THE IMPACTS OF URBANIZATION ON

ENDANGERED FLORIDA KEY DEER

A Dissertation

by

PATRICIA MOODY HARVESON

Submitted to the Office of Graduate Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

December 2005

Major Subject: Wildlife and Fisheries Sciences

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ABSTRACT

The Impacts of Urbanization on Endangered Florida Key Deer.

(December 2005)

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Conservation of native wildlife is becoming increasingly difficult due to continued human population growth and expansion. As the human population continues to increase, so does the rate of consumption of our natural resources. As competition for resources between man and wildlife continues, it is important to understand the effects of urbanization on species. Endangered Key deer (*Odocoileus virginianus clavium*) are endemic to the Florida Keys archipelago stretching southwest off the southern tip of peninsular Florida. Key deer range is restricted to the Lower Florida Keys with approximately 60% residing on Big Pine Key and 15% residing on No Name Key which have undergone rapid human population growth and development over the past 30 years. Urban development and its associated risk factors (i.e., habitat loss and fragmentation, deer domestication, and deer–vehicle collisions) have been cited as the greatest threat to the Key deer population. For my dissertation research, I evaluated the impacts of 30 years of development on the Key deer population. My results suggest that increased habitat fragmentation and increased road traffic have created areas of varying habitat quality and mortality risk and have resulted in a source-sink system for Key deer on Big

Pine Key. In my examination of Key deer metapopulation dynamics, I found a low probability of deer colonizing 2 target outer islands (Sugarloaf and Cudjoe) through dispersal alone in the next 20 years. Further, I examined the impacts of urbanization on changes in Key deer population dynamics, behavior, and morphology. Collectively, my results suggest that over the past 30 years Key deer have become more urbanized, which in turn has influenced Key deer behavior and population viability. Behavioral adaptations due to deer plasticity appear to have provided Key deer with mechanisms to persist in a changing environment due to urbanization. However, the future ability of Key deer to persist in a continuously urbanizing environment cannot be predicted. At some threshold, urban development would become unsustainable, and, unlike other forms of habitat change or environmental disturbances, urban development is in most cases irreversible, requiring careful planning in habitat conservation strategies.

DEDICATION

To my children:

Katie and Jake

ACKNOWLEDGEMENTS

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

CHAPTER I

INTRODUCTION

The dissertation is divided into chapters, each of which represents an independent, stand-alone paper with a distinct research focus. While each chapter has its own unique research objectives, the purpose of each is the same: to increase our understanding of the impacts of urbanization on Key deer (*Odocoileus virginianus clavium*) population ecology. Thus, some information is repeated among chapters (i.e., study area description, Key deer background, scientific names).

Key deer are an endangered subspecies of white-tailed deer endemic to the Florida Keys. The majority of Key deer (approximately 75%) reside on Big Pine Key and No Name Key (Lopez 2001). Over the last 30 years, Big Pine and No Name Keys have experienced a 10-fold increase in human population growth and urban development (Monroe County Growth Management Division 1992). Urban development and its associated risks are considered the greatest threat to Key deer (Lopez et al. 2003). The overall goal of my dissertation research was to better understand the population dynamics of Key deer. Each chapter was designed to answer specific biological and conservation questions about Key deer and the impacts of urbanization on the population. While each chapter is written as an independent paper, they are all related in that they address the current gaps in our knowledge and answer the most pertinent questions relevant to Key deer conservation.

The format and style follow the Journal of Wildlife Management.

In chapter II, I examined the effects of urbanization and fragmentation on the habitat and population dynamics of Key deer on Big Pine Key (BPK), Florida. Increased human population growth on BPK, has resulted in increased development of houses and roadways. However, most of the high traffic volume and deer-vehicle collisions occur in the southern portion of the island. My goal was to assess the habitat quality differences and associated risk factors on the Key deer population. Specifically, I tested the research hypothesis that north BPK (NBPK) was a source and south BPK (SBPK) was a sink by comparing (1) Key deer viability for both populations using a population model, (2) Key deer dispersal between NBPK and SBPK and its effects on population viability, and (3) habitat quality between NBPK and SBPK and its associated risks to Key deer.

In chapter III, I examined the possibility of Key deer dispersing and creating additional viable populations on islands within their current range which is a conservation objective of the U.S. Fish and Wildlife Service – National Key Deer Refuge. The majority of Key deer are concentrated on 2 islands (Big Pine and No Name Keys) making them vulnerable to environmental catastrophes such as hurricanes. My research objectives were to develop a metapopulation model for Key deer and to evaluate the probability of deer colonization of peripheral islands through natural dispersal. Specifically, my objectives were to (1) evaluate the effects of distance and dispersal rate on Key deer island subpopulations, and (2) estimate the probability of Key deer colonizing surrounding islands with a viable population.

In chapter IV, I evaluated an alternative method for estimating survival of Key deer. Annual survival is an important population parameter that influences population growth and is a key component in simulation modeling. Reliable and cost-effective methods of obtaining survival estimates are necessary to make conservation decisions based on model-predicted population trends. My research objective was to evaluate the use of life table survival estimates for Key deer using roadkill data. Specifically, my objectives were to (1) calculate Key deer survival using roadkill data (life table), and (2) compare these survival estimates to previously published survival estimates calculated from radiotelemetry data.

In the last chapter, I evaluated the impacts of 30 years of urban growth and development in the Keys on the Key deer population. I examined changes in Key deer population dynamics, behavior, and morphology from 1970 to 2000. Specifically, I evaluated whether urbanization has negatively impacted Key deer by testing the following research hypotheses: (1) Key deer have become more urbanized due to increased human population growth and development; (2) increased urbanization and anthropogenic risk factors have negatively impacted Key deer population dynamics; (3) increased urbanization has resulted in changes in Key deer behavior and increased domestication; and (4) increased urbanization has resulted in long-term physiological effects on Key deer morphology.

CHAPTER II

SOURCE–SINK DYNAMICS

SYNOPSIS

The endangered Florida Key deer (*Odocoileus virginianus clavium*) is endemic to the Florida Keys, Florida, with Big Pine Key (BPK) supporting the majority (approximately 60%) of the population. Habitat loss and fragmentation have altered the amount of available habitat creating areas of varying suitability; north BPK (NBPK) is believed to contain more optimal habitat as compared to south BPK (SBPK), which is more developed and fragmented. I evaluated the source–sink dynamics of Key deer using a sex- and stage-structured, stochastic matrix model. Model results indicated the NBPK population of Key deer was increasing ($\lambda = 1.02$), whereas the SBPK population was decreasing ($\lambda = 0.87$). Without dispersal from the north, the SBPK deer population has a 97% probability of falling below 25 individuals (quasi-extinction threshold) in the next 20 years. The higher risk to Key deer in SBPK can be explained by relative habitat quality differences between the 2 areas. House density, amount of roads, number of fences, and amount of development were all greater in SBPK. Collectively, study results indicate that SBPK can be described as an ecological sink with a nonviable population supplemented by deer dispersal from NBPK (source). Care should be taken to preserve the source population and its habitat. Thus, I propose limiting future development in NBPK (high-quality source habitat). The US 1 highway corridor project has the potential to decrease Key deer mortality due to vehicle collisions, and I recommend that

future management goals continue to address mortality factors on SBPK (low-quality sink habitat).

INTRODUCTION

Various theoretical models are used to aid conservationists in understanding population dynamics in heterogeneous and fragmented landscapes (Pulliam 1988, Akçakaya 2000, Morris 2003). Source–sink models can occur in systems where different demographic rates are attributed to different quality habitats (Holt 1985, Pulliam 1988, Dias 1996). Habitats of high quality (sources) yield a demographic surplus (births > deaths) whereas habitats of low quality (sinks) yield a demographic loss (deaths > births; Dias 1996). In a source–sink system, dispersal from a source population to a sink is necessary to prevent the sink population from going extinct. Although widely accepted, source–sink dynamics are difficult to quantify with most examples represented by plant, avian, or small mammal species (Watkinson and Sutherland 1995, Dias 1996, Diffendorfer 1998, Walters 2001). Some debate has occurred over the validity of source–sink claims in the literature where methods have been questioned and other factors suggested such as pseudo-sinks, maladaptive responses, and various dispersal mechanisms (Watkinson and Sutherland 1995, Diffendorfer 1998, Remes 2000). Despite the problems in identifying source–sink dynamics, attempting to understand their function in fragmented landscapes is imperative to the overall recovery and management of endangered species.

Florida Key deer, the smallest subspecies of white-tailed deer in the United States, are endemic to the Florida Keys on the southern end of peninsular Florida

(Hardin et al. 1984). Key deer occupy 17 islands in the Lower Keys with the majority (approximately 60%) residing on BPK (Fig. 2.1) (Lopez 2001). Over the last 30 years, BPK has experienced a 10-fold increase in human population growth and urban development (Monroe County Growth Management Division 1992). Urban development and its associated risks are considered the greatest threat to the Key deer population (Lopez et al. 2003). These risks include habitat loss and fragmentation, deer domestication, and deer–vehicle collisions (Hardin 1974, Folk and Klimstra 1991*b*, Folk 1992, Lopez et al. 2003). In a recent study, Lopez et al. (2003) reported that deer–vehicle collisions accounted for 50% of the total Key deer mortality on BPK. They reported that increases in urban development and habitat fragmentation contributed to higher, yet variable, risks for Key deer with the greatest risk in SBPK, which has greater development, as compared to NBPK, which has more protected land (Fig. 2.1) (Lopez et al. 2003). Furthermore, Lopez et al. (2003) also suggested that NBPK, with its high-quality habitat and high Key deer densities, was a potential source for SBPK, which was characterized as low-quality habitat with low Key deer densities.

I examined the impacts of urbanization and fragmentation on the population dynamics of Key deer. While previous research has reported a difference in mortality rates by area (Lopez et al. 2003), I explore the impacts of these rates to the viability of the Key deer subpopulations (NBPK and SBPK) and the overall population on BPK. I also explored the differences between habitat on NBPK and SBPK to identify risk factors influencing Key deer mortality rates.
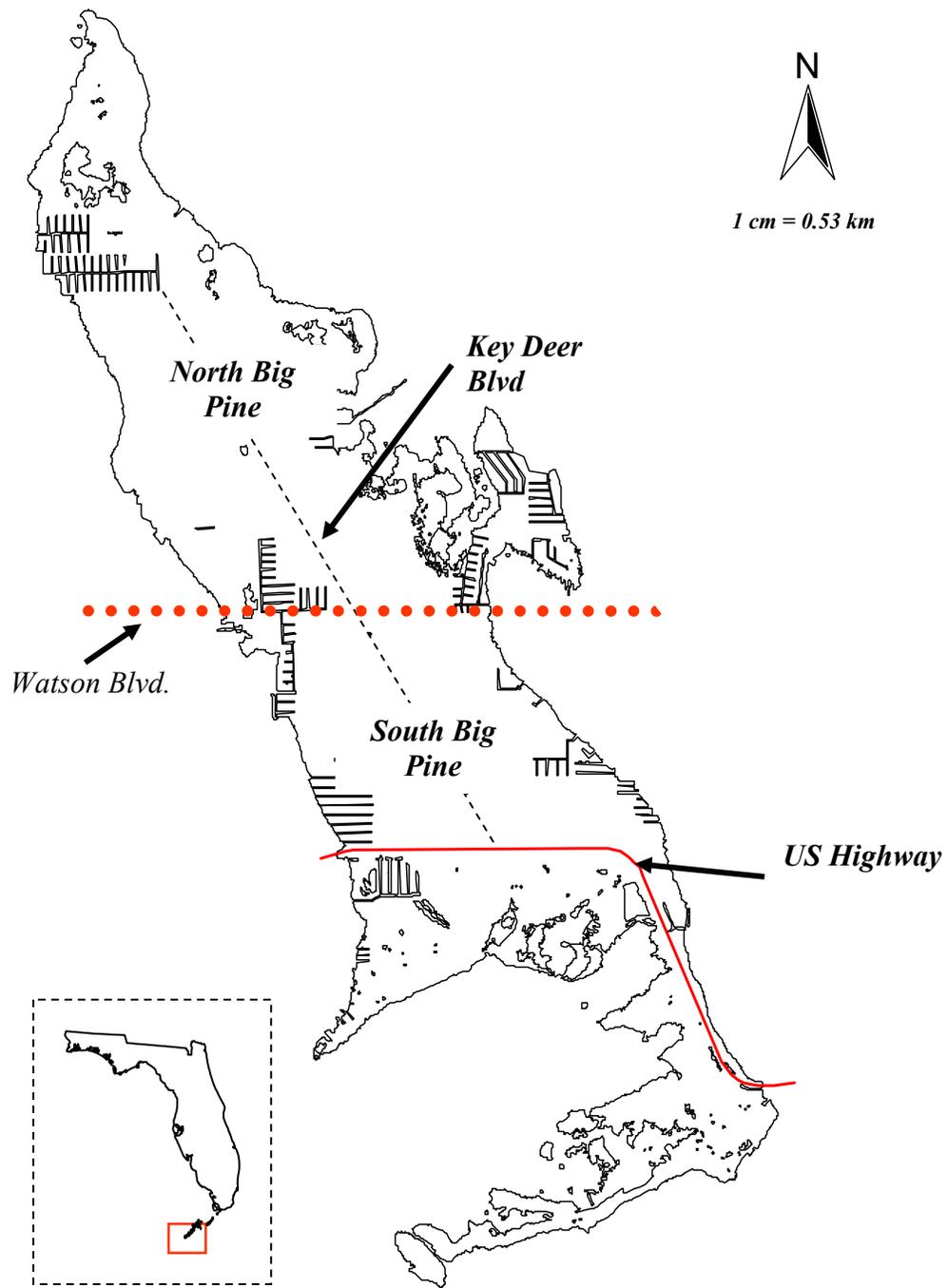
Fig. 2.1. Big Pine Key, Florida, with Watson Boulevard dividing north Big Pine Key (NBPK) and south Big Pine Key (SBPK).

I tested the research hypothesis that NBPK was a source and SBPK was a sink by comparing (1) Key deer viability for both populations using a population model, (2) Key deer dispersal between NBPK and SBPK and its effects on population viability, and (3) habitat quality between NBPK and SBPK and its associated risks to Key deer (Boyce 1992, Burgman et al. 1993, Akçakaya 2000). I predicted that the greater developed and fragmented habitat on SBPK was a sink that would not be viable in the absence of dispersal from NBPK.

STUDY AREA

The Florida Keys are a chain of small islands located southwest of the southern tip of peninsular Florida in Monroe County, Florida. Key deer range includes 17 islands in the Lower Florida Keys comprising a total of 9,836 ha. Big Pine Key (Fig. 2.1) is the largest island (2,531 ha) within this range. Watson Boulevard runs across the middle of BPK from east to west dividing the island in half (Fig. 2.1). The area north of Watson Boulevard is defined as NBPK (1,238 ha), and the area located to the south of Watson is defined as SBPK (1,293 ha). Island soil types vary from marl deposits to bare rock of the oolitic limestone formation (Dickson 1955). Vegetation varies by elevation with red (*Rhizophora mangle*), black (*Avicennia germinans*), and white mangroves (*Laguncularia racemosa*), and buttonwood (*Conocarpus erectus*) forests occurring near sea level (maritime zones). As elevation increases inland, maritime zones transition into hardwood (e.g., Gumbo limbo [*Bursera simaruba*], Jamaican dogwood [*Piscidia piscipula*]) and pineland (e.g., slash pine [*Pinus elliottii*], saw palmetto [*Serenoa*

repens]) upland forests with vegetation intolerant of salt water (Dickson 1955, Folk 1992).

METHODS

Model Overview

I modeled the NBPK and SBPK deer populations with a sex- and stage-structured, stochastic matrix model using RAMAS Metapop (Akçakaya 2002). I classified Key deer into 3 age classes: fawn (<1 year), yearling (1–2 year), and adult (≥2 year) (Lopez et al. 2003). I was limited to 3 age classes due to the difficulty in aging deer beyond 3 years (Jacobson and Reiner 1989, Dimmick and Pelton 1994). I used density independence in the model to provide a conservative assessment in population growth when populations were below carrying capacity (Ginzburg et al. 1990). Model parameter estimates were taken from comprehensive Key deer studies from December 1968 to June 1972, and January 1998 to December 2000 on BPK (Hardin 1974, Silvy 1975, Lopez 2001). I used sensitivity and elasticity estimates to examine the effects of each parameter on model results (Caswell 2001).

Model Parameters

*Survival.—*Lopez et al. (2003) estimated Key deer survival and variance estimates from radiomarked animals by sex, age, and area using a known-fate model framework in Program MARK (Table 2.1) (White and Burnham 1999). For fawn survival, I used conservative estimates reported by Lopez (2001) adjusted to compensate for presumed overestimation reported by Lopez et al. (2003).

Fecundity.—Fecundity was estimated based on Key deer necropsy data (Hardin 1974, Lopez 2001). Annual deer maternity was 1.05 fawns per breeding female (yearlings and adults; Hardin 1974). Key deer <1 year-of-age were not reproductively active (Hardin 1974, Folk and Klimstra 1991*a*). Furthermore, Hardin (1974) reported fetal sex ratios were male biased (59% males) in the Key deer population. From these data, fecundity estimates for yearlings ($Fy = RMSy$) and adults ($Fa = RMSa$) were determined as described by Akçakaya et al. (1999), where R is equal to the female fetal sex ratio, M is equal to maternity, and Sy and Sa are equal to yearling and adult survival, respectively (Table 2.1).

Initial Abundances.—Initial abundances used in model simulations were determined from mark–resight estimates based on 247 road count surveys conducted from 1998 to 2001 on BPK (Lopez et al. 2004*a*). A stable age distribution was assumed for both populations with 299 deer in NBPK (107 fawns, 49 yearlings, and 143 adults) and 106 deer in SBPK (39 fawns, 20 yearlings, and 47 adults; Akçakaya 2002).

Dispersal.—I defined dispersal as the permanent movement of a deer from its birth place to the place where it reproduced (Caughley and Sinclair 1994). Key deer dispersal primarily occurs in the yearling age class (Silvy 1975, Lopez 2001). I assumed that dispersal for Key deer would occur following the transition of fawns into the next age class (as yearlings); therefore, using radiotelemetry data, I identified animals radiomarked as fawns or young yearlings with ≥20 locations from 1998 to 2000. I examined movement of yearlings from location as fawns to location as adults. Key deer dispersal was estimated from 36 deer (19 M, 17 F) between NBPK and SBPK.

Table 2.1. Annual model parameter estimates and standard errors for Florida Key deer by sex and age on north Big Pine Key (NBPK) and south Big Pine Key (SBPK), Florida, 1968–1972 and 1998–2000.

Parameter	Sex	Age	NBPK		SBPK	
			Mean	SE	Mean	SE
Survival[a]	F	Fawn	0.470	0.061	0.470	0.061
		Yearling	0.848	0.033	0.710	0.082
		Adult	0.848	0.033	0.710	0.082
	M	Fawn	0.470	0.061	0.470	0.061
		Yearling	0.583	0.060	0.412	0.099
		Adult	0.583	0.060	0.412	0.099
Fecundity[b]	F_f	Yearling	0.365	0.087	0.306	0.096
		Adult	0.365	0.087	0.306	0.096
	F_m	Yearling	0.525	0.126	0.440	0.139
		Adult	0.525	0.126	0.440	0.139

[a] Survival estimates from Lopez (2001) and Lopez et al. (2003).

[b] Fecundity estimates represent recruitment by sex class (e.g., F_f = female recruitment in model, F_m = male recruitment in model; Hardin 1974, Lopez 2001).

Environmental and Demographic Stochasticity

Environmental Stochasticity.—Environmental fluctuations in natural populations often result in unpredictable and variable vital rates (i.e., survival, fecundity; Akçakaya 2000). Environmental stochasticity can be incorporated into a model through matrix selection (randomly sampling from matrices based on vital rates in good and bad years) or by randomly sampling vital rates from normal distributions based on the mean and variance of each rate (Akçakaya 1991, Akçakaya 2000). I was unable to calculate vital rate matrices for good and bad years (due to lack of data), thus, survival and fecundity rates were varied based on a mean stage matrix and a standard deviation matrix for each area (Table 2.1).

Demographic Stochasticity.—Akçakaya (2000) recommended using demographic stochasticity in population models for rare species. I incorporated demographic stochasticity in model simulations by sampling the number of survivors and the number of individuals dispersing in a local population from a binomial distribution. Since the average number of offspring per female Key deer is >1 (1.05; Hardin 1974), I could not sample from a binomial distribution but instead assumed the data followed a Poisson distribution (Akçakaya 1991, Akçakaya 2000).

Model Use and Risk

The finite rate of increase (λ) is the proportional population growth under stable distribution, no density dependence, no stochasticity, and no dispersal (Akçakaya 2002). A value >1 indicates the population is growing, while a value <1 indicates the population is declining. I used 3 measures to evaluate the viability of both Key deer

populations: (1) the λ of each population, (2) the risk of each population going extinct in 20 years, and (3) the risk of each population falling below 25 individuals (quasi-extinction) in 20 years (Akçakaya 2000). I also examined the effects of dispersal on each population and its projected growth or decline. I simulated the population dynamics of Key deer for all of BPK, NBPK (without dispersal), and SBPK (without dispersal). For each local population, I ran 10,000 simulations over a 20-year period.

Habitat Quality

I evaluated 6 relative indicators of habitat quality for NBPK and SBPK: (1) number of houses, (2) amount of roads (km), (3) amount of fenced area (ha), (4) amount of developed land (ha), (5) amount of preferred habitat for Key deer (ha), and (6) amount of avoided habitat for Key deer (ha). Lopez (2001) reported Key deer preferred upland areas (hammock, pineland, and developed) and avoided lowlands (freshwater marsh, buttonwood, and mangrove). Existing spatial data (MacAulay et al. 1994, Lopez 2001) were quantified in ArcView (ESRI 1999). I also summarized mortality data collected by National Key Deer Refuge (NKDR) biologists from direct sightings, citizen reports, or observation of turkey vultures (*Cathartes aura*) from 1990 to 2000 (Lopez et al. 2003).

RESULTS

As predicted, model results suggest the Key deer population increased (λ = 1.02, variance = 0.015) on NBPK whereas the deer population on SBPK declined (λ = 0.87, variance = 0.017). Differences in λ are reflected in population trajectories for NBPK and SBPK (Fig. 2.2). The model predicted a low terminal extinction risk (<1%) for BPK and NBPK deer. However, when SBPK was modeled separately, terminal extinction

risk increased to 24% (Fig. 2.3). I found similar trends in risk of quasi-extinction. Overall, the model predicted a low risk (<1%) of quasi-extinction for the BPK population.

Estimated Key deer dispersal differed between sex and area (NBPK: female = 0%, male = 33%; SBPK: female = 0%, male = 10%). In the absence of dispersal from NBPK to SBPK, risk of quasi-extinction for SBPK deer was 97% (Fig. 2.3). Model sensitivity and elasticity estimates indicated that adult female survival had the most influence on model matrices. Results indicated that the models were least sensitive to male input parameters and were most sensitive to adult female survival. Sensitivity results for females were as follows: NBPK yearling fecundity = 0.07, adult fecundity = 0.33, fawn survival = 0.31, yearling survival = 0.14, adult survival = 0.72; SBPK yearling fecundity = 0.09, adult fecundity = 0.37, fawn survival = 0.29, yearling survival = 0.16, adult survival = 0.68.

The higher risk of extinction for SBPK deer in the model can be explained by relative habitat quality differences between the 2 areas. For most variables, SBPK contained poorer habitat quality and greater risk factors than NBPK (Table 2.2). Preferred and avoided habitats occurred in almost equal amounts on NBPK and SBPK. However, a greater proportion of preferred habitat on SBPK was developed (31%) as compared to NBPK (16%). Of 836 mortalities recorded on BPK between 1990 and 2000, more occurred on SBPK (576, 69%) than on NBPK (260, 31%).
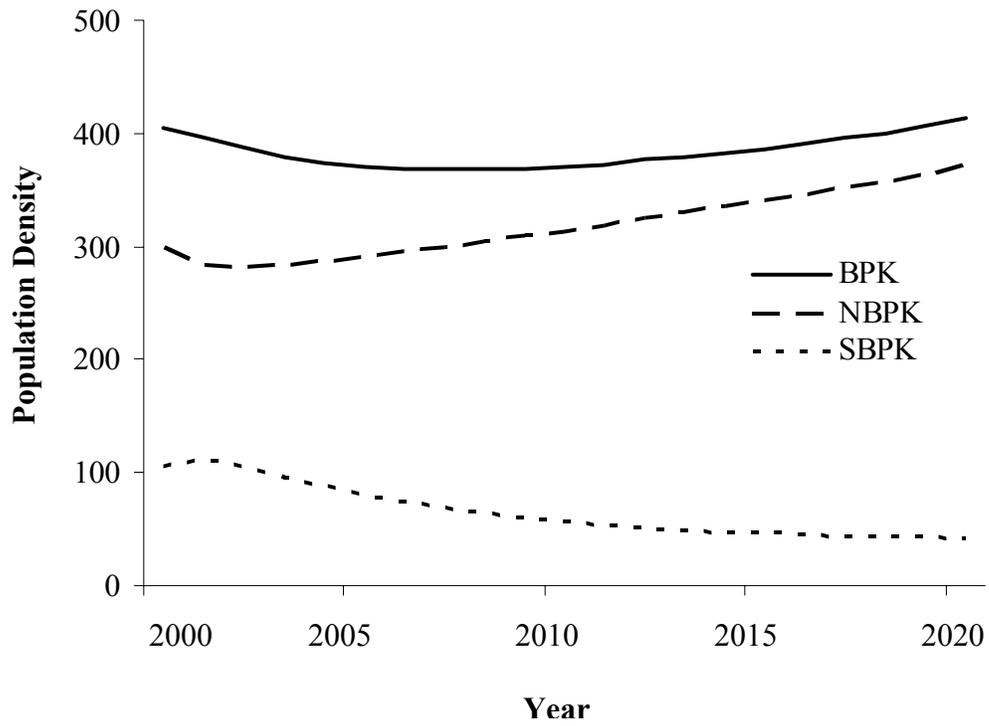
Fig. 2.2. Simulated population trajectory for Florida Key deer on Big Pine Key (BPK),

north Big Pine Key (NBPK), and south Big Pine Key (SBPK), Florida, 2000–2020.
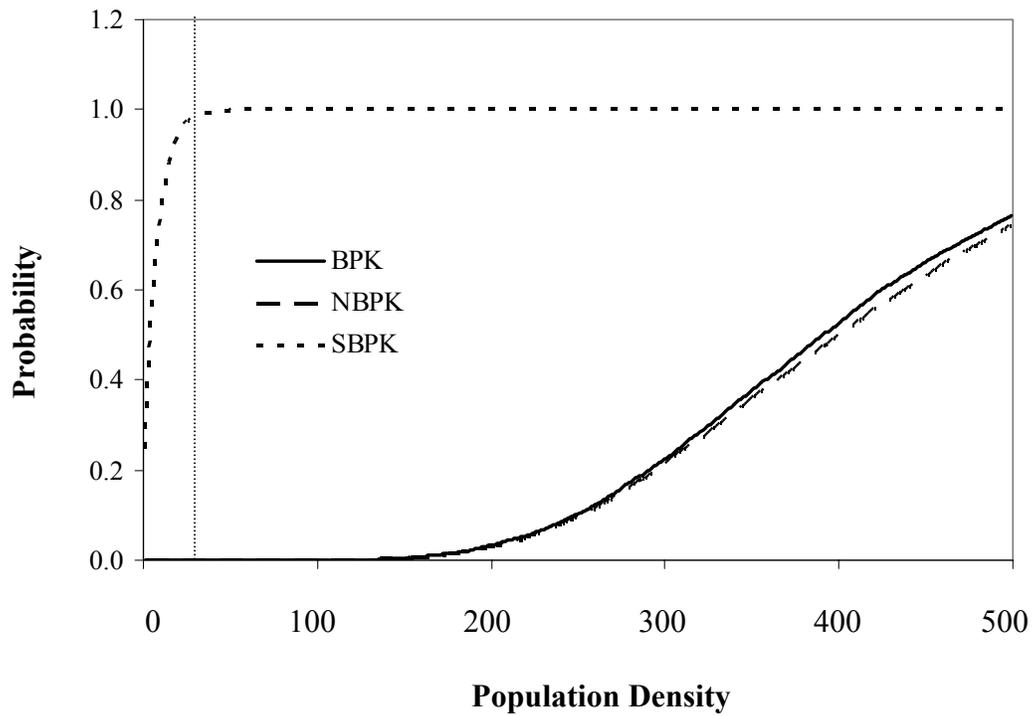
Fig. 2.3. Risk curve (probability of falling below 25 individuals in 20 years, vertical

dotted line) by population abundance for Key deer on Big Pine Key (BPK), north Big

Pine Key (NBPK, no dispersal to south), and south Big Pine Key (SBPK, no dispersal to

north), Florida, 2000–2020.

Table 2.2. Anthropogenic factors, habitat availability, and associated risks to Florida Key deer on north Big Pine Key (NBPK) and south Big Pine Key (SBPK), Florida, 2000.

Risk	NBPK		SBPK		Area of greater risk
	No.	%	No.	%	
Houses	1,082	40.4	1,597	59.6	SBPK
Roads (km)	51	40.2	75	59.8	SBPK
Fenced area (ha)	31	34.7	58	65.3	SBPK
Developed land (ha)	205	33.6	405	66.4	SBPK
Preferred habitat[a] (ha)	703	49.5	717	50.5	Same
Avoided habitat[b] (ha)	535	48.1	576	51.9	Same

[a] Preferred habitat includes pineland, hammock, and developed areas (Lopez 2001).

[b] Avoided habitat includes freshwater marsh, buttonwood, and mangrove areas (Lopez 2001).

DISCUSSION

Source–Sink Dynamics

Three elements are necessary in identifying source–sink systems: (1) heterogeneous habitat and habitat-specific demographic rates; (2) active or passive dispersal; and (3) λ in source habitats should be >1, whereas λ in sink habitats should be <1 (Holt 1985, Pulliam 1988, Dias 1996, Diffendorfer 1998). Model results support the hypothesis that the Key deer population on BPK is a source–sink system due to anthropogenic factors.

First, Key deer occupy a limited range within a highly fragmented landscape. Analyses indicate habitat on BPK is divided between areas of high quality (NBPK) and low quality (SBPK) (Table 2.2). Urban development, including houses, fences, and roads (which present the greatest mortality risk to Key deer [Lopez et al. 2003]), was higher for SBPK. Furthermore, the majority (69%) of Key deer mortalities occurred on SBPK despite higher deer densities reported by Lopez (2001) on NBPK. I attribute the greater risk on SBPK to low habitat quality, greater amount of urban development, and greater amount of roads, which pose an especially high mortality risk to Key deer because of the high traffic volume on US 1 highway (Lopez et al. 2003). As a result, I propose differing demographic rates (i.e., survival and fecundity) observed in the study are attributed to differences in habitat quality between SBPK and NBPK.

Second, Key deer are active dispersers selecting habitat based on a variety of density-dependent factors such as availability, competition, and reproductive opportunities (Hardin 1974, Lopez 2001). Active dispersers select habitats based on

differences in habitat quality, and individuals may choose to leave a source habitat whenever their expected reproductive success is higher in the sink (Pulliam 1988, Dias 1996, Diffendorfer 1998). Key deer on BPK dispersed from areas of high fitness and low reproductive opportunity (NBPK) to areas of low fitness and high reproductive opportunity (SBPK). I came to this conclusion because (1) dispersal was higher from NBPK than SBPK, and (2) only male deer dispersed. Female deer did not lack reproductive opportunities on NBPK and thus remained in the source where survival was greatest. High deer densities on NBPK reduced opportunities for yearling males to reproduce, and I hypothesize they dispersed to SBPK for greater reproductive opportunities (Hardin 1974, Silvy 1975, Lopez 2001). While the conservative nature of the dispersal estimates were chosen to quantify overall immigration and emigration of subpopulations, I should note the lack of female dispersers could be a result of the small sample sizes used in estimating dispersal. Therefore, dispersal may be underestimated and other explanations such as maladaptive response or despotism might be the actual or contributing underlying factors to Key deer dispersal (Remes 2000, Conradt and Roper 2003).

Third, habitat-specific demographic rates and active dispersal from source to sink habitat collectively result in varying fitness between local populations (Holt 1985, Pulliam 1988, Dias 1996, Diffendorfer 1998). Assuming λ as a measure of local population fitness (Caughley 1977), model results indicated rate of increase for NBPK is >1 ($\lambda = 1.02$) and for SBPK is <1 ($\lambda = 0.87$). This supports the premise that NBPK is a source whereas SBPK is a sink. Sink habitats, however, can be categorized as absolute

or pseudo-sinks (Watkinson and Sutherland 1995). In an absolute sink, the removal of immigration would result in extinction, and in a pseudo-sink, the removal of immigration would only result in population decrease and not extinction. The Key deer model for SBPK indicates that without dispersal from NBPK, the population has a 25% probability of extinction in 20 years (Fig. 2.3). These results suggest the SBPK population is an absolute sink supplemented by dispersal from NBPK. In contrast, NBPK is characterized by high-quality habitat and higher survival and fecundity rates. The model indicates the Key deer population on NBPK is increasing despite emigration and can be considered a source.

MANAGEMENT IMPLICATIONS

Collectively, study results indicate that SBPK can be described as an ecological sink supplemented by deer dispersal from NBPK. Dias (1996) noted that in a source–sink system, preserving only sink habitats will likely lead to population extinction. Care should be taken, especially when dealing with endangered species such as the Key deer, to preserve the source population and its habitat. Thus, I propose limiting future development in NBPK (high-quality source habitat). Future management goals should continue to address mortality factors on SBPK while still recognizing and preserving the important source population in NBPK. The US 1 highway corridor project, which includes the construction of fences and underpasses, has the potential to reduce Key deer mortality in SBPK due to vehicle collisions by 10% annually. This increase in survival could possibly stabilize ($\lambda = 1.0$) the SBPK deer population so it would no longer be a sink.

CHAPTER III

METAPOPULATION DYNAMICS

SYNOPSIS

Metapopulation dynamics of species that occur in patchy or island subpopulations are an important consideration in the conservation of endangered species. Key deer (*Odocoileus virginianus clavium*) are endemic to the Florida Keys and occur on 11 island-complexes in the Lower Keys from Big Pine Key to Sugarloaf Key. While deer numbers have increased notably, the majority of the population occurs on 2 of the islands, Big Pine and No Name Keys. Deer dispersal between islands is possible due to short distances between islands and shallow water. Key deer have been documented to actively disperse between islands but at very low rates (11% males, 3% females). However, increased population densities could possibly increase dispersal rates as island populations on Big Pine and No Name Keys approach carrying capacity. I examined the probability of deer colonization of peripheral islands using a sex- and stage-structured metapopulation model. My objectives were to (1) evaluate the effects of distance and dispersal rate on Key deer island subpopulations, and (2) estimate the probability of Key deer colonizing surrounding islands with a viable population. Results suggest that over the next 20 years, the Key deer population could colonize 6 of the 11 island-complexes with viable populations. However, of the remaining 5 islands, 3 lack the resources to support a minimum viable population; and while Cudjoe and Sugarloaf Keys have the potential to support >200 deer each, they are not projected to increase to above 20 deer by 2020 regardless of dispersal rate due to distance from source population.

INTRODUCTION

Metapopulation theory is commonly applied in the understanding and conservation of endangered species by wildlife conservationists. A metapopulation is a collection of local populations occupying separate patches of habitat in a landscape linked by emigration and immigration (Levin 1979, Meffe and Carroll 1997). Though the concept of metapopulation has been studied in the past (e.g., Howe et al. 1991, Rolstad 1991, Wootton and Bell 1992, Akçakaya et al. 1995, Dias 1996, Donovan et al. 1995, Hanski 1997), few studies have evaluated the metapopulation dynamics of large and long-lived animals (Beier 1993, Doak 1995, Harrison and Taylor 1997, Gaona et al. 1998, Walters 2001), particularly in landscapes fragmented due to urban development. Landscape fragmentation and habitat deterioration typically result in the establishment of new metapopulations with varying rates of movement between subpopulations (Hanski 1997). The rate of emigration and immigration between subpopulations depends on the species ability to disperse and the juxtaposition of patches. Dispersal plays an important role in the metapopulation dynamics of a species, and can be altered with dramatic landscape changes (e.g., urban development).

The Florida Keys archipelago is a collection of island habitat patches occupied by the endangered Florida Key deer. Previous researchers have described the Key deer as a metapopulation comprised of local island subpopulations in the Lower Florida Keys (Lopez 2001). Deer dispersal between islands is possible due to short distances and shallow water; however, little is known about the role of dispersal in the Key deer population. Lopez (2001) reported that Key deer actively dispersed between Big Pine

and No Name Keys but at very low rates (<10% males, <3% females; Lopez 2001). It is unknown whether Key deer disperse to other islands or at what rates. Understanding Key deer dispersal and its effects on the dynamics of the metapopulation is essential for the management of this endangered species. For example, the colonization of peripheral islands is a necessary step in the recovery of Key deer (USFWS 1999). By modeling Key deer metapopulation dynamics, I examined the possibility of future colonizations under various dispersal strategies.

Social animals, such as white-tailed deer, form matrilineal groups where females remain in their natal area. While previous research has shown varying degrees of male deer dispersal, female dispersal is usually rare, even during times of high population density and low reproductive fitness (Greenwood 1980, Halls 1984, Clutton-Brock et al. 1985, Lopez 2001). However, Albon et al. (1992) studied an island metapopulation of red deer and found that as population density increased, family bonds began to break down and dispersal increased. Other research has suggested that Key deer lack strong philopatry (family ties) exhibited by other white-tailed deer due to the absence of predators and migration (Hardin et al. 1976) and these weaker social bonds may enhance dispersal of Key deer to other islands. Thus, I examined the effects of various dispersal scenarios on Key deer metapopulation dynamics. Specifically, I evaluated the probability of deer colonization of peripheral islands using a sex- and stage-structured metapopulation model. My objectives were to (1) evaluate the effects of distance and dispersal rates on Key deer island subpopulations, and (2) estimate the probability of Key deer colonizing surrounding islands with a viable population.

BACKGROUND INFORMATION

Endangered Key deer are endemic to the Florida Keys and occur on 11 island-complexes in the Lower Florida Keys from Big Pine Key to Sugarloaf Key (Fig. 3.1) (Hardin et al. 1984). An island-complex is a collection of islands in close proximity to each other separated by shallow waters. Islands within a complex are "bridged" together during low tides (i.e., sea bottom is exposed), thus, can be considered to be a functionally single island (Folk 1992). The majority of Key deer (approximately 75%) reside on Big Pine Key and No Name Key (Lopez 2001). Over the last 30 years, Big Pine and No Name Keys have experienced a 10-fold increase in human population growth and urban development (Monroe County Growth Management Division 1992). Urban development and its associated risks are considered the greatest threat to Key deer (Lopez et al. 2003). Key deer are also at risk to environmental catastrophes such as hurricanes (Lopez et al. 2000). While the Key deer populations on these 2 islands have increased, the majority of the metapopulation occupies a small geographic area. The establishment of additional deer populations on other islands is a management goal of the U.S. Fish and Wildlife Service (USFWS) and a necessary step in Key deer recovery (USFWS 1999).

MODEL OVERVIEW

The model represents the dynamics of the Key deer metapopulation in the 11 island-complexes of the Florida Keys. The model consists of 11 submodels (one for each island-complex). The model parameters are based on the estimates from the main island, Big Pine Key. Each submodel is identical to the main model except for the initial

population abundance and carrying capacity. The model is driven by the dispersal rate of male and female yearling and adult deer from the main island. Dispersal rates for each of these sex and stage classes are held constant for all 11 island-complexes. Dispersal from the main island, Big Pine Key, flows out to the 5 island-complexes surrounding it in a "stepping-stone" fashion following a tier-system (Fig. 3.2). Dispersal between tiers can only occur in successive, ascending fashion (i.e., dispersal from tier 1 would occur to tier 2 followed by tier 3, etc.).
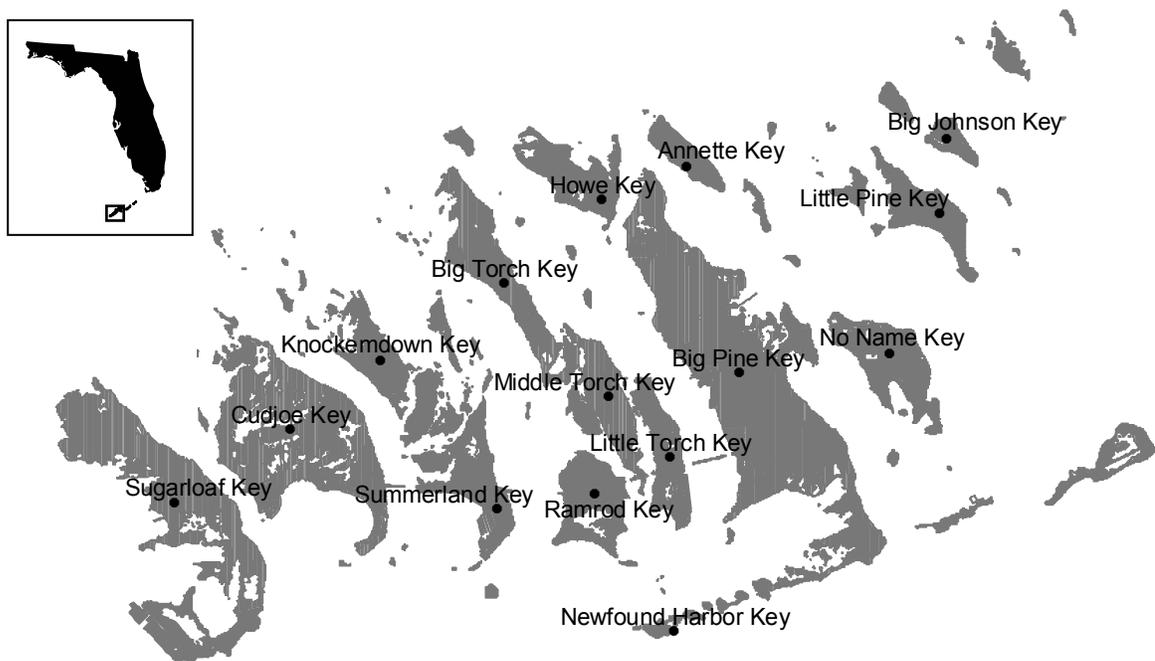
Fig. 3.1. Map of the Lower Florida Keys, Florida.
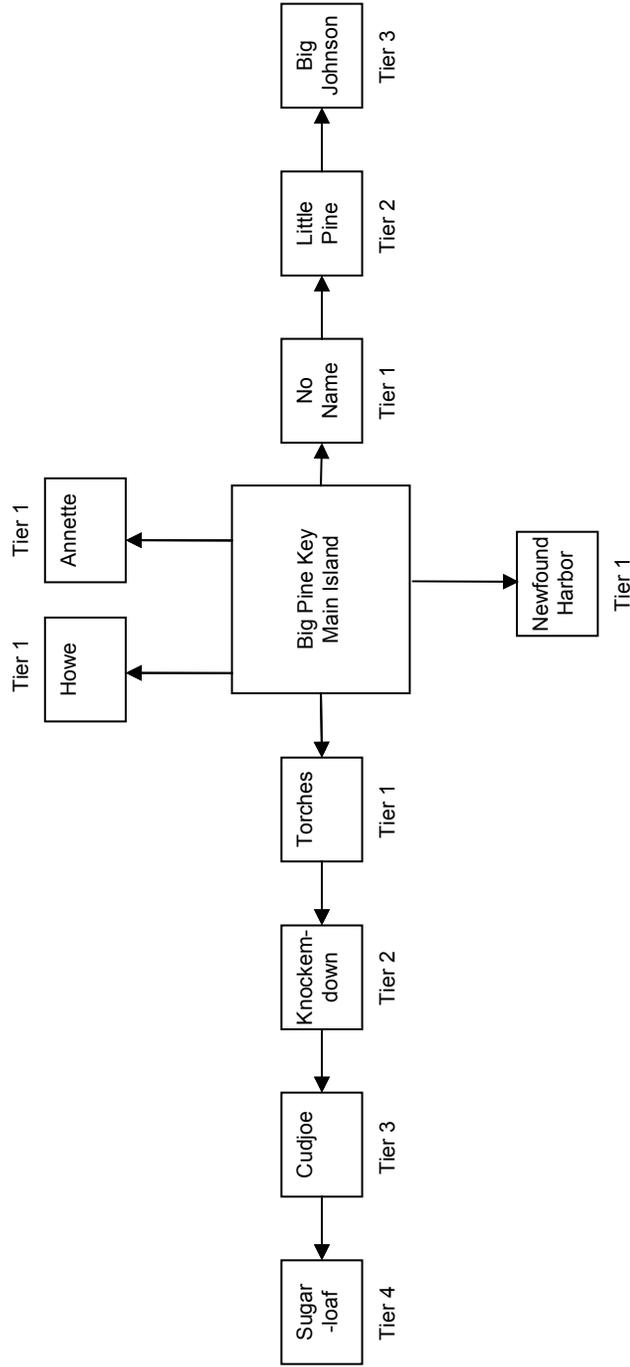
Fig. 3.2. Conceptual model representing the dispersal routes of Key deer among the 11 island-complexes in

the Lower Florida Keys.

MODELS DESCRIPTION

The metapopulation model was developed as a stochastic compartment model

based on difference equations ($\Delta t = 1$ year) and consists of 11, structurally-identical,

sub-models, each representing a sex- and age-structured subpopulation for each island-

complex:

$$N_{i,j,t+1} = N_{i,j,t} + (n_{j,t} - m_{i,j,t} - e_{i,j,t} - r_{i,j,t}) * \Delta t, \text{ for } i = 0 \tag{1}$$

$$N_{i,j,t+1} = N_{i,j,t} + (r_{i-1,j,t} + i_{i-1,j,t} - m_{i,j,t} - e_{i,j,t} - r_{i,j,t}) * \Delta t, \text{ for } i > 0 \tag{2}$$

where $N_{i,j,t}$ represents the number of females ($j = 1$) or males ($j = 2$) in age class i at the

beginning of time t, $n_{j,t}$ represents the number of females or males born into age class 0

during time t, and $m_{i,j,t}$, $e_{i,j,t}$, $i_{i,j,t}$, and $r_{i,j,t}$ represent the number of females or males in age

class i dying, emigrating from the island, immigrating to the island, and remaining on the

island, respectively, during time t.

Natality

Maximum natality rate was estimated as 1.05 fawns per reproductively mature

female per year, including yearlings (1 - 2 years old) and adults ($\geq$ 2 years old) (Hardin

1974); Key deer fawns ($<$ 1 year old) are not reproductively active (Hardin 1974, Folk

and Klimstra 1991a). Sex ratio at birth was estimated as 41% females and 59% males

(Hardin 1974). Thus,

$$n_{j,t} = \sum_{i=1}^{i=2} (0.41 * k1_t) * N_{i,1,t}, \text{ for } j = 1 \tag{3}$$

$$n_{j,t} = \sum_{i=1}^{i=2} (0.59 * k1_t) * N_{i,1,t}, \text{ for } j = 2 \tag{4}$$

where $k1_t$ represents a density-dependent natality rate (fawns born per reproductively mature female per year), which decreases linearly from 1.05 to 0 as population size increases from 0 to a number (K') somewhat larger than the carrying capacity of the island-complex (K) (Table 3.1). The value of K' for each island-complex was adjusted such that natality was just sufficient to offset mortality when the simulated population size reached K. Estimates of K were obtained following the methodology used previously for No Name and Big Pine Keys (Lopez 2001, Lopez et al. 2004*b*); each island-complex was classified into 6 habitat types using digital vegetation coverages, the area (ha) of each habitat type was multiplied by the corresponding Key deer habitat selection ratio (a weighting factor based on relative deer use), and carrying capacity was estimated as the sum of these values.

Mortality

Estimates of age- and sex-specific mortality rates were based on survival estimates (proportion of individuals surviving to the next age class, $k2_{i,j}$) obtained from Key deer studies conducted on Big Pine and No Name Keys (Hardin 1974, Silvy 1975, Lopez 2001) (Table 3.2). Survival estimates for yearlings and adults were calculated from radiocollared animals using a known-fate model (Program MARK, White and Burnham 1999, Lopez 2001); fawn survival was estimated by adjusting the model-fitted estimates to compensate for presumed overestimation due to small sample sizes (Lopez 2001:160).

Table 3.1. Carrying capacities (*K*) of the 11 island-complexes estimated following the methodology used previously for No Name and Big Pine Keys (Lopez 2001, Lopez et al. 2004*b*). Also presented are areas and estimates of Key deer abundance in the year 2000 (Lopez et al. 2004*a*, R. Lopez, unpublished data). Island complexes are classified into tiers based on distance from Big Pine Key.

| Tier | | Deer Carrying | |
Island-Complex	Area (ha)	Capacity (*K*)	Deer Density
Main			
Big Pine	2,549	517	406
Tier 1			
Annette	222	26	6
Howe	373	50	16
Newfound Harbor	76	12	10
No Name	471	90	78
Torches/Ramrod	1,714	287	94
Tier 2			
Knockemdown/Summerland	1,019	155	8
Little Pine	382	61	16
Tier 3			
Big Johnson	154	19	0
Cudjoe	1,319	217	6
Tier 4			
Sugarloaf	1,399	224	6

Table 3.2. Estimates of age- and sex-specific survival rates for Key deer on Big Pine and No Name Keys (Hardin 1974, Silvy 1975, Lopez 2001), and estimates of age- and sex-specific emigration rates for Key deer moving from Big Pine Key to No Name Key (Lopez 2001).

Age Class[a] (i)	Mean (SE) Survival (proportion surviving per year)		Mean (SD) Emigration (proportion emigrating per year)	
	Females ($j = 1$)	Males ($j = 2$)	Females ($j = 1$)	Males ($j = 2$)
Fawn ($i = 0$)	0.470 (0.061)	0.470 (0.061)	0.000 (0.000)	0.000 (0.000)
Yearling ($i = 1$)	0.824 (0.071)	0.569 (0.089)	0.032 (0.047)	0.107 (0.056)
Adult ($i = 2$)	0.842 (0.030)	0.597 (0.054)	0.032 (0.047)	0.107 (0.056)

[a]Age classes defined as fawn (0-1 year), yearling (1-2 years), and adult (≥ 2 years).

Each year of simulated time, a value for each $k2_{i,j}$ is drawn randomly from the normal distribution (truncated at zero) (Akçakaya 1991, Grant et al. 1997) generated by the mean and standard error corresponding to that age- and sex-class (Table 3.2).

Thus,

$$m_{i,j,t} = (1 - k2_{i,j}) * N_{i,j,t} \qquad (5)$$

Emigration and Immigration

Estimates of age- and sex-specific emigration rates (proportion of individuals leaving the island-complex per year, $k3_{i,j}$) were based on estimates of dispersal from Big Pine Key to No Name Key (Lopez 2001) (Table 3.2) and calculated as:

$$e_{i,j,t} = k3_{i,j} * N_{i,j,t} \qquad (6)$$

Estimates of age- and sex-specific immigration ($i_{i,j,t}$) to the different island-complexes were based on geographical location and the assumption that all emigrating individuals move away from Big Pine Key. Big Pine is the main source population and is the only population that disperses to more than one other island-complex. Individuals emigrating from Big Pine are distributed among adjoining (tier 1) island-complexes (Table 3.1, Fig. 3.2); No Name, Newfound Harbor, and Howe each received 25% of the Big Pine emigrants ($i_{i,j,t} = 0.25 * e_{i-1,j,t}$) because of their close proximity, and Torches and Annette each received 12.5% of the Big Pine emigrants ($i_{i,j,t} = 0.125 * e_{i-1,j,t}$) because of their further distance from Big Pine. All emigrants from island complexes other than Big Pine are immigrants ($i_{i,j,t} = e_{i-1,j,t}$) to the island-complex in the next tier to which they are connected (Table 3.1, Fig. 3.2).

The number of individuals in each age- and sex-class remaining on the same island-complex (and advancing age class $i + 1$) during time t is calculated as:

$$r_{i,j,t} = N_{i,j,t} - m_{i,j,t} - e_{i,j,t.} \tag{7}$$

MODEL CALIBRATION AND SENSITIVITY ANALYSIS

Initial Abundances

Initial abundances used in model simulations beginning in 2000 were determined from mark-resight estimates based on 247 road count surveys conducted from 1998-2001 on Big Pine Key (Lopez et al. 2004a). Density on the other islands was estimated using trip cameras and Lincoln-Peterson mark-recapture statistics (R. Lopez, unpublished data). A stable age distribution was assumed for the Big Pine Key population and the other island populations were proportionally divided into each sex- and stage-class (0.125 for female and male fawns and yearlings; and 0.25 for female and male adults) (Table 3.2). To evaluate the model, I used the 1971 Big Pine Key deer population (170) as the initial abundance proportionally divided into each sex- and age-class as described above (Lopez et al. 2004a).

I varied dispersal in simulations to observe the effects of different female dispersal rates on the Key deer population. I assumed that male dispersal would be double female dispersal. I used the female dispersal estimate reported by Lopez (2001) as the medium rate (0.03). I set high female dispersal as the rate at which the projected metapopulation began to decline (0.05). Thus, I ran model simulations using none ($f = 0$, $m = 0$), low ($f = 0.01$, $m = 0.02$), medium ($f = 0.03$, $m = 0.06$), and high ($f = 0.05$, $m = 0.10$) dispersal rates (Fig. 3.3). I assumed dispersal to be density dependent and adjusted

each rate with a simple linear model where dispersal was maximum when the population

equaled *K*, and dispersal was 0 when the population equaled 0.
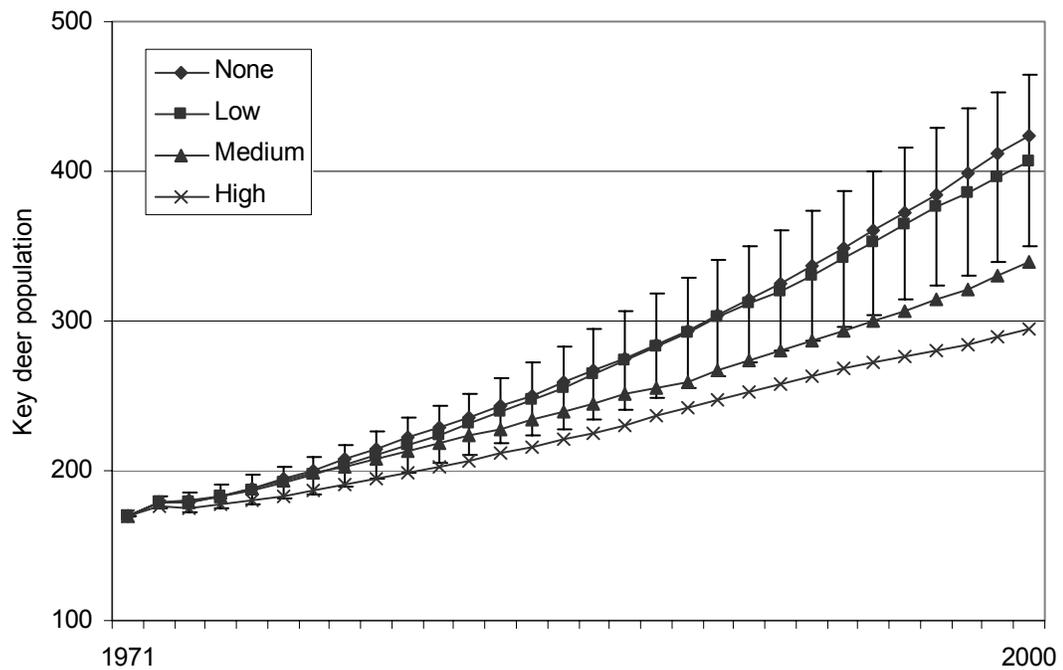
Fig. 3.3. Evaluation projections of the Big Pine Key deer population under none (f = 0,

m = 0), low (f = 0.01, m = 0.02), medium (f = 0.03, m = 0.06), and high (f = 0.05, m =

0.10) dispersal scenarios in the Lower Florida Keys, 1971-2000. Vertical bars represent

± 1 SD of the mean, based on 120 replicate stochastic simulations under low dispersal.

POPULATION PROJECTIONS

I ran 120 Monte Carlo simulations over a 20-year period under each of the 4 dispersal scenarios (none, low, medium, and high). The model predicted an increase in the total metapopulation under each scenario with medium dispersal producing the highest population increase and no dispersal producing the smallest increase (Fig. 3.4). Big Pine Key (where the majority of the Key deer population resides) also increased under each dispersal scenario (Fig. 3.5). Big Pine Key deer density was projected at 100% of K with no and low dispersal, 98% K with medium dispersal, and 88% K with high dispersal.

In analyzing model results, I defined a viable island population as ≥50 deer. I chose 50 as the minimum viable population size because, historically, it is the lowest known Key deer population size which resulted in an increase (Dickson 1955). I defined a successful colonization as having a deer population at ≥50% of K. Model results varied based on the input dispersal scenario (Table 3.3). Number of successful colonizations and viable populations, respectively, by dispersal scenario were as follows: none, 7, 3; low 7, 3; medium 7, 6; and high 8, 6.

To evaluate the risk of quasi-extinction, I calculated the probability of each island-complex having <50 deer during each timestep (2000-2020) and under each dispersal scenario (none, low, medium, and high). Big Pine, No Name, and Torches/Ramrod had 0% probability of dropping below 50 deer under all dispersal scenarios from 2000-2020. Newfound Harbor, Annette, Big Johnson, Cudjoe, and Sugarloaf had a 100% probability of dropping below 50 deer under all dispersal

scenarios. Howe, Knockemdown/Summerland, and Little Pine had variable probabilities of dropping below 50 deer (Figs. 3.6, 3.7, 3.8).
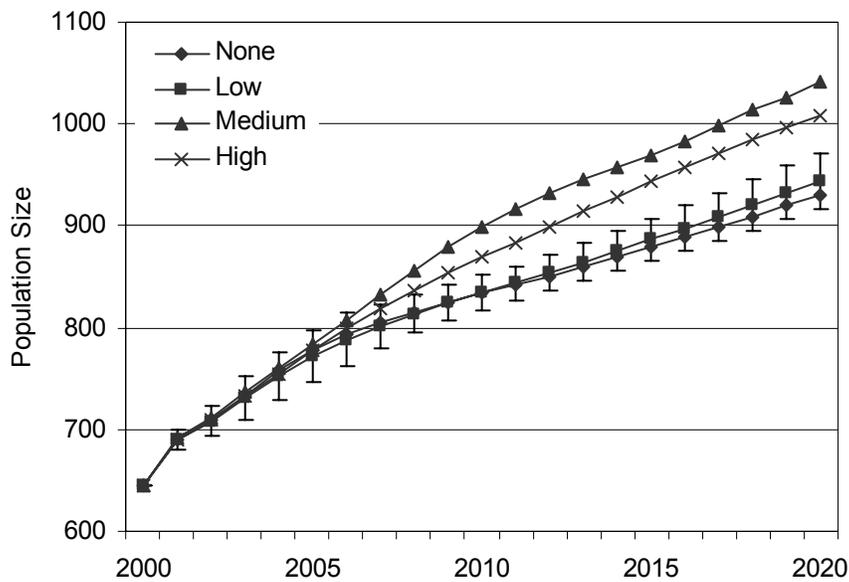
Fig. 3.4. Model projected Key deer metapopulations under 4 dispersal scenarios (none, f = 0, m = 0; low, f = 0.01, m = 0.02; medium, f = 0.03, m = 0.06; and high, f = 0.05, m = 0.10) in the Lower Florida Keys, 2000-2020. Vertical bars represent ± 1 SD of the mean, based on 120 replicate stochastic simulations under low dispersal.
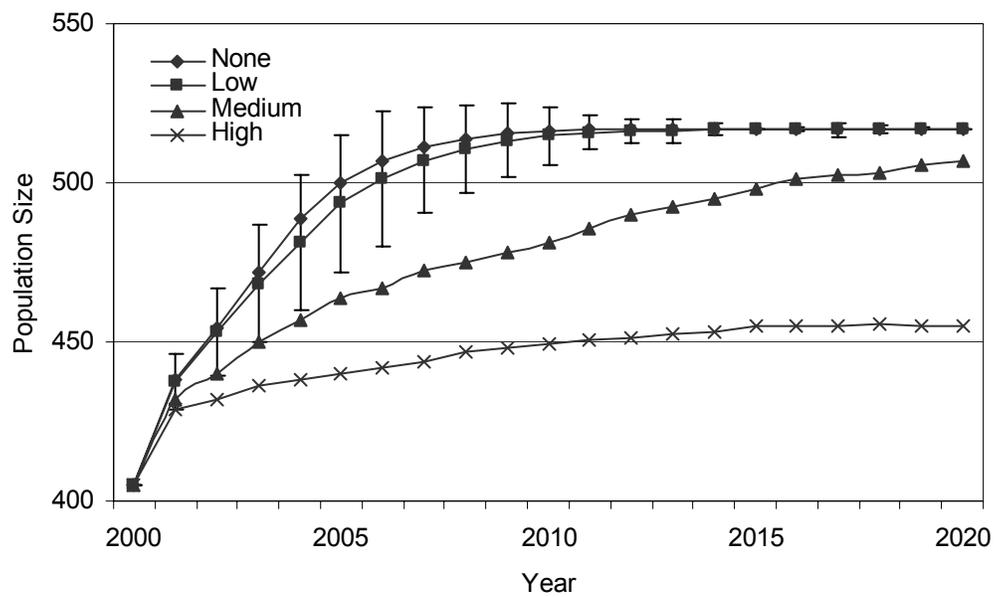
Fig. 3.5. Model projected Big Pine Key deer populations under 4 dispersal scenarios

(none, f = 0, m = 0; low, f = 0.01, m = 0.02; medium, f = 0.03, m = 0.06; and high, f =

0.05, m = 0.10) in the Lower Florida Keys, 2000-2020. Vertical bars represent ± 1 SD

of the mean, based on 120 replicate stochastic simulations under low dispersal.

Table 3.3. Model projected Key deer populations for each island-complex under 3 dispersal scenarios (none, f = 0, m = 0; medium, f = 0.03, m = 0.06; and high, f = 0.05, m = 0.10) in the Lower Florida Keys, 2000-2020.

Tier	No Dispersal			Low Dispersal		
Island-Complex	Mean	SD	% of K[a]	Mean	SD	% of K
Main						
Big Pine	517	0.8	100	517	0.0	100
Tier 1						
Annette	16	4.8	63	16	4.8	62
Howe	37	8.5	74	44	8.0	88
Newfound Harbor	12	0.9	98	12	0.9	98
No Name	90	0.0	100	90	0.4	100
Torches/Ramrod	177	25.5	62	174	24.0	61
Tier 2						
Knockemdown/Summerland	19	6.8	12	19	6.1	12
Little Pine	35	8.2	57	35	8.3	57
Tier 3						
Big Johnson	0	0.0	0	0	0.0	0
Cudjoe	17	6.2	8	16	5.8	8
Tier 4						
Sugarloaf	16	5.6	7	16	5.9	7
Metapopulation	931	30.9	n/a	944	28.0	n/a

[a]K = island carrying capacity; percentages > 100 are due to demographic stochasticity.

Table 3.3. Continued.

Tier	Medium Dispersal			High Dispersal		
Island-Complex	Mean	SD	% of K^a	Mean	SD	% of K
Main						
Big Pine	507	17.9	98	455	43.8	88
Tier 1						
Annette	26	0.0	100	26	0.0	100
Howe	50	0.0	100	50	0.1	100
Newfound Harbor	12	0.5	102	15	0.7	124
No Name	90	0.1	100	90	0.1	100
Torches/Ramrod	204	24.4	71	182	20.6	64
Tier 2						
Knockemdown/Summerland	50	9.8	32	74	9.9	48
Little Pine	61	0.0	100	61	0.6	100
Tier 3						
Big Johnson	9	4.1	48	19	0.1	100
Cudjoe	17	6.2	8	17	5.7	8
Tier 4						
Sugarloaf	17	5.6	8	14	4.6	6
Metapopulation	1,041	33.5	n/a	1,008	49.9	n/a

[a]K = island carrying capacity; percentages > 100 are due to demographic stochasticity.
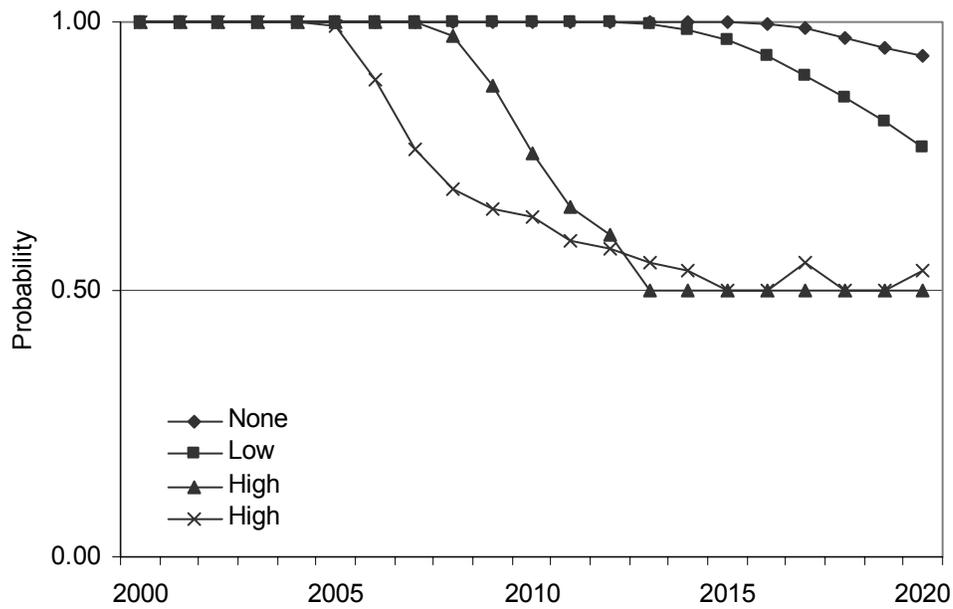
Fig. 3.6. Model projected probability of Howe Key (tier 1, Fig. 3.2) dropping below 50 Key deer under 4 dispersal scenarios (none, f = 0, m = 0; low, f = 0.01, m = 0.02; medium, f = 0.03, m = 0.06; and high, f = 0.05, m = 0.10) in the Lower Florida Keys, 2000-2020.
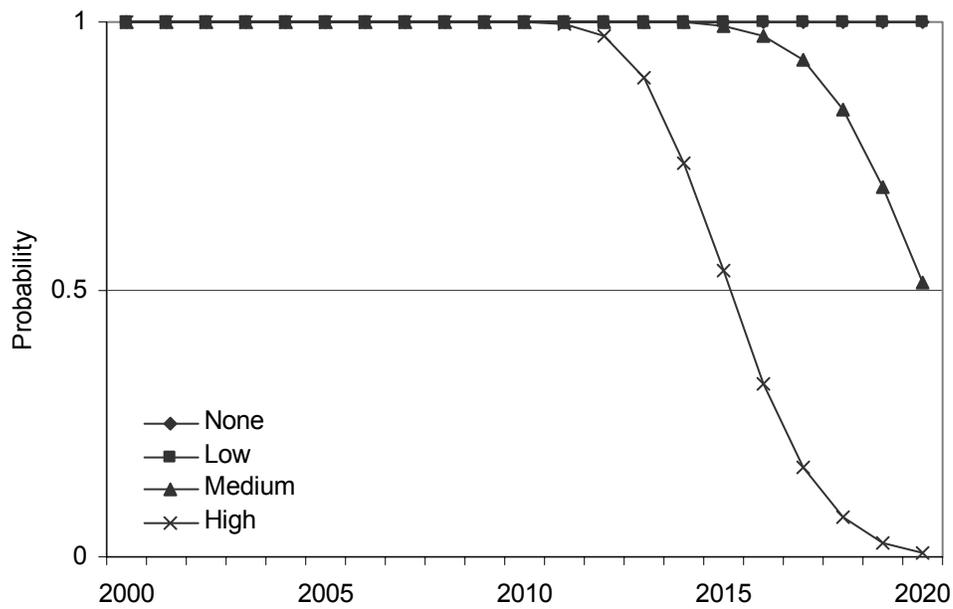
Fig. 3.7. Model projected probability of Knockemdown/Summerland Complex (tier 2,

Fig. 3.2) dropping below 50 Key deer under 4 dispersal scenarios (none, f = 0, m = 0;

low, f = 0.01, m = 0.02; medium, f = 0.03, m = 0.06; and high, f = 0.05, m = 0.10) in the

Lower Florida Keys, 2000-2020.
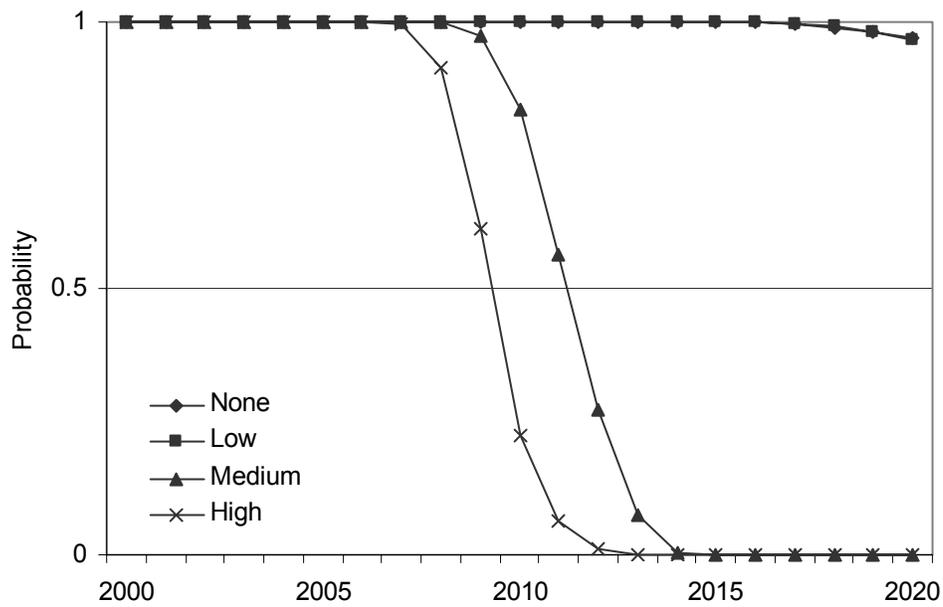
Fig. 3.8. Model projected probability of Little Pine Key (tier 2, Fig. 3.2) dropping

below 50 Key deer under 4 dispersal scenarios (none, f = 0, m = 0; low, f = 0.01, m =

0.02; medium, f = 0.03, m = 0.06; and high, f = 0.05, m = 0.10) in the Lower Florida

Keys, 2000-2020.

DISCUSSION

The majority of the Key deer population is located on Big Pine Key. Current deer estimates suggest that the Big Pine Key population is at 78% island carrying capacity and is increasing (Lopez et al. 2004*a*). Previous studies have demonstrated density dependence in white-tailed deer populations (McCullough 1979, Halls 1984). I included density dependence in the Key deer metapopulation model to evaluate the dynamics of the population as it approaches *K*. Deer dispersal was assumed to increase linearly as density increased toward *K*. I assumed that as deer densities increased that competition for territory and resources would increase the likelihood of dispersal to other islands (Kammermeyer and Marchington 1976). I assumed an outward dispersal from islands of high density (the main island, Big Pine Key) to islands of lower density. As other island populations increased (due to births and immigration), they were also modeled to disperse with each island population flowing outward from the main population to peripheral islands like stepping stones (Fig. 3.2). I used this dispersal model to evaluate the possibility of Key deer establishing other viable populations on islands with suitable habitat within the current Key deer range in the Lower Florida Keys.

By modeling various dispersal scenarios, I was able to evaluate the effects of this unknown parameter on population projections for individual island-complexes and the metapopulation. The greatest differences were found using the no and low dispersal scenarios. Under these scenarios, half as many peripheral island-complexes were projected to reach a population ≥50 as with the medium and high scenarios. Howe,

Knockemdown/Summerland, and Little Pine were most affected by dispersal scenario (Figs. 3.6, 3.7, 3.8). However, all models predicted successful colonizations on all tier 1 islands (Table 3.3) suggesting that dispersal has occurred in previous years. In evaluating dispersal scenarios (none, low, medium, and high), the reported population increase on Big Pine Key from 1971 to 2000 (170 to 406; Lopez et al. 2004*a*) was most closely matched by the low dispersal rate (170 to 407). Therefore, I will concentrate my discussion using the results from the low dispersal scenario.

Of the 11 island-complexes I identified in the lower Keys as possessing suitable habitat for Key deer, 3 would not support a deer population ≥50 deer based on my estimate of carrying capacity. Because of the low deer densities that these islands (Big Johnson, Annette, and Newfound Harbor) could support, they are considered supporting islands which can contribute habitat resources and genetic migration during times of hardship, as well as serve as stepping-stones to other islands. The remaining 8 islands can or do support viable deer populations according to my estimates. Big Pine, No Name, and Torches/Ramrod each had initially-estimated deer populations ≥50 in 2000. Under the low dispersal scenario, no additional island-complexes were projected to increase to viable populations (≥50). Under medium and high dispersal, the models projected that in 20 years, Howe, Knockemdown/Summerland, and Little Pine will increase to viable populations. However, all model simulations suggest that Cudjoe and Sugarloaf will have low deer densities (<20) despite their relatively high carrying capacities (217 and 224, respectively). Projected populations for Cudjoe (tier 3) and Sugarloaf (tier 4) were similar under all dispersal scenarios (Table 3.3) suggesting that

dispersal rate (whether none, low, medium, or high) will have little effect on population size and that distance from the main island (tier level) and the size of intermediate islands will have the greatest effect on population size within the 20 year timeframe of the model.

The potential benefits of establishing additional Key deer populations on Cudjoe and Sugarloaf Keys are numerous. First, the current greatest threat to Key deer is urban development and its associated risk factors (e.g., loss of habitat, habitat fragmentation, and increased deer mortality due to vehicle collisions). The majority of Key deer reside on Big Pine Key where high urban development in the south has created an ecological sink primarily due to high roadkill mortalities of Key deer (chapter II). While measures are underway to reduce deer roadkill mortality on Big Pine Key, the effects of future policy (e.g., the pending Habitat Conservation Plan and lifting of the building moratorium) are unknown. Second, the Florida Keys are prone to hurricanes and while impacts on Key deer during previous hurricanes have been minimal (Lopez et al. 2000) the potential exists for a hurricane to severely impact the population because of its limited geographic range. Sugarloaf and Cudjoe are the furthest islands from Big Pine making them desirable choices for increasing the populations range to minimize the potential catastrophic effects of a hurricane. Finally, additional local populations on Cudjoe and Sugarloaf Keys will provide increased population growth and genetic heterogeneity further bolstering this endangered population towards recovery.

MANAGEMENT IMPLICATIONS

The conservation and management of an endangered species such as Key deer requires a thorough understanding of the demographic and environmental factors influencing its population dynamics. While much is known about Key deer biology (e.g., survival, maternity, habitat use), changes in habitat conditions due to urban development will continue to prove challenging to managers. Habitat loss and fragmentation combined with increasing deer densities will have unpredictable effects on parameters such as dispersal. Through the use of simulation modeling, I examined the potential effects of changes in dispersal rates on the Key deer metapopulation. According to my results, under all modeled scenarios the establishment of viable populations on Cudjoe and Sugarloaf by dispersal alone is unlikely within the next 20 years. I recommend the use of other methods (e.g., translocations) to supplement deer numbers on these islands in order to establish viable populations.

CHAPTER IV

SURVIVAL ESTIMATES

SYNOPSIS

Obtaining reliable survival estimates is important in the management of wildlife populations, particularly for the construction of computer simulation models. While many methods exist to estimate survival, many of these methods may be cost-prohibitive or time consuming (e.g., radiotelemetry). Life tables can provide survival estimates using data routinely collected by some management agencies such as roadkills. I compared annual survival estimates from life tables using roadkill data to those calculated from radiotelemetry data. Life table survival estimates were similar to radiocollared deer except for adult females where the life table estimate (0.546) was considerably lower than the radiotelemetry estimate (0.846). I attribute the lower survival in the life table estimate to (1) the variance in mortality risk and (2) unequal distribution of female deer on the study area. Results suggest that life tables based on roadkills could potentially be a useful tool in estimating survival for Key deer (*Odocoileus virginianus clavium*). Combining results from these 2 methods provide verification and insight into the factors regulating population growth.

INTRODUCTION

Estimating wildlife population demographics is an important component in construction of simulation models (e.g., harvest models, population viability analyses) used to predict population trends. Annual survival is an important population parameter that influences population growth (White and Bartmann 1998, Krebs 1999, Rabe et al.

2002), and is a key component in the development of these models. For example,

population viability analyses (PVA) are commonly used in endangered species

management (Boyce 1992, Akçakaya 2000) and requires precise survival estimates.

Many methods for estimating survival exist, however, each of these methods have their

own benefits and problems (Krebs 1999). Estimating survival from radiotelemetry data,

for example, can provide precise estimates yet often at great expense (Krebs 1999).

Limited or declining budgets of many wildlife management agencies may prohibit the

use of radiotelemetry data in estimating survival (Rabe et al. 2002). Alternative

approaches to estimating survival might include the use of mark-recapture data, age-

composition data, or life tables (Krebs 1999).

Life tables can be used to estimate age-specific mortality or survival from an

assumed cohort using various methods including age at death, age of remains, and age

distribution of a population (Caughley 1977, Krebs 1999). Although data collection for

life tables also can be expensive (Caughley 1977), some agencies routinely collect

roadkill (mortalities due to vehicle collisions) data that can be used in the construction of

life tables. Use of already collected data could be a cost-effective way for agencies to

estimate important population parameters for managing wildlife populations. Caughley

(1977) cautioned against the improper use of these methods and violations of

assumptions in life table construction. For example, to estimate survival from carcasses

or skulls, the population must have a stable age distribution and a known rate of increase

(Caughley 1977). Potential biases pertaining to data collection include the use of hunter-

harvest mortalities, seasonal collection (i.e., winter or summer only), or mortalities

resulting from rare events such as catastrophes. Each of these situations could produce biased survival estimates. Care should be taken to evaluate the accuracy of survival estimates based on life table data, and, whenever possible, these estimates should be validated with estimates derived by other means such as radiotelemetry data. Use of erroneous survival estimates in making management decisions could have potentially devastating effects on a population, especially in the management of an endangered species like Florida Key deer.

Key deer are a sub-species of white-tailed deer endemic to the Florida Keys (Hardin et al. 1984). Urban development and habitat fragmentation threaten the Key deer population with 50% of Key deer mortality attributed to deer-vehicle collisions (Lopez et al. 2003). Since 1968, the U.S. Fish and Wildlife Service (USFWS) has collected Key deer mortality data (Lopez et al. 2004*a*) as part of a long-term monitoring program. Additionally, radiotelemetry data has been collected during 2 separate studies from December 1968 to June 1972, and January 1998 to December 2000 (Hardin 1974, Silvy 1975, Lopez 2001). Survival estimates for Key deer using radiotelemetry data were recently reported (Lopez 2001, Lopez et al. 2003), which offers a unique opportunity to compare survival estimates from different sources (i.e., radiotelemetry versus roadkill data). My research objectives were to (1) calculate Key deer survival using roadkill data (life table), and (2) compare these survival estimates to previously published survival estimates calculated from radiotelemetry data.

STUDY AREA

The Florida Keys are a chain of small islands approximately 200-km long extending southwest from peninsular Florida. Big Pine Key (BPK; 2,548 ha) and No Name Key (NNK; 461 ha) are within the boundaries of the National Key Deer Refuge (NKDR) in Monroe County, and support approximately 75% of the deer population (Lopez 2001). Soil types vary from marl deposits to bare rock of the oolitic limestone formation (Dickson 1955). Island vegetation varies by elevation with red (*Rhizophora mangle*), black (*Avicennia germinans*), and white mangroves (*Laguncularia racemosa*), and buttonwood (*Conocarpus erectus*) forests occurring near sea level (maritime zones). As elevation increases inland, maritime zones transition into hardwood (e.g., Gumbo limbo [*Bursera simaruba*], Jamaican dogwood [*Piscidia piscipula*]) and pineland (e.g., slash pine [*Pinus elliottii*], saw palmetto [*Serenoa repens*]) upland forests with vegetation intolerant of salt water (Dickson 1955, Folk 1992).

METHODS

Life Table Data

Since 1968, NKDR staff have recorded deer mortalities as part of recovery efforts. Direct sightings, citizen reports, or observation of turkey vultures (*Cathartes aura*) helped locate most dead animals. Animals collected were held frozen prior to necropsy examination or necropsied immediately. Carcass quality or ability to determine cause of death ranged from good to marginal (Nettles 1981, Nettles et al. 2002). Age, sex, body mass, and cause of death were recorded for each animal using procedures described by Nettles (1981), and all mortality locations were recorded.

Life tables were constructed for roadkilled Key deer by sex and age using the USFWS mortality data collected from 1995-2000 (Table 4.1). Mortality data were combined between islands due to small sample sizes on No Name Key. Age-specific survival was estimated by sex- and age-classes assuming a stable age distribution and an instantaneous population growth rate of 0.037 (Caughley 1977, Krebs 1999, Lopez et al. 2004*a*). The estimated number of deer dying for each age interval was calculated using

$$d'_x = d_x e^{rx}$$

where,

d_x = actual number of carcasses in each age class

r = instantaneous population growth rate

x = age class

e = base of natural logarithms (i.e., 2.71828).

Survival (p_x) was calculated using

$$p_x = 1 - \left(\frac{d_x}{\sum_{y=x}^{\infty} d_y} \right) \text{ (Caughley 1977).}$$

Deer of unknown sex or age were not used in calculations. For comparison purposes, a weighted mean adult survival rate was calculated by grouping yearly age classes ≥ 2 (Caughley 1977).

Table 4.1. Sex and age composition of roadkilled Florida Key deer collected by U.S.

Fish and Wildlife Service on Big Pine and No Name Keys, Florida, 1995-2000.

	Roadkilled deer	
Age (years)	Female	Male
0	44	77
1	29	69
2	25	42
3	20	27
4	10	20
5	2	5
6	2	2
7	1	0
8	2	0
Total	135	242

Radiotelemetry Data

Lopez et al. (2003) recently reported Key deer survival estimates based on 314 radiocollared animals by sex, age, and area (north BPK, south BPK, and NNK). Due to constraints from small sample sizes, I was not able to construct life tables for each of these areas. Thus, I used the pooled survival estimates reported in Lopez (2001) for comparison purposes. Deer were classified into 3 age groups: fawn (< 1 year old), yearling (1-2 years old), and adult (≥ 2 years old). Annual Key deer survival was estimated using a known-fate model framework in Program MARK (White and Burnham 1999, Lopez et al. 2003).

I used survival estimates calculated from radiotelemetry data as a benchmark for comparison under the assumption that these estimates best reflected actual Key deer survival rates. Life table survival estimates were compared to radiotelemetry survival estimates for each sex and age category (Lopez 2001). Life table survival estimates were considered significantly different if they fell outside of the 95% confidence intervals for radiotelemetry survival estimates (Johnson 1999) reported by Lopez (2001).

RESULTS

A total of 377 deer (135 females, 242 males) roadkill mortalities were recorded by USFWS biologists from 1995-2000. Key deer survival estimates derived from life tables were generally similar to rates calculated from radiocollared deer (Table 4.2, Fig. 4.1). The only exception was for adult females where the life table estimate (0.546) was considerably lower than the radiotelemetry estimate (0.846) (Table 4.2).

Table 4.2. Comparison of annual Key deer survival estimates by source (roadkills, radiotelemetry), sex, and age-class on Big Pine and No Name Keys, Florida.

Sex Age	Roadkill[a] Survival	Radiotelemetry[b] Survival	95% LCI	95% UCI
Female				
Fawn	0.692	0.615	0.275	0.834
Yearling	0.697	0.824	0.628	0.923
Adult	0.546*	0.842	0.772	0.892
Male				
Fawn	0.698	0.743	0.454	0.895
Yearling	0.596	0.569	0.379	0.721
Adult	0.490	0.597	0.483	0.695

*Survival estimates calculated from roadkill data that are significantly different.

[a] Roadkill data collected by U.S. Fish and Wildlife Service from 1995-2000.

[b] Radiotelemetry estimates from 1968-1972 and 1998-2000 (Lopez 2001).
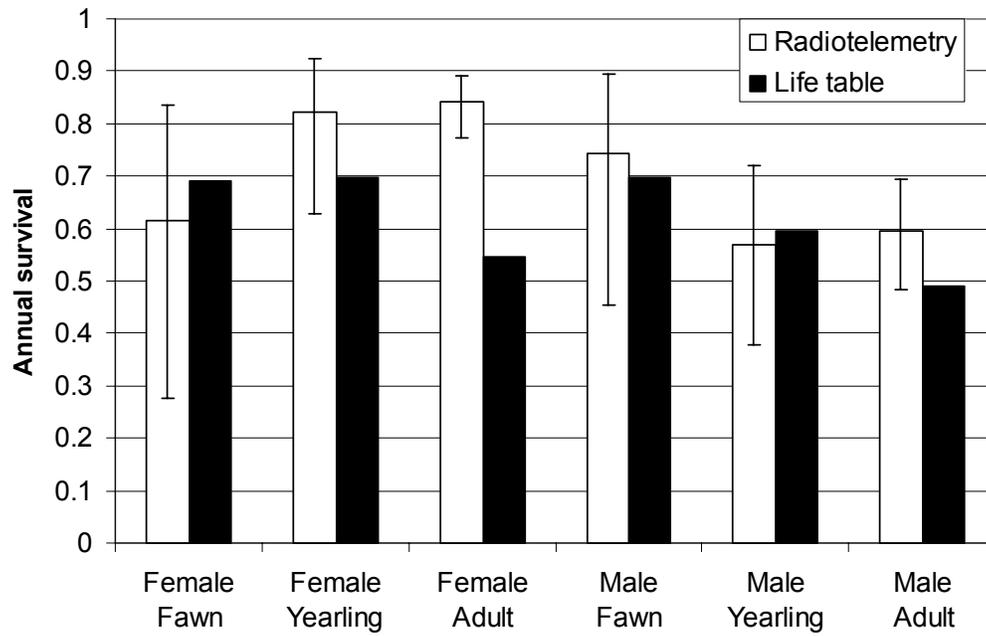
Fig. 4.1.   Comparison of Key deer survival estimates calculated using life tables and radiotelemetry (with 95% confidence intervals) on Big Pine and No Name Keys, Florida.

DISCUSSION

Construction of life tables with roadkill data for white-tailed deer requires several assumptions that may introduce bias (Caughley 1977); thus, results should be viewed cautiously. For example, I found a difference in estimated survival of adult females between the 2 data sources (Fig. 4.1). Accurate survival estimates for adult females are particularly important as these estimates tend to have a significant impact on large ungulate population trends (White and Bartmann 1998, Rabe et al. 2002). Overestimating adult female survival could have detrimental effects in both endangered species and game population management. However, in this study life table survival estimates underestimated adult female survival which would produce a lower, more conservative estimate of population growth. I attribute the lower survival observed from the roadkill data to 2 factors. First, mortality risk is not constant over the entire area. For example, Lopez et al. (2003) reported Key deer road mortality was higher in the southern half of BPK, which has greater amounts of urban development and traffic. Deer-vehicle collisions account for the majority (74%, Lopez et al. 2003) of Key deer mortality recorded by USFWS biologists. Second, deer density by sex is not evenly distributed across the islands. The deer population on the northern half of BPK is 75% female while the southern half of BPK is only 56% female (Lopez et al. 2003).

MANAGEMENT IMPLICATIONS

I found life table survival estimates to be similar to those derived from radiotelemetry data, suggesting a potential alternative for estimating survival of Key deer. The long-term monitoring of Key deer mortality by USFWS biologists offers

managers such an opportunity. In this study, comparing results from both methods

provided survival estimate verification and insight into the factors regulating Key deer

population growth. Wildlife managers who consider the use of life table data in

estimating survival should be aware of the potential biases. However, despite potential

biases, resource managers that collect roadkill data (e.g., state and national parks,

refuges, and forests) may be able to provide an adequate and cost-effective estimate for

survival of deer using life tables.

CHAPTER V

IMPACTS OF URBANIZATION

SYNOPSIS

Rapid human population growth and urbanization have had a negative impact on

species biodiversity. As competition for resources between man and wildlife continues,

it is important to understand the effects of urbanization on species. Endangered Key

deer (*Odocoileus virginianus clavium*) are endemic to the Florida Keys which have

undergone rapid human population growth and development over the past 30 years. My

study objectives were to evaluate the impacts of development on Key deer habitat use,

population dynamics, behavior, and morphology. Results suggest that Key deer use

urban areas more today than 30 years ago. This increase in urban use does not appear to

be the result of increased availability of urban areas but rather a behavioral change in

resource selection by deer. As predicted, behavioral differences in deer related to urban

use also were found. However, contrary to my predictions, survival was higher for more

urban deer than for less urban deer suggesting a positive relationship between deer urban

use and survival. Analysis of Key deer body mass also was converse to my predictions

as deer weights appear to have increased over the past 30 years. Collectively, my results

suggest that over the past 30 years Key deer have become more urbanized and that deer

plasticity has allowed them to adapt and persist in an urbanizing environment. However,

the future ability of Key deer to persist in an environment with continued urban

development is unknown. At some threshold, urban development would become

unsustainable and unlike other forms of habitat change or environmental disturbances,

urban development is in most cases irreversible requiring careful planning in habitat conservation strategies.

INTRODUCTION

Conservation of native wildlife is becoming increasingly difficult due to continued human population growth and expansion. As the human population continues to increase, so does the rate of consumption of our natural resources. In fact, human population growth is cited as the greatest threat to species biodiversity (Meffe and Carroll 1997). The proliferation of housing has been identified as a primary mechanism of this impact, and an accelerating threat to biodiversity (Liu et al. 2003). Expansion of road networks and higher traffic levels on current roads are direct impacts of household proliferation. In the contiguous United States, roads and roadsides cover approximately 1% of the surface area, and impact 22% of it ecologically (Forman 2000). The expansion of households and roadways degrades wildlife habitat via fragmentation, outright destruction, facilitation of invasive exotic invasion, and wildlife-vehicle collisions (Gelbard and Harrison 2003, Lopez et al. 2003). As competition for resources between man and wildlife continues, it is important to understand the effects of urbanization on species.

The decline of many native wildlife species has been attributed to increased urbanization and anthropogenic impacts. The Florida Key deer is one such example. The endangered Florida Key deer is the smallest subspecies of white-tailed deer in the United States. Key deer are endemic to the Florida Keys archipelago stretching southwest off the southern tip of peninsular Florida (Hardin et al. 1984). Key deer range

is restricted to the Lower Florida Keys with approximately 60% residing on Big Pine

Key (BPK) and 15% residing on No Name Key (NNK; Folk 1992, Lopez 2001). During

the early 1900s, Key deer numbers declined due to unregulated hunting (Hardin et al.

1984). In 1940, the total Key deer population was estimated at <50 animals (Hardin et

al. 1984). In an effort to conserve and protect the deer, the National Key Deer Refuge

was established in 1957 and incorporated 3,457 ha of the historic Key deer range. One

third of the refuge's upland area is located in BPK and NNK. The establishment of the

refuge and increased law enforcement have resulted in the subsequent growth of the Key

deer population. The Key deer population grew to an estimated 300-400 animals by

1974 (Klimstra et al. 1974). Further, Key deer populations on BPK and NNK were

estimated to have grown by 240% between 1971 (~200 deer) and 2001 (~453-517 deer;

Lopez et al. 2004*a*).

Urbanization in the Keys began in the early 1900s with relatively slow

population growth until the 1970s. In 1970, there were 565 houses on BPK (Lopez

2001). Land development prior to 1970 occurred primarily in mangrove and

buttonwood areas. This resulted in the conversion of lowland areas to uplands (Lopez et

al. 2004*b*). For example, most home construction occurred on the periphery of the island

and these low elevation tidal areas were filled to create higher elevations for

development. During the housing boom of the late 1970s and early 1980s, development

moved to upland areas. The U.S. Census Bureau estimated the 1980 human population

on BPK at 2,350 with 1,681 housing units. By 2000, the BPK human population

increased to 4,206 and housing units increased to 2,453 (U.S. Census Bureau). Lopez et

al. (2004*b*) estimated that 610 ha (24%) and 31 ha (7%) were developed on BPK and

NNK, respectively. Urbanization resulted in changes in habitat types and in the amount

of usable space for Key deer (Fig. 5.1). The footprint from homes, businesses, and roads

removed approximately 232 ha from usable Key deer habitat (Lopez et al. 2004*b*).

Has urban development been beneficial or detrimental to Key deer?

I examine in this chapter whether the past 30 years of urban development on

BPK has been beneficial or detrimental to Key deer. Previous studies on Key deer have

suggested conflicting answers. For example, urbanization has resulted in the

modification of habitat and the creation of more upland habitat preferred by deer (Lopez

et al. 2004*b*). Over the last 30 years, BPK has experienced a 10-fold increase in human

population growth and urban development (Monroe County Growth Management

Division 1992) yet during this time, the Key deer population has grown by 240% (Lopez

et al. 2004*a*). If I evaluate the impacts of urbanization with deer population growth

alone, it would appear that urbanization has not harmed and may have possibly benefited

the deer (Peterson et al. 2004, Lopez et al. 2004*b*).
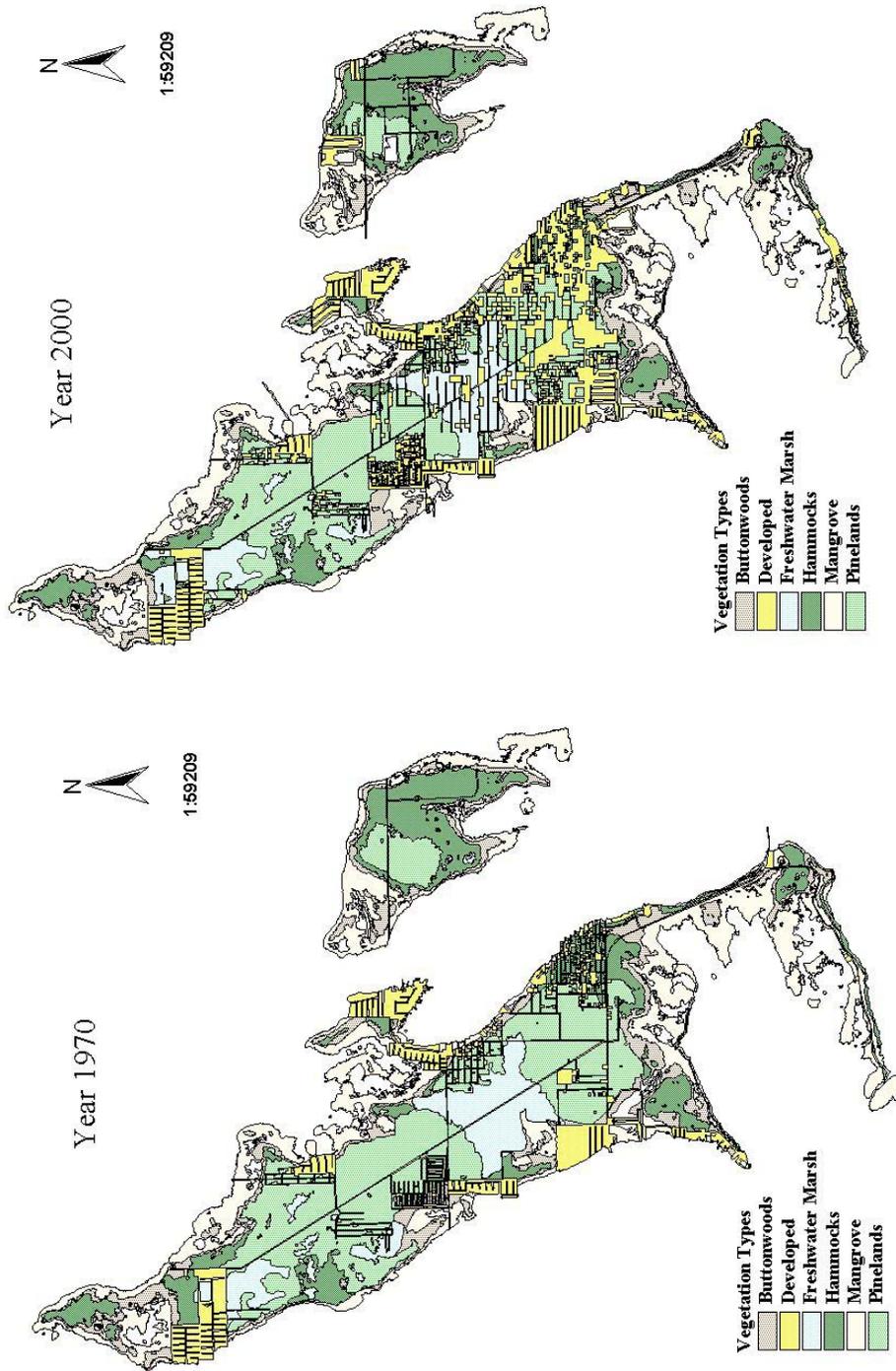
Fig. 5.1. Big Pine Key and No Name Key habitat maps during the historic (1970) and current (2000) study periods.

Conversely, urban development and its associated risks have also been reported as the greatest threat to the Key deer population (Lopez et al. 2003). Prior to the 1940s, humans were directly responsible for the early exploitation and near extinction of Key deer. While deer mortality due to illegal hunting is now minimal, other anthropogenic impacts including habitat loss and fragmentation, deer domestication, and deer–vehicle collisions have been cited as risk factors for Key deer (Hardin 1974, Folk and Klimstra 1991*b*, Folk 1992, Lopez et al. 2003). In a recent study, deer-vehicle collisions were cited as the primary mortality factor for Key deer accounting for 50% of total Key deer mortality on BPK (Lopez et al. 2003). Anthropogenic factors also accounted for other means of mortality for Key deer including entanglement in fences, drowning in swimming pools, and attacks by dogs. Urbanization and particularly the high traffic volume on the US 1 highway have created areas of varying habitat quality and mortality risk on BPK. In chapter II, I reported that fragmentation and high deer mortality in south BPK has created a non-viable sink population that is supplemented by a source population in north BPK. Furthermore, Peterson et al. (2004) examined the effects of urbanization on Key deer fawn mortality and range size. While the authors found fawn mortality decreased over the past 30 years, they also found that range sizes decreased and suggested that if ranges continued to shrink fawn mortality could increase due to lack of resources (Peterson et al. 2004). Finally, Peterson et al. (2005) reported changes in deer sociobehavior due to illegal feeding. The authors suggested that increased group sizes and densities around feeders were indicators of increased domestication of Key deer (Peterson et al. 2005).

Research Objectives

My goal was to evaluate the impacts of 30-years of increasing urban growth on the Key deer population. Key deer have been the focus of 2 comprehensive studies. The first study (Hardin 1974, Silvy 1975; hereafter referred to as historic) was conducted from December 1968 through June 1972 during a time of low human population density and urbanization (0.22 houses/ha) on BPK. The second study (Lopez 2001; hereafter referred to as current) was conducted from January 1998 through December 2000 during a time of high human population density and urbanization (0.96 houses/ha) on BPK. I evaluated whether urbanization has negatively impacted Key deer by testing the following research hypotheses:

1. Key deer have become more urbanized due to increased human population growth and development.

2. Increased urbanization and anthropogenic risk factors have negatively impacted Key deer population dynamics.

3. Increased urbanization has resulted in changes in Key deer behavior and increased domestication.

4. Increased urbanization has resulted in long-term physiological effects on Key deer morphology. Specific predictions as related to each of these hypotheses are included (Table 5.1).

Table 5.1. Summary of research hypotheses and predictions used to determine the impacts of urbanization on the Key deer population by population variable.

Population Variable

Habitat Use

> *Ha:* Key deer have become more urbanized due to increased human population growth and development.

> *Predictions:* 1. Urban use by Key deer has increased between study periods.

> 2. Urban use by Key deer has increased for all deer rather than dichotomized populations (i.e., urban vs. wild deer).

> 3. Habitat selection by Key deer has changed and selection for urban areas has increased.

Population Dynamics

> *Ha:* Increased urbanization and anthropogenic risk factors have negatively impacted Key deer population dynamics.

> *Predictions:* 1. Survival for more urbanized deer (higher percent urban use) is lower due to higher mortality risk.

> 2. Recruitment (fawn:doe ratio) has decreased between study periods.

Table 5.1. Continued.

Population Variable

Behavior

    Ha: Increased urbanization has resulted in changes in Key deer behavior and increased domestication.

    Predictions: 1. Flight distance (the distance a person can approach before a deer flees) is shorter in urban than in wild areas.

        2. Flight distances are negatively correlated with deer urban use.

        3. Range sizes are negatively correlated with deer urban use.

        4. Group sizes are higher during the current study.

        5. Group sizes in urban areas are higher than in wild areas.

        6. Density in urban areas is higher during the current study.

Morphology

    Ha: Increased urbanization has resulted in long-term physiological effects on Key deer morphology.

    Predictions: 1. Body mass of captured adult Key deer has decreased between study periods.

        2. Carcass weights from Key deer mortalities will have decreased between study periods.

STUDY AREA

The Florida Keys are a chain of small islands approximately 200-km long extending southwest from peninsular Florida in Monroe County, Florida. Soil types vary from marl deposits to bare rock of the oolitic limestone formation (Dickson 1955). Island vegetation varies by elevation with red (*Rhizophora mangle*), black (*Avicennia germinans*), and white mangroves (*Laguncularia racemosa*), and buttonwood (*Conocarpus erectus*) forests occurring near sea level (maritime zones). As elevation increases inland, maritime zones transition into hardwood (e.g., Gumbo limbo [*Bursera simaruba*], Jamaican dogwood [*Piscidia piscipula*]) and pineland (e.g., slash pine [*Pinus elliottii*], saw palmetto [*Serenoa repens*]) upland forests with vegetation intolerant of salt water (Dickson 1955, Folk 1992).

METHODS

Data Collection

Key deer were captured, marked, and/or radiocollared during 2 separate study periods from December 1968 through June 1972 (historic; Hardin 1974, Silvy 1975) and from January 1998 through December 2000 (current, Lopez 2001) on BPK and NNK. Capture techniques included the use of portable drive-nets, drop-nets, and hand capture (Silvy 1975, Silvy et al. 1975, Lopez 2001). Captured deer were physically restrained for an average of 10-15 minutes, ear tattooed, and radiocollared with battery-powered mortality-sensitive radiotransmitters (AVM Electonics Corporation, Champaign, Illinois, USA, 1968-1972; Advanced Telemetry Systems, Isanti, Minnesota, USA, 1998-2000).

Deer were classified by sex and age when captured. Three age-classes were used: fawn (<1 year), yearling (1-2 years), and adult (≥2 years; Lopez et al. 2003). Deer were monitored and telemetry locations were recorded 6-7 times per week at random intervals. With each day divided into 6 4-hr segments, 1 4-hr segment was randomly selected each day to locate all deer. If a mortality signal was detected, deer were immediately located and necropsied to determine cause of death (Nettles 1981). Deer were censored from the data set after their last known encounter if their radios failed or disappeared (Pollock et al. 1989). Deer locations were recorded on maps and entered into a GIS database (ArcView GIS, ESRI 1999).

Habitat Use

I evaluated the hypothesis that Key deer have become more urbanized due to human population growth and development by examining changes in percent urban use and habitat selection by deer between the 2 studies. Habitat use estimates were obtained from radiocollared deer.

Urban Use.—I tested the prediction that Key deer use of urban areas had increased between the historic (1968-1972) and current (1998-2000) study periods using radiotelemetry data. Deer locations were classified by habitat as urban and wild (non-urban). I estimated urban use by deer by calculating (1) the percent of urban radiotelemetry locations per deer and (2) the percent of urban area in each deer's range. I tested for differences in urban percent of locations and ranges between the current and historic study period using t-tests for unequal variances (Ott and Longnecker 2001). I also evaluated the concept of an "urban deer." It has been suggested that urban use by

deer was a continuum rather than dichotomously polarized as urban and wild deer (Peterson et al. 2005). Thus, I constructed histograms to evaluate the distribution of urban use by deer.

Habitat Selection.—I tested the prediction that urbanization has altered Key deer selection of habitats by increasing the use of developed (urban) areas using radiotelemetry data collected in the historic (1968-1972) and current (1998-2000) study periods. Vegetation coverage maps from the Advanced Identification of Wetlands Project (MacAulay et al. 1994) were used to classify habitat into 6 vegetation types (hammock, pineland, freshwater marsh, buttonwood, mangrove, and developed; Lopez et al. 2004*b*). Historical vegetation coverages were created by reclassifying developed areas to original vegetation types as described by Lopez et al. (2004*b*). I evaluated first-, second-, and third-order habitat selection (Johnson 1980, Lopez et al. 2004*b*) by Key deer. First-order selection was defined as the habitat use by deer radiotelemetry locations compared to habitat availability in the study area. Second-order selection was defined as habitat use in a deer's range compared to habitat availability in the study area. Third-order selection was defined as habitat use by deer point locations compared to habitat availability in a deer's range. I calculated a habitat selection ratio for each deer as $S = ([U + 0.001] / [A + 0.001])$, where U was equal to observed use and A to expected use (Manly et al. 2000). I calculated the mean ratio for current and historic deer to identify differences in habitat use between periods. I limited analysis to BPK because historic data did not include NNK.

Population Dynamics

I tested the hypothesis that increased urbanization and anthropogenic risk factors have negatively impacted Key deer population dynamics by evaluating (1) the influence of deer urban use on survival and (2) changes in recruitment rate between study periods.

Survival.—I predicted that percent urban use by deer would be negatively correlated with survival due to increased exposure to anthropogenic risk factors for deer. I used telemetry data collected from radiocollared deer during the historic (1968-1972) and current (1998-2000) study periods. Lopez et al. (2003) reported that fawn survival differed from yearling and adult survival for both sexes, thus, I only included yearling and adult radiocollared deer on BPK in analysis. Survival estimates were estimated using a known-fate model framework in Program MARK (White and Burnham 1999). Encounter history files were created for individual deer including sex, area (north and south BPK), and study period for input into Program MARK. A covariate for urban use by each deer based on the percent of urban radiotelemetry locations was also included. Twelve models were constructed based on sex, area, study, and urban use and combinations of each were evaluated using program MARK. Models were evaluated based on Akaike's Information Criterion (AIC_c) and the highest ranking model was selected to estimate survival (Burnham and Anderson 1998).

Recruitment.—I tested the prediction that recruitment has decreased between study periods using fawn and doe counts from deer census data. Road counts were conducted on various survey routes to estimate population density and structure from 1969-2001 on BPK and NNK. I limited data to the "Big Pine Key 44-mile" route

(BPK44m, Lopez 2001) because this survey was conducted in both the historic and current periods and covers the entire island. The BPK44m route is 71 km from the northern tip to the southern tip of the island. Weekly road counts were conducted ½ hour before sunrise (1969-1972) and 1½ hours before sunset (1998-2001). Road count data includes the location, sex, and age of deer seen. Fawn-doe ratios were calculated for each census observation group. Seasons were defined as winter (January-March), spring (April-June), summer (July-September), and fall (October-December). I used an ANOVA to test for differences in ratios by period and season. I used the Kolmogorov-Smirnov statistic with a Lilliefors significance level for testing normality and Levene's test for equality of error variances (SPSS 12.0.1, Ott and Longnecker 2001). Fawn:doe ratios were not normally distributed and were square root transformed ($\sqrt{y}$).

Behavior

I evaluated the hypothesis that increased urbanization has resulted in changes in Key deer behavior and increased domestication by examining the effect of urban use on Key deer flight distance and range size. I also examined differences in group size and density between urban and wild areas and study periods.

Flight Distance.—I tested the prediction that deer in urban areas are "tamer" and more approachable than deer in wild (non-urban) areas using flight distance data gathered from radiocollared deer during the current study period (1998-2000). I defined flight distance as the distance at which a person can approach a deer before it flees. Data were gathered during the regular monitoring of radiocollared deer during the current study. If deer were visually located, additional information was gathered including the

distance at which the deer fled when approached by the observer (flight distance). I used this data to examined whether deer flight distance varied based on the type of habitat (urban vs. wild) the deer was located in. I used t-tests assuming unequal variances to compare mean flight distance between urban and non-urban areas. I used a nonparametric Kruskal-Wallis test to compare mean flight distance between habitat types because data did not exhibit normality even when transformed. Further, I predicted that flight distance was negatively correlated with urban use. I explored the possible relationship between deer urban use and flight distance using regression analysis. I predicted that as urban use increased, flight distance would decrease. Thus, I tested whether urban use significantly influenced flight distance using linear regression and predicted a negative slope. Data were not normally distributed and were square root transformed ($\sqrt{y}$).

Range Size.—I tested the prediction that Key deer range sizes were negatively correlated with urban use using range estimates calculated from radiotelemetry data during the historic (1968-1972) and current (1998-2000) study periods. Annual Key deer ranges were calculated using a 95% fixed-kernel home-range estimator (Worton 1989, Seaman et al. 1998, Seaman et al. 1999) with the animal movement extension in ArcView (Hooge and Eichenlaub 1997). Calculation of the smoothing parameter (kernel width) was used in generating kernel range estimates (Silverman 1986). Only deer with $\geq$ 175 locations were used to calculate annual estimates. Only BPK deer ranges were included in analysis. For deer with >1 annual range estimate, only the most recent range was included. I used analysis of covariance to evaluate the relationship between annual

range size and urban use by deer. I included sex as a factor because of known sex differences in range size (Lopez et al. 2005) and urban use as a covariate. Urban use was calculated as the number of telemetry locations in urban areas divided by the total number of telemetry locations for each deer. I used the Kolmogorov-Smirnov statistic with a Lilliefors significance level for testing normality and Breush-Pagan test for equality of error variances (SPSS 12.0.1, Ott and Longnecker 2001). Annual range size data were not normally distributed and were natural log transformed [ln(y)].

Group Size and Density.—I tested 2 predictions regarding Key deer grouping behavior: (1) that group sizes in urban areas are greater than in wild areas, and (2) that group sizes in the current period are higher than in the historic period. I used BPK44m survey data from 1971 (January-December) and 1999 (January-December) to estimate average group size and density in urban and wild areas on BPK. Survey data included the location, age, sex, and markings of observed deer. I classified observation area as urban if developed and all other areas as wild. Seasons were defined as winter (January-March), spring (April-June), summer (July-September), and fall (October-December). I calculated the seasonal mean group size for each survey observation during 1971 and 1999. Group size data were not normal and I used Mann-Whitney and Kruskal-Wallis nonparametric tests for differences between year, area, and season.

Using the same survey data, I also tested the prediction that Key deer densities in urban areas are greater during the current than during the historic study period. I calculated the proportion of deer seen in urban and wild areas for each survey. I then classified each survey by season and calculated means and 95% confidence intervals. I

used ANOVA to test for differences in the proportion of deer seen in urban areas by season for each study period. I also estimated deer densities for urban and wild areas in 1971 and 1999 by multiplying the proportion of deer seen in each area and year by previously reported deer estimates on BPK (Lopez et al. 2004*a*).

Morphology

I evaluated the hypothesis that urbanization has resulted in long-term physiological effects on Key deer morphology by comparing (1) Key deer body mass of captured deer during the historic and current study periods, and (2) carcass weights of adult Key deer mortalities from 1969-2003.

I tested the prediction that adult deer body mass (weight) has decreased between study periods using live deer capture data. Deer were captured during the historic (1968-1972) and current (1998-2000) study periods and body mass, age, and sex were recorded. I included only adult deer in analysis due to small sample sizes for fawns and yearlings. If a deer was captured and weighed more than once during a study, I used the mean weight for that deer in analysis. I compared the average body mass of adult deer by sex and study period. I tested for differences between mean weights by period using t-tests with equal variances not assumed for each sex.

I also tested the prediction that Key deer body mass has decreased using U.S. Fish and Wildlife Service (USFWS) mortality data. Key deer mortality data has been actively collected and recorded by USFWS NKDR staff since 1968. Key deer mortalities were located by direct sightings, citizen reports, or observation of turkey vultures (*Cathartes aura*). Collected carcasses were necropsied immediately or held

frozen prior to necropsy examination. Carcass quality or ability to determine cause of

death ranged from good to marginal (Nettles 1981, Nettles et al. 2002). Age, sex, body

mass, and cause of death were recorded for each animal using procedures described by

Nettles (1981), and all mortality locations were recorded. I used USFWS key deer

mortality data from 1969-2003 to examine weight trends for adult deer. I grouped data

into 5-year categories and graphed mean weights with 95% confidence intervals for male

and female deer separately.

RESULTS

Habitat Use

Urban Use.—I analyzed the urban use (%) for 180 radiocollared deer (131

current, 49 historic). As predicted, my analysis indicated that mean percent urban use by

Key deer differed significantly between periods for point location ($t = 4.946$, $P < 0.001$)

and range ($t = 2.319$, $P = 0.022$) estimates. For both point locations and ranges, urban

use by deer was greater in the current period than in the historic period (Fig. 5.2). I also

predicted the distribution of urban use by deer was continuous and not dichotomous.

Histograms indicate that urban use by deer differed by period but was relatively

continuous during both periods (Fig. 5.3).

Habitat Selection.—Total deer used in analysis was 143 (94 current, 49 historic).

I analyzed first-, second-, and third-order habitat selection (Johnson 1980) by

radiocollared deer in the historic (1968-1972) and current (1998-2000) study periods.

Habitat selection ratios were interpreted as follows: > 1, habitat selected in greater

proportion than available (preferred); < 1, habitat selected in lesser proportion than

available (avoided), and = 1, habitat selected in proportion to availability (used

proportionately). In the current study, deer preferred developed areas under all 3 orders

of selection (Figs. 5.4, 5.5, 5.6). However, deer in the historic study preferred developed

areas only under second-order analysis. Deer in the historic study avoided urban areas

under first- and third-order analyses.
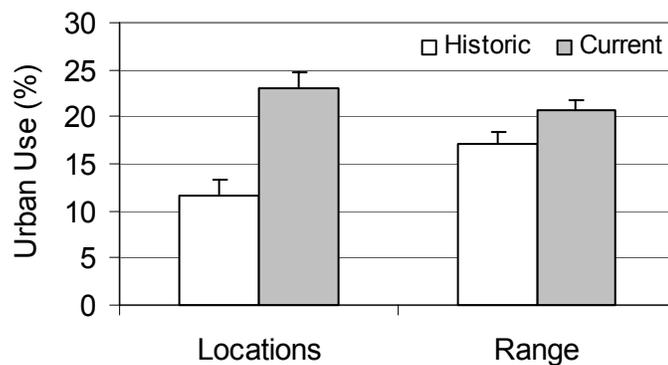
Fig. 5.2. Key deer urban use (mean, 1SE) for point locations and annual ranges by

period (historic, 1968-1972; current, 1998-2000).

(a)
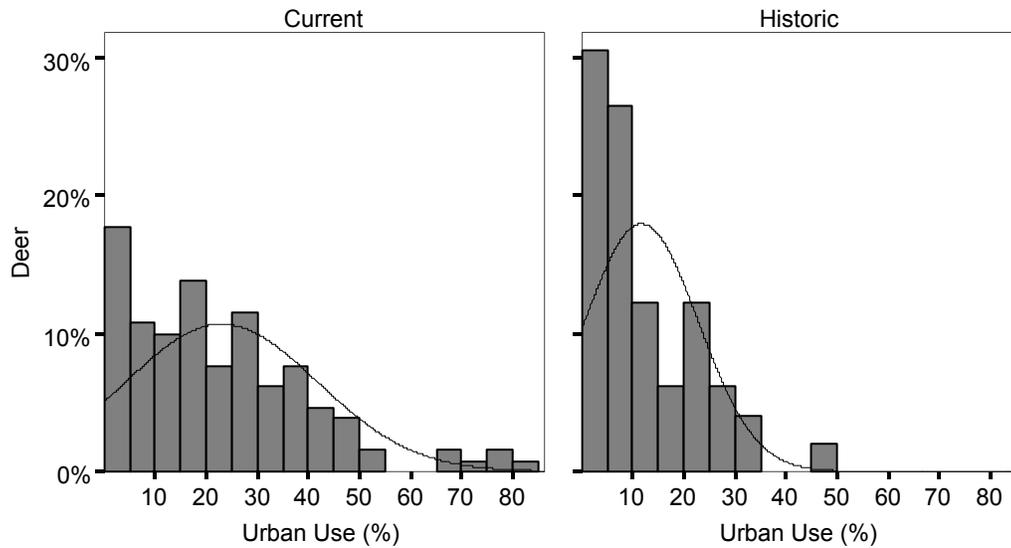
(b)
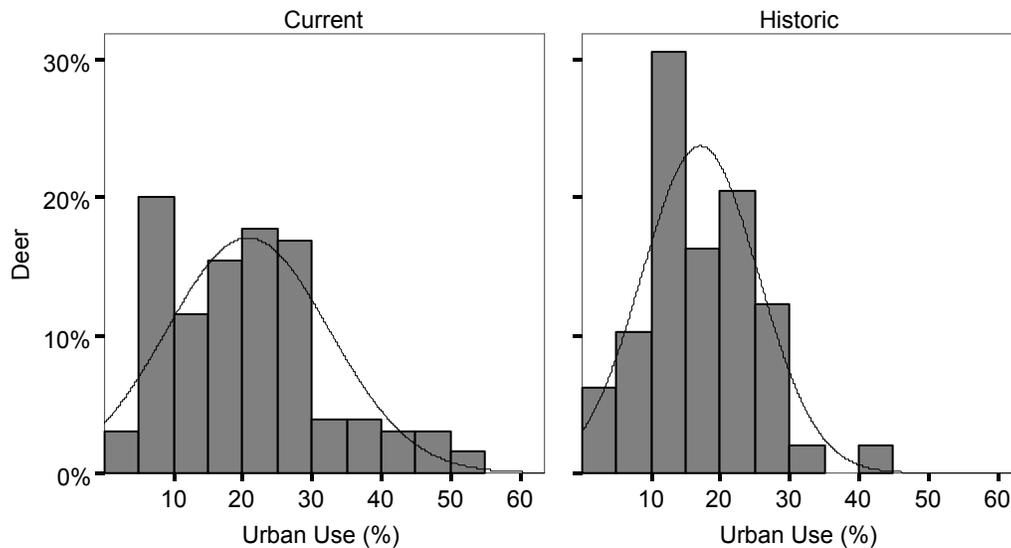
Fig. 5.3. Histograms of (a) percent urban radiotelemetry locations and (b) percent urban area in ranges for Key deer by period (historic, 1968-1972; current, 1998-2000).
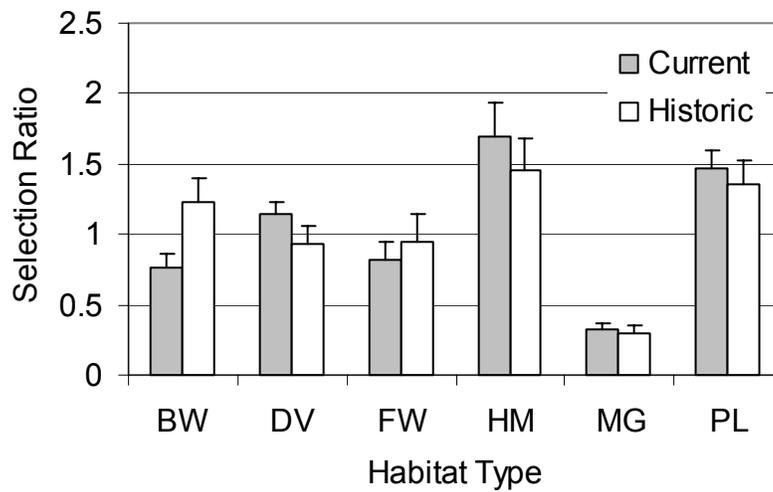
Fig. 5.4. Key deer habitat first-order (point-study area, Johnson 1980) selection ratios

(mean, 1SE), by period and habitat type (BW = buttonwood, DV = developed, FW =

freshwater marsh, HM = hammock, MG = mangrove, PL = pineland) during the historic

(1968-1972) and current (1998-2000) study periods on Big Pine Key, Florida.
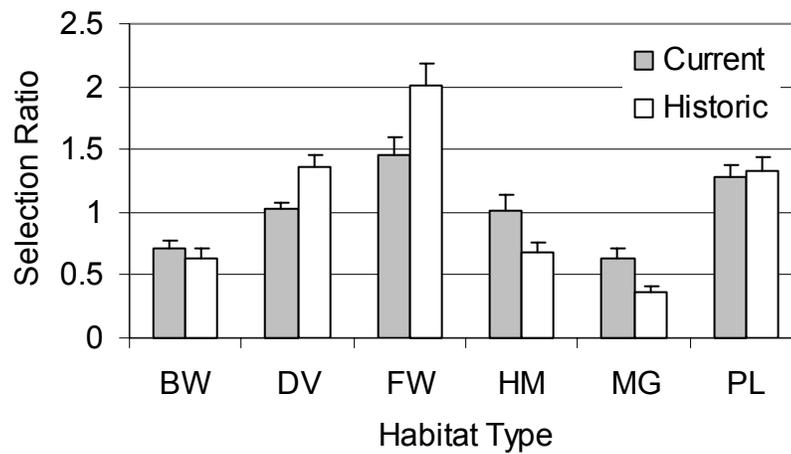
Fig. 5.5. Key deer habitat second-order (range-study area, Johnson 1980) selection ratios (mean, 1SE), by period and habitat type (BW = buttonwood, DV = developed, FW = freshwater marsh, HM = hammock, MG = mangrove, PL = pineland) during the historic (1968-1972) and current (1998-2000) study periods on Big Pine Key, Florida.
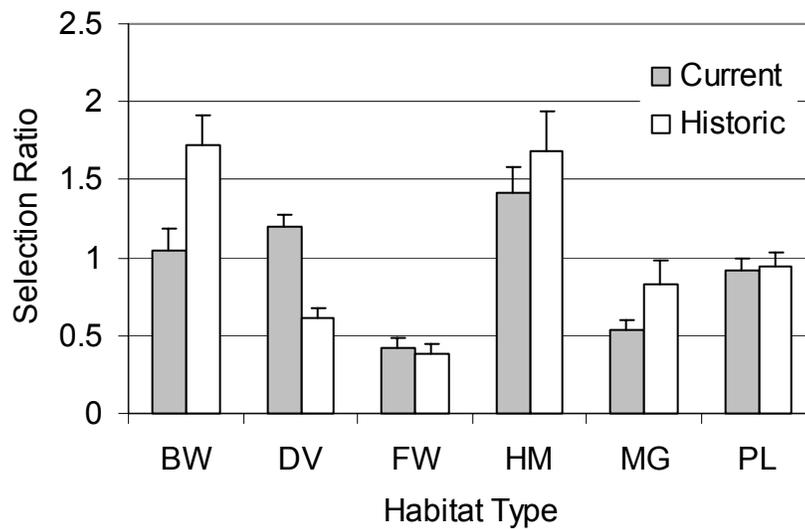
Fig. 5.6. Key deer third-order (point-range, Johnson 1980) habitat selection ratios

(mean, 1SE), by period and habitat type (BW = buttonwood, DV = developed, FW =

freshwater marsh, HM = hammock, MG = mangrove, PL = pineland) during the historic

(1968-1972) and current (1998-2000) study periods on Big Pine Key, Florida.

Population Dynamics

Survival.—Key deer survival was estimated using 107 radiocollared yearling and adult deer on BPK. Models were ranked based on AICc score calculated in program MARK. The highest ranking model included sex, study period, and urban use (Table 5.2). Thus, this model was used to estimate Key deer survival and evaluate the influence of each factor on Key deer survival. Overall, Key deer survival was higher in the historic study than in the current study, and survival was higher for female than for male deer (Table 5.3). Within each of these categories (study*period*sex) survival was positively correlated with urban use by deer. These results were contradictory to my predictions and suggest that more urbanized deer have higher survival than deer that are less urbanized (Fig. 5.7).

Recruitment.—Total fawn:doe ratios used in analysis was 95 (25 historic, 70 current). Results indicate that mean fawn-doe ratios differed by period ($F = 14.963$, $P < 0.001$) and season ($F = 61.100$, $P < 0.001$) with no period*season interaction ($F = 1.293$, $P = 0.282$) and an adjusted $R^2 = 0.729$. As predicted, fawn:doe ratios were lower during the current study than during the historic study. Specifically, current fawn:doe ratios were significantly lower than historic ratios for fall and winter seasons (Fig. 5.8).

Table 5.2. Candidate models and selection results for estimated survival for yearling and adult Key deer ($n = 107$) on Big Pine Key, Florida.

Candidate Model	No. of Parameters	Δ_i^{1}	Akaike Weight (w_i)	Evidence Ratio (w_1 / w_i)
$S_{Sex, Study, Urban Use}$	4	0.000	0.28	1.00
$S_{Sex, Study}$	3	0.455	0.22	1.27
$S_{Sex, Study, Area, Urban Use}$	5	1.357	0.14	2.00
$S_{Sex, Study, Area}$	4	1.404	0.13	2.15
$S_{sex, Area}$	3	1.725	0.11	2.55
S_{Sex}	2	3.401	0.05	5.66
$S_{Sex, Urban Use}$	3	4.665	0.03	9.33
S_{Area}	2	5.685	0.02	14.00
S_{Study}	2	6.614	0.01	28.00
$S_{Urban Use}$	2	10.503	0.00	53.84
$S_{Seasonal Urban Use}$	12	13.372	0.00	>100
$S_{Urban Use/changes w/each interval}$	5	31.091	0.00	>100

[1]Minimum $AIC_c = 146.607$

Table 5.3. Annual survival estimates and variances for yearling and adult Key deer on Big Pine Key by period (historic, 1968-1972; current, 1998-2000) and sex.

Sex	Study Period	Range Value	Urban Use Covariate (Prop.)	Annual Survival Estimate	Annual Survival SE
Male	Historic	Minimum	0.03	0.903	0.358
		Median	0.13	0.922	0.367
		Maximum	0.24	0.939	0.367
	Current	Minimum	0.03	0.473	0.000
		Median	0.27	0.638	0.005
		Maximum	0.92	0.899	0.379
Female	Historic	Minimum	0.00	0.976	0.251
		Median	0.06	0.979	0.226
		Maximum	0.34	0.991	0.149
	Current	Minimum	0.00	0.825	0.105
		Median	0.22	0.889	0.200
		Maximum	0.71	0.962	0.273

Male Annual Survival

Female Annual Survival
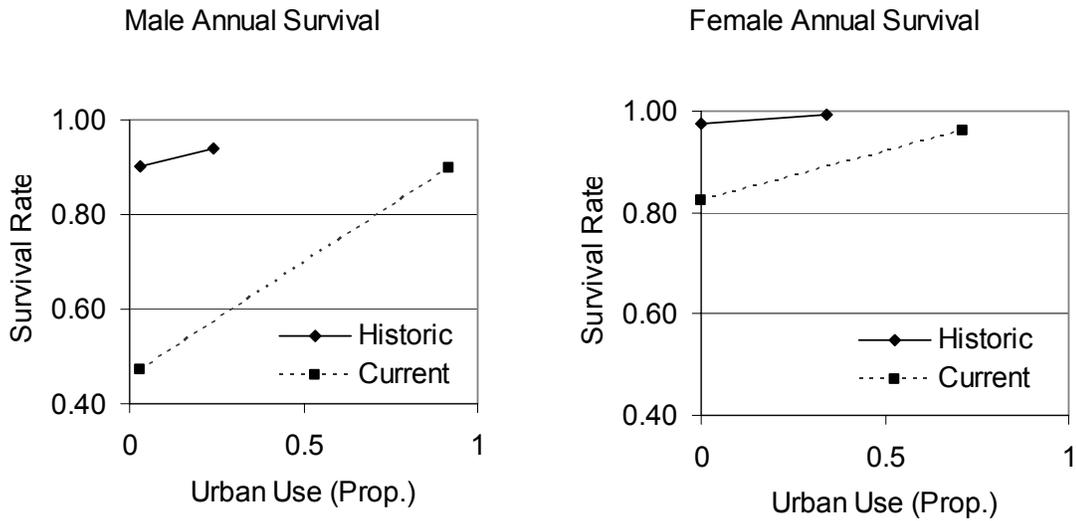
Fig. 5.7. Annual survival for yearling and adult Key deer on Big Pine Key, Florida by period (historic, 1968-1972; current, 1998-2000), sex, and urban use (minimum and maximum values).
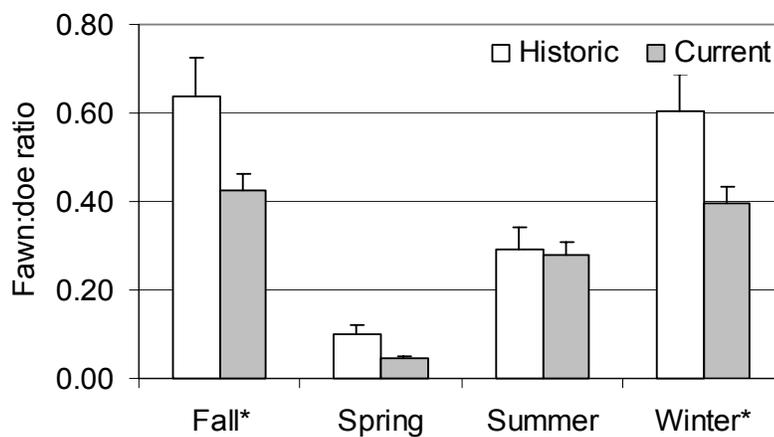
Fig. 5.8. Key deer fawn-doe ratios (mean, 1SE) by season and period (historic, 1968-1972; current, 1998-2000). Asterisk (*) indicates significant difference at $\alpha = 0.05$.

Behavior

Flight Distance.—I examined possible differences in Key deer flight distance

based on habitat type using 746 observations from 117 deer on BPK and NNK from

1998 to 2000. Contrary to my predictions, I found no significant difference (t = -0.822,

P = 0.412) between mean flight distance in urban (mean = 6.3 ft) and non-urban (mean =

7.3 ft) areas (Fig. 5.9). I also tested for differences between mean flight distances in 5

different habitat types. Freshwater marsh was excluded from analysis because of small

sample size (n = 1). Mean flight distance was greatest for buttonwood (mean = 13.4 ft)

and lowest for pineland (mean = 5.7 ft), however, differences between types were not

significant (X^2 = 8.304, df = 4, P = 0.081) (Fig. 5.10).

I tested the relationship between urban use and flight distance using data

collected from 117 deer on BPK and NNK. Linear regression analysis suggests that

urban use by deer is a significant predictor of flight distance (F = 6.514, P = 0.012) with

an adjusted R^2 = 0.045. As urban use increased, flight distance decreased (slope =

-1.959). These results were consistent with my predictions, however, the low adjusted

R^2 value suggests that urban use explained only a small portion of the variability in flight

distance.
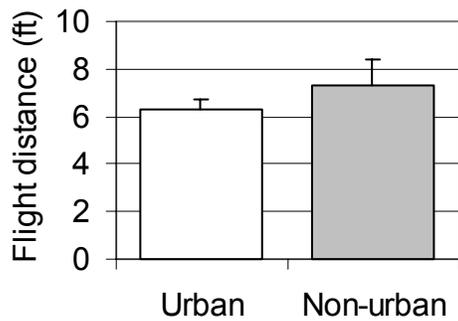
Fig. 5.9. Key deer flight distance (mean, 1SE) in urban and non-urban areas on Big Pine and No Name Keys, Florida, 1998-2000.
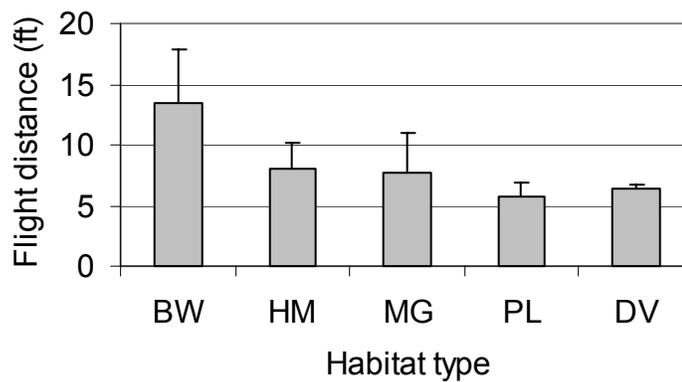
Fig. 5.10. Key deer flight distance (mean, 1SE) by habitat type (BW = buttonwood, DV = developed, FW = freshwater marsh, HM = hammock, MG = mangrove, PL = pineland) on Big Pine and No Name Keys, Florida, 1998-2000.

Range Size.—I removed 4 ranges with standardized residuals ±3.0. Total annual

ranges used in analysis was 45 (12 male, 33 female). Transformed data were normal but

did not have equal variances. Analysis of covariance suggests that mean range size

differed by sex ($F = 18.718$, $P < 0.001$) and was influenced by urban use of deer ($F =$

10.957, $P = 0.002$) with adjusted $R^2 = 0.413$. Analysis results confirmed my prediction

that as percent urban use by deer increases, range size decreases (Fig. 5.11).

Group Size and Density.—I calculated average group size and deer density in

urban and wild areas using 26 and 48 surveys conducted from January-December 1971

and January-December 1999, respectively. Contrary to my predictions, no differences

were found between deer mean group size by area (urban or wild; $P = 0.591$) or season

($P = 0.294$) in 1971 (historic study, Fig. 5.12). However, results for the current study

(1999) were as predicted with significant differences found between deer mean group

size by area ($P = 0.021$) and season ($P < 0.001$) in 1999 (current study, Fig. 5.13). I also

predicted that group sizes have increased between study periods. As expected, average

yearly group size in 1999 was significantly greater than in 1971 for both urban ($P <$

0.001) and wild ($P < 0.001$) areas (Fig. 5.14).
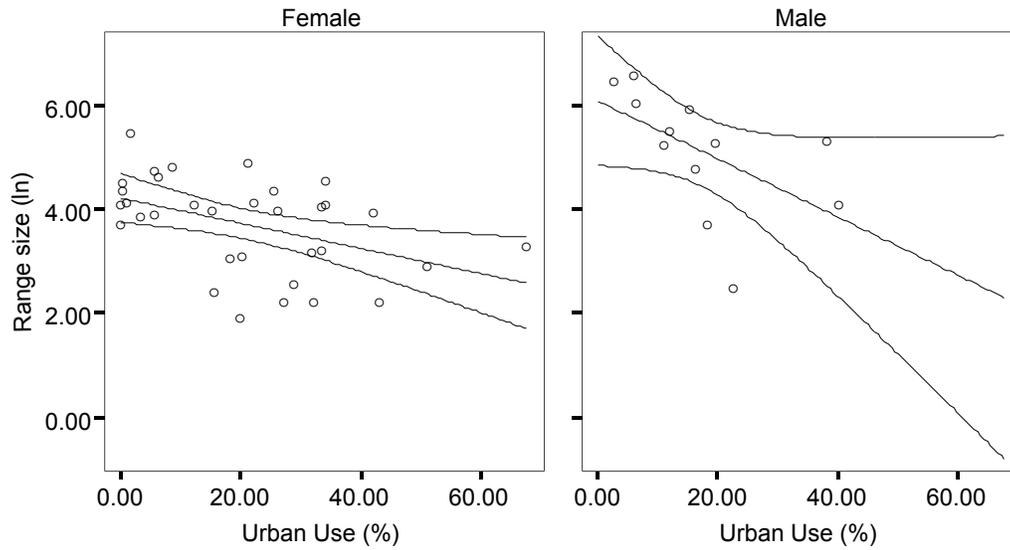
Fig. 5.11. Trends of adult Key deer annual range size (ln transformation) by urban use for females and males on Big Pine Key, 1968-2000.
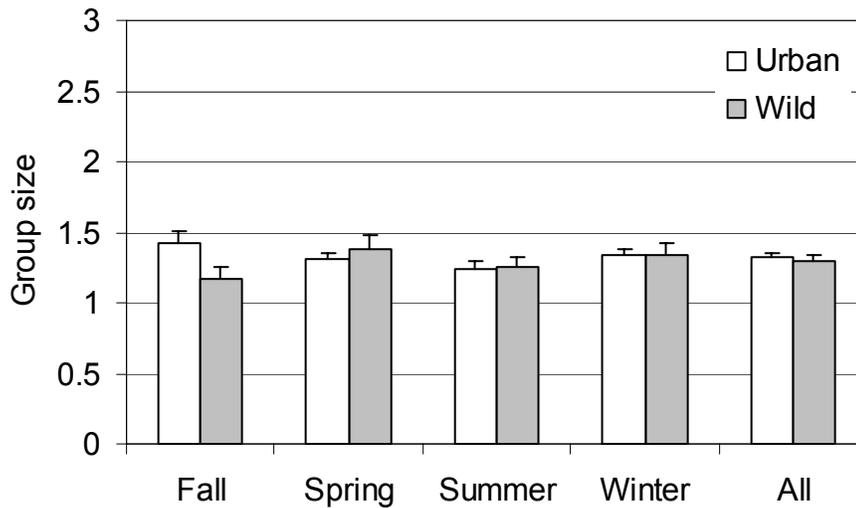
Fig. 5.12. Key deer group size (mean, 1SE) by season in urban and wild areas during Big Pine Key 44-mile deer surveys in 1971 (historic).
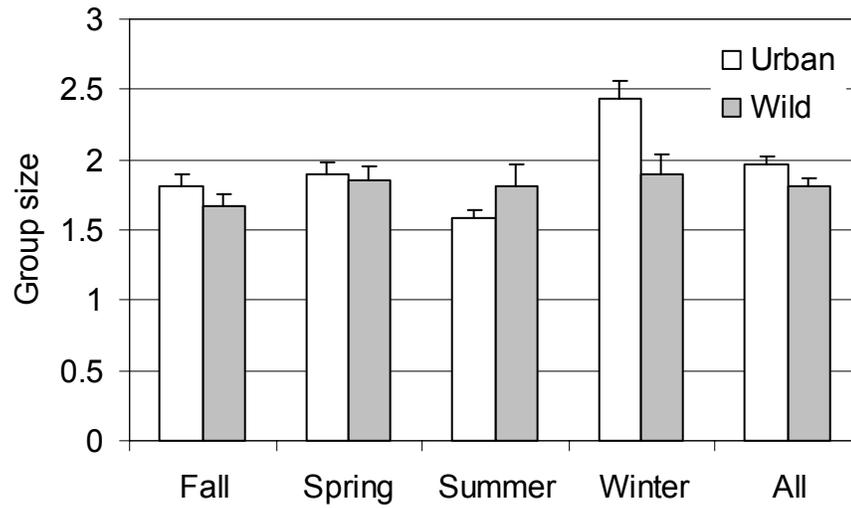
Fig. 5.13. Key deer group size (mean, 1SE) by season in urban and wild areas during

Big Pine Key 44-mile deer surveys in 1999 (current).
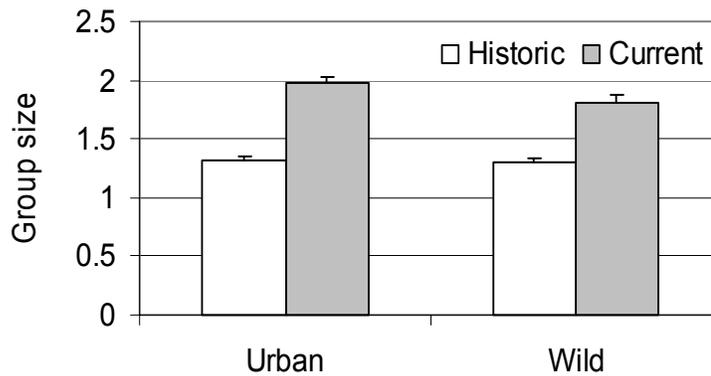
Fig. 5.14. Key deer group size (mean, 1SE) in urban and wild areas during Big Pine Key

44-mile deer surveys in 1971 (historic) and 1999 (current).

I also calculated the proportion of total deer seen in urban areas for each survey conducted in 1971 and 1999. Results indicate that the proportion of urban deer did not differ seasonally in 1971 ($F = 2.643$, $P = 0.071$) (Fig. 5.15). However, seasonal differences were found in 1999 ($F = 7.553$, $P < 0.001$) (Fig. 5.15). In the current survey, the proportion of deer seen in urban areas was significantly greater in winter than summer ($P < 0.001$) and fall ($P = 0.007$). The overall proportion of deer seen in urban areas in 1971 (0.70) was slightly, although not significantly, higher than in 1999 (0.65). This was contradictory to what I predicted especially considering the proportion of urban area available on BPK was smaller in 1971 (0.13) than in 1999 (0.24). I also estimated deer density by area and calculated the number of deer per hectare in urban and wild areas for each year (Table 5.4). Estimated deer densities varied by area with deer occurring at much higher densities in urban areas. Deer densities in urban areas were similar although slightly higher in 1971 (0.51 deer/ha) than in 1999 (0.43 deer/ha). The greatest difference was found in wild areas where deer density was much higher in 1999 (0.07 deer/ha) than in 1971 (0.03 deer/ha).
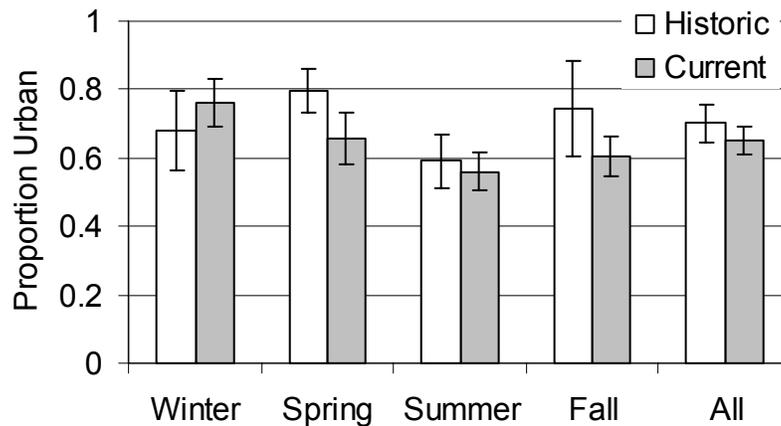
Fig. 5.15. Key deer seen (proportion) in urban areas during Big Pine Key 44-mile deer

surveys in 1971 (historic) and 1999 (current).

Table 5.4. Group size and density estimates for Key deer in urban and wild areas on Big

Pine Key, Florida, 1971 and 1999.

	Urban Areas		Wild Areas	
	1971	1999	1971	1999
Group size (mean)	1.31	2.23	1.29	1.92
Proportion of deer observed	0.70	0.65	0.30	0.35
Estimated deer density	169	264	72	142
Proportion of available habitat[a]	0.13	0.24	0.87	0.76
Density (deer/ha)	0.51	0.43	0.03	0.07

[a]Area calculated from 1970 and 2000 vegetation maps.

Morphology

I analyzed the weights of 252 adult deer captured during the historic (72 female, 38 male) and current (87 female, 55 male) study periods. Adult deer capture weights differed significantly between periods for both males ($t = 3.760$, $P < 0.001$) and females ($t = 5.249$, $P < 0.001$). I predicted that weights for Key deer have decreased between study periods. However, results for both female and male deer were contrary to what I predicted. Current adult female weights (mean = 70.5 lbs) were significantly greater than historic weights (mean = 63.3 lbs) (Fig. 5.16). Likewise, current adult male weights (mean = 94.1 lbs) were also significantly greater than historic weights (mean = 80.3 lbs) (Fig. 5.16). In my review of the USFWS mortality data, I removed data outliers (6 female, 3 male) and analyzed 605 male and 366 female mortality weights from 1969-2003. Contrary to my predictions, mortality data indicated increasing trends in weight by time for both males and females similar to analysis results for capture data (Fig. 5.17).
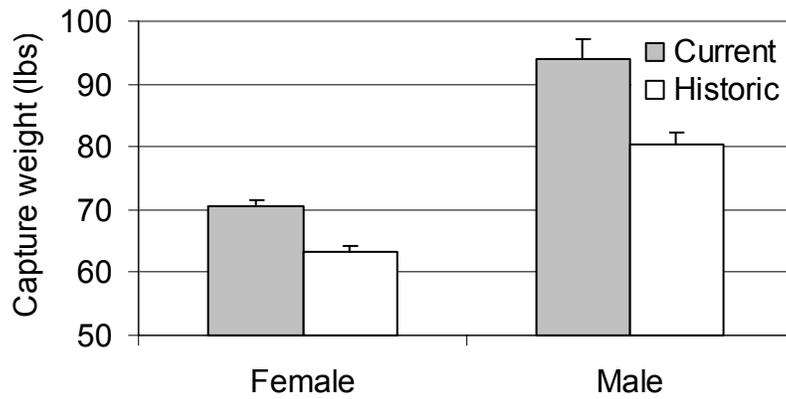
Fig. 5.16. Key deer capture weights (mean, 1SE) for adult deer by sex and period (historic, 1968-1972; current, 1998-2000).
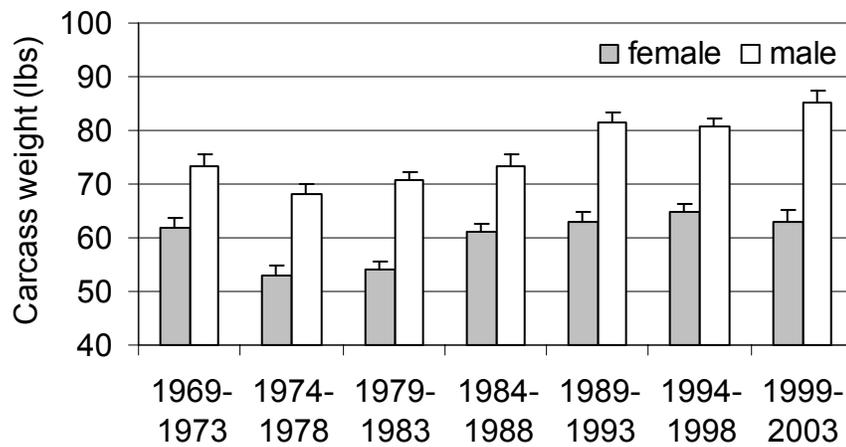
Fig. 5.17. Key deer carcass weights (mean, 1SE) from U.S. Fish and Wildlife Service mortality data for adult female and male deer in 5-year increments, 1969-2003.

DISCUSSION

Urbanization of Key Deer

Urban Use.—Radiotelemetry data suggests Key deer are more urbanized today than they were 30 years ago. The percent of deer telemetry locations in urban areas in the current study was nearly double the percent in the historic study (Fig. 5.2). Similarly, the percent of urban area in radiocollared deer ranges was significantly greater in the current study than in the historic study (Fig. 5.2). I also found that deer use of urban areas was fairly continuous and not polarized at either end of the scale (Fig. 5.3). This suggests that deer are not dichotomously "wild" or "urban" but are using different types of habitat to varying degrees. However, it appears from this analysis that deer in the historic study were at the lower end of the continuum of urban use when compared to the current study (Fig. 5.3).

Habitat Selection.—To better understand the factors influencing deer use of habitat, I compared habitat selection by deer between study periods. For instance, the amount of urban area available to deer has increased from 13% to 24% between the study periods (Table 5.4). Thus, I examined whether the increase in urban use by deer was due to increased availability by calculating a habitat selection ratio. I analyzed first-, second-, and third-order habitat selection (Johnson 1980) by radiocollared deer in the historic and current study periods. Since the focus of this paper is on the effects of urbanization on Key deer, I will limit my discussion to deer use of urban or developed habitat. In the current study, deer preferred urban areas under all 3 orders of habitat selection; whereas deer in the historic study preferred urban areas under only 1 order of

selection (Figs. 5.4, 5.5, 5.6). These results suggest that the increased use of urban areas by Key deer is not a function of increased availability but rather a change in deer behavior or preference.

Population Dynamics

Survival.—I evaluated the influence of deer urban use on survival for Key deer. My results suggest that urban use is an important factor in explaining deer survival (Table 5.2). However, results were converse to my predictions. I predicted that increased threats due to anthropogenic risk factors (i.e., deer-vehicle collisions, dog attacks, fence entanglement) in urban areas would result in decreased survival for more urbanized deer. However, my analysis indicated that as percent urban use by deer increases, survival also increases (Fig. 5.7). These results suggest that the urbanization of Key deer does not have a negative impact on the survival of the deer. While there are many possible explanations for these results, I speculate that this positive relationship between deer urban use and survival is due to behavioral adaptations by Key deer. More urbanized Key deer may have learned to avoid or compensate for urban risk factors such as roads, dogs, fences, and swimming pools. This may explain why male deer with low urban use had the lowest survival rate. White-tailed deer are territorial and the current deer population density on BPK is high (Lopez et al. 2004*a*) which may require yearling males to disperse greater distances to find a home range. This dispersal through urban areas may lead to lower survival due to inexperience with anthropogenic risk factors. If this explanation is correct, then the urbanization of Key deer may be a behavioral response stimulated by urban development and perpetuated by natural selection.

Recruitment.—I examined changes in recruitment by calculating seasonal fawn:doe ratios from surveys conducted in the historic and current study periods. As predicted, ratios were lower during the current study and significantly lower in fall and winter (Fig. 5.8). There are many possible explanations for this decrease in recruitment. Lopez et al. (2003) reported that urbanization and its associated risk factors are the primary mortality causes for Key deer. Thus, increased fawn mortality would be the most obvious explanation. However, Peterson et al. (2004) found Key deer fawn mortality has decreased since the historic study. Another possible explanation is decreased fitness in deer due to lack of resources or higher stress. However, my analysis of Key deer morphology suggests that this is not the likely cause since deer body mass has increased since the historic study. Therefore, I suspect that decreased recruitment is a density dependent response to the Key deer population nearing carrying capacity (K) (Halls 1984) and predict that without increased dispersal to other islands, recruitment will continue to decrease in the future as deer densities exceed K.

Behavior

Flight Distance.—Domestication of Key deer has been speculated but few studies confirm these claims (Petersen et al. 2005). I addressed the issue of domestication by examining the flight distance or the closest distance a person can approach a deer before it flees. I found flight distances in urban areas were smaller, though not significantly different, than in wild areas. However, I attribute the lack of significance to the high variability of deer responses in non-urban areas (Figs. 5.9, 5.10). Further, I found that on an individual deer basis, flight distance decreased as urban use increased. This

confirms my predictions and suggests that the more urbanized a deer is, the more

approachable or tamer it is. Although this relationship was significant, the model had a

small adjusted R^2 value suggesting a weak relationship and that further research is

warranted.

Range Size.—Lopez et al. (2005) reported Key deer range sizes have decreased

as a density dependent response to increased deer densities. I evaluated the influence of

urbanization on range size and found that as urban use by deer increased, range size

decreased (Fig. 5.11). In other words, the more urbanized a deer was, the smaller its

annual range size. These results are consistent with my predictions and I offer several

explanations for these results. First, urban areas may provide adequate resources in a

smaller area than wild areas. Deer that spend more time in urban areas may not have to

travel as far to meet all of their nutritional needs. Second, differences in deer densities

between urban and wild areas may be influencing range size. Deer densities are higher

in urban areas (Table 5.4) and decreased range sizes for urbanized deer may be a density

dependent response. Third, changes in range size could be a result of changes in Key

deer behavior due to increased domestication.

Group Size.—I analyzed survey data for differences in group size between study

periods (historic and current) and area type (urban or wild). I found differences between

periods within urban and wild areas (Fig. 5.14). In the current study, Key deer group

size was significantly greater than in the historic study for all seasons in both urban and

wild areas (Fig. 5.13). During the historic study, group sizes were fairly constant in all

seasons and all area types (Fig. 5.12). However, during the current study, group sizes

varied seasonally with winter in urban areas having the largest group sizes (Fig. 5.13).

This increase in group size indicates a modification in deer behavior. Peterson et al.

(2005) found similar results with larger group sizes associated with increased urban

feeder use by Key deer.

　　　　Density.—I examined the effects of urbanization on Key deer density and

distribution on BPK. According to my analysis, the proportion of deer seen in urban

areas was similar during both study periods (Fig. 5.15). This was contradictory to what I

expected considering the proportion of available urban area on BPK almost doubled

from the historic (13%) to the current (24%) study (Table 5.4). Thus to better

understand deer densities, I estimated the proportion of deer in each area during each

study and calculated the number of deer per hectare. For both study periods, deer

densities were much higher in urban areas than in wild areas. Deer densities in urban

areas were similar in the historic (0.51 deer/ha) and current (0.43 deer/ha) studies (Table

5.4). The greatest increase was seen in wild areas where deer density in the current

study (0.07 deer/ha) was over twice that of the historic study (0.03 deer/ha).

　　　　These results suggest the greatest amount of deer population growth has occurred

in the wild areas of BPK. While the amount of wild area decreased from 1971-1999, the

estimated deer population in wild areas increased by 97%. Furthermore, the amount of

urban area increased from 1971-1999, yet the estimated deer population in urban areas

only increased by 56%. One possible explanation is that increased mortality risks in

urban areas has resulted in slower deer population growth. However, my survival

analysis suggests otherwise and I hypothesize that, because deer density in urban areas

remained relatively constant between time periods, deer densities in urban areas may have been saturated in 1971 and increases in the deer population in these areas were the result of increased development of urban areas. I suspect that the deer population in urban areas is being regulated by density dependent and anthropogenic factors (i.e., deer feeders, vehicle traffic) while the deer population in wild areas (which was below carrying capacity) was able to exhibit rapid growth due to greater resource availability. However, I do not expect this trend to continue as the deer population in wild areas is likely approaching K and density dependent population regulation will continue to decrease the rate of growth (Halls 1984). Additionally, continued urbanization in the future will likely create increased mortality risks to Key deer due to increased fragmentation and vehicle traffic. I suspect this because of results from chapter II where the more urbanized, southern portion of BPK has higher deer mortality than the less urbanized, northern portion of the island.

Morphology

Lastly, I examined the effects of urbanization on Key deer body mass over the past 30 years. Changes in deer morphology have been reported in the literature as density dependent responses. Studies have shown that as deer densities increase and resources become limited, deer fitness and body mass decrease (Leberg and Smith 1993, Pettorelli et al. 2002, Keyser et al. 2005). I evaluated the effects of urbanization on Key deer morphology by analyzing changes in body mass (weight) between the historic and current study periods. During this time frame, the Key deer population on BPK increased from 247 to 406 animals. Nettles et al. (2002) reported increases in

population-limiting diseases in Key deer which is characteristic of a population that is at or above *K*-carrying capacity. I predicted the decrease in wild habitat due to urbanization combined with the increase in deer population size would have resulted in a decrease in deer body mass. However, my results suggest the opposite. Both female and male adult radiocollared deer weighed significantly more in the current than in the historic study (Fig. 5.16). Likewise, examination of USFWS deer mortality data indicated similar trends in the weights of adult deer carcasses from 1969-2003 (Fig. 5.17). So why then, has the Key deer population (which was well below *K*-carrying capacity in the historic study) increased in body weight? I hypothesize that urbanization has increased the amount of resources available to deer through the conversion of lowlands to uplands and the associated availability of urban resources (i.e., nonnative vegetation, refuge, and freshwater; Lopez et al. 2004*b*). However, I anticipate that this trend in increased body mass will not continue as the deer population reaches and exceeds *K* either through deer population growth and/or loss of usable space due to continued urbanization.

MANAGEMENT IMPLICATIONS

Human population growth and urban development in the Keys have altered the native habitat and impacted endemic species on the islands. The impacts of urbanization on Key deer have been the focus of speculation in the literature for over 30 years. My goal was to answer the question, has urbanization been beneficial or detrimental to Key deer? To answer this, I evaluated the impacts of urbanization on Key deer in a comprehensive study examining deer habitat use, population dynamics, behavior, and

morphology. My results suggest that Key deer have become more urbanized, using urban areas more today than they did 30 years ago. This urbanization of deer has resulted in the modification of Key deer behavior. For example, urban deer have smaller range sizes and shorter flight distances suggesting they are tamer and more domesticated. However, more urban deer have higher survival than their counterparts suggesting that deer have adapted to urban environments and that this behavioral adaptation has resulted in a positive impact on their ability to survive in urban areas. Problems still exist with mortality factors heavily impacting some portions of the deer population. Lower survival associated with less urban male deer are cause for concern.

Collectively, my results suggest that over the past 30 years Key deer have adapted to their urbanizing environment. Whether these behavioral adaptations (e.g., domestication) are desired is debatable, however, they appear to have provided Key deer with mechanisms to persist in an urbanizing environment. Furthermore, deer plasticity appears to have allowed Key deer to adapt and persist in a changing environment due to urbanization. The future ability of Key deer to persist in a continuously urbanizing environment is unknown. The impacts of additional urbanization on the deer population cannot be predicted. At some point, development will become unsustainable and it is impossible to predict where that point lies. Unfortunately, the negative impacts of urbanization on species often are not realized until after the damage has been done and these impacts are often irreversible. On a broader scale, more and more species will continue to be faced with the challenges of a changing environment as human population growth and urbanization continues to increase. The ability of species to adapt to these

changes will be a determining factor in their future success as the competition for

resources between man and wildlife continues.

CHAPTER VI

CONCLUSIONS AND IMPLICATIONS

In chapters II – V, I examined some of the current most pressing issues relevant to the conservation of Key deer (*Odocoileus virginianus clavium*) and the impacts of urbanization on the population. In this chapter, I highlight my research findings and provide recommendations for the future management of Key deer in the Lower Florida Keys.

First, I conclude that south Big Pine Key (SBPK) can be described as an ecological sink supplemented by deer dispersal from north Big Pine Key (NBPK) (chapter II). Dias (1996) noted that in a source–sink system, preserving only sink habitats will likely lead to population extinction. Care should be taken, especially when dealing with endangered species such as the Key deer, to preserve the source population and its habitat. Thus, I propose limiting future development in NBPK (high-quality source habitat). Future management goals should continue to address mortality factors on SBPK while still recognizing and preserving the important source population in NBPK. The US 1 highway corridor project, which includes the construction of fences and underpasses, has the potential to reduce Key deer mortality in SBPK due to vehicle collisions by 10% annually. This increase in survival could possibly stabilize ($\lambda = 1.0$) the SBPK deer population so it would no longer be a sink.

Second, the role of dispersal in the conservation and management of an endangered species such as Key deer requires a thorough understanding of the demographic and environmental factors influencing its population dynamics (chapter

III). While much is known about Key deer biology (e.g., survival, maternity, habitat use), changes in habitat conditions due to urban development will continue to prove challenging to managers. Habitat loss and fragmentation combined with increasing deer densities will have unpredictable effects on parameters such as dispersal. Through the use of simulation modeling, I examined the potential effects of changes in dispersal rates on the Key deer metapopulation and found that under all modeled scenarios the establishment of viable populations on Cudjoe Key and Sugarloaf Key by dispersal alone is unlikely within the next 20 years. I recommend the use of other methods (e.g., translocations) to supplement deer numbers on these islands in order to establish viable populations.

Third, survival estimates are yet another important population parameter important in the recovery and conservation of endangered populations (chapter IV). Previous estimates of survival have included the use of labor-intensive radiotelemetry data. An alternative method of estimating Key deer survival would allow USFWS biologists to obtain annual survival estimates using already collected mortality data. In comparing roadkill mortality data to radiotelemetry data, I found survival estimates to be similar, suggesting a possible alternative for estimating survival of Key deer. The long-term monitoring of Key deer mortality by USFWS biologists offers managers such an opportunity. Wildlife managers who consider the use of life table data in estimating survival should be aware of the potential biases. However, despite potential biases, resource managers that collect roadkill data (e.g., state and national parks, refuges, and

forests) may be able to provide an adequate estimate for survival of deer using life tables.

Finally, I evaluated the impacts of human population growth and urban development on Key deer over 30 years (chapter V). My chapter objective was to determine whether urbanization has been beneficial or detrimental to Key deer. To answer this, I evaluated changes in deer habitat use, population dynamics, behavior, and morphology. My results suggest that Key deer have become more urbanized, using urban areas more today than they did 30 years ago. This urbanization of deer has resulted in the modification of Key deer behavior. For example, urban deer have smaller range sizes and shorter flight distances suggesting they are tamer and possibly more domesticated. However, more urban deer have higher survival than their counterparts suggesting that deer have adapted to urban environments and that this behavioral adaptation has resulted in a positive impact on their ability to survive in urban areas. Problems still exist with mortality factors heavily impacting some portions of the deer population. Lower survival associated with less urban male deer are cause for concern.

In conclusion, collectively my results suggest that over the past 30 years Key deer have adapted to their urbanizing environment. Whether these behavioral adaptations (e.g., domestication) are desired is debatable, however, they appear to have provided Key deer with mechanisms to persist in an urban environment. This is not to say that continued urbanization would benefit Key deer; the future ability of Key deer to persist in an increasingly urban environment is unknown and cannot be predicted. Obviously, a threshold of urban development must exist where the Key deer population

can no longer be sustainable. Unlike other forms of habitat change or environmental

disturbances (e.g., forest succession, wild fire), urban development is in most cases

irreversible requiring careful planning in habitat conservation strategies. On a broader

scale, more and more species will continue to be faced with the challenges of a changing

environment as human population growth and urbanization continues to increase. The

ability of species to adapt to these changes will be a determining factor in their future

success as the competition for resources between man and wildlife continues.

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