWI)*HAR + strata(StepID) + cluster(id)	13	-241484.31	482994.62	4.59	0.09
3	(Hammock + Swamp + Marsh + Flatwoods + Road + Edge + SWI)*HAR + strata(StepID) + cluster(id)	15	-241582.90	483195.82	205.78	0.00
5	(Swamp + Marsh + Flatwoods + Road + Edge + SWI) * HAR + strata(StepID) + cluster(id)	13	-241601.56	483229.13	239.10	0.00
7	(Swamp + Marsh + Flatwoods + Edge + SWI) * HAR + strata(StepID) + cluster(id)	11	-241604.98	483231.96	241.93	0.00
6	(Swamp + Marsh + Flatwoods + Road + SWI) * HAR + strata(StepID) + cluster(id)	11	-241611.21	483244.42	254.39	0.00
12	(Marsh + Prairie + Road + Edge + SWI)*HAR + strata(StepID) + cluster(id)	11	-241638.46	483298.93	308.90	0.00
11	(Hammock + Flatwoods + Prairie + Road + Edge + SWI)*HAR + strata(StepID) + cluster(id)	13	-241757.32	483540.64	550.61	0.00
10	(Hammock + Swamp + Prairie + Road + Edge + SWI)*HAR + strata(StepID) + cluster(id)	13	-241820.64	483667.28	677.25	0.00
13	(Swamp + Flatwoods + Road + Edge + SWI)*HAR + strata(StepID) + cluster(id)	11	-241850.95	483723.90	733.87	0.00
14	(Flatwoods + Edge + SWI)*HAR + strata(StepID) + cluster(id)	7	-241895.66	483805.31	815.28	0.00
15	(Swamp + Edge + Road + SWI)*HAR + strata(StepID) +	9	-241934.57	483887.14	897.11	0.00

			cluster(id)					
		9	(Road + Edge + SWI) * HAR + strata(StepID) + cluster(id)	7	-241978.14	483970.28	980.25	0.00
		8	Hammock + Swamp + Marsh + Flatwoods + Prairie + Road + edge + SWI + HAR strata(StepID) + cluster(id)	8	-249700.82	499417.64	16427.61	0.00
		Null	1 + strata(StepID) + cluster(id)	0	-1280707.64	2561415.28	2078425.25	0.00
Male								
	Wet							
		Global	**(Hammock + Swamp + Marsh + Flatwoods + Prairie + Road + Edge + SWI) * HAR + strata(StepID) + cluster(id)**	**17**	**-83017.80**	**166069.62**	**0.00**	**1**
		4	(Hammock + Swamp + Marsh + Flatwoods + Prairie + SWI)*HAR + strata(StepID) + cluster(id)	13	-83027.42	166080.84	11.22	0
		3	(Hammock + Swamp + Marsh + Flatwoods + Road + Edge + SWI)*HAR + strata(StepID) + cluster(id)	15	-83029.64	166089.30	19.68	0
		12	(Marsh + Prairie + Road + Edge + SWI)*HAR + strata(StepID) + cluster(id)	11	-83068.73	166159.46	89.84	0
		5	(Swamp + Marsh + Flatwoods + Road + Edge + SWI) * HAR + strata(StepID) + cluster(id)	13	-83077.26	166180.52	110.90	0
		6	(Swamp + Marsh + Flatwoods + Road + SWI) * HAR + strata(StepID) + cluster(id)	11	-83082.65	166187.30	117.68	0
		7	(Swamp + Marsh + Flatwoods + Edge + SWI) * HAR + strata(StepID) + cluster(id)	11	-83084.48	166190.96	121.34	0
		11	(Hammock + Flatwoods + Prairie + Road + Edge + SWI)*HAR + strata(StepID) + cluster(id)	13	-83131.06	166288.14	218.52	0
		10	(Hammock + Swamp + Prairie + Road + Edge + SWI)*HAR + strata(StepID) + cluster(id)	13	-83143.12	166312.26	242.64	0
		13	(Swamp + Flatwoods + Road + Edge + SWI)*HAR + strata(StepID) + cluster(id)	11	-83192.53	166407.08	337.45	0
		9	(Road + Edge + SWI) * HAR + strata(StepID) + cluster(id)	7	-83202.07	166418.15	348.53	0
		14	(Flatwoods + Edge + SWI)*HAR + strata(StepID) + cluster(id)	7	-83202.33	166418.66	349.04	0
		15	(Swamp + Edge + Road + SWI)*HAR + strata(StepID) + cluster(id)	9	-83202.36	166422.72	353.10	0

	8	Hammock + Swamp + Marsh + Flatwoods + Prairie + Road + edge + SWI + HAR strata(StepID) + cluster(id)	8	-101084.57	202185.14	36115.52	0
	Null	1 + strata(StepID) + cluster(id)	0	-485334.09	970668.18	804598.56	0
Dry							
	3	**(Hammock + Swamp + Marsh + Flatwoods + Road + Edge + SWI)*HAR + strata(StepID) + cluster(id)**	**15**	**-79636.89**	**159303.80**	**0.00**	**0.78**
	Global	(Hammock + Swamp + Marsh + Flatwoods + Prairie + Road + Edge + SWI) * HAR + strata(StepID) + cluster(id)	17	-79636.14	159306.31	2.50	0.22
	5	(Swamp + Marsh + Flatwoods + Road + Edge + SWI) * HAR + strata(StepID) + cluster(id)	13	-79684.18	159394.38	90.58	0.00
	4	(Hammock + Swamp + Marsh + Flatwoods + Prairie + SWI)*HAR + strata(StepID) + cluster(id)	13	-79693.89	159413.79	109.99	0.00
	10	(Hammock + Swamp + Prairie + Road + Edge + SWI)*HAR + strata(StepID) + cluster(id)	13	-79698.74	159423.49	119.68	0.00
	7	(Swamp + Marsh + Flatwoods + Edge + SWI) * HAR + strata(StepID) + cluster(id)	11	-79710.00	159442.01	138.21	0.00
	11	(Hammock + Flatwoods + Prairie + Road + Edge + SWI)*HAR + strata(StepID) + cluster(id)	13	-79738.45	159502.92	199.12	0.00
	6	(Swamp + Marsh + Flatwoods + Road + SWI) * HAR + strata(StepID) + cluster(id)	11	-79741.14	159504.29	200.48	0.00
	13	(Swamp + Flatwoods + Road + Edge + SWI)*HAR + strata(StepID) + cluster(id)	11	-79766.76	159555.53	251.72	0.00
	15	(Swamp + Edge + Road + SWI)*HAR + strata(StepID) + cluster(id)	9	-79768.92	159555.85	252.05	0.00
	12	(Marsh + Prairie + Road + Edge + SWI)*HAR + strata(StepID) + cluster(id)	11	-79770.08	159562.16	258.36	0.00
	9	(Road + Edge + SWI) * HAR + strata(StepID) + cluster(id)	7	-79798.96	159611.93	308.13	0.00
	14	(Flatwoods + Edge + SWI)*HAR + strata(StepID) + cluster(id)	7	-79838.78	159691.57	387.77	0.00
	8	Hammock + Swamp + Marsh + Flatwoods + Prairie + Road + edge + SWI + HAR strata(StepID) + cluster(id)	8	-101228.69	202473.38	43169.58	0.00
	Null	1 + strata(StepID) + cluster(id)	0	-1280707.64	2561415.28	2402111.48	0.00

Table 12. Seasonal selection coefficients (Est), standard errors (SE), F-statistics (F-stat), 95% confidence intervals (Lower, Upper), and p-values from step selection functions fit to seasonal data subsets for all male white-tailed deer eligible for analyses (wet season *n* = 70; dry season *n* = 78) during January 2015 - – December 2018 Big Cypress National Preserve and Florida Panther National Wildlife Refuge with respect to distance to habitat and surface water index (SWI).

Season	**Term**	**Est**	**SE**	**F-Stat**	**Lower**	**Upper**	**p-value**
Dry							
	Hammock	-0.020	0.007	-0.821	-0.069	0.028	0.412
	Swamp	-0.011	0.006	-0.468	-0.055	0.034	0.640
	Marsh	-0.103	0.006	-3.400	-0.162	-0.044	0.001
	Flatwoods	-0.037	0.006	-1.276	-0.093	0.020	0.202
	Prairie	-0.044	0.005	-2.016	-0.087	-0.001	0.044
	Road	-0.022	0.005	-0.715	-0.081	0.038	0.475
	Edge	0.012	0.006	0.508	-0.034	0.058	0.611
	SWI	-0.012	0.004	-0.806	-0.040	0.017	0.420
	Hammock:SWI	-0.013	0.004	-1.204	-0.034	0.008	0.229
	Swamp:SWI	0.027	0.004	2.853	0.008	0.045	0.004
	Marsh:SWI	0.010	0.004	0.727	-0.017	0.037	0.467
	Flatwoods:SWI	-0.025	0.004	-2.224	-0.047	-0.003	0.026
	Prairie:SWI	-0.015	0.004	-1.499	-0.034	0.005	0.134
	Road:SWI	-0.032	0.003	-2.741	-0.054	-0.009	0.006
	Edge:SWI	-0.040	0.004	-3.291	-0.063	-0.016	0.001
Wet							
	Hammock	0.011	0.007	0.484	-0.034	0.056	0.629
	Swamp	0.024	0.005	1.261	-0.013	0.062	0.207
	Marsh	-0.168	0.006	-6.808	-0.217	-0.120	<0.001
	Flatwoods	-0.008	0.006	-0.309	-0.056	0.041	0.757
	Prairie	-0.042	0.005	-2.490	-0.075	-0.009	0.013
	Road	-0.036	0.005	-1.532	-0.083	0.010	0.126
	Edge	-0.010	0.005	-0.421	-0.055	0.035	0.674
	SWI	-0.056	0.004	-4.038	-0.083	-0.029	<0.001
	Hammock:SWI	-0.005	0.003	-0.469	-0.027	0.017	0.639
	Swamp:SWI	0.019	0.003	2.556	0.005	0.034	0.011
	Marsh:SWI	-0.008	0.004	-0.655	-0.031	0.016	0.513
	Flatwoods:SWI	-0.011	0.003	-1.181	-0.030	0.007	0.238
	Prairie:SWI	-0.016	0.003	-2.758	-0.027	-0.005	0.006
	Road:SWI	-0.022	0.003	-2.101	-0.043	-0.002	0.036
	Edge:SWI	-0.012	0.003	-1.220	-0.032	0.007	0.222

Table 13. Seasonal selection coefficients (Est), standard errors (SE), F-statistics (F-Stat), 95% confidence intervals (Lower, Upper), and p-values from four step selection functions fit to seasonal data subsets for all female white-tailed deer eligible for analyses (wet season *n* = 120; dry season *n* = 143) during January 2015 – December 2018 in Big Cypress National Preserve and Florida Panther National Wildlife Refuge with respect to distance to habitat and surface water index (SWI).

Season	**Term**	**Est**	**SE**	**F-Stat**	**Lower**	**Upper**	**p-value**
Dry							
	Hammock	0.006	0.021	0.285	-0.036	0.048	0.776
	Swamp	0.057	0.021	2.650	0.015	0.099	0.008
	Marsh	-0.052	0.017	-3.116	-0.084	-0.019	0.002
	Flatwoods	-0.045	0.024	-1.863	-0.093	0.002	0.062
	Prairie	-0.049	0.018	-2.765	-0.084	-0.014	0.006
	Road	0.011	0.017	0.659	-0.022	0.045	0.510
	Edge	-0.034	0.017	-2.041	-0.066	-0.001	0.041
	SWI	-0.004	0.013	-0.270	-0.030	0.023	0.787
	Hammock:SWI	-0.003	0.007	-0.452	-0.017	0.011	0.651
	Swamp:SWI	0.023	0.006	3.719	0.011	0.036	<0.001
	Marsh:SWI	0.052	0.009	5.512	0.033	0.070	<0.001
	Flatwoods:SWI	-0.006	0.009	-0.680	-0.025	0.012	0.497
	Prairie:SWI	-0.005	0.008	-0.676	-0.021	0.010	0.499
	Road:SWI	-0.024	0.006	-3.699	-0.036	-0.011	<0.001
	Edge:SWI	-0.033	0.010	-3.432	-0.052	-0.014	0.001
Wet							
	Hammock	0.010	0.021	0.491	-0.031	0.052	0.624
	Swamp	0.039	0.022	1.768	-0.004	0.082	0.077
	Marsh	-0.103	0.018	-5.785	-0.138	-0.068	<0.001
	Flatwoods	-0.077	0.025	-3.051	-0.127	-0.028	0.002
	Prairie	-0.050	0.019	-2.692	-0.087	-0.014	0.007
	Road	-0.023	0.020	-1.142	-0.061	0.016	0.253
	Edge	-0.025	0.020	-1.272	-0.064	0.014	0.203
	SWI	-0.057	0.013	-4.302	-0.083	-0.031	<0.001
	Hammock:SWI	0.016	0.008	2.109	0.001	0.031	0.035
	Swamp:SWI	0.019	0.008	2.394	0.003	0.034	0.017
	Marsh:SWI	0.067	0.012	5.854	0.045	0.090	<0.001
	Flatwoods:SWI	-0.007	0.010	-0.663	-0.026	0.013	0.507
	Prairie:SWI	-0.018	0.009	-1.947	-0.036	0.000	0.052
	Road:SWI	-0.021	0.008	-2.496	-0.038	-0.005	0.013
	Edge:SWI	-0.025	0.012	-2.149	-0.048	-0.002	0.032

Table 14. Seasonal selection coefficients (Est), standard errors (SE), F-statistics (F-stat), 95% confidence intervals (Lower, Upper), and p-values from step selection functions fit to seasonal data subsets for all male white-tailed deer eligible for analyses (wet season *n* = 70; dry season *n* = 78) during January 2015 – December 2018 in Big Cypress National Preserve and Florida Panther National Wildlife Refuge with respect to distance to habitat, surface water (SWI), and panther activity rates (PAR).

Season	**Term**	**Est**	**SE**	**F-Stat**	**Lower**	**Upper**	**p-value**
Dry							
	Hammock	0.105	0.011	2.981	0.036	0.174	0.003
	Swamp	-0.099	0.009	-3.649	-0.153	-0.046	<0.001
	Marsh	-0.106	0.010	-3.057	-0.175	-0.038	0.002
	Flatwoods	0.025	0.009	0.759	-0.039	0.088	0.448
	Road	0.058	0.008	1.510	-0.017	0.134	0.131
	Edge	0.079	0.009	2.601	0.020	0.139	0.009
	SWI	-0.049	0.007	-2.212	-0.093	-0.006	0.027
	PAR	-0.042	0.018	-1.919	-0.084	0.001	0.055
	Hammock:PAR	-0.017	0.009	-1.923	-0.034	0.000	0.054
	Swamp:PAR	-0.003	0.009	-0.476	-0.017	0.010	0.634
	Marsh:PAR	-0.012	0.010	-0.721	-0.043	0.020	0.471
	Flatwoods:PAR	-0.001	0.010	-0.078	-0.022	0.021	0.938
	Road:PAR	0.001	0.009	0.127	-0.012	0.013	0.899
	Edge:PAR	0.004	0.009	0.570	-0.011	0.020	0.569
	SWI:PAR	-0.004	0.007	-0.318	-0.026	0.019	0.750
Wet							
	Hammock	0.098	0.011	3.154	0.037	0.160	0.002
	Swamp	-0.011	0.008	-0.487	-0.055	0.033	0.626
	Marsh	-0.142	0.010	-4.289	-0.206	-0.077	<0.001
	Flatwoods	0.039	0.008	1.013	-0.037	0.115	0.311
	Prairie	-0.039	0.009	-1.401	-0.095	0.016	0.161
	Road	-0.027	0.008	-0.848	-0.089	0.035	0.396
	Edge	0.015	0.008	0.451	-0.050	0.080	0.652
	SWI	-0.059	0.006	-3.147	-0.096	-0.022	0.002
	PAR	-0.002	0.016	-0.105	-0.031	0.028	0.916
	Hammock:PAR	-0.011	0.009	-1.078	-0.030	0.009	0.281
	Swamp:PAR	0.006	0.008	0.675	-0.011	0.023	0.499
	Marsh:PAR	0.008	0.009	0.816	-0.012	0.028	0.415
	Flatwoods:PAR	-0.005	0.009	-0.512	-0.024	0.014	0.609

Prairie:PAR	0.014	0.008	2.049	0.001	0.026	0.040
Road:PAR	0.007	0.009	1.056	-0.006	0.021	0.291
Edge:PAR	0.011	0.008	1.408	-0.004	0.023	0.159
SWI:PAR	0.028	0.006	5.192	0.017	0.039	<0.001

Table 15. Seasonal selection coefficients (Est), standard errors (SE), F-statistics (F-stat), 95% confidence intervals (Lower, Upper), and p-values from step selection functions fit to seasonal data subsets for all female white-tailed deer eligible for analyses (wet season *n* = 120; dry season *n* = 143) during January 2015 – December 2018 in Big Cypress National Preserve and Florida Panther National Wildlife Refuge with respect to habitat, surface water (SWI), and panther activity rates (PAR).

Season	**Term**	**Est**	**SE**	**F-Stat**	**Lower**	**Upper**	**p-value**
Dry							
	Hammock	0.025	0.006	0.831	-0.035	0.085	0.406
	Swamp	0.023	0.005	0.784	-0.035	0.082	0.433
	Marsh	-0.114	0.005	-4.700	-0.162	-0.067	<0.001
	Flatwoods	-0.083	0.006	-2.752	-0.143	-0.024	0.006
	Prairie	-0.071	0.005	-2.697	-0.123	-0.019	0.007
	Road	-0.004	0.005	-0.173	-0.045	0.038	0.863
	Edge	-0.017	0.005	-0.728	-0.062	0.029	0.467
	SWI	-0.018	0.004	-0.957	-0.056	0.019	0.339
	PAR	-0.011	0.010	-0.845	-0.038	0.015	0.398
	Hammock:PAR	0.002	0.005	0.350	-0.009	0.013	0.727
	Swamp:PAR	-0.001	0.005	-0.199	-0.012	0.010	0.842
	Marsh:PAR	0.007	0.005	1.224	-0.004	0.018	0.221
	Flatwoods:PAR	0.006	0.006	0.911	-0.007	0.020	0.363
	Prairie:PAR	-0.005	0.005	-0.829	-0.015	0.006	0.407
	Road:PAR	0.007	0.005	1.132	-0.005	0.019	0.257
	Edge:PAR	-0.004	0.005	-0.717	-0.015	0.007	0.473
	SWI:PAR	0.006	0.004	1.376	-0.003	0.015	0.169
Wet							
	Hammock	0.020	0.006	0.610	-0.044	0.083	0.542
	Swamp	0.003	0.005	0.089	-0.058	0.063	0.929
	Marsh	-0.142	0.005	-5.736	-0.191	-0.094	<0.001
	Flatwoods	-0.106	0.006	-3.289	-0.168	-0.043	0.001
	Prairie	-0.084	0.005	-2.944	-0.140	-0.028	0.003
	Road	-0.056	0.005	-2.202	-0.106	-0.006	0.028
	Edge	-0.017	0.005	-0.592	-0.074	0.039	0.554
	SWI	-0.058	0.004	-3.125	-0.094	-0.022	0.002
	PAR	-0.014	0.010	-1.148	-0.037	0.010	0.251
	Hammock:PAR	-0.006	0.005	-1.315	-0.015	0.003	0.189
	Swamp:PAR	0.001	0.005	0.256	-0.008	0.010	0.798

Marsh:PAR	-0.001	0.005	-0.223	-0.012	0.009	0.823
Flatwoods:PAR	0.009	0.006	1.536	-0.002	0.020	0.125
Prairie:PAR	0.001	0.005	0.281	-0.009	0.012	0.779
Road:PAR	0.004	0.005	0.740	-0.007	0.016	0.459
Edge:PAR	0.006	0.005	1.250	-0.003	0.016	0.211
SWI:PAR	0.025	0.004	4.884	0.015	0.034	<0.001

Table 16. Seasonal selection coefficients (Est), standard errors (SE), F-statistics (F-stat), 95% confidence intervals (Lower, Upper), and p-values from step selection functions fit to seasonal data subsets for all male white-tailed deer eligible for analyses (wet season *n* = 70; dry season *n* = 78) during January 2015 – December 2018 in Big Cypress National Preserve and Florida Panther National Wildlife Refuge with respect to distance to habitat, surface water (SWI), and human activity rates (HAR).

Season	**Term**	**Est**	**SE**	**F-Stat**	**Lower**	**Upper**	**p-value**
Dry							
	Hammock	0.105	0.011	3.005	0.036	0.173	0.003
	Swamp	-0.099	0.009	-3.659	-0.152	-0.046	<0.001
	Marsh	-0.107	0.010	-3.058	-0.176	-0.039	0.002
	Flatwoods	0.025	0.009	0.757	-0.039	0.089	0.449
	Road	0.059	0.008	1.514	-0.017	0.134	0.130
	Edge	0.079	0.009	2.593	0.019	0.138	0.010
	SWI	-0.049	0.007	-2.213	-0.093	-0.006	0.027
	HAR	-0.039	0.020	-1.104	-0.109	0.030	0.270
	Hammock:HAR	0.017	0.010	1.551	-0.005	0.040	0.121
	Swamp:HAR	-0.008	0.009	-1.129	-0.023	0.006	0.259
	Marsh:HAR	-0.006	0.010	-0.376	-0.039	0.026	0.707
	Flatwoods:HAR	0.014	0.010	0.938	-0.015	0.042	0.348
	Road:HAR	0.005	0.009	0.508	-0.014	0.024	0.612
	Edge:HAR	-0.003	0.009	-0.298	-0.021	0.015	0.766
	SWI:HAR	-0.003	0.007	-0.271	-0.022	0.016	0.787
Wet							
	Hammock	0.098	0.011	3.161	0.037	0.159	0.002
	Swamp	-0.011	0.008	-0.500	-0.056	0.033	0.617
	Marsh	-0.142	0.010	-4.290	-0.207	-0.077	<0.001
	Flatwoods	0.040	0.008	1.033	-0.035	0.115	0.302
	Prairie	-0.041	0.009	-1.445	-0.097	0.015	0.149
	Road	-0.027	0.008	-0.846	-0.088	0.035	0.397
	Edge	0.015	0.008	0.459	-0.050	0.080	0.646
	SWI	-0.060	0.006	-3.137	-0.098	-0.023	0.002
	HAR	-0.009	0.020	-0.264	-0.074	0.056	0.792
	Hammock:HAR	0.004	0.010	0.342	-0.019	0.027	0.732
	Swamp:HAR	-0.001	0.008	-0.137	-0.017	0.015	0.891
	Marsh:HAR	0.005	0.010	0.450	-0.015	0.024	0.653
	Flatwoods:HAR	0.004	0.009	0.472	-0.013	0.021	0.637

Prairie:HAR	-0.012	0.009	-0.959	-0.036	0.013	0.337
Road:HAR	0.014	0.009	1.266	-0.008	0.036	0.205
Edge:HAR	0.010	0.009	1.075	-0.008	0.029	0.283
SWI:HAR	0.000	0.006	-0.038	-0.015	0.015	0.970

Table 17. Seasonal selection coefficients (Est), standard errors (SE), F-statistics (F-stat), 95% confidence intervals (Lower, Upper), and p-values from step selection functions fit to seasonal data subsets for all female white-tailed deer eligible for analyses (wet season: *n* = 120; dry season *n* = 143) during January 2015 – December 2018 in Big Cypress National Preserve and Florida Panther National Wildlife Refuge with respect to distance to habitat, surface water index (SWI), and human activity rates (HAR).

Season	**Term**	**Est**	**SE**	**F-Stat**	**Lower**	**Upper**	**p-value**
Dry							
	Hammock	0.025	0.006	0.831	-0.035	0.085	0.406
	Swamp	0.024	0.005	0.796	-0.035	0.082	0.426
	Marsh	-0.114	0.005	-4.682	-0.161	-0.066	<0.001
	Flatwoods	-0.084	0.006	-2.759	-0.143	-0.024	0.006
	Prairie	-0.071	0.005	-2.703	-0.123	-0.020	0.007
	Road	-0.004	0.005	-0.201	-0.046	0.037	0.840
	Edge	-0.017	0.005	-0.721	-0.062	0.029	0.471
	SWI	-0.018	0.004	-0.950	-0.056	0.019	0.342
	HAR	-0.034	0.012	-1.122	-0.094	0.025	0.262
	Hammock:HAR	0.005	0.005	0.611	-0.010	0.020	0.541
	Swamp:HAR	-0.006	0.005	-1.196	-0.016	0.004	0.232
	Marsh:HAR	-0.003	0.005	-0.392	-0.017	0.011	0.695
	Flatwoods:HAR	0.008	0.006	1.258	-0.005	0.022	0.208
	Prairie:HAR	-0.017	0.005	-2.843	-0.028	-0.005	0.004
	Road:HAR	0.006	0.005	0.903	-0.007	0.019	0.367
	Edge:HAR	-0.003	0.005	-0.540	-0.015	0.009	0.589
	SWI:HAR	-0.001	0.004	-0.214	-0.011	0.009	0.830
Wet							
	Hammock	0.019	0.006	0.597	-0.044	0.083	0.551
	Swamp	0.003	0.005	0.100	-0.057	0.063	0.920
	Marsh	-0.142	0.005	-5.744	-0.191	-0.094	<0.001
	Flatwoods	-0.105	0.006	-3.275	-0.168	-0.042	0.001
	Prairie	-0.084	0.005	-2.937	-0.139	-0.028	0.003
	Road	-0.056	0.005	-2.200	-0.106	-0.006	0.028
	Edge	-0.017	0.005	-0.588	-0.073	0.040	0.557
	SWI	-0.059	0.004	-3.166	-0.095	-0.022	0.002
	HAR	-0.010	0.013	-0.541	-0.047	0.027	0.589
	Hammock:HAR	0.003	0.006	0.520	-0.008	0.015	0.603
	Swamp:HAR	0.002	0.005	0.398	-0.007	0.010	0.691

Marsh:HAR	0.000	0.005	0.051	-0.009	0.009	0.960
Flatwoods:HAR	0.014	0.006	2.407	0.003	0.025	0.016
Prairie:HAR	-0.018	0.005	-3.836	-0.028	-0.009	<0.001
Road:HAR	0.011	0.005	2.713	0.003	0.020	0.007
Edge:HAR	0.003	0.005	0.774	-0.005	0.012	0.439
SWI:HAR	-0.001	0.004	-0.230	-0.010	0.008	0.818

Table 18. Integrated step selection coefficients (Est), standard error (SE), F-statistics (F-Stat), 95% confidence intervals (Lower, Upper), and p-values for all female white-tailed deer eligible for analysis during the wet season (May-October; *n* = 120) during January 2015 – December 2018 in Big Cypress National Preserve and Florida Panther National Wildlife Refuge with respect to distance to habitat, turn angle (cos(TA)), step length (log(SL)), and surface water index (SWI).

Term	**Est**	**SE**	**F-Stat**	**Lower**	**Upper**	**p-value**
Edge	-8.670	6.143	-0.936	-26.828	9.488	0.349
Edge:cos(TA)	8.487	6.146	0.916	-9.677	26.651	0.360
Edge:log(SL)	0.032	0.002	3.755	0.015	0.048	<0.001
Edge:SWI	-0.026	0.002	-2.228	-0.048	-0.003	0.026
Flatwoods	18.356	5.999	2.126	1.434	35.278	0.033
Flatwoods:cos(TA)	-18.467	6.002	-2.137	-35.401	-1.532	0.033
Flatwoods:log(SL)	0.004	0.002	0.506	-0.012	0.020	0.613
Flatwoods:SWI	-0.008	0.002	-0.766	-0.027	0.012	0.444
Hammock	-11.918	5.506	-1.527	-27.217	3.382	0.127
Hammock:cos(TA)	11.934	5.509	1.527	-3.381	27.249	0.127
Hammock:log(SL)	-0.001	0.002	-0.105	-0.015	0.014	0.916
Hammock:SWI	0.016	0.002	2.116	0.001	0.031	0.034
Marsh	32.461	6.061	4.045	16.733	48.190	<0.001
Marsh:cos(TA)	-32.274	6.064	-4.022	-48.001	-16.548	<0.001
Marsh:log(SL)	-0.061	0.002	-7.478	-0.076	-0.045	<0.001
Marsh:SWI	0.067	0.002	5.810	0.045	0.090	<0.001
Prairie	7.880	5.631	0.946	-8.444	24.203	0.344
Prairie:cos(TA)	-7.916	5.634	-0.950	-24.243	8.412	0.342
Prairie:log(SL)	-0.003	0.002	-0.437	-0.019	0.012	0.662
Prairie:SWI	-0.021	0.002	-2.209	-0.039	-0.002	0.027
Road	-0.534	5.453	-0.068	-15.897	14.830	0.946
Road:cos(TA)	0.633	5.456	0.081	-14.742	16.007	0.936
Road:log(SL)	-0.022	0.002	-2.939	-0.037	-0.007	0.003
Road:SWI	-0.021	0.002	-2.495	-0.038	-0.005	0.013
Swamp	4.787	5.558	0.572	-11.607	21.180	0.567
Swamp:cos(TA)	-4.824	5.560	-0.576	-21.229	11.581	0.564
Swamp:log(SL)	0.014	0.002	1.801	-0.001	0.029	0.072
Swamp:SWI	0.019	0.002	2.454	0.004	0.034	0.014
SWI	23.153	3.985	3.196	8.954	37.352	0.001
SWI:cos(TA)	-23.301	3.987	-3.216	-37.499	-9.102	0.001

SWI:log(SL)	0.017	0.001	3.835	0.008	0.025	<0.001

Table 19. Integrated step selection coefficients (Est), standard error (SE), F-statistics (F-Stat), 95% confidence intervals (Lower, Upper), and p-values for all male white-tailed deer eligible for analysis during the wet season (May-October; *n* = 70) during January 2015 – December 2018 in Big Cypress National Preserve and Florida Panther National Wildlife Refuge.

Term	**Est**	**SE**	**F-Stat**	**Lower**	**Upper**	**p-value**
Edge	10.712	9.215	0.791	-15.829	37.254	0.429
Edge:cos(TA)	-10.831	9.220	-0.798	-37.420	15.757	0.425
Edge:log(SL)	0.019	0.003	1.793	-0.002	0.041	0.073
Edge:SWI	-0.014	0.003	-1.328	-0.034	0.006	0.184
Flatwoods	16.494	8.477	1.543	-4.458	37.447	0.123
Flatwoods:cos(TA)	-16.525	8.481	-1.543	-37.518	4.468	0.123
Flatwoods:log(SL)	0.002	0.003	0.153	-0.018	0.021	0.879
Flatwoods:SWI	-0.011	0.003	-1.129	-0.030	0.008	0.259
Hammock	-2.076	8.177	-0.182	-24.401	20.250	0.855
Hammock:cos(TA)	1.762	8.181	0.154	-20.599	24.123	0.877
Hammock:log(SL)	0.055	0.003	4.406	0.030	0.079	<0.001
Hammock:SWI	-0.008	0.003	-0.695	-0.031	0.015	0.487
Marsh	67.122	9.265	4.174	35.602	98.642	<0.001
Marsh:cos(TA)	-67.075	9.270	-4.169	-98.611	-35.539	<0.001
Marsh:log(SL)	-0.047	0.003	-4.445	-0.067	-0.026	<0.001
Marsh:SWI	-0.009	0.004	-0.690	-0.034	0.016	0.490
Prairie	18.088	8.210	1.515	-5.313	41.489	0.130
Prairie:cos(TA)	-18.135	8.214	-1.516	-41.573	5.303	0.129
Prairie:log(SL)	-0.001	0.003	-0.083	-0.025	0.023	0.934
Prairie:SWI	-0.017	0.003	-2.801	-0.028	-0.005	0.005
Road	-13.504	8.447	-1.738	-28.736	1.728	0.082
Road:cos(TA)	13.568	8.452	1.742	-1.699	28.836	0.082
Road:log(SL)	-0.017	0.003	-1.564	-0.039	0.004	0.118
Road:SWI	-0.024	0.003	-2.236	-0.045	-0.003	0.025
Swamp	-18.565	8.365	-1.439	-43.855	6.725	0.150
Swamp:cos(TA)	18.423	8.369	1.427	-6.878	43.724	0.154
Swamp:log(SL)	0.031	0.003	3.364	0.013	0.049	0.001
Swamp:SWI	0.020	0.003	2.559	0.005	0.036	0.010
SWI	28.571	6.292	2.728	8.046	49.096	0.006
SWI:cos(TA)	-28.794	6.295	-2.747	-49.334	-8.253	0.006
SWI:log(SL)	0.029	0.002	4.946	0.018	0.041	<0.001

Table 20. Integrated step selection coefficients (Est), standard error (SE), F-statistics (F-Stat), 95% confidence intervals (Lower, Upper), and p-values for all female white-tailed deer eligible for analysis during the dry season (November-April; *n* = 143) during January 2015 – December 2018 in Big Cypress National Preserve and Florida Panther National Wildlife Refuge with respect to distance to habitat, turn angle (cos(TA)), step length (log(SL)), and surface water index (SWI).

Term	**Est**	**SE**	**F-Stat**	**Lower**	**Upper**	**p-value**
Edge	-11.513	6.355	-1.369	-27.997	4.970	0.171
Edge:cos(TA)	11.383	6.359	1.353	-5.108	27.875	0.176
Edge:log(SL)	0.020	0.002	2.582	0.005	0.035	0.010
Edge:SWI	-0.033	0.002	-3.495	-0.052	-0.015	<0.001
Flatwoods	-1.345	6.198	-0.129	-21.719	19.029	0.897
Flatwoods:cos(TA)	1.269	6.201	0.122	-19.136	21.674	0.903
Flatwoods:log(SL)	0.006	0.002	0.730	-0.009	0.020	0.465
Flatwoods:SWI	-0.007	0.003	-0.771	-0.025	0.011	0.441
Hammock	1.674	5.590	0.204	-14.444	17.792	0.839
Hammock:cos(TA)	-1.697	5.593	-0.206	-17.852	14.458	0.837
Hammock:log(SL)	0.005	0.002	0.710	-0.008	0.018	0.477
Hammock:SWI	-0.003	0.002	-0.470	-0.017	0.011	0.638
Marsh	-1.392	6.318	-0.141	-20.691	17.906	0.888
Marsh:cos(TA)	1.659	6.322	0.168	-17.650	20.967	0.866
Marsh:log(SL)	-0.063	0.002	-8.461	-0.078	-0.049	<0.001
Marsh:SWI	0.051	0.003	5.585	0.033	0.069	<0.001
Prairie	13.843	5.631	1.483	-4.447	32.133	0.138
Prairie:cos(TA)	-13.943	5.634	-1.494	-32.237	4.352	0.135
Prairie:log(SL)	0.008	0.002	1.037	-0.007	0.024	0.300
Prairie:SWI	-0.008	0.002	-1.033	-0.024	0.007	0.302
Road	-15.227	5.595	-1.853	-31.333	0.878	0.064
Road:cos(TA)	15.314	5.598	1.863	-0.799	31.426	0.062
Road:log(SL)	-0.013	0.002	-1.896	-0.026	0.000	0.058
Road:SWI	-0.023	0.002	-3.672	-0.036	-0.011	<0.001
Swamp	-9.589	5.810	-1.148	-25.956	6.777	0.251
Swamp:cos(TA)	9.671	5.813	1.158	-6.702	26.043	0.247
Swamp:log(SL)	-0.004	0.002	-0.543	-0.019	0.011	0.587
Swamp:SWI	0.024	0.002	3.907	0.012	0.036	<0.001
SWI	8.883	4.504	1.084	-7.172	24.937	0.278
SWI:cos(TA)	-8.956	4.507	-1.093	-25.023	7.111	0.275

SWI:log(SL)	0.015	0.002	3.364	0.006	0.023	0.001

Table 21. Integrated step selection coefficients (Est), standard error (SE), F-statistics (F-Stat), 95% confidence intervals (Lower, Upper), and p-values for all male white-tailed deer eligible for analysis during dry seasons (November-April; *n* = 78) during January 2015 – December 2018 in Big Cypress National Preserve and Florida Panther National Wildlife Refuge with respect to distance to habitat, turn angle (cos(TA)), step length (log(SL)), and surface water index (SWI).

Term	**Est**	**SE**	**F-Stat**	**Lower**	**Upper**	**p-value**
Edge	-4.255	10.102	-0.347	-28.281	19.771	0.729
Edge:cos(TA)	4.219	10.107	0.344	-19.822	28.259	0.731
Edge:log(SL)	0.009	0.003	1.230	-0.006	0.024	0.219
Edge:SWI	-0.040	0.004	-3.355	-0.063	-0.016	0.001
Flatwoods	24.588	9.940	1.985	0.307	48.869	0.047
Flatwoods:cos(TA)	-24.638	9.945	-1.986	-48.948	-0.328	0.047
Flatwoods:log(SL)	0.001	0.003	0.128	-0.015	0.017	0.899
Flatwoods:SWI	-0.025	0.004	-2.277	-0.046	-0.003	0.023
Hammock	13.851	9.257	1.205	-8.686	36.389	0.228
Hammock:cos(TA)	-14.084	9.262	-1.223	-36.649	8.481	0.221
Hammock:log(SL)	0.036	0.003	3.825	0.017	0.054	<0.001
Hammock:SWI	-0.014	0.004	-1.329	-0.034	0.007	0.184
Marsh	3.135	9.994	0.271	-19.568	25.838	0.787
Marsh:cos(TA)	-3.010	9.999	-0.260	-25.746	19.726	0.795
Marsh:log(SL)	-0.042	0.003	-4.997	-0.059	-0.026	<0.001
Marsh:SWI	0.010	0.004	0.740	-0.017	0.037	0.459
Prairie	35.649	9.205	3.219	13.946	57.351	0.001
Prairie:cos(TA)	-35.834	9.210	-3.236	-57.540	-14.129	0.001
Prairie:log(SL)	0.023	0.003	2.569	0.005	0.041	0.010
Prairie:SWI	-0.018	0.004	-1.898	-0.037	0.001	0.058
Road	-18.430	8.844	-1.576	-41.342	4.483	0.115
Road:cos(TA)	18.537	8.849	1.583	-4.417	41.491	0.113
Road:log(SL)	-0.021	0.003	-2.423	-0.038	-0.004	0.015
Road:SWI	-0.031	0.004	-2.728	-0.053	-0.009	0.006
Swamp	-16.676	9.409	-1.280	-42.209	8.857	0.201
Swamp:cos(TA)	16.648	9.414	1.277	-8.894	42.189	0.201
Swamp:log(SL)	0.005	0.003	0.656	-0.010	0.019	0.512
Swamp:SWI	0.025	0.004	2.761	0.007	0.044	0.006
SWI	11.037	6.957	1.107	-8.498	30.571	0.268
SWI:cos(TA)	-11.162	6.960	-1.118	-30.722	8.398	0.263
SWI:log(SL)	0.023	0.002	4.857	0.014	0.032	<0.001

Table 22. Sex- and season-specific directional relationships of the interactive effects of distance to cover type and surface water index (SWI) by step length (log(SL)) and distance to cover type and surface water index (SWI) by turn angle (cos(TA)) from integrated step selection analyses for white-tailed deer (F wet *n* = 120, F dry *n* = 143; M wet *n* = 70, M dry *n* = 78) during January 2015 - December 2018 in Big Cypress National Preserve and Florida Panther National Wildlife Refuge. Empty values indicate no significant interaction.

	Male		**Female**	
Parameter	**Dry Season**	**Wet Season**	**Dry Season**	**Wet Season**
log(SL):SWI	+	+	+	+
log(SL):Flatwoods				
log(SL):Hammock	+	+		
log(SL):Marsh	-	-	-	-
log(SL):Prairie	+			
log(SL):Swamp		+		
log(SL):Edge[P]			+	+
log(SL):Road[P,H]	-			-
cos(TA):SWI		-		-
cos(TA):Flatwoods	-			-
cos(TA):Hammock				
cos(TA):Marsh		-		-
cos(TA):Prairie	-			
cos(TA):Swamp				
cos(TA):Edge[P]				
cos(TA):Road[P,H]				

[P] proxy for predation risk

[H] proxy for human disturbance

Table 23. Model selection results including number of parameters, corrected Akaike information criteria (AIC_c), change in AIC_c (ΔAIC_c), and model weight (W) for models used to predict male and female (≥1 year old) white-tailed deer detection rates (independent camera detections/h) at camera traps on the Big Cypress National Preserve and Florida Panther National Wildlife Refuge during February-October 2015.

Model	**Parameters**	**AICc**	**ΔAICc**	**W**
Male Deer				
Trail:Season + Trail:Time + Season:Time	17	12601	0	1
Season:Time	11	12803	202	0
Trail:Time + Season	9	12926	325	0
Trail + Season + Time	8	12935	335	0
Female Deer				
Trail:Season + Trail:Time + Season:Time	17	16116	0	1
Trail:Time + Season	9	16512	396	0
Season:Time	11	16943	826	0
Trail + Season + Time	8	17004	888	0
Panther				
Trail:Season + Trail:Time + Season:Time	17	5307	0	1
Trail:Time + Season	9	5354	47	0
Trail + Season + Time	8	5384	77	0
Season:Time	11	5410	103	0

Table 24. Total number of white-tailed deer detections at trail cameras in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge during January 2015 – December 2017.

	Site	**2015**	**2016**	**2017**	**Study**
All deer	North Addition Lands	12687	13382	5729	31798
	Bear Island	16076	12186	10801	39063
	FPNWR	30909	27455	23665	82029
Males	North Addition Lands	3304	2895	1266	7465
	Bear Island	2044	2327	1615	5986
	FPNWR	4876	4912	5157	14945
Females	North Addition Lands	4268	4913	2368	11549
	Bear Island	6140	4395	3780	14315
	FPNWR	13112	12442	9917	35471
Fawns	North Addition Lands	1352	1891	492	3735
	Bear Island	2332	1282	1676	5290
	FPNWR	3535	3019	2750	9304

Table 25. Average number of white-tailed deer detections per camera day in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge during January 2015 – December 2017. Temporal trends can be viewed in Figs. 22, 24, 26, 28.

	Site	**2015**	**2016**	**2017**	**Study**
All deer	North Addition Lands	0.618	0.615	0.312	0.514
	Bear Island	0.841	0.566	0.515	0.638
	FPNWR	1.484	1.275	1.130	1.296
Males	North Addition Lands	0.160	0.133	0.062	0.118
	Bear Island	0.105	0.108	0.077	0.097
	FPNWR	0.235	0.228	0.247	0.237
Females	North Addition Lands	0.208	0.226	0.132	0.189
	Bear Island	0.321	0.205	0.180	0.234
	FPNWR	0.631	0.578	0.474	0.561
Fawns	North Addition Lands	0.066	0.087	0.027	0.060
	Bear Island	0.126	0.060	0.080	0.077
	FPNWR	0.167	0.139	0.129	0.145

Table 26. The average daily proportion of cameras with at least one white-tailed deer detection in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge during the January 2015 – December 2017. Temporal trends can be viewed in Figs. 23, 25, 27, 29.

	Site	**2015**	**2016**	**2017**	**Study**
All deer	North Addition Lands	0.123	0.114	0.079	0.105
	Bear Island	0.154	0.124	0.117	0.131
	FPNWR	0.216	0.200	0.201	0.206
Males	North Addition Lands	0.055	0.045	0.030	0.043
	Bear Island	0.042	0.042	0.038	0.041
	FPNWR	0.082	0.076	0.081	0.080
Females	North Addition Lands	0.054	0.058	0.041	0.051
	Bear Island	0.091	0.067	0.059	0.072
	FPNWR	0.129	0.126	0.116	0.124
Fawns	North Addition Lands	0.012	0.015	0.008	0.012
	Bear Island	0.021	0.007	0.013	0.013
	FPNWR	0.023	0.024	0.023	0.023

Table 27. Total number of detections of white-tailed deer predators in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge during January 2015 – December 2017.

Predator	**Site**	**2015**	**2016**	**2017**	**Study**
Panther	North Addition Lands	1255	1503	621	3379
	Bear Island	1448	926	636	3010
	FPNWR	587	775	611	1973
Black Bear	North Addition Lands	2691	1257	764	4712
	Bear Island	2341	1808	2561	6710
	FPNWR	3009	2110	1822	6491
Bobcat	North Addition Lands	775	1246	583	2604
	Bear Island	1443	1216	1486	4145
	FPNWR	801	1085	868	2754
Coyote	North Addition Lands	19	17	23	59
	Bear Island	31	37	18	86
	FPNWR	139	8	32	179
Alligator	North Addition Lands	91	41	36	14
	Bear Island	47	3	19	25
	FPNWR	111	27	50	34

Table 28. Average number of predator detections per camera day in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge during January 2015 – December 2017. Temporal trends can be viewed in Figs. 30, 32, 34, 36, 38.

Predator	**Site**	**2015**	**2016**	**2017**	**Study**
Panther	North Addition Lands	0.062	0.069	0.031	0.054
	Bear Island	0.078	0.043	0.030	0.050
	FPNWR	0.029	0.036	0.029	0.031
Black Bear	North Addition Lands	0.130	0.058	0.037	0.074
	Bear Island	0.121	0.084	0.125	0.110
	FPNWR	0.144	0.097	0.087	0.110
Bobcat	North Addition Lands	0.038	0.057	0.029	0.042
	Bear Island	0.079	0.057	0.070	0.068
	FPNWR	0.039	0.050	0.041	0.043
Coyote	North Addition Lands	0.001	0.001	0.001	0.001
	Bear Island	0.002	0.002	0.001	0.001
	FPNWR	0.007	<0.001	0.001	0.002
Alligator	North Addition Lands	0.001	0.002	0.002	<0.001
	Bear Island	<0.001	<0.001	<0.001	0.001
	FPNWR	0.002	0.001	0.002	0.002

Table 29. The average daily proportion of cameras with at least one predator detection in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge during January 2015 – December 2017. Temporal trends can be viewed in Figs. 31, 33, 35, 37, 39.

Predator	**Site**	**2015**	**2016**	**2017**	**Study**
Panther	North Addition Lands	0.038	0.040	0.022	0.033
	Bear Island	0.047	0.030	0.021	0.033
	FPNWR	0.018	0.023	0.018	0.020
Black Bear	North Addition Lands	0.051	0.026	0.019	0.032
	Bear Island	0.056	0.045	0.051	0.051
	FPNWR	0.052	0.036	0.033	0.040
Bobcat	North Addition Lands	0.030	0.041	0.021	0.031
	Bear Island	0.055	0.038	0.049	0.047
	FPNWR	0.028	0.035	0.030	0.031
Coyote	North Addition Lands	0.001	0.001	0.001	0.001
	Bear Island	0.002	0.001	0.001	0.001
	FPNWR	0.006	<0.001	0.001	0.002
Alligator	North Addition Lands	0.001	0.001	0.001	<0.001
	Bear Island	<0.001	<0.001	<0.001	<0.001
	FPNWR	0.001	<0.001	0.002	0.001

Table 30. Abundance estimates, standard deviation (SD), bias, and root mean squared error (RMSE) for 500 simulated data sets for each trail camera design. Data were simulated using values of abundance (N) and detection parameters estimated from the three years of camera and telemetry data from each of the three study sites in South Florida (AL - North Addition Lands, BI - Bear Island, and FP - Florida Panther National Wildlife Refuge). The design with the lowest RMSE was considered to be optimal in terms of bias and precision.

Site	**Design**	**Description**	**Cameras**	**Abundance (N)**	**Estimated N**	**SD**	**Bias**	**RMSE**
AL	1	Status quo	60	75	75.40	19.78	0.40	14.28
AL	2	Paired	40	75	74.86	22.56	-0.14	16.65
AL	3	Random	20	75	74.46	29.38	-0.54	24.43
AL	4	Random	30	75	74.74	24.53	-0.26	19.22
AL	5	Random	40	75	74.44	22.38	-0.56	16.73
AL	6	On-trail	20	75	75.55	30.27	0.55	22.80
AL	7	On-trail	30	75	75.92	24.99	0.92	18.67
AL	8	On-trail	40	75	75.53	22.96	0.53	16.38
AL	9	On-trail	50	75	74.78	21.39	-0.22	15.12
AL	10	On-trail	60	75	75.41	20.65	0.41	14.02
BI	1	Status quo	60	200	200.83	46.90	0.83	31.52
BI	2	Paired	40	200	203.70	53.33	3.70	38.45
BI	3	Random	20	200	196.88	62.35	-3.21	47.77
BI	4	Random	30	200	199.61	57.42	-0.39	44.41
BI	5	Random	40	200	200.02	52.03	0.02	37.04
BI	6	On-trail	20	200	200.95	67.12	0.95	50.09
BI	7	On-trail	30	200	198.56	58.35	-1.44	45.14
BI	8	On-trail	40	200	198.96	54.29	-1.04	37.69
BI	9	On-trail	50	200	199.69	51.20	-0.31	33.66
BI	10	On-trail	60	200	200.65	49.13	0.65	31.37
FP	1	Status quo	60	250	250.94	54.70	0.94	35.72
FP	2	Paired	40	250	248.27	59.43	-1.73	43.63
FP	3	Random	20	250	249.25	67.70	-0.75	57.65
FP	4	Random	30	250	252.02	62.18	2.02	46.35
FP	5	Random	40	250	249.08	59.08	-0.92	41.74
FP	6	On-trail	20	250	248.11	66.81	-1.89	51.20
FP	7	On-trail	30	250	247.08	62.18	-2.92	44.26
FP	8	On-trail	40	250	249.96	58.94	-0.04	40.22
FP	9	On-trail	50	250	249.11	56.04	-0.89	36.46
FP	10	On-trail	60	250	251.17	54.85	1.17	32.83

Table 31. Cost analysis of aerial surveying and trail camera surveying designs for a 14-day sampling period. The camera design options are described in the Optimal Monitoring Design section. Field equipment costs for camera surveying includes costs of trail cameras, camera boxes, memory cards, batteries, and GPS units. Field equipment costs for aerial surveying includes flight cost and purchase of GPS units. Field labor costs for camera surveying includes estimated labor for deploying and retrieving trail cameras. Data processing costs for camera surveying includes estimated labor for photo tagging and statistical analyses. Annual Cost in Year 1 is defined as the initial purchase of equipment and labor. Annual Cost in Subsequent Years does not include the initial costs of purchasing equipment and labor. The 5 Year Cost Total is the estimated cost to implement a 5-year monitoring program. Field labor and data processing hours are converted to dollars using a $20/hr payscale.

Survey Method	**Field Surveying ($)**	**Field Labor (hr)**	**Field Labor ($)**	**Data Processing (hr)**	**Data Processing ($)**	**Annual Cost in Year 1 ($)**	**Annual Cost in Subsequent Years ($)**	**5 Year Cost Total ($)**
Design 1	14,374	88	1,760	24	480	16,614	2,240	25,574
Design 2	9,774	56	1,120	22	440	11,334	1,560	17,574
Design 3	5,174	32	640	20	400	6,214	1,040	10,374
Design 4	7,474	48	960	20	400	8,834	1,360	14,274
Design 5	9,774	56	1,120	22	440	11,334	1,560	17,574
Design 6	5,174	24	480	20	400	6,054	880	9,574
Design 7	7,474	32	640	20	400	8,514	1,040	12,674
Design 8	9,774	40	800	22	440	11,014	1,240	15,974
Design 9	12,074	56	1,120	22	440	13,634	1,560	19,874
Design 10	14,374	64	1,280	24	480	16,134	1,760	23,174
Aerial	6,327	16	320	10	200	6,847	6,273	31,937

Table 32. Cost analysis of aerial surveying and trail camera surveying designs for an 8-week sampling period. The camera design options are described in the Optimal Monitoring Design section. Field equipment costs for camera surveying includes costs of trail cameras, camera boxes, memory cards, batteries, and GPS units. Field equipment costs for aerial surveying includes flight cost and purchase of GPS units. Field labor costs for camera surveying includes estimated labor for deploying and retrieving trail cameras. Data processing costs for camera surveying includes estimated labor for photo tagging and statistical analyses. Annual Cost in Year 1 is defined as the initial purchase of equipment and labor. Annual Cost in Subsequent Years does not include the initial costs of purchasing equipment and labor. The 5 Year Cost Total is the estimated cost to implement a 5 year monitoring program. Field labor and data processing hours are converted to dollars using a $20/hr payscale.

Survey Method	Field Surveying ($)	Field Labor (hr)	Field Labor ($)	Data Processing (hr)	Data Processing ($)	Annual Cost in Year 1 ($)	Annual Cost in Subsequent Years ($)	5 Year Cost Total ($)
Design 1	14,374	88	1,760	72	1,440	17,574	3,200	30,374
Design 2	9,774	56	1,120	64	1,280	12,174	2,400	21,774
Design 3	5,174	32	640	56	1,120	6,934	1,760	13,974
Design 4	7,474	48	960	56	1,120	9,554	2,080	17,874
Design 5	9,774	56	1,120	56	1,120	12,014	2,240	20,974
Design 6	5,174	24	480	56	1,120	6,774	1,600	13,174
Design 7	7,474	32	640	56	1,120	9,234	1,760	16,274
Design 8	9,774	40	800	64	1,280	11,854	2,080	20,174
Design 9	12,074	56	1,120	64	1,280	14,474	2,400	24,074
Design 10	14,374	64	1,280	72	1,440	17,094	2,720	27,974
Aerial	6,327	16	320	10	200	6,847	6,273	31,937

FIGURES

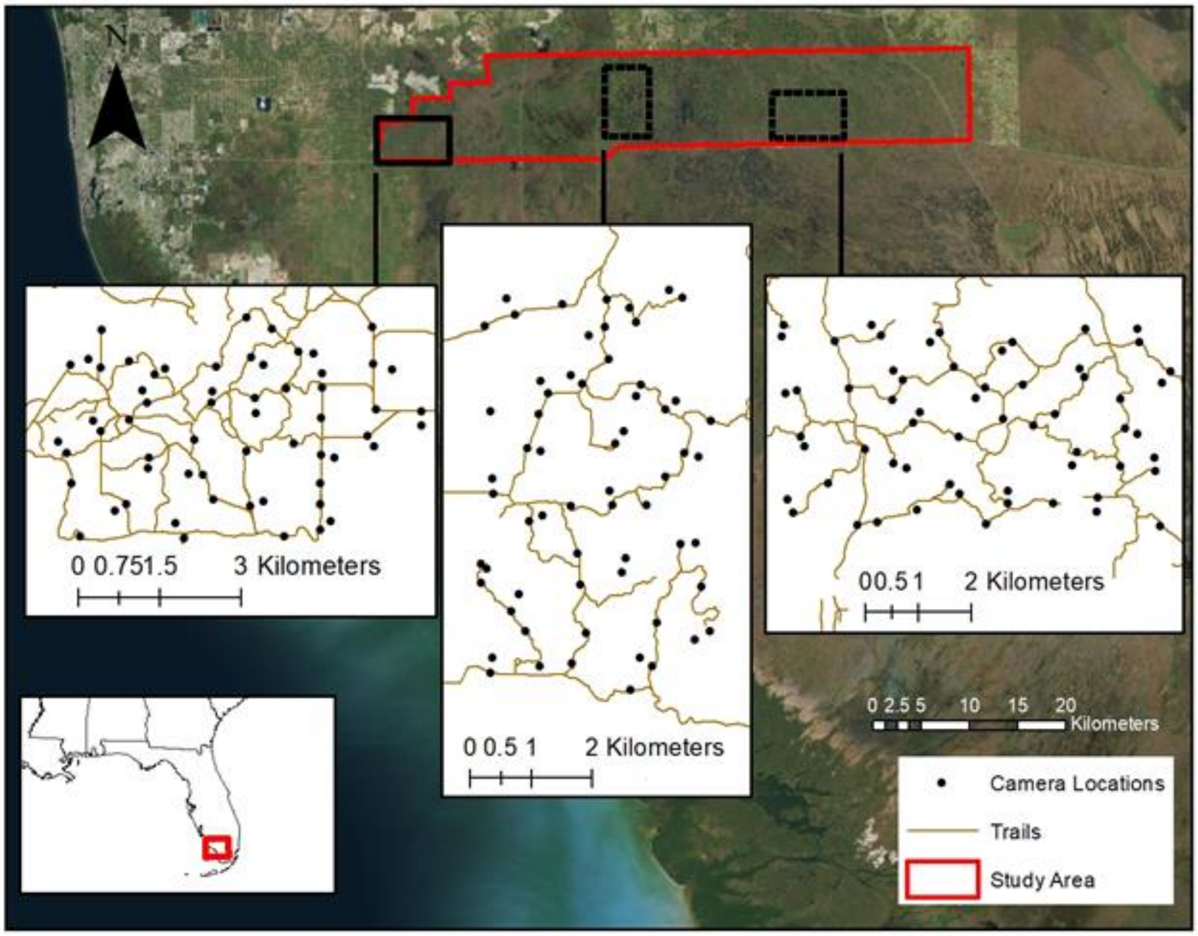
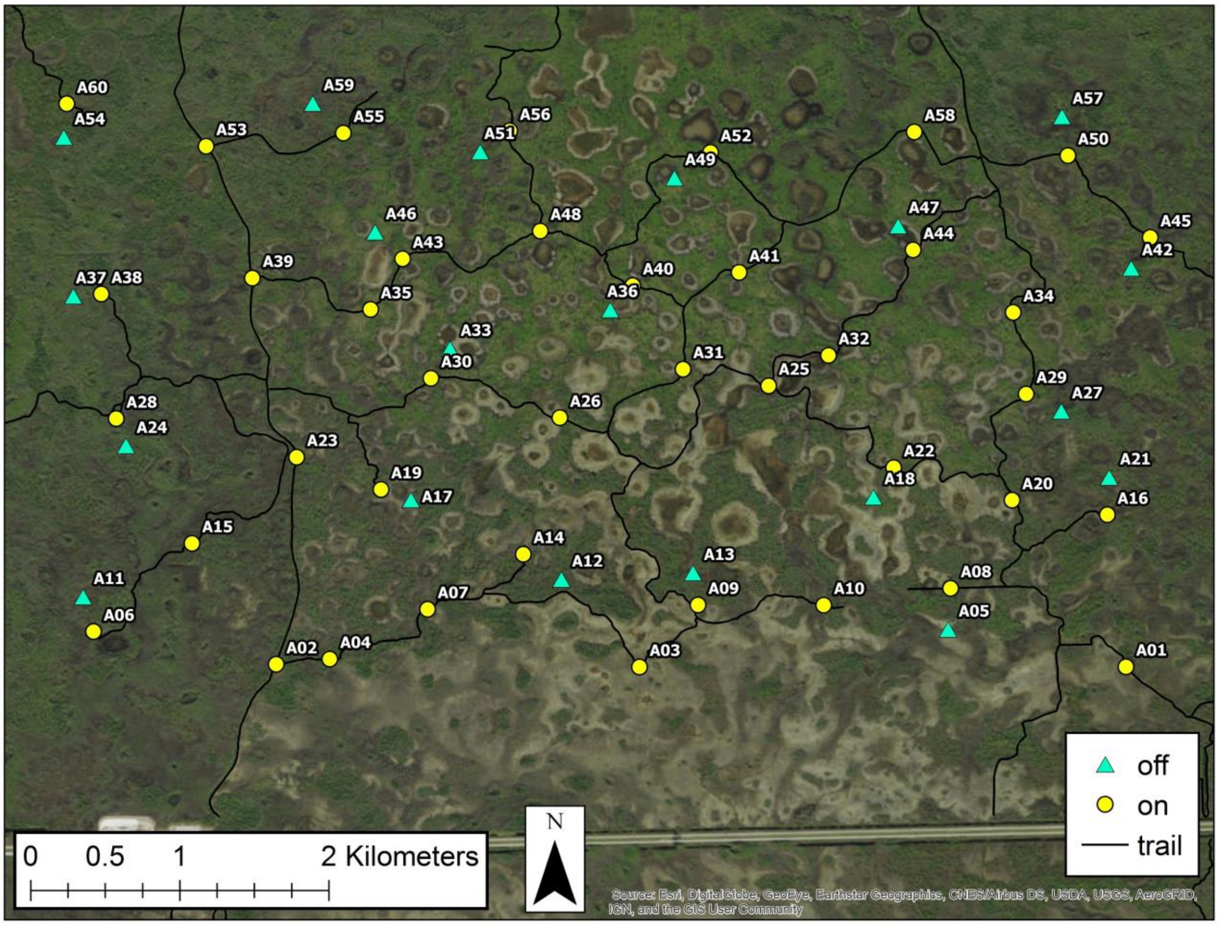


Figure 1. Locations of the 180 cameras used to monitor white-tailed deer in in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge during 1 January 2015 – 31 December 2017. The central and eastern grids (dashed boundary lines) were located in Big Cypress National Preserve. The western grid (solid boundary line) was in the Florida Panther National Wildlife Refuge.



Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community

Figure 2. Locations of the 60 cameras used to monitor white-tailed deer between 1 January 2015 – 31 December 2017 within the North Addition Lands study site in Big Cypress National Preserve. Yellow circles indicate on-trail cameras, and blue triangles indicate off-trail cameras.

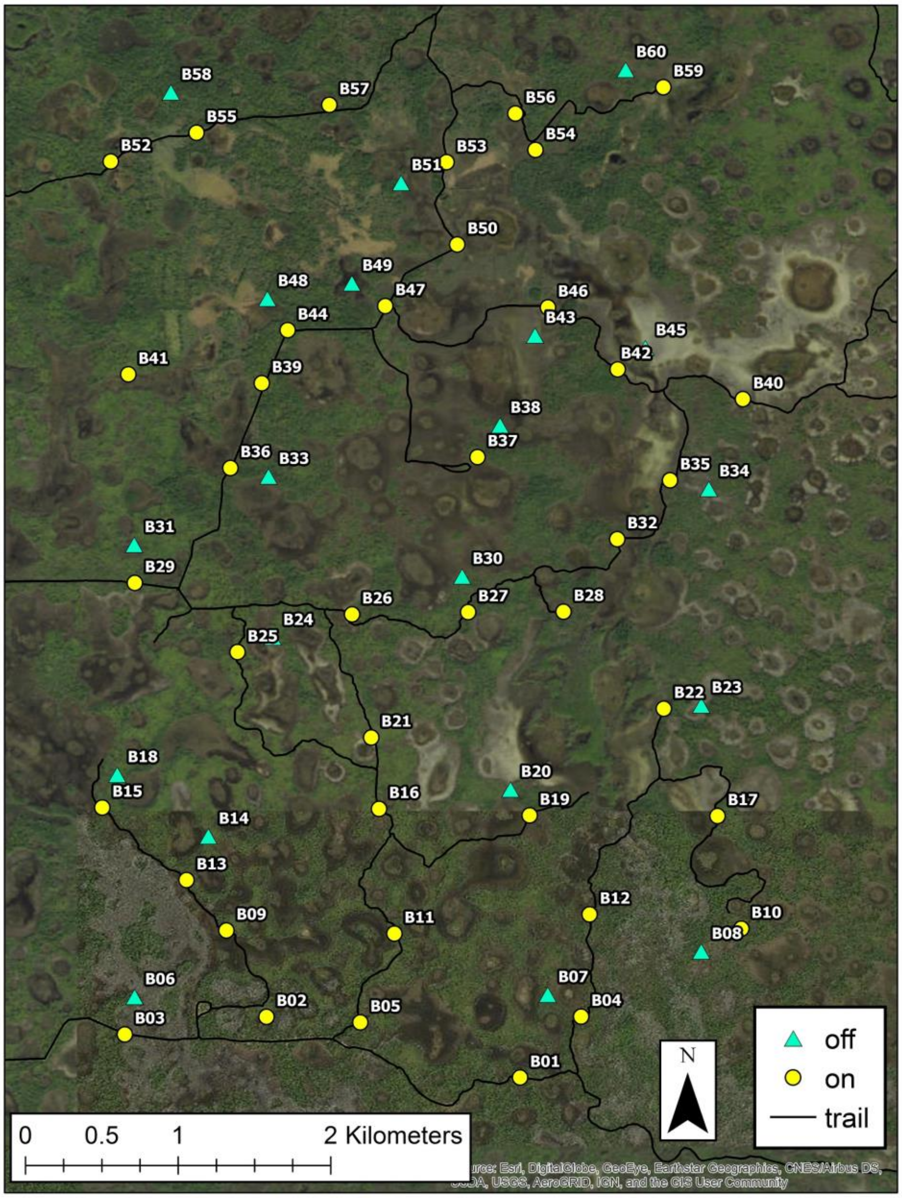


Figure 3. Location of the 60 cameras used to monitor white-tailed deer between 1 January 2015 – 31 December 2017 within the Bear Island study site in Big Cypress National Preserve. Yellow circles indicate on-trail cameras, and blue triangles indicate off-trail cameras.

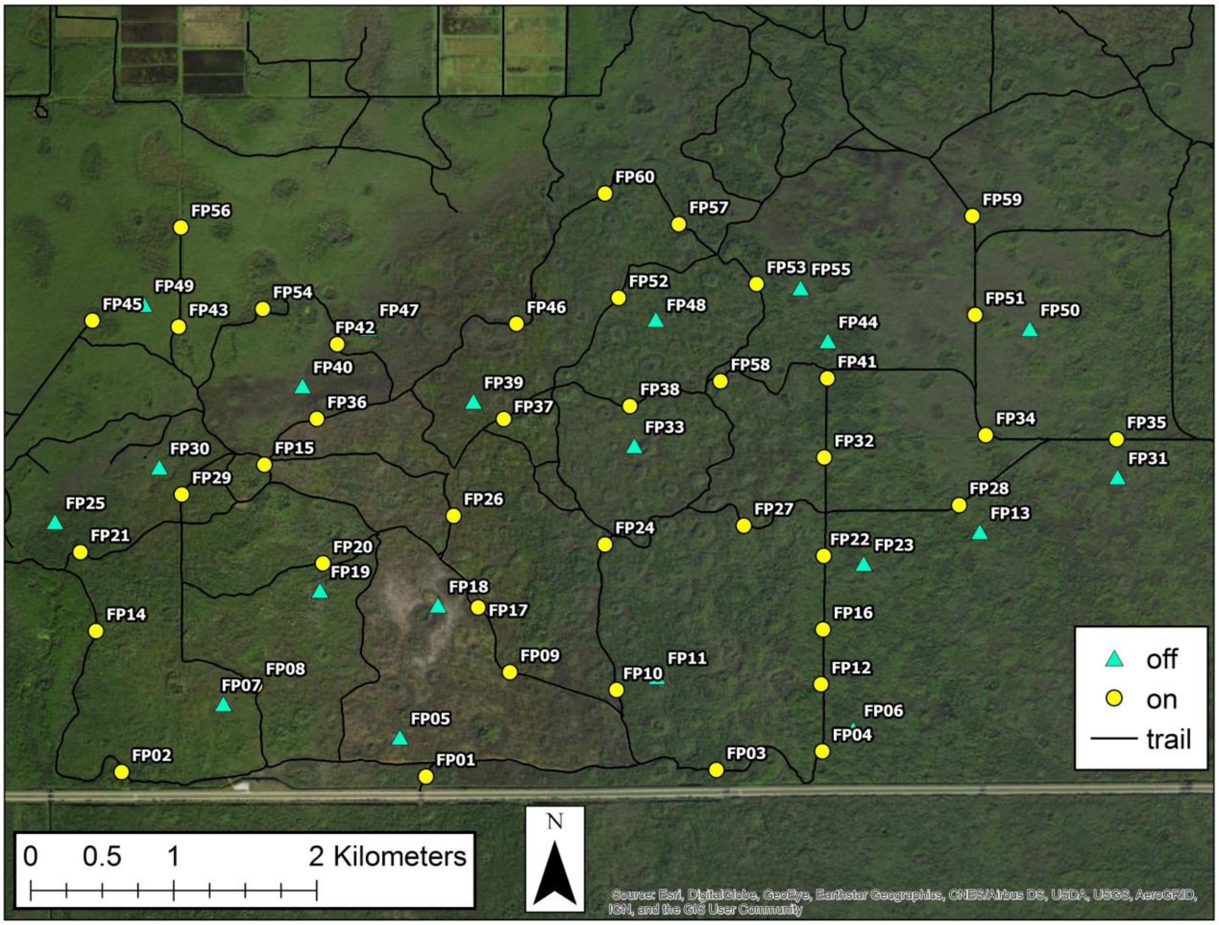


Figure 4. Location of the 60 trail cameras used to monitor white-tailed deer between 1 January 2015 – 31 December 2017 within the Florida Panther National Wildlife Refuge. Yellow circles indicate on-trail cameras, and blue triangles indicate off-trail cameras.

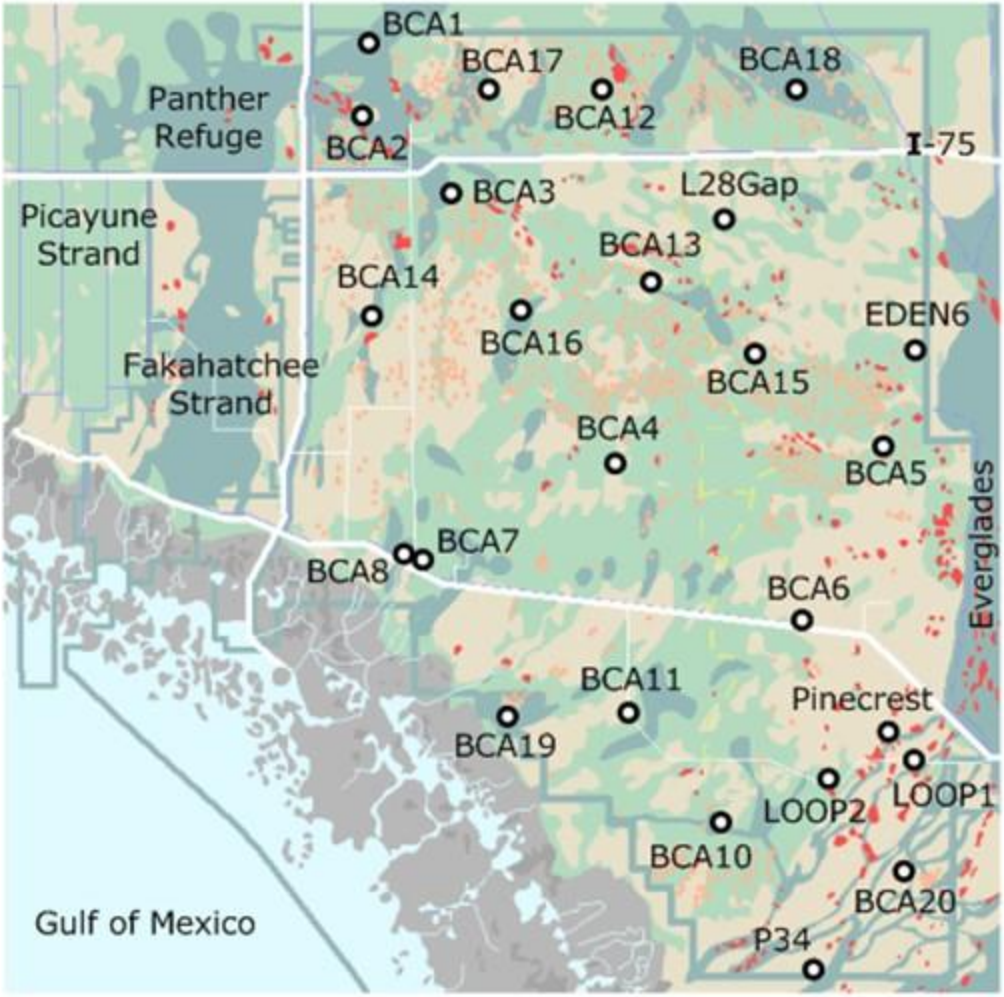


Figure 5. Surface water wells across the Big Cypress Basin, South Florida. DBHYDRO surface water wells BCNPA 1, BCNPA 2, BCNPA 12, BCA 17, and BCA 18 (DBHYDRO 2019) were used to estimate daily surface water values within Big Cypress National Preserve and Florida Panther National Wildlife Refuge between 1 January 2015 – 31 December 2018. Figure courtesy of Robert Sobczak, Big Cypress National Preserve.

a

Habitat Type	Surface Water Estimate (ft.)
Marsh/Pond Apple (MA)	0
Tall Cypress (TC)	0.5
Marl Prairie (MP)	1
Hydric Pines (HP)	1.5
Mesic Pines (MP)	1.85
Mesic Hammock (MH)	2.05

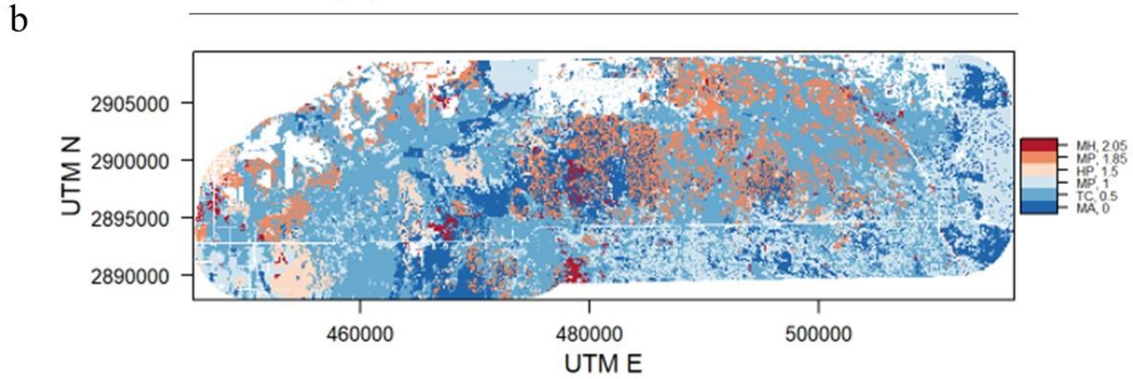


Figure 6. Surface water depths needed to inundate six of the most common habitat types in the Big Cypress Basin, South Florida. Surface water depth are depicted a) in tabular form, and b) as a map. White areas represent no data. DBHYDRO surface water wells BCNPA 1, BCNPA 2, BCNPA 12, BCA 17, and BCA 18 (DBHYDRO 2019) were used to estimate daily surface water estimates within Big Cypress National Preserve and Florida Panther National Wildlife between 1 January 2015 – 31 December 2018.

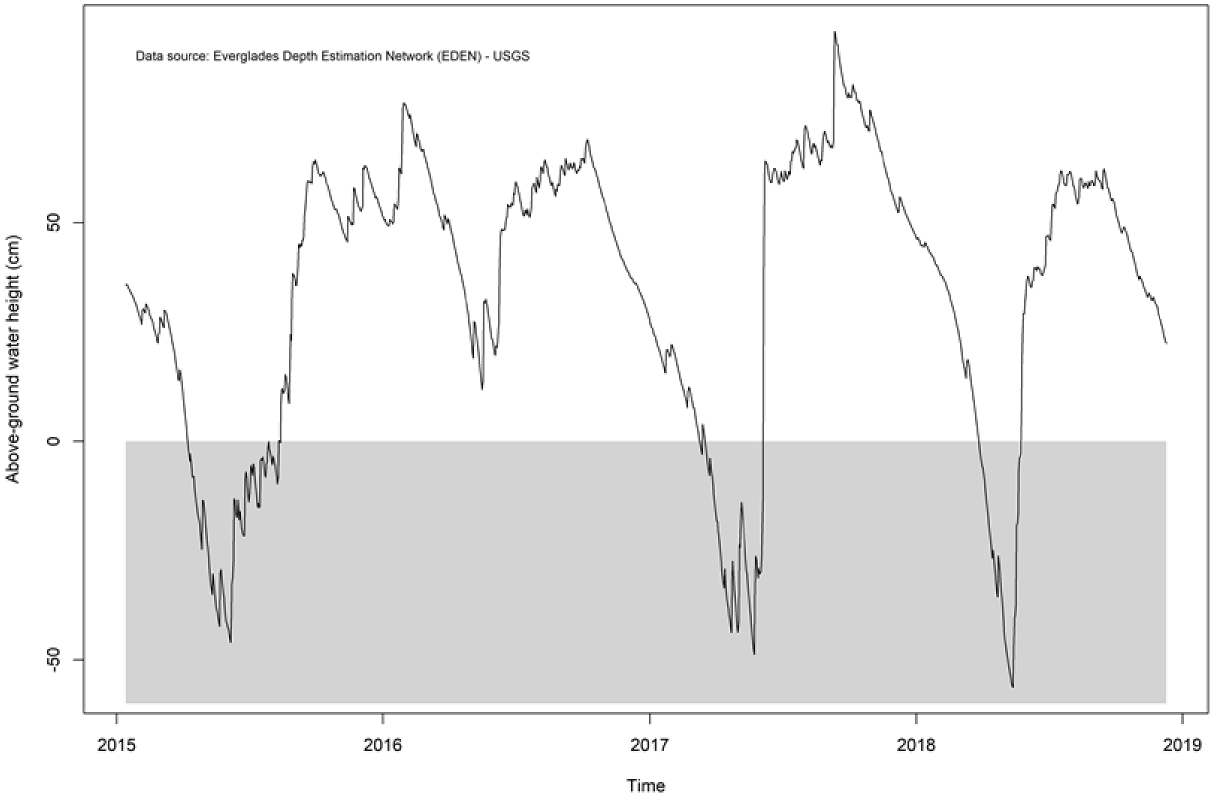


Figure 7. Average water level during the study period from January 2015 – December 2018 in Big Cypress National Preserve and Florida Panther National Wildlife Refuge. Grey area indicates dry conditions. Water level data courtesy of the Everglades Depth Estimation Network (EDEN) project and the US Geological Survey.

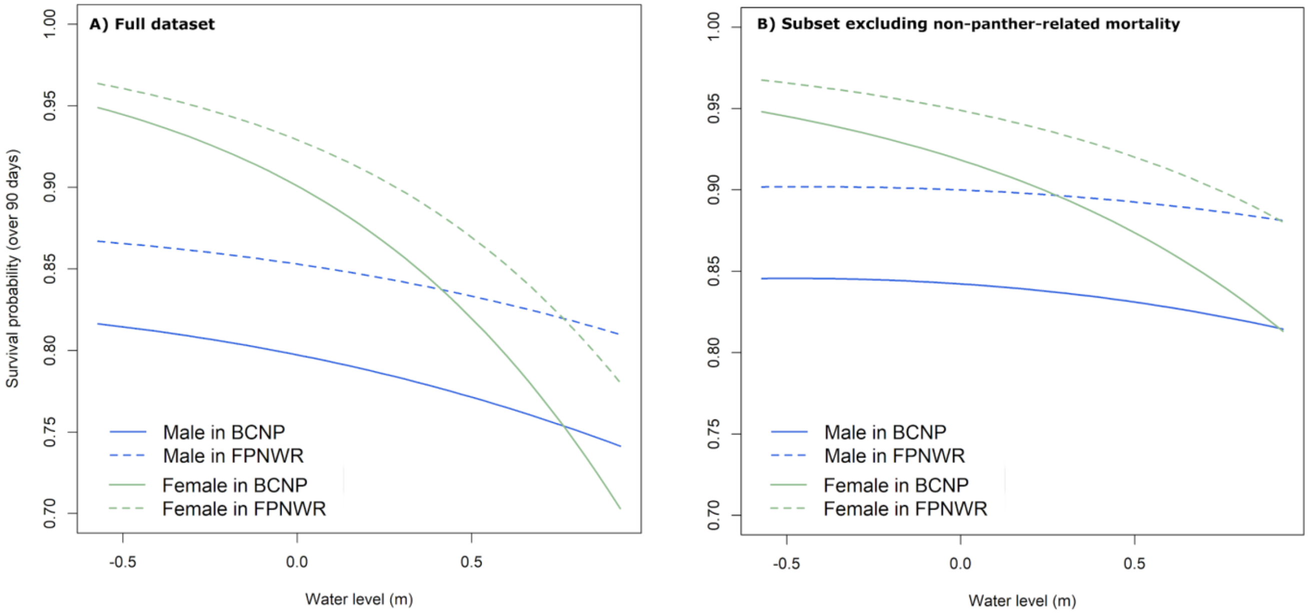


Figure 8. White-tailed deer survival probability as a function of water level, sex, and management area (at peak fawning during first year of study [24 January]) for A) the whole dataset and for B) the subsetted dataset with panther only mortality (*n* = 241) from January 2015 – December 2018 in Big Cypress National Preserve (BCNP) and Florida Panther National Wildlife Refuge (FPNWR).

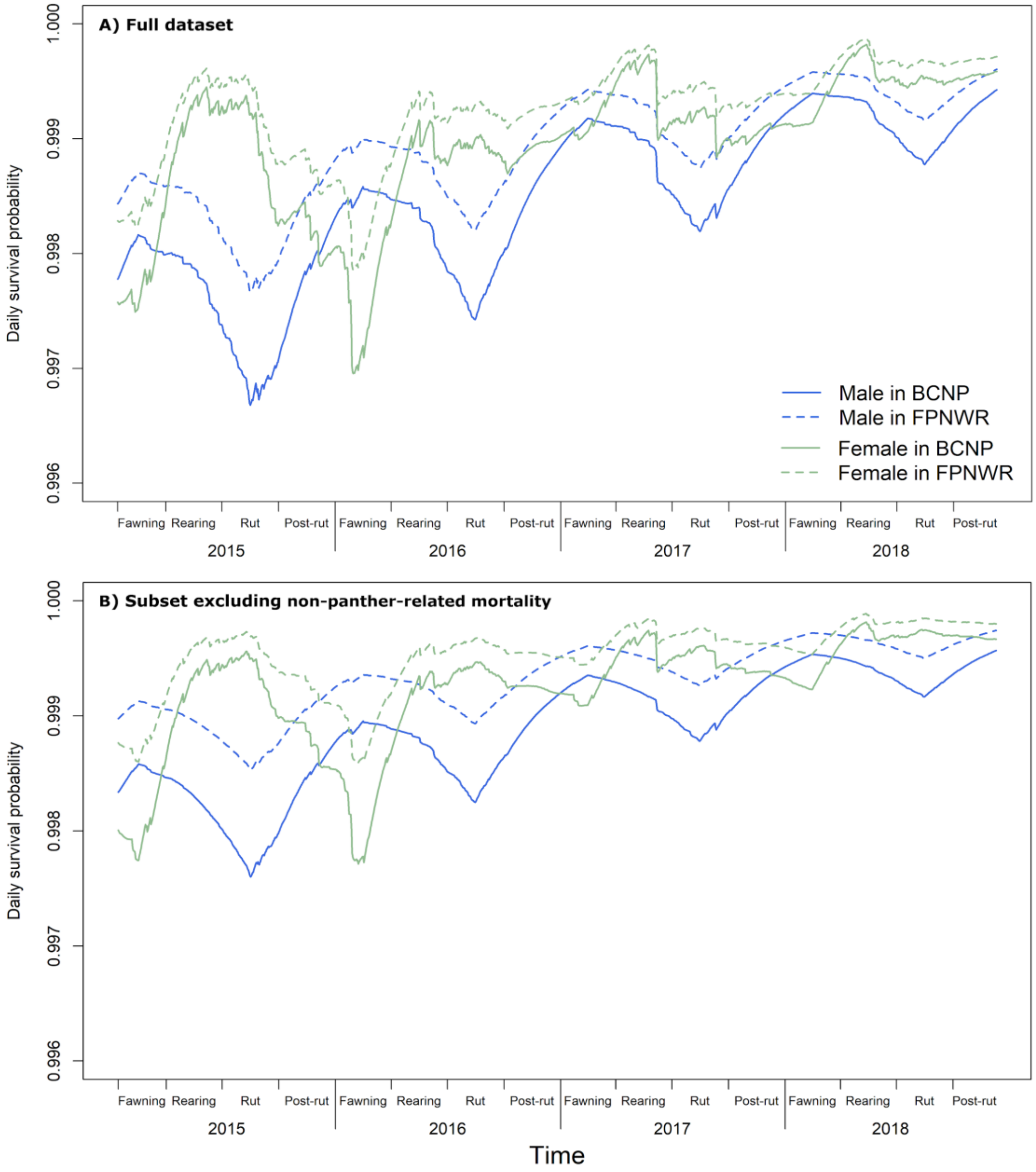


Figure 9. Daily white-tailed deer survival probability over the course of the study period as a function of time, sex, management area, and water level for A) the whole dataset and for B) the subset of data with panther only mortality (*n* = 241) from January 2015 – December 2018 in Big Cypress National Preserve (BCNP) and Florida Panther National Wildlife Refuge (FPNWR).

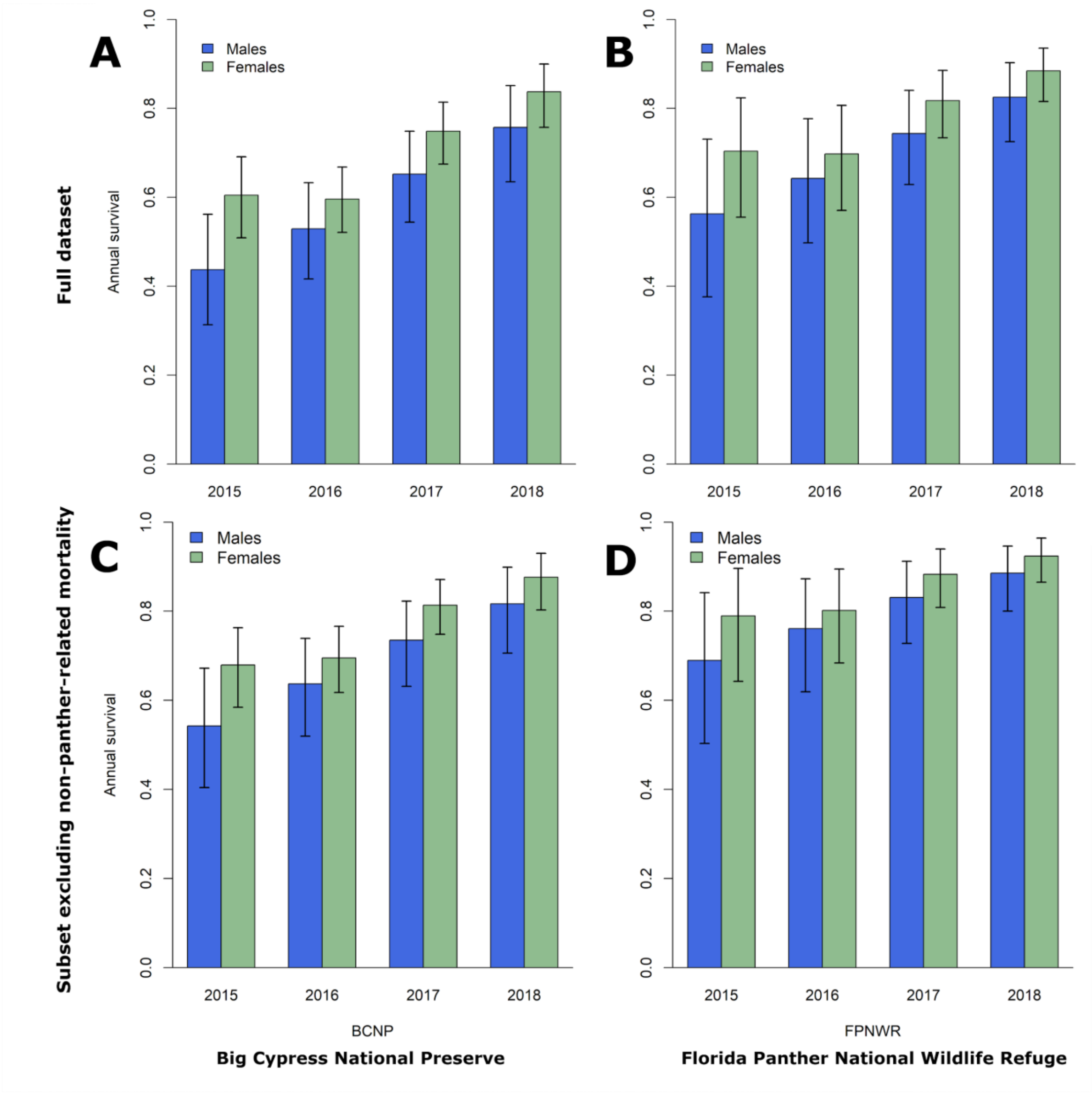


Figure 10. Estimated annual survival probability for male and female white-tailed deer in A) Big Cypress National Preserve (BCNP), based on the whole dataset, B) Florida Panther National Wildlife Refuge (FPNWR), based on the whole dataset, C) Big Cypress National Preserve (BCNP), based on the subset of data with panther only mortality, and D) in Florida Panther National Wildlife Refuge (FPNWR), based on the subset of data with panther only mortality from January 2015 – December 2018.

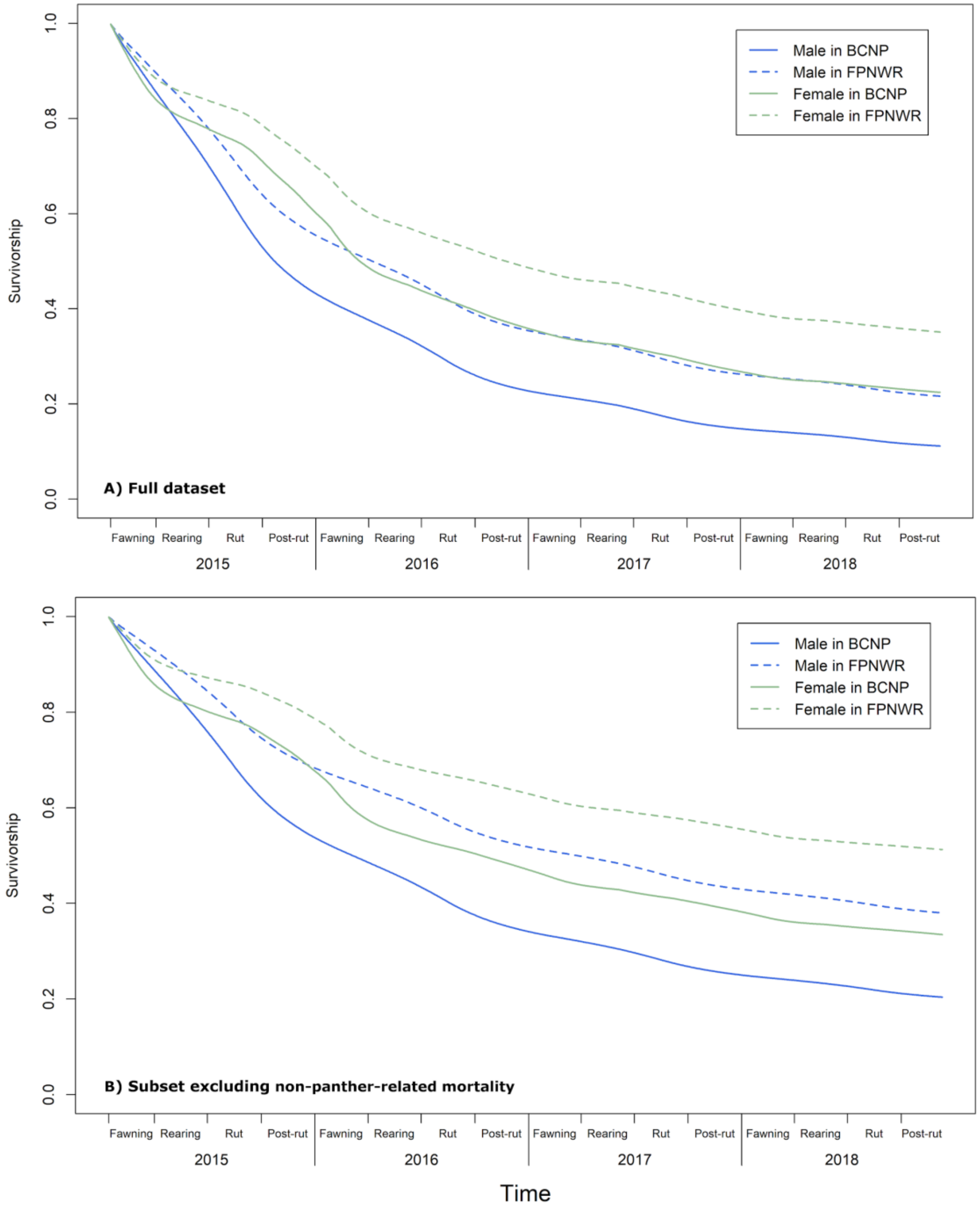


Figure 11. White-tailed deer survivorship function over the course of study the accounting for A) all sources of mortality, and B) panther mortality only (*n* = 241) from January 2015 - December 2018 in Big Cypress National Preserve (BCNP) and Florida Panther National Wildlife Refuge (FPNWR).

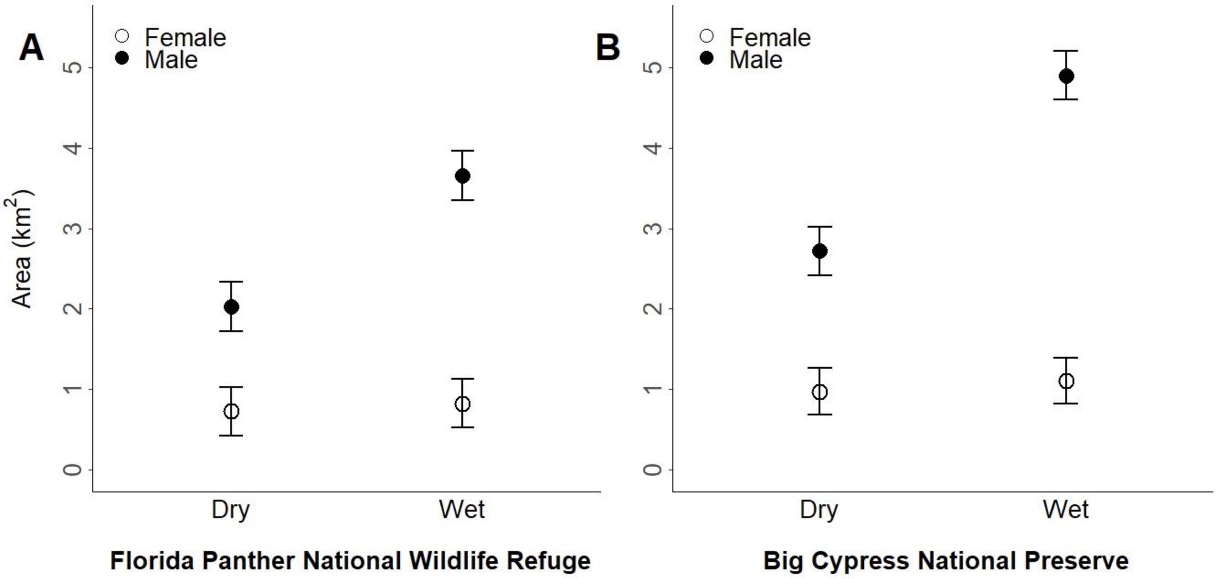


Figure 12. Property-specific mean seasonal home range sizes (km^2) for males and females during hydrological seasons (wet, May - October; dry, November - April) in A) Florida Panther National Wildlife Refuge and B) Big Cypress National Preserve for all deer qualifying for analyses from May 2015 – October 2018 (*n* = 188). Error bars indicate 95% confidence intervals.

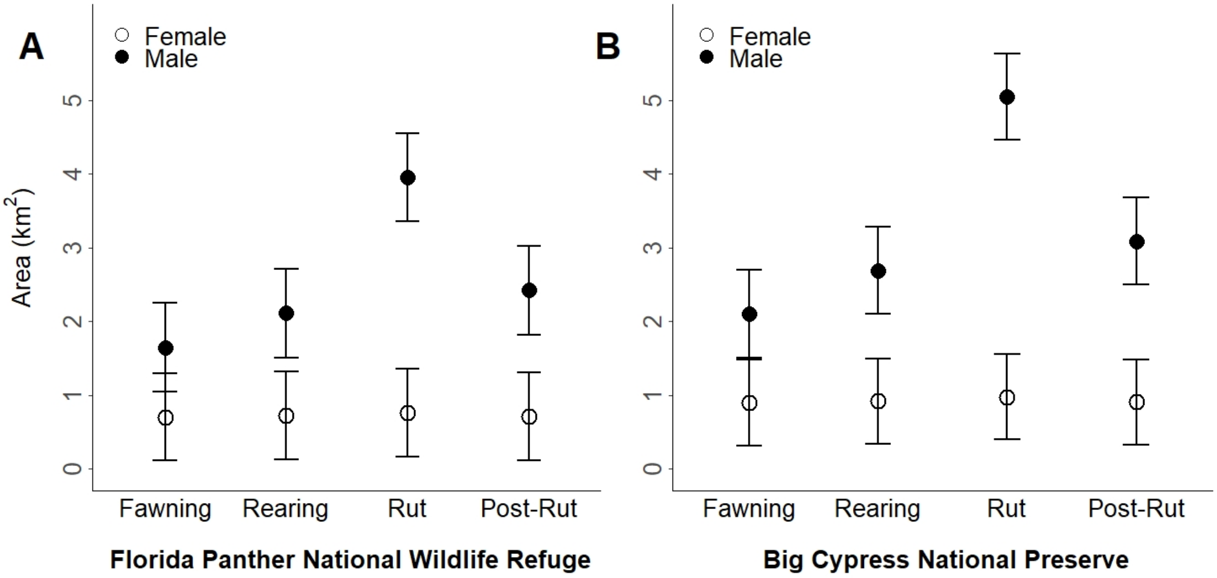


Figure 13. Property-specific mean seasonal home range size (km^2) for male and female white-tailed deer during the four biological seasons (fawning, January – March; fawn-rearing, April – June; rut, July – September; post-rut, October – December) in A) Florida Panther National Wildlife Refuge and B) Big Cypress National Preserve for all deer qualifying for analyses from January 2015 – December 2018 (*n* = 188). Error bars indicate 95% confidence intervals.

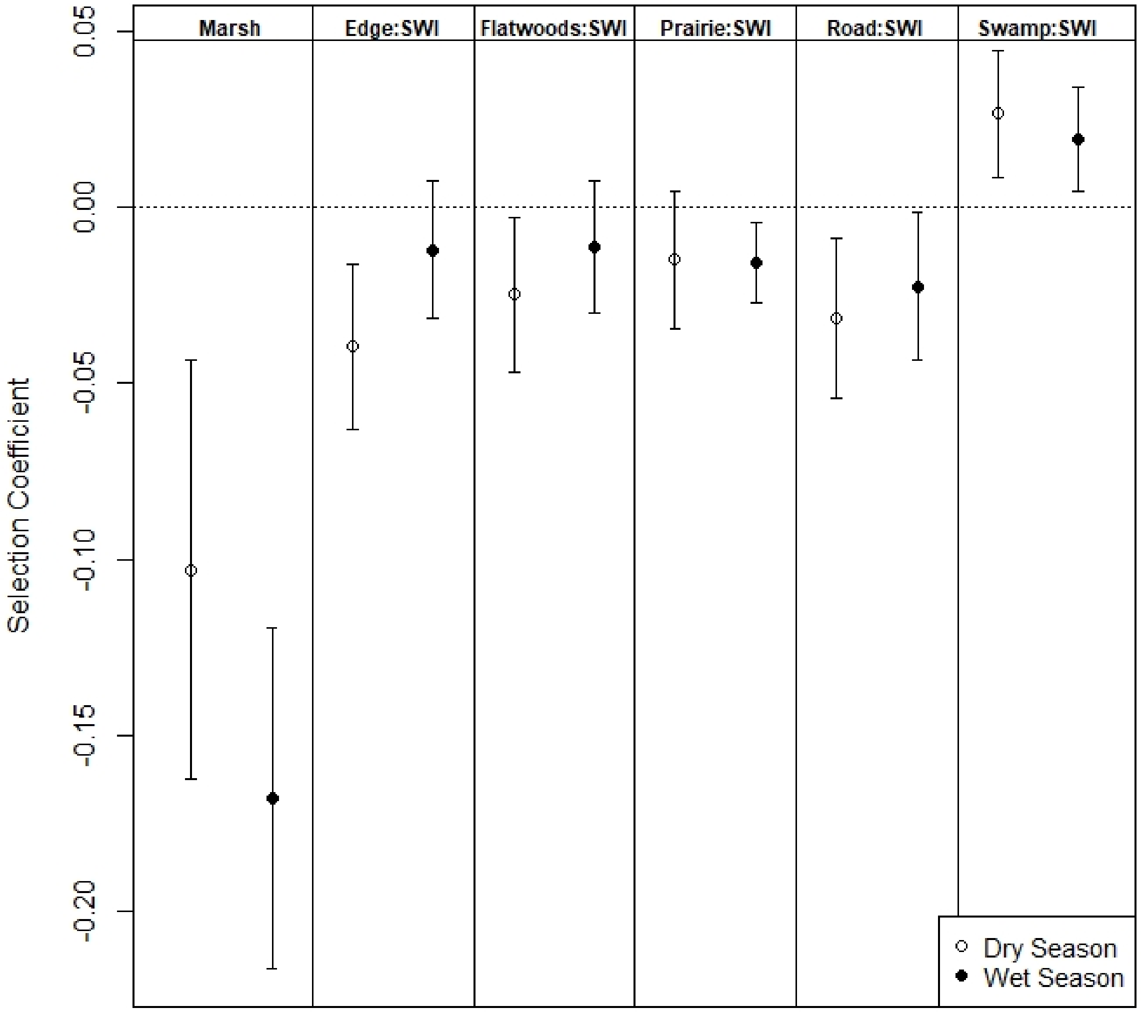


Figure 14. Significant season-specific selection coefficients for all male white-tailed deer eligible for analysis (Wet: *n* = 70; Dry: *n* = 78) during January 2015 – December 2018 in Big Cypress National Preserve and Florida Panther National Wildlife Refuge with respect to distance to habitat or feature and surface water index (SWI). Negative values indicate selection for a habitat or feature, and positive values indicate avoidance. Error bars indicate 95% confidence intervals, and the dotted line indicates no selection.

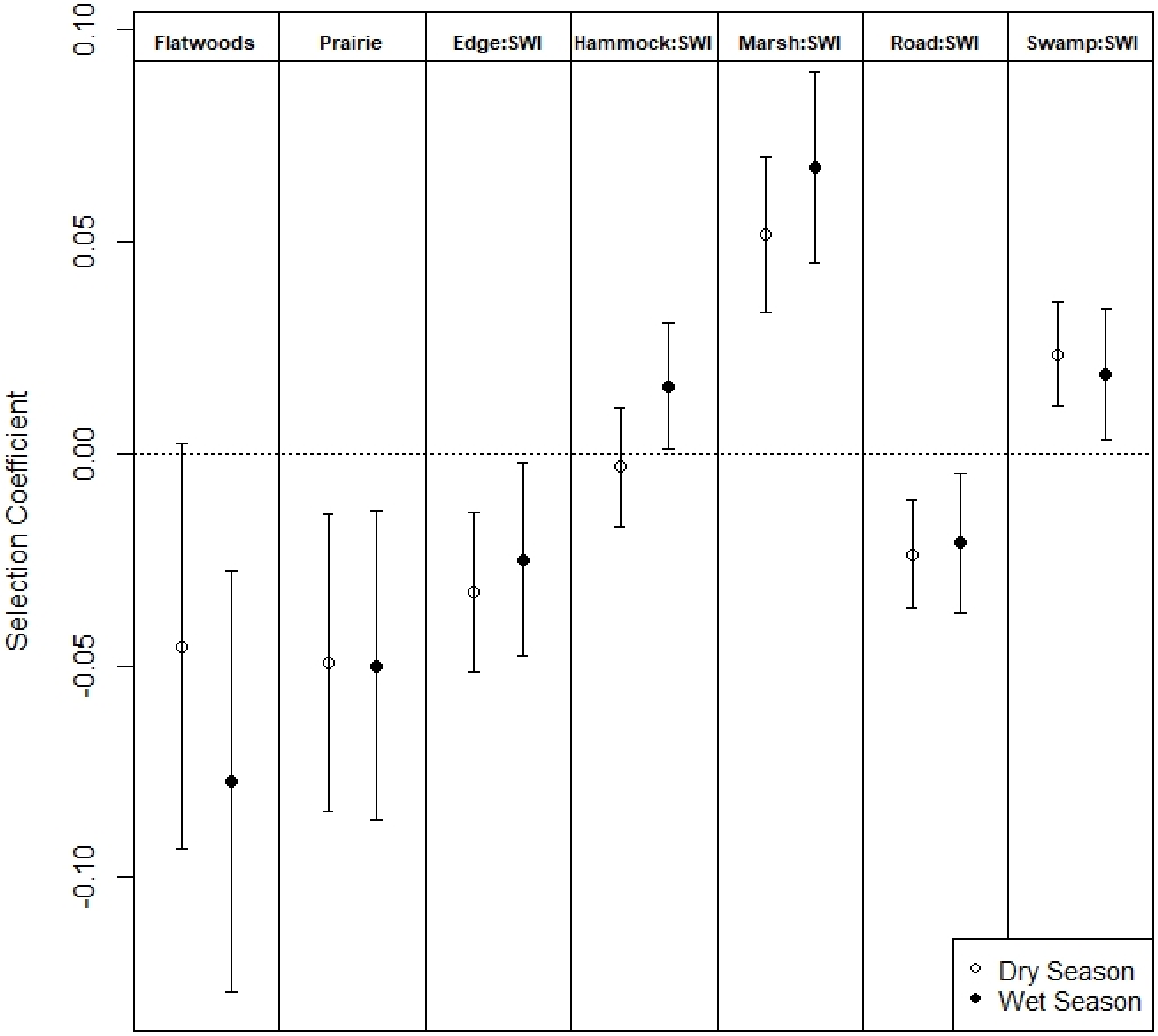


Figure 15. Significant season-specific selection coefficients for all female white-tailed deer eligible for analysis (Wet: *n* = 120; Dry: *n* = 143) during January 2015 – December 2018 in Big Cypress National Preserve and Florida Panther National Wildlife Refuge with respect to distance to habitat or feature and surface water index (SWI). Negative values indicate selection for a habitat or feature, and positive values indicate avoidance. Error bars indicate 95% confidence intervals, and the dotted line indicates no selection.

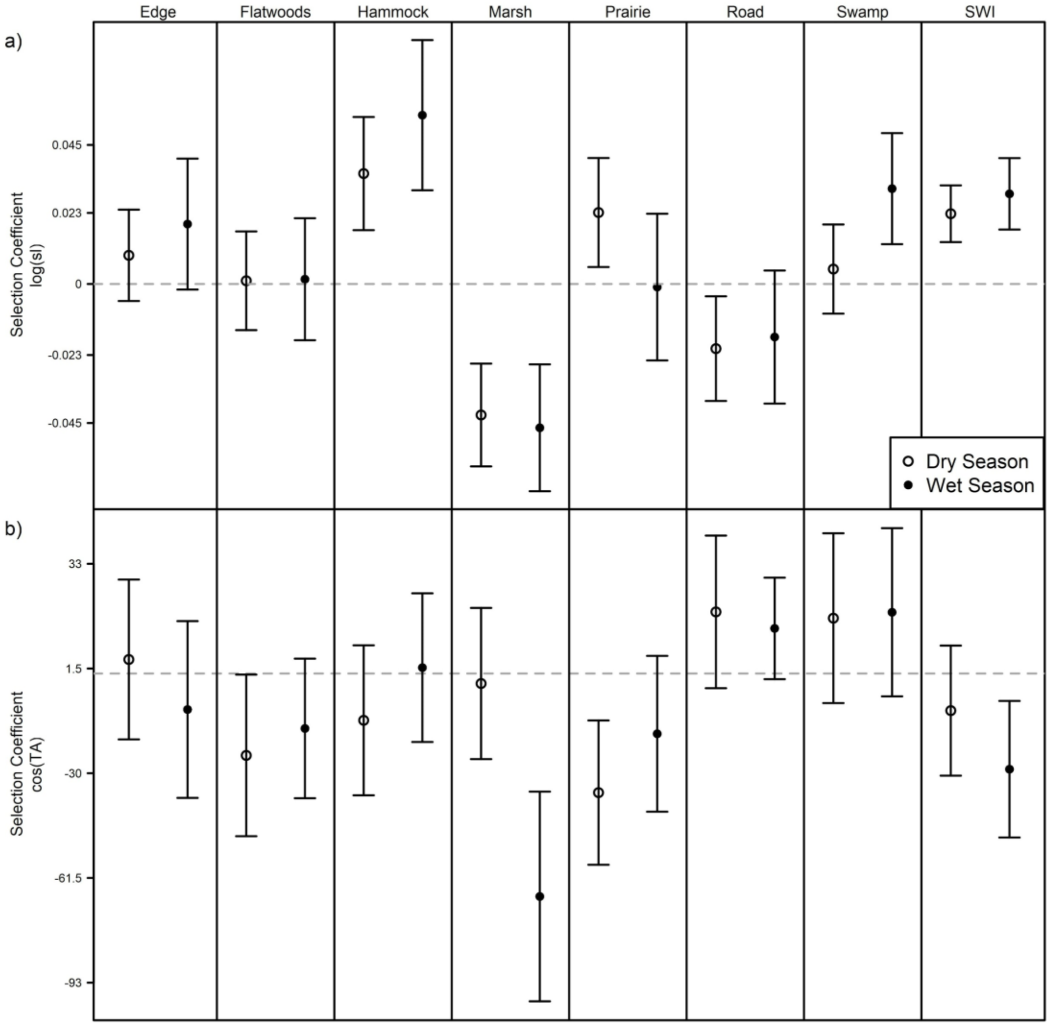


Figure 16. Season-specific interactions between a) step length (log(sl)) and surface water index (SWI) and distance to habitats and b) turn angle (cos(TA)) and surface water index (SWI) and distance to habitats for all male white-tailed deer qualifying for analyses (Dry: *n* = 78; Wet: *n* = 70) during January 2015 - December 2018 in Big Cypress National Preserve and Florida Panther National Wildlife Refuge. For distance-based covariates, positive step length and turn angle values indicate shorter step lengths and more linear paths near a habitat, respectively. Error bars indicate 95% confidence intervals, and the dotted line indicates no selection.

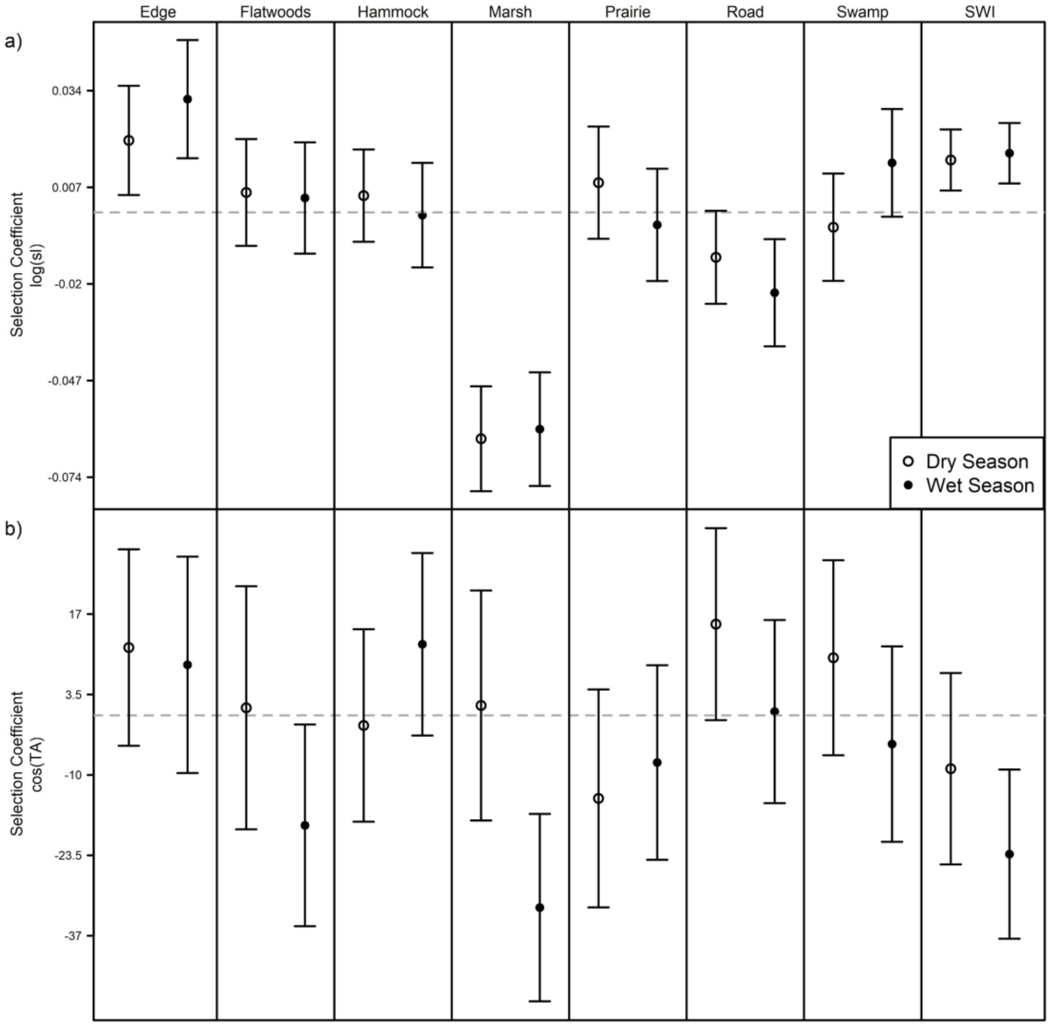


Figure 17. Season-specific interactions between a) step length (log(sl)) and surface water index (SWI) and distance to habitats and b) turn angle (cos(TA)) and surface water index (SWI) and distance to habitats for all female white-tailed deer qualifying for analyses (Dry: *n* = 143; Wet: *n* = 120) during January 2015 - December 2018 in Big Cypress National Preserve and Florida Panther National Wildlife Refuge. For distance-based covariates, positive step length and turn angle values indicate shorter step lengths and more linear paths near a habitat, respectively. Error bars indicate 95% confidence intervals, and the dotted line indicates no selection.

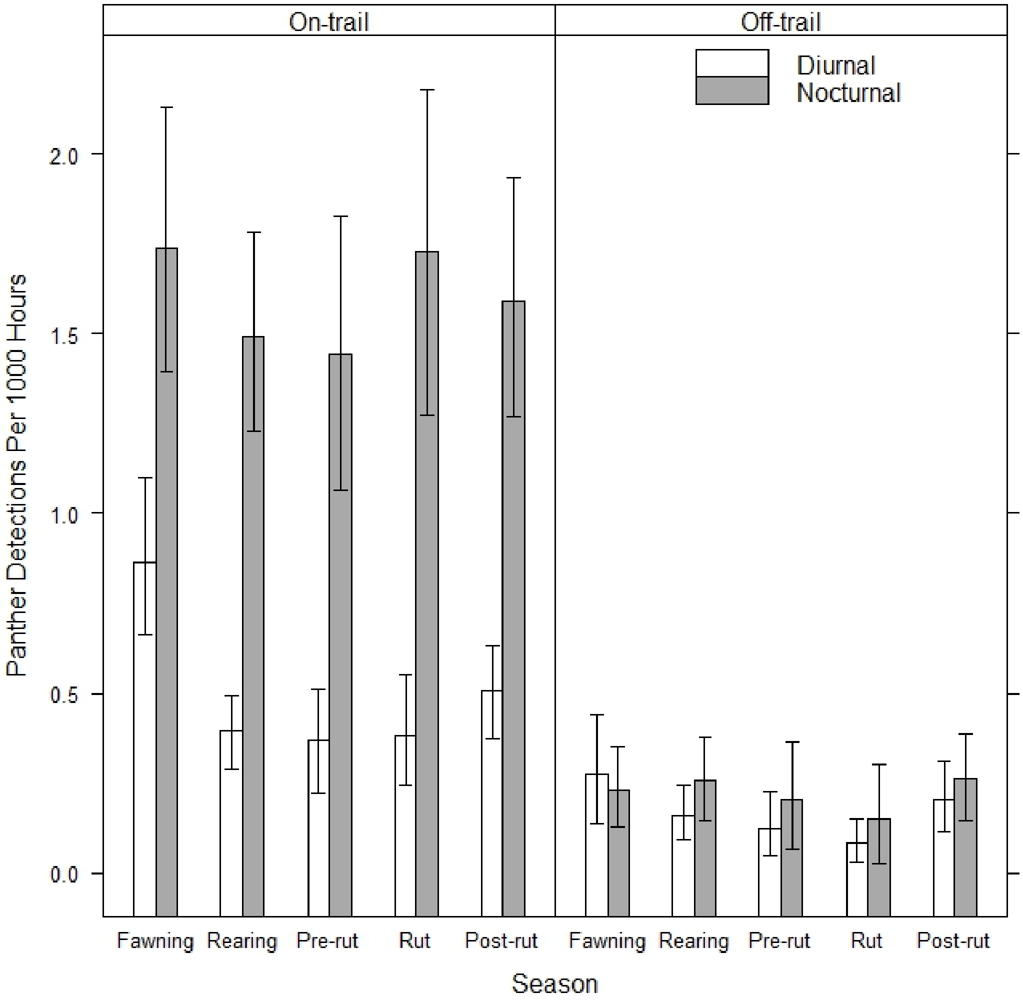


Figure 18. Diurnal (sunrise to sunset) and nocturnal (sunset to sunrise) panther detections per 1,000 hours at on- and off-trail camera traps during biological seasons of white-tailed deer during February – October 2015 in Big Cypress National Preserve and Florida Panther National Wildlife Refuge. Biological seasons include fawning (February - March), rearing (April - June), pre-rut (July), rut (August), and post-rut (September - October). Error bars indicate bootstrapped 95% confidence intervals.

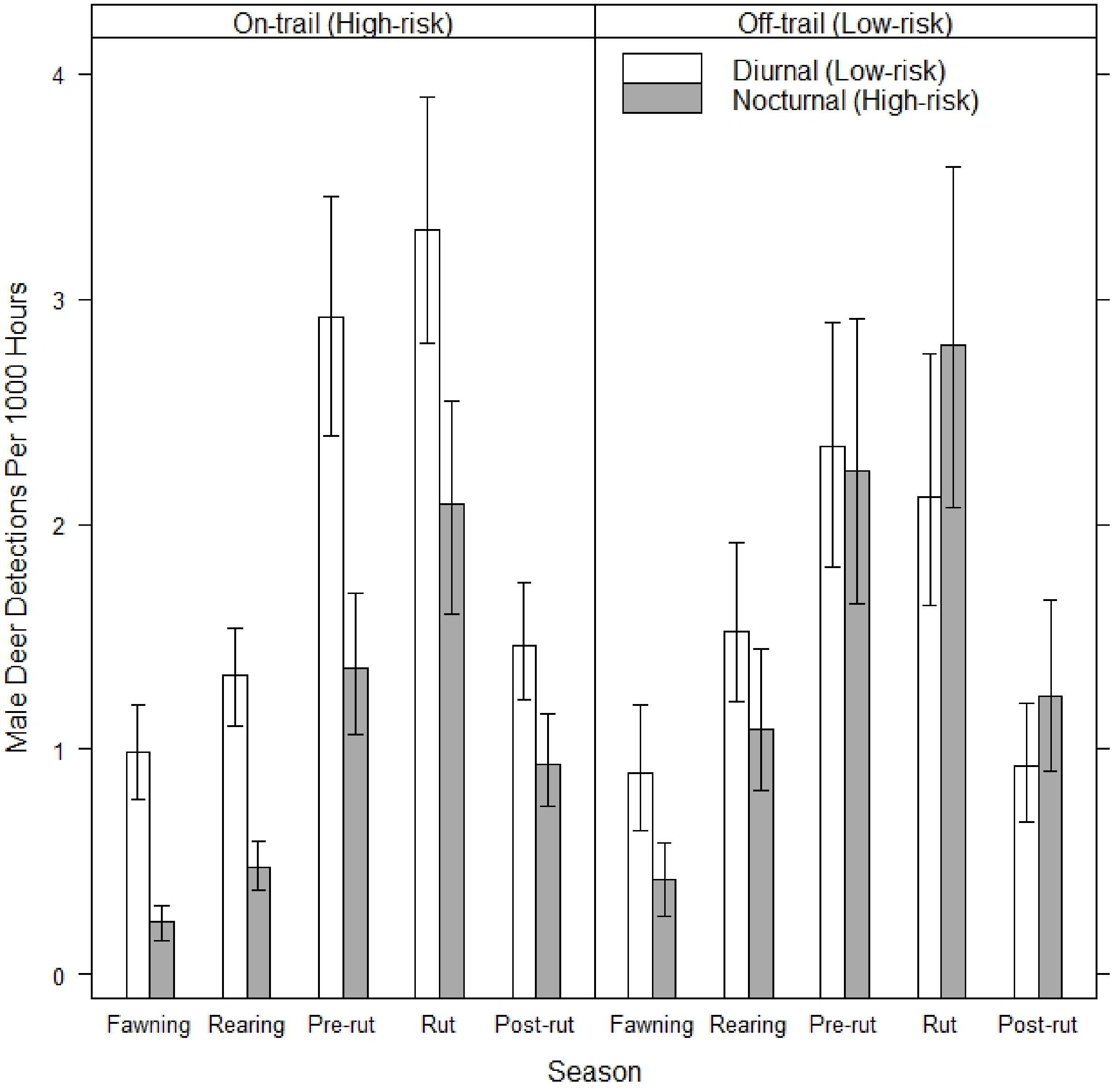


Figure 19. Diurnal (sunrise to sunset) and nocturnal (sunset to sunrise) male white-tailed deer detections per 1,000 hours at on- and off-trail camera traps during biological seasons of white-tailed deer during February – October 2015 in Big Cypress National Preserve and Florida Panther National Wildlife Refuge. Biological seasons include fawning (February - March), rearing (April - June), pre-rut (July), rut (August), and post-rut (September - October). Error bars indicate bootstrapped 95% confidence intervals.

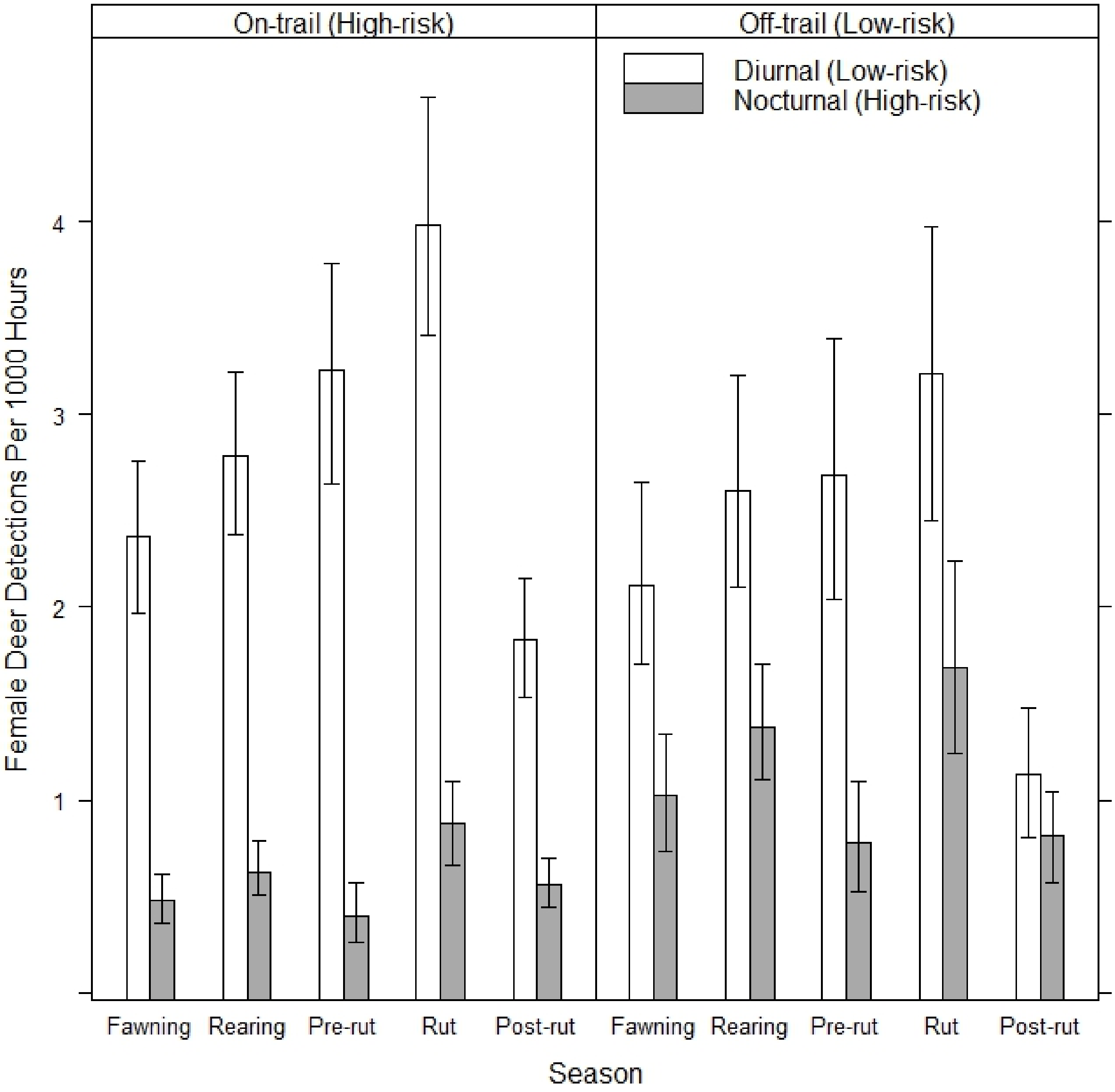


Figure 20. Diurnal (sunrise to sunset) and nocturnal (sunset to sunrise) female white-tailed deer detections per 1,000 hours at on- and off-trail camera traps during biological seasons of white-tailed deer during February – October 2015 in Big Cypress National Preserve and Florida Panther National Wildlife Refuge. Biological seasons include fawning (February - March), rearing (April - June), pre-rut (July), rut (August), and post-rut (September - October). Error bars indicate bootstrapped 95% confidence intervals.

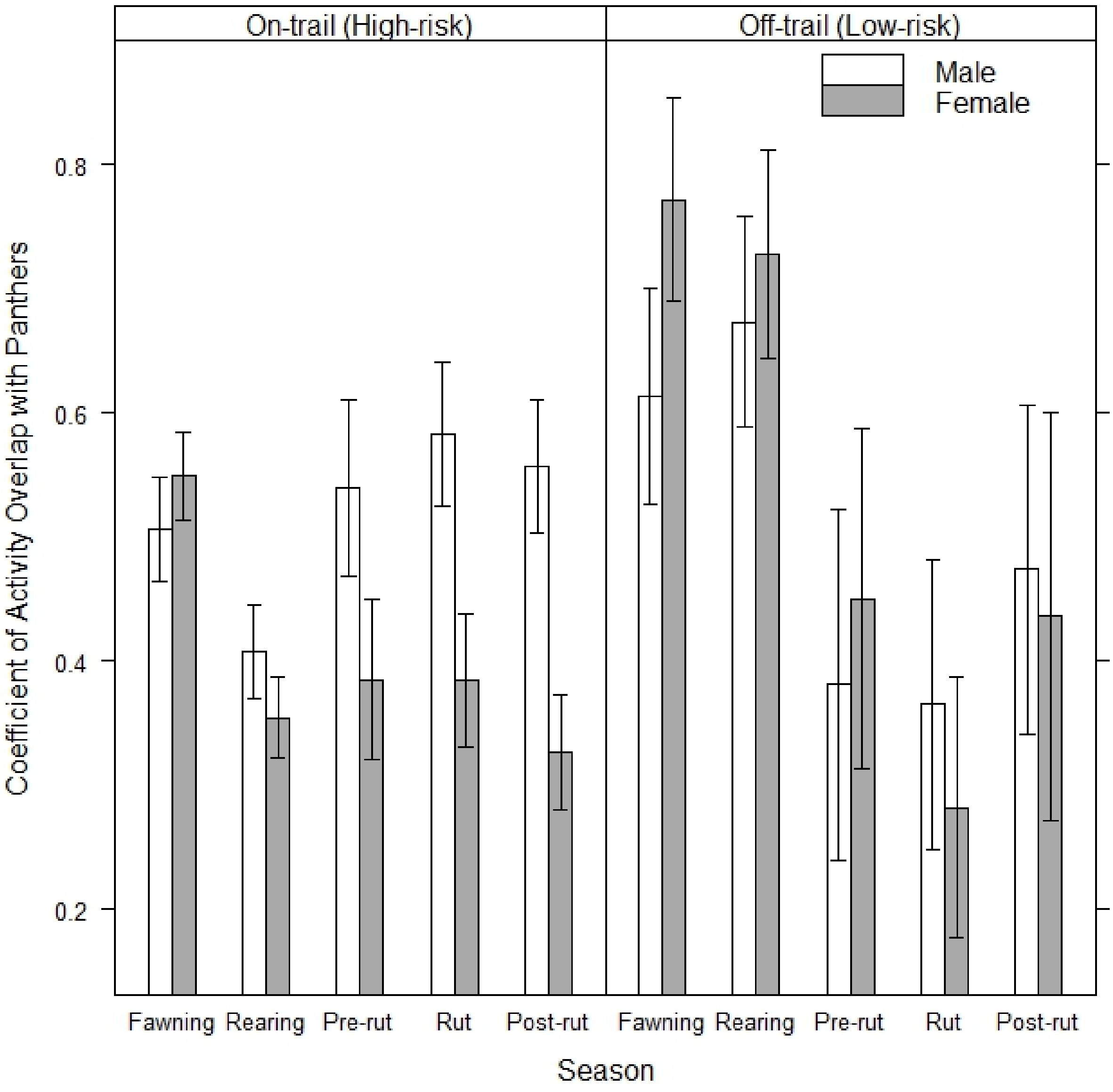


Figure 21. Overlap of male and female deer activity patterns with panther activity at on- and off-trail camera traps during biological seasons of white-tailed deer during February – October 2015 Big Cypress National Preserve and Florida Panther National Wildlife Refuge. Biological seasons include fawning (February - March), rearing (April - June), pre-rut (July), rut (August), and post-rut (September - October). Error bars indicate bootstrapped 95% confidence intervals.

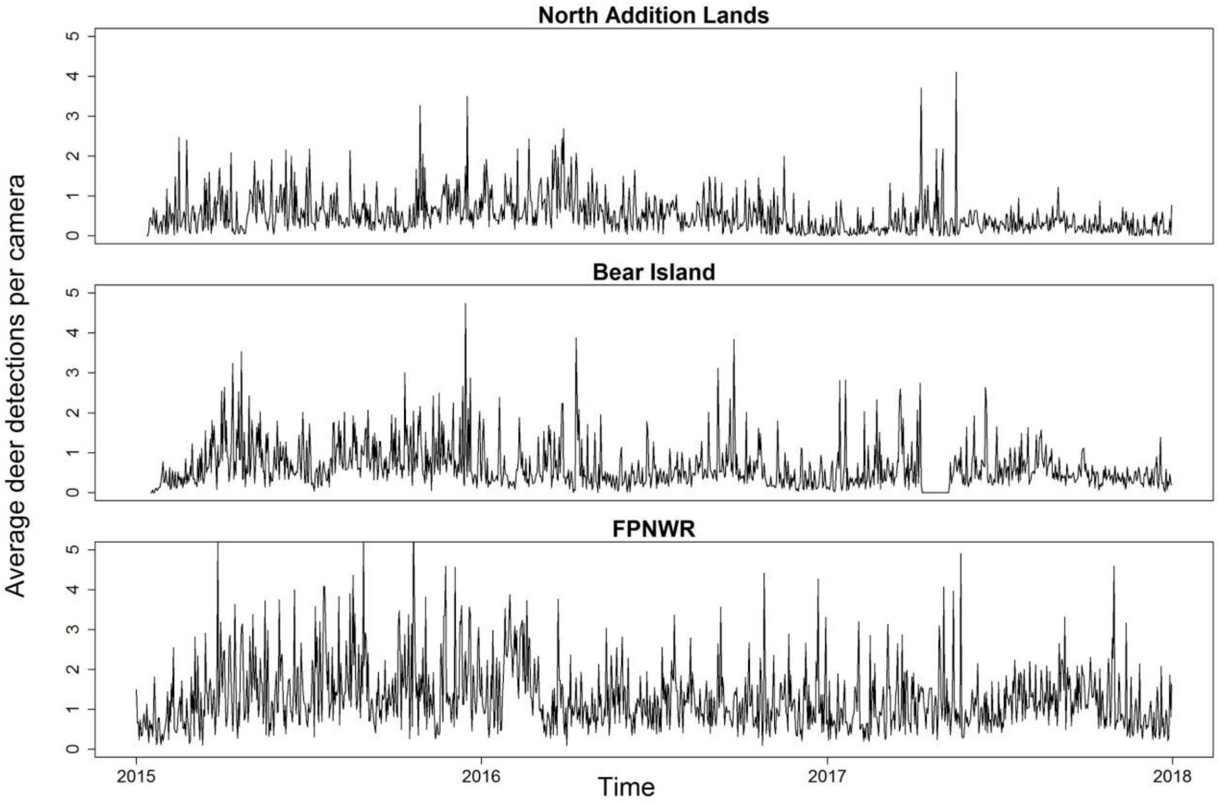


Figure 22. Daily white-tailed deer detections per trail camera day across the three trail camera sites in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge (FPNWR) from 1 January 2015 – 31 December 2017. We removed cameras from Bear Island during April - May 2017 due to wildfires.

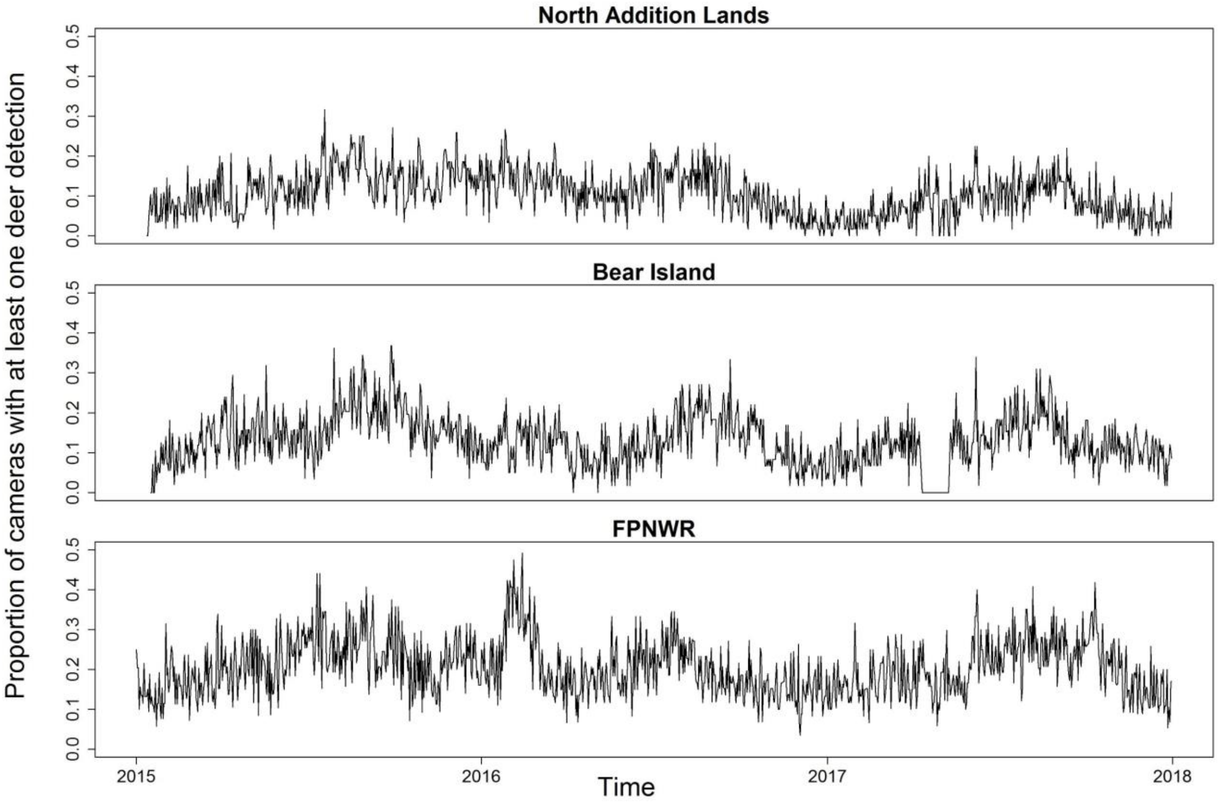


Figure 23. The proportion of trail cameras with at least one white-tailed deer detection on each day from 1 January 2015 – 31 December 2017 at the three trail camera sites in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge (FPNWR). We removed cameras from Bear Island during April - May 2017 due to wildfires.

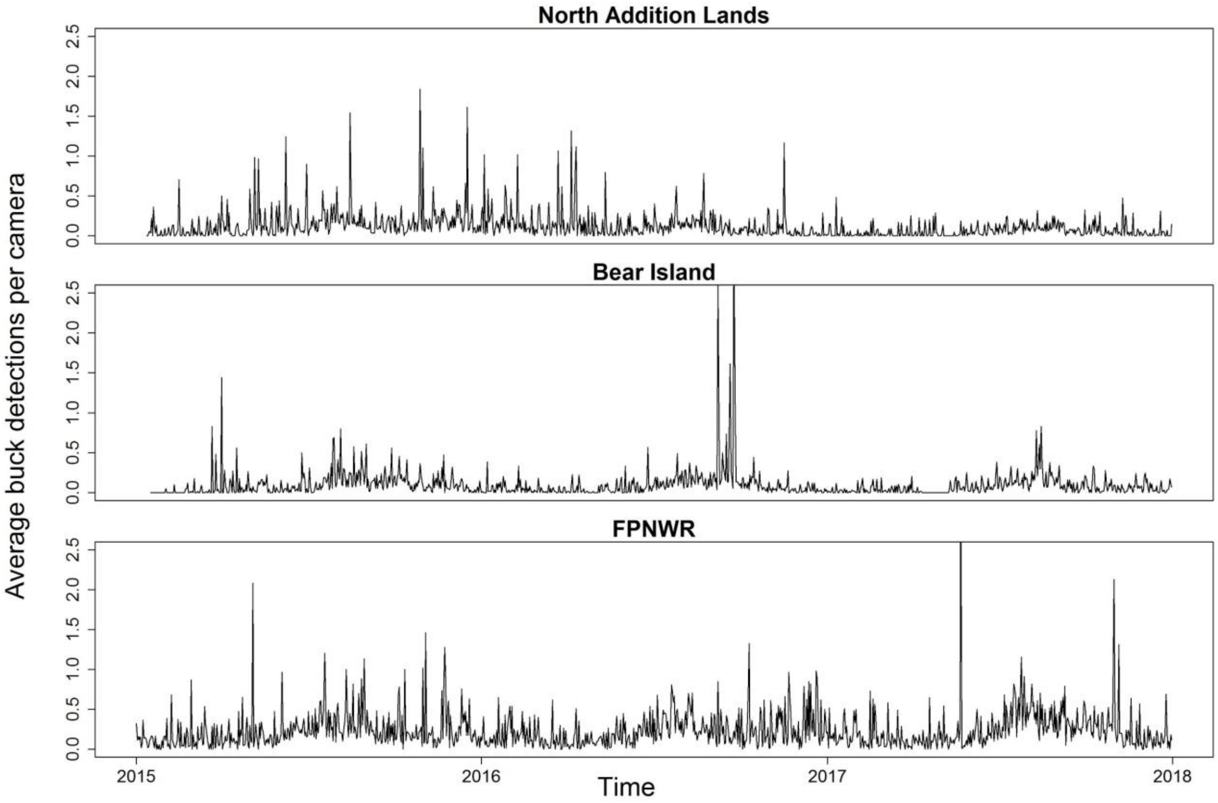


Figure 24. Daily male white-tailed detections per trail camera day across the three trail camera sites in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge (FPNWR) during 1 January 2015 – 31 December 2017. We removed cameras from Bear Island during April - May 2017 due to wildfires.

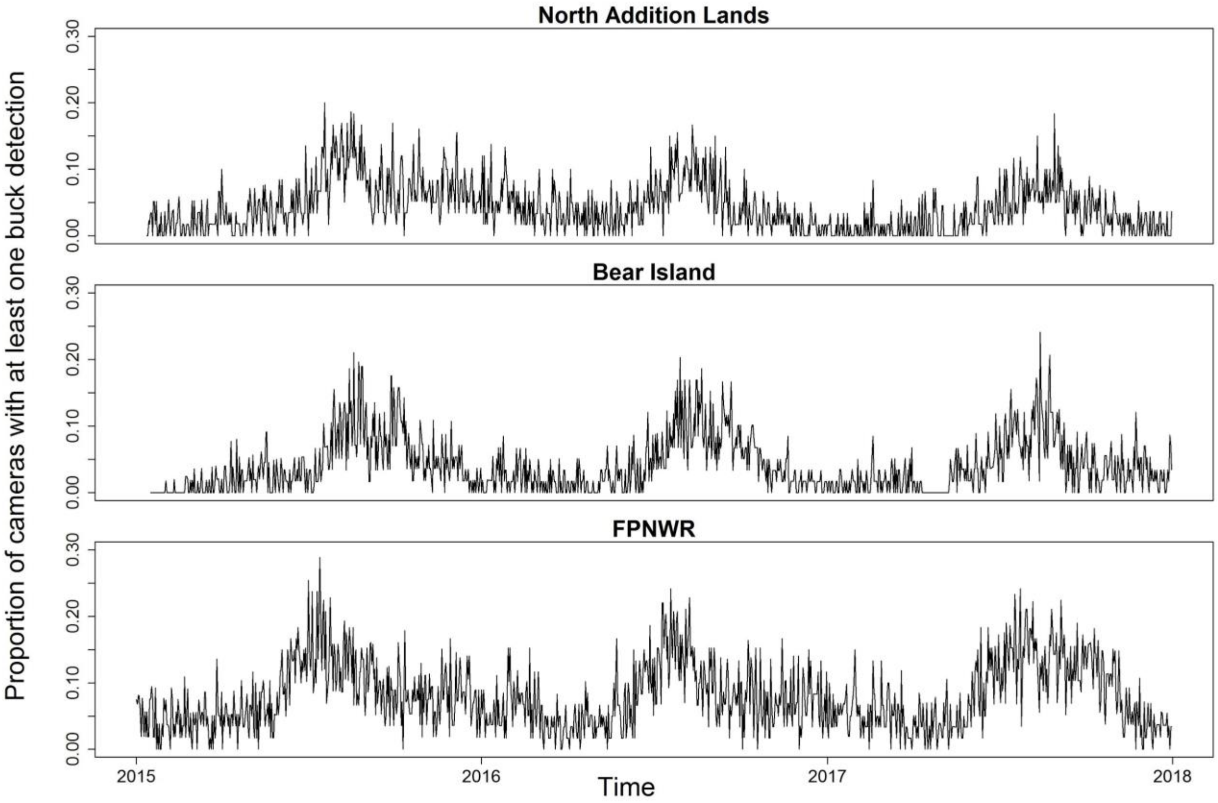


Figure 25. The proportion of trail cameras with at least one male white-tailed deer detection on each day from 1 January 2015 – 31 December 2017 at the three trail camera sites in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge (FPNWR). We removed cameras from Bear Island during April - May 2017 due to wildfires.

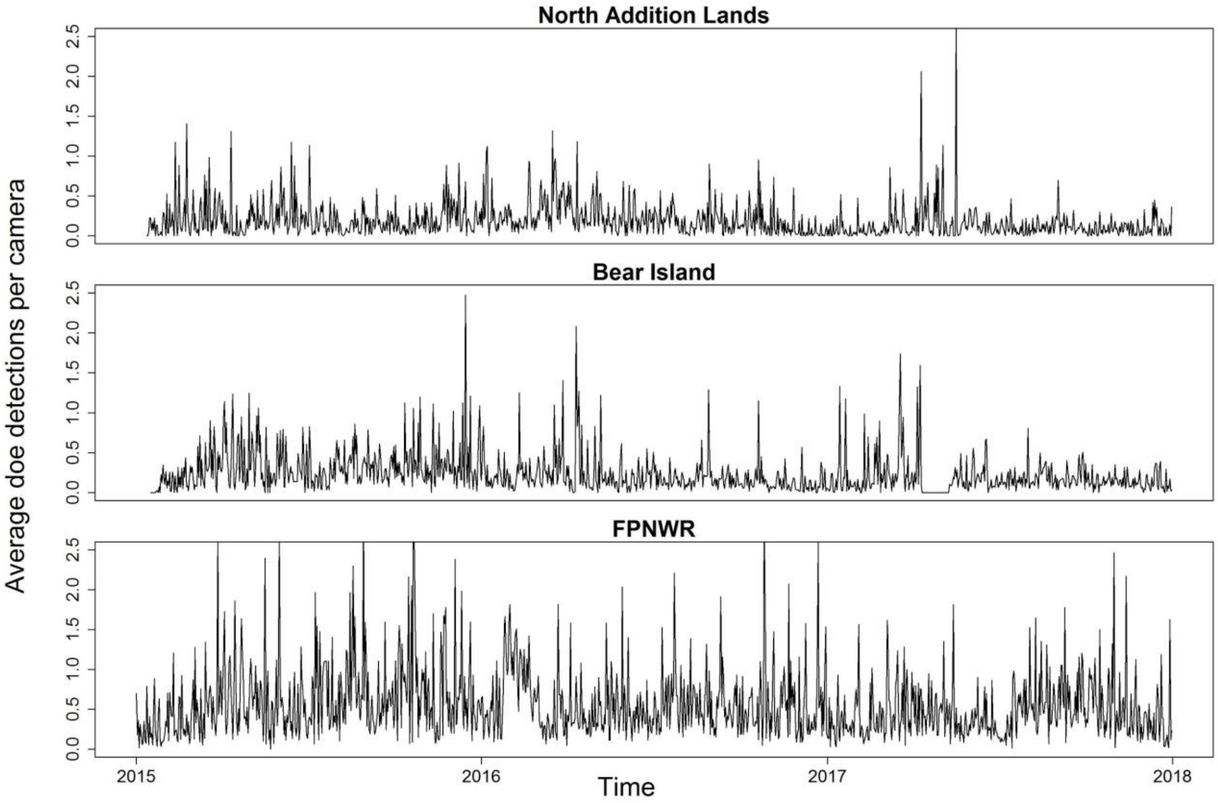


Figure 26. Daily female white-tailed detections per trail camera day across the three trail camera sites in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge (FPNWR) from 1 January 2015 – 31 December 2017. We removed cameras from Bear Island during April - May 2017 due to wildfires.

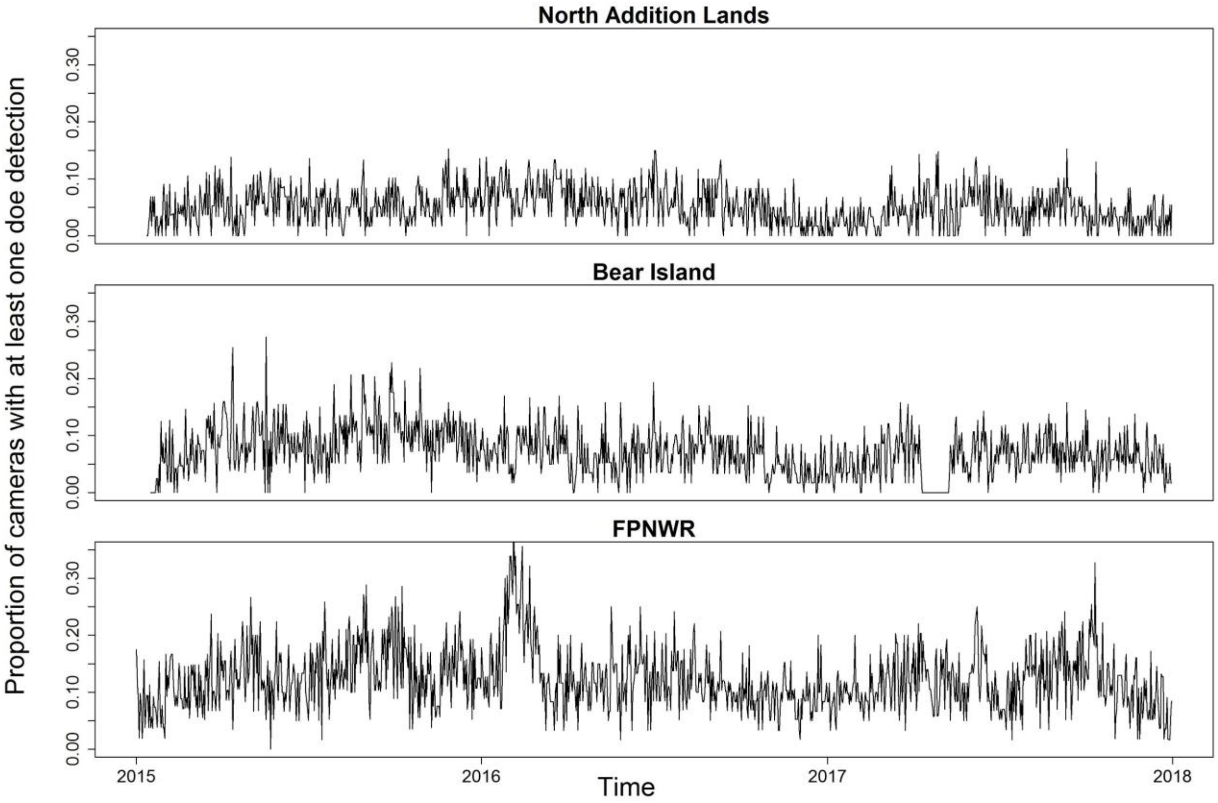


Figure 27. The proportion of trail cameras with at least one female white-tailed deer detection on each day from 1 January 2015 – 31 December 2017 at the three trail camera sites in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge (FPNWR). We removed cameras from Bear Island during April - May 2017 due to wildfires.

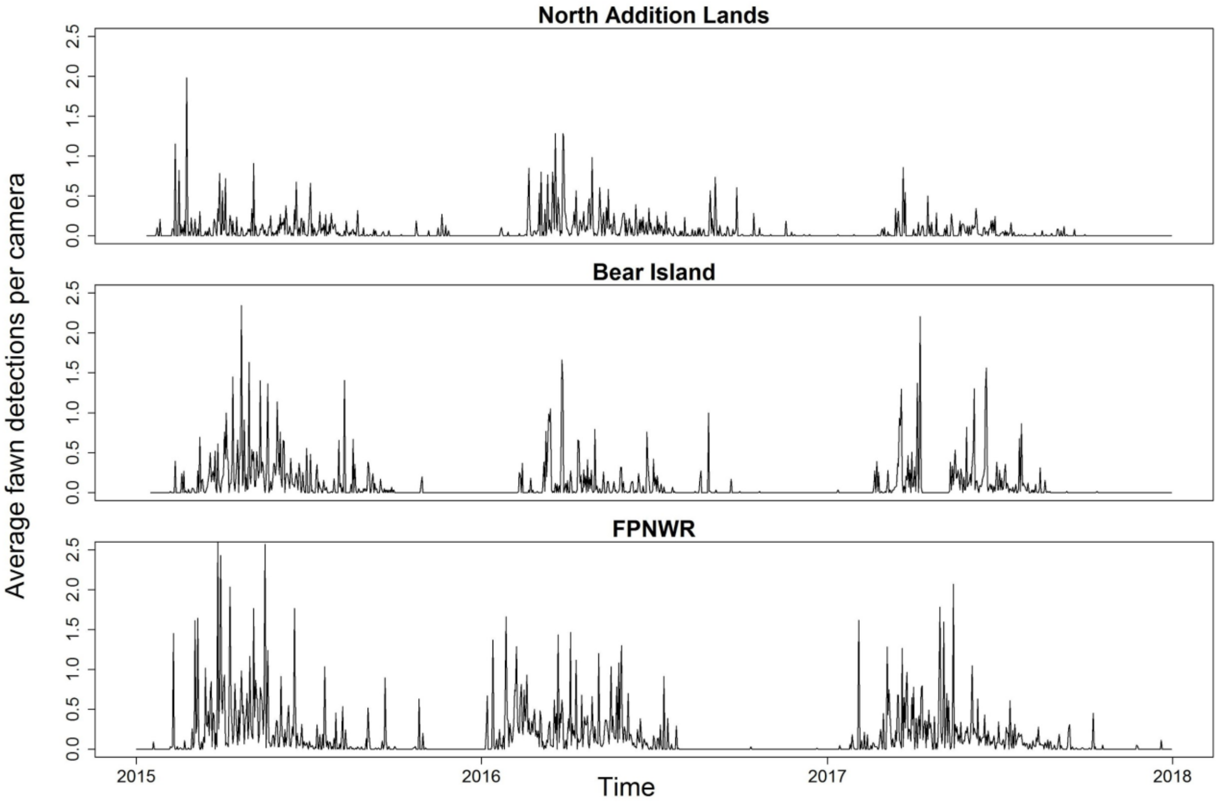


Figure 28. Daily white-tailed deer spotted fawn detections per trail camera day from 1 January 2015 – 31 December 2017 across the three trail camera sites in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge (FPNWR). We removed cameras from Bear Island during April - May 2017 due to wildfires.

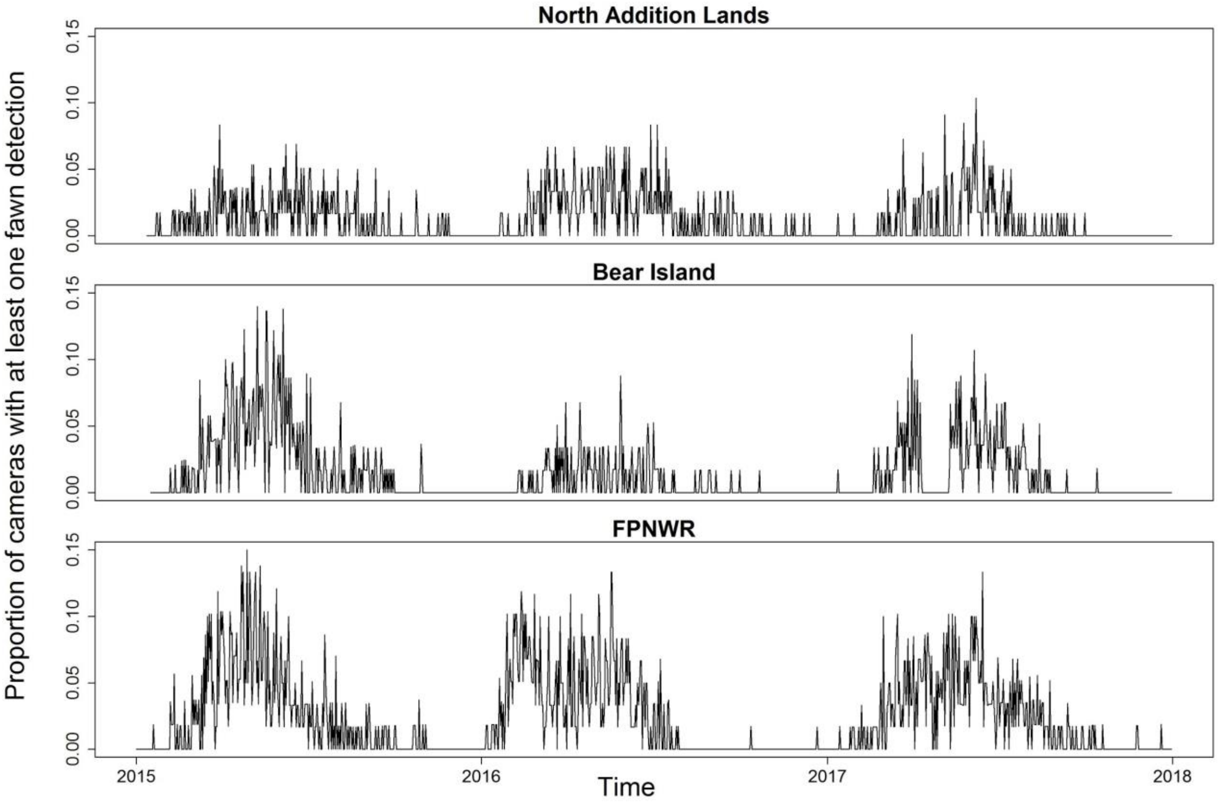


Figure 29. The proportion of trail cameras with at least one white-tailed deer spotted fawn detection on each day from 1 January 2015 – 31 December 2017 across the three trail camera sites in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge (FPNWR). We removed cameras from Bear Island during April - May 2017 due to wildfires.

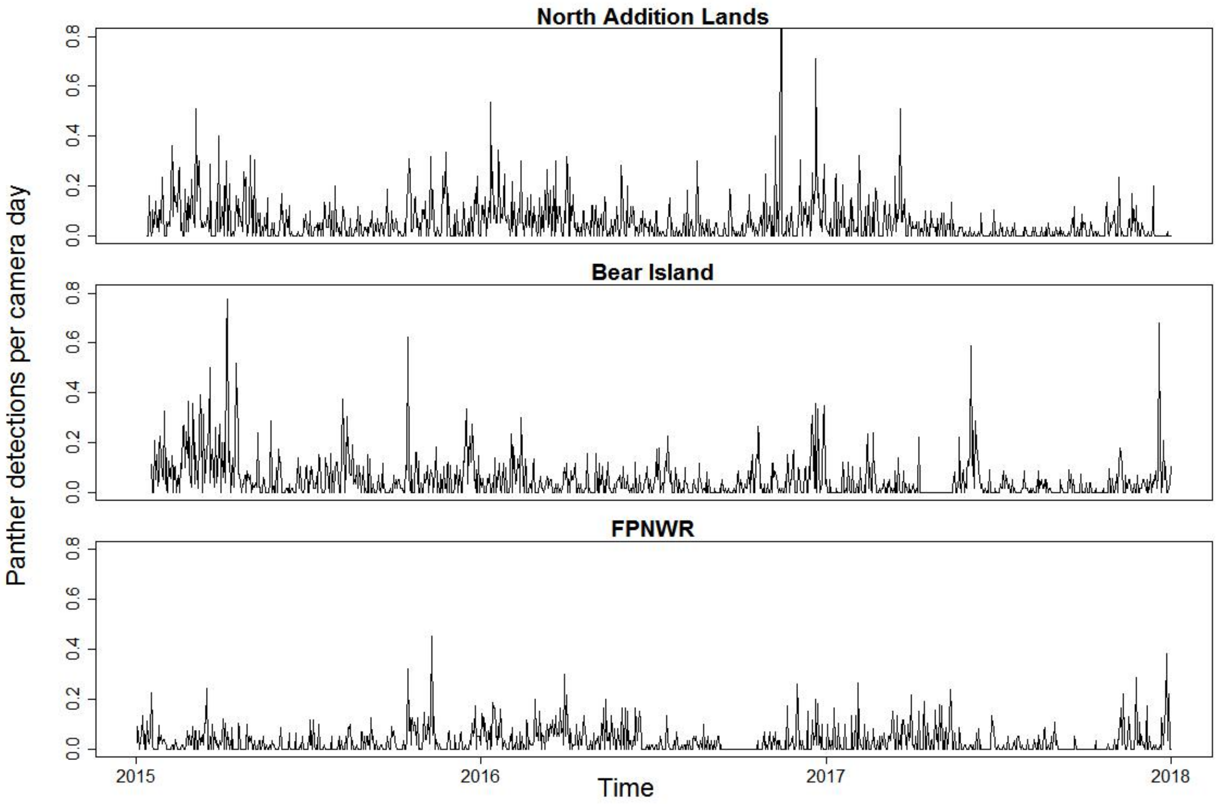


Figure 30. Daily panther detections per trail camera day across North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge (FPNWR)from 1 January 2015 – 31 December 2017. We removed cameras from Bear Island during April - May 2017 due to wildfires.

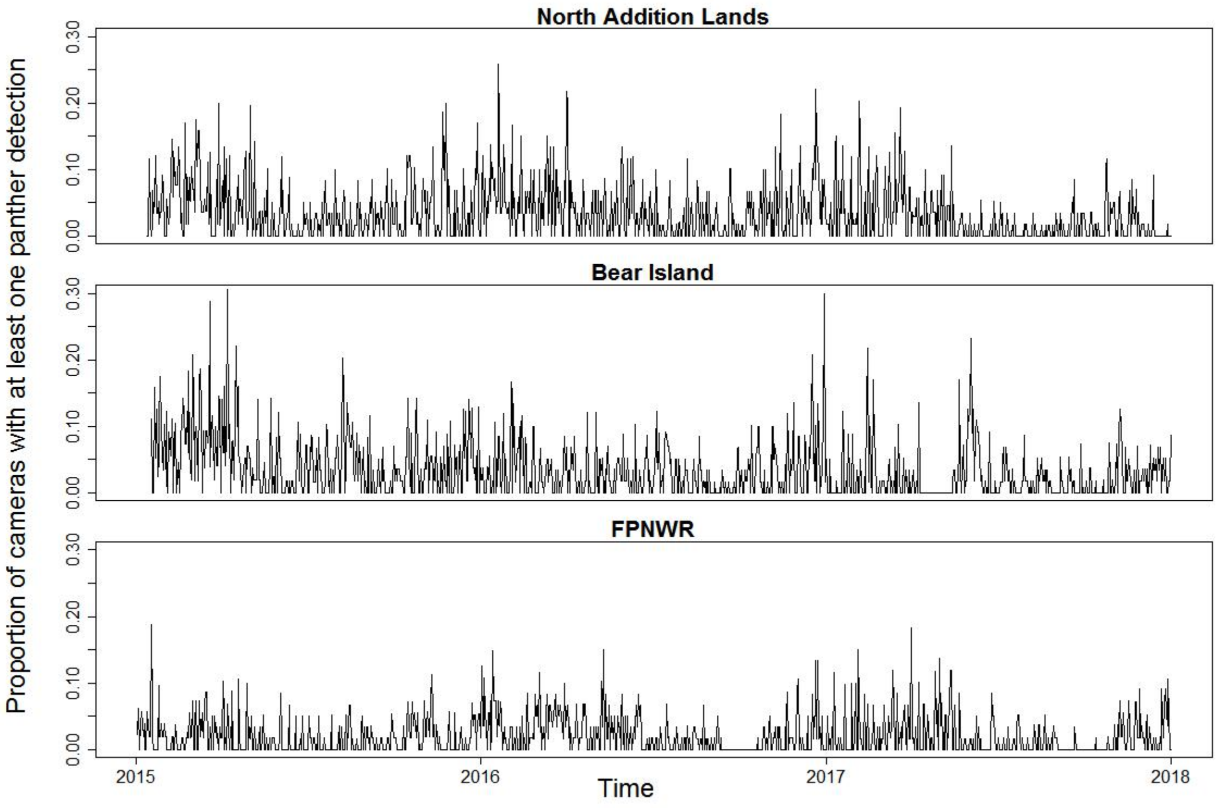


Figure 31. The proportion of trail cameras with at least one panther detection on each day from 1 January 2015 – 31 December 2017 across the three trail camera sites North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge (FPNWR).. We removed cameras from Bear Island during April - May 2017 due to wildfires.

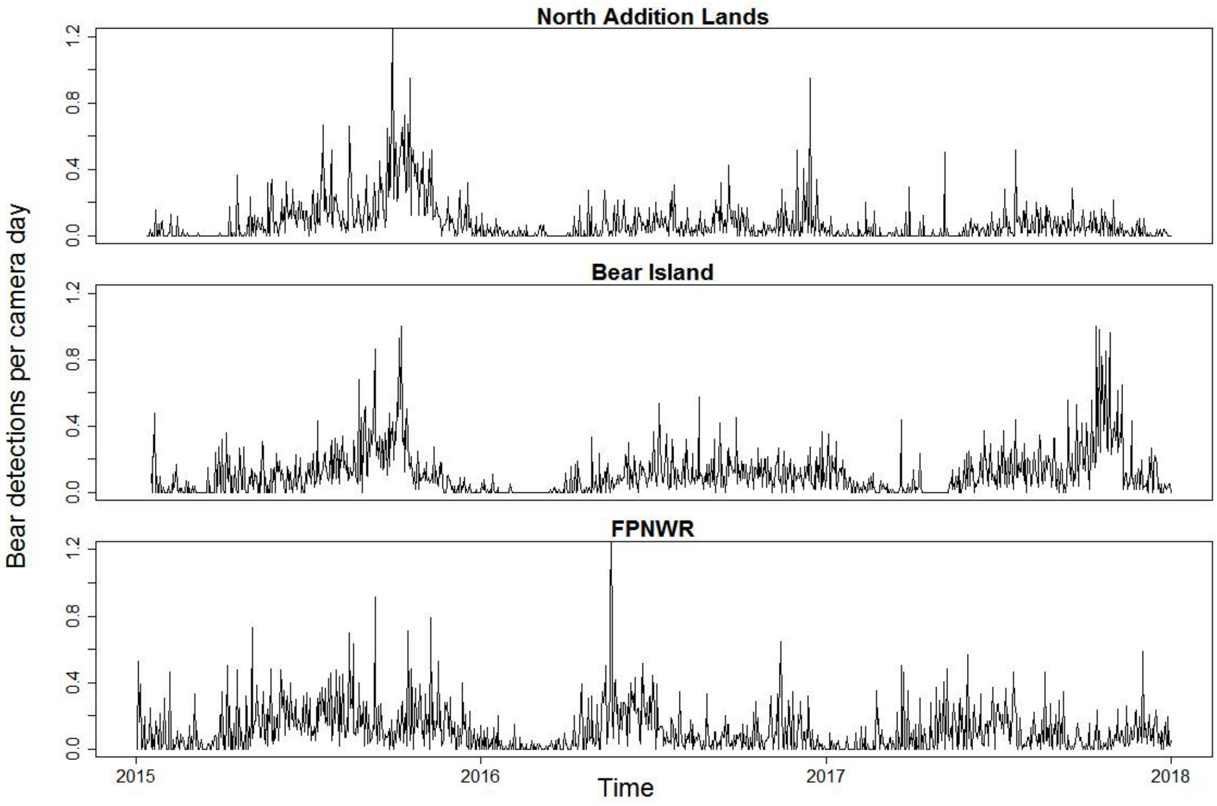


Figure 32. Daily bear detections per trail camera day across the three trail camera sites North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge (FPNWR) from 1 January 2015 – 31 December 2017. We removed cameras from Bear Island during April - May 2017 due to wildfires.

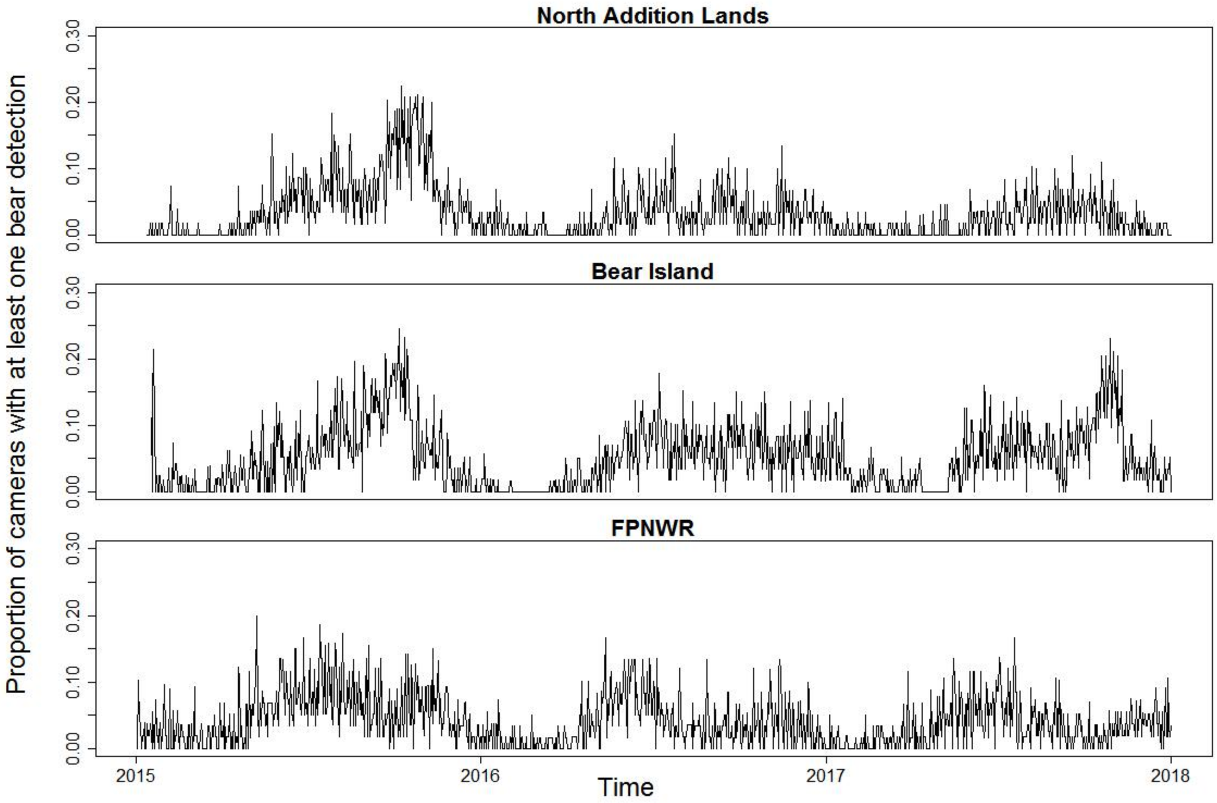


Figure 33. The proportion of trail cameras with at least one bear detection on each day from 1 January 2015 – 31 December 2017 across the three trail camera sites North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge (FPNWR). . We removed cameras from Bear Island during April - May 2017 due to wildfires.

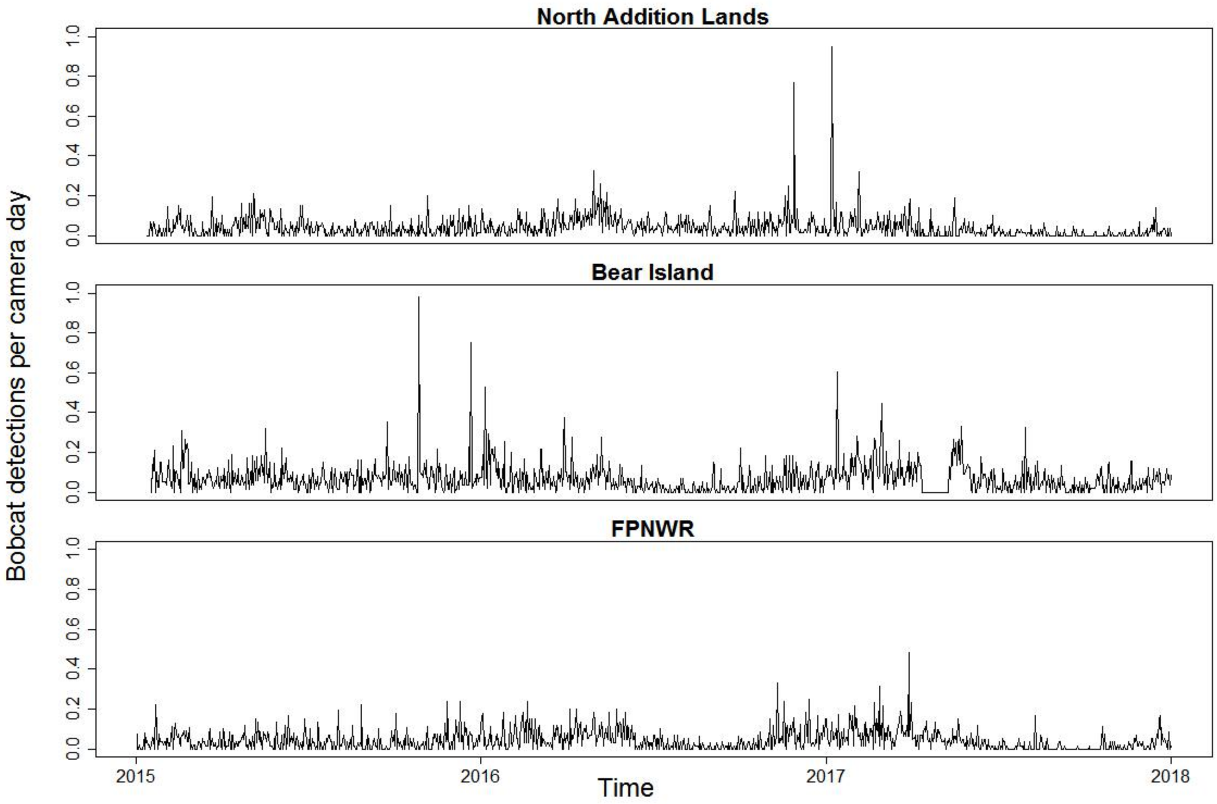


Figure 34. Daily bobcat detections per trail camera day in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge (FPNWR) from 1 January 2015 – 31 December 2017. We removed cameras from Bear Island during April - May 2017 due to wildfires.

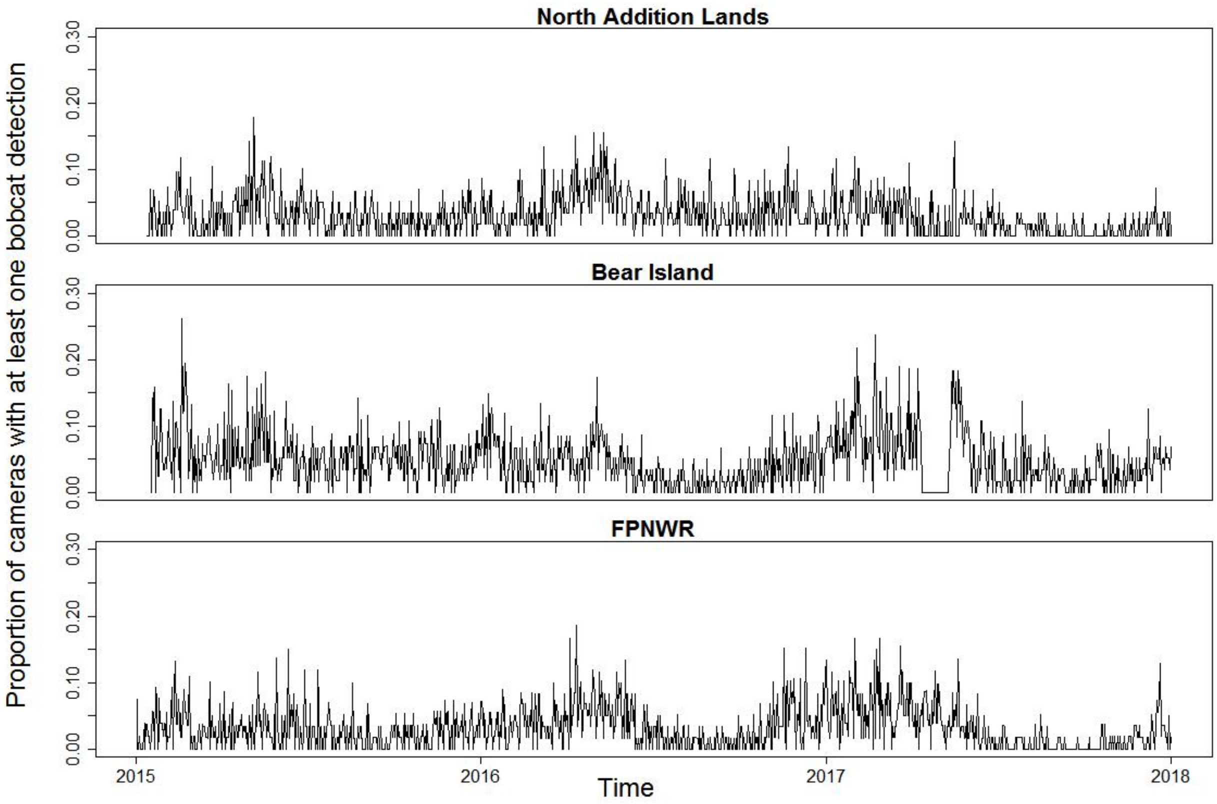


Figure 35. The proportion of trail cameras with at least one bobcat detection on each day from 1 January 2015 – 31 December 2017 across the three trail camera sites North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge (FPNWR). We removed cameras from Bear Island during April - May 2017 due to wildfires.

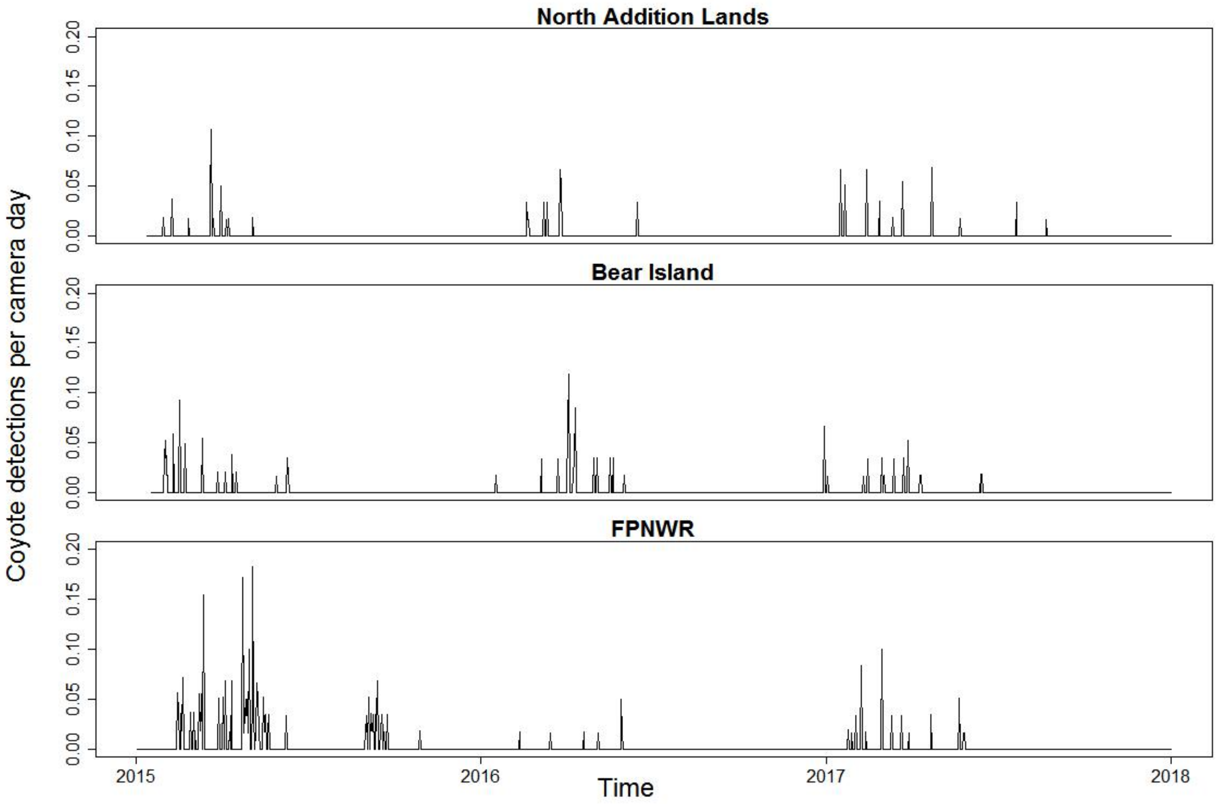


Figure 36. Daily coyote detections per trail camera day across the three trail camera sites in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge (FPNWR) from 1 January 2015 – 31 December 2017. We removed cameras from Bear Island during April - May 2017 due to wildfires.

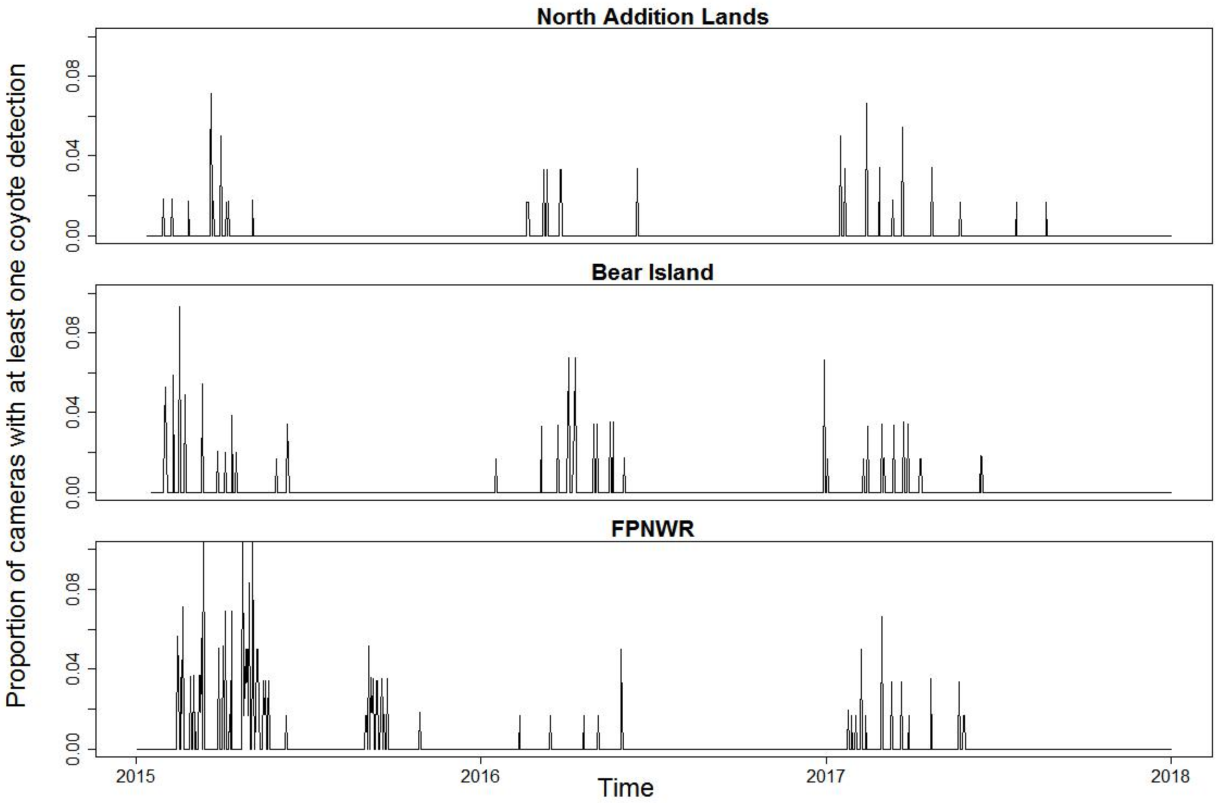


Figure 37. The proportion of trail cameras with at least one coyote detection on each day from 1 January 2015 – 31 December 2017 across the three trail camera sites in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge (FPNWR). We removed cameras from Bear Island during April - May 2017 due to wildfires.

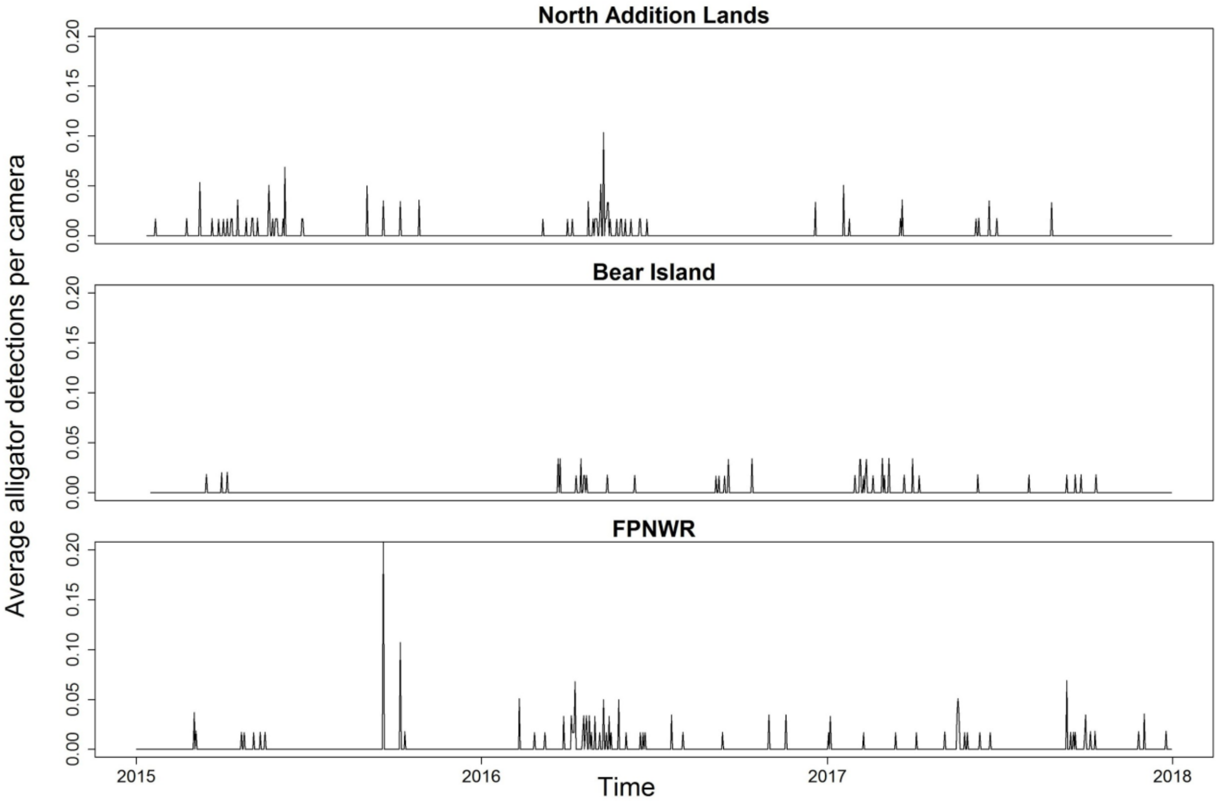


Figure 38. Daily alligator detections per trail camera day across the three trail camera sites in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge (FPNWR) from 1 January 2015 – 31 December 2017. We removed cameras from Bear Island during April - May 2017 due to wildfires.

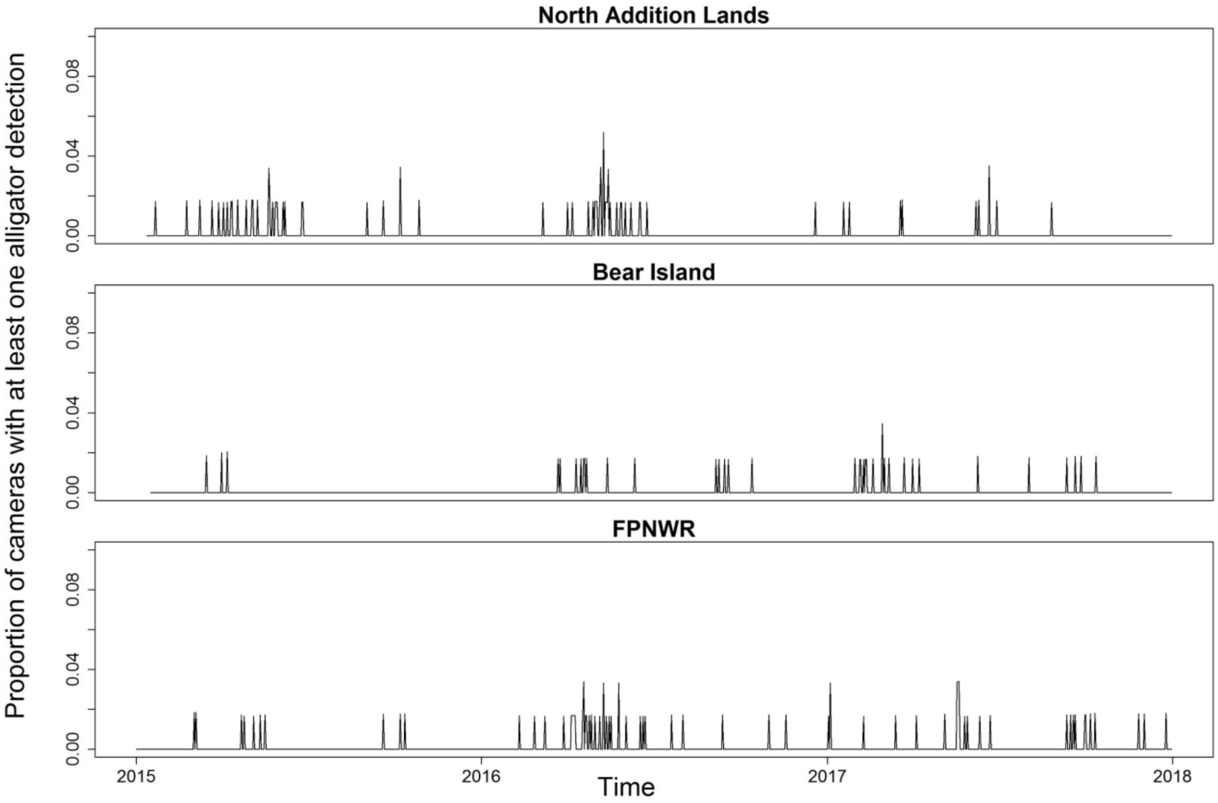


Figure 39. The proportion of trail cameras with at least one alligator detection on each day from 1 January 2015 - 31 December 2017 across the three trail camera sites in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge (FPNWR). We removed cameras from Bear Island during April - May 2017 due to wildfires.

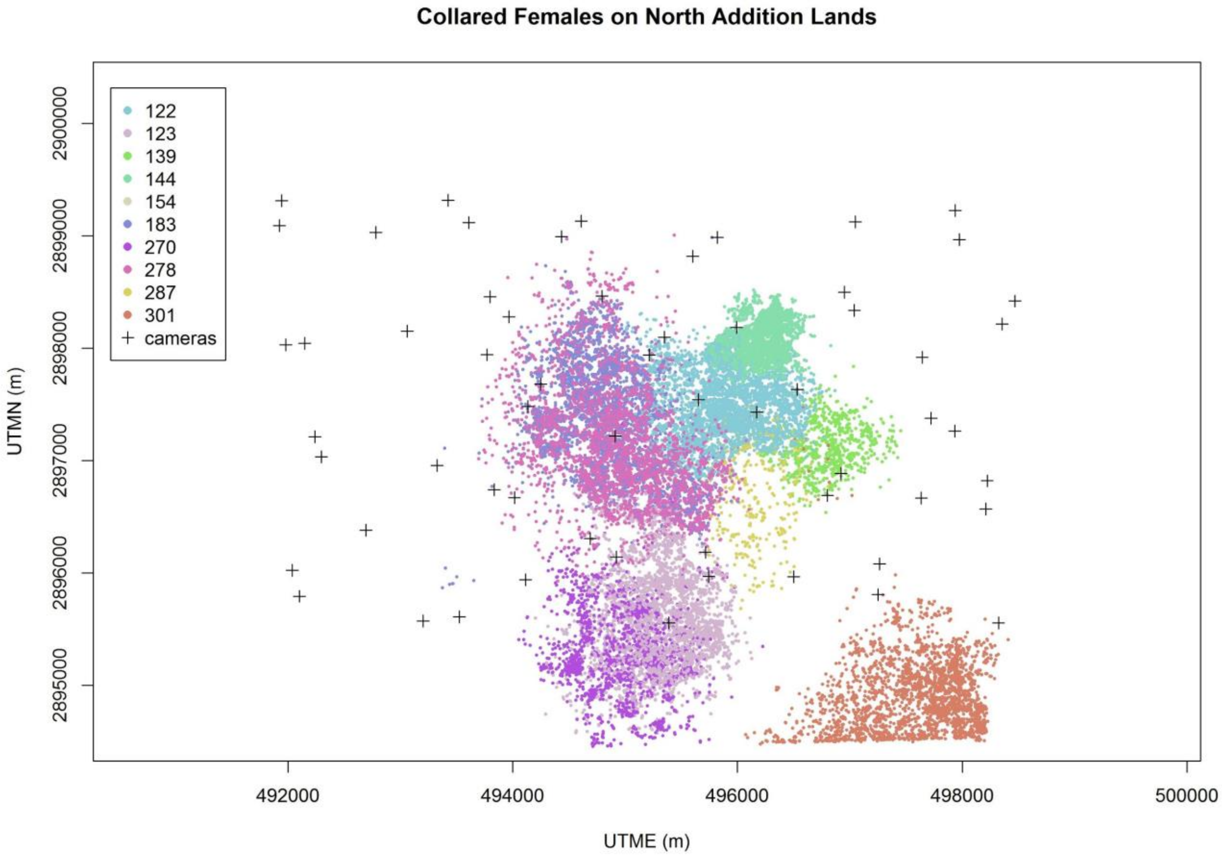


Figure 40. Telemetry locations of 10 GPS-collared white-tailed deer females during 1 January 2015 - 31 December 2017 on North Addition Lands in Big Cypress National Preserve used to estimate the detection probability parameters for the spatial capture-recapture model.

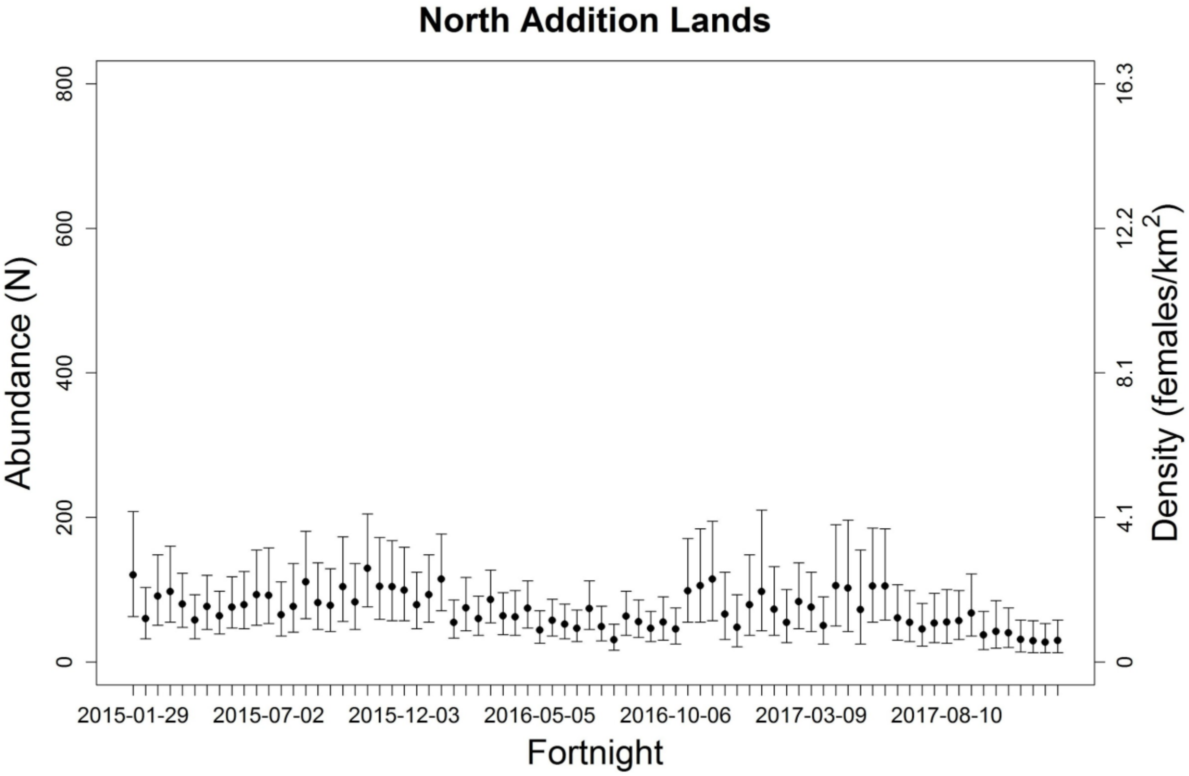


Figure 41. Biweekly estimates and 95% credible intervals of abundance and density for female white-tailed deer on North Addition Lands, Big Cypress National Preserve during 1 January 2015 – 31 December 2017.

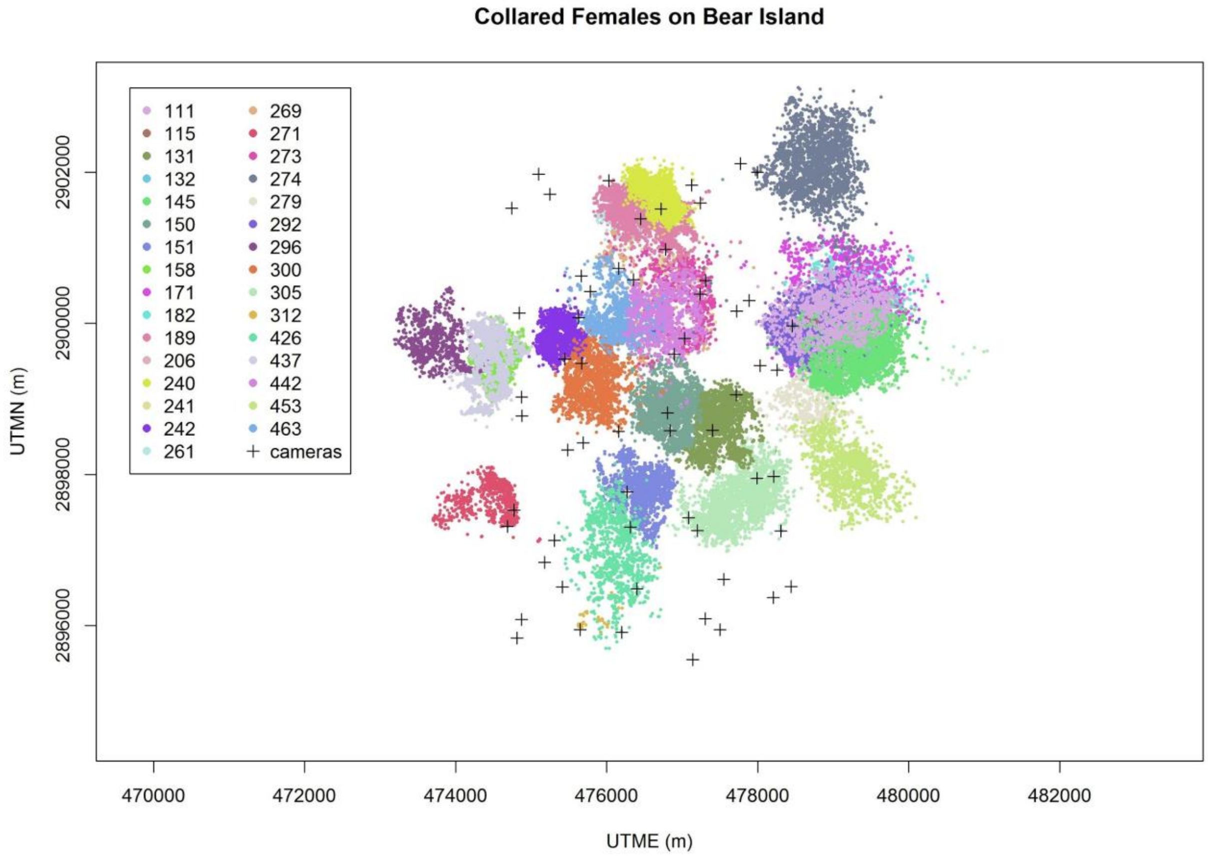


Figure 42. Telemetry locations of 31 GPS-collared female white-tailed deer on Bear Island, Big Cypress National Preserve, used to estimate the detection probability parameters for the spatial capture-recapture model during 1 January 2015 – 31 December 2017.

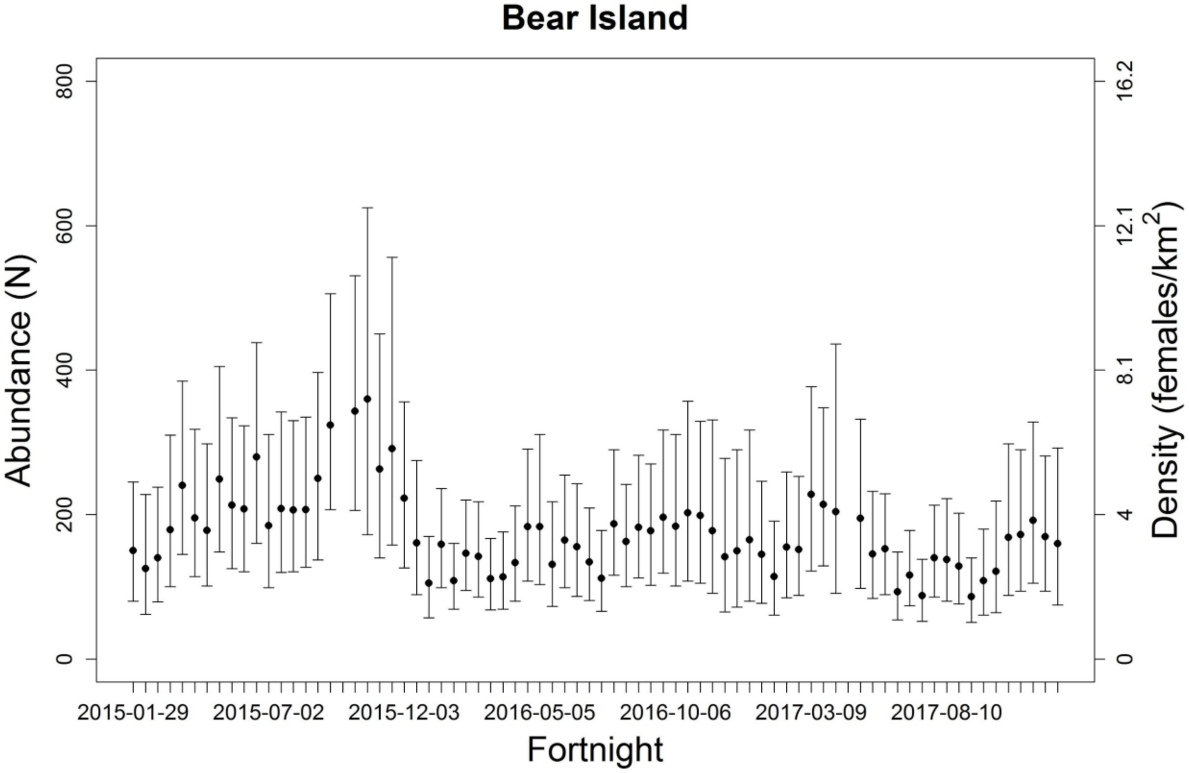


Figure 43. Biweekly estimates and 95% credible intervals of abundance and density for female white-tailed deer on Bear Island, Big Cypress National Preserve during 1 January 2015 – 31 December 2017. White-tailed deer density estimates for two fortnights were excluded due to models failing to converge.

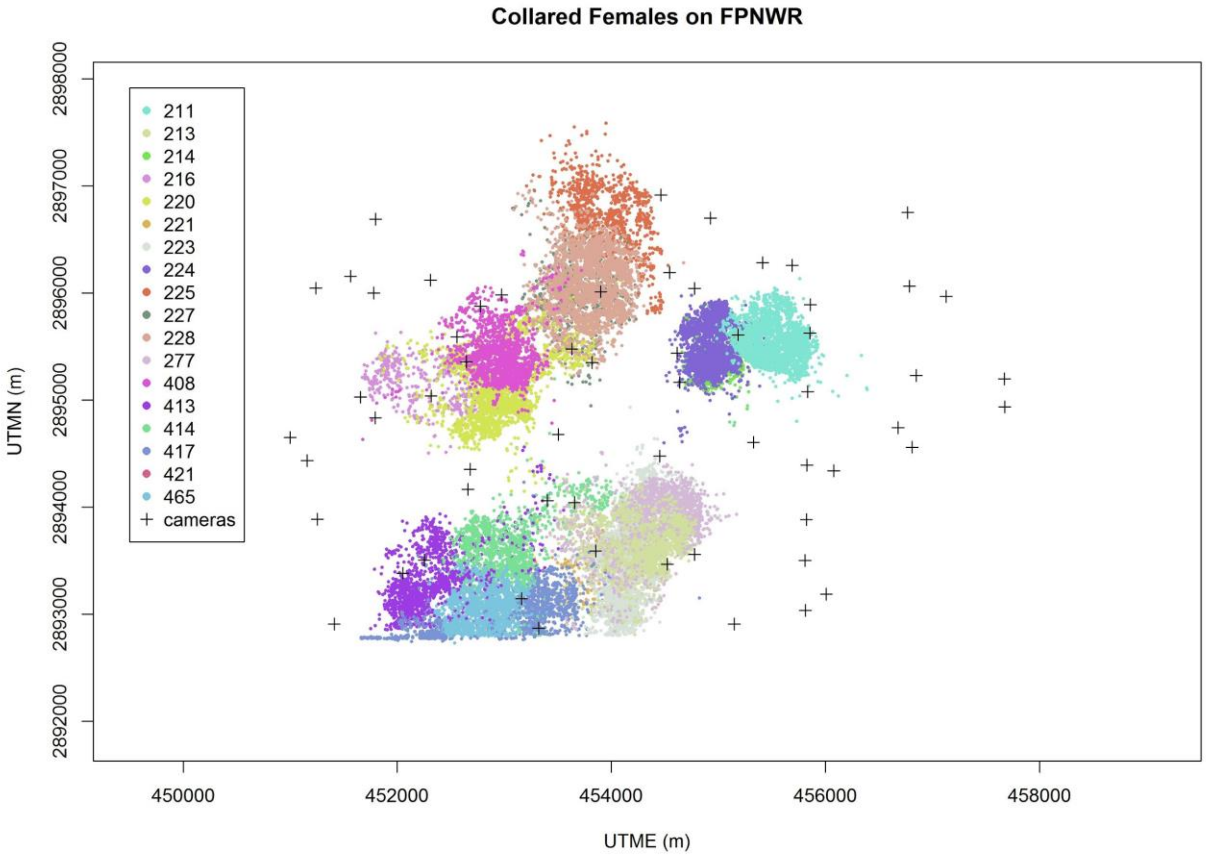


Figure 44. Telemetry locations of 18 GPS-collared female white-tailed deer on Florida Panther National Wildlife Refuge (FPNWR) during 1 January 2015 – 31 December 2017, used to estimate the detection probability parameters for the spatial capture-recapture model.

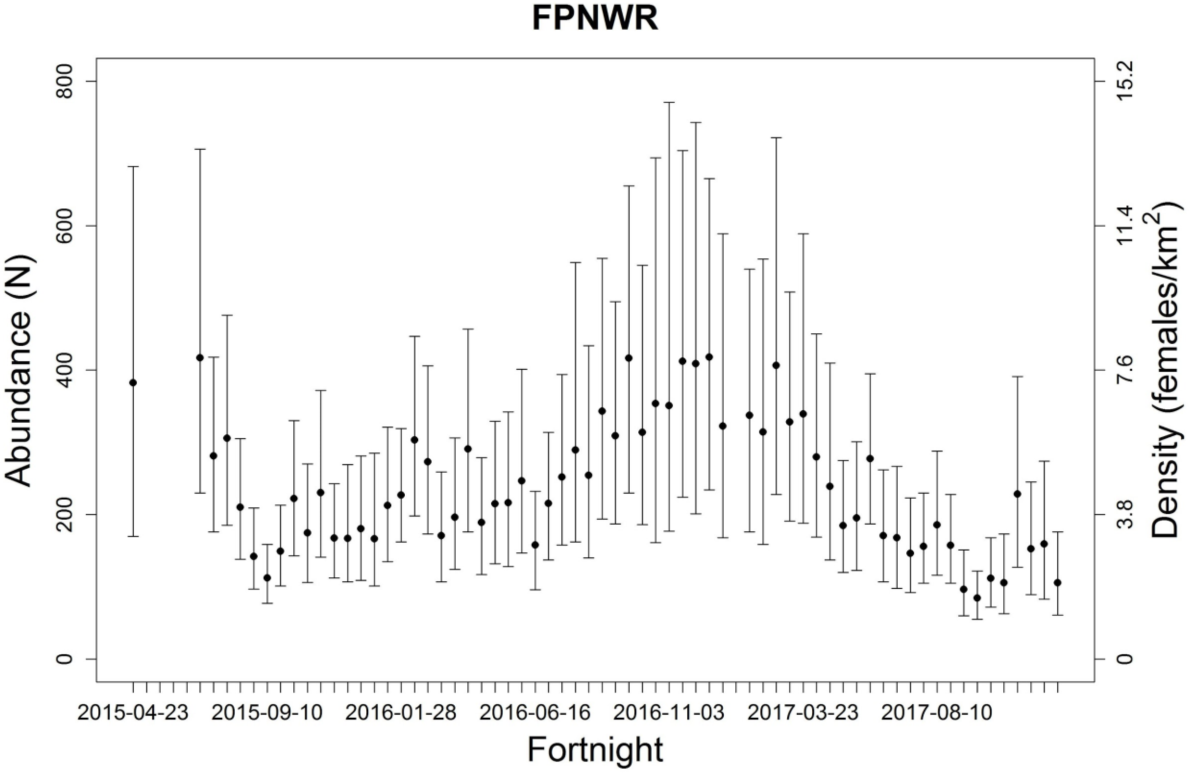


Figure 45. Biweekly estimates and 95% credible intervals of abundance and density for female white-tailed deer on Florida Panther National Wildlife Refuge (FPNWR), during 1 January 2015 – 31 December 2017. White-tailed deer density estimates for five fortnights were excluded due to models failing to converge.

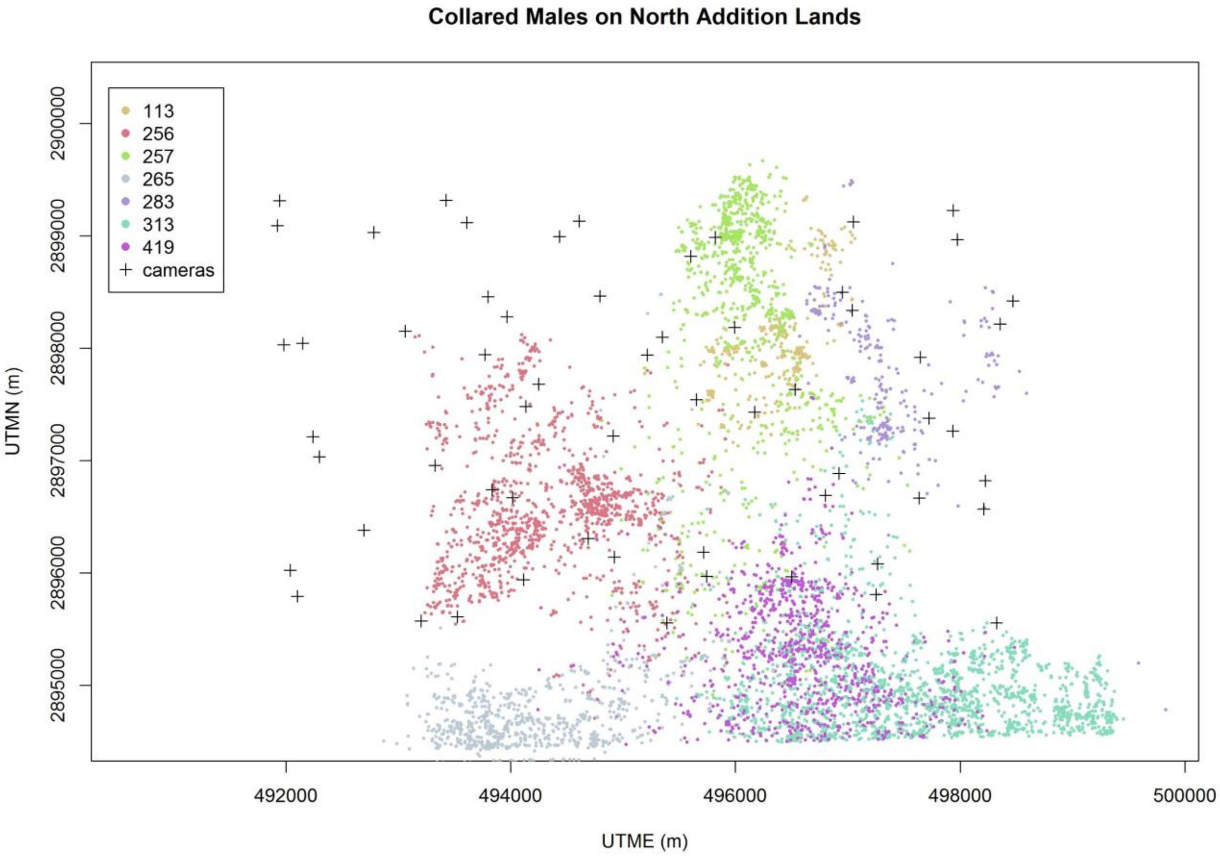


Figure 46. Telemetry locations of 7 GPS-collared male white-tailed deer on North Addition Lands, Big Cypress National Preserve, during 1 January 2015 – 31 December 2017 used to estimate the detection probability parameters for the spatial capture-recapture model.

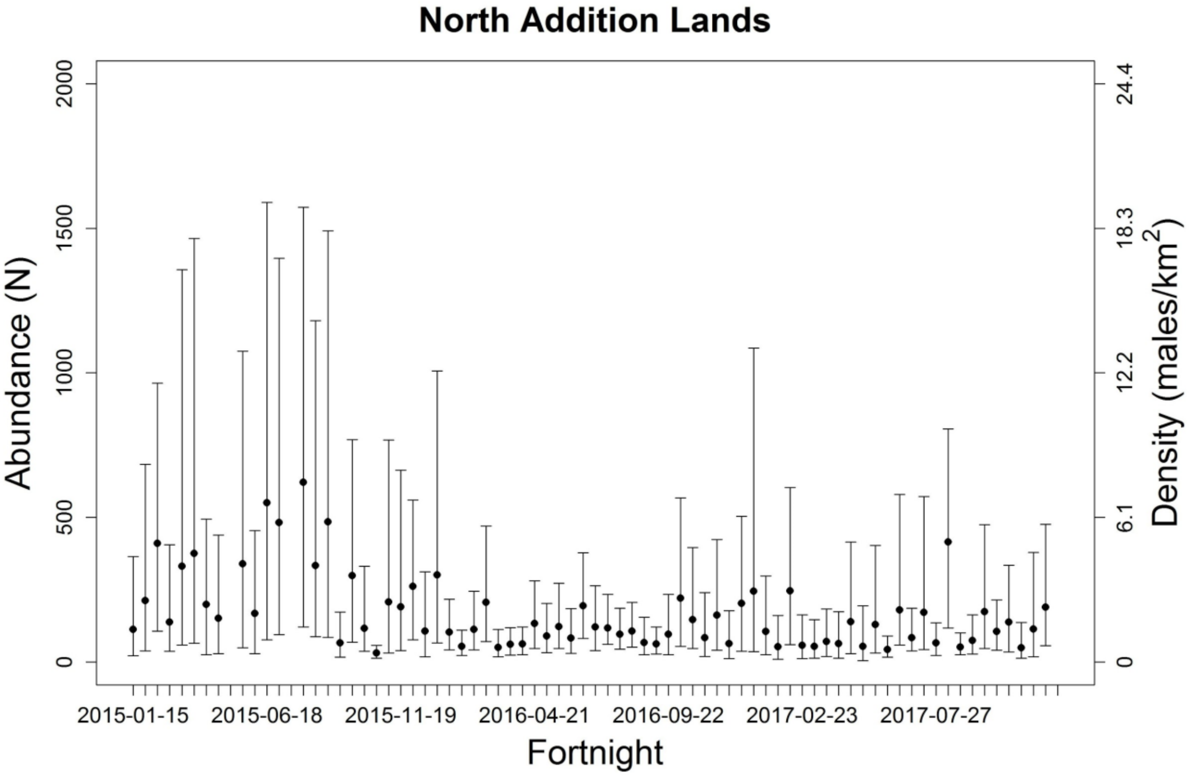


Figure 47. Biweekly estimates and 95% credible intervals of abundance and density for male white-tailed deer on North Addition Lands, Big Cypress National Preserve during 1 January 2015 – 31 December 2017. White-tailed deer density estimates for two fortnights were excluded due to models failing to converge.

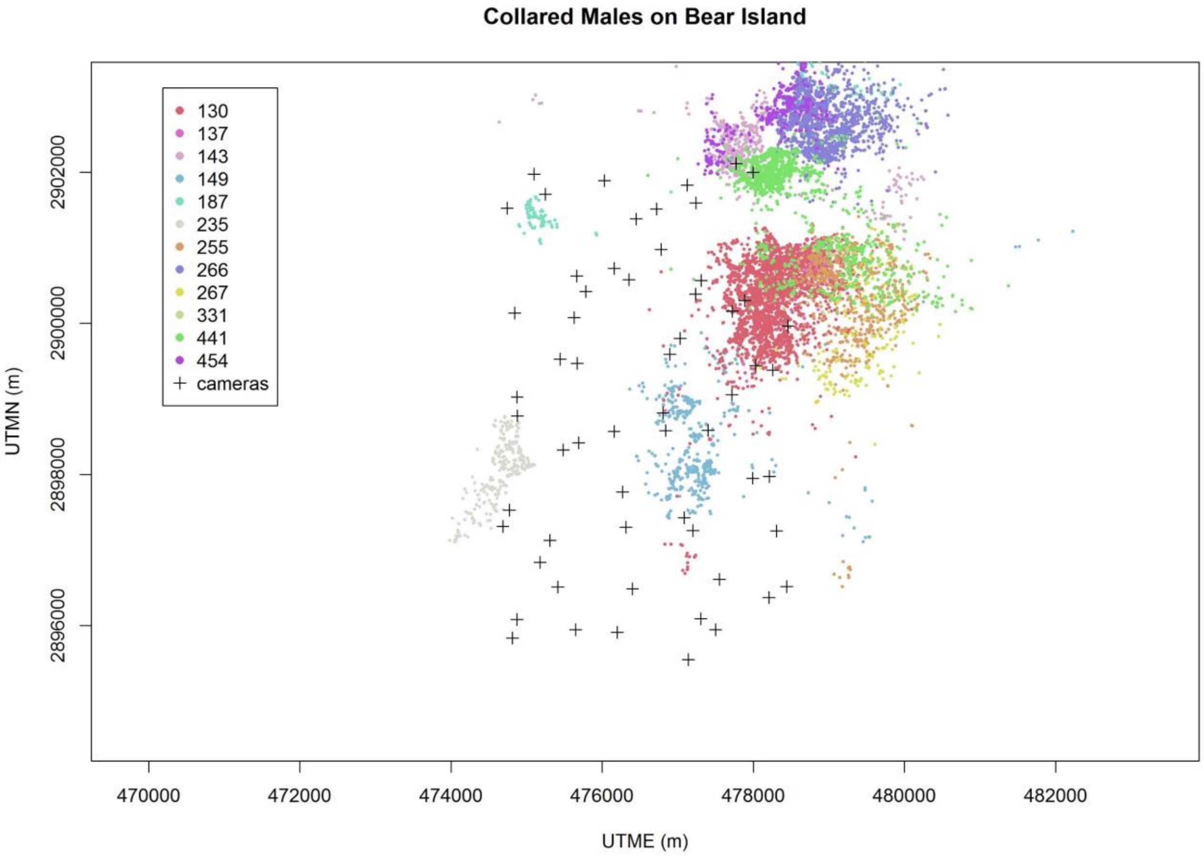


Figure 48. Telemetry locations of 13 GPS-collared male white-tailed deer on Bear Island, Big Cypress National Preserve during 1 January 2015 – 31 December 2017, used to estimate the detection probability parameters for the spatial capture-recapture model.

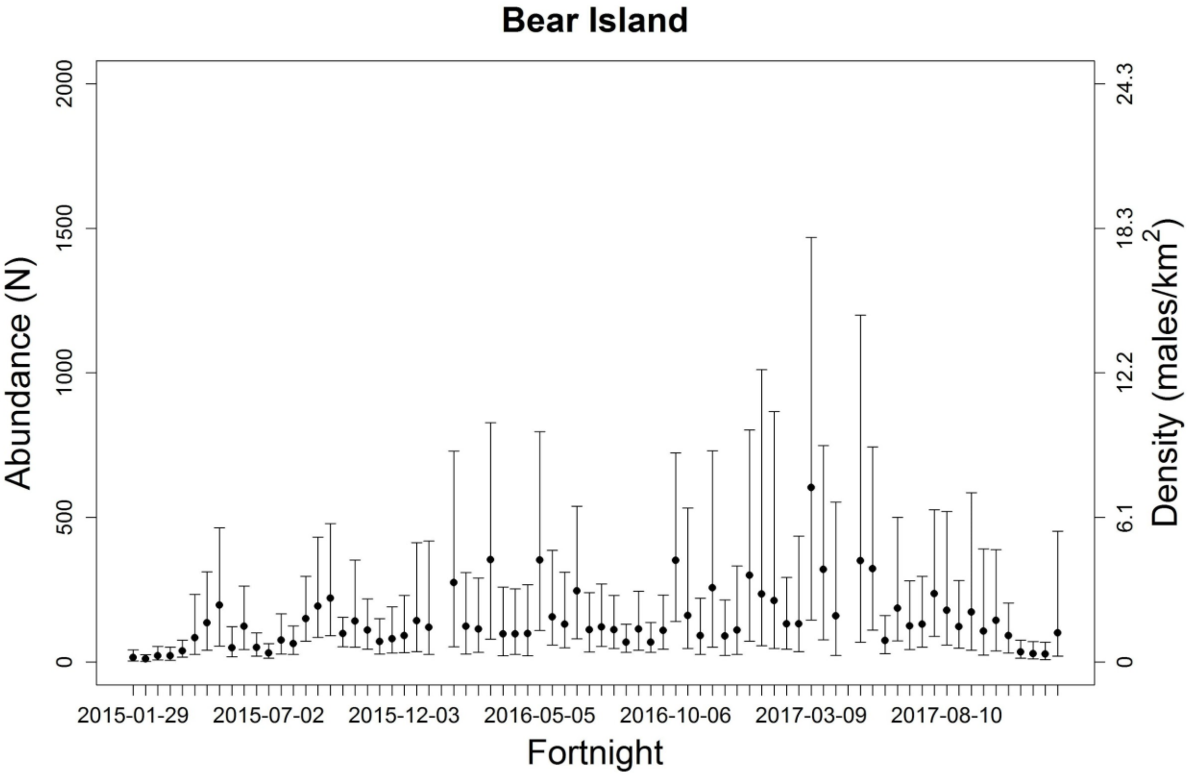


Figure 49. Biweekly estimates and 95% credible intervals of abundance and density for male white-tailed deer on Bear Island, Big Cypress National Preserve, during 1 January 2015 – 31 December 2017. Density estimates for two fortnights were excluded due to models failing to converge.

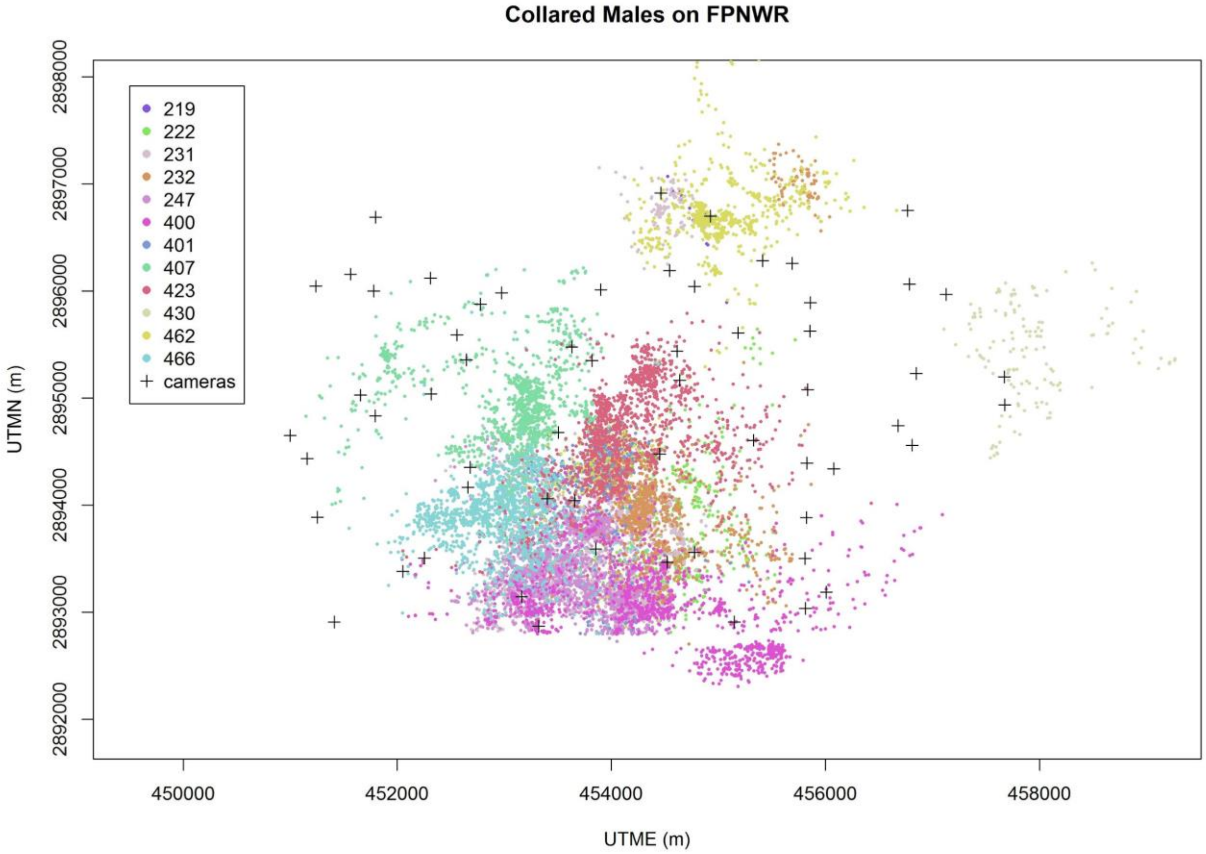


Figure 50. Telemetry locations of 12 GPS-collared male white-tailed deer on Florida Panther National Wildlife Refuge (FPNWR), during 1 January 2015 – 31 December 2017, used to estimate the detection probability parameters for the spatial capture-recapture model.

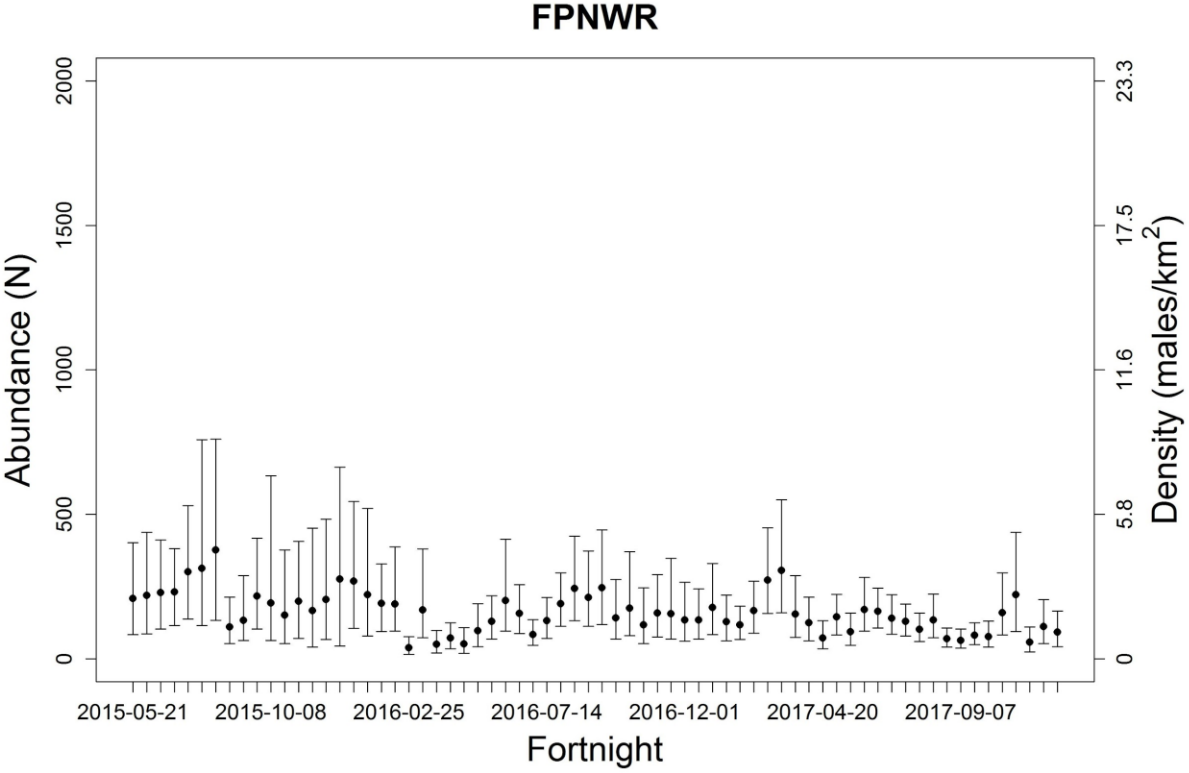


Figure 51. Biweekly estimates and 95% credible intervals of abundance and density for male white-tailed deer on Florida Panther National Wildlife Refuge (FPNWR), during 1 January 2015 – 31 December 2017.

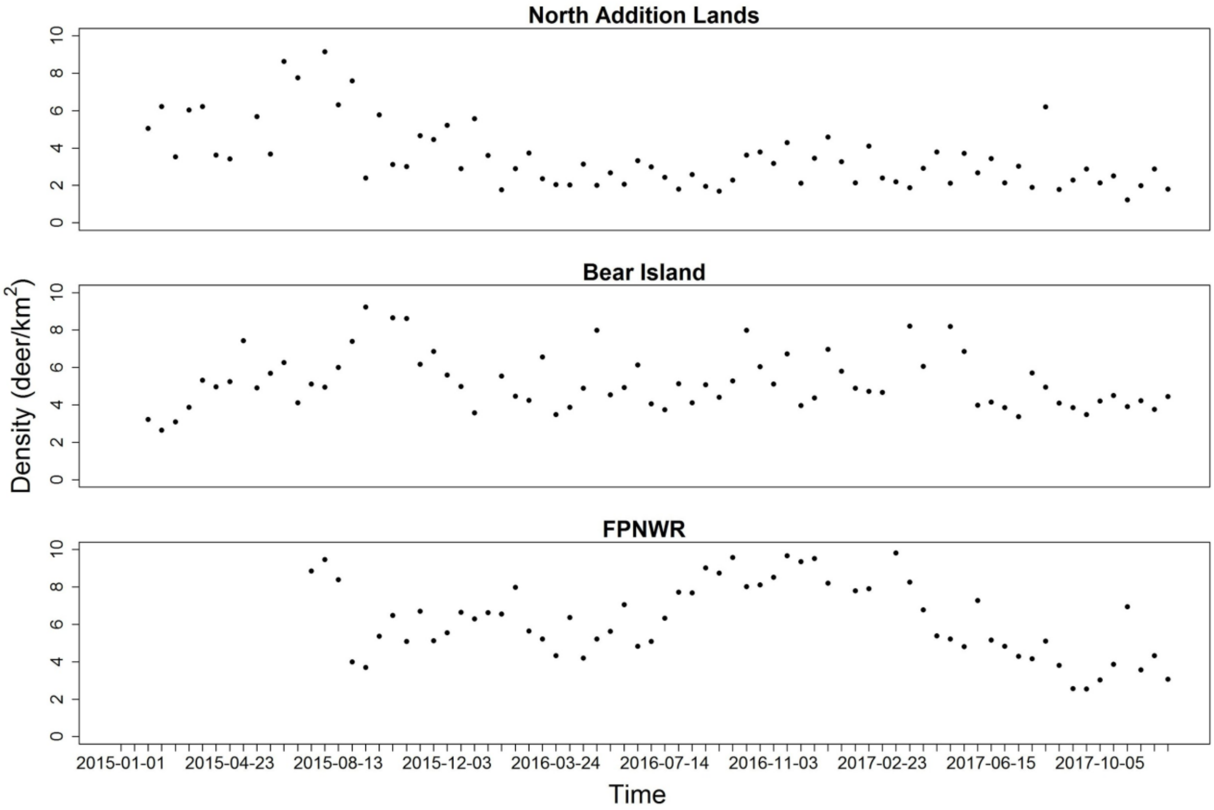


Figure 52. Biweekly adult white-tailed deer density estimates for North Addition Lands and Bear Island and Florida Panther National Wildlife Refuge (FPNWR). Adult white-tailed deer density estimates were calculated for fortnight periods in which both male and female densities were reliably estimated between 1 January 2015 – 31 December 2017. Adult white-tailed deer density represents combined mean density estimates of males and females.

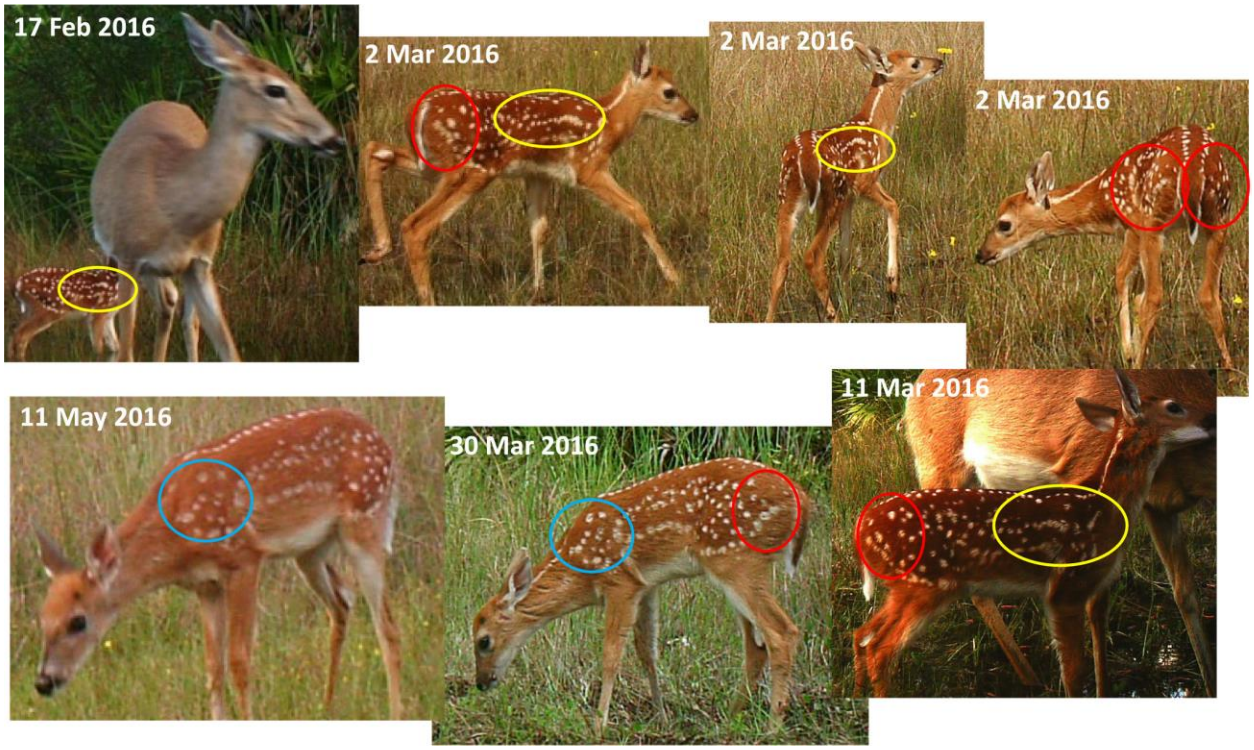


Figure 53. One of the 269 fawns detected during the South Florida Deer Study during the 2015 and 2016 fawning seasons. Fawns were uniquely identifiable because of distinctive spot patterns on the left (blue ovals) and right (yellow ovals) sides. We used a single camera design at each survey location and were able to use photos either of the hindquarters (red ovals) taken from behind the fawn or the shoulders taken from in front of the fawn to link the spot patterns on each side to the same individual ID.

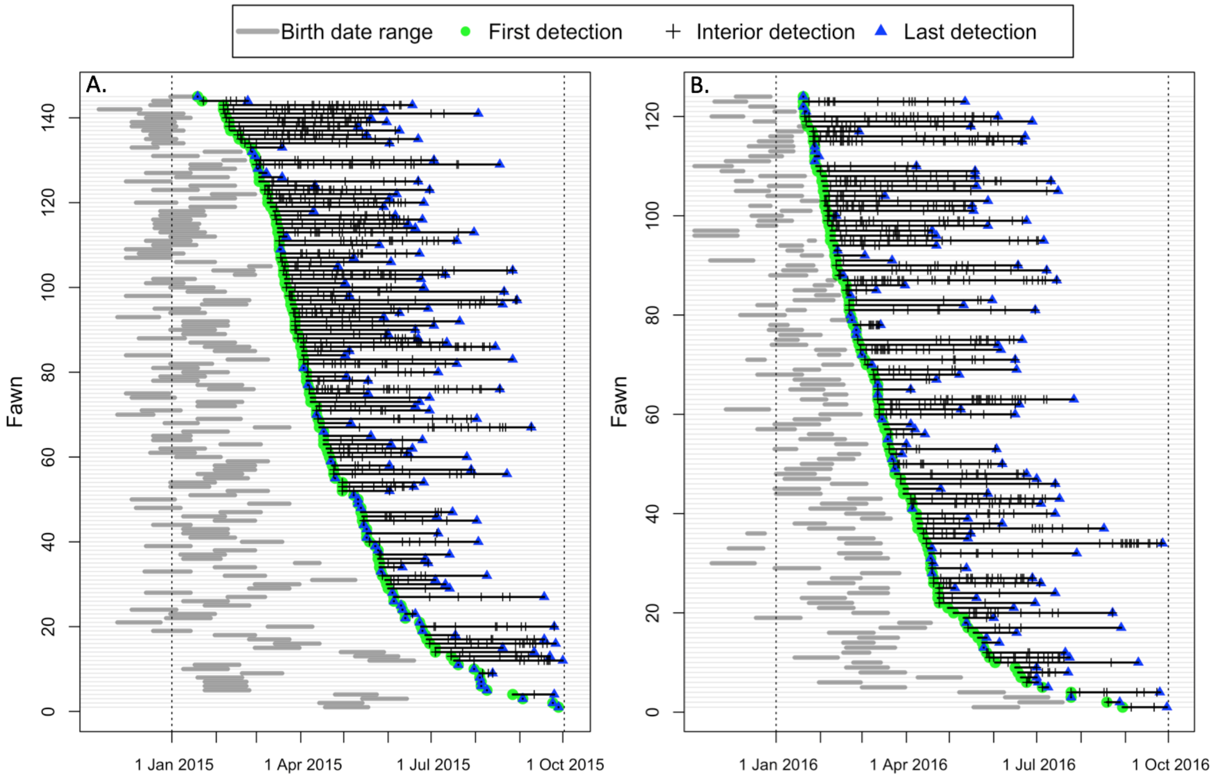


Figure 54. The birth date ranges (grey bars), first detection events (green circles), subsequent detection events (crosses), and last detection events (blue triangles) for the 269 uniquely identified fawns across all three study sites in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge over the (A) 2015 and (B) 2016 fawning seasons. Vertical dashed lines at 1 January and 1 October of each year indicate the time period in which the trail camera data were collected.

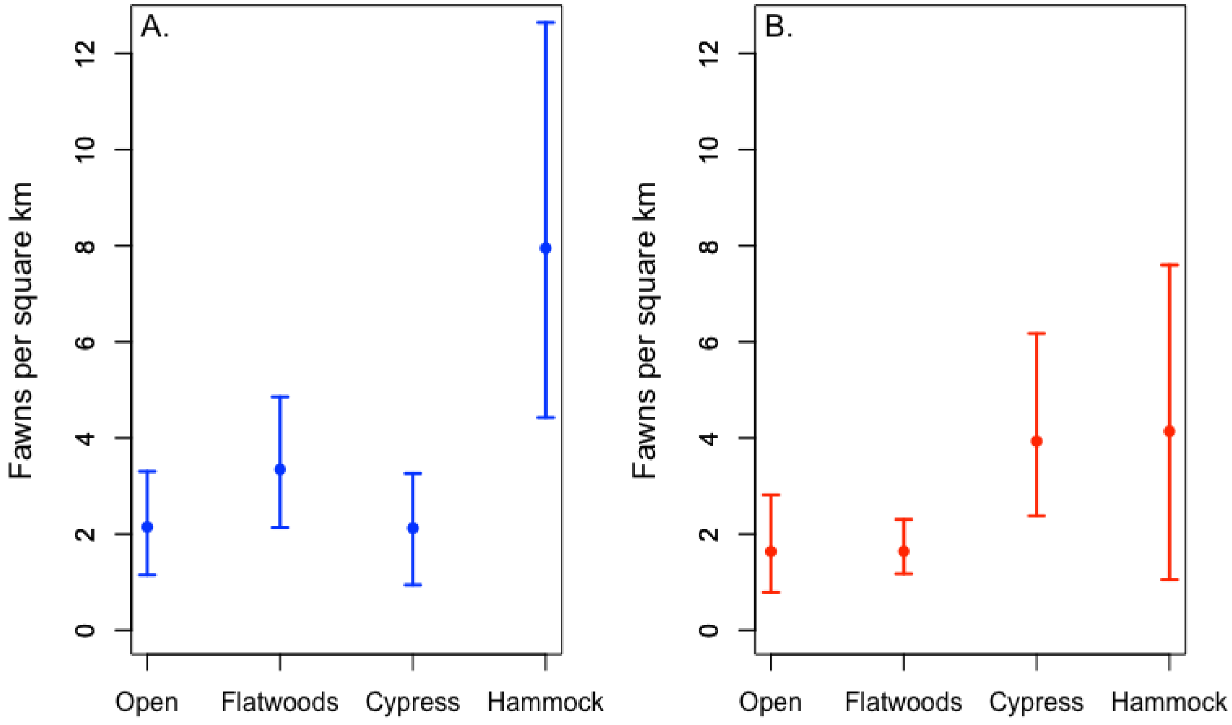


Figure 55. Estimated density and 95% credible intervals of fawn birth locations in fawns per square kilometer for each of the four vegetation classes across all three camera sites during the (A) 2015 fawning season, and (B) 2016 fawning season in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge.

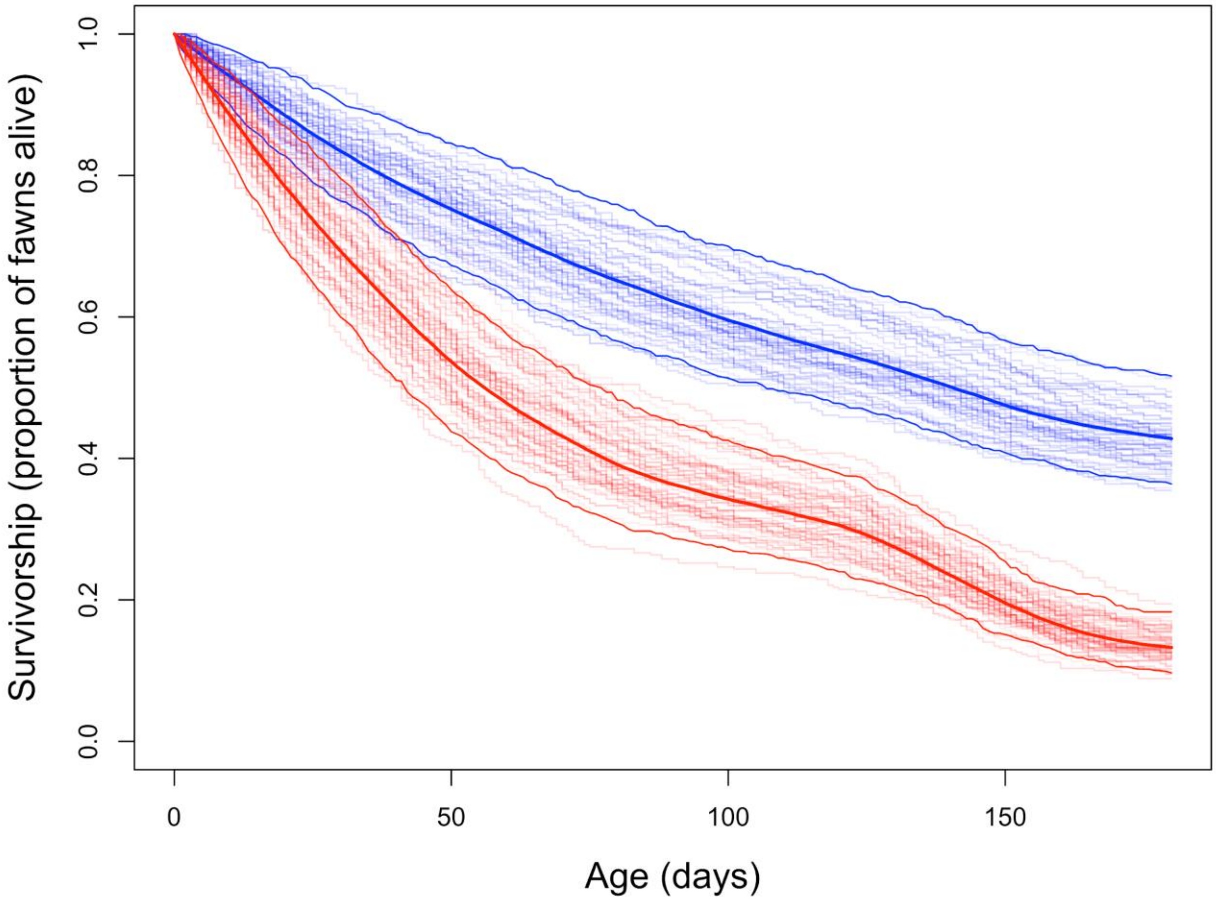


Figure 56. Fawn survivorship curves from the analysis of encounter data of 269 uniquely identified individuals from North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge. The blue curve indicates survivorship for the 2015 fawning season and the red curve indicates survivorship for the 2016 fawning season. The thickest lines are the posterior means. Thinner lines are 95% credible intervals. Faded lines are posterior samples.

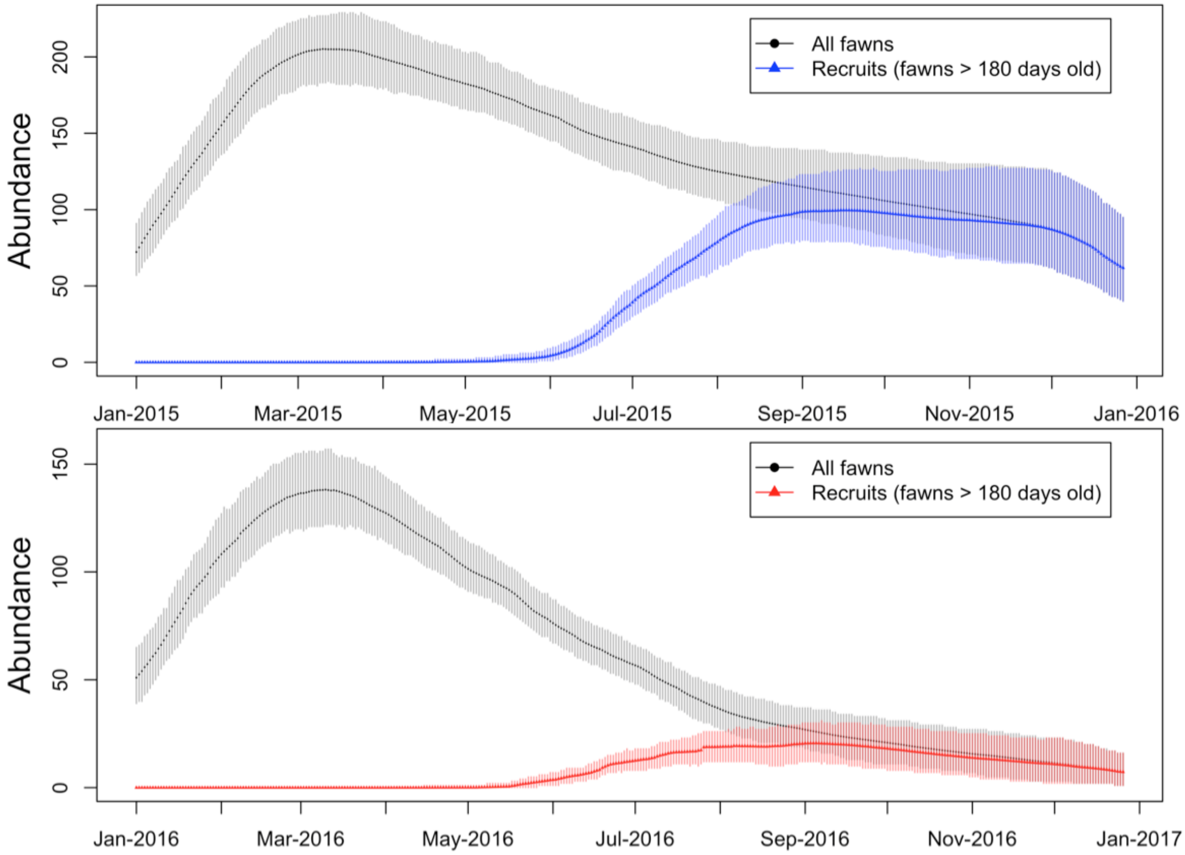


Figure 57. The number of fawns and recruits (fawns ≥180 days) alive throughout all three camera sites during the 2015 (top) and 2016 (bottom) fawning seasons in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge. Vertical lines along the curve indicate the 95% credible intervals for each estimate of fawn and recruit abundance.

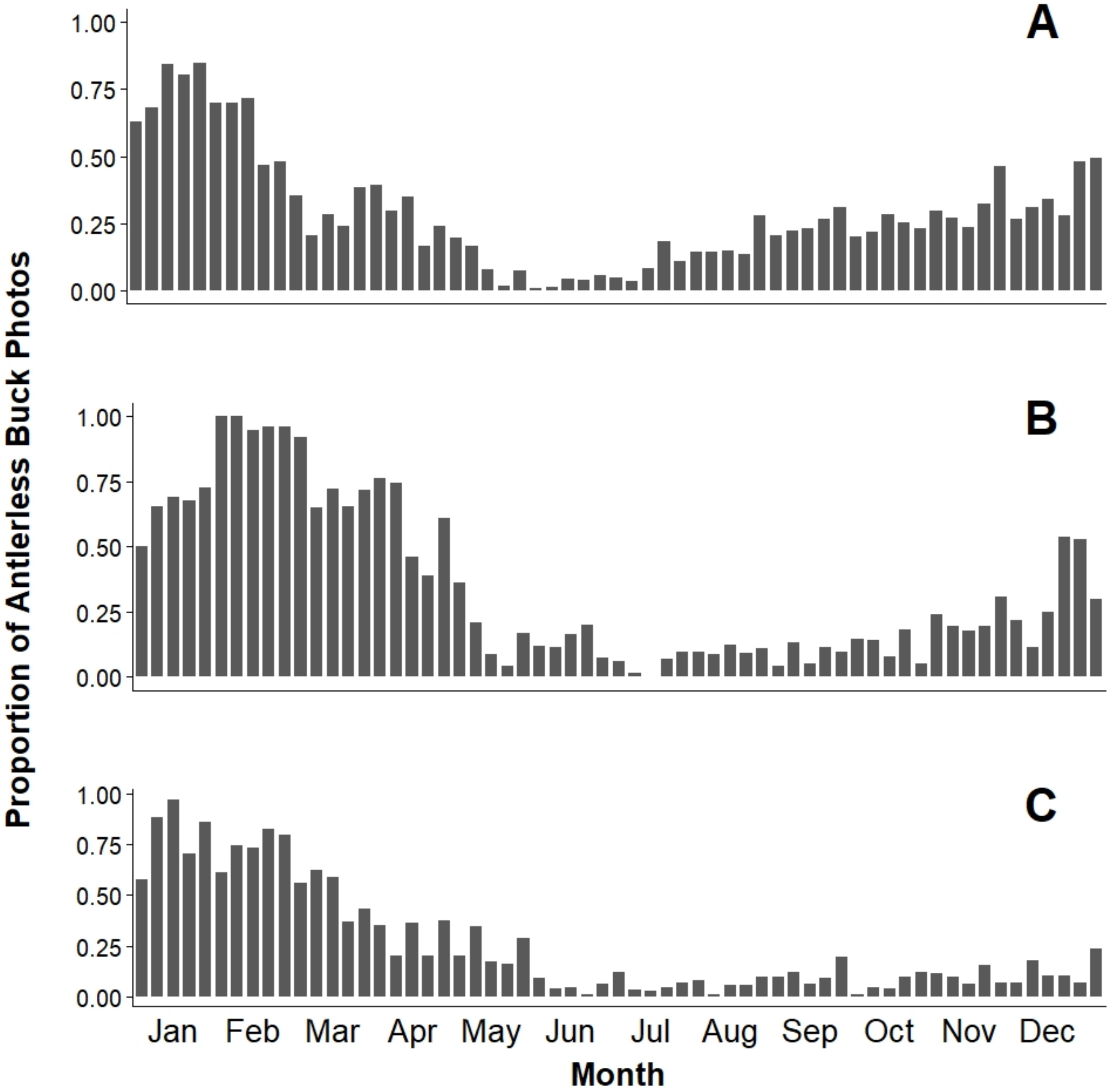


Figure 58. Proportion of daily antlerless male photos to daily total male photos across the year in A) Florida Panther National Wildlife Refuge, B) Bear Island, and C) North Addition Lands. Data were collected January 2015 - December 2018 in South Florida.

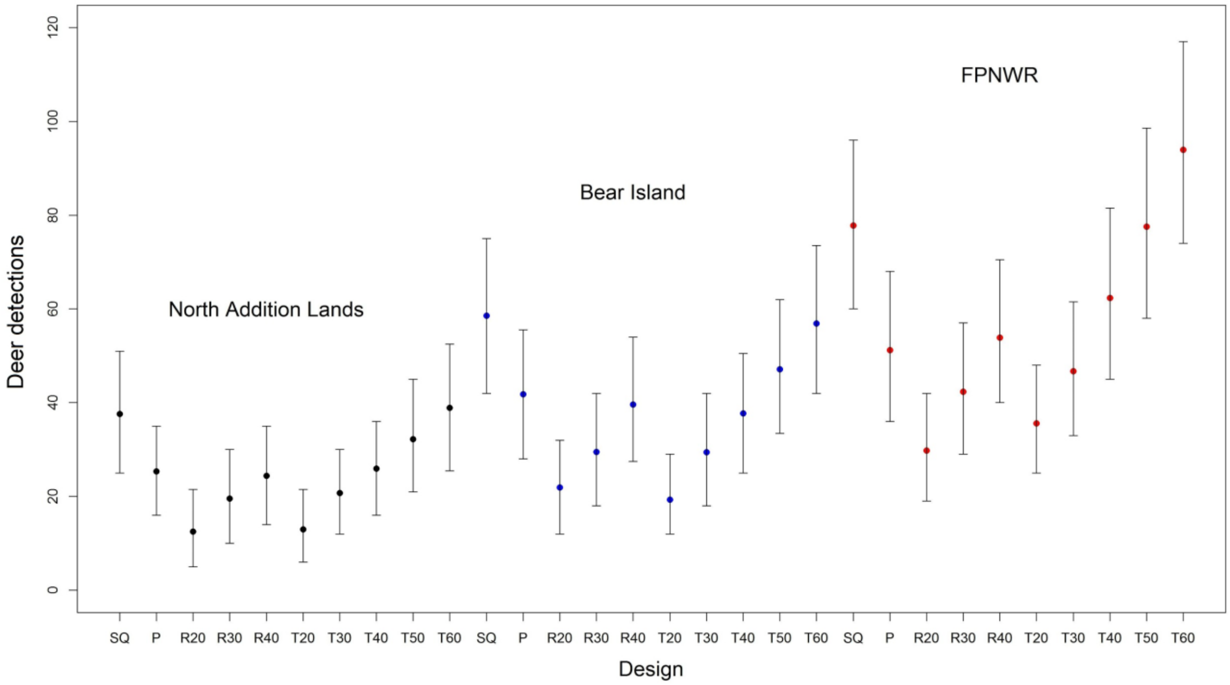


Figure 59. Total number of white-tailed deer detections averaged across 500 simulated datasets for each camera design collected in North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge. A detection event is defined as the detection of at least one deer at a camera during a 24 h sampling occasion. The camera design options are described in the Optimal Monitoring Design section.

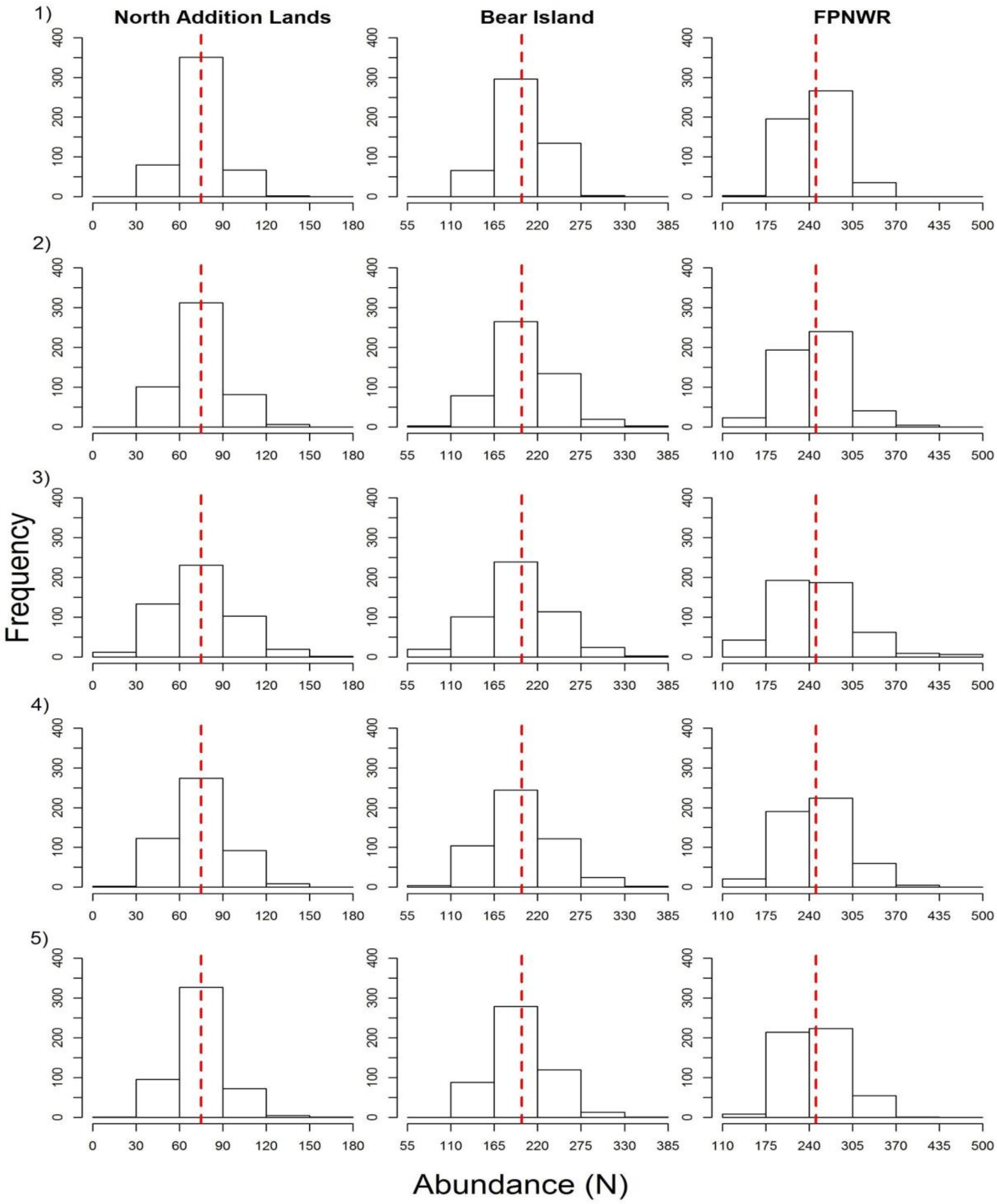


Figure 60. Abundance estimates from 500 simulated datasets for camera designs for North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge: 1) status quo, 2) 40 paired on- and off-trail cameras, 3) 20 random cameras, 4) 30 random cameras, and 5) 40 random cameras. The dashed line indicates the value of abundance used to simulate the data.

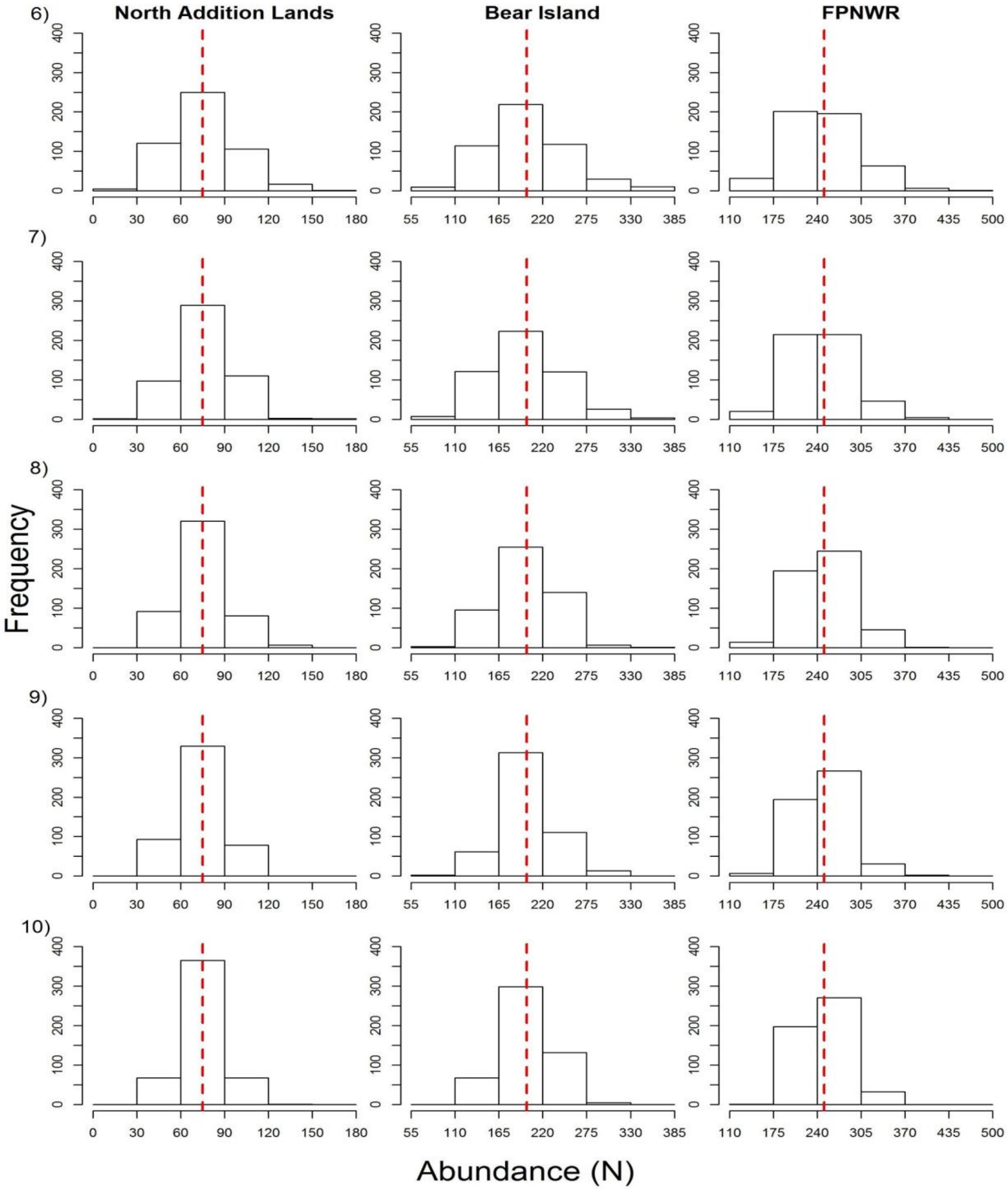


Figure 61. Abundance estimates from 500 simulated datasets for camera designs for North Addition Lands, Bear Island, and Florida Panther National Wildlife Refuge: 6) 20 on-trail cameras, 7) 30 on-trail cameras, 8) 40 on-trail cameras, 9) 50 on-trail cameras, 10) 60 on-trail cameras. The dashed line indicates the value of abundance used to simulate the data.

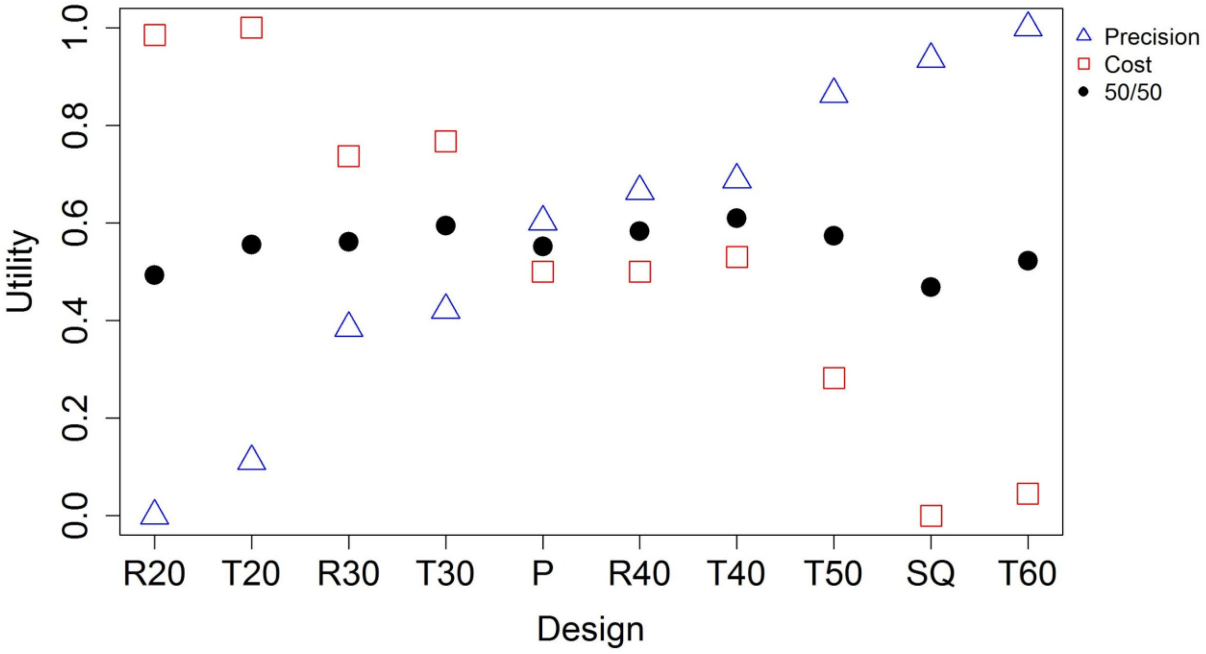


Figure 62. Camera design performance for achieving the objectives of maximizing precision, minimizing bias, and minimizing costs. Three objective weightings were assessed to evaluate tradeoffs. The “precision” uses 100% weighting for minimizing root mean squared error (RMSE), while cost uses 100% weighting for minimizing costs. The 50/50 objective gives equal weight to minimizing RMSE and cost. A utility of 1 indicates high performance, while a utility of 0 indicates poor performance relative to the camera designs implemented. The camera design options are described in the Optimal Monitoring Design section. Using the 50/50 objective, the optimal design was T40, which involves 40 on-trail cameras and no off-trail cameras.

APPENDICES

Appendix A. Glossary

Abbreviations

AIC – Akaike's Information Criterion

AL – North Addition Lands Management Unit in Big Cypress National Preserve

BCNP – Big Cypress National Preserve

BI – Bear Island Management Unit in Big Cypress National Preserve

CI – Confidence interval when using frequentist statistics. A statistic that can be used to describe the range of values defined by a specific probability within which an unobserved parameter lies.

CrI – Credible interval when using Bayesian statistics. A statistic that can be used to describe the range of values defined by a specific probability within which an unobserved parameter lies.

CERP – Comprehensive Everglades Restoration Plan

dBBMM – Dynamic Brownian Bridge Movement Model

EDEN – Everglades Depth Estimation Network

ENP - Everglades National Park

FNAI – Florida Natural Areas Inventory

FPNWR – Florida Panther National Wildlife Refuge

FWC – Florida Fish and Wildlife Commission

GLMM – Generalized linear mixed model

iSSA – Integrated step-selection analysis

MCMC – Markov chain Monte Carlo

ORV – Off-road vehicle

RMSE – Root mean squared error. A statistic that can be used to describe bias and precision of an estimator. Low RMSE indicates low bias and high precision.

SCR – Spatial capture-recapture

SSF – Step-selection function

SWI – Surface Water Index

UD – Utilization distribution. A 3-dimensional probability distribution which provide the probability density that an animal is found at a given point in space. In this report, UDs are the 2-dimensional area included within the 95% isopleth.

WAIC – Watanabe-Akaike information criterion

Spatial Capture-Recapture

Spatial capture-recapture (SCR) is an extension of traditional capture-mark-recapture (CMR) methods. Unlike traditional CMR methods, which produce estimates of abundance but not density, SCR models yield spatially explicit estimates of abundance and density. In addition, SCR models account for spatial variation in detection probability that arises from the distance between animals and traps.

Table A1. Common parameters and notation used in SCR models.

Parameter	Short Description	Long Description
N	Abundance	The number of individuals in the population within the spatial region $\mathcal{S}$
$\mathcal{S}$	Spatial region	Polygon within which the Nindividuals occur. This was defined by a 1 km buffer placed around each of the three camera arrays.
D	Density	Abundance in the spatial region $\mathcal{S}$ divided by the area of $\mathcal{S}$
λ_0	Baseline encounter rate	The expected number of detections of an individual at a trap during a single occasion when the distance between its home range center and the trap is zero. The encounter rate, and hence detection probability, decrease with distance.
σ	Spatial scale parameter	This parameter describes how encounter rate changes with distance between a home range center and a trap. This parameter is often proportional to home range size because animals with large home ranges can be detected farther from their home range center than animals with small home ranges.
λ_{ijk}	Encounter rate	The expected number of detections of an individual at a trap during a single occasion. This is a function of λ_0, σ, s_i, and $x_{\boldsymbol{j}}$.
p_{ijk}	Detection probability	The probability of detecting an individual at a trap during a single occasion.
$x_{\boldsymbol{j}}$	Trap coordinates	The spatial coordinates of trap j
$\boldsymbol{s}_i$	Home range center coordinates	The spatial coordinates of the home range center for individual i
y_{ijk}	Encounter history data	The number of detections of individual i at trap j on occasion k
n_{jk}	Occurrence data	A binary variable indicating if at least one individual was detected at trap $j$on occasion k

Appendix B. Noteworthy Photographs

White-tailed Deer Capture Photos





White-tailed Deer Trail Camera Photos


HCO ScoutGuard
02.16.2016 15:23:26


HCO ScoutGuard
03.02.2016 10:21:02


HCO ScoutGuard
08.29.2017 11:34:53


HCO ScoutGuard
07.02.2016 13:40:35


HCO ScoutGuard
08.05.2017 14:54:01


HCO ScoutGuard
08.08.2017 13:21:47

Predator Community Trail Camera Photos


HCO ScoutGuard
06.08.2016 11:18:44


HCO ScoutGuard
06.24.2016 15:31:48


HCO ScoutGuard
12.10.2015 08:34:43


HCO ScoutGuard
05.04.2015 14:34:41


HCO ScoutGuard
09.26.2017 18:16:43


HCO ScoutGuard
03.16.2015 10:51:11

Environmental Trail Camera Photos

Appendix C. Capture data summary by capture method.

Table C1. Capture data summary by capture method, sex, location and year (*n* = 294) for deer captured January 2015 to December 2018 in Big Cypress National Preserve and Florida Panther National Wildlife Refuge.

	2015				2016				2017				Grand Total
	Net-gunning	Darting	Rocket netting	Total	Net-gunning	Darting	Rocket netting	Total	Net-gunning	Darting	Rocket netting	Total	
BCNP													
Female	67	-	-	67	55	-	-	55	24	-	-	24	146
Male	38	-	-	38	27	-	-	27	12	-	-	12	77
Total	105	-	-	105	82	-	-	82	36	-	-	36	223
FPNWR													
Female	1	3	14	18	4	-	6	10	-	5	9	14	42
Male	-	1	5	6	-	4	3	7	1	13	1	15	28
Fawn	-	-	1	1	-	-	-	-	-	-	-	-	1
Total	1	4	20	25	4	4	9	17	1	18	10	29	71
Grand Total	106	4	20	130	86	4	9	99	37	18	10	65	294

Appendix D. Land Cover Classification Data and Reclassification Schematic

Table D1. Land cover classification data and reclassification schematic. The data classification scheme presented here is derived from the Florida Natural Area Inventory (FNAI) cooperative land cover, version 3.2 site-level land cover data. Land cover classifications are grouped into a land cover reclassification types used in this study. These cover types were classified using unsupervised classification following the FNAI Classification System.

Original Site Land Cover Name	Spectral Class	Land Cover Reclassification Name
Rural Open Forested	18311	Agriculture
Orchards/Groves	18332	Agriculture
Vineyard and Nurseries	18334	Agriculture
Other Agriculture	18335	Agriculture
Row Crops	183311	Agriculture
Field Crops	183312	Agriculture
Improved Pasture	183313	Agriculture
Citrus	183321	Agriculture
Ornamentals	183343	Agriculture
Fallow Cropland	1833151	Agriculture
Mesic Hammock	1120	Hardwood Hammock
Cabbage Palm	1125	Hardwood Hammock
Rockland Hammock	1130	Hardwood Hammock
Mixed Hardwood-Coniferous	1400	Hardwood Hammock
Successional Hardwood Forest	1410	Hardwood Hammock
Hydric Hammock	2232	Hardwood Hammock
Prairie Hydric Hammock	22322	Hardwood Hammock
Cabbage Palm Hammock	22323	Hardwood Hammock
Oak-Cabbage Palm Forests	183111	Hardwood Hammock
Unimproved/Woodland Pasture	183314	Hardwood Hammock
Cypress/Tupelo(incl Cy/Tu mixed)	2210	Hardwood Swamp
Cypress	2211	Hardwood Swamp
Isolated Freshwater Swamp	2213	Hardwood Swamp
Strand Swamp	2214	Hardwood Swamp
Other Coniferous Wetlands	2220	Hardwood Swamp
Other Hardwood Wetlands	2230	Hardwood Swamp
Mixed Wetland Hardwoods	2233	Hardwood Swamp
Other Wetland Forested Mixed	2240	Hardwood Swamp
Cypress/Hardwood Swamps	2241	Hardwood Swamp
Cypress/Pine/Cabbage Palm	2242	Hardwood Swamp
Dome Swamp	22131	Hardwood Swamp
South Florida Bayhead	22312	Hardwood Swamp

Marshes	2120	Marshes
Isolated Freshwater Marsh	2121	Marshes
Glades Marsh	2125	Marshes
Sawgrass	2131	Marshes
Floating/Emergent Aquatic Vegetation	2140	Marshes
Cultural - Palustrine	2400	Marshes
Depression Marsh	21211	Marshes
Lacustrine	3000	Open Water
Natural Lakes and Ponds	3100	Open Water
Cultural - Lacustrine	3200	Open Water
Artificial/Farm Pond	3210	Open Water
Artificial Impoundment/Reservoir	3220	Open Water
Quarry Pond	3230	Open Water
Canal	4210	Open Water
Ditch/Artificial Intermittent Stream	4220	Open Water
Mesic Flatwoods	1311	Pine Forests
Scrubby Flatwoods	1312	Pine Forests
Wet Flatwoods	2221	Pine Forests
Hydric Pine Flatwoods	22211	Pine Forests
Hydric Pine Savanna	22212	Pine Forests
Cabbage Palm Flatwoods	222112	Pine Forests
Palmetto Prairie	1340	Prairie
Prairies and Bogs	2110	Prairie
Wet Prairie	2111	Prairie
Mixed Scrub-Shrub Wetland	2112	Prairie
Marl Prairie	2113	Prairie
Highway Rights of Way	1812	Roads
Transportation	1840	Roads
Shrub and Brushland	1500	Shrub
Rural Open	1831	Shrub
Vegetative Berm	1811	Urban
Low Intensity Urban	1821	Urban
Rural Structures	1832	Urban
Communication	1850	Urban
Utilities	1860	Urban
Extractive	1870	Urban
Sand & Gravel Pits	1872	Urban
Oil & Gas Fields	1874	Urban
Urban Open Land	18211	Urban
Residential, Low Density	18212	Urban
Commercial and Services	18223	Urban
Industrial	18224	Urban
Urban Open Forested	182111	Urban
Melaleuca	7200	NoData
Brazilian Pepper	7300	NoData
Exotic Wetland Hardwoods	7400	NoData

Table D2. Land cover classification data and reclassification schematic for assigning all vegetation types to types outlined in Table 2. The data classification scheme presented here is derived from the Florida Natural Area Inventory (FNAI) cooperative land cover, version 3.2 site-level land cover data. These cover types were classified using unsupervised classification following the FNAI Classification System.

Original Site Land Cover Name	Spectral Class	Land Cover Reclassification Name
Rural Open Forested	18311	Agriculture
Orchards/Groves	18332	Agriculture
Vineyard and Nurseries	18334	Agriculture
Other Agriculture	18335	Agriculture
Row Crops	183311	Agriculture
Field Crops	183312	Agriculture
Unimproved/Woodland Pasture	183314	Agriculture
Citrus	183321	Agriculture
Ornamentals	183343	Agriculture
Fallow Cropland	1833151	Agriculture
Cypress/Tupelo(incl. Cy/Tu mixed)	2210	Cypress
Cypress	2211	Cypress
Isolated Freshwater Swamp	2213	Cypress
Strand Swamp	2214	Cypress
Other Coniferous Wetlands	2220	Cypress
Other Hardwood Wetlands	2230	Cypress
Mixed Wetland Hardwoods	2233	Cypress
Other Wetland Forested Mixed	2240	Cypress
Cypress/Hardwood Swamps	2241	Cypress
Cypress/Pine/Cabbage Palm	2242	Cypress
Dome Swamp	22131	Cypress
South Florida Bayhead	22312	Cypress
Wet Flatwoods	2221	Hydric Pines
Hydric Hammock	2232	Hydric Pines
Hydric Pine Flatwoods	22211	Hydric Pines
Hydric Pine Savanna	22212	Hydric Pines
Prairie Hydric Hammock	22322	Hydric Pines
Cabbage Palm Hammock	22323	Hydric Pines
Cabbage Palm Flatwoods	222112	Hydric Pines
Prairies and Bogs	2110	Marl Prairie
Wet Prairie	2111	Marl Prairie
Mixed Scrub-Shrub Wetland	2112	Marl Prairie
Marl Prairie	2113	Marl Prairie
Marshes	2120	Marshes

Isolated Freshwater Marsh	2121	Marshes
Glades Marsh	2125	Marshes
Sawgrass	2131	Marshes
Floating/Emergent Aquatic Vegetation	2140	Marshes
Cultural - Palustrine	2400	Marshes
Depression Marsh	21211	Marshes
Cabbage Palm	1125	Mesic Hammock
Rockland Hammock	1130	Mesic Hammock
Rural Open	1831	Mesic Hammock
Mesic Hammock	1120	Mesic Pines
Mesic Flatwoods	1311	Mesic Pines
Scrubby Flatwoods	1312	Mesic Pines
Palmetto Prairie	1340	Mesic Pines
Mixed Hardwood-Coniferous	1400	Mesic Pines
Successional Hardwood Forest	1410	Mesic Pines
Rural Structures	1832	Mesic Pines
Lacustrine	3000	Open Water
Natural Lakes and Ponds	3100	Open Water
Cultural - Lacustrine	3200	Open Water
Artificial/Farm Pond	3210	Open Water
Artificial Impoundment/Reservoir	3220	Open Water
Quarry Pond	3230	Open Water
Canal	4210	Open Water
Highway Rights of Way	1812	Urban
Low Intensity Urban	1821	Urban
Transportation	1840	Urban
Communication	1850	Urban
Utilities	1860	Urban
Urban Open Land	18211	Urban
Residential, Low Density	18212	Urban
Commercial and Services	18223	Urban
Industrial	18224	Urban
Urban Open Forested	182111	Urban
Improved Pasture	183313	Case-by-case*
Oak - Cabbage Palm Forests	183111	Case-by-case*
Shrub and Brushland	1500	Case-by-case*
Vegetative Berm	1811	NA
Extractive	1870	NA
Sand & Gravel Pits	1872	NA
Oil & Gas Fields	1874	NA
Ditch/Artificial Intermittent Stream	4220	NA
Melaleuca	7200	NA
Brazilian Pepper	7300	NA
Exotic Wetland Hardwoods	7400	NA

* reclassification where surrounding land cover types and aerial imagery determined the land cover assigned to the relative spectral class

Appendix E. Home Ranges

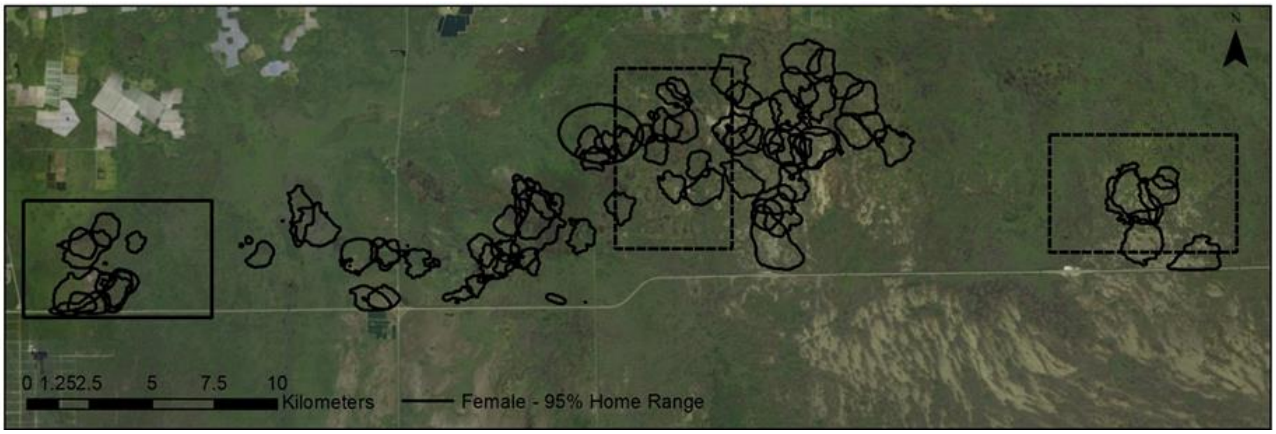


Figure E1. Home ranges for all male deer that lived at least one year and collected 80% of their scheduled GPS locations in the three study sites in the Big Cypress Basin. The Addition Lands (dashed) and Bear Island (dashed) were in Big Cypress National Preserve. The western site (solid boundary line) was in the Florida Panther National Wildlife Refuge.

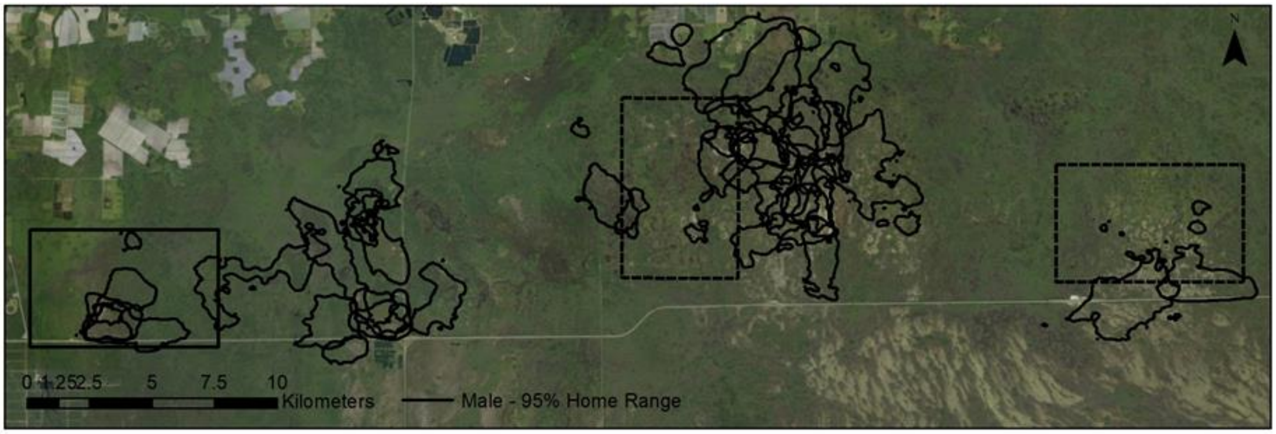


Figure E2. Home ranges for all female deer that lived at least one year and collected 80% of their scheduled GPS locations in the three study sites in the Big Cypress Basin. The Addition Lands (dashed) and Bear Island (dashed) were in Big Cypress National Preserve. The western site (solid boundary line) was in the Florida Panther National Wildlife Refuge.

Figure E3. Home ranges for all female (a) and male (b) deer that lived at least one year and collected 80% of their scheduled GPS locations in deer in the Florida Panther National Wildlife Refuge during 1 January 2015 – 31 December 2017.

Figure E4. Home ranges for all female (a) and male (b) deer that lived at least one year and collected 80% of their scheduled GPS locations in deer in the North Addition Lands management unit of Big Cypress National Preserve during 1 January 2015 – 31 December 2017.

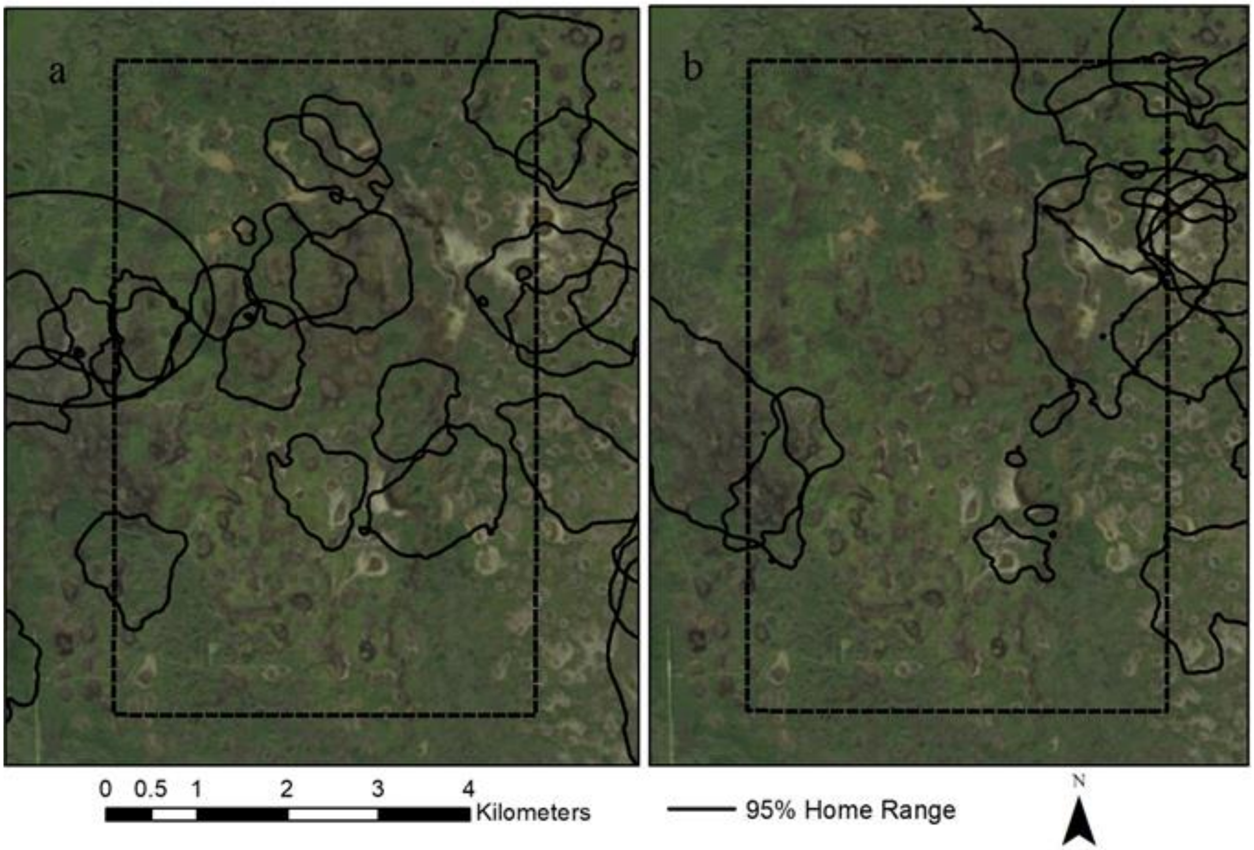


Figure E5. Home ranges for all female (a) and male (b) deer that lived at least one year and collected 80% of their scheduled GPS locations in deer in the Bear Island management unit of Big Cypress National Preserve during 1 January 2015 – 31 December 2017.

Appendix F. Spatial Time-series Maps of White-tailed Deer Detection Data

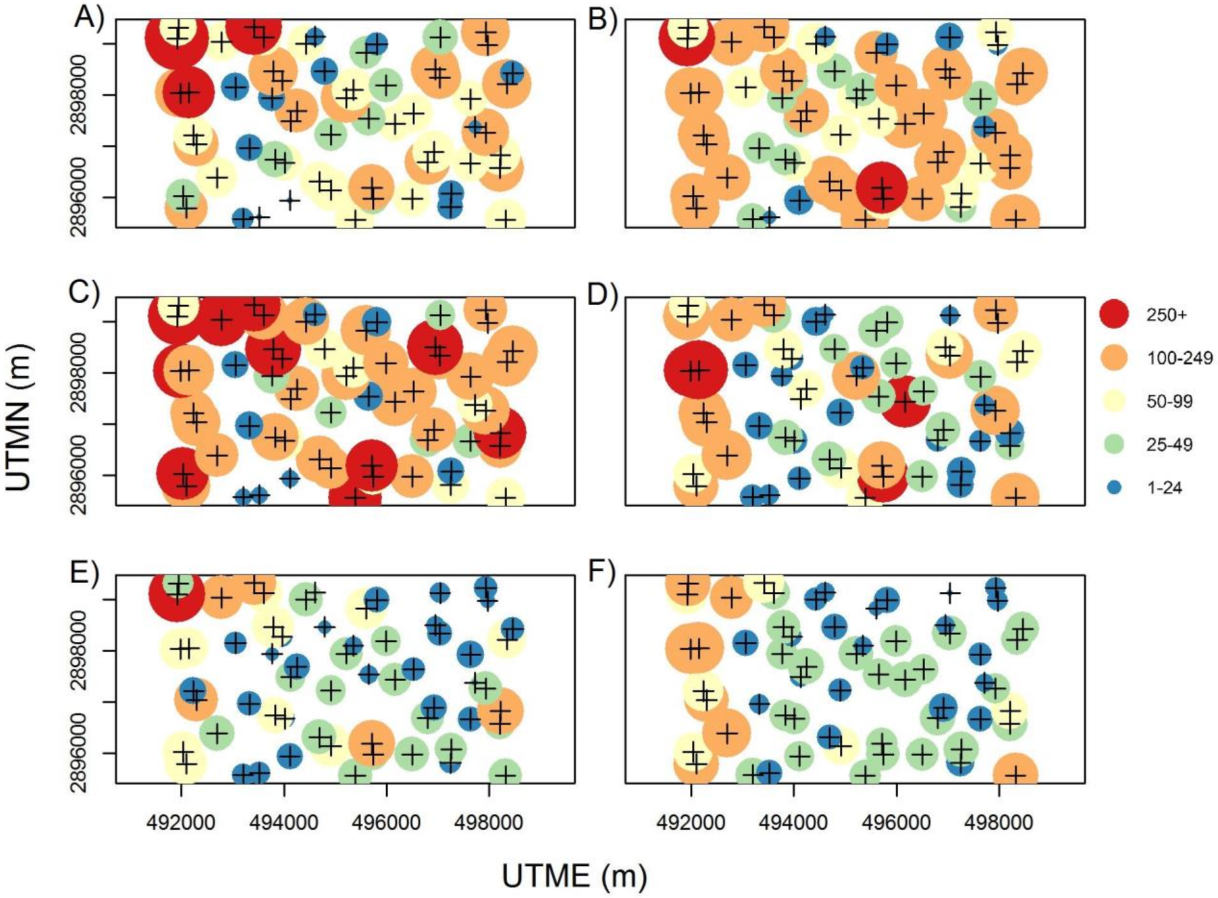


Figure F1. Maps of white-tailed deer camera detections for 6-month periods on North Addition Lands. The size of each point reflects the number of detections. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

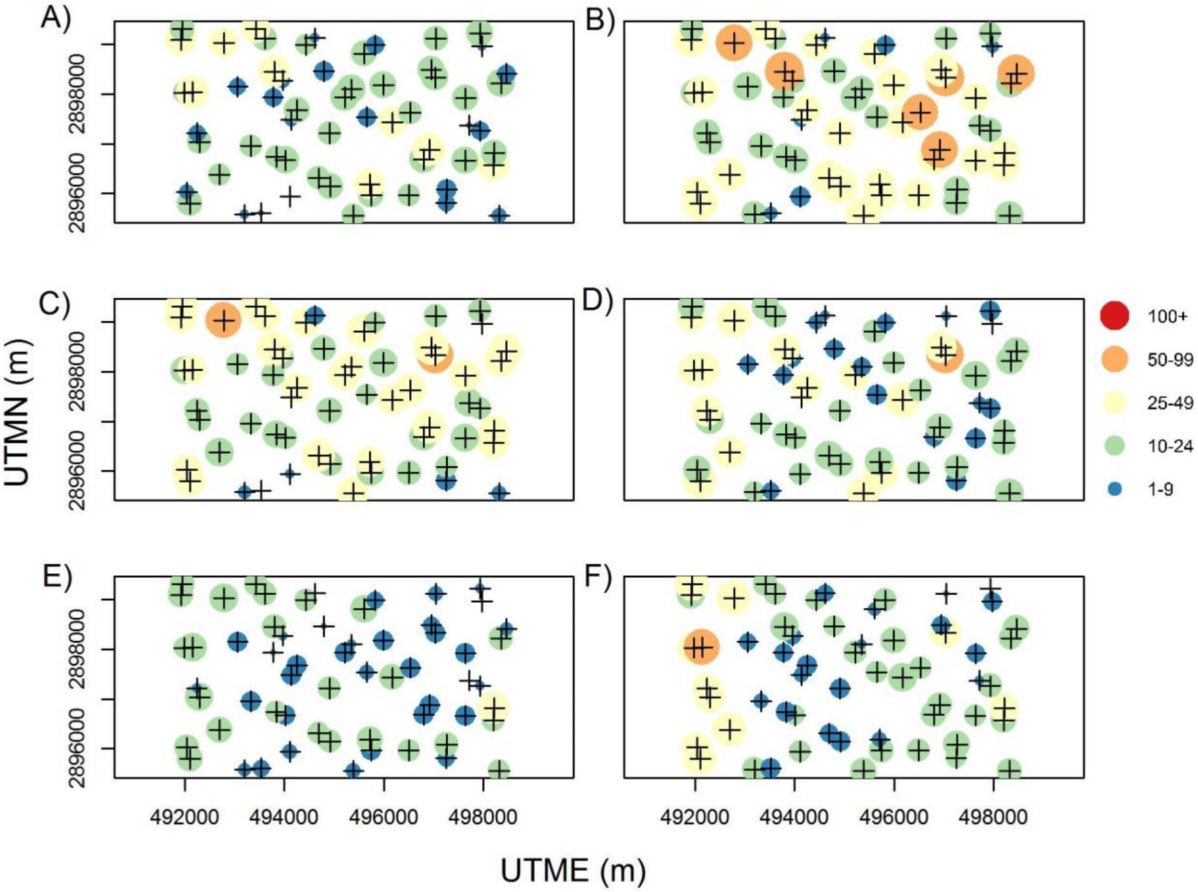


Figure F2. Maps of white-tailed deer daily occurrences for 6-month periods on North Addition Lands. An occurrence is defined as the detection of at least one deer at a camera during a 24 h sampling occasion. The size of each point reflects the number of daily occurrences. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

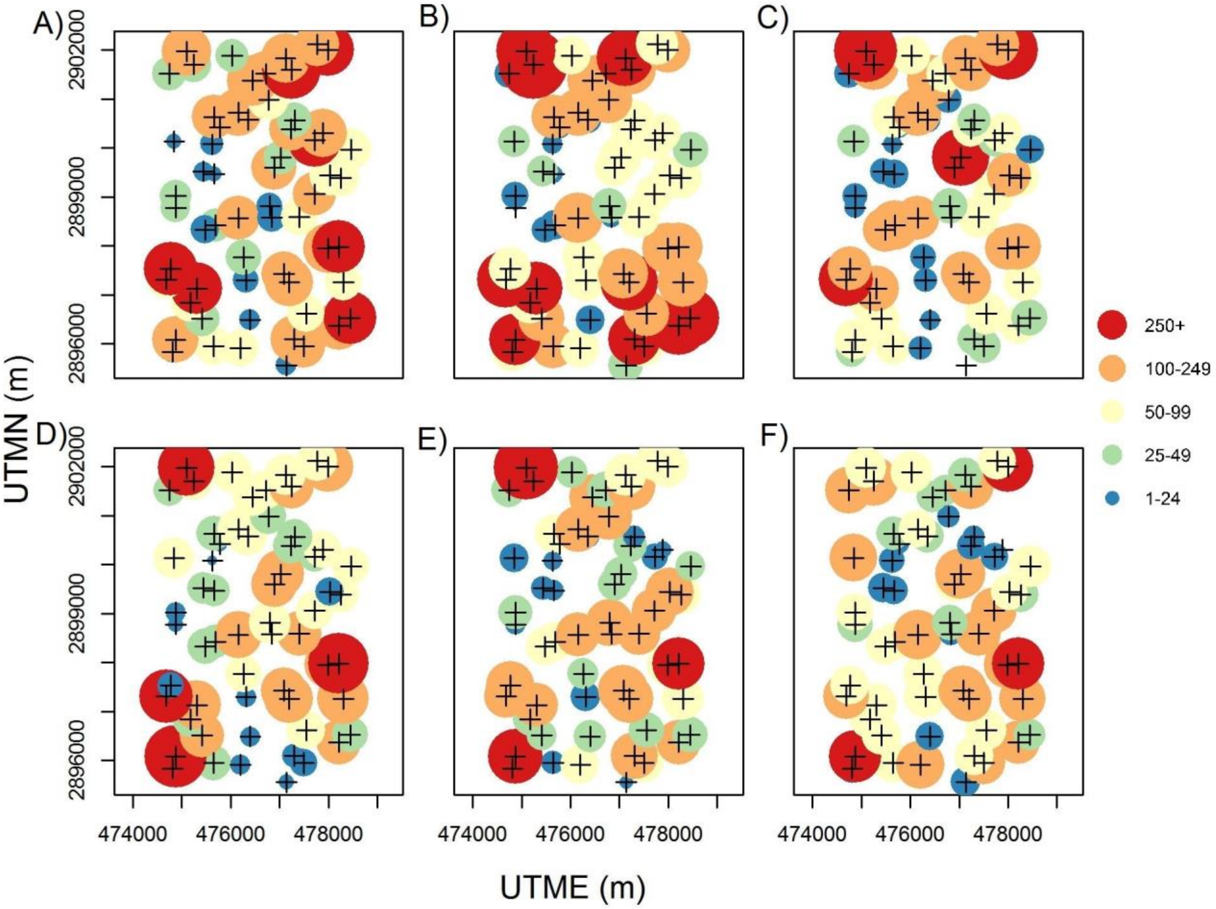


Figure F3. Maps of white-tailed deer camera detections for 6-month periods on Bear Island. The size of each point reflects the number of detections. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

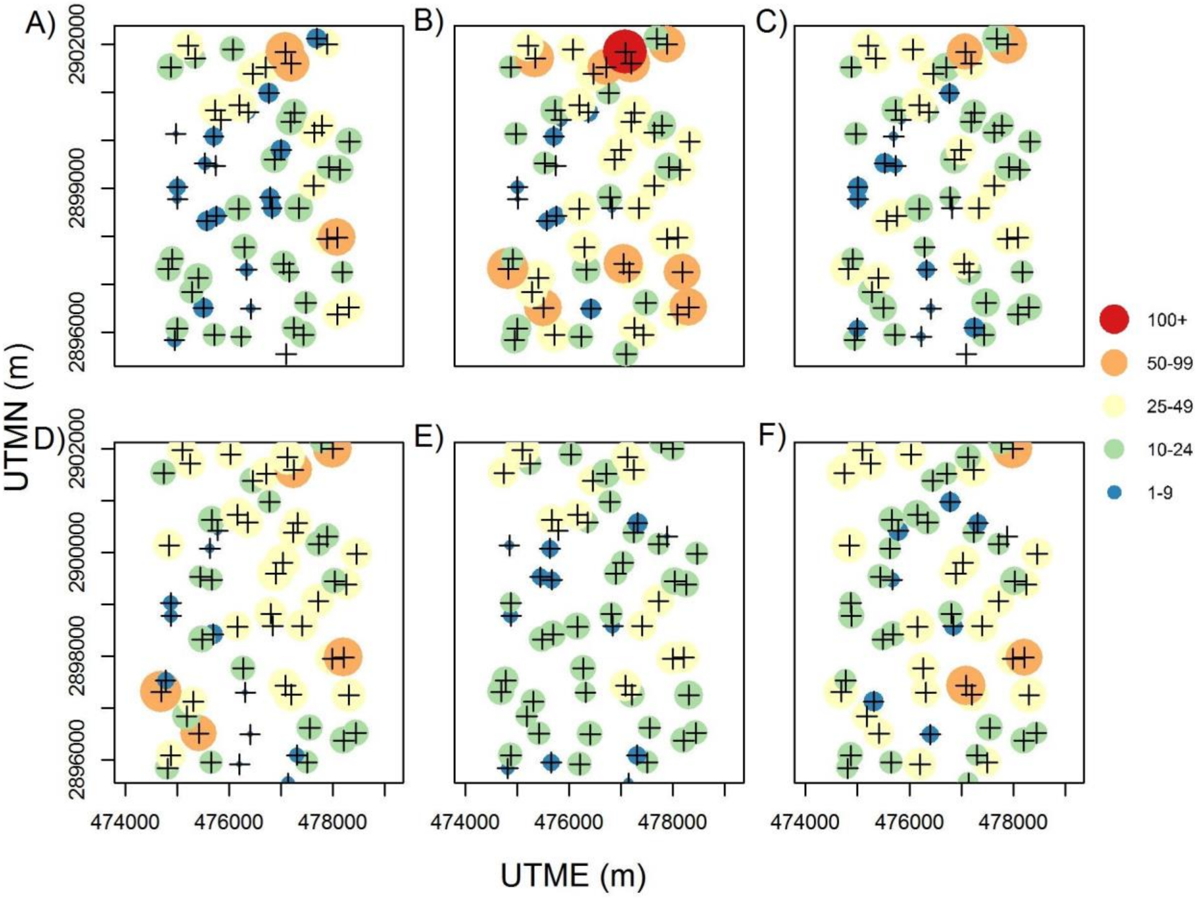


Figure F4. Maps of white-tailed deer daily occurrences for 6-month periods on Bear Island. An occurrence is defined as the detection of at least one deer at a camera during a 24 h sampling occasion. The size of each point reflects the number of daily occurrences. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

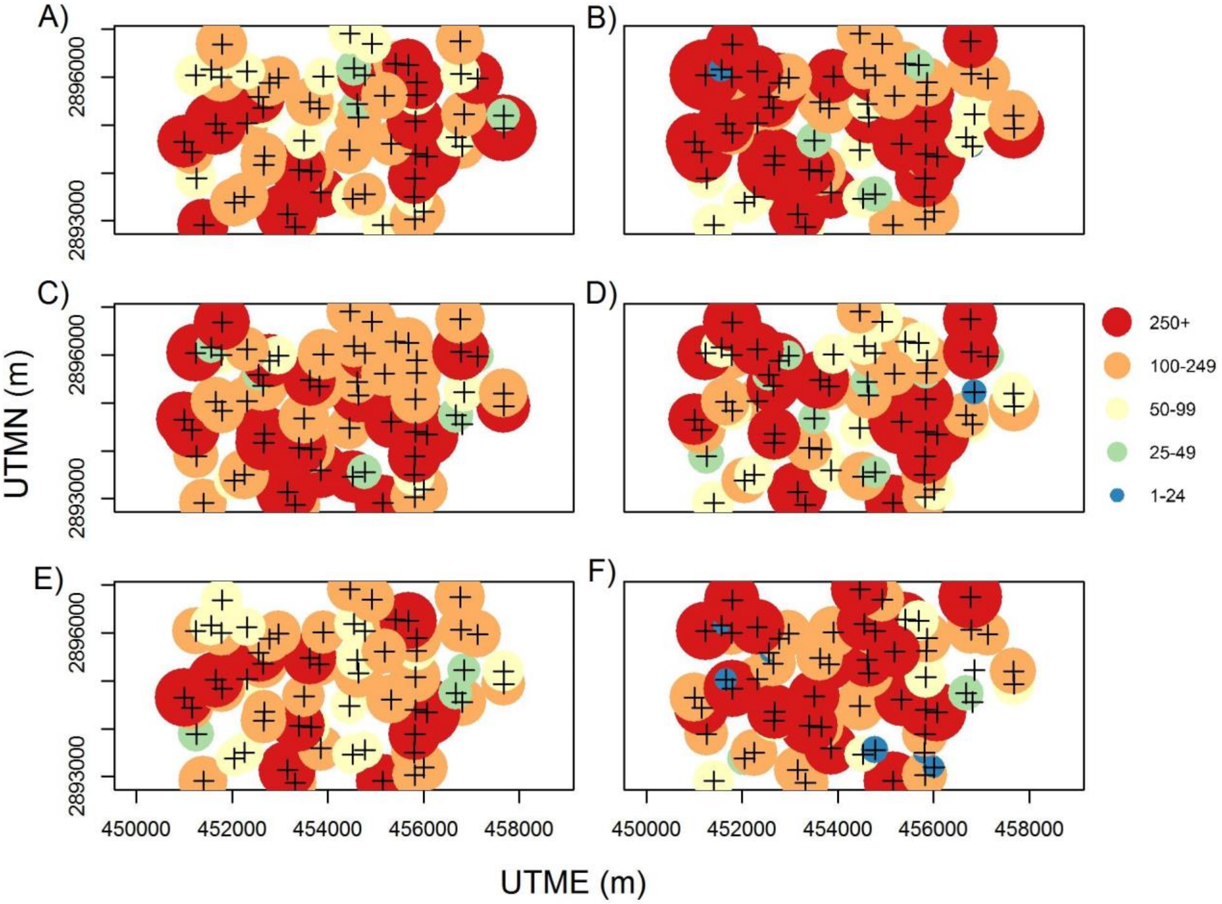


Figure F5. Maps of white-tailed deer camera detections for 6-month periods on Florida Panther National Wildlife Refuge. The size of each point reflects the number of detections. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

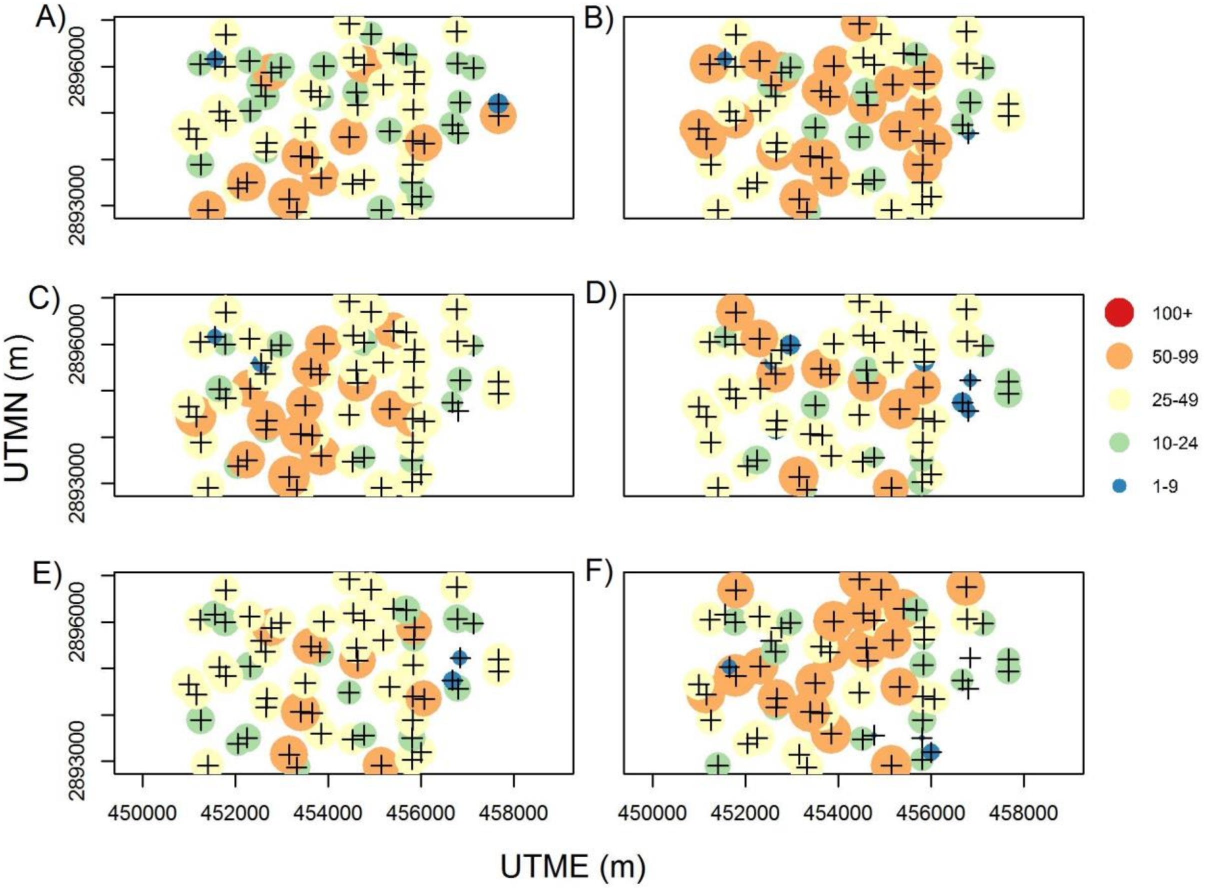


Figure F6. Maps of white-tailed deer daily occurrences for 6-month periods on Florida Panther National Wildlife Refuge. An occurrence is defined as the detection of at least one deer at a camera during a 24 h sampling occasion. The size of each point reflects the number of daily occurrences. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

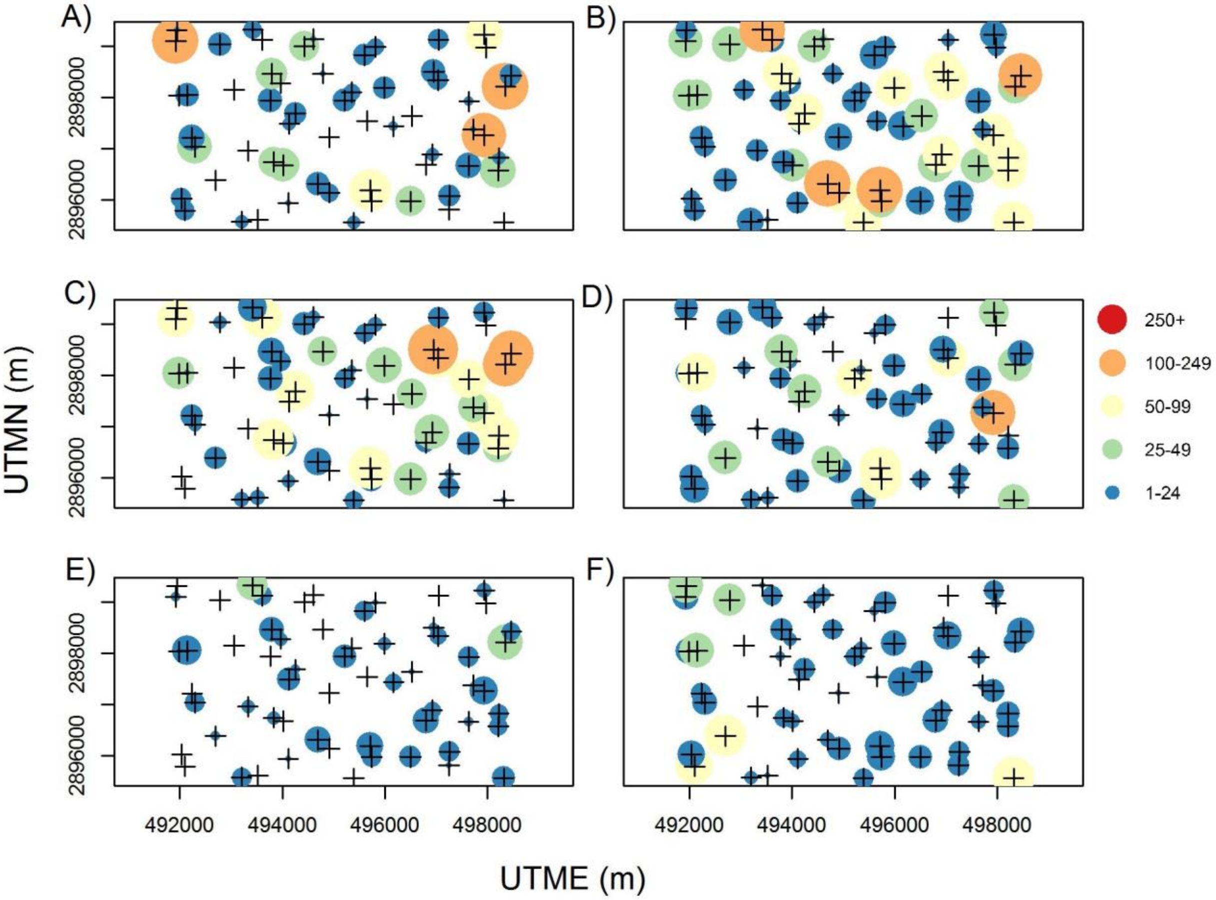


Figure F7. Maps of male white-tailed deer camera detections for 6-month periods on North Addition Lands. The size of each point reflects the number of detections. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

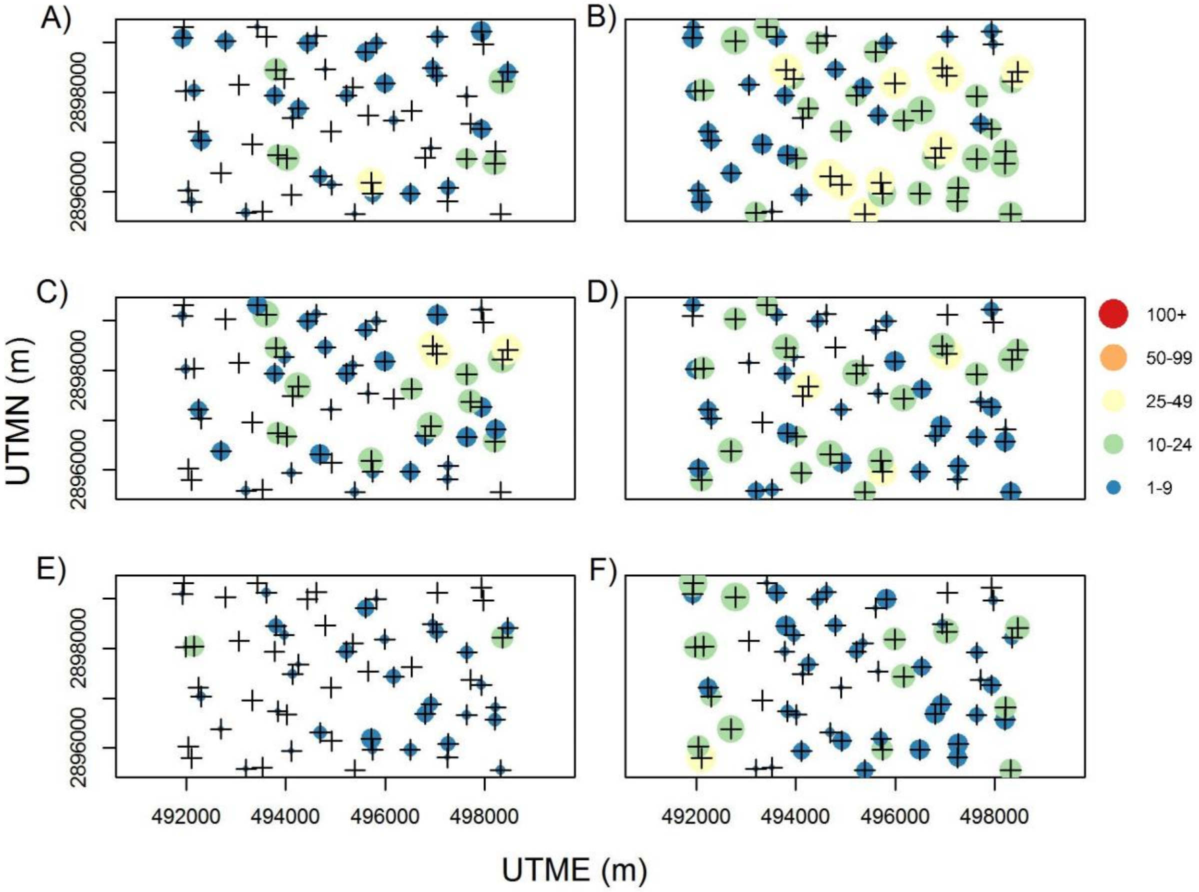


Figure F8. Maps of male white-tailed deer daily occurrences for 6-month periods on North Addition Lands. An occurrence is defined as the detection of at least one deer at a camera during a 24 h sampling occasion. The size of each point reflects the number of daily occurrences. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

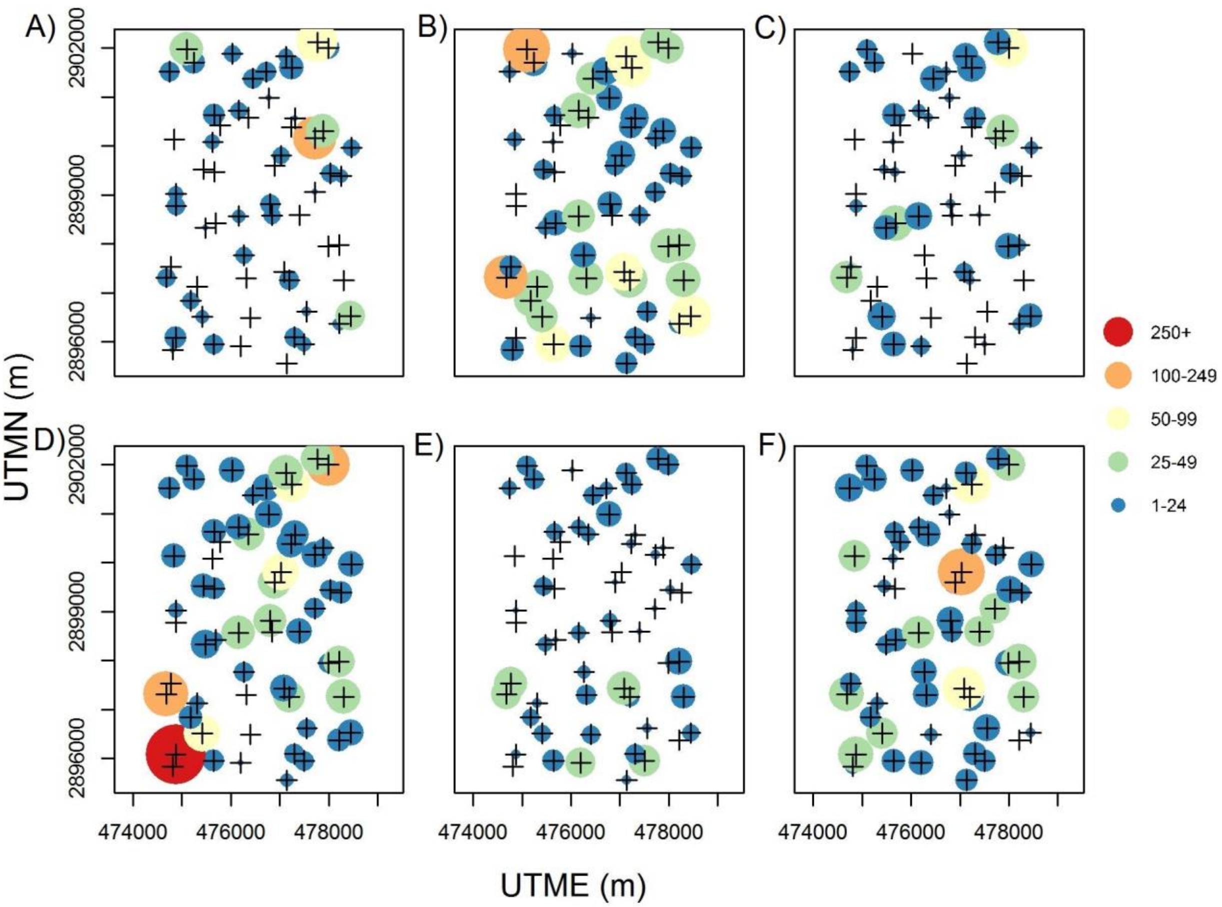


Figure F9. Maps of male white-tailed deer camera detections for 6-month periods on Bear Island. The size of each point reflects the number of detections. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

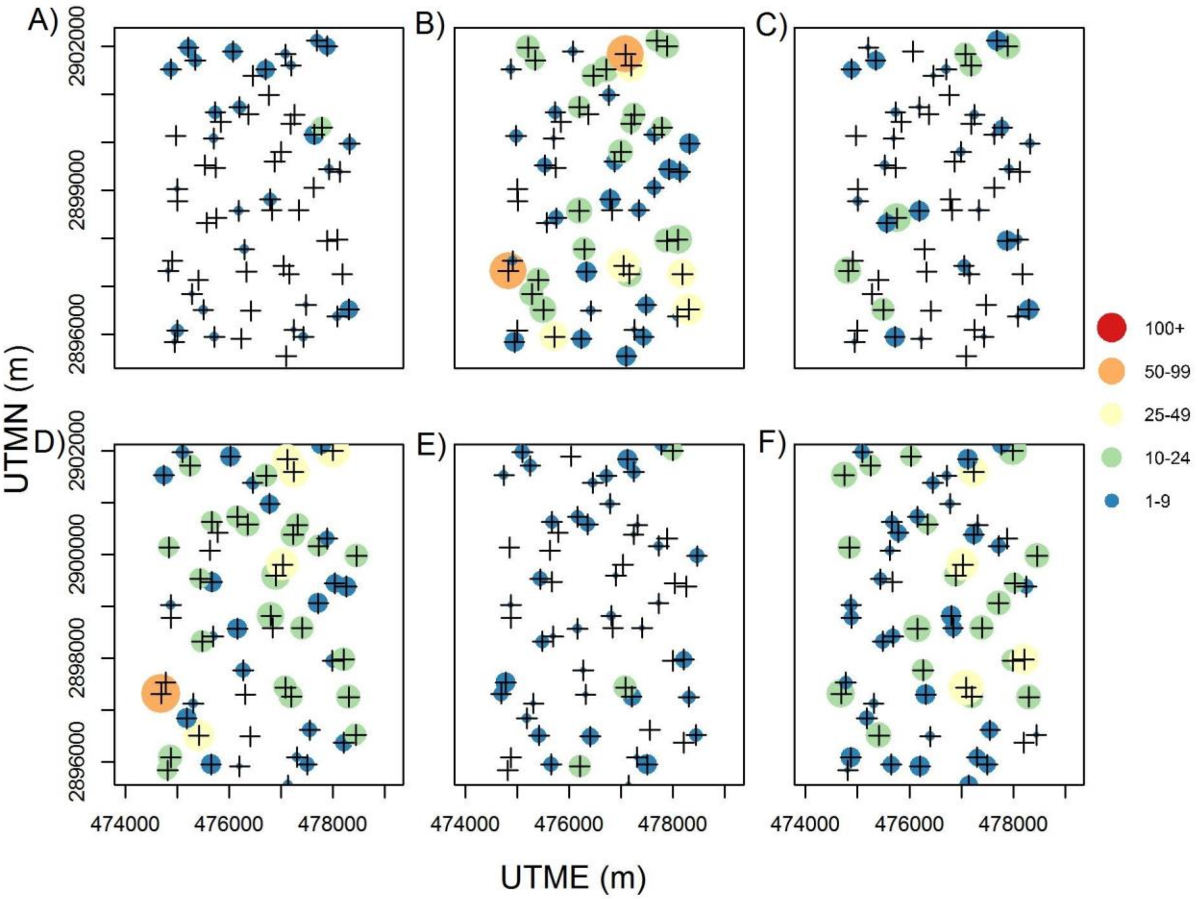


Figure F10. Maps of male white-tailed deer daily occurrences for 6-month periods on Bear Island. An occurrence is defined as the detection of at least one deer at a camera during a 24 h sampling occasion. The size of each point reflects the number of daily occurrences. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

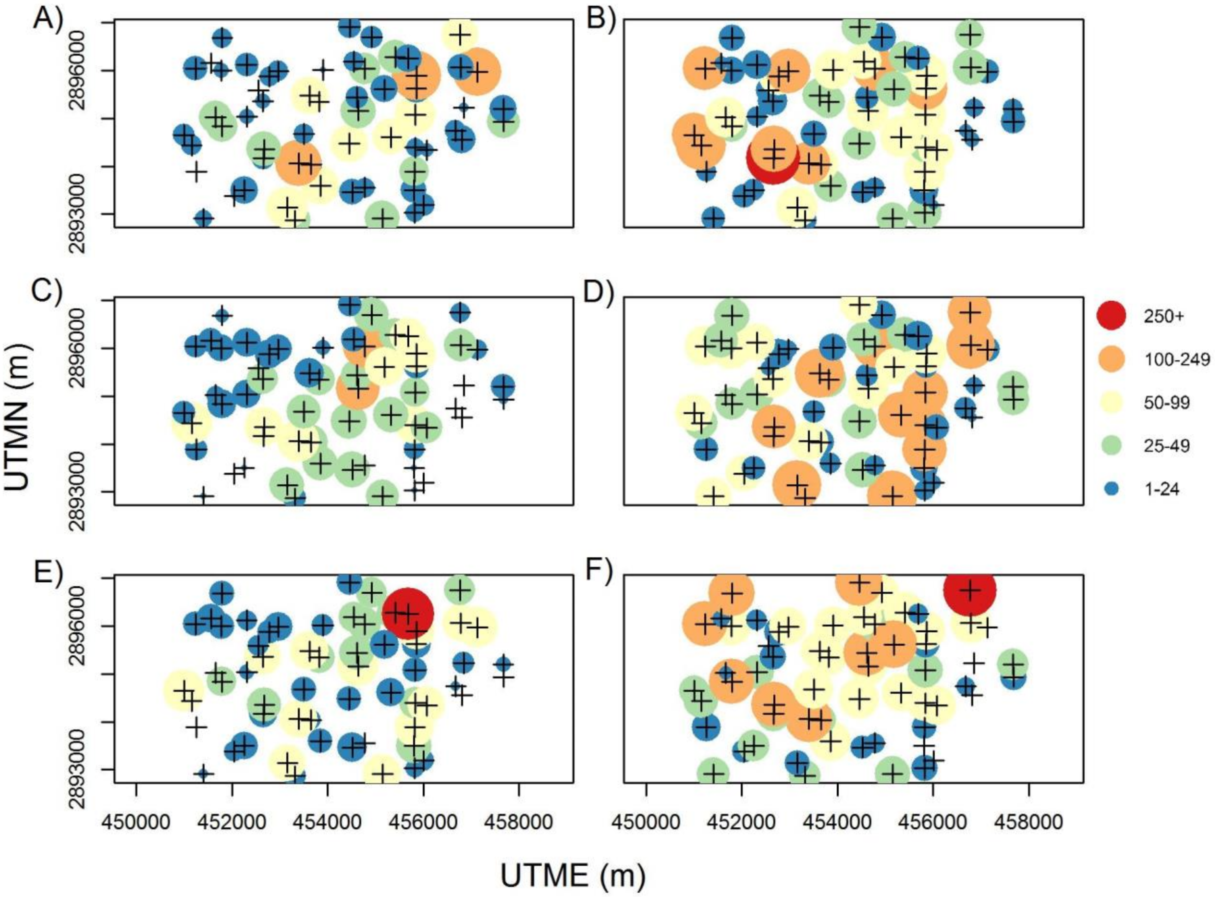


Figure F11. Maps of male white-tailed deer camera detections for 6-month periods on Florida Panther National Wildlife Refuge. The size of each point reflects the number of detections. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

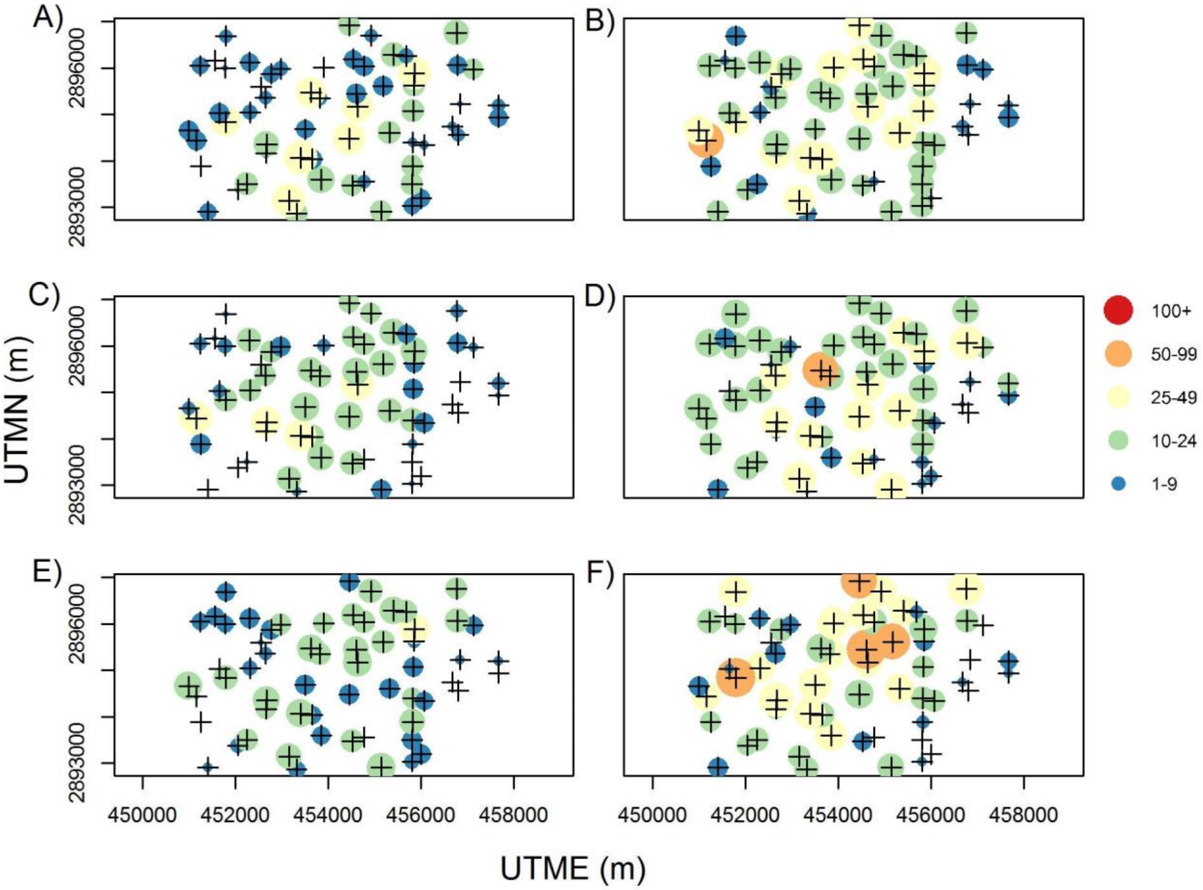


Figure F12. Maps of male white-tailed deer daily occurrences for 6-month periods on Florida Panther National Wildlife Refuge. An occurrence is defined as the detection of at least one deer at a camera during a 24 h sampling occasion. The size of each point reflects the number of daily occurrences. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

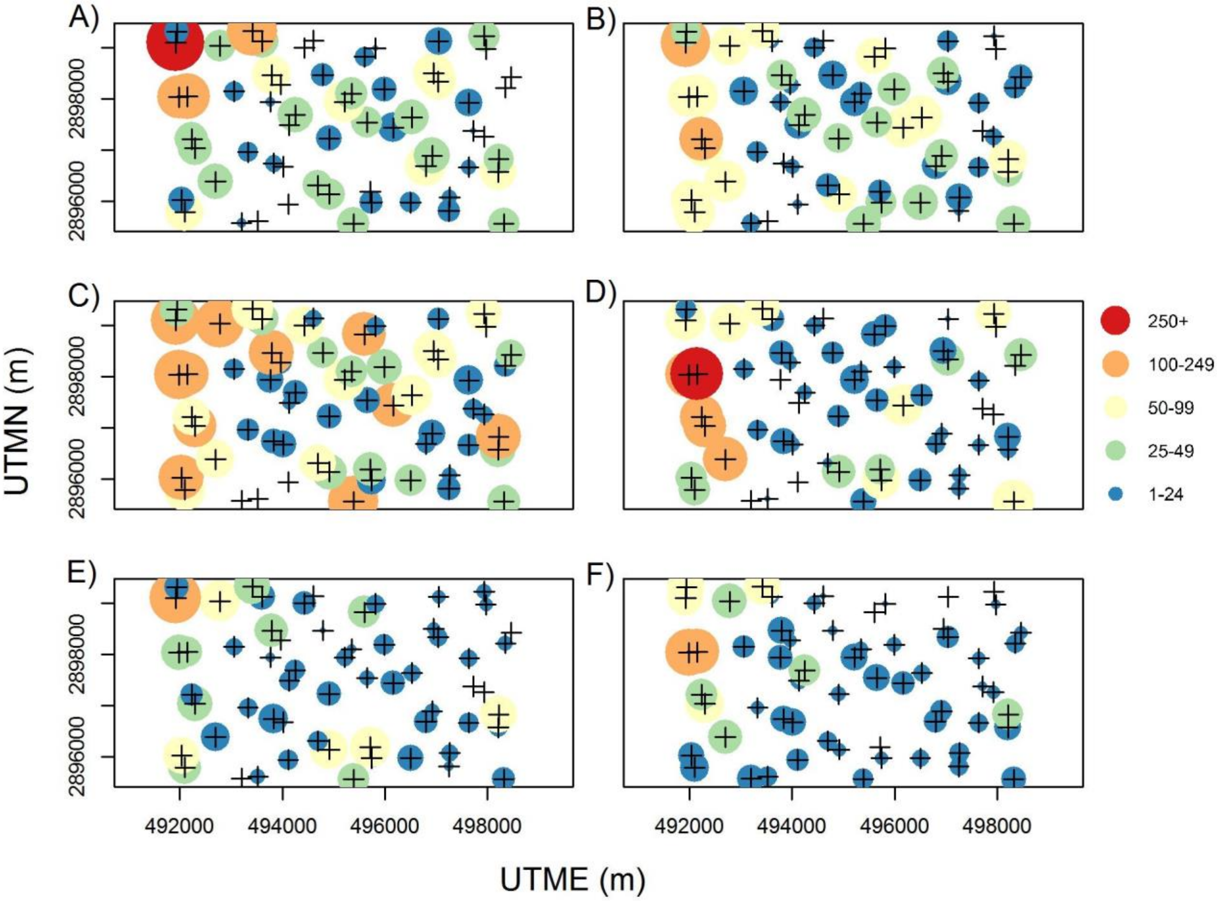


Figure F13. Maps of female white-tailed deer camera detections for 6-month periods on North Addition Lands. The size of each point reflects the number of detections. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

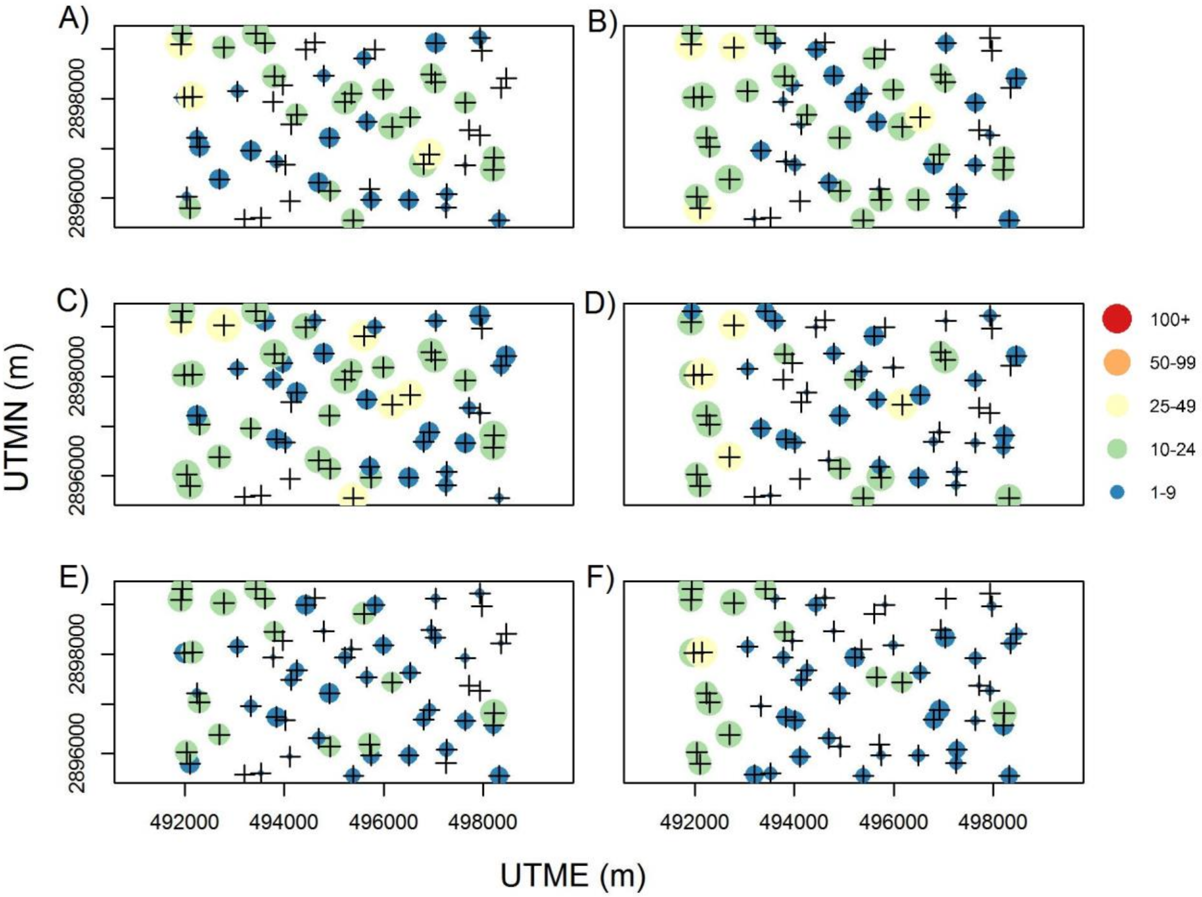


Figure F14. Maps of female white-tailed deer daily occurrences for 6-month periods on North Addition Lands. An occurrence is defined as the detection of at least one deer at a camera during a 24 h sampling occasion. The size of each point reflects the number of daily occurrences. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

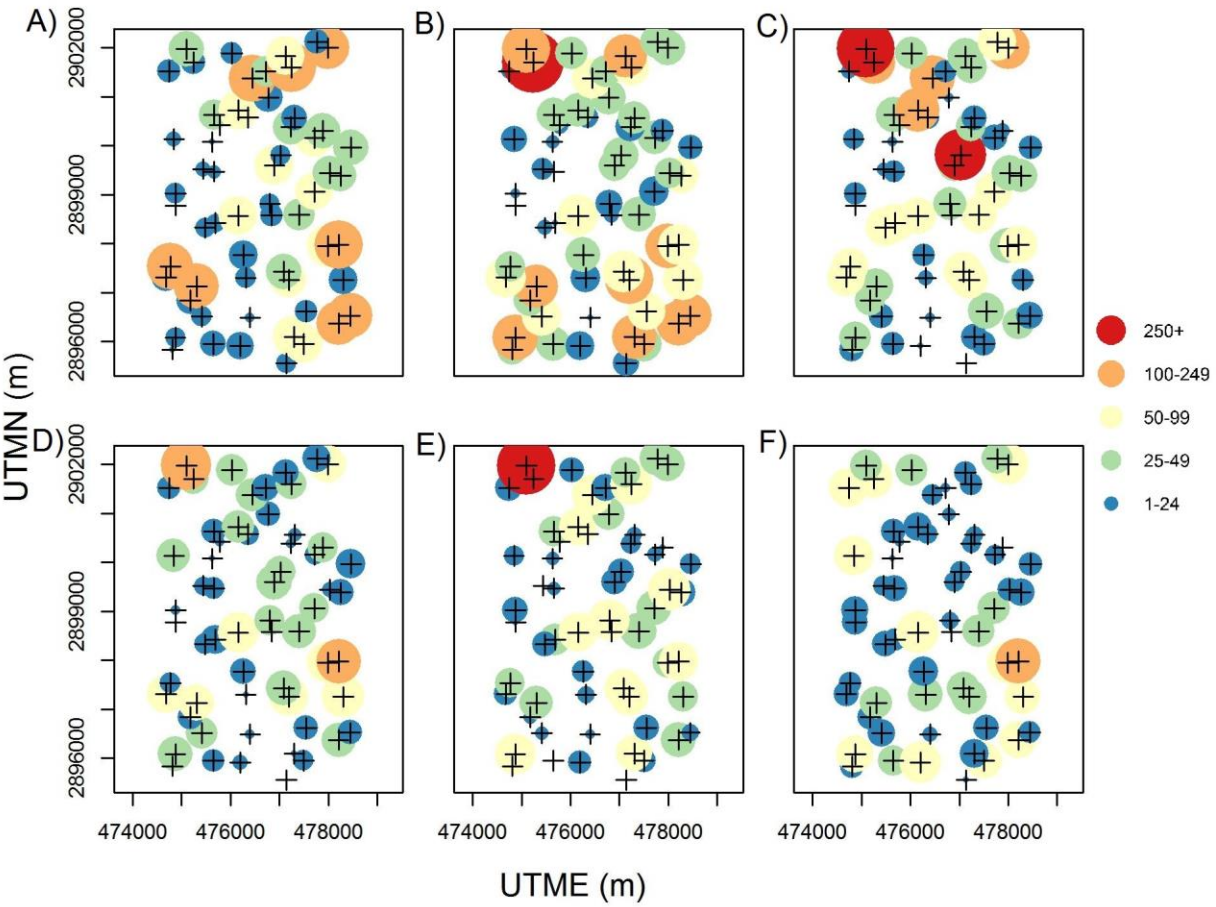


Figure F15. Maps of female white-tailed deer camera detections for 6-month periods on Bear Island. The size of each point reflects the number of detections. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

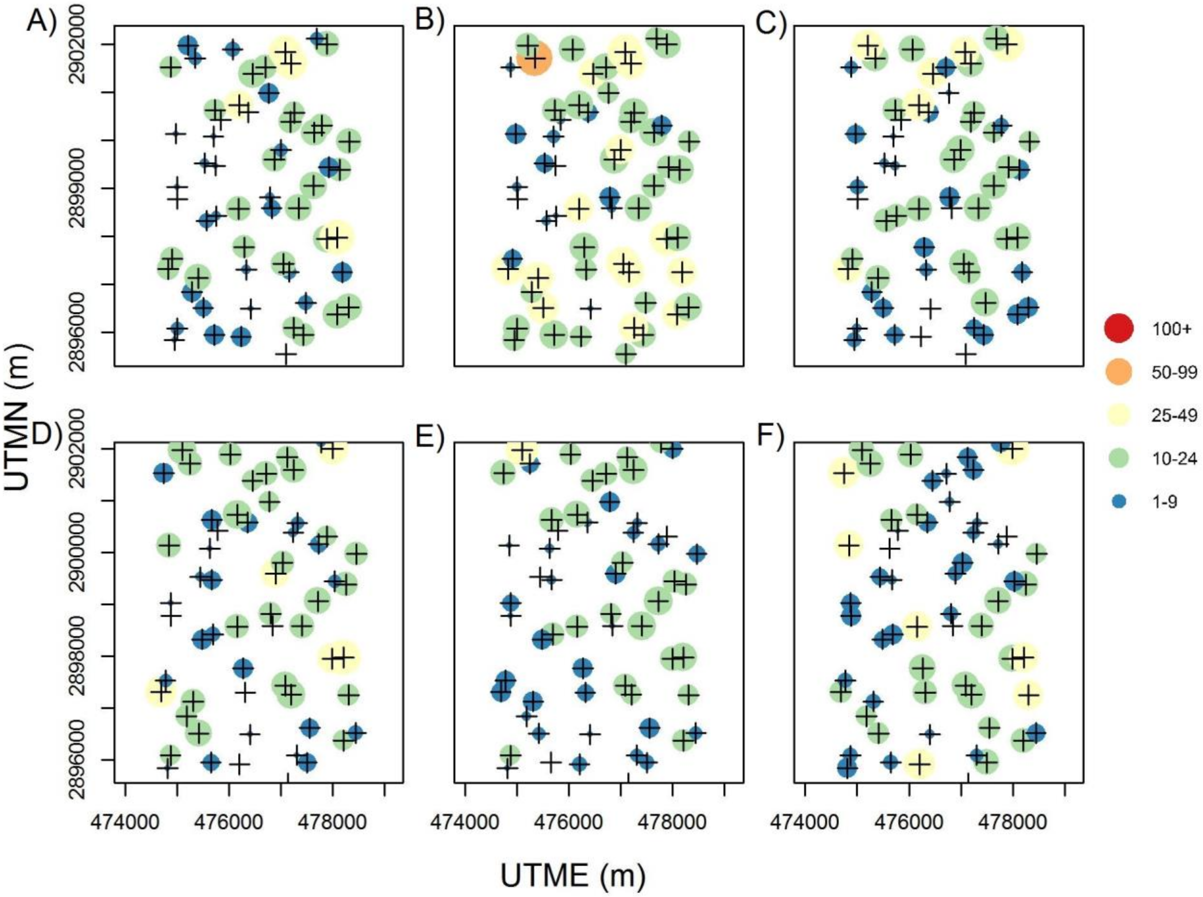


Figure F16. Maps of female white-tailed deer daily occurrences for 6-month periods on Bear Island. An occurrence is defined as the detection of at least one deer at a camera during a 24 h sampling occasion. The size of each point reflects the number of daily occurrences. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

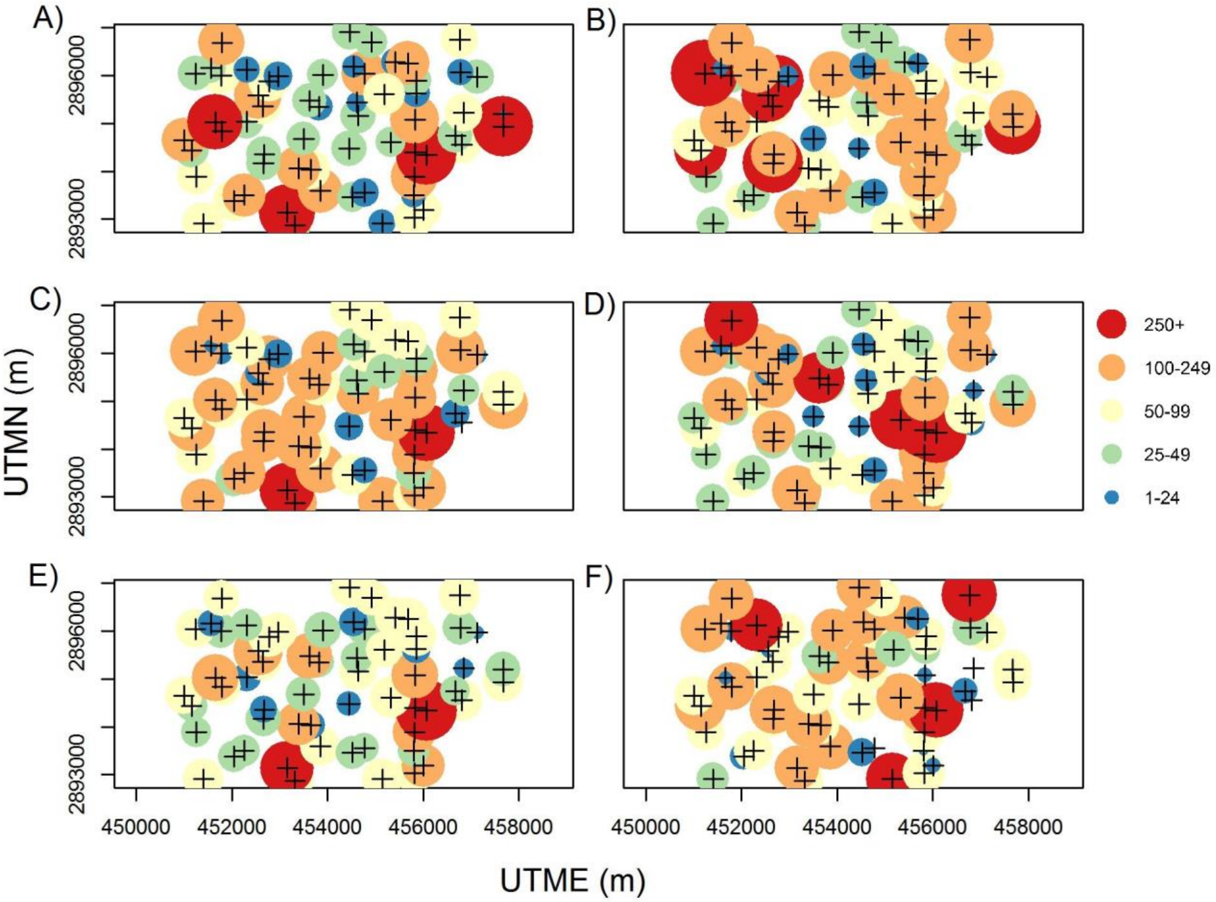


Figure F17. Maps of female white-tailed deer camera detections for 6-month periods on Florida Panther National Wildlife Refuge. The size of each point reflects the number of detections. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

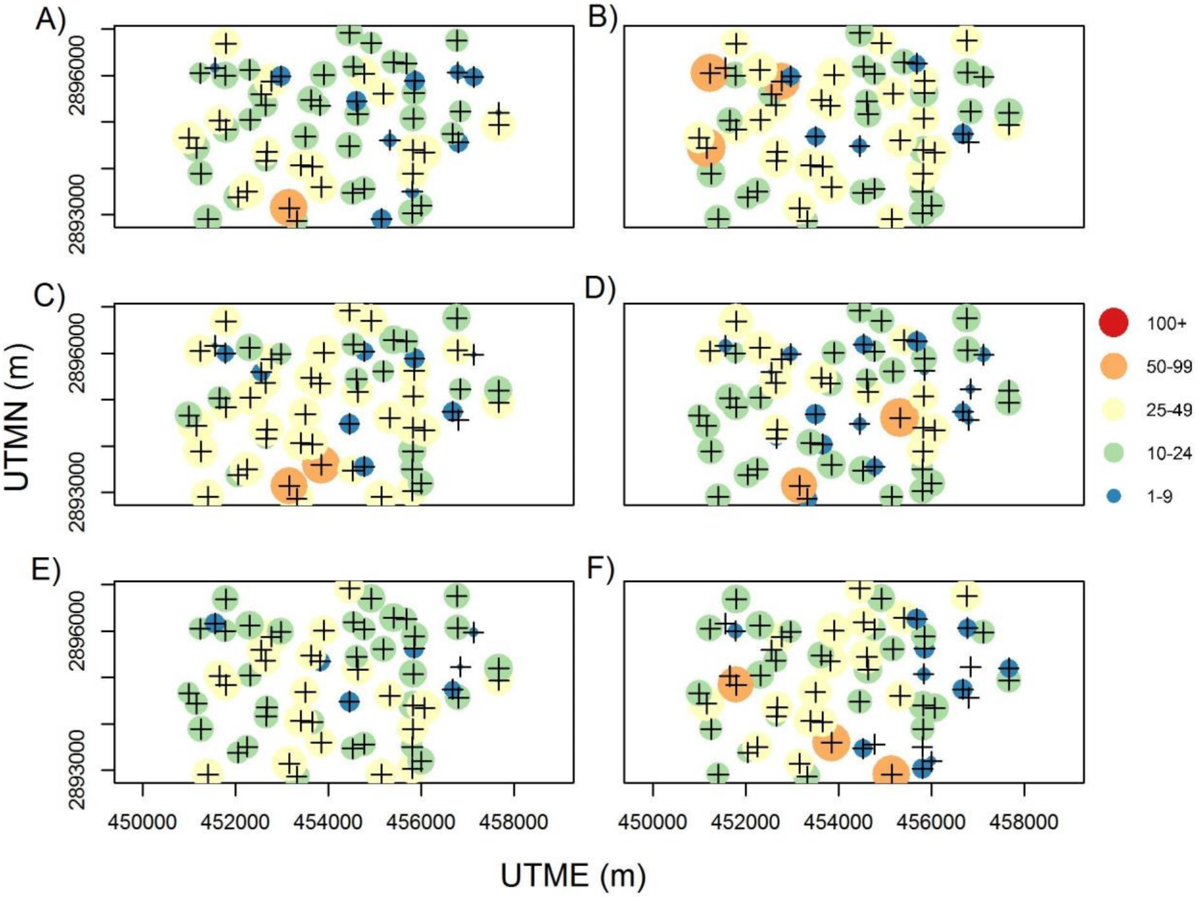


Figure F18. Maps of female white-tailed deer daily occurrences for 6-month periods on Florida Panther National Wildlife Refuge. An occurrence is defined as the detection of at least one deer at a camera during a 24 h sampling occasion. The size of each point reflects the number of daily occurrences. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

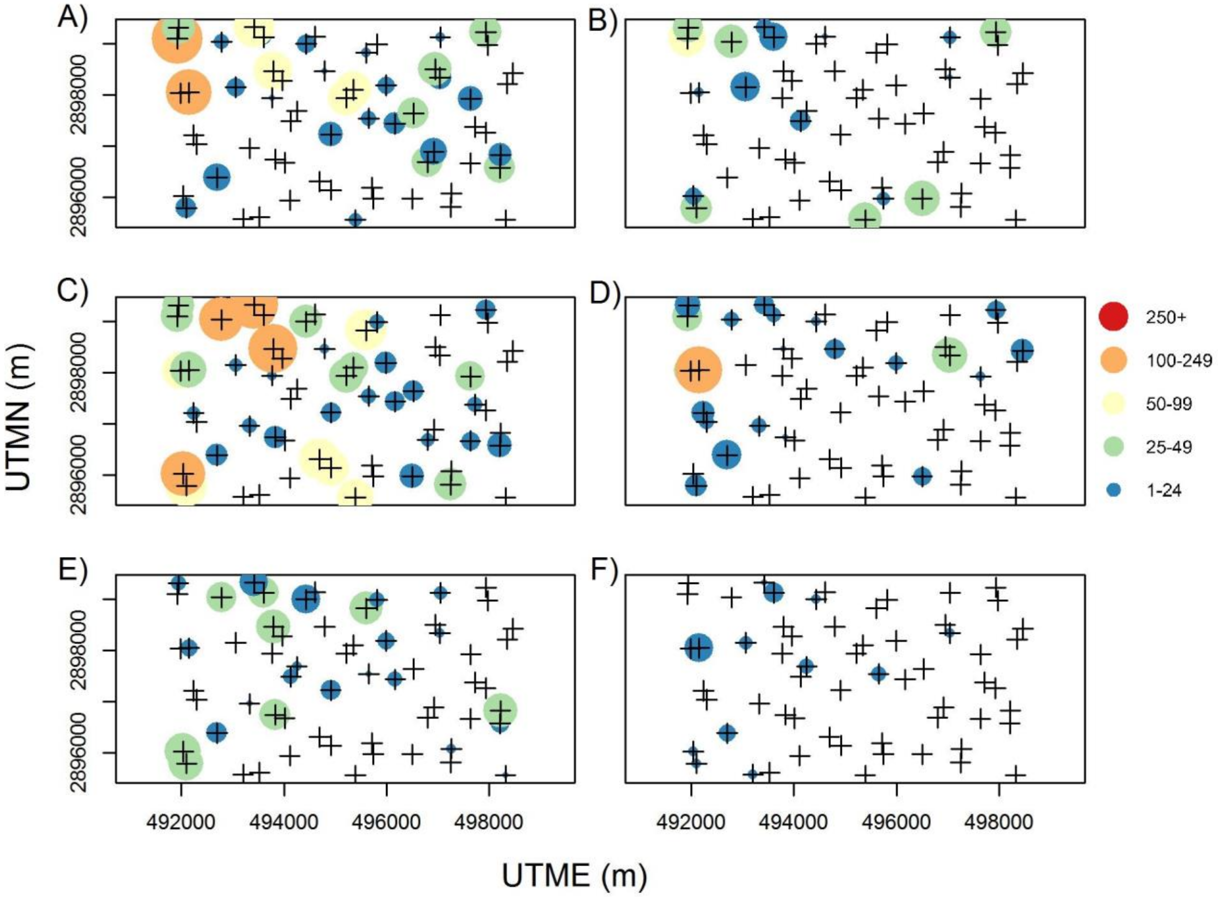


Figure F19. Map of spotted fawn camera detections for 6-month periods on North Addition Lands. The size of each point reflects the number of detections. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

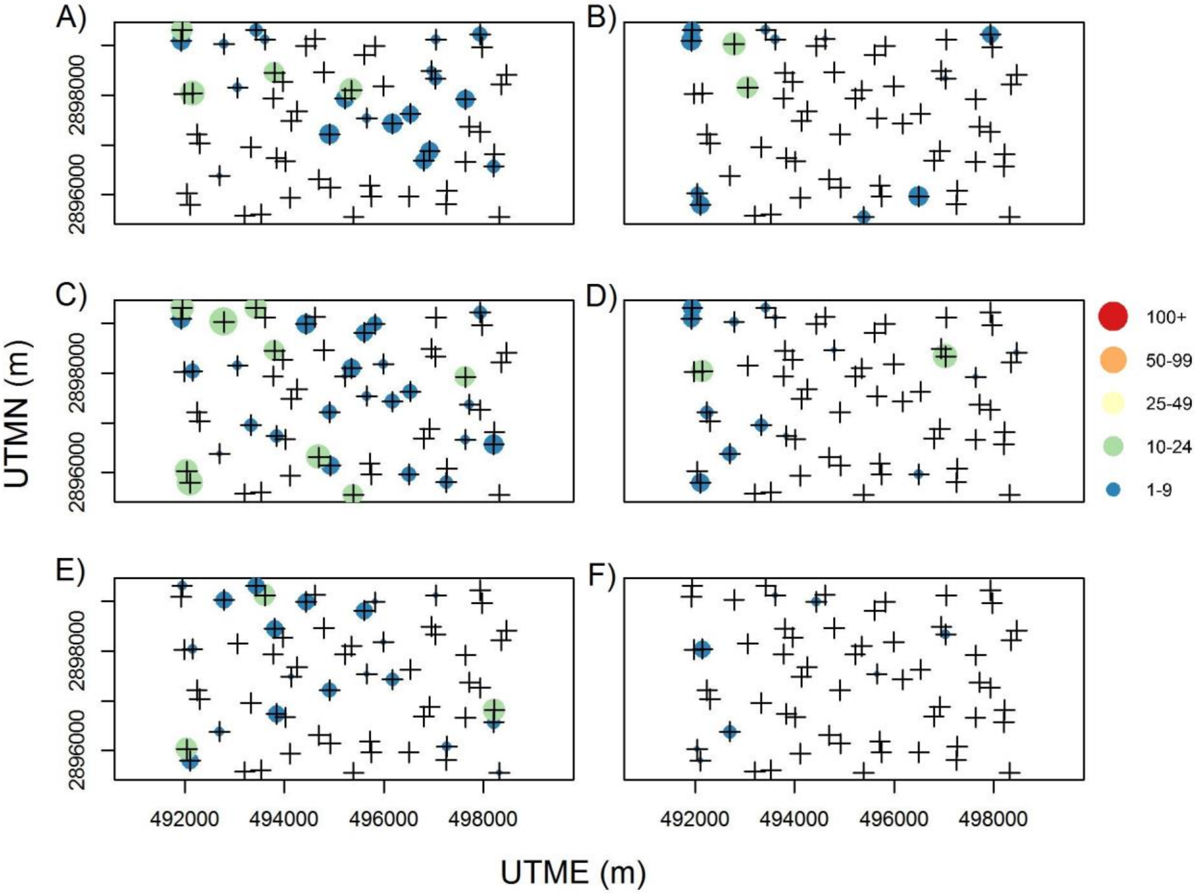


Figure F20. Maps of spotted fawn daily occurrences for 6-month periods on North Addition Lands. An occurrence is defined as the detection of at least one deer at a camera during a 24 h sampling occasion. The size of each point reflects the number of daily occurrences. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

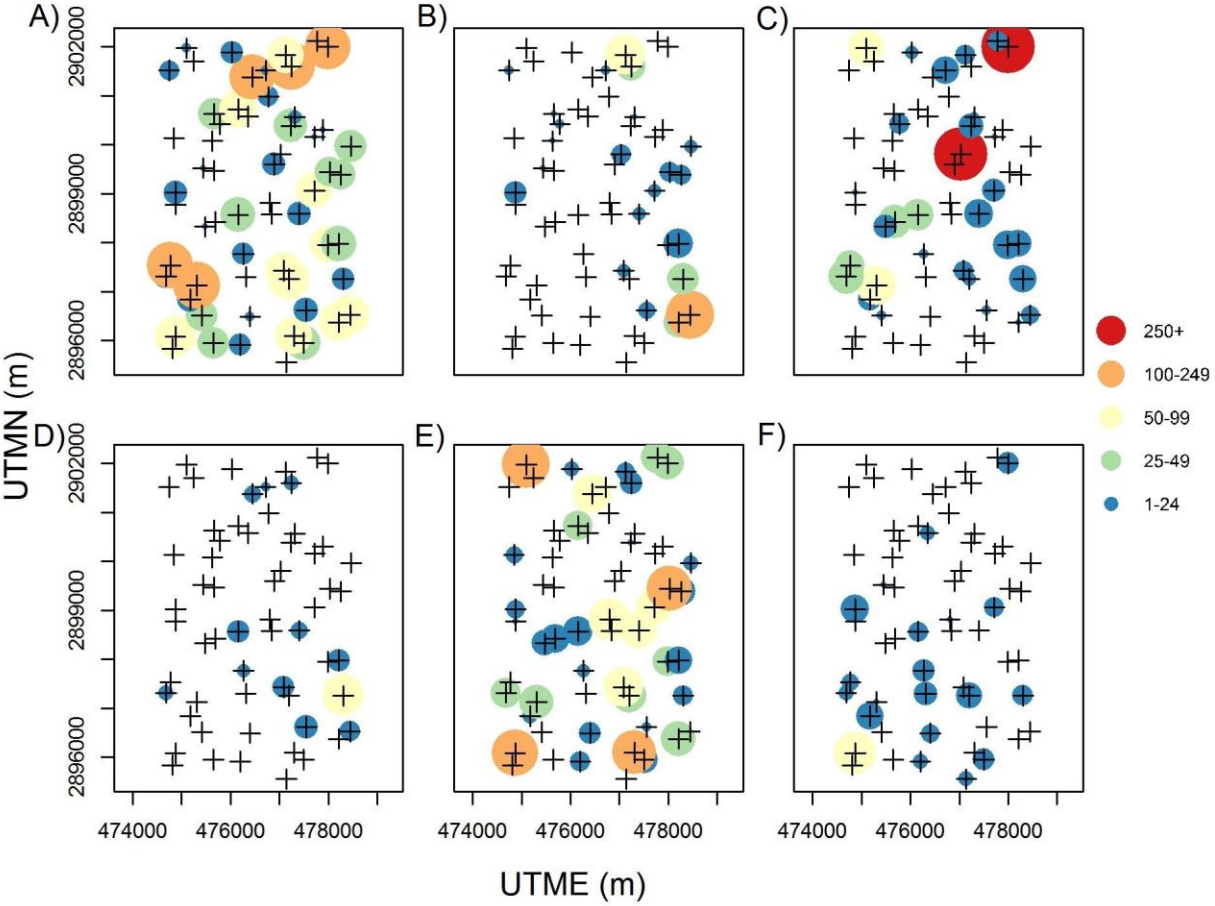


Figure F21. Maps of spotted fawn camera detections for 6-month periods on Bear Island. The size of each point reflects the number of detections. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

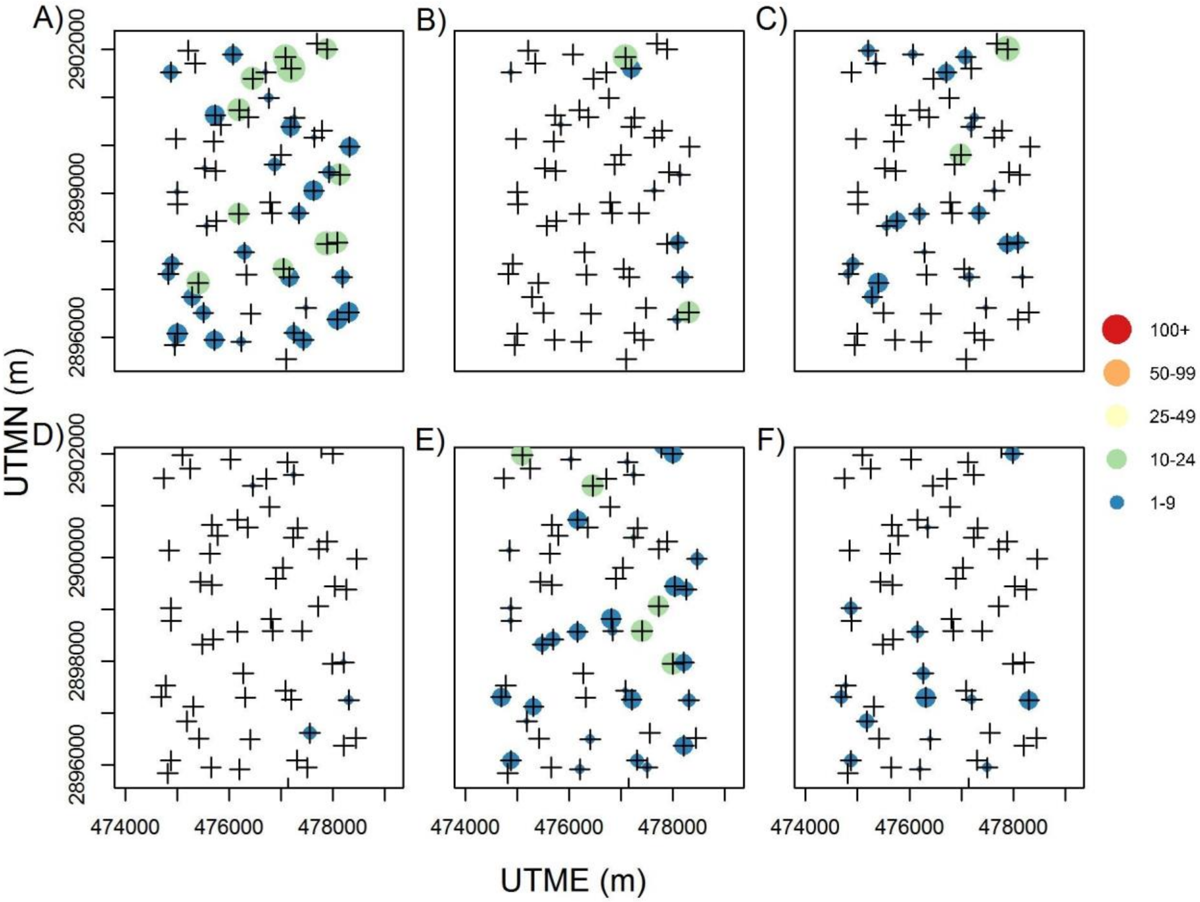


Figure F22. Maps of spotted fawn daily occurrences for 6-month periods on Bear Island. An occurrence is defined as the detection of at least one deer at a camera during a 24 h sampling occasion. The size of each point reflects the number of daily occurrences. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

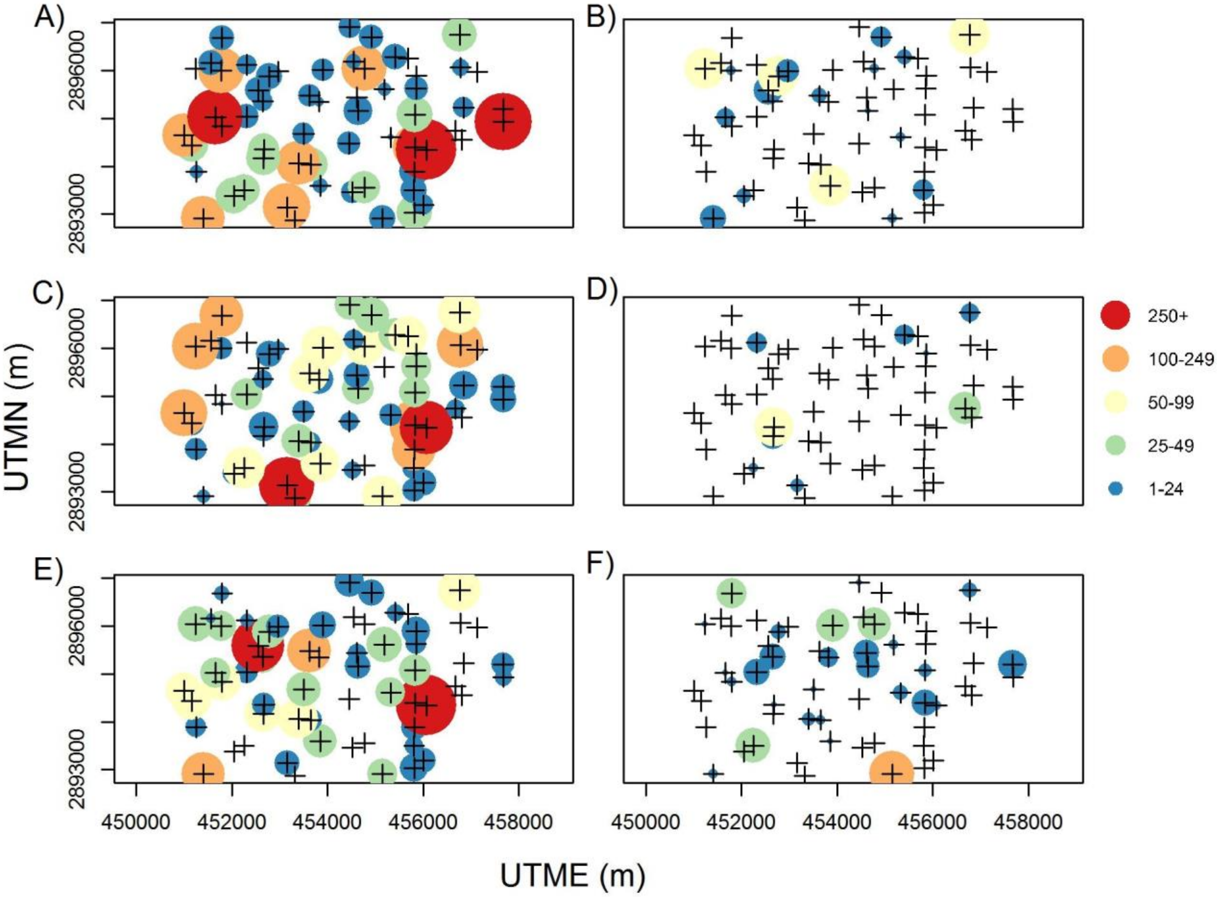


Figure F23. Maps of spotted fawn camera detections for 6-month periods on Florida Panther National Wildlife Refuge. The size of each point reflects the number of detections. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

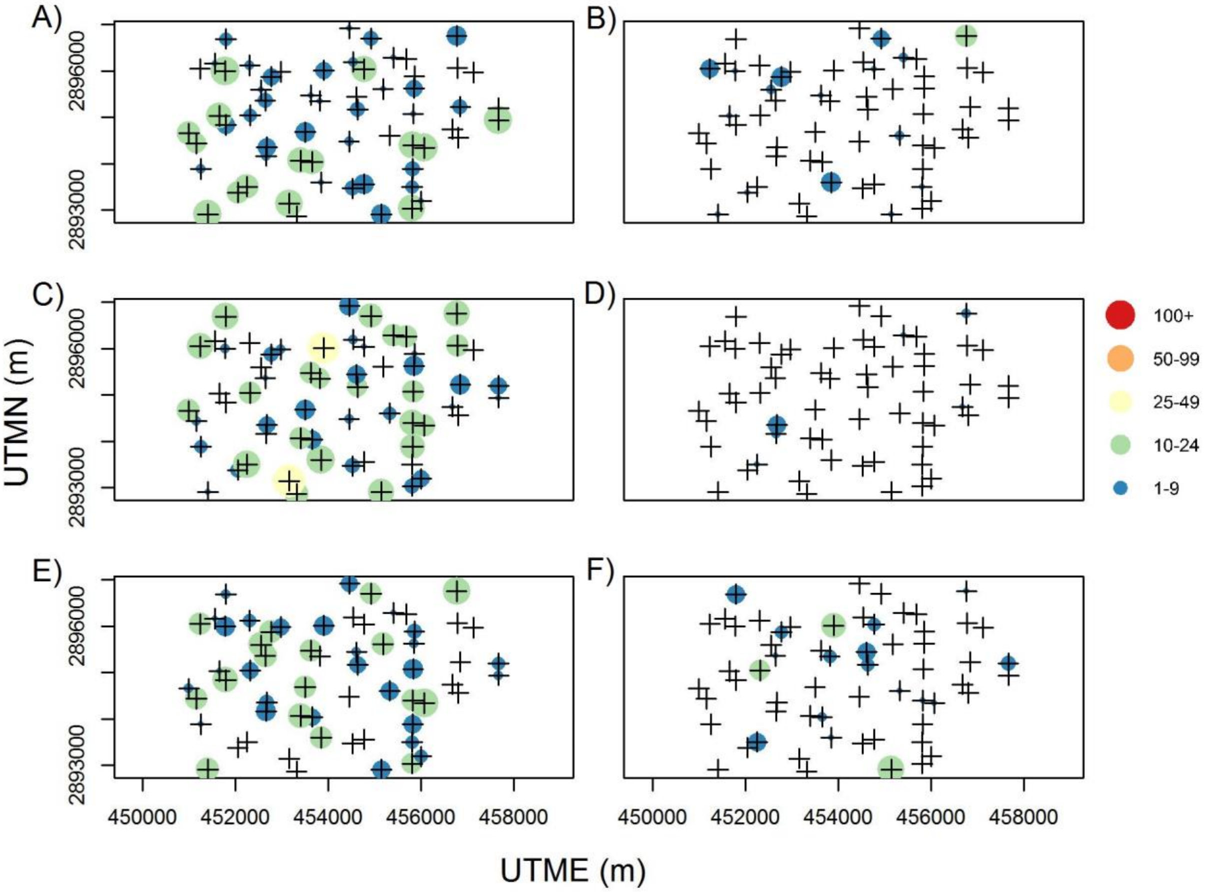


Figure F24. Maps of fawn daily occurrences for 6-month periods on Florida Panther National Wildlife Refuge. An occurrence is defined as the detection of at least one deer at a camera during a 24 h sampling occasion. The size of each point reflects the number of daily occurrences. A) January - June 2015; B) July - December 2015; C) January - June 2016; D) July - December 2016; E) January - June 2017; F) July - December 2017.

Appendix G. SCR Model λ and σ Estimates

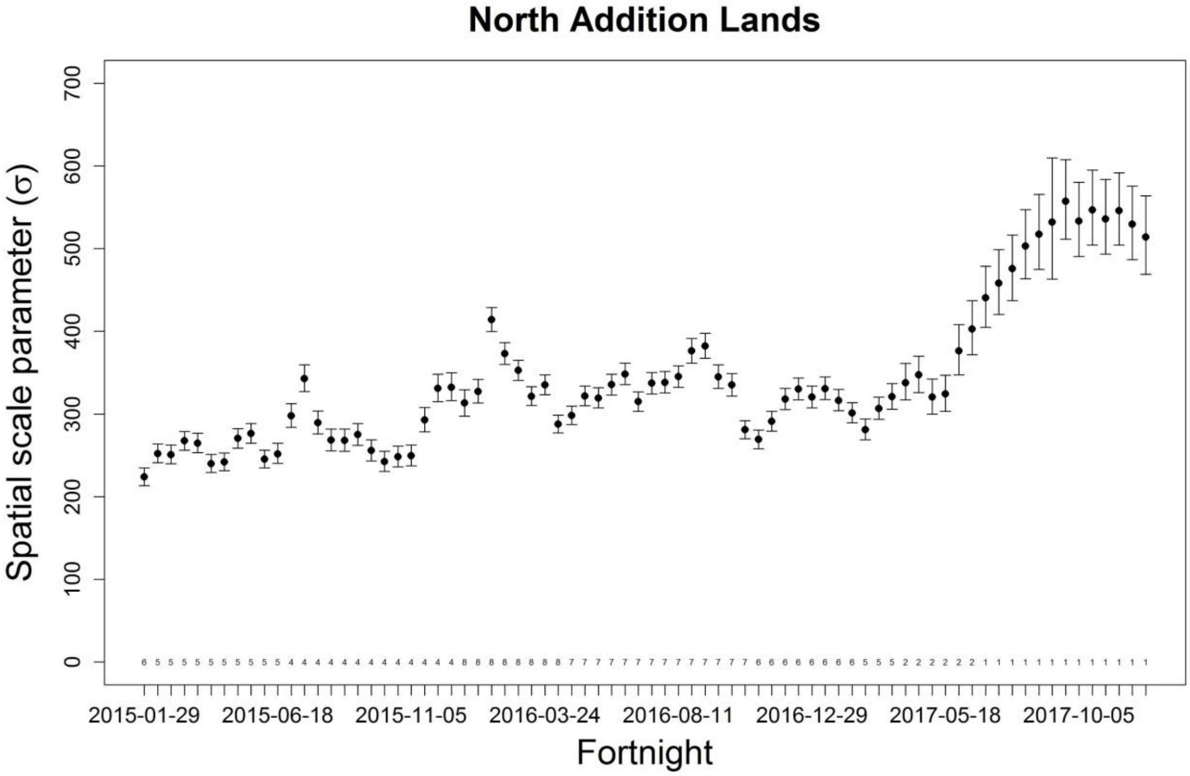


Figure G1. Biweekly estimates and 95% credible intervals of the spatial scale parameter (σ) for GPS-collared female deer on North Addition Lands. Sample size for each fortnight is indicated along the bottom.

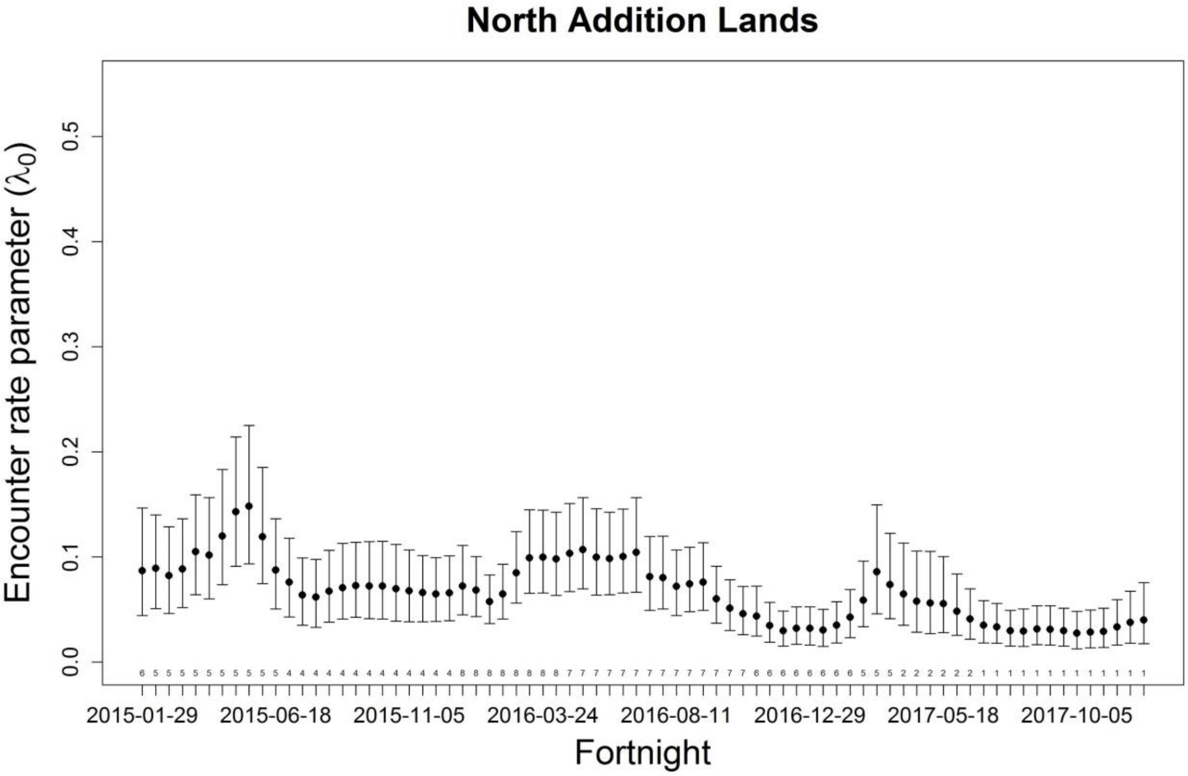


Figure G2. Biweekly estimates and 95% credible intervals of the baseline encounter rate parameter (λ_0) for GPS-collared female deer on North Addition Lands. Sample size for each fortnight is indicated along the bottom.

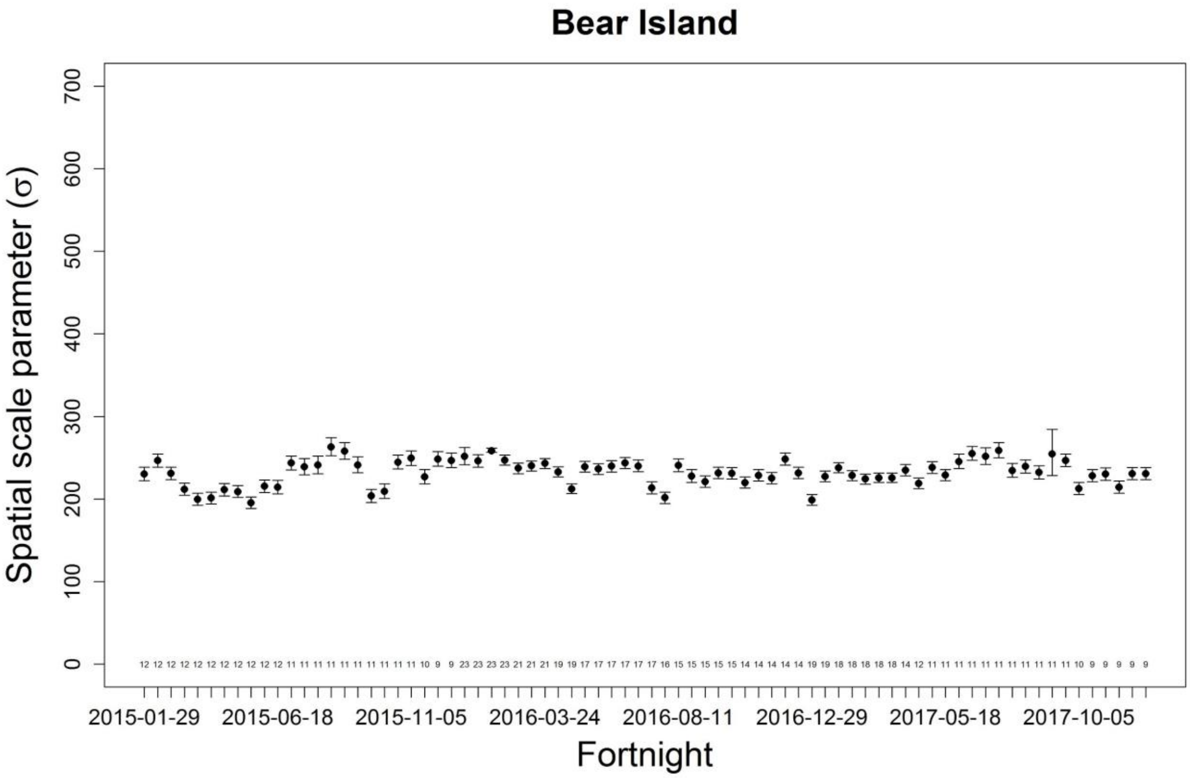


Figure G3. Biweekly estimates and 95% credible intervals of the spatial scale parameter (σ) for GPS-collared female deer on Bear Island. Sample size for each fortnight is indicated along the bottom.

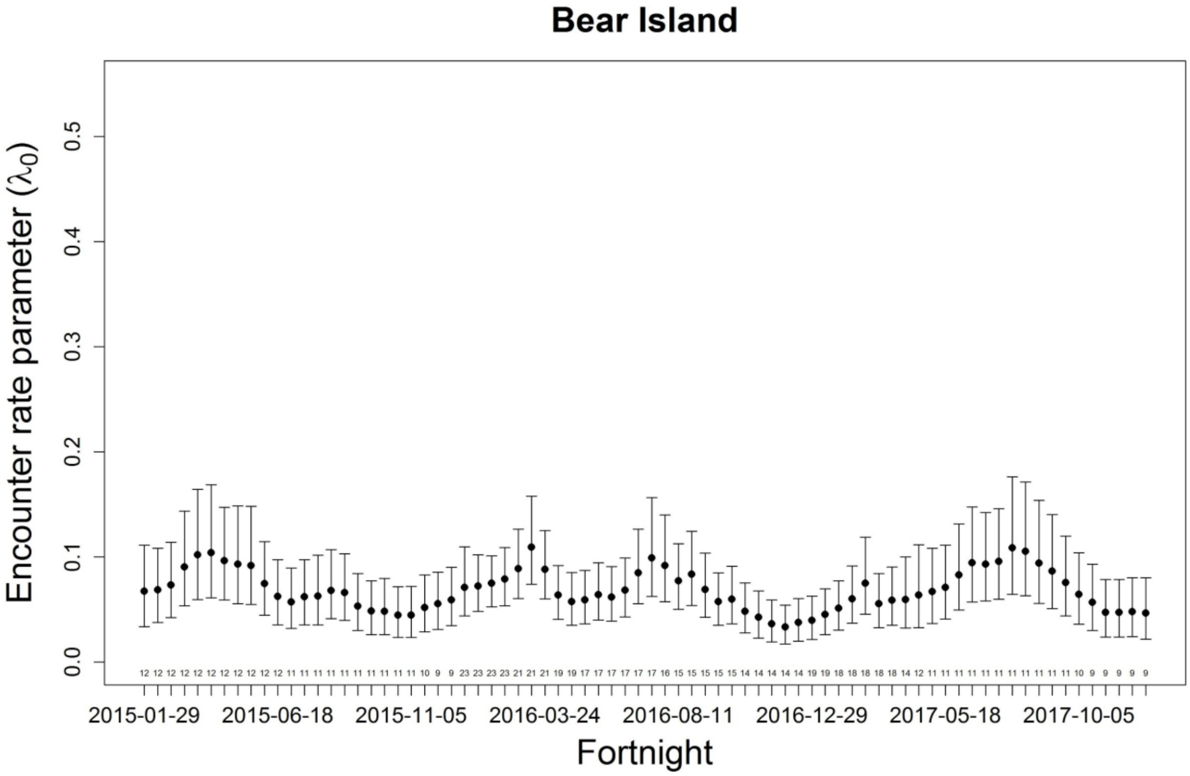


Figure G4. Biweekly estimates and 95% credible intervals of the baseline encounter rate parameter (λ_0) for GPS-collared female deer on Bear Island. Sample size for each fortnight is indicated along the bottom.

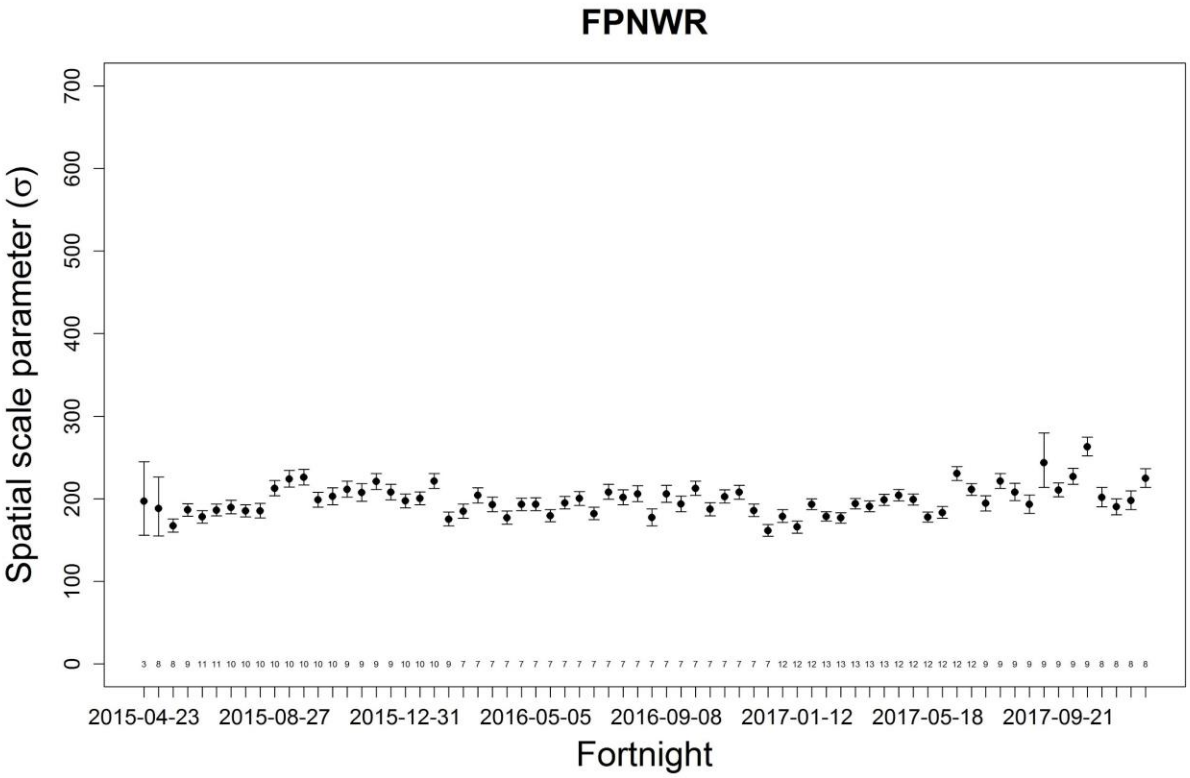


Figure G5. Biweekly estimates and 95% credible intervals of the spatial scale parameter (σ) for GPS-collared female deer on Florida Panther National Wildlife Refuge (FPNWR). Sample size for each fortnight is indicated along the bottom.

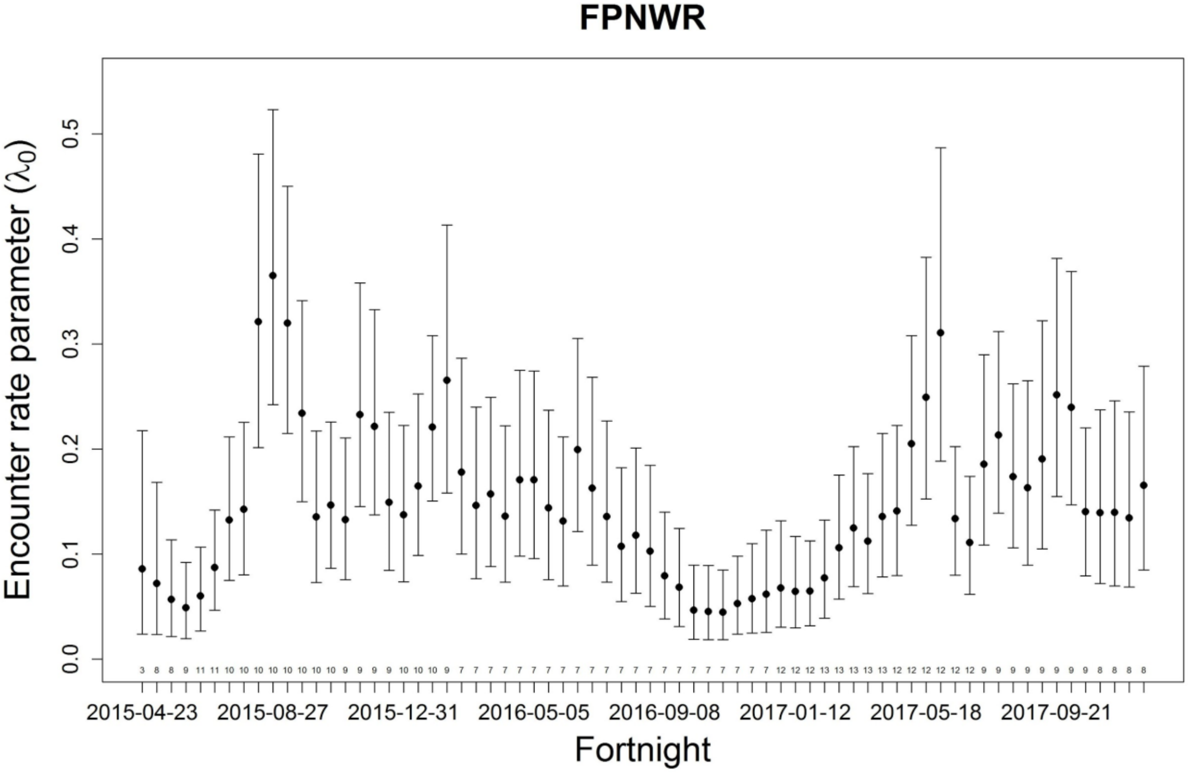


Figure G6. Biweekly estimates and 95% credible intervals of the baseline encounter rate parameter (λ_0) for GPS-collared female deer on Florida Panther National Wildlife Refuge (FPNWR). Sample size for each fortnight is indicated along the bottom.

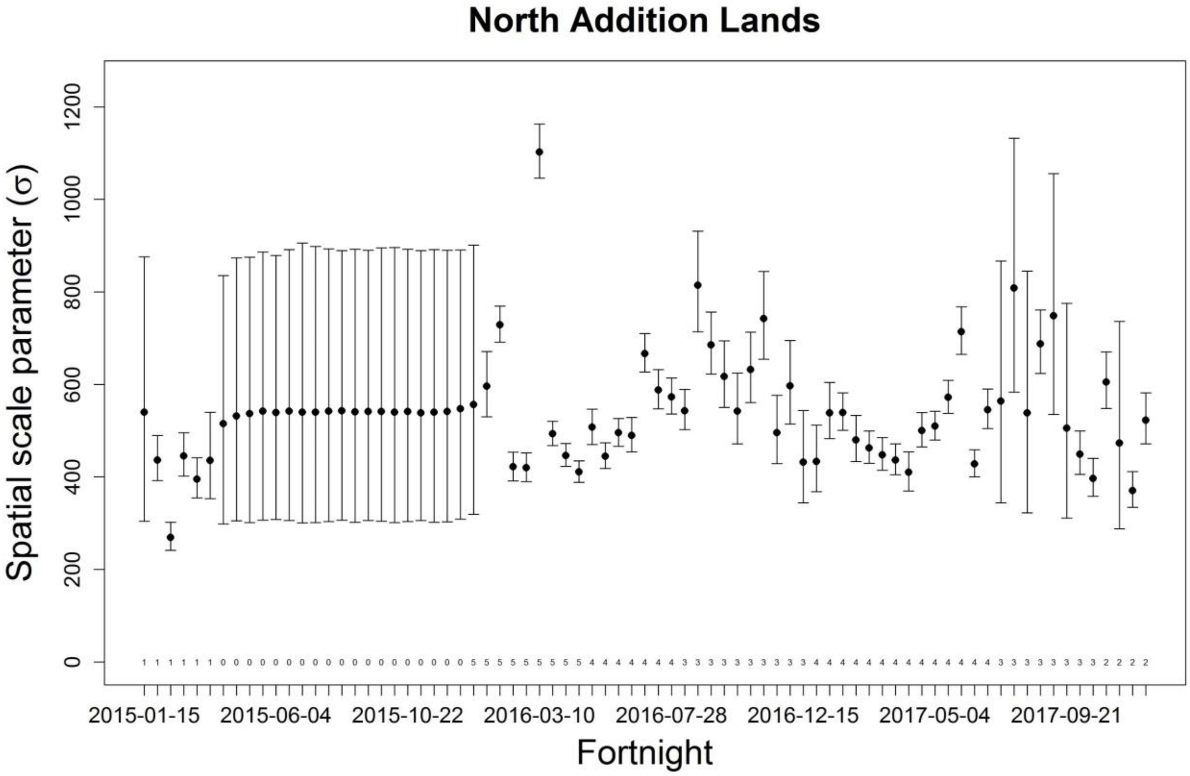


Figure G7. Biweekly estimates and 95% credible intervals of the spatial scale parameter (σ) for GPS-collared male deer on North Addition Lands. Sample size for each fortnight is indicated along the bottom.

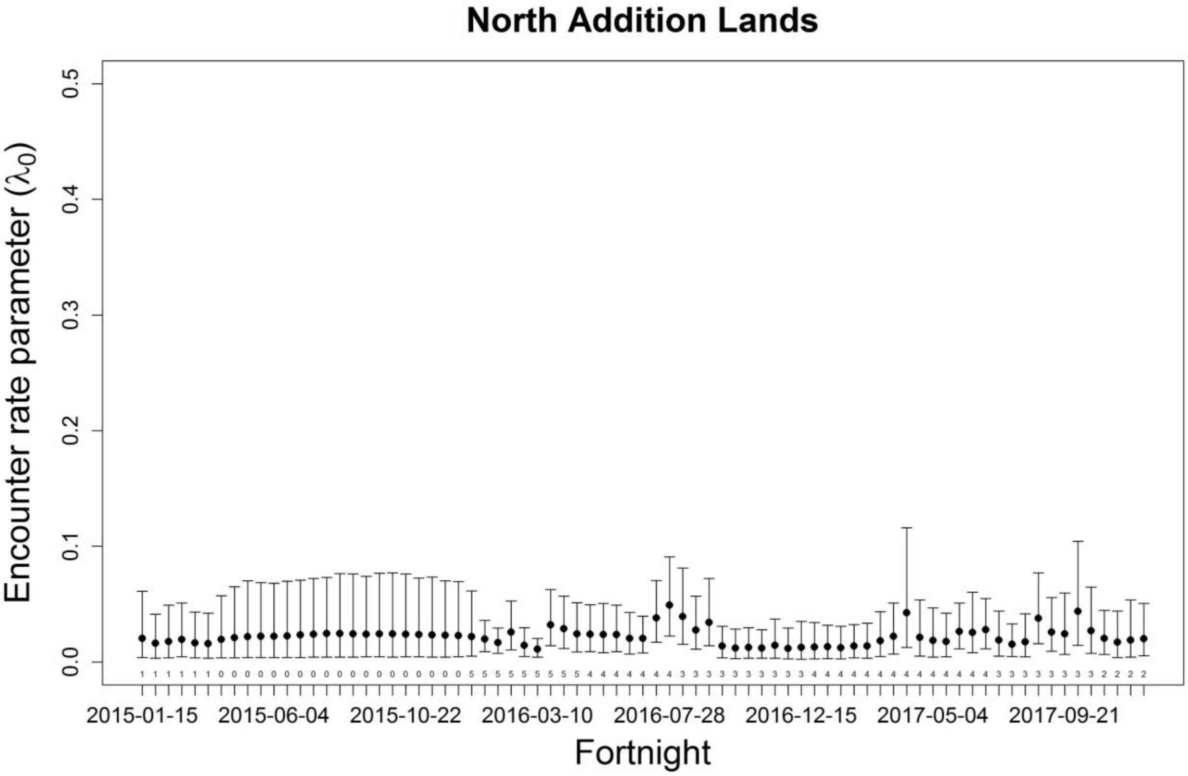


Figure G8. Biweekly estimates and 95% credible intervals of the baseline encounter rate parameter (λ_0) for GPS-collared male deer on North Addition Lands. Sample size for each fortnight is indicated along the bottom.

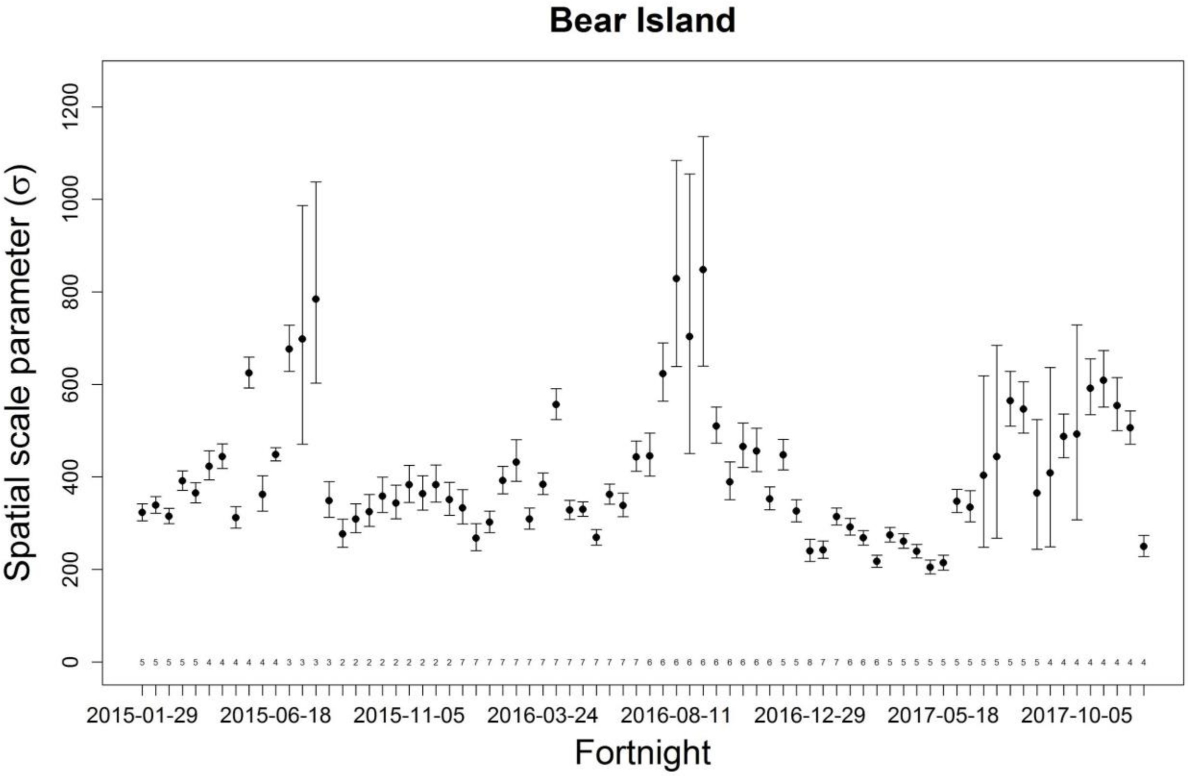


Figure G9. Biweekly estimates and 95% credible intervals of the spatial scale parameter (σ) for GPS-collared male deer on Bear Island. Sample size for each fortnight is indicated along the bottom.

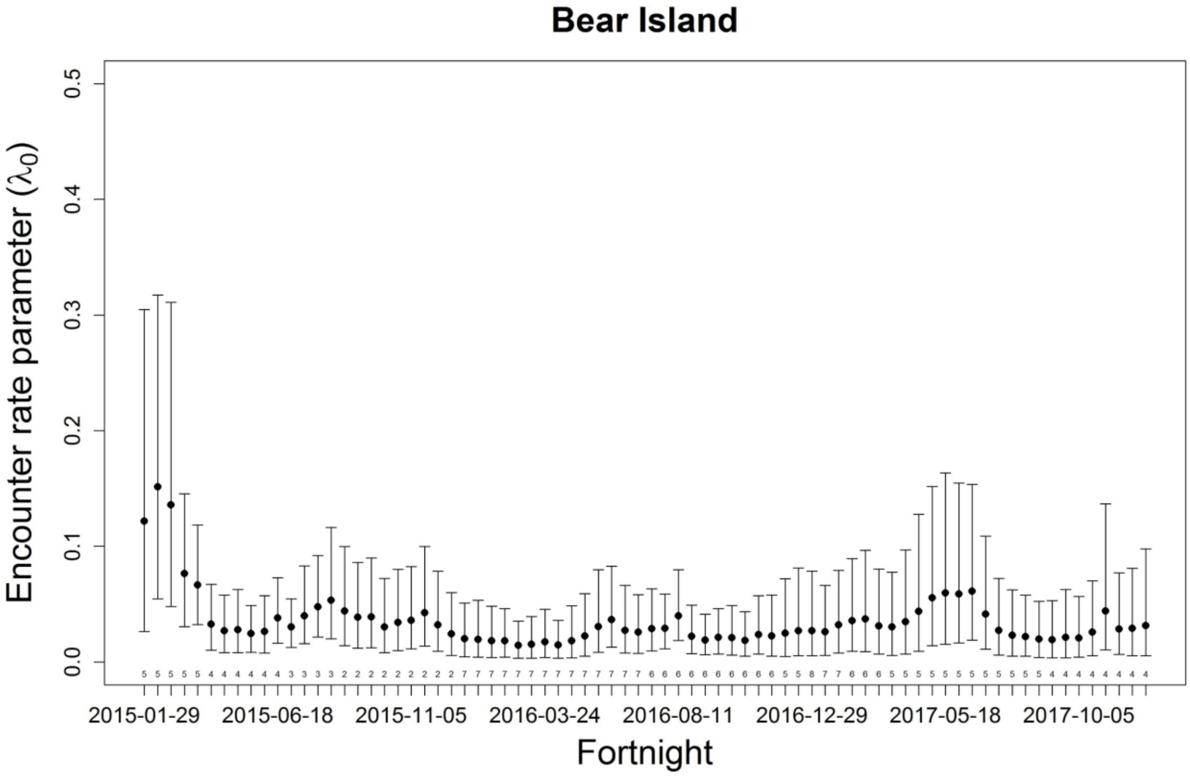


Figure G10. Biweekly estimates and 95% credible intervals of the baseline encounter rate parameter (λ_0) for GPS-collared male deer on Bear Island. Sample size for each fortnight is indicated along the bottom.

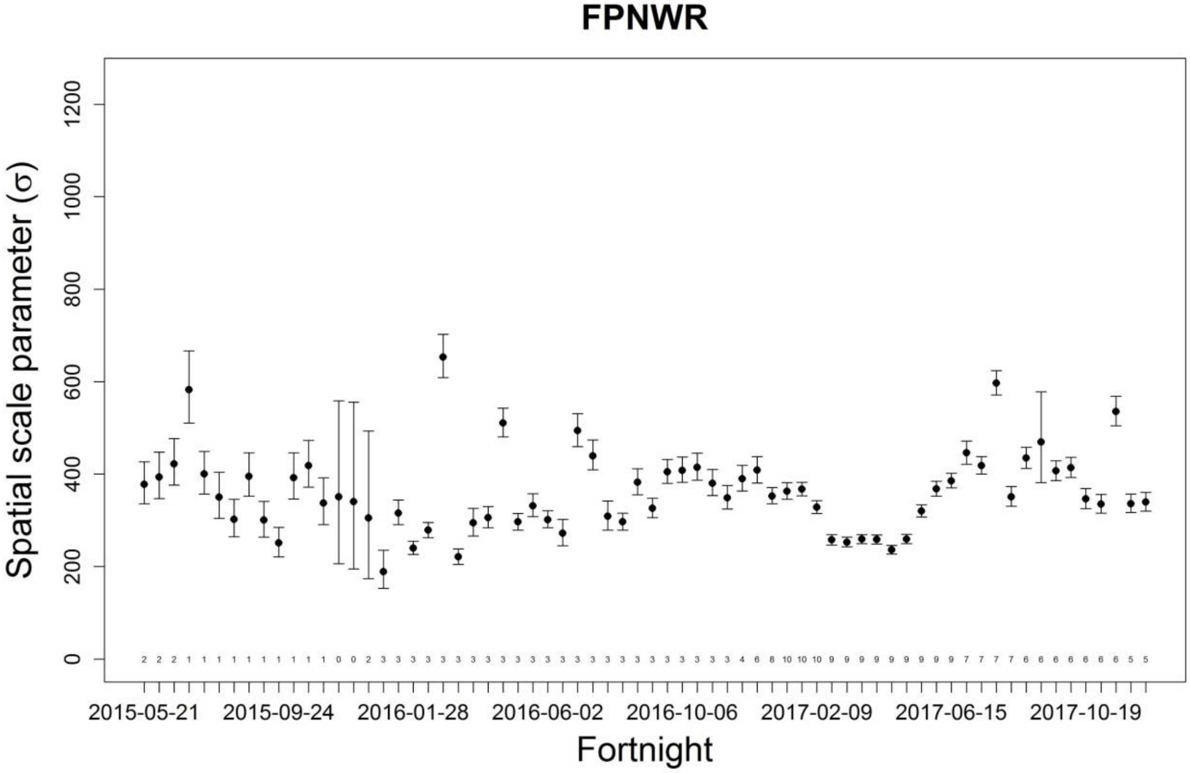


Figure G11. Biweekly estimates and 95% credible intervals of the spatial scale parameter (σ) for GPS-collared male deer on Florida Panther National Wildlife Refuge (FPNWR). Sample size for each fortnight is indicated along the bottom.

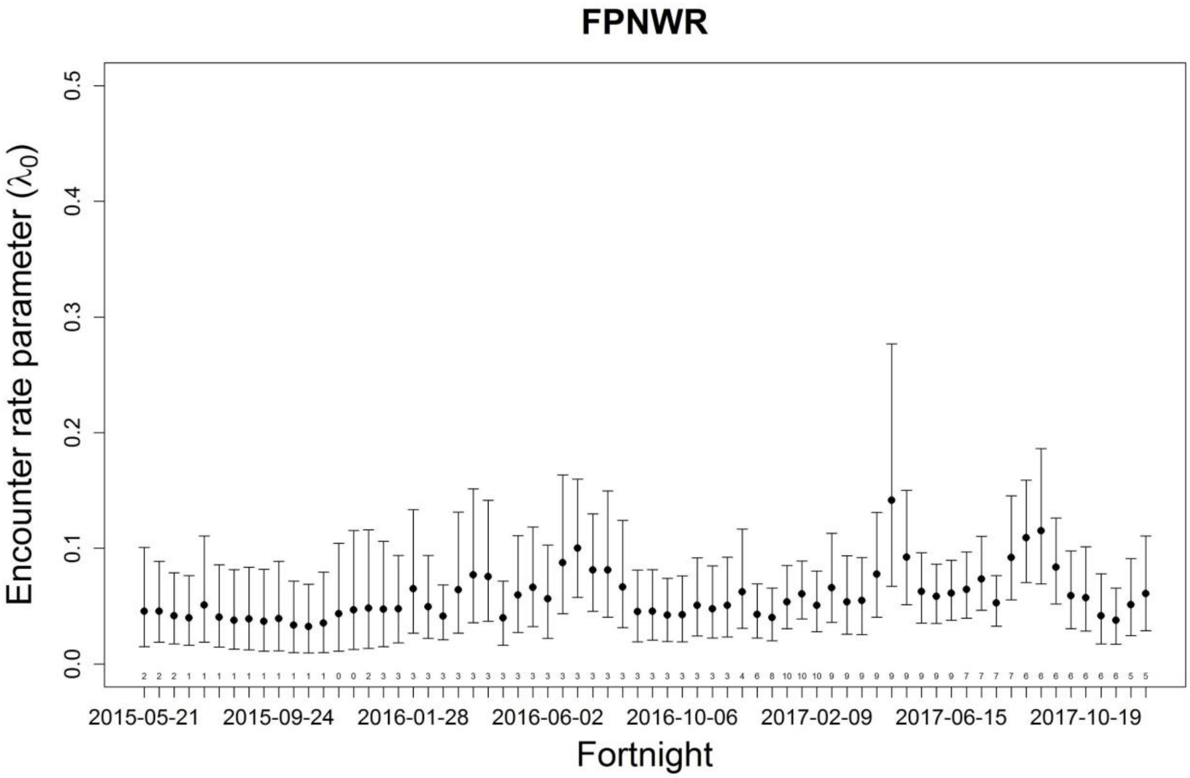


Figure G12. Biweekly estimates and 95% credible intervals of the baseline encounter rate parameter (λ_0) for GPS-collared male deer on Florida Panther National Wildlife Refuge (FPNWR). Sample size for each fortnight is indicated along the bottom.

Appendix H. Maps of Trail Camera Designs

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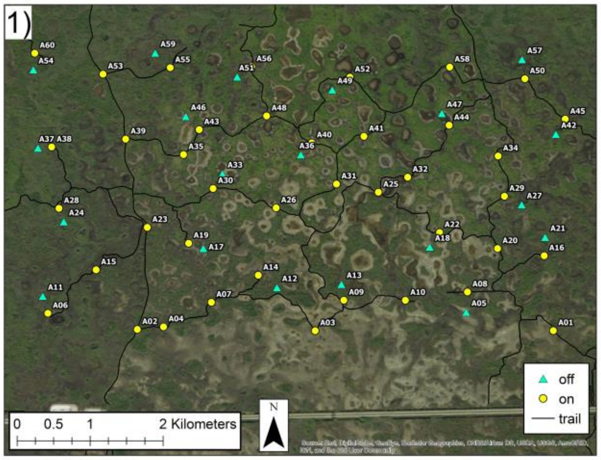

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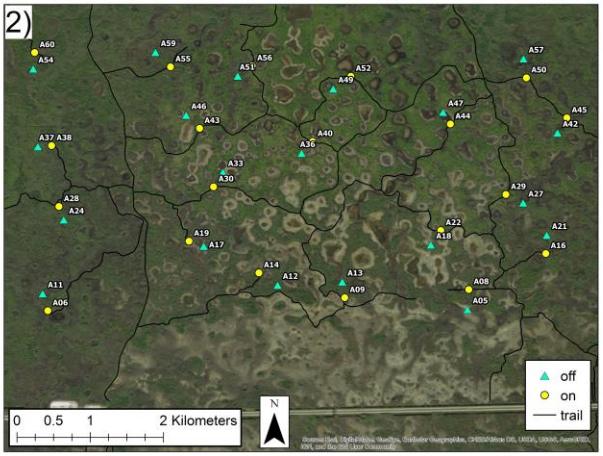

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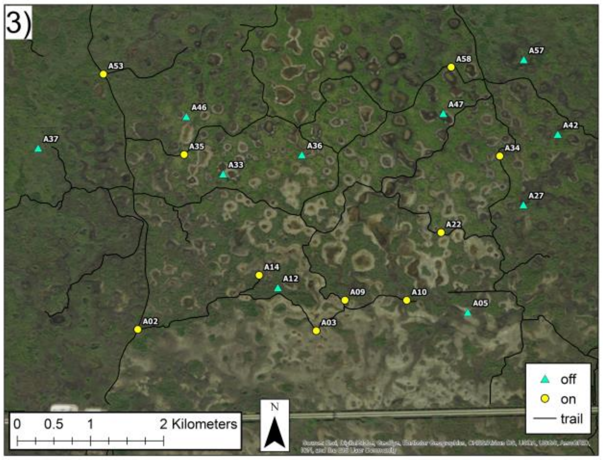

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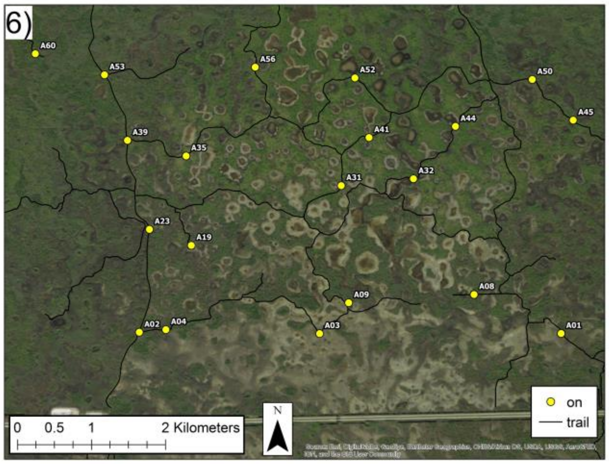

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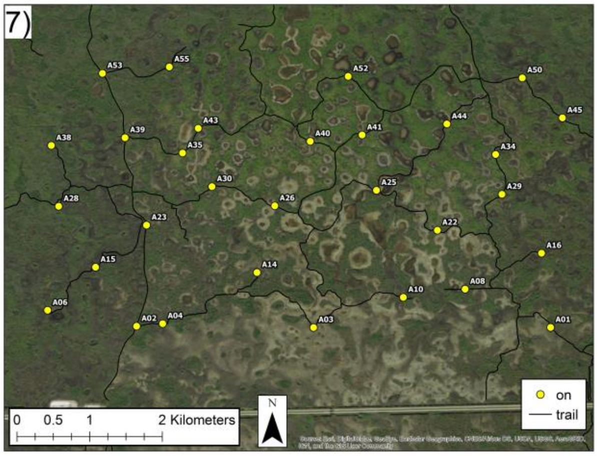

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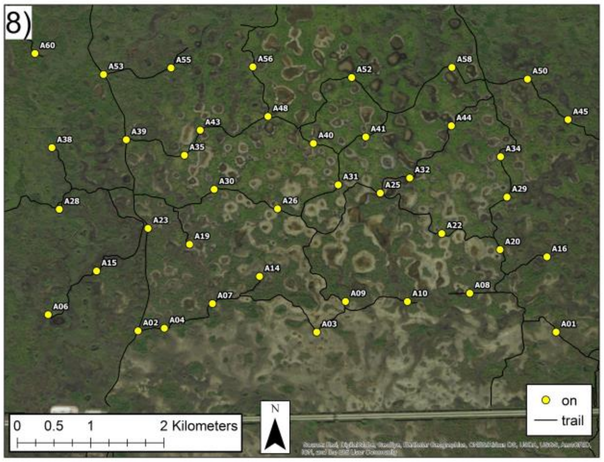


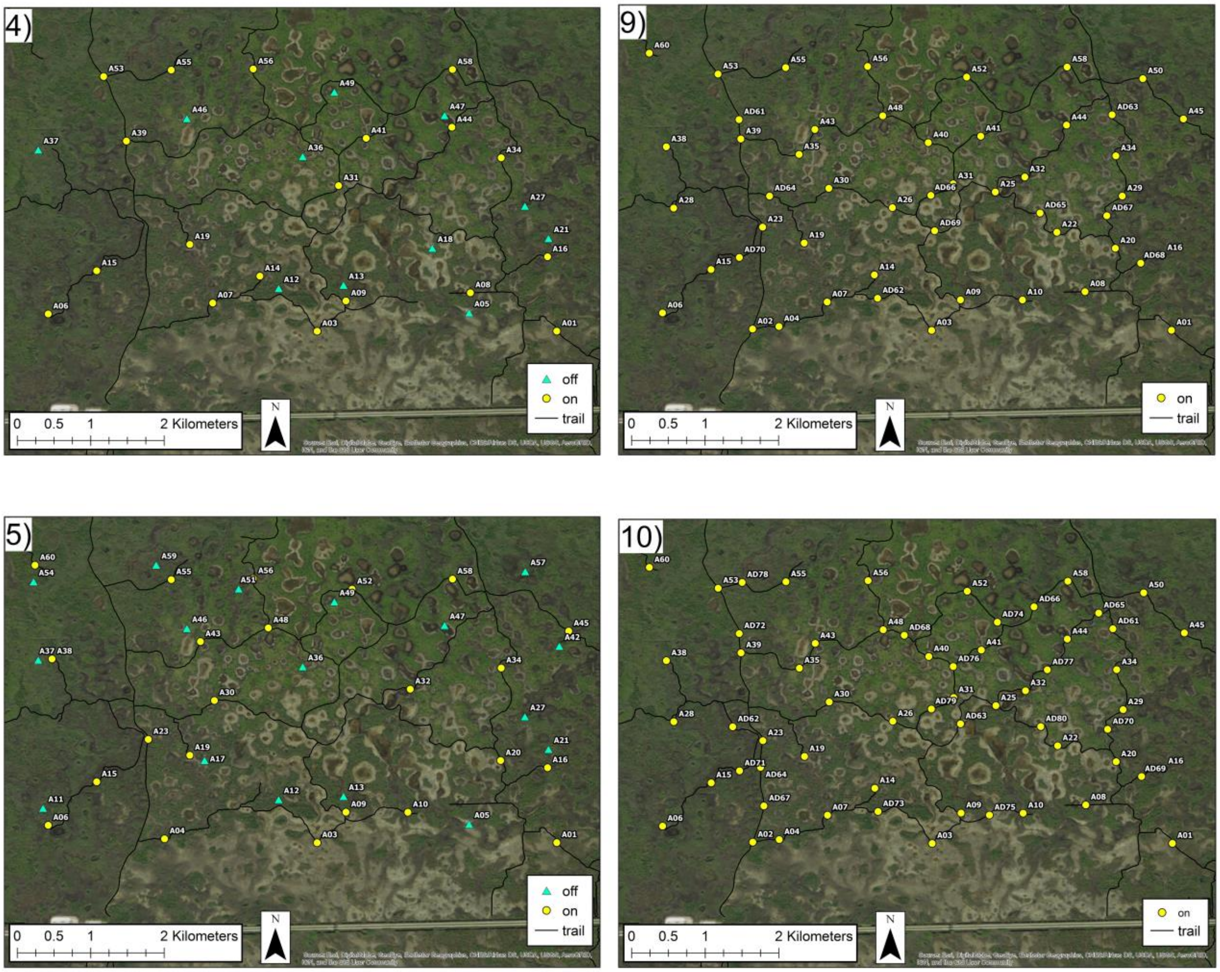
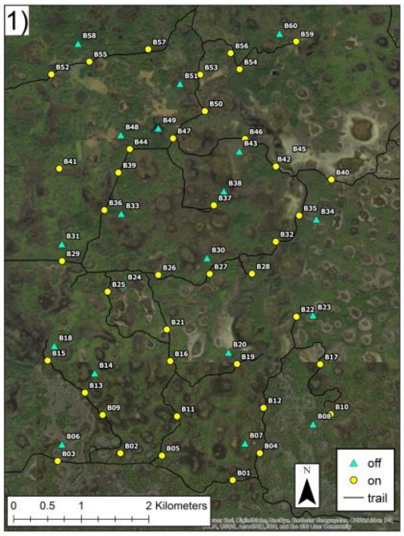


Figure H1. Location of the cameras on North Addition Lands for the camera design simulation study. Left column - 1) Status quo: 40 on-trail and 20 off-trail cameras, 2) 40 paired on- and off-trail cameras, 3) 20 random cameras, 4) 30 random cameras, 5) 40 random cameras. Right column - 6) 20 on-trail cameras, 7) 30 on-trail cameras, 8) 40 on-trail cameras, 9) 50 on-trail cameras, 10) 60 on-trail cameras.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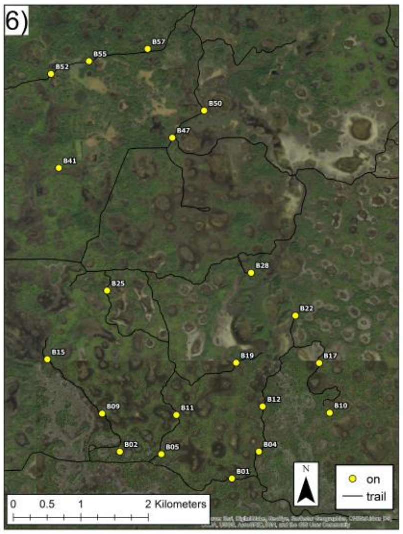
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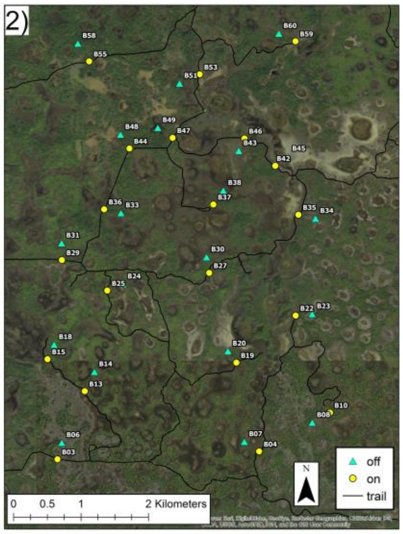
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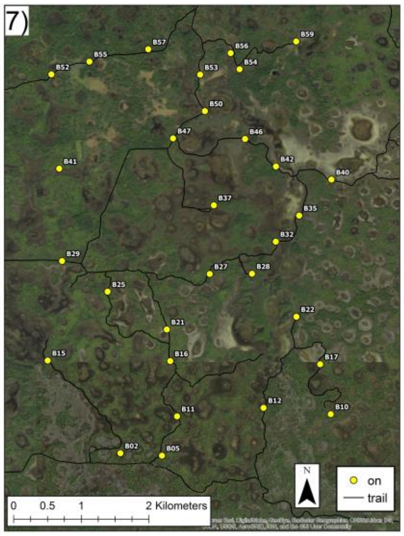
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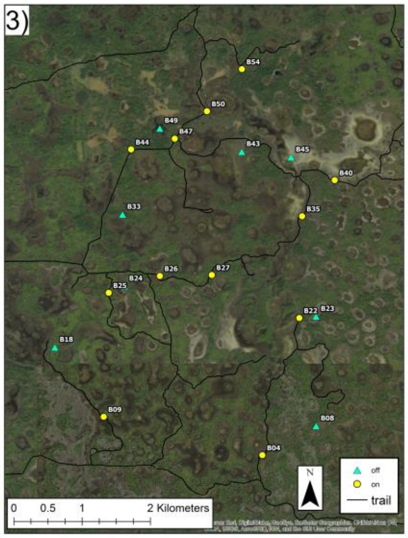
on
trail
0 0.5 1 2 Kilometers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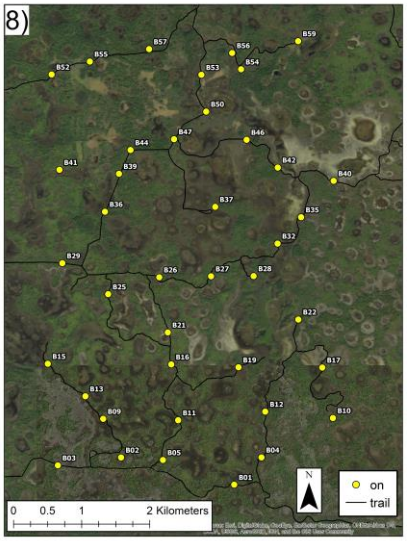
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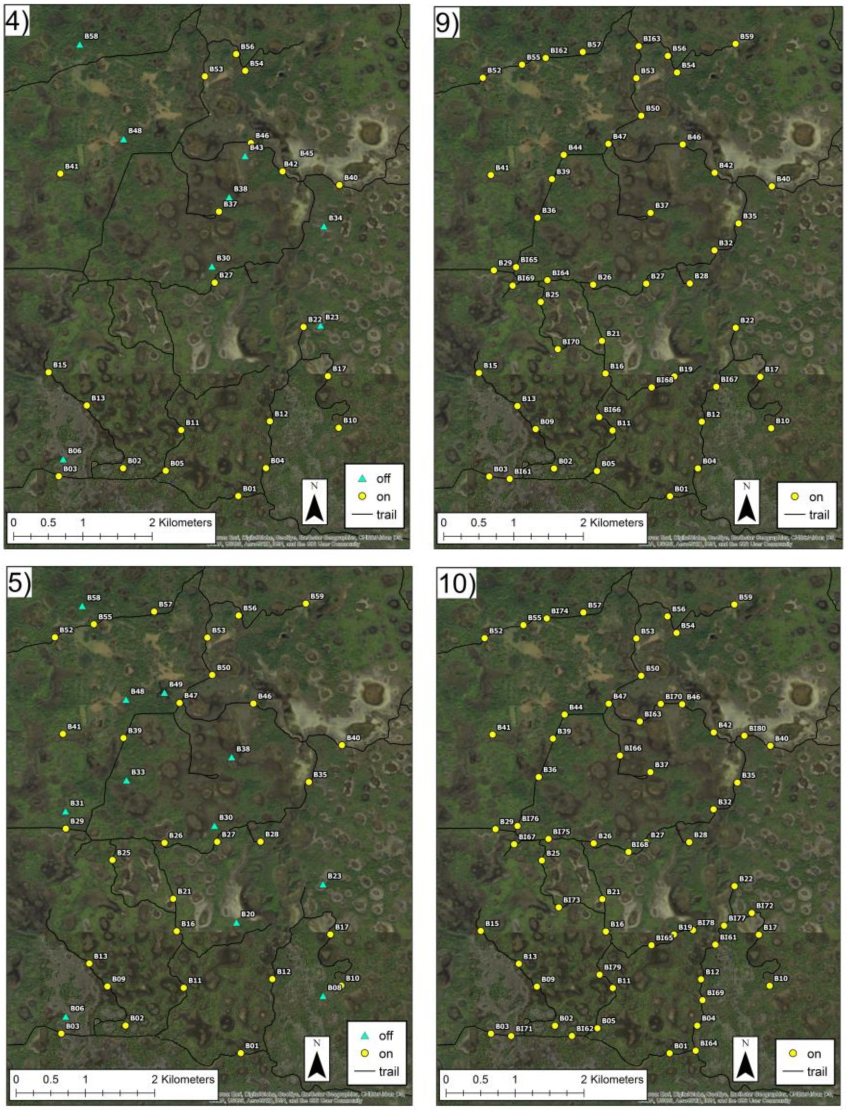
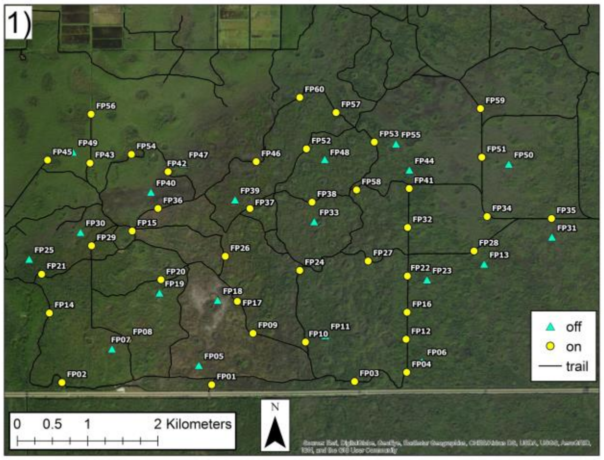


Figure H2. Location of the cameras on Bear Island for the camera design simulation study. Left column - 1) Status quo: 40 on-trail and 20 off-trail cameras, 2) 40 paired on- and off-trail cameras, 3) 20 random cameras, 4) 30 random cameras, 5) 40 random cameras. Right column - 6) 20 on-trail cameras, 7) 30 on-trail cameras, 8) 40 on-trail cameras, 9) 50 on-trail cameras, 10) 60 on-trail cameras.

1)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on
trail
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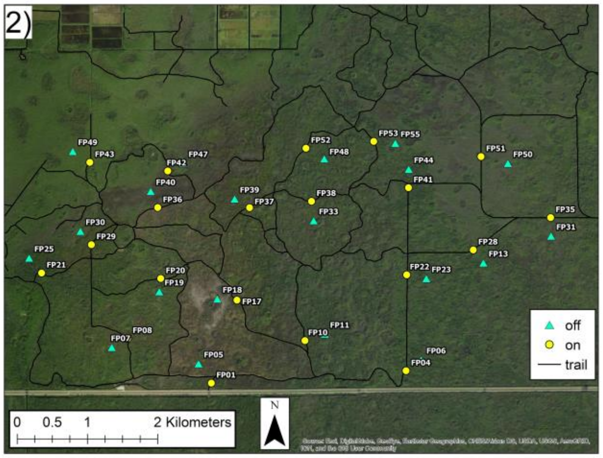
0.5
1
2 Kilometers
N

2)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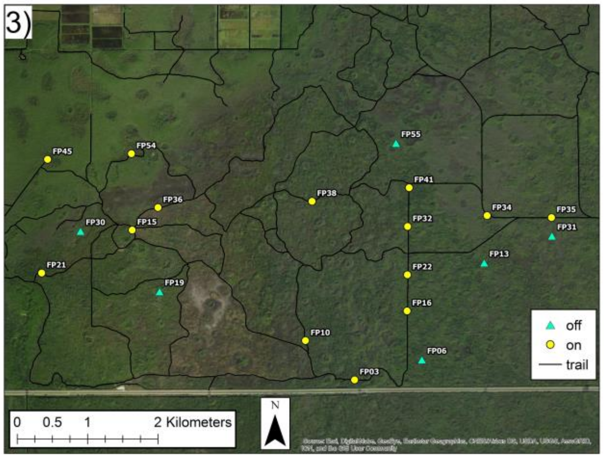
3)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2 Kilometers
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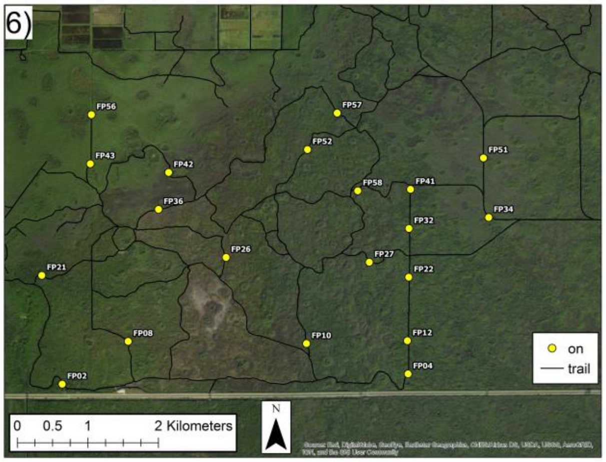
6)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2 Kilometers
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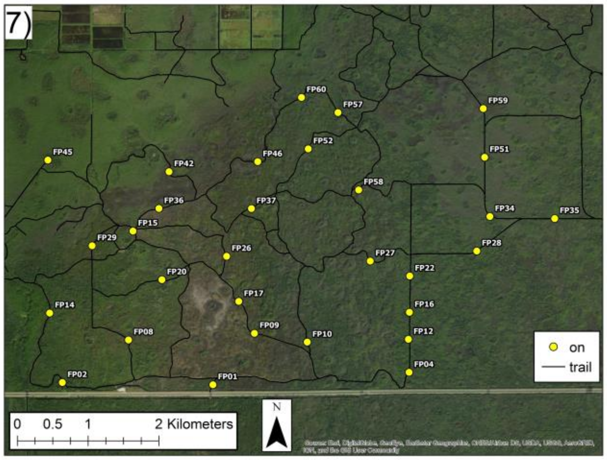
7)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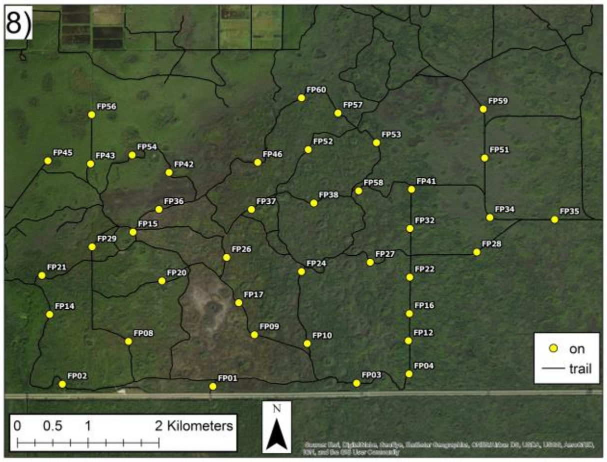
8)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trail
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2 Kilometers
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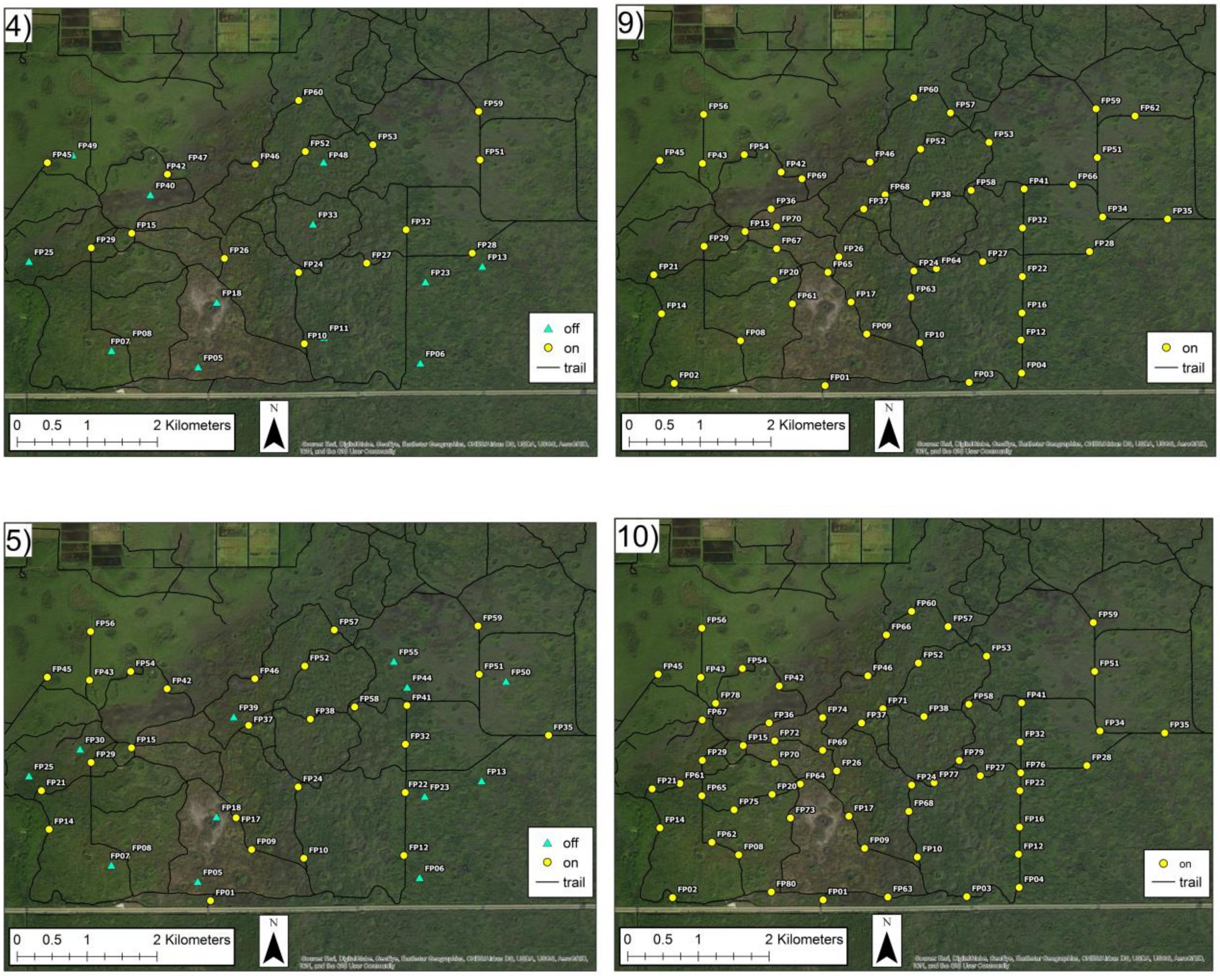

Figure H3. Location of the cameras on Florida Panther National Wildlife Refuge for the camera design simulation study. Left column - 1) Status quo: 40 on-trail and 20 off-trail cameras, 2) 40 paired on- and off-trail cameras, 3) 20 random cameras, 4) 30 random cameras, 5) 40 random cameras. Right column - 6) 20 on-trail cameras, 7) 30 on-trail cameras, 8) 40 on-trail cameras, 9) 50 on-trail cameras, 10) 60 on-trail cameras.