SEEING SPOTS IN THE SWAMP: USING CAMERA DATA TO ESTIMATE WHITE-TAILED DEER FAWN SURVIVAL AND RECRUITMENT IN SOUTH FLORIDA

by

KRISTIN NICOLE ENGBRETSEN

(Under the Direction of

Richard B. Chandler and Karl V. Miller)

ABSTRACT

In South Florida, white-tailed deer (Odocoileus virginianus) are an important game species and primary prey of the endangered Florida panther (Puma concolor coryi). Due to a potential decline in South Florida deer populations, updated information is needed about fawn survival and recruitment. During the 2016 fawning season, I deployed 180 passive cameras in South Florida, identified 123 individual fawns, and used a novel spatial capture-recapture model to estimate number of fawns born and recruited. An estimated 211 (95% CI 180-250) fawns were born in the 10,941-ha study area, and the density of birth locations was greatest in cypress forests and in areas with more frequent fire. Only 37 (95% CI 27-48) of 211 fawns survived to 180 days, indicating 18% of fawns reached sexual maturity. However, 2016 was an extraordinarily wet year and multi-year studies are needed to determine if recruitment is high enough to offset mortality in this population.

INDEX WORDS: camera trap, fawn recruitment, Florida, neonate, Odocoileus virginianus, productivity, SCR, spatial capture-recapture, survival, white-tailed deer
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DEDICATION

This thesis is dedicated to my four wonderful grandparents, Mr. Toley and Helen Engebretsen and Rev. Roscoe and Loretta Wilson, who have been a constant source of love and encouragement in my life. Thank you each for teaching me how to live and love by your exemplary lives, for encouraging me to take risks and work hard to achieve my goals, and for supporting every dream, hobby, or activity I have pursued since I was little. I have been so blessed to have these four incredible and strong people in my life.
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CHAPTER 1
INTRODUCTION AND LITERATURE REVIEW

INTRODUCTION

Information about spatial and temporal variation in density is needed to effectively manage wildlife populations, whether for ensuring sustainable harvest of game species or encouraging population growth for species of conservation concern. Understanding the factors influencing trends in abundance and density requires reliable estimates of vital rates, including mortality and recruitment. Recruitment, when juveniles survive to reproductive maturity, depends on both fecundity, the per-capita rate that females produce young, and juvenile survival. It is often assumed that population-level recruitment rates of many temperate ungulates, such as white-tailed deer (*Odocoileus virginianus*; hereafter, deer), are relatively stable across years (McCullough 1987). However, research shows that recruitment rates may vary over time due to changes in habitat quality (Tollefson et al. 2011), introduction of novel predators (Gulsby et al. 2015, Chitwood et al. 2015), availability of seasonal mast (e.g., *Quercus* spp. acorns; Wentworth et al. 1992) and changes in weather patterns (Ginnett and Young 2000, Ciuti et al. 2015). Therefore, reliable information about local recruitment rates is needed to effectively manage ungulate populations.

The white-tailed deer is the most culturally and economically important game species in North America and has been studied extensively in a variety of habitats. In South Florida, deer are important for recreational hunting and as the primary prey of the Florida panther (*Puma concolor coryi*; Maehr et al. 1990), which has been the subject of extensive conservation effort
Ensuring a sustainable deer population is critical to the continued recovery of the Florida panther.

It can be difficult to estimate juvenile survival and recruitment rates directly, and a variety of methods have been developed for inference, such as collaring neonates and monitoring fawn to doe (fawn:doe) ratios. However, many of these traditional methods are limited by logistical challenges or unreasonable assumptions. A novel open-population spatial capture-recapture (SCR) method has been recently developed that appears to hold promise in estimating recruitment rate while avoiding the shortcoming of other common methodologies (Chandler et al., in review).

LITERATURE REVIEW
Methodology

Estimating juvenile survival and cause-specific mortality rates often involves deploying VHF-telemetry collars on a sample of neonates and monitoring them daily to a known age or until collar failure (Steigers and Flinders 1980, Diefenbach et al. 2003). Many studies of white-tailed deer fawns have used grid-searches for fawns during the peak fawning season (Ballard et al. 1999) or observations of doe maternal behavior (Nelson and Woolf 1987) to detect parturition events and capture and collar neonate fawns. More recent studies rely on capturing pregnant females and equipping them with a Vaginal Implant Transmitter (VIT), which allows for capture of the neonate within a few hours of birth and often at the birth site, thereby providing more information about neonate survival in the first few days of life when the mortality rate can be highest (Carstensen et al. 2009, Jackson and Ditchkoff 2013, Gilbert et al. 2014, Shuman et al. 2017). However, due to the initial adult capture, subsequent neonate capture, and months of
intensive monitoring of all collared individuals, neonate capture studies can be invasive and costly. Telemetry studies often require multiple teams of researchers to monitor VITs and collars several times a day for at least 12-16 weeks after the last fawn birth (Carstensen et al. 2009, Shuman et al. 2017). Because of the intensive labor and financial resource requirements, the implications of these studies may be limited by the spatial extent that can be sampled or by the number of individuals that can be captured and monitored (Jackson and Ditchkoff 2013, Watine 2015).

Although collaring neonates has disadvantages regarding sample size limitations and resource requirements, it does provide precise information about survival for the collared sample. If collared animals are monitored until the age of recruitment (typically 6 months for white-tailed deer studies), this method can also provide a precise number of fawns recruited in the sample (Jackson and Ditchkoff 2013). However, small sample sizes and the potential for capture-induced bias in survival may make extrapolation to the rest of the population tenuous. Additionally, most neonate survival studies conclude around 12-16 weeks and do not continue following juveniles to the recruitment age (Carstensen et al. 2009, Shuman et al. 2017).

Many methods have been developed to estimate recruitment rate that utilize fawn:doe ratios attainable through population surveys. One of the most commonly-used methods to estimate deer demographics and recruitment rate is the baited camera survey. This technique consists of identifying individual bucks that visit baited camera traps during a given survey period (Jacobsen et al. 1997). The ratio of buck pictures to doe and fawn pictures can be used to calculate the number of does and fawns in the population (Jacobsen et al. 1997, Koerth and Kroll 2000). Males often dominate bait sites, which can create a bias in detection probability among bucks, does, and juveniles (McCoy et al. 2011). Nevertheless, this method has been used to index
recruitment during fall surveys, with the assumption that surviving fawns will always be documented at the camera with their dam. However, the age of the fawn also affects the likelihood that the fawn will travel and forage with the dam (Chitwood et al. 2017). For example, when fawns are young, they will likely be infrequently detected. Once mobile, fawns may heavily use feeders, causing an overestimation of fawns (McCoy et al. 2011). These biased counts can result in underestimates or overestimates of fawn recruitment, which suggests that baited surveys should only be used to index the trends in recruitment through time, rather than provide a point estimate of recruitment (Chitwood et al. 2017).

Another method used to estimate fawn recruitment is population reconstruction, such as the sex-age-kill model, which uses demographic data from harvested individuals to recreate the sex and age structure of the population in prior years (Roseberry and Woolf 1991, Skalski and Millspaugh et al. 2002). However, this model assumes population closure, is affected by changes in hunting pressure, and can be biased by hunter selectivity or population growth (Millspaugh et al. 2009). Additionally, population reconstruction cannot provide time-sensitive estimates of recruitment, since individuals must grow old enough to be harvested before they provide data for the reconstruction model.

Distance sampling utilizes replicated surveys of either line transects, point counts, aerial transects, or spotlight routes to count individuals and calculate population size and demographic parameters, including recruitment (DeYoung 1985, Stoll et al. 1991, Schwarz and Seber 1999, LaRue et al. 2007, DeYoung 2011, Anderson et al. 2013). Spotlight counts and roadside surveys have been frequently used by state and federal agencies, but research suggests that they can be ineffective and biased by highly variable detection rates (McCullough 1982, Roberts et al. 2006, LaRue et al. 2007, Collier et al. 2013). Aerial surveys are biased by the assumption of equal
detectability of all demographic groups, as well as limitations to detection in certain cover types
(Caughley 1975, DeYoung 1985, LaRue et al. 2007). Aerial methods are also expensive due to
operating costs of aircraft and the necessity to conduct multiple flights for replication (Potvin et
al. 2004, LaRue et al. 2007). Therefore, while a variety of distance sampling methods have been
used to estimate herd demographics and recruitment, they are limited in their ability to provide
accurate, dependable, and cost-effective estimations of population size or recruitment for making
management decisions. Although each of these traditional methods (baited camera surveys,
population reconstruction, and distance sampling) can be used to estimate population size,
density, and survival, and then infer recruitment, they can be prohibitively expensive to carry out
and do not yield robust and reliable estimates of the number of fawns recruited.

Capture-recapture methods have been used to quantify population demographics in a
variety of species (Schwarz and Seber 1999). The suite of capture-recapture methods includes
both physical capture-mark-recapture (CMR) of individuals (Eberhardt 1969, Guthery 1980,
Grimm et al. 2014) and mark-resight methods, where resighting can occur through harvest,
visual surveys, or camera detections (White and Bartmann 1983, Minta and Mangel 1989,
Martorello et al. 2001, Roberts et al. 2006). Studies of elusive or rare species have utilized CMR
methods that lack physical capture of individuals, but instead depend entirely on non-invasive
captures. These non-invasive techniques can use genetic ‘capture’ of individuals through hair
snares, fecal collection, and other genetic matter (Waits and Paetkau 2005). Alternately, study
designs that use passive camera traps to initially capture and then recapture animals have been
developed for a variety of species, especially those can be individually identified through body
morphology or unique marks (Karanth 1995, Trolle and Kery 2003, Silver et al. 2004, Weingartha
et al. 2015). These CMR methods can provide information about population size, but they are
limited in their ability to estimate population density because they lack explicit information about the extent of the area sampled.

A rapidly developing class of new CMR models makes use of the spatial information at each capture, allowing researchers to identify the effective area sampled and account for individual variation in capture probability. These spatially-explicit capture-recapture (SCR) methods (Efford 2004, Borchers and Efford 2008, Royle and Young 2008, Royle et al. 2013) use the same data collection process as traditional CMR methods, but allow inference about spatial and temporal variation in density as well. These models use the locations where individuals were detected to estimate the distribution of home range centers (also referred to as activity centers in the SCR literature) throughout the area of interest. Capture probability is modeled as a decreasing function of the distance between an individual’s activity center and a trap or detector. SCR methods have been used with uniquely identifiable species through a grid of cameras or other detectors (Thornton and Pekins 2015, Weingarth et al. 2015). However, these methods typically assume demographic closure, so they cannot be used to estimate recruitment. Recent developments of open-population SCR models have provided methods to estimate survival and recruitment (Gardner et al. 2010, Ergon and Gardner 2014, Chandler and Clark 2014). Although these models allow more robust estimates of density and other demographic parameters, they do not incorporate data on age-specific survival and detection rates, which can vary dramatically when focusing on juvenile individuals. Additionally, these open-population SCR models do not allow exploration of spatial and temporal variation in birth rate or juvenile survival, which are two critical components in estimating juvenile recruitment.

An open-population SCR method focusing on estimating juvenile recruitment through passive detection data has recently been developed to address these concerns (Chandler et al., in
review). This model allows age-dependent detection and mortality probabilities to be estimated, avoiding the bias in ignoring age-related variation. This model additionally allows the variation in the underlying juvenile survival and birth processes to be examined. This method was first demonstrated with a small dataset of passive camera photos on white-tailed deer, but it has not been applied to assess hypotheses about the factors influencing survival and recruitment.

**Deer Recruitment in South Florida**

Due to low quality, mineral-deficient forage in South Florida, deer body size, antler mass and productivity are lower than in many other populations in temperate North America (Harlow and Jones 1965, Harlow 1972, Garrison and Gedir 2006, Jones et al. 2010). Annual breeding patterns throughout Florida are less constrained by seasonally harsh climatic variations. As such, breeding can occur in Florida during much of the year. For example, peak breeding throughout the state ranges from February in northern Florida to August in southern Florida (Richter and Labisky 1985), where fetal measurement data indicate that peak breeding occurs near 10 August (+/- 16 days; Richter and Labisky 1985). In South Florida, seasonal precipitation patterns result in summer flooding, and fawn drop is timed to match the dry season from January to March (Richter and Labisky 1985, Garrison and Gedir 2006). A VHF-telemetry study of neonates in South Florida’s Big Cypress National Preserve (BCNP) found the peak fawning date to be 22 February (Labisky et al. 1995). An independent dataset on breeding chronology using doe collections from the Everglades Wildlife Management Area estimated 19 February as the peak fawning date (Florida Fish and Wildlife Conservation Commission, unpublished data). However, fawning in this area has been documented throughout a wide range of dates, from November to March (Fleming et al. 1994). These estimates differ markedly from many other white-tailed deer
studies in the Southeast, where peak fawn drop occurs typically in a tight window but varies locally from May to July (Watine 2015, Shuman et al. 2017).

Productivity of white-tailed deer in South Florida is lower than in most other areas of the species’ range and may vary due to the quality of vegetation in a doe’s home range (Richter and Labisky 1985, McCown 1991, Labisky et al. 1995). Collections of South Florida deer indicate a pregnancy rate of 10% for fawns, 91% for 1.5 year olds, and 100% for 2.5 year olds and up (Richter and Labisky 1985). In a five-year study in the Bear Island Unit of BCNP, 93% of adult does were pregnant with an average of 1.3 fetuses (McCown 1991). Only 25% of females in an independent BCNP study produced twins (Labisky et al. 1995), although white-tailed does throughout the rest of their range frequently produce twins (Jones et al. 2010, Fortin et al. 2015, Green et al. 2017, Shuman et al. 2017). None of 380 sampled deer throughout the state of Florida produced triplets (Richter and Labisky 1985). South Florida’s fecundity rate of 1.18-1.26 fetuses/pregnant doe (Richter and Labisky 1985, Fleming et al. 1994) is on the low end of comparable white-tailed doe fecundity rates throughout their extensive range, where adult does may average 1.2 to 2.27 fetuses/pregnant doe (DeYoung 2011, Fortin et al. 2015, Green et al. 2017).

Estimates of South Florida fawn survival also vary locally and may be sensitive to seasonal hydrology and severe weather events. The 12-month fawn survival rate was 56% (N=34, 1987-1990) in the dry prairie of the eastern Everglades (Eisenberg and Sunquist 1994). A study conducted in the wet prairies of the Everglades National Park (ENP) and BCNP monitored collared fawns for 18 months. The mean survival of 80 fawns to 12 months was 23.3% in both sites over three years. However, the mean survival of 55 fawns (censored fawns excluded) to 18 months was only 15% (4 of 27 fawns) in BCNP and 4% (1 of 28 fawns) in ENP (Labisky et al.
1995). Additionally, only a few fawns less than a week old were included in this sample, leaving a gap in knowledge about survival probability of young neonates. Visual monitoring of collared does from a typical rainfall year to a significant flood year indicated a 10-fold decrease in apparent fawn production (MacDonald-Beyers and Labisky 2005), supporting Fleming et al.’s (1994) finding of a significant reduction in fawn recruitment when average standing water level rose over 45 cm in marshes. These results suggest that the differences in hydrological schemes among habitats and across years may greatly impact fawn survival (Labisky et al. 1995). These studies were also conducted before the Florida panther population began to increase in the mid-1990s due to genetic restoration (Pimm et al. 2006, FWC 2017).

The structure of the predator community likely influences fawn survival to recruitment in South Florida. Beyond being the primary prey for the Florida panther, South Florida deer are also important prey for alligators (*Alligator mississippiensis*), black bears (*Ursus americanus*), bobcats (*Lynx rufus*), and coyotes (*Canis latrans*) (Maehr and Brady 1984, Land 1991, Labisky and Boulay 1998, McCown and Scheick 2007). In many deer populations, coyotes have been shown to significantly reduce survival and recruitment of deer fawns (Cook et al. 1971, Kilgo et al. 2014, Nelson et al. 2015, Chitwood et al. 2015). However, coyotes are infrequent in BCNP, only reaching South Florida within the past few decades (McCown and Scheick 2007). Most coyote detections or sightings in BCNP are of young dispersing individuals, and there is no known documented coyote reproduction or denning in BCNP (Shindle, D., US Fish and Wildlife Service, personal communication). Additionally, coyote diet studies in Florida indicate a low prevalence of white-tailed deer relative to smaller prey items (Thornton et al. 2004, Watine 2015). Therefore, coyotes are unlikely to be a large factor in survival of South Florida fawns. In contrast, black bears are common in South Florida, with a density of 0.132 bears/km² in BCNP.
Bears are opportunistic and effective predators of white-tailed deer fawns within their first few weeks of life (Shuman et al. 2017). However, South Florida bears are largely dormant in early spring. Bear activity does not increase until the beginning of April, when fawns are older and have become highly mobile (unpublished data, this study). Additionally, bear predation on deer in South Florida also appears to constitute only a small and opportunistic portion of their diet (Maehr and Brady 1984). Bobcats are frequent in South Florida and can heavily affect fawn survival in many areas (Land 1991, Labisky et. al 1995, Nelson et al. 2015, Shuman et al. 2017), as well as predating adult deer (Land 1991, MacDonald-Beyers and Labisky 2005). In one study, 31 of 53 fawn deaths (58%) were attributed to bobcat mortality in ENP and BCNP, which peaked in June and July when fawns were 3-5 months old (Labisky et. al 1995, Labisky and Boulay 1998). However, there is little information about current bobcat density in BCNP. Due to intensive conservation efforts, Florida panther populations have recovered from 20-30 individuals in the early 1990’s to 120-230 individuals in the most recent estimates (Pimm et al. 2006, FWC 2017). There is little known about the potential impacts of this panther recovery on bobcat populations or bobcat prey selection (Eisenberg and Sunquist 1994). Panthers are very effective ambush predators in depredating adult deer, and they often predate fawns as well (Maehr et al. 1990, Eisenberg and Sunquist 1994, Dalrymple and Bass 1996). Additionally, both American alligators and invasive Burmese pythons (*Python bivitattus*) have been known to predate adult deer and fawns (Epstein et al. 1983, MacDonald-Beyers and Labisky 2005, Boback et al. 2016) and may additionally impact fawn survival, especially during the wet season.

There have been no recent estimates of fawn recruitment in South Florida, apart from FWC’s annual recruitment aerial surveys performed over the Everglades Wildlife Management
Area and surrounding state lands. These surveys provided an estimated 35.44% recruitment rate (ranging from 17%-46.33%) from 2007-2017 by using the fawn:doe ratios observed during the aerial surveys (Ward, M. R. Florida Fish and Wildlife Conservation Commission, unpublished data). However, aerial surveys are known to be biased by detectability (Caughley 1975), and these surveys are primarily used to track trends in fawn recruitment and population size instead of providing a precise estimate.

**STUDY SITE**

The Big Cypress Basin (BCB) of South Florida lies northwest of the large, low-lying Everglades Swamp, which covers most of the southern tip of Florida. The BCB is generally higher in elevation than the Everglades, with a thin layer of marl or sand characterizing the soil, which gives the Big Cypress Basin more distinct topography and vegetation types than in the Everglades (McPherson 1974). The BCB is characterized as a tropical savannah, which has clear wet and dry seasons (Hela 1952). Mean annual rainfall is 53 inches per year, with 60-80% of the rain falling between May and September (McPherson 1974, Duever et al. 1986). During this wet season, the shallow aquifers fill and sheets of water flow southwest over much of Big Cypress National Preserve (BCNP). From October to April, the BCB gets relatively little rainfall and much of the seasonally-inundated land dries. The mean annual air temperature is 23°C, and mean monthly temperatures range from 14°C in January to 28°C in August (Duever et al. 1986).

The dominant vegetation types of the BCB are determined by topography, hydroperiod, and soil type. Pine flatwood forests tend to have an open overstory and absent or sparse mid-story. The canopy is dominated by slash pine (*Pinus elliotti*) and cabbage palm (*Sabal palmetto*), and the groundcover includes saw palmetto (*Serenoa repens*) and grasses from the *Panicum*,
Aristida, and Andropogon genera. Pine flatwoods are maintained by frequent fire, which allows diverse plant communities to flourish and reduces hardwood abundance (Loveless 1959, McPherson 1974, Duever et al. 1986). Cypress forests and mixed swamps are characterized by bald and pond cypress trees (*Taxodium distichum* and *T. ascendens*), and they also often include pond apple (*Annona glabra*) or pop ash (*Fraxinus caroliniana*) sloughs and willow heads (*Salix spp.*). Small shrubs and hardwoods trees, such as wax myrtle (*Morella cerifera*), are common in the sparse understory of cypress forests. Mixed swamps are more densely forested by cypress, willow, pop ash, and pond apple in the deeper areas, with cabbage palm, red maple (*Acer rubrum*), wax myrtle, and bays (*Persea spp.* and *Magnolia spp.*) distributed throughout. Due to the thick vegetation and frequent to year-round water inundation, wildfires rarely burn through the mixed swamp (McPherson 1974, Duever et al. 1986). Hardwood hammocks are elevated slightly more than the swamps, marshes, and prairies, which allows hardwood species such as cabbage palm, red maple, laurel oak (*Quercus laurifolia*), water oak (*Q. nigra*) and live oak (*Q. virginiana*) to dominate the dense overstory, which can be up to 20 m high. Fire is rare in these communities due to the understory of shrubs, vines, ferns, and epiphytes that retain high humidity, leading to a fire-resistant climax community (McPherson 1974, Duever et al. 1986). Marshes are characterized by sparse vegetation, peaty soils, low elevations, and are inundated for 225-275 days per year. The emergent wetland vegetation found in marshes typically consists of cattails (*Typha spp.*), sawgrass (*Cladium mariscus*), rushes (*Juncus spp.*), and alligator flag (*Thalia geniculata*) (McPherson 1974, Duever et al. 1986). Wet prairies have shorter hydroperiods than marshes and are dominated by relatively short mixed grasses and sedges with occasional dwarfed cypress trees (Duever et al. 1986). They often form the transition between marshes and cypress sloughs (Loveless 1959). Dry prairies are slightly higher in elevation than
wet prairies, seldom inundated, contain more grasses, sedges, and saw palmetto, and include some shrubs and small trees (McPherson 1974, Duever et al. 1986). Wetland scrub is a transitional vegetation type, which includes small cypress trees, willows, and other hardwood trees and shrubs less than 6 m high in poorly drained soil with frequent standing water. These areas occasionally have scattered pines or bays but no true overstory canopy (Kawula 2009).

OBJECTIVES

The objectives of this research were to estimate fawn survival and recruitment in South Florida to inform future management decisions in the region. To do this, I uniquely identified and monitored individual white-tailed deer fawns from first detection to 180 days of age using passive camera data. I then utilized a novel SCR model (Chandler et al., in review) to analyze the fawn encounter history data and to estimate fawn survival to recruitment at a broad spatiotemporal scale. I also evaluated multiple environmental factors, such as vegetation, fire history, water level, and human disturbance, to determine their relative influence on the spatial variation in density of fawn birth locations and fawn detection rate.

THESIS FORMAT

This thesis is presented in manuscript format. Chapter 1 presents a literature review of methodology and previous regional research pertinent to the focus of this thesis. Chapter 2 applies the novel SCR methodology to camera trap data on white-tailed deer fawns in South Florida. It includes results on spatial and temporal variation in fawn survival and recruitment, the effects of environmental variables on these rates, and estimates of landscape-scale survival to
recruitment. Chapter 3 discusses general conclusions of the research, management implications of the estimates, and future possibilities to utilize this methodology.

LITERATURE CITED


CHAPTER 2

SPATIAL AND TEMPORAL VARIATION IN WHITE-TAILED DEER FAWN SURVIVAL AND RECRUITMENT IN SOUTH FLORIDA

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ABSTRACT

In South Florida, the white-tailed deer (*Odocoileus virginianus*) is an economically and culturally important game species and the primary prey of the endangered Florida panther (*Puma concolor coryi*). Due to recent declines in deer populations in South Florida, more information is needed about the factors influencing fawn survival and recruitment. We deployed 180 cameras throughout the Florida Panther National Wildlife Refuge and the Big Cypress National Preserve. We visually identified 123 unique fawns in 5,199 fawn photos from 1 December 2015 to 30 June 2016 and created spatially-referenced capture histories for each fawn. We utilized a novel spatial capture-recapture (SCR) model to estimate the number of fawns born and the survival rate to the recruitment age of 180 days. The estimated peak fawning date was 25 January 2016. An estimated 211 (95% CI 180-250) fawns were born in the 10,941-ha study area, and the density of birth locations was greatest in cypress forests and in areas that frequently burned during the previous twenty-years. However, only 37 (95% CI 27-48) of the 211 fawns survived to 180 days. Fawn activity, measured by fawn detection rate, increased with water depth and decreased with higher levels of human activity and vehicle traffic. Our results suggest that a relatively low proportion of fawns (18%) reach sexual maturity in South Florida, but our study was conducted during an extremely wet spring, which was among the wettest dry seasons on record. Additional data are needed to determine if the observed survival to recruitment rate is representative of the system or demonstrates a worst-case scenario. Future research will reveal how such low survival to recruitment, whether abnormally low or representative of true recruitment in the area, might affect future viability of the deer population.
INTRODUCTION

The white-tailed deer (*Odocoileus virginianus*; hereafter, deer) is the most culturally and economically important game species in North America and has been studied extensively across its wide range (Conover 2011, Heffelfinger 2011). Information about the abundance and productivity of white-tailed deer populations is needed to inform management decisions and assess population viability. 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 research on white-tailed deer fawn survival has revealed a wide range of neonatal mortality rates, from 9.3% to 90% (DeYoung 2011), with variation primarily attributed to differing predation rates (Nelson et al. 2015, Shuman et al. 2017). However, numerous variables, including the quality and quantity of vegetation, availability of cover, previous fire history of an area, suite of predator species present, and human disturbance, can influence fawn production and fawn survival rate (Taber and Dasmann 1957, Nelson and Woolf 1987, Berger 2007, Tollefson et al. 2011, Shuman et al. 2017).

Several key environmental variables can impact the spatial variation in fawn birth locations and fawn survival rate. The vegetative composition selected during fawning and rearing provides safety to the doe and neonates during parturition and weaning through the availability of concealment cover, which often reduces predation risk to fawns (Nelson and Woolf 1987, Cherry et al. 2017, Grovenburg et al. 2012). However, the open sightlines present in
some vegetation types may act to reduce fawn predation risk and increase survival probability when ambush-style predators are present (Nelson and Woolf 1987, Kauffman et al. 2007, Cherry et al. 2018). The quality of forage in the habitat selected by the lactating doe and weaning fawn provides critical nutrition and directly impacts fawn growth, survival, and recruitment (Tollefson et al. 2011). The fire history of a landscape influences both the distribution and extent of vegetative communities (Loveless 1959, Bond and Keeley 2005) and the nutritional quality of forage available to pregnant does, which can increase doe nutritional state and affect fawn survival (Taber and Dasmann 1957, Harlow and Jones 1965, Stransky and Harlow 1981). Thus, fire on the landscape can result in patches of high-quality and desirable fawning and weaning habitat.

Human activity and disturbance in an area can also impact selection of birthing locations and fawn survival rates. Humans may act as a “shield” of protection for mammalian prey species, where human-averse predators reduce their hunting pressure in areas with frequent human activity. This allows parturient females to select areas with a perceived lower risk of predation in which to deliver their young (Berger 2007). This trade-off may result in prey species sacrificing higher-quality vegetation and cover by selecting for cover types with lower perceived predation risk to ensure their own and their offspring’s survival (Christianson and Creel 2010). Additionally, survival rates may be lower for fawns living near human-influenced landscapes or roads (Watine 2015), regardless of a lower perceived predation risk.

Non-invasive spatial capture-recapture (SCR) methods have been developed to estimate survival and recruitment (Gardner et al. 2010, Royle et al. 2013), while accounting for variation in detection probability. Detection probability is strongly influenced by activity rate, which may be affected by factors such as human activity and changes in the environment. Although reduced
predation risk from predators may increase prey species use of roads and other human-influenced areas (Berger 2007), white-tailed deer in heavily-hunted regions alter their movement patterns to avoid interaction with roads and humans (Bonnot et al. 2013, Little et al. 2016). A prey species’ spatiotemporal behavioral responses to this harvest pressure and predation risk may influence individual detection probability in SCR studies. Environmental factors such as increased water level in a home range may also impact detection probability by influencing deer movement and activity rates (MacDonald-Beyers and Labisky 2005).

Although research has shown that environmental variables can impact fawn production and individual survival, it is often assumed that population-level recruitment rates of many temperate ungulates, such as white-tailed deer, are relatively stable across years in populations maintained at a constant density relative to the carrying capacity (McCullough 1987). However, recent studies have demonstrated that annual average recruitment rates may vary due to changes in habitat quality (Tollefson et al. 2011), introduction of novel predators (Gulsby et al. 2015, Chitwood et al. 2015), availability of seasonal mast (eg., Quercus spp. acorns; Wentworth et al. 1992), and local weather patterns (Ginnett and Young 2000, Ciuti et al. 2015). Therefore, up-to-date information about local recruitment rates is needed to effectively manage ungulate populations.

In South Florida, white-tailed deer experience unique pressures from seasonal flooding, low-quality forage, and a diverse predator community, including the federally endangered Florida panther (Puma concolor coryi; Comiskey et al. 2002, Pimm et al. 2006, FWC 2017). Deer in South Florida are also prey for alligators (Alligator mississippiensis), black bears (Ursus americanus), bobcats (Lynx rufus), coyotes (Canis latrans), and recently, the invasive Burmese python (Python bivittatus) (Maehr and Brady 1984, Maehr et al. 1990, Labisky and Boulay 1998,

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 doe (Richter and Labisky 1985, McCown 1991, Fleming et al. 1994). Fawn survival in South Florida can be highly variable due to local environmental conditions. A three-year study of collared fawns in South Florida’s Big Cypress National Preserve (BCNP) and Everglades National Park (ENP) found 12-month fawn survival rate to be 0% and 42.6% in two concurrently-sampled study sites (Labisky et al. 1995). However, overall fawn survival to 18 months was 9% (5 of 55 fawns) in this same study (Labisky et. al 1995). There have been no other recent estimates of fawn recruitment in South Florida, apart from the Florida Fish and Wildlife Conservation Commission’s (FWC) annual aerial recruitment surveys performed over the Everglades Wildlife Management Area and surrounding state lands. These data indicate that fawn:doe ratios averaged 35.44% (ranging from 17%-46.33%) from 2007-2017, with a recruitment rate estimate of 17% in 2016 (Ward, M. R., Florida Fish and Wildlife Conservation Commission, unpublished data). However, aerial surveys are often biased by detectability (Caughley 1975) and are primarily used to track trends in fawn recruitment.

In recent years, decreases in deer hunter success rates have been recorded at some hunter check stations in BCNP (McBride and Scott 2017). Additionally, FWC’s aerial surveys over BCNP have indicated a declining deer population in some units (Garrison et al. 2011). Meanwhile, Florida panther populations have increased from 20-30 individuals in the early 1990’s to between 120-230 individuals today due to conservation efforts (Pimm et al. 2006,
FWC 2017). The reported deer decline and concurrent increase in populations of the deer’s primary predator have raised concerns about the viability of the white-tailed deer population in South Florida.

We examined white-tailed deer productivity and fawn survival to recruitment in South Florida using a newly developed spatial capture-recapture (SCR) model (Chandler et al., in review) applied to encounter histories of fawns uniquely identified using their spot patterns. We used data from 180 passive cameras to estimate (1) fawn productivity in the study area, (2) the impact of factors affecting spatial variation in birth locations, (3) fawn survival rate to the recruitment age, and (4) the impact of factors affecting fawn activity and detection rate. We investigated the effects of vegetation type, fire history, time since fire, and relative abundance of vegetation through the Normalized Difference Vegetation Index on spatial variation in the density of fawn birth locations. We also evaluated the effects of seasonal changes in groundwater levels, camera trap placement on a road or established trail, human activity, and vehicle use in our study grids on fawn detection rates.

METHODS

Study Site

The Big Cypress Basin (BCB) of South Florida lies northwest of the large, low-lying Everglades Swamp, which covers most of the southern tip of Florida. The BCB is generally higher in elevation than the Everglades, with more diverse topography and vegetation (McPherson 1974). The BCB is a tropical savannah with distinct wet and dry seasons (Hela 1952). Mean annual rainfall is 135 cm per year, with 60-80% of the rain falling between May and September (McPherson 1974, Duever et al. 1986). During this wet season, the shallow
aquifers fill and sheets of water flow southwest over much of BCNP along the slight elevational decline of 9 cm/km (Labisky et al. 1995). From October to April, the BCB gets relatively little rainfall and much of the seasonally-inundated land dries. The mean annual air temperature is 23°C, and mean monthly temperatures range from 14°C in January to 28°C in August (Duever et al. 1986).

The dominant vegetation types of the BCB are determined by topography, hydroperiod, and soil type. Pine flatwood forests tend to have a relatively open overstory and absent or sparse mid-story. The canopy is dominated by slash pine (*Pinus elliottii*) and cabbage palm (*Sabal palmetto*), and the groundcover includes saw palmetto (*Serenoa repens*) and grasses from the *Panicum*, *Aristida*, and *Andropogon* genera. Pine flatwoods are maintained by frequent fire, which allows diverse plant communities to flourish and reduces hardwood abundance (Loveless 1959, McPherson 1974, Duever et al. 1986). Cypress forests and mixed swamps are characterized by bald and pond cypress trees (*Taxodium distichum* and *T. ascendens*), and they also often include pond apple (*Annona glabra*) or pop ash (*Fraxinus caroliniana*) sloughs and willow heads (*Salix* spp.). Small shrubs and hardwoods trees, such as wax myrtle (*Morella cerifera*), are common in the sparse understory of open cypress forests. Mixed swamps are more densely forested by cypress, willow, pop ash, and pond apple in the deeper areas, with cabbage palm, red maple (*Acer rubrum*), wax myrtle, and bays (*Persea* spp. and *Magnolia* spp.) distributed throughout. Due to the thick vegetation and frequent to year-round water inundation, wildfires rarely burn completely through the mixed swamp (McPherson 1974, Duever et al. 1986). Hardwood hammocks are elevated slightly more than the swamps, marshes, and prairies, which allows hardwood species such as cabbage palm, red maple, laurel oak (*Quercus laurifolia*), water oak (*Q. nigra*) and live oak (*Q. virginiana*) to dominate the dense overstory,
which is up to 20 m high. Fire is rare in these communities due to the understory of shrubs, vines, ferns, and epiphytes that retain high humidity, leading to a fire-resistant climax community (McPherson 1974, Duever et al. 1986). Marshes are characterized by sparse vegetation, peaty soils, low elevations, and are inundated for 225-275 days per year. The emergent wetland vegetation found in marshes typically consists of cattails (*Typha* spp.), sawgrass (*Cladium mariscus*), rushes (*Juncus* spp.), and alligator flag (*Thalia geniculata*) (McPherson 1974, Duever et al. 1986). Wet prairies have shorter hydroperiods than marshes and are dominated by relatively short mixed grasses and sedges with occasional dwarfed cypress trees (Duever et al. 1986). They often form the transition between marshes and cypress sloughs (Loveless 1959). Dry prairies are slightly higher in elevation than wet prairies, seldom inundated, contain more grasses, sedges, and saw palmetto, and include some shrubs and small trees (McPherson 1974, Duever et al. 1986). Wet scrubland is a transitional vegetation type, which includes small cypress trees, willows, and other hardwood trees and shrubs less than 6 m high in poorly drained soil with frequent standing water. These areas occasionally have isolated pines or bays but no true overstory canopy (Kawula 2009).

This study occurred on the Florida Panther National Wildlife Refuge (FPNWR) and in two management units of the Big Cypress National Preserve (BCNP): North Addition Lands (AL) and Bear Island (BI) (Figure 2.1). FPNWR is closed to public access and has an extensive network of off-road vehicle (ORV) trails and graded roads, which are used primarily as prescribed fire breaks and for research and management activities. There is no hunting allowed on FPNWR. The Add Lands management unit is open to the public for hiking and limited permit-only hunting, but access to the unit is restricted to foot-travel. However, AL has an extensive network of hiking trails and a few graded roads which are maintained as fire breaks.
and used infrequently by administrative personnel. The Bear Island management unit provides the most public access of the three units, with multiple graded roads leading to developed campsites and licensed hunting allowed. An extensive ORV and hiking trail network is also present in BI, and multiple trails are open for year-round public ORV use. Across the 53,333-ha rectangular study area of these three management units, the vegetation cover was 38.25% cypress, 20.39% flatwoods, 5.7% hammock, and 35.65% open canopy, which included marsh, wet and dry prairies, and wet scrubland (Figure 2.2, Table 2.1).

The reproductive season for white-tailed deer in South Florida is less synchronized than in more temperate parts of the white-tailed deer’s range (Richter and Labisky 1985, Garrison and Gedir 2006). Fetal measurements indicate that peak breeding in BCNP occurs in mid-August (Richter and Labisky 1985). Fawn drop is timed to match the peak of the dry season (Richter and Labisky 1985, Garrison and Gedir 2006), which occurs in mid-February (Labisky et al. 1995). However, fawning in this area has been documented throughout a wide range of dates, from November to March (Fleming et al. 1994).

**Camera Deployment**

We deployed 180 white-flash motion-sensor cameras (HCO Outdoor Products, Norcross, GA, USA) across the study area in three arrays of 60 cameras to cover the gradients of hydrology and hunting pressure across the region. We placed one array of 60 cameras in each of the FPNWR, BI, and AL management units (Figure 2.1). We overlaid a 700-m² grid over each 6x5 km rectangular study site grid and used ArcGIS 10.2 (Environmental Systems Research Institute, Redlands, CA, USA) to place proposed locations for the cameras in the midpoints of those cells. In each array, we deployed forty cameras on established roads or maintained hiking trails and the remaining twenty cameras approximately 250 m from a road or hiking trail. We placed the on-
trail cameras on the closest suitable tree to the center of the 700-m² grid cell. To increase off-trail detection probability, we deployed these cameras along the most well-established game path or habitat edge near the center of the 700-m² grid cell. All cameras were oriented north as much as possible to avoid direct sunlight. We initially set cameras approximately 0.33 m off the ground but raised and lowered them to a maximum height of 1 m throughout the study to account for seasonal fluctuations in water level. Camera density was chosen to maximize area sampled while allowing for multiple cameras within individual home ranges to increase the potential for spatial recaptures. We visited all camera locations approximately once a month to service cameras, download data, and clear vegetation to avoid visual obstruction and reduce false triggering of the cameras. The vegetation cover within the three camera grids was 29.29% cypress, 29.1% flatwoods, 35.5% open canopy, and 5.5% hardwood hammock (Table 2.1). Of our 180 cameras, 21.1% were placed in cypress, 42.8% in flatwoods, 31.1% in open canopy, and 5% in hammock.

**Fawn Identification and Aging**

We uniquely identified fawns in photos using their spots, which are distinctive from birth until approximately five to six months old (Figure 2.3). As described in Chandler et al. (*in review*), we identified unique spot patterns using images from each side of the fawn, as well as photos from directly behind or in front of the fawn, to confirm an individual fawn ID.

We estimated birth date ranges for all fawns based on morphological characteristics in the photos. We were unable to age fawns using the standard method of hoof growth when using our non-invasive camera-detection method. However, Sams et al. (1996) found that body mass, chest girth, body length, hind foot length, and head length were useful factors in determining age of white-tailed fawns when measuring hoof growth is not possible. We considered body size of 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 skilled observers with multiple seasons of experience in capturing and collaring deer of various ages independently viewed all images from first to last detection of each fawn. 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. Each observer created a range of possible birth dates by estimating the minimum and maximum age of a fawn in each photo. The birth date ranges created by each observer were very similar. Discrepancies were resolved by combining the earliest minimum and latest maximum birth date from the two observers to create the most conservative range.

**Data Processing**

Fawns often remained in front of cameras for multiple minutes, resulting in many non-independent photo detections. We removed non-independent detections using a 1 hour detection threshold. Specifically, we discarded detections of a fawn if it had been previously detected at the same camera within one hour. We then created spatially-referenced capture histories indicating the detections of each fawn at each camera during the study. We used all fawn photos between 1 December 2015 and 30 June 2016 from all three camera grids. We developed a binary camera operational status matrix to account for camera malfunctions.

**Model Fitting**

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 of Chandler et al. (*in review*). This model is a type of open population spatial capture-recapture (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 can be modeled using spatially-referenced covariates. Individual lifetimes are modeled using a failure-time approach (Chandler et al, in review), with a survival rate that can be modeled as a function of individual-specific, time-varying covariates, including age. Detection probability is 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. (in review), the probability of detecting a fawn far from its birth location is assumed to increase with age, as the fawn becomes more mobile. Detection parameters can also be modeled using trap-specific, temporally-varying covariates.

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 multiple spatially-referenced environmental variables, including vegetation type, relative abundance of vegetation through Normalized Difference Vegetation Index (NDVI), fire history, and time since fire, to evaluate effects on the number of fawns born and spatial variation in birth location density (Table 2.1). 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 (Figure 2.2). We 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 does and fawns (Suppl. Table 1). These categories were pine flatwoods, cypress forests (including mixed swamp), hardwood hammock, and open canopy (including marsh, wet and dry prairie, and wet scrub) using the vegetative descriptions provided by the Florida Land Cover Classification System (Kawula 2009), Florida Land Use, Cover and Forms Classification System (FDOT 1999), and the FNAI Guide to the Natural Communities of Florida: 2010 edition (FNAI 2010).

We calculated the NDVI for the entire study site on February 16, 2016 to quantify the relative abundance of vegetation and cover available for fawns during the peak of the first critical weeks of life (Table 2.1). Fire history was determined separately for the FPNWR and BCNP management units (Figure 2.4). We used FPNWR burn unit level records of wildfire and prescribed burns to calculate fire frequency between January 1, 1995 and December 31, 2015. On the two BCNP grids, we used ArcGis 10.5 (ESRI, Redlands, CA, USA) to stack individual raster layers detailing each fire that occurred within our study area and calculated the frequency of fire at each 30x30-m pixel between January 1, 1995 and December 31, 2015. Time since fire was calculated using fire data within the last thirty years for both FPNWR and BCNP (Figure 2.5). We used January 1, 2016 as the cutoff date for “pre-study” fires, as fawning peaks from January through March across our study area. In FPNWR, we recorded the date of the most recent burn or wildlife in each burn unit. In BCNP, we used ArcGis 10.5 to stack all fire raster layers in our study area and isolated the date of the most recent fire in each pixel. We then computed the difference in days between the most recent fire in each pixel or unit and January 1,
2016 as the time since fire for each pixel, using 10950 days (30 years) as the maximum cutoff (Table 2.1).

Camera-level covariates hypothesized to influence detection rates included trail status, daily water depth, human activity, and vehicle activity (Table 2.1). Trail status was represented as a binary variable, with zero indicating an off-trail camera and one indicating an on-trail camera. Daily water depth was a camera-specific, time-varying covariate quantified using depth gauges in marshes across BCNP. We averaged the daily mean groundwater depth from 3 deep-water ground wells maintained by the Everglades Depth Estimation Network (EDEN) project and the US Geological Survey. We used an index of average ground elevation in each vegetative community to adjust the mean water depth measurement at each camera (Sobczak, R., National Park Service, personal communication), which allowed us to account for elevational differences in vegetation types leading to corresponding variation in standing groundwater levels. Human activity and vehicle activity were calculated by the number of photos taken of humans and vehicles, respectively, per camera per day. This provided a camera-specific, time-varying rate for both humans and vehicles, adjusted by camera operational status.

Bayesian methods were used for statistical inference. Analysis was performed using a Gibbs sampler written in program R (R Core Team, 2016). We first ran the basic model with no covariates, then we began evaluating each covariate independently, retaining those covariates with significant effect sizes. If the 95% credible interval around the parameter estimate did not include zero, we retained the variable into the next round of models. We fit four final models which each contained the 4 vegetation types and the fire history from 1991 to 2015 (Table 2.2). Each of these four final models included a different detection covariate (Humans, Vehicles, Water, Trail) (Table 2.2).
RESULTS

Photo Data Summary

Trail cameras were operational for 37,492 of 38,340 (97.8%) of the possible trap days in our 213-day study period. Our complete dataset included 5,242 photos of spotted fawns captured between 1 December 2015 and 30 June 2016. We were unable to use 29 (0.55%) of these photos, which were taken during rain or during low light conditions, and we removed these photos from all subsequent analysis.

We processed 5,213 images of spotted fawns taken that were sufficient for individual identification. Although we only used one camera at each survey location, most fawns spent substantial time in front of the cameras, which allowed us to capture images of both sides. However, several fawns walked past a camera at a broadside angle that did not allow us to link a one-sided image to the other side. In ambiguous cases where two one-sided images could not be confirmed to be the same fawn, we assigned a unique ID to one side and discarded the other side, resulting in 14 (0.27%) otherwise sufficient photos discarded. We then used the remaining 5,199 photos to identify 123 unique individuals across the study site: 28 in the Add Lands grid, 31 in the Bear Island grid, and 64 in FPNWR. We linked both sides of a fawn together for 97 of 123 (79%) detected fawns. The remaining 26 of 123 (21%) fawns consisted of an ID with only images of one side. However, we could confidently conclude, based on the age of the fawn and the location of each detection, that these one-sided individuals were distinct from every other one-sided individual detected in that camera grid.

The first fawn detection occurred on 20 January 2016, and we detected multiple fawns through 30 June 2016, the end of data collection (Figure 2.6). After the first detection of each identified fawn, we discarded all subsequent detection of that fawn at the same camera within a
one hour window to produce 898 independent detection events (Figure 2.7). Fawn detections occurred on 125 of our 180 deployed cameras. Seventy-eight of the 123 individuals were detected only at one camera location, 32 were detected at 2 cameras, 6 were detected at 3 cameras, 4 at 4 cameras, and 3 at 5 cameras. Twenty-five of the 123 fawns were only detected on one occasion. The two most frequently detected fawns were detected on 35 and 36 occasions each. We detected two fawns in the same image on 8 occasions, which only occurred in the FPNWR grid. We are unable to confirm that these fawns were twin pairs, but similarities in size of the fawns and the exclusive association of one doe with each pair, suggest that, at a minimum, 4 does may have produced twins in FPNWR during our study. Because twinning appears to be very rare in our study area, we modeled all birth locations as statistically independent events, conditional on the covariate effects.

The mean length of the estimated birth window for the 123 identified fawns was 23 days, with a range from 4 days to 34 days (Figure 2.6). The birth windows ranged from the earliest possible birthdate of 6 November 2015 to the latest possible birth date of 10 May 2016. Using the midpoint of each birthdate range, we found that the mean birth dates in BI and AL were 12 February 2016 and 13 February 2016, respectively. However, the mean birth date in FPNWR was 6 January 2016, over one month earlier than in the two grids in BCNP. We used a window of 180 days for possible detection of each fawn, starting with the first potential birthdate for each individual, to account for spot loss around 6 months of age, and thus our inability to continue detecting individuals past 6 months old.
Model Results and Covariate Effects

The estimated mean birth date across our study site was 25 January 2016 (± SD 2.8 days) for all detected and undetected fawns in the study (Figure 2.8). An estimated 211 (95% CI 180-250) fawns were born in the 10,941-ha camera grid area, defined by the 800-m buffers around the camera locations, with the highest number of fawns alive at one time peaking in March (Figure 2.9). Extrapolated estimates for the complete study area of 53,333-ha, which includes the three camera grids and the area between the grids, yielded 1,033 fawns born (95% CI 834-1271). Fawn birth location density was highest in cypress, and lowest in hardwood hammocks (Table 2.3, Figures 2.10, 2.11, 2.12). Density of fawns born in flatwoods and open canopy were similar. Fawn birth location density was significantly higher in areas that had experienced more fire within the past twenty years (Table 2.3, Figures 2.11, 2.12, 2.13), but days since fire did not have a significant impact on fawn birth density. Similarly, NDVI did not have a strong effect in explaining variation in birth location density.

Fawn survival to 30 days was 75%, but survival to 120 days declined to 38%. By the recruitment age of 180 days, survival further declined to 18% (Figure 2.14). Of the estimated 211 fawns born within the 10,941-ha camera grid area, 37 (95% CI 27-48) fawns survived to 180 days (Figures 2.9, 2.14). Throughout the extrapolated 53,333-ha complete study area, an estimated 181 (95% CI 131-234) of 1,033 fawns born reached the recruitment age of 180 days.

Fawn activity rate, as interpreted from the daily detection rate of fawns, significantly decreased with both increasing human activity and vehicle traffic. However, fawn detection rate increased with mean standing groundwater depth (Figure 2.15). The trail status of the camera, whether the cameras was deployed on- or off- an established trail or road, did not have a
significant effect on fawn detection rate, although there was a trend towards higher detection rates on trails and roads (Figure 2.16).

**DISCUSSION**

To effectively manage wildlife populations, estimates of adult mortality and juvenile recruitment rates are needed. Understanding recruitment is particularly important when managing a species with highly variable juvenile survival, such as white-tailed deer (DeYoung 2011). We applied a novel non-invasive SCR method 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 1990s (Land 1991, Labisky et al. 1995) and evaluated the landscape-level environmental variables that impact fawn birth location and activity rates in this region.

Fawn production was lower in South Florida than in many other areas of the white-tailed deer’s range, which is primarily attributable to low-quality nutrition throughout the region (Harlow and Jones 1965, Duever et al. 1986), leading to low fecundity rates. Pregnancy rates for adult deer in BCNP are relatively high (93%) (McCown 1991), however young-of-year fawn (≤1 year) pregnancy rates in South Florida are low (10%) (Richter and Labisky 1985). Additionally, only 25% of adult females produce twins (Richter and Labisky 1985), contributing to South Florida’s low average fecundity rate, estimated between 1.18-1.26 fetuses/pregnant doe (Richter and Labisky 1985, Fleming et al. 1994). The density of fawn birth locations ranged from 0.0296-5.632 fawns per km², which agrees with expected low fawn production throughout this region.
Density of fawn birth locations was highest in cypress vegetation, similar in flatwoods and cypress, and lowest in hardwood hammock. The nutritional quality of forage available to does and fawns within each vegetation type and the presence of open sightlines to reduce predation risk may help to explain patterns of fawn birth location density in certain vegetation types. Open cypress forests have relatively sparse understories that may permit easier predator detection by adult female deer. Although cypress forests do flood yearly, fawns may have been born around the drier edges of the cypress domes. Fawning occurs at the peak of the dry season, so seasonal flooding may not preclude does and fawns from using these areas. Hardwood hammocks were estimated to have the lowest density of fawns, which may be attributable to the low-quality forage caused by poor soils in late successional forests (Harlow and Jones 1965, Comiskey et al. 2002) and the thick understory of the hammocks precluding open sightlines to detect predators. However, because hammock vegetation was only represented by 5.5% of the study grid, our sample within that vegetation type likely was not sufficient to precisely estimate fawn density, evidenced by the large credible intervals around that estimate.

We found an effect of fire frequency, but not time since fire, on the density of fawn birth locations. Fire can increase the nutritional content of forage and encourage the growth of more herbaceous material (Stransky and Harlow 1981), which may support higher fawn production and recruitment in a nutritionally-stressed deer population (Taber and Dasmann 1957, Harlow and Jones 1965, Stransky and Harlow 1981). We expected does to select birth locations in areas with recent fire, but we did not detect a significant effect of time since fire on birth locations. The initial effects of fire on increased palatability and nutrition in fresh growth only persist for a few months (Stransky and Harlow 1981, Main and Richardson 2002), although Vogl and Beck (1970) noted that deer still selected for an area where a high-intensity fire had occurred 8 years
prior. The three most recent fires in each camera grid ranged from 6 to 14 months (FP), 1.6 to 6 years (AL), and 2.5 to 4.7 years (BI), and the longest time since fire in some areas exceeded our cutoff date of 30 years. This high variability in fire return interval throughout the study area may have precluded our ability to detect fine-scale temporal selection of the most recently burned areas. Our analysis did, however, reveal a strong positive effect of increased fire frequency in the past twenty years on the density of fawn birth locations. Fire can shape the distribution of vegetation communities across the landscape (Loveless 1959, Vogl and Beck 1970, Bond and Keeley 2005) and provide open sightlines to reduce predation risk from ambush predators (Cherry et al. 2018). Conversely, concealment cover can be very important for young fawns and does due to offering protection from cursorial predator hunting strategies (Cherry et al. 2017). Thus, a home range offering a mosaic of different levels of post-fire rough and regrowth, as well as habitat with open sightlines, would likely provide ideal habitat for deer in South Florida.

The BCB is characterized by seasonal inundation in the lower-lying vegetation types, namely marsh, wet prairie and scrub, and cypress. In extremely wet years, the higher-elevation pines and hammocks can temporarily flood as well. Due to human alteration of the natural flooding cycles through draining, re-routing floodwaters, and establishment of canal systems (Duever et al. 1986, Light and Dineed 1994), an increasingly intense cycle of floods and droughts in the past few decades has had a drastic impact on deer survival rates and movement patterns (MacDonald-Beyers and Labisky 2005, Garrison et al. 2011). Previous research indicated that water depths above 50 cm severely impact adult deer movement, survival, and productivity (MacDonald-Beyers and Labisky 2005). Additionally, the same 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 recruitment (Fleming et al. 1994, Garrison et al. 2011). Mean daily water levels from January to May 2016 ranged from 60-85 cm in marsh vegetation, exceeding the level found to put fawns at risk (Fleming et al. 1994). While we were unable to test the effects of annual differences in fawn recruitment between a dry fawning season and wet fawning season, we did evaluate the short-term effects of fluctuating water levels on fawn daily detection rates. With temporary increases in water level, fawn detection rate also increased, indicating increased fawn activity and movement rates. Deer may increase their activity rates to obtain necessary nutrition during high water events, potentially due to the energetic needs of pregnant or lactating does. It is possible that the increased activity of fawns resulted from following the dam during a period of nutritionally-driven increased activity. Another possible explanation of increased detection rates with increased water levels could be that 60% of our cameras were placed on trails or roads. Some of these roads were elevated grades, and fawns may use these elevated paths more frequently during times of high water to avoid traveling through the surrounding flooded wetlands. However, placement of cameras on-or off-trails did not significantly influence our detection rates.

Fawn detection rate decreased with increasing human activity and vehicle traffic, indicating that does and fawns avoided disturbance caused by human recreation, including hunting, hiking, and ORV use. However, we also saw a trend towards more frequent fawn detections at on-trail cameras, although this effect was not significant. These seemingly-conflicting patterns may be explained through spatiotemporal variation in deer use of roads. Roads may provide a human shield effect to provide safety from mammalian predators (Berger 2007). Conversely, this increased activity of fawns on roads may merely be caused by the increased elevation of some roads in our system, which permits easy travel and drier conditions.
However, previous work in our study area found that does increased their activity and travel on roads and trails during diurnal periods, when panthers were less active (Crawford 2017). Regardless of the motivation to use roads, does and fawns appear to alter their spatial and temporal use of roads and human-dominated areas to avoid both the high-risk times for panther predation and increased human activity (Bonnot et al. 2013, Crawford 2017). Future work should investigate the extent to which fawn activity on roads varies over diel periods.

Although our study design allowed us to evaluate the effects of landscape-scale covariates such as vegetation, fire, water, and human disturbance on fawn density and detection among and within the camera grids, some of these covariates varied among the three management units more than they did within the units. We designed our camera grids in the three management units to purposely avoid confounding environmental conditions by sampling across a continuous gradient of human activity, hunting pressure, vegetation types, and hydrology patterns. However, some of the most profound variation we observed occurred among, not within, the three management units. Notably, far more fawns were born in FPNWR relative to the two other management units (N=64 in FPNWR, N=28 in Add Lands, and N=31 in Bear Island). FPNWR had a higher density of graded roads and trails throughout our camera grid, as well as the most recent fires and the highest number of fires, and it is possible that factors other than those we measured may be responsible for this variation. Manipulative experiments would be useful for determining the extent to which variables such as fire frequency directly impact fawn production, survival, and recruitment in South Florida.

The fawn survivorship curve showed a relatively high survival rate to 30 days (75%) compared to many other white-tailed fawn survival studies in the Southeast. 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, Nelson et al. 2015, 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 38% of fawns surviving to 120 days and only 18% of fawns surviving to 180 days. The difference between the pattern of our 2016 cohort’s survivorship curve and survivorship curves of other fawn survival studies may be due to sampling methodology. In studies relying on capture of fawns, radio-instrumentation could impact survival of young fawns. Although research indicates there is low risk of marking-induced abandonment in deer (Powell et al. 2005), there may be non-abandonment effects on fawns, such as reduced fitness from wearing a collar, including weight loss or collar injury, or a higher risk of predation due to higher visibility or inability to flee as effectively. This may cause the neonate mortality rate in fawn-collaring studies to be overestimated at young ages. Simultaneous use of SCR techniques with a traditional fawn collaring study to estimate survival would clarify if the differences we observed are real or influenced by the technique used.

Alternately, the unique pattern of fawn survivorship may be due to the structure of the predator community in South Florida. Coyotes are increasingly implicated as very efficient primary predators of fawns during the first few weeks of life (Thornton et al. 2004, Kilgo et al. 2014, Nelson et al. 2015, Chitwood et al. 2015), but coyotes in Florida do not utilize fawns or adult deer as frequently as they do throughout the rest of their range (Thornton et al. 2004, Watine 2015). Additionally, coyote density in our study site is very low, and there is no known documented coyote reproduction or denning in BCNP (Shindle, D. US Fish and Wildlife Service, personal communication). During 2016, we documented an average of only 21 photos of coyotes
per camera grid. Black bears are also highly effective fawn predators at consuming neonates during the first few weeks of life (Shuman et al. 2017). However, bears are not effective at catching fawns past three weeks old, and due to the late January fawn drop in South Florida, most fawns are 2-4 months old by March and April when bears become active after their winter dormancy. Additionally, bear density in Big Cypress is relatively low, at 0.132 bears/km$^2$; Humm et al. 2017) compared to a density of 0.66 bears/km$^2$ in an area where bears have been found to cause high fawn mortality (Hooker 2010, Shuman et al. 2017). The relative low predation pressure from coyotes and bears in our study area likely shapes the comparatively high survival observed within the first 30 days of fawn life. However, bobcats can be effective in predating both adult and juvenile deer (Land 1991, Labisky et al. 1995, MacDonald-Beyers and Labisky 2005, Nelson et al. 2015, Shuman et al. 2017) and were noted to cause 58% (31 of 53) of fawn deaths in BCNP and ENP, primarily when fawns were between 3-5 months old (Labisky et al. 1995). Panthers are also extremely effective ambush predators in depredating adult deer and fawns (Maehr et al. 1990, Dalrymple and Bass 1996, Garrison and Gedir 2006). Therefore, bobcats and panthers are likely more important sources of predation to older fawns. Additionally, alligators and invasive Burmese pythons are known to take both adult deer and fawns (Epstein et al. 1983, Land 1991, MacDonald-Beyers and Labisky 2005, Boback et al. 2016), especially in flooded habitats. This variety of predators likely shapes the continual decline in fawn survivorship we observed between three- to six-months, when many other white-tailed fawn studies report a marked decrease in fawn mortality.

Recent reports of decline in South Florida deer populations have raised concerns about the stability of these populations. Deer density has been estimated in BCNP from 3.5 deer/km$^2$ to 4 deer/km$^2$ (Labisky et al. 1995), but this study was conducted over 20 years ago. Since then,
Florida panther populations have increased from merely 20-30 individuals in the early 1990’s to between 120-230 individuals today due to extensive conservation effort (Pimm et al. 2006, FWC 2017). Recently, FWC’s annual aerial deer population surveys over BCNP have observed apparent declines in deer numbers (Garrison et al. 2011). Although there is a high level of human recreation in some of our study areas, hunting pressure is not a significant source of mortality in our study sites, with only 4 deaths of 239 collared deer attributable to harvest in our survival sample from 2015-2017 (unpublished data, this study). However, in some units of BCNP, decreases in deer hunter success rates have been recorded at hunter check stations over the past decade (McBride and Scott 2017), supporting concerns of deer decline. The largest source of adult mortality in this study was depredation by Florida panther, followed by bobcat (unpublished data, this study). Due to high depredation rates, South Florida’s deer population almost certainly exhibits a lower annual survival rate now than the 81.3% survival rate found in BCNP in the late 1980s and early 1990s (Land 1991). Therefore, ensuring the sustainability of South Florida’s deer population depends on adequate replacement through fawn recruitment to balance the losses sustained from adult mortality.

We estimated that annual fawn recruitment to 180 days is 18%. However, we were unable to monitor survival past 6 months, when spot patterns fade and fawns develop adult pelage. Previous research in ENP estimated 56% fawn survival to 12-months (N=34, 1987-1990) (Eisenberg and Sunquist 1994). A study that followed 13 collared fawns in Bear Island from 6 months to 12 months saw 92.1% survival of those individuals (Land 1991). Contrary to these high survival estimates, a study in both BCNP and ENP estimated 12-month fawn survival at 23.3% (11%, 34%, 0.03% for each year) averaged over both sites. However, they found 18-month survival of these fawns in both sites to be only 9% (Labisky et al. 1995). These varied
results indicate that the steep fawn mortality rate in the first 6 months of life may decline somewhat as fawns age. The mortality risk to juvenile white-tailed deer in South Florida evidently remains substantial through adulthood and can be highly variable depending on annual environmental influences.

Survival and recruitment estimates suggest that very few fawns born in this system will survive to reproductive maturity. However, our study was conducted during an extremely wet fawning season and includes only one year of data. It is likely that the extremely wet spring conditions resulted in a lower-than-typical survival rate for fawns in 2016. However, a consistently low survival rate over multiple years, along with a potential reduction in adult survival, may indicate a cause for concern for the viability of South Florida’s deer population. Multiple fawning seasons of data are needed to assess the annual variability in fawn survival and recruitment over time. Future research should continue to monitor both the adult and fawn survival rates of this population as part of a comprehensive population viability analysis. Additionally, further work in this system should evaluate how spatial and temporal environmental covariates, such as vegetation, fire, water level, and human disturbance, can influence the individual survival rates of fawns, as we demonstrated here with the density of fawn birth locations. Similar studies as ours should also continue to utilize our novel SCR method of estimating fawn survival and further evaluate the effectiveness of this method by conducting a concurrent fawn-collaring study.

**MANAGEMENT IMPLICATIONS**

Our research concurs with previous studies indicating low fawn survival in this system. Our evaluation of environmental variables suggests that increased fire frequency on the
landscape is beneficial for fawns in South Florida. We found that fawns frequently used roads for travel during our study; however, roads that experienced high disturbance from human activity and vehicle traffic were avoided. Management agencies and private land owners in South Florida may be able to increase fawn survival and recruitment by increasing fire frequency and limiting human and vehicular access. Most importantly, our work necessitates continued monitoring of fawn survival to the recruitment age and adult survival rate in these populations to ensure an adequate prey base for the Florida panther. Population viability analyses based on updated productivity, recruitment, and survival rates are warranted for effective future management.

LITERATURE CITED


Florida Fish and Wildlife Conservation Commission and Florida Natural Areas Inventory. 2016. Cooperative Land Cover version 3.2 Raster. Tallahassee, Florida, USA.

Florida Natural Areas Inventory 2010. Guide to the Natural Communities of Florida. Institute of Science and Public Affairs at The Florida State University. Tallahassee, Florida, USA.


Table 2.1. Summary statistics of environmental predictor variables used in the models of fawn birth location density and daily detection rate, including vegetation type (hammock, cypress, flatwoods, and open canopy), number of fires in recent history, days since fire, NDVI, daily human photo rate, daily vehicle photo rate, and mean water level.

<table>
<thead>
<tr>
<th>Environmental Variable</th>
<th>Study Area (53,333 ha)</th>
<th>3 Grids (10,941 ha)</th>
<th>Add Lands</th>
<th>Bear Island</th>
<th>FPNWR</th>
</tr>
</thead>
<tbody>
<tr>
<td>% Cover of Cypress</td>
<td>36.5</td>
<td>29.3</td>
<td>30.7</td>
<td>8.4</td>
<td>50.3</td>
</tr>
<tr>
<td>% Cover of Flatwoods</td>
<td>19.4</td>
<td>29.1</td>
<td>37.9</td>
<td>18.4</td>
<td>31.2</td>
</tr>
<tr>
<td>% Cover of Open Canopy</td>
<td>34.0</td>
<td>35.5</td>
<td>30.8</td>
<td>64.2</td>
<td>11.2</td>
</tr>
<tr>
<td>% Cover of Hammock</td>
<td>5.4</td>
<td>5.5</td>
<td>0.6</td>
<td>9.0</td>
<td>7.3</td>
</tr>
<tr>
<td>Median (Range) Number of Fires</td>
<td>1 (0,7)</td>
<td>2 (0, 7)</td>
<td>3 (0, 7)</td>
<td>1 (0, 3)</td>
<td>4 (0, 6)</td>
</tr>
<tr>
<td>Median Days Since Fire</td>
<td>-</td>
<td>3161</td>
<td>3161</td>
<td>5343</td>
<td>1367</td>
</tr>
<tr>
<td>(Range) Days Since Fires</td>
<td>-</td>
<td>(155, 10950)</td>
<td>(182, 10950)</td>
<td>(610, 10950)</td>
<td>(155, 10950)</td>
</tr>
<tr>
<td>Median NDVI Score</td>
<td>0.159</td>
<td>0.166</td>
<td>0.147</td>
<td>0.160</td>
<td>0.201</td>
</tr>
<tr>
<td>Mean (Range) Humans/Day</td>
<td>-</td>
<td>0.49 (0, 170)</td>
<td>0.73 (0, 170)</td>
<td>0.52 (0, 65)</td>
<td>0.22 (0, 170)</td>
</tr>
<tr>
<td>Mean (Range) of Vehicles/Day</td>
<td>-</td>
<td>0.18 (0, 140)</td>
<td>0.15 (0, 140)</td>
<td>0.26 (0, 51)</td>
<td>0.12 (0, 131)</td>
</tr>
<tr>
<td>Median Water Level at Cam</td>
<td>-</td>
<td>1.06</td>
<td>1.03</td>
<td>1.25</td>
<td>0.91</td>
</tr>
<tr>
<td>(Range) Water Level at Cam</td>
<td>-</td>
<td>(-0.43, 1.81)</td>
<td>(-0.46, 1.78)</td>
<td>(-0.25, 2.00)</td>
<td>(-0.58, 1.66)</td>
</tr>
</tbody>
</table>
Table 2.2. Estimates of fawn abundance (N) and recruitment rates (R) from the four final models fitted to the fawn encounter history data. Deviance represents the posterior mean deviance, which we used with credible intervals to select the best model.

<table>
<thead>
<tr>
<th>Density Covariates</th>
<th>Detection Covariates</th>
<th>N</th>
<th>N (95% CI)</th>
<th>R</th>
<th>R (95% CI)</th>
<th>Deviance</th>
</tr>
</thead>
<tbody>
<tr>
<td>Vegetation + Fire</td>
<td>Human Activity c</td>
<td>211.3</td>
<td>180, 250</td>
<td>37.1</td>
<td>27, 48</td>
<td>7387.68</td>
</tr>
<tr>
<td>Vegetation + Fire</td>
<td>Daily Water Level d</td>
<td>211.2</td>
<td>179, 249</td>
<td>37.6</td>
<td>27, 47</td>
<td>7393.27</td>
</tr>
<tr>
<td>Vegetation + Fire</td>
<td>Trail Status e</td>
<td>207.1</td>
<td>178, 240</td>
<td>37.6</td>
<td>27, 49</td>
<td>7397.08</td>
</tr>
<tr>
<td>Vegetation + Fire</td>
<td>Vehicle Activity c</td>
<td>205.1</td>
<td>174, 243</td>
<td>38.5</td>
<td>27, 50</td>
<td>7397.24</td>
</tr>
</tbody>
</table>

a. Includes all 4 vegetation types (hammock, cypress, flatwoods, open canopy)
b. Number of fires at each point between 1995 and 2015
c. Daily rate of photos per day
d. Adjusted groundwater level at the camera location
e. On- or off-trail
Table 2.3. Posterior summary statistics (mean, SD, and 95% confidence intervals) from output of the best-fit model in Table 2.2.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Mean</th>
<th>SD</th>
<th>2.50%</th>
<th>97.50%</th>
</tr>
</thead>
<tbody>
<tr>
<td>Asymptote of sigma&lt;sup&gt;a&lt;/sup&gt;</td>
<td>310.320</td>
<td>8.77313</td>
<td>293.37148</td>
<td>327.59745</td>
</tr>
<tr>
<td>Rate at which sigma increases with age</td>
<td>20.621</td>
<td>1.69934</td>
<td>17.11271</td>
<td>23.94371</td>
</tr>
<tr>
<td>Baseline encounter rate&lt;sup&gt;b&lt;/sup&gt;</td>
<td>-2.1268</td>
<td>0.05933</td>
<td>-2.24583</td>
<td>-1.99067</td>
</tr>
<tr>
<td>Effect on encounter rate</td>
<td>-0.1758</td>
<td>0.08712</td>
<td>-0.36915</td>
<td>-0.02266</td>
</tr>
<tr>
<td>Mean birth date&lt;sup&gt;c&lt;/sup&gt;</td>
<td>56.0219</td>
<td>2.8125</td>
<td>50.5392</td>
<td>61.20898</td>
</tr>
<tr>
<td>Standard deviation of birth date distribution</td>
<td>30.6595</td>
<td>2.10998</td>
<td>26.53611</td>
<td>35.29944</td>
</tr>
<tr>
<td>Probability of birth location</td>
<td>-4.3586</td>
<td>0.27898</td>
<td>-4.89309</td>
<td>-3.78859</td>
</tr>
<tr>
<td>Effect 1 on probability of birth location</td>
<td>-3.5488</td>
<td>3.07004</td>
<td>-9.55528</td>
<td>0.67123</td>
</tr>
<tr>
<td>Effect 2 on probability of birth location</td>
<td>0.7037</td>
<td>0.39049</td>
<td>-0.11335</td>
<td>1.32649</td>
</tr>
<tr>
<td>Effect 3 on probability of birth location</td>
<td>0.354</td>
<td>0.35186</td>
<td>-0.29383</td>
<td>1.15535</td>
</tr>
<tr>
<td>Effect 4 on probability of birth location</td>
<td>0.2691</td>
<td>0.09906</td>
<td>0.08455</td>
<td>0.47848</td>
</tr>
<tr>
<td>Shape parameter of lifetime distribution</td>
<td>1</td>
<td>0</td>
<td>1</td>
<td>1</td>
</tr>
<tr>
<td>Scale parameter of lifetime&lt;sup&gt;e&lt;/sup&gt; (mean lifetime)</td>
<td>106.9762</td>
<td>13.716</td>
<td>82.46583</td>
<td>135.66754</td>
</tr>
<tr>
<td>Abundance of fawns born (N)</td>
<td>211.2804</td>
<td>18.39878</td>
<td>180</td>
<td>250</td>
</tr>
<tr>
<td>Abundance of fawns recruited (R)</td>
<td>37.1072</td>
<td>5.37916</td>
<td>27</td>
<td>48</td>
</tr>
<tr>
<td>Deviance</td>
<td>7387.6762</td>
<td>36.3908</td>
<td>7323.2555</td>
<td>7460.40928</td>
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<sup>a</sup> Scale parameter of the detection function, which is expected to increase with fawn age and stabilize at asymptote representing home range size of the doe
<sup>b</sup> Detection probability
<sup>c</sup> Number of days after 1 December 2015
<sup>d</sup> Effects on probability of birth location help explain spatial variation in birth density
<sup>e</sup> Measured in days
Figure 2.1. Locations of the 180 passive trail cameras in the Florida Panther National Wildlife Refuge and two units of Big Cypress National Preserve, Bear Island and the North Addition Lands, from 1 December 2015 to 30 June 2016.
Figure 2.2. The major vegetation types within the study area in Florida Panther National Wildlife Refuge and Big Cypress National Preserve in South Florida, USA. The vegetation types were reclassified from FNAI Cooperative Land Cover classification data (FWC/FNAI 2016)
Figure 2.3. One of 123 individual fawns identified using unique spot patterns, showing the distinct groupings of spots on the right (yellow ovals), left (blue ovals), and directly behind (red ovals) used to distinguish individuals.
Figure 2.4. The three camera grids exhibited a range of fire frequencies in the past 20 years (1995 - 2015). FPNWR is under a frequent prescribed fire rotation in designated burn units. The other two units (BI and AL) have received fewer prescribed fires but have been affected by wildfire.
Figure 2.5. The three camera grids exhibited a range of days since fire in the past 30 years. FPNWR tends to have units with very recent fire, due to their frequent prescribed fire rotation. However, the peripheral units have not burned since the land was acquired ~30 years ago. Bear Island and Add Lands both tend to have less frequent fires.
Figure 2.6. The range of estimated birth date windows (grey bars), occurrence of first detection (green circle), subsequent detections (crosses), and last detection (blue circle) for 123 uniquely identified fawns on three study areas in South Florida from 1 December 2015 to 30 June 2016. Vertical dashed lines indicate period of passive camera data collection.
Figure 2.7. Camera locations (+) and number of detections of fawns on 3 study sites in South Florida (N=60 cameras per study grid) during 2016. The size of the blue circle represents the number of independent detections of fawns at that camera trap, where larger circles indicate more detections.
Figure 2.8. The peak shows the distribution of birth dates of all fawns in a 10941-ha study area in South Florida, with a mean of 25 January 2016. The blue dots indicate the estimated birth date of each of the 123 uniquely identified individuals. The red line indicates a previous estimate of peak breeding (7 February) based on fetal morphometric data from the same region.
Figure 2.9. The number of fawns and recruits alive throughout a 10941-ha study area comprising three camera grids in South Florida during fawning season 2016. Recruits are fawns ≥ 180 days.
Figure 2.10. Estimated density of fawn birth locations in each of the four vegetation types, based on posterior mean estimates from the best-fit model in Table 2.2. Vertical lines indicate the 95% CI for each estimate.
Figure 2.11. Estimated density of fawn birth locations in fawns per square km throughout a 53,333-ha study area in South Florida, derived from spatial encounter data of 123 fawns between 1 December 2015 and 30 June 2016.
Figure 2.12. Estimated density of fawn birth locations in fawns per square km throughout the 10,941-ha camera grids in South Florida, derived from spatial encounter data of 123 fawns between 1 December 2015 and 30 June 2016.
Figure 2.13. The effect of fire frequency on fawn birth location density in flatwoods vegetation between 1 December 2015 and 30 June 2016 in South Florida. Each of the other vegetation types show a similar pattern of increasing density of fawn birth locations with increasing frequency of fire. The heavy black line indicates posterior mean, and the light black lines indicate the 95% CI.
Figure 2.14. Fawn survivorship curve derived from analysis of encounter data of 123 uniquely identified individuals in a 10941-ha study area comprising three camera grids in South Florida during fawning season 2016. Survivorship rates at 30 days, 120 days, and 180 days are circled.
Figure 2.15. The significant effects of three continuous variables on the daily detection rate of fawns in South Florida between 1 December 2015 and 30 June 2016, where each estimate
indicates the probability of detecting a fawn when a camera is placed in the center of a fawn’s home range. The heavy black line indicates posterior mean, and the light black lines indicate the 95% CI.
Figure 2.16. The effect of trail status on the daily detection rate of fawns in South Florida between 1 December 2015 and 30 June 2016, where the estimate indicates the probability of detecting a fawn when a camera is placed in the center of a fawn’s home range. The lines indicate the 95% CI around each estimate. This effect was not statistically significant.
CHAPTER 3
CONCLUSIONS, MANAGEMENT IMPLICATIONS, AND FUTURE DIRECTIONS

White-tailed deer populations depend on sufficient fawn production and recruitment to mitigate losses from adult mortality. In South Florida, deer are a highly valued species for recreational hunting, as well as for cultural and aesthetic reasons. Adult deer are also the primary prey of the endangered Florida panther, which is dependent on a viable deer population for its continued recovery (Comiskey et al. 2002, Pimm et al. 2006, FWC 2017). South Florida deer are also depredated by alligators, black bears, bobcats, coyotes, and invasive Burmese pythons (Maehr and Brady 1984, Maehr et al. 1990, Labisky and Boulay 1998, MacDonald-Beyers and Labisky 2005, McCown and Scheick 2007, Boback et al. 2016). Due to these many sources of potential mortality, understanding the productivity and survival of deer fawns in this system is essential to managing this population effectively.

My research found evidence that average fawn survival rate at young ages (<30 days) was greater in South Florida than in many traditional fawn collaring studies throughout the Southeast United States. However, the fawn survivorship curve in South Florida does not follow the pattern seen in other parts of the deer’s range, which is often characterized by a drastic decrease in fawn mortality rate between two to six months (Nelson et al. 2015, Shuman et al. 2017). Instead, I found a consistently high fawn mortality rate to the recruitment age of 180 days, resulting in 18% survival to recruitment across the study site. Fawn mortality risk did not appear to decrease significantly from birth to 180 days. This may be due to the unique predator

I found the highest densities of fawn birth locations in areas that had a higher frequency of fires within the previous twenty years, likely due to the increased forage quality and improved sightlines to reduce predation risk caused by the fires (Cherry et al. 2018). I also found higher fawn densities in areas dominated by cypress. Some types of cypress forests may provide both good cover and open sightlines for does and fawns. I found lower densities of fawns born in hardwood hammocks, which is likely due to the thick vegetation and cover, as well as low-quality forage availability for the doe during that very nutritionally-stressful time. Fawn activity, as measured by fawn detection rate, was reduced during increased levels of human activity and vehicle traffic, indicating that does and fawns decreased or shifted their activity around areas and times of higher human disturbance. Fawn activity increased as standing groundwater rose during my study, possibly due to the increased movement required for fawns and does to access sufficient nutrition during high water events. Overall, my research revealed a unique pattern in fawn survivorship in the South Florida population and estimated the fawn survival to recruitment rate (18%) for the first time in over 20 years.

This research demonstrated a novel non-invasive methodology, which ensures that the research had no adverse effects on doe and fawn activity, survival, or movement throughout the study. Future tests of this non-invasive methodology should be designed with a concurrent fawn collaring study to assess if both methods provide similar estimates of survival to the recruitment rate and to reveal any biases associated with this new methodology.
Future research in this system should explore the effects of environmental variables on fawn survival rate. While I estimated age-specific fawn survival, I did not evaluate landscape-level influences of vegetation and cover, fire history, human disturbance, or standing water level on spatial variation in individual fawn survival rate. Additionally, multi-year studies are particularly necessary to account for the annual variation in fawn recruitment, as well as to provide a source of repeatability for recruitment estimation. My research was conducted in an extremely wet fawning season, which may have greatly reduced the fawn survival rate and may provide a worst-case-scenario estimate. I encourage continued work in this system to more thoroughly understand the average survival and recruitment process of white-tailed deer fawns in South Florida to result in the most effective management strategy.

The results of my research may be helpful to explore potential management actions, such as increasing the frequency of prescribed fire, which may benefit fawn survival and recruitment. However, continued monitoring of both adult mortality rates and fawn survival to recruitment rates using multi-year studies is necessary to fully understand the viability of this population as both a recreational resource and a prey base for the Florida panther.

LITERATURE CITED


Supplemental Table 1. We cropped FNAI Cooperative Land Cover version 3.2 Raster data around our study site and reclassified the 72 FNAI vegetation types into 7 major categories. Listed are the original classifications and our new reclassification title. The first 4 vegetation types (Open, Cypress, Flatwoods, and Hammock) were incorporated into our analyses.

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