

ABSTRACT

WALDSTEIN, ARIELLE HOPKINS. Raccoon Ecology and Management on Cape Lookout National Seashore, North Carolina. (Under the direction of Theodore R. Simons).

Removal of predators to protect prey species is a growing but contentious practice in wildlife management and the efficacy of this technique is often questioned. Predator removal is often done *ad hoc* and experimental manipulations and formal assessments of this technique are rare. Raccoons are considered a serious threat to threatened piping plovers, endangered sea turtles and several species of ground nesting shorebirds at Cape Lookout National Seashore, North Carolina. The National Park Service administers Cape Lookout and is generally reluctant to manage native species, preferring nature to self regulate ecosystems; however, generalist predators such as raccoons often exist at high densities due to anthropogenic influences. In such cases, protected areas such as national parks and wildlife refuges are becoming the last refuges for many rare, threatened and endangered species. As a result, managers at Cape Lookout needed to assess the feasibility of predator control as one of several management options. The first objective of my research was to document the ecology of raccoons on a single island (South Core Banks) of Cape Lookout in 2007 and 2008 and assess their role in the system, establishing baseline numbers for abundance and density. My second objective was to reduce the raccoon population and evaluate the response of 1) raccoon movement patterns and population dynamics, 2) predation rates on select prey species, and 3) productivity of prey species in the subsequent nesting season. I found that raccoon density on South Core Banks was relatively low compared to densities reported in the

literature and that animals were concentrated in areas of high human activity. Typical of an undeveloped barrier island, food resources were scarce and spatially clumped, further supporting spatial variability in raccoon density. The population appears to be limited in winter when food availability is low and mortality rates are highest in spring. Telemetry data indicate that most raccoons forage in the sound-side marshes of the island and that relatively few individuals forage on the beach where they are likely to encounter nesting shorebirds and turtles. To date we have observed little change in raccoon movement patterns following removal. Nest predation rates dropped for all species of concern in 2009 as productivity rose for all species except sea turtles. These preliminary results suggest that predator management may be a useful tool for protecting sensitive bird and turtle populations at Cape Lookout National Seashore. Monitoring of raccoons and prey species will continue for the next two years to determine the overall efficacy of predator management at this site.

Raccoon Ecology and Management on Cape Lookout National Seashore, North Carolina

by
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DEDICATION

To my Dad for teaching me to keep an open, inquiring mind and for his love, support and encouragement throughout my life.

BIOGRAPHY

I was born in Exeter, New Hampshire and grew up in Vermont. My family lived in a rural area and I was exposed from a very young age to wildlife and domestic livestock and developed a love for animals and the outdoors. I got my bachelor's degree in Zoology from McGill University in Canada where I was able to begin developing a knowledge and love for ecology and conservation. I was also able to experience field research for the first time and became a technician on a project studying the breeding biology of Eastern Loggerhead Shrikes. After graduation I joined the Peace Corps and spent two years in Senegal, West Africa in a village bordering a national park. I worked in community development focusing on fostering an ideal for protection of the environment through sustainable ecotourism activities. When I returned from the Peace Corps I took another technician job working with Common Loon monitoring in New Hampshire and finally came to NC State to pursue my Masters degree. The work for my Master's degree has greatly developed by skills and interest in management of species in protected areas because these are fast becoming the last refuges for the persistence of so many species.

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CHAPTER 1

Ecology of Raccoons on the Outer Banks of North Carolina

ABSTRACT

Raccoons (*Procyon lotor*) are common meso-predators of the southeastern United States that can thrive in human dominated landscapes. Barrier island populations are not well studied and they are of particular interest because of their effects on native species of conservation concern. We used mark-recapture studies based on live trapping, camera trapping, and radio-telemetry to monitor a raccoon population on the Outer Banks of North Carolina for two years, calculating abundance, density, home range size and overlap, activity patterns, movement and survival. We removed 149 individuals (~50% estimated population size) from the population over one year and gathered data on body condition, litter size, breeding season, sexual maturity, diet, parasites and age structure. Abundance and density were lower than other island populations, but home ranges were relatively large. Seasonal variations in movement and home range size indicate this population may be limited during the winter. Litter size and breeding season were comparable to other island and mainland populations although the age at first breeding was later than that reported in other studies. Diet analysis indicated that most animals foraged in marsh habitats. The population age structure was characterized by older

animals (> 2 year of age), typical of unharvested raccoon populations exposed to few mortality sources.

INTRODUCTON

The raccoon (*Procyon lotor*) has a broad distribution that spans North and Central America, due in large part to their ability to adapt to a variety of resources and environmental conditions. Once historically important as a furbearer, focus on this omnivore has more recently shifted to their interactions with humans, either as a nuisance or potential zoonotic disease vector, or as a significant predator of rare and endangered wildlife. Raccoons now inhabit a number of unique environments, such as coastal islands, where their role as predators (Hartmann et al. 1997, Ellis et al. 2007) can be amplified. Raccoons are considered a significant predator of beach-nesting shorebirds and sea turtles throughout the Atlantic coastal zone. High raccoon densities have been correlated with low productivity of American Oystercatchers (Davis et al. 2001), sea turtles (Engeman et al. 2006, Ratnaswamy and Warren 1998, Stancyk et al. 1980), black skimmers and least terns (Erwin et al. 2001). Populations along the Atlantic seaboard can reach high densities due to the absence of large predators and recreational activities that provide shelter and food (Barton and Roth 2007, Ratnaswamy et al. 1997, Smith and Engeman 2002) but even at low densities, raccoons

can be devastating predators of ground nesting species (Ellis et al. 2007).

Although mainland raccoons have been well studied, particularly throughout the eastern United States (Gehrt 2003), published information on Atlantic coastal and island populations is limited to a few locations: St. Catherines Island, GA (Hudson 1978, Harman and Stains 1979, Lotze 1979, Anderson 1981), South Carolina (Hable et al. 1992), Virginia (Hanlon et al. 1989) and Florida (Bigler et al. 1977, Bigler et al. 1981, Barton and Roth 2007). Island populations often include protected areas such as national parks and wildlife refuges in which raccoon populations are not subject to mortality from hunting or trapping. Under these circumstances, mammalian predator populations can differ strikingly in demographics (Weston and Brisbin 2003) and survival (Gehrt and Fritzell 1999) from populations that are subject to harvest or other forms of management.

Recent efforts to model populations of raccoons (Martin et al. 2010) and other meso-predators (Roth et al. 2008) have shown that site-specific information on parameters such as survivorship, age structure, and sex ratios, diet and movement are often necessary to model predator populations that can be used to protect local populations of rare and endangered prey species. In this paper we provide information on the biology and demographics of a barrier island raccoon population to improve our knowledge of predator-prey dynamics in Atlantic coastal environments.

MATERIALS AND METHODS

Study Area.— The Outer Banks of North Carolina are comprised of approximately 320 km of barrier islands extending from Currituck to Cape Fear. Cape Lookout National Seashore (CLNS) is comprised of four islands located at the southern end of the Outer Banks adjacent to Beaufort, North Carolina. South Core Banks is the southernmost island within the Cape Lookout National Seashore boundary. It is 35 km long and varies in width from 0.26 to 2.35 km, averaging approximately 0.5 km (Fig. 1.1). South Core Banks is oriented northeast-southwest and exhibits a low profile landscape of wide, bare beaches with low dunes covered by scattered grasses, flat grasslands bordered by dense vegetation, and large expanses of salt marsh along Core Sound that separates the island from the mainland (Godfrey and Godfrey 1976).

Abundance and density estimates.— We used box traps to capture and mark a total of 131 raccoons over three sampling periods between May 2007 and June 2008. We transported unmarked animals to a field station where they were anaesthetized with an IM injection of 10mg/kg Telazol[®] (100mg/ml) (Elkin-Sinn, Incorporated, Cherry Hill, NJ). We weighed and measured all animals, extracted a premolar tooth for aging and fitted most animals with a one-inch wide polyurethane coated nylon dog collar with individually numbered livestock ear tags (National Band and Tag Company, Newport, KY) attached. We fitted a subset of 60 animals with radio-collars (ATS[®] Inc., Asanti, MN) with collar mounted numbered ear tags from May 2007 to June 2008. The radio-

transmitters weighed 58g and were equipped with mortality sensors (ATS Inc., Asanti, MN). Following recovery from anesthesia, all animals were released at the point of capture.

We used infrared trail monitors (TrailMaster™) with 35mm cameras using 400 ASA film (hereafter called camera traps) to recapture marked individuals for population estimation. Twenty cameras, spaced approximately 1.8 km apart, were set up in May 2007 along the entire length of the island. Each camera trap was armed from 21:00 to 5:00 hours for six periods ranging in length from 21 to 48 days between 9 May 2007 and 27 August 2008. We baited camera traps with a variety of food and a scent lure (Cronk's Outdoor Supplies, Wiscasset, ME). The ratio of marked to unmarked animals captured on the cameras was used to generate a Lincoln-Petersen population estimate for each sampling period (Williams et al. 2002 pp.290-296). We used methods different from the typical Lincoln-Petersen model. Typically when using the two-sample Lincoln-Petersen estimator, animals are captured, marked and released during the first sample and the number of marked animals from the first sample recaptured in the second sample is counted. In our study animals were live-trapped and marked outside of the camera sampling periods and unmarked animals captured during the camera sampling periods were not marked when captured. The number of marks in the population during each period was not derived from how many animals were captured and marked in the first sample but instead the number of marks in the population for each period was estimated

as the number of marked animals released from the live-trapping sessions corrected for period mortality derived from the Kaplan-Meier survival curve. We estimated density by calculating the area of the polygon created by the camera locations in ArcMap (Version 9.2, ESRI Redlands, CA) with an added boundary strip with width calculated from home range diameter. We calculated home range diameter using the average length of home ranges along the island length.

Home range size, overlap, activity patterns and movement.— We collected location and movement data from radio-collared animals using a portable receiver and a three-element hand-held Yagi antenna. We obtained 20 locations on each animal during each of four seasons (summer: mid-May to mid-August, fall: mid-August to mid-November, winter: mid-November to mid-February, spring: mid-February to mid-May) from May 2007 to August 2009. Locations were equally divided among four time periods: 00:00-6:00, 6:00-12:00, 12:00-18:00 and 18:00-00:00. Consecutive locations were taken a minimum of eight hours apart.

Due to the island's narrow width (<2.35 km at any point), we determined that triangulation was not necessary to delineate a home range, assuming that animals use the entire width of the island. We recorded telemetry locations by traversing an access road and recording locations with a geographic positioning system where the signal vector was approximately perpendicular to the nearshore beach. Locations for all animals were plotted in ArcMap and home ranges were drawn within the longitudinal bounds given by

the telemetry locations. The resulting polygon areas were calculated using Hawth's Tools in ArcMap and were compared across sexes and seasons. Home range overlap was analyzed graphically in ArcMap by comparing the percentage of an animal's home range area that overlapped with another animal. Home range overlap was compared between males, between males and females and between female raccoons. Activity distributions were represented by the movement of animals along the longitudinal axis of the island (linear home ranges). The mean distance between all points in each animal's activity distribution and the distribution midpoint was plotted to determine foray length and the associated standard deviation. Foray length represents how far telemetry locations are from the center of an individual's home range and thus provide an index of movement and space-use. We analyzed foray length and variation by season and by sex. The proportion of hourly locations in which animals were moving versus stationary was analyzed by sex and season to determine differences in activity patterns throughout the day. Beach foraging was of particular interest because it may be related to the presence of nesting sea turtles and birds. We therefore analyzed telemetry locations during the hours of the night when animals are foraging to determine the frequency with which animals were using the beach.

Removal trapping.— As part of a study to evaluate raccoon removal as a management tool for nesting sea turtles and birds we used leg-hold (EGG, Lil' Grizz and Duffer) and box (Tomahawk and Havahart) traps to capture and humanely euthanized 149 raccoons

from South Core Banks between August 2008 and April 2009. Animals were euthanized using an IV injection of Beuthanasia solution (pentobarbital sodium and phenytoin sodium) (Schering-Plough Animal Health Corp., Omaha, NE) following complete anesthesia with 100 mg/ml Telazol, as approved by the American Veterinary Medical Association (AVMA 2007) protocol for animal euthanasia and by the Institutional Animal Care and Use Committee at North Carolina State University (NCSU) (Protocol# 07- 0120-0). Euthanized animals were frozen and then transported to the NCSU College of Veterinary Medicine where they were necropsied in January and April 2009.

Age Distribution.— We extracted the first premolar tooth of each animal, live-trapped and euthanized. We sent the extracted teeth to be analyzed at a lab (Matson's, Milltown, MT) for cementum analysis to estimate animal age (Johnson 1970).

General assessment.— We took weights and measurements from each live-trapped and euthanized animal to calculate body condition based on the index developed by Prange et al. (2003): $CI = (\text{body mass [kg]} / \text{body length [cm]}) \times 100$. Each animal's sex was recorded to calculate body condition differences between the sexes as well as an overall sex ratio for the population.

Diet.— We removed GI tracts from each necropsied animal and analyzed stomach and colon contents within each season. We weighed and identified items to the most precise classification level possible. We measured the percent of the total stomach or colon content weight and the frequency of occurrence for each diet item.

Parasites and disease.-We removed and identified to species all ectoparasites from live-trapped and euthanized animals. We also identified any endoparasites present in the stomachs of euthanized animals. Radio-collared animals that died of natural causes in the field were recovered and taken to the NCSU College of Veterinary Medicine for necropsy to determine cause of death.

Sexual maturity, litter size and breeding season.— We removed reproductive tracts from each necropsied animal. We separated male reproductive tracts into baculum and testes. We cleaned, dried, weighed and measured the baculum and weighed the testes. We based male sexual maturity on baculum length and weight (Johnson 1970, Sanderson 1961). We classified any male with a baculum weight of (>2.5g) and length (>90mm) as sexually mature following the procedure of Kramer et al. (1999). We used the age data collected from tooth analysis to determine the age of sexual maturity for males. We used testes weight to estimate breeding season. We removed the uterus from all females and examined them for evidence of uterine scars. Females were considered sexually mature if the uterus contained fetuses or if uterine scars were present (Johnson 1970). These data were combined with age data from the tooth analysis to establish the age of sexual maturity in females. We assumed that the darkest set of uterine scars represented the most recently implanted placentii (Sanderson 1987) and used the number of fetuses present to calculate mean litter size. We estimated the age of fetuses following Llewellyn (1953) and used the calculated fetal age to estimate time of breeding.

Survival.— We calculated survival using a staggered entry Kaplan-Meier design (Pollock et al. 1989) with data from radio-collared animals of known fates. Animals that died within two weeks of collaring (n=4) were eliminated from the analysis. When the exact date of death was unknown, the midpoint between the last known date alive and the date of recovery was used. We performed an analysis of all animals together as well as separating males and females. Because of small numbers of yearlings and juveniles captured, we pooled data for all age classes.

RESULTS

Abundance and density estimates.— Abundance estimates ranged from 137 to 287 animals between the six sampling periods with an average estimate of 199.87 (SE=22.88) individual animals (Table 1.1). We used the average of all population estimates to calculate a density of 4.84 animals/ km².

Home range size, overlap and movement.— We constructed 18 home ranges for summer 2007, 36 for fall 2007, 28 for winter 2008, 24 for spring 2008 and 25 for summer 2008 in ArcMap. The home ranges of many animals overlapped considerably (Fig. 1.2). Average home range overlap over all seasons was 32% (n=244, SE=0.04). Male to male home range overlap was 25% (n=98, SE=0.02), male to female overlap was 34% (n=118, SE=0.06) and female to female overlap was 50% (n=28, SE=0.18). Male-male home

range overlap was significantly different than female-female overlap ($t=-2.054$, $df=241$, $p=0.041$, $\alpha=0.05$). Average home range size over all seasons was 139.9 ha ($n=148$, $SE=12.3$ ha). A one-way ANOVA for all seasons was significant ($F=3.69$, $p<0.001$, $\alpha=0.05$). We compared means using a t-test and found that average home range area for the fall 2007 period ($n=36$ animals, $mean=215.61$ ha, $SE=40.80$ ha) was significantly larger than the average home range area in any other season (Table 1.2). Average home range area for males during the summer 2007 period was significantly larger than the average home range area for females during the same period ($t=4.0665$, $df=30.46$, $p<0.001$). Males also had larger home ranges than females in the spring 2008 period ($t=2.706$, $df=21.83$, $p=0.013$) (Table 1.2). Home ranges in all other seasons were similar between the sexes.

Foray length was not significantly different among seasons for all animals combined. However, males showed longer average foray length than females in summer 2007 (males $n=8$, $mean=0.398$ km, $SE=0.886$ km, females $n=10$, $mean=0.141$ km, $SE=0.023$ km, $t=2.815$, $df=7.93$, $p=0.023$), summer 2008 (males $n=16$, $mean=0.438$, $SE=0.047$, females $n=9$, $mean=0.172$, $SE=0.043$, $t=4.182$, $df=21.78$, $p<0.001$) and spring 2008 (males $n=14$, $mean=0.510$ km, $SE=0.116$ km, females $n=10$, $mean=0.196$ km, $SE=0.031$ km, $t=2.622$, $df=14.76$, $p=0.019$). No significant differences in male and female foray length were found at other times of year (Fig. 1.3).

Animals were most active between 18:00 and 6:00 in all seasons (Fig. 1.4).

Females showed a higher proportion of active locations during the daylight hours than males.

Use of beach habitats was highest in the summer and lowest in the winter, with intermediate levels of activity in the fall and spring months. Males (n=801, locations on the beach=4.5%) foraged more often on the beach than females (n=499, locations on the beach=1.6%) when data from all seasons were combined ($\chi^2=5.09$, $p=0.024$, $\alpha=0.05$). No significant differences were found between males and females in individual seasons.

Age Distribution.— The age distribution was generated from 229 animals caught between May 2007 and April 2009 showed that most animals were two (n=38), three (n=52) or four (n=40) years old (Fig. 1.5).

General assessment.— Average weight for males was 4.05 kg (n=152, SE=0.088 kg) over the study period which was significantly higher than the average weight for females (n=66, mean=3.18 kg, SE=0.079 kg) ($t=7.331$, $df=196$, $p<0.001$). Animals examined in the summer and spring seasons were often emaciated and dehydrated and animals examined in the fall and winter appeared in better overall condition. Our condition index data show the highest values in the October 2007 and December 2008 and the lowest values in May and June of 2007 and 2008 and April 2009 (Fig. 1.6). Trends were similar between males and females. We captured nearly twice as many males (n=152) as females (n=66).

Parasites and Disease. — We examined the contents of 128 stomachs for

endoparasites and found roundworms of the genus *Physaloptera* in 23 stomachs, we found no other evidence of endoparasites. We also removed ticks (*Dermacentor variabilis*) from the skin of 41 (n=131) animals. Two radio-collared animals died in the field of mysterious causes. Based on gross necropsy lesions, these animals were determined to have likely died of a parvovirus infection.

Diet.— Twenty-five percent (n=32) of the 128 stomachs examined were empty. Of the remaining 96 animals with stomach contents 48.21% of the total weight of stomach contents were items that are not digestible. These items are reported in Table 1.3 under the column for Leaves/Fiber and Other. The category Other represents wood, endoparasites, rocks, sand, cotton and seashells. Seasonal patterns of all food items identified from 96 animals are represented in Tables 1.3 and 1.4. The relative importance of any food item is represented by the percentage that food item makes up of the total weight of all food items consumed over all stomachs or colons and by the frequency of occurrence over all stomachs or colons.

The most important food sources over all seasons based on percent total weight were (in order of greatest to least importance) crustaceans (crabs), fruit, fish and birds. The most important food sources over all seasons based on percent frequency were crustaceans (crabs), fruit, insects, and birds in order of importance. Seasonal changes in food habits were apparent. Fruits were most common in August stomachs (n=3) and colons (n=1), fruits and fish were most common in December stomachs (n=35) and

colons (24) and crustaceans were by far most common in April stomachs (n=58) and colons (n=55). The most common crustaceans by weight and frequency were fiddler crab (*Uca pugilator*) and molecrabs (*Emerita talpoida*) followed by blue crab (*Callinectes sapidus*). The most common fruit was yaupon (*Ilex vomitoria*) followed by eastern redcedar (*Juniperus virginiana*). Although birds did not represent a high percentage of the overall weight of food items consumed, bird feathers did occur in a high frequency of overall stomachs (8.3%, n=96) especially in December (16.7%, n=35).

Sexual maturity, litter size and breeding season.— Eighty-one percent of males and 73% of females captured were sexually mature. Males under one year old were not sexually mature based on baculum size. Forty-two percent (n=12) of one-year-olds, 85.7% (n=13) of two-year-olds and 89.5% of three-year-olds (n=19) were sexually mature. All males four-years-old and above (n=38) were sexually mature. Females under two-years-old were not sexually mature (n=6). Ninety percent (n=9) of two-year-old females, 77.8% (n=9) of three-year-old females, 90% (n=10) of four-year-old females, 100% (n=2) five-year-old females, 100% (n=2) seven-year-old females, 75% (n=4) eight-year-old females and 100% (n=2) nine-year-old females had bred in the last two seasons based on placental scarring. Average litter size calculated by uterine scars was 2.56 (n=25, SE=0.19) and average litter size calculated by the number of fetuses present was 2.67 (n=9, SE=0.17). Average fetus weight was 11.1g (n=26, SE=3.1) and average fetus length was 60.9mm (n=26, SE=8.2). Fetus age based on crown to rump

length ranged from 12 days to 66 days (Llewellyn 1953). Using fetus age, the breeding season for this population was estimated to begin in mid-February with births increasing through March and April (animals were not collected between May and January). Testes weights averaged 15.27g (n=40, SE=0.39g) in April, 4.46g (n=4, SE=0.46g) in August and 13.77g (n=35, SE=0.57g) in December.

Survival.— Overall annual survival was estimated at 0.83 (SE=0.03) when males and females were analyzed together. When males and females were analyzed separately, male annual survival was 0.87 (SE=0.04) and female annual survival was 0.85 (SE=0.09). Survival for all animals was 0.95 in the fall 2007 period and stayed steady through the winter 2007-2008 period but dropped to 0.91 during the spring 2008 period and dropped again to 0.83 in the summer 2008 period.

DISCUSSION

The raccoon density we found on South Core Banks (4.84 animals/km²) was comparable to some islands (Big Key, Florida (2.2-6.5 animals/km²) (Bigler et al. 1981) but low compared to other published densities for island populations (Wolf Island, Ontario (8.4 animals/km²) (Rosatte et al. 2007), Ten Thousand Islands, Florida (7.7 animals/km²) (Hoff et al. 1977). Density can be affected by many factors including resource availability and distribution as well as human activity. On South Core Banks our highest

trapping success was in areas where human activity was the highest. Indeed raccoons in those areas also exhibited a high degree of home range overlap (Fig. 1.2). Overall raccoon density on SCB was not abnormally high when compared to similar sites suggesting that although resources may be abundant in certain areas, they are not evenly distributed across the landscape in high abundance. In addition, factors such as limited shelter, low diversity of vegetation and the generally harsh coastal environment could keep limit this population. In fact, the average home range on South Core Banks (135.5 ha) was large compared to other island populations (34.0 ha Parramore Island, Virginia (Hanlon et al. 1989), 84.0 ha (Hable et al. 1992), 38.0 ha Saint Catherine's Island, Georgia (Lotze 1979)) which suggests that animals need to range widely to secure sufficient resources. Home ranges were particularly large in the fall indicating that winter is likely the most resource limited season (Rabinowitz 1981). Winters on the barrier islands are typically cold and stormy, supplementary food resources (fishing by catch, nesting birds and turtles, and garbage associated with recreational visitors) are lacking, and the barrier island environment is barren with limited biodiversity (Oosting 1954). Our condition index data indicate that animals are in their best condition in the early winter and show drastically lower condition index values in spring. Similar trends have been found at other sites in Minnesota (Mech et al. 1968) and Tennessee (Moore and Kennedy 1985), and by Pitt et al. (2008a) who found that raccoons in Manitoba had

higher levels of body fat in the fall than in the spring and that the level of body fat in the spring was inversely related to winter severity.

Inadequate den sites and limited cover may also contribute to poor spring condition. Raccoons on the Outer Banks frequently den beneath the thick mats of black needlerush (*Juncus roemerianus Scheele*) that accumulate along marsh edges. These den sites are not typical of raccoon populations, although Mech et al. (1966) found that raccoons in a bog habitat in Minnesota did use ground beds in cattail marshes, alder swamps, or cedar swamps. Most studies report a preference for denning in tree cavities (Berner and Gysel 1967, Dorney 1954, Henner et al. 2004) rock outcrops (Berner and Gysel 1967) or ground burrows dug by other species (Stains 1956). We observed two animals denning beneath uprooted trees and another that denned in the attic of an abandoned building. Raccoons likely choose den sites based on what is available. On South Core Banks large trees are rare and concentrated on one end of the island only. Abandoned buildings likely represent the most adequate den sources but are also highly concentrated in small, widely scattered areas.

Females had significantly smaller home ranges and exhibited shorter foray lengths in spring and summer suggesting that they restrict their movements during these months, likely because of parturition and cub rearing (Anderson and Willis 1982). In general we found females moved more than males except in these months. We found that females overlapped home ranges with each other significantly more than did males and that male-

female spatial overlap was minimal. Females will generally distribute themselves relative to resources and a high degree of female overlap further supports the idea of spatially clumped resources on South Core Banks. Our results are contrary to those of Gehrt and Fritzell (1998) who found that males and females in Texas were often aggregated, and that male and female home range overlap was extensive. Furthermore, they found that females rarely consorted with other adults. They noted that female-female overlap was variable but increased especially where water was scarce. We saw evidence of dehydration in our population and fresh water is likely limiting this population as well. It is also possible that even where there is overlap in home ranges between females, females still avoid each other (Pitt et al. 2008b).

Several researchers have proposed that raccoon predation on beach nesting species is often caused by a few individuals (Barton and Roth 2007, Engeman et al. 2003, Ratnaswamy et al. 1997, Ratnaswamy and Warren 1998). We found that few individuals in our population used beach habitats and that most animals foraged in marsh habitats. Indeed the most common items in the diet (both the highest percent of total weight and frequency of occurrence) were fiddler crabs and yaupon berries both of which are restricted to marsh habitats (Harman and Stains 1979). Some of the insect species in the diet, particularly *Dyscinetus morator*, favor moist, rich soils typical of island marshes but not found on beaches or upland areas (D. Stephan, *personal communication*). However, one of the most common crab species, *Emerita talpoida* can be found on the beaches

indicating that foraging does take place there. Although we found that male raccoons spent more time on the beaches than females, females do use beach habitats especially in the summer when nesting turtles and birds are abundant suggesting that young animals may learn beach foraging from their mothers. Engeman et al. (2003) found evidence of this type of cultural transmission on Hobe Sound National Wildlife Refuge in Florida.

We found no evidence in the diet analysis that ghost crabs (*Ocypode quadrata*) were of dietary importance. Barton and Roth (2007) found two out of 23 raccoons collected from stomachs at Merritt Island and Archie Carr National Wildlife Refuges contained ghost crab remains. Based on this finding they suggested that raccoons might regulate ghost crab populations and proposed that raccoon removal could cause ghost crab populations to increase. We found ghost crabs in only one out of 96 stomachs examined and found little evidence that raccoons regulate ghost crab populations at Cape Lookout National Seashore (T. Wolcott, *personal communication*).

We found nematodes of the genus *Physaloptera* in 18% of stomachs sampled. No other endoparasites were recorded. Harkema and Miller (1964) reported that *Physaloptera* was common in raccoon stomachs in coastal North Carolina as were nematodes of the genus *Gnathostoma*. Other studies of raccoon endoparasites have found high loads of *Physaloptera* in Tennessee (Bafundo et al. 1980), Kentucky (Cole and Shoop 1987), Kansas (Robel et al. 1989), and Illinois (Snyder and Fitzgerald 1985).

Parasitism and disease has been reported as a key factor in the mortality of raccoons in Alabama (Johnson 1970) and other areas of the United States (Gehrt 2003).

We trapped nearly twice as many males than females in this study but we do not believe this ratio reflects the true sex ratio. Other studies have shown that male raccoons are more easily caught in live traps than females (Gehrt and Fritzell 1996) and this can produce an apparently biased sex-ratio where none exists. Evidence of animals in breeding condition in our population first appeared in mid-February and increased through April. Low testes weights for animals collected in August suggest a peak of breeding in April and May although our data were not sufficient to calculate the exact peak or end to the breeding season. McKeever (1958) found that the breeding season for raccoons in Georgia and Florida extended from mid-February to August and Bigler et al. (1981) found a breeding season extending from March through August in an estuarine mangrove swamp in Southern Florida, both of which agree with our findings. Northern populations may have earlier breeding seasons due to pronounced seasonal changes in weather (Johnson 1970, Stains 1956). Most males became sexually mature by two years of age, although some did not reach maturity until four years of age; much older than other studies in the southeastern United States (Dunn and Chapman 1983). Similarly, most females were sexually mature by two years of age although four out of 29 females in the older age classes showed no evidence of breeding in the two years prior to their death. Evidence that females may not breed every year is also consistent with other

studies (Gehrt 2003). However, Stuewer (1943) found up to 50% of females in Michigan and Ohio mated in their first year and Junge and Sanderson (1982) found that 73% of females mated before their first birthday. We found no evidence of such early breeding in our study. A delay in breeding could be due to resource scarcity. Zeveloff and Doer (1981) found a litter size of 2.9 (n=14) for raccoons in western North Carolina and Dunn and Chapman (1982) found an average litter size of 3.2 (n=10) for raccoons in Maryland, similar to our average litter size of 2.62 (n=34) on South Core Banks.

Studies of raccoon survival have focused primarily on harvested populations (Clark et al. 1989, Fritzell and Greenwood 1984, Hasbrouck et al. 1992). Hunting and trapping can result in lower survival rates for harvested populations. Hasbrouck et al. (1992) reported adult annual survival of 0.53 for a hunted population in Iowa and Mankin et al. (1999) found an annual survival rate of 0.74 in a hunted population in Illinois. Our estimated annual survival of 0.83 is similar to the 0.84 reported by Gehrt and Fritzell (1999) for an unharvested population of raccoons in South Texas and 0.80 reported by Urbanek et al. (2009) for an unharvested population in Illinois. Wilson (2002) reported survival of females in an unexploited raccoon population in Illinois as 0.84 in fall/winter but only 0.68 in spring/summer. The population on South Core Banks does include animals that are much older (>8 years, n=14) than those reported in other accounts of unharvested raccoon populations, suggesting that raccoon mortality rates on the Outer

Banks are unusually low. An older age structure can also be a product of scarce resources which are limiting recruitment (Clutton-Brock et al. 1987).

Detailed information on predators populations in protected areas like National Seashores can help guide management efforts for other species. This study has shown efforts to control raccoon populations may be most effective in the early spring when populations are lower and animals are in poor physical condition. Outer Banks raccoon populations may also be limited by the availability of suitable den sites. Limiting anthropogenic den sites, such as abandoned buildings, may also help limit populations that benefit from human activity. Although raccoon density on South Core Banks was not higher than densities at comparable sites (Bigler et al. 1981), average home range size for this population was large, suggesting that food sources are scarce and widely distributed over the island. Indeed 25% of stomachs examined were empty and a further 48% of stomach contents were non-digestible. We found evidence that resources are not only scarce but clumped. We observed our highest trapping success in areas of high human activity suggesting that resources are concentrated in those areas. Thus efforts to eliminate or further reduce anthropogenic food may prevent the raccoon population from becoming artificially high. We hypothesize that the scarcity of food and harsh winters may explain delayed breeding in the barrier island population. If so, management which reduces the size of the population may also reduce the age at first breeding. Recent

models of optimal predator control (Martin et al. 2010) may prove useful in assessing different predator management strategies.

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TABLES AND FIGURES

Table 1.1: Population estimates of raccoon on South Core Banks, NC based on camera trapping from 5/18/07 to 8/25/08

Sampling Period	Marked Caught	Unmarked Caught	Total Caught	Total Population	Marked in Lincoln- Petersen Estimate	SE
5/18-6/30/07	85	112	197	57	136.80	11.61
7/1-8/15/07	150	105	255	94	159.80	8.34
8/16-10/1/07	141	104	245	96	166.81	9.12
10/2-11/17/07	74	91	165	96	214.05	18.36
11/18-2/17/08	26	41	67	96	234.50	35.30
6/21-8/25/08	38	69	107	102	287.21	36.93
				Mean	199.86	22.88

Table 1.2: Home range area of raccoons on South Core Banks, NC by sex and season

Season and Sex	N	Home range area (ha)	SE (ha)
Summer 2007	33	109.94	9.48
Male	20	134.33	11.27
Female	13	72.42	10.24
Fall 2007	36	215.62	40.80
Male	19	249.27	68.67
Female	17	178.00	40.21
Winter 2007-08	28	143.55	29.56
Male	17	167.22	46.97
Female	11	106.97	18.12
Spring 2008	24	112.21	11.14
Male	14	134.03	14.32
Female	10	81.67	13.01
Summer 2008	27	96.14	8.40
Male	18	103.54	10.11
Female	9	81.34	14.60

Table 1.3: Principal food items by weight and frequency from stomach content analysis of raccoons on South Core Banks, NC during three seasons.

Month	August		December		April		All Seasons		
#Animals	3		35		58		96		
	%	%	%	%	%	%			%
Foods	Weight	Freq.	Weight	Freq.	Weight	Freq.	%	Weight	Freq.
<i>Vitis rotundifolia</i>	34.1	30.0	-	-	-	-	0.2		1
<i>Ilex vomitoria</i>	-	-	20.2	26.7	-	-	8.6		8.3
<i>Juniperus virginiana</i>	-	-	1	6.7	-	-	0.4		2.1
<i>Smilax auriculata</i>	-	-	0.3	3.3	-	-	0.1		1
<i>Opuntia drummondi</i>	-	-	-	-	2.1	3.5	1.2		2.1
Unknown fruit	-	-	0.02	3.3	-	-	0.01		2.1
Leaves/Fiber	15.1	70.0	20.7	60	28	62.1	24.2		58.3
<i>Emerita talpoida</i>	-	-	-	-	7.1	5.2	4		3.1
<i>Callinectes sapidus</i>	-	-	-	-	4.6	5.2	2.6		3.1
<i>Uca pugilator</i>	-	-	0.4	3.3	1.3	5.2	0.9		4.2
<i>Ocypode quadrata</i>	-	-	-	-	0.6	1.7	0.3		1
<i>Limulus polyphemus</i>	-	-	-	-	0.06	1.7	0.03		1
<i>Panopeus herbstii</i>	-	-	2.5	3.3	-	-	1.1		1
Unknown crab	-	-	2.5	10	9.8	24.1	6.6		17.7

Table 1.3 Continued

Month	August		December		April		All Seasons	
# Animals	3		35		58		96	
	%		%	%	%	%	%	%
Foods	Weight	Freq.	Weight	Freq.	Weight	Freq.	Weight	Freq.
Unknown crustacean	30.8	30.0	-	-	0.4	5.2	0.5	4.2
Unknown mollusk	-	-	0.02	3.3	-	-	0.01	1
Insect: Diptera	20.0	30.0	-	-	-	-	0.01	1
<i>Lethocerus</i> sp.	-	-	-	-	1.8	1.7	1	1
<i>Dyscinetus morator</i>	-	-	-	-	0.7	3.4	0.4	2.1
<i>Chrysops</i> sp.	-	-	-	-	0.3	8.6	0.2	5.2
<i>Phyllophaga</i> sp.	-	-	0.2	6.7	0.04	1.7	0.1	3.1
<i>Solenopsis invicta</i>	-	-	-	-	0.01	1.7	0.01	1
<i>Edessa florida</i>	-	-	0.04	3.3	-	-	0.02	1
<i>Neoconocephalus</i>	-	-	-	-	0.01	1.7	0.01	1
<i>Anguilla rostrata</i>	-	-	-	-	4	5.2	2.3	3.1
Unknown fish	-	-	0.3	2.8	0.6	3.4	0.9	2.1
<i>Opsanus</i> sp.	-	-	-	-	2.9	1.7	1.6	1
<i>Symphurus plagiusa</i>	-	-	17.3	2.8	-	-	7	1
<i>Lumbricus</i> sp.	-	-	-	-	0.3	1.7	0.2	1
Unknown bird	-	-	2.5	16.7	3.4	5.2	3	8.3
Hair: <i>Procyon lotor</i>	-	-	0.02	10	10.7	13.8	6.2	11.5
Hair: <i>Mus musculus</i>	-	-	-	-	2.2	6.9	1.2	4.2

Table 1.3 Continued

Month	August		December		April		All Seasons	
# Animals	3		35		58		96	
	%		%	%	%	%	%	%
Foods	Weight	Freq.	Weight	Freq.	Weight	Freq.	Weight	Freq.
<i>Mus musculus</i>	-	-	2.5	2.9	-	-	1.1	1.0
Other	-	-	29.5	48.6	19.08	37.9	24	40.6

Table 1.4: Principal foods items by weight and frequency from colon content analysis of raccoons on South Core Banks, NC during three seasons.

Month	August		December		April		All season	
# Animals	1		24		55		80	
	%	%	%	%	%	%	%	%
Foods	Weight	Freq	Weight	% Freq	Weight	Freq	Weight	Freq
<i>Physalis maritime</i>	69.6	100.0	0.4	4.2	-	-	0.6	2.5
<i>Juniperus virginiana</i>	-	-	21.25	12.5	-	-	5.6	3.8
<i>Ilex vomitoria</i>	-	-	5	33.3	0.6	1.8	1.8	11.3
<i>Opuntia drummondii</i>	-	-	-	-	0.09	1.8	0.06	1.3
Unknown fruit	-	-	-	-	0.3	10.9	0.2	7.5
Leaves/Fiber	30.4	100.0	33.5	54.2	10.62	36.4	14.7	1.3
<i>Panopeus herbstii</i>	-	-	7.7	4.2	-	-	2.1	1.3
<i>Uca pugilator</i>	-	-	-	-	5	7.3	3.6	5
<i>Callinectes sapidus</i>	-	-	-	-	4.9	3.6	3.6	2.5
<i>Emerita talpoida</i>	-	-	-	-	4.1	3.6	2.9	2.5
<i>Cancer irroratus</i>	-	-	-	-	3.5	1.8	2.5	1.3
Unknown crab	-	-	1.1	4.2	60.9	47.3	42.9	33.8
Unknown shrimp	-	-	-	-	4.3	3.6	3.1	2.5
Unknown crustacean	-	-	-	-	1.7	3.6	1.3	2.5
Unknown mollusk	-	-	0.05	4.2	-	-	0.01	1.3

Table 1.4 Continued

Month	August		December		April		All Seasons	
#Animals	1		24		55		80	
	%	%	%	%	%	%	%	
Foods	Weight	Freq	Weight	Freq	Weight	Freq	Weight	% Freq
<i>Dyscinetus morator</i>	-	-	-	-	0.07	1.8	0.05	1.3
<i>Dermacentor variabilis</i>	-	-	-	-	0.02	1.8	0.01	1.3
<i>Dicaelus elongates</i>	-	-	-	-	0.25	0.9	0.2	0.65
<i>Galerita</i> sp.					0.25	0.9	0.2	0.65
Unknown fish	-	-	0.9	4.2	-	-	0.3	1.3
Unknown bird	-	-	1.7	12.5	-	-	0.5	3.8
Hair: <i>Procyon lotor</i>	-	-	0.3	8.3	0.9	3.6	0.7	5
Hair: <i>Mus musculus</i>	-	-	-	-	0.1	1.8	0.1	1.3
Other	-	-	28.1	37.5	2.4	16.4	12.97	22.5

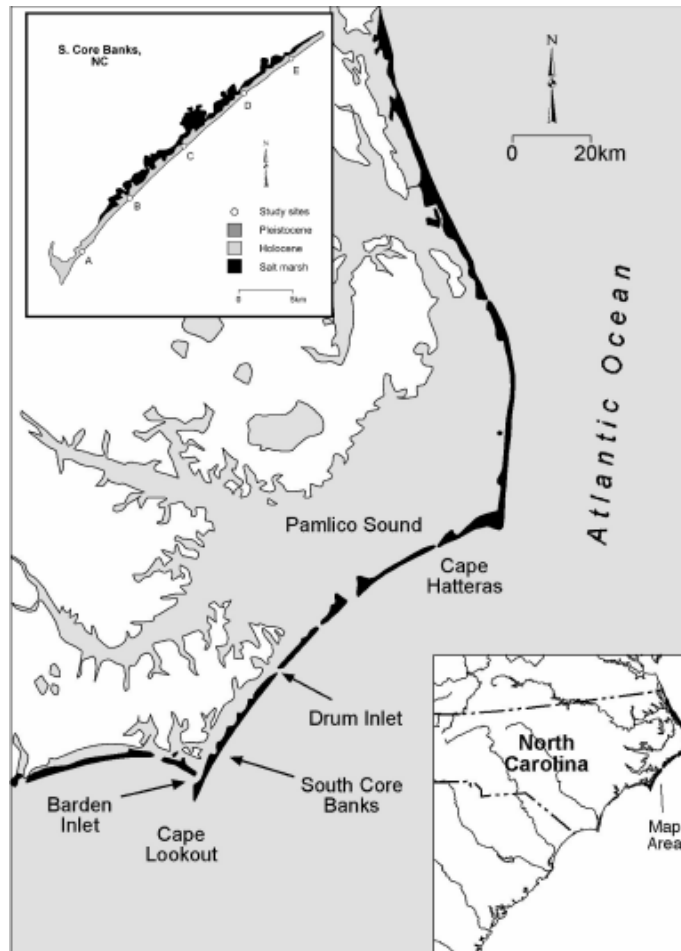


Figure 1.1: South Core Banks, Cape Lookout National Seashore, NC.

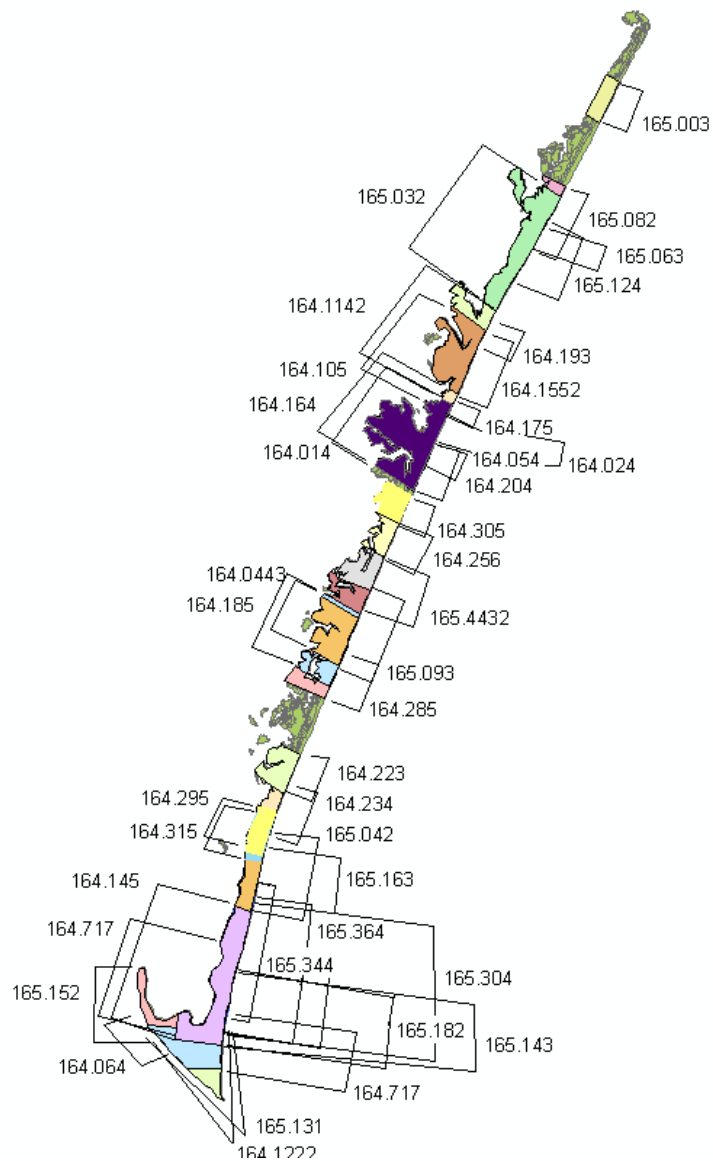


Figure 1.2: Home range overlap of raccoons from radio-telemetry on South Core Banks, NC from May 2007 to August 2008 (n=244, mean=0.45ha, SE=0.02ha)

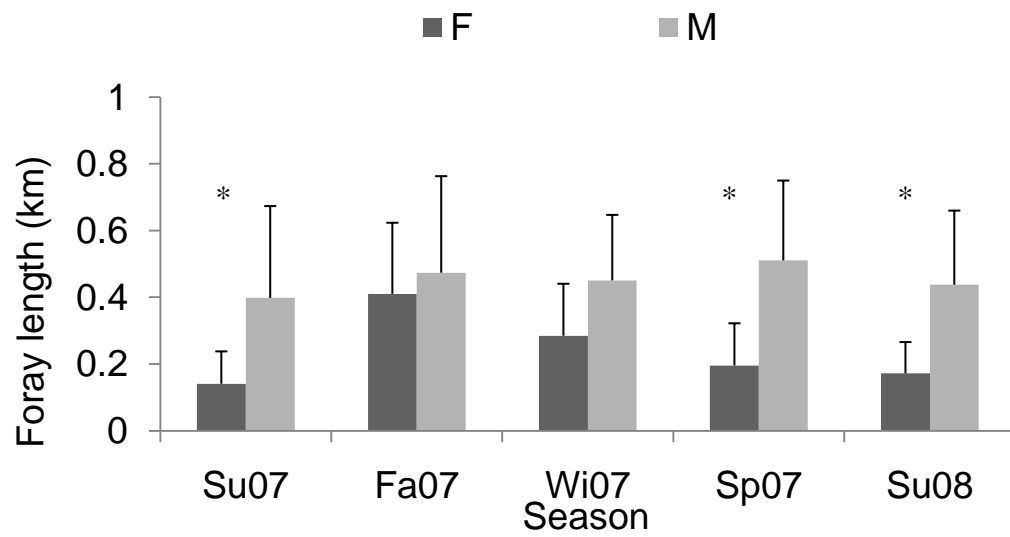


Figure 1.3: Foray length of raccoons on South Core Banks, NC by sex and season 2007 and 2008. Error bars are plus and minus one standard deviation of the mean. The (*) indicates significance at $p < 0.05$.

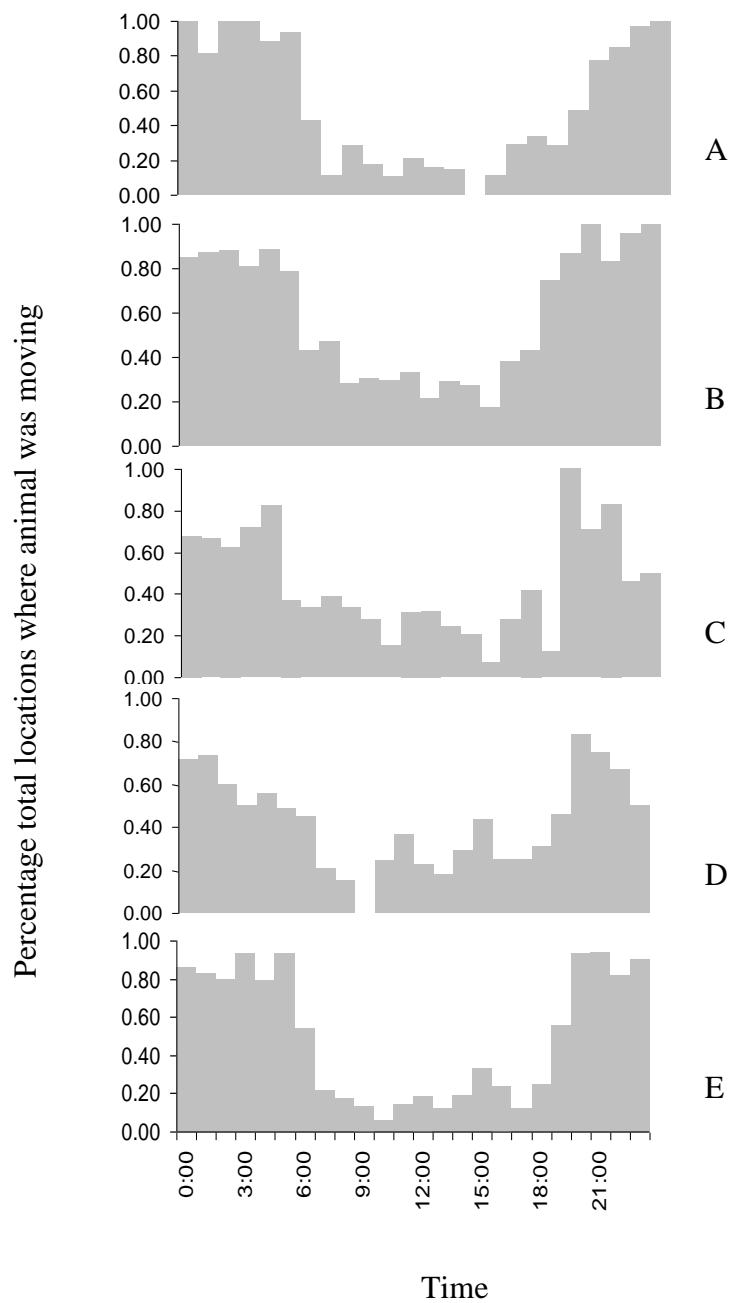


Figure 1.4: Percentage of total locations per hour where animal was moving by season. (A) Summer 2007 (n=597) (B) Fall 2007 (n=795) (C) Winter 2007 (n=540) (D) Spring 2008 (n=526) (E) Summer 2008 (n=575).

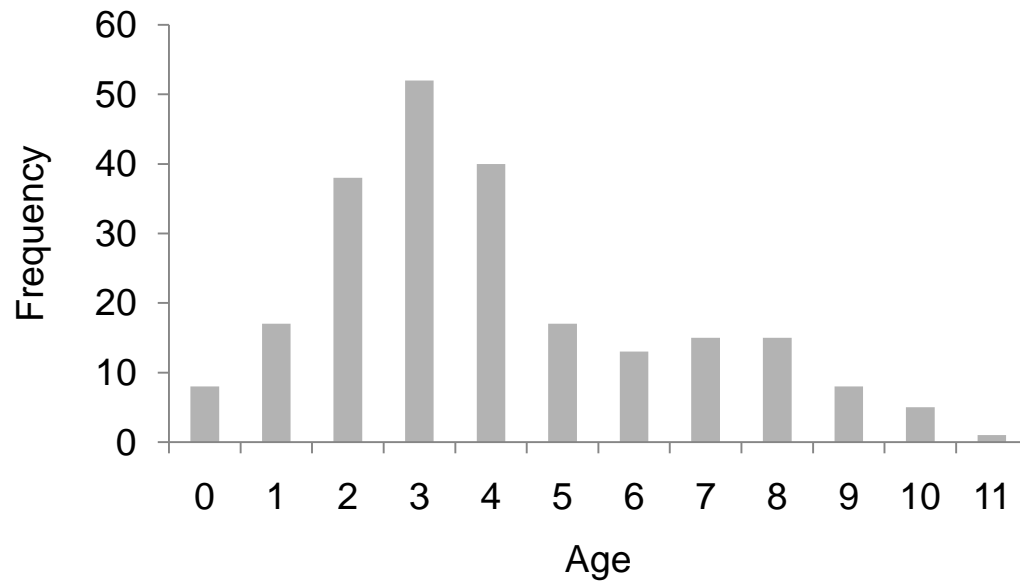


Figure 1.5: Age distribution for raccoons captured on South Core Banks, NC, May to October 2007 (n=70), May 2008 to April 2009 (n=159).

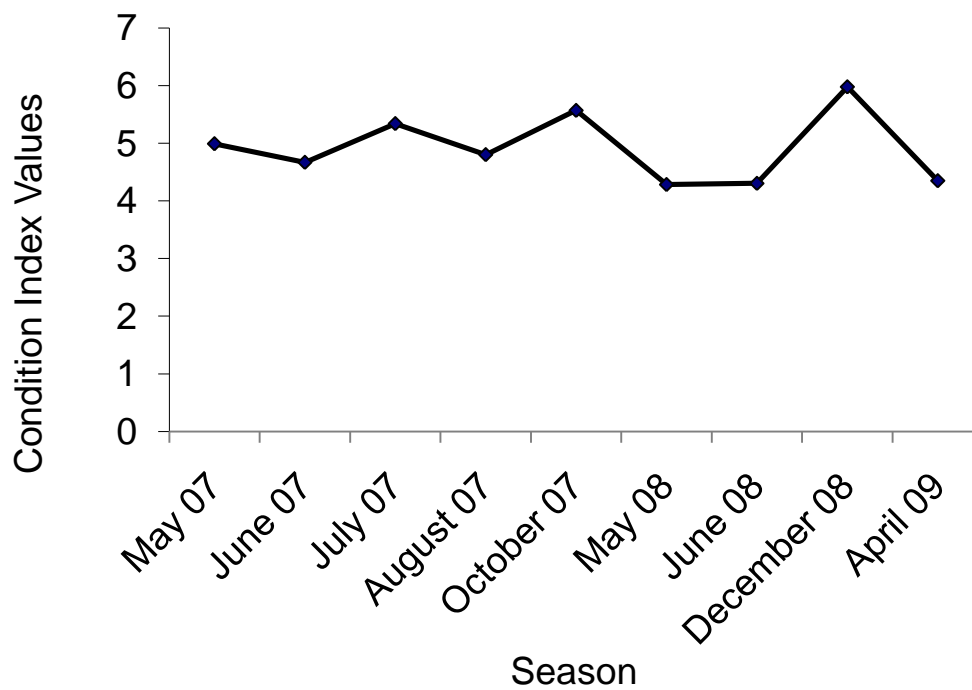


Figure 1.6: Body condition index values for raccoons captured on South Core Banks, NC by season (May 07 n=38, June 07 n=19, July 07 n=37, August 07 n=9, October 07 n=6, May 08 n=13, June 08 n=6, December 08 n=82, April 09 n=67).

CHAPTER 2

Testing the Assumptions of Camera Trapping for Estimating Abundance and Density

ABSTRACT

Camera traps remotely sample animal populations and can be used to make inferences about a variety of population attributes if the estimation techniques use accurate local information or adjust for sampling biases. We tested some common camera trap assumptions by using local information gathered from radio-telemetry. We used camera traps on a 2859 hectare barrier island off the coast of North Carolina to re-sight raccoons (*Procyon lotor*) and estimate population size, survival, and home range. We then compared these estimates to information collected from radio-collared animals on the same island. We captured 131 animals and marked them with numeric tags attached to a neck collar. We established a transect of camera traps along the length of the island and collected photographic recaptures (~5000 trap nights) between May 2007 and August 2008. We applied Lincoln-Petersen and Cormack-Jolly-Seber models to the resight data to estimate raccoon survival, abundance and density. Information from a subset of 37 radio-collared animals was used to assess capture heterogeneity and the accuracy of our parameter estimates. We calculated an average abundance of 199.87 animals (SE=22.88)

across several sampling periods. Although we assumed equal probability of capture in our estimates, we found that only 41% of animals known to be alive during the study were re-sighted by our camera-traps. Our abundance estimates for each period of the study increased over time, suggesting an increasing trend in detection. Raccoon density estimates based on radio telemetry data were nearly two times higher than estimates derived from the camera trap data and the Mean Maximum Distance Moved (MMDM) method. Factors such as camera placement, variations in the home range of individual animals, habitat, and trap response, especially when cameras were baited, contributed to bias in our estimates. We also tested a natural marking system with applied numeric tags in a double-tagging experiment to explore accuracy and sources of error associated with natural marks. We found that accuracy of identification depended on photograph quality, individual distinctiveness and orientation to the camera. We found that observers agreed on an incorrect identification 25% of the time, indicating that non-robust forms of natural marking system validation where double-tagging is not used could be fraught with observer error.

INTRODUCTION

Camera traps are becoming a popular method for sampling animal populations (Dillon and Kelly 2008, Goswami et al. 2007, Karanth 1995, Mace et al. 1994, Noss et al. 2003).

While live-trapping often requires significant labor and funds, can modify the behavior of captured animals, and sometimes results in animal mortality during handling (Moore and Kennedy 1985), camera traps can minimize or eliminate these issues. Camera traps can also be used within a mark-recapture framework to assess abundance, density, survival, recruitment and other aspects of animal ecology (Karanth and Nichols 1998). Properly deployed, camera traps are non-invasive and therefore preferable to other mark-recapture methods for sampling rare, cryptic, or elusive species, especially those that demonstrate a strong trap response (Carbone et al. 2001, Kays and Slauson 2008).

Despite the advantages of camera traps, mark-recapture using these devices is subject to biases due to issues of detectability similar to those encountered in live trapping studies. Detection may depend on the sex (Gehrt and Fritzell 1995), age (Garshelis and Noyce 2006), habitat preferences (Kelly and Holub 2008) and behavioral responses (Moore and Kennedy 1985, Wegge et al. 2004) of the target animals, as well as season (Gehrt and Fritzell 1995) and camera density (Karanth 1995, Dillon 2005). Unfortunately, little work has been done to evaluate these potential sources of bias for camera trapping studies. Many studies using camera traps continue to ignore detection, although a few have conducted radio-telemetry surveys simultaneously with camera sampling, allowing them to gather meaningful data on sources of bias (Dillon 2005, Dillon and Kelly 2008, Noss et al. 2003, Soisalo and Cavalcanti 2006).

Estimating density of an animal population when using camera traps has become an issue of special concern for scientists and managers. Information on density is the logical extension of abundance and requires accurate spatial information about how the target animals use their landscape. The challenge is that the effective trapping area of the camera trap array and the home range of the animal are unknown and must be estimated. Density in camera trapping is traditionally defined by adding a boundary strip to the sampling area in an effort to compensate for home range extension around that area (Dillon and Kelly 2008, Karanth 1995, Karanth and Nichols 1998, Soisalo and Cavalcanti 2006). In camera trapping, the width of the boundary strip has historically been measured using Mean Maximum Distance Moved (MMDM) between camera stations as a proxy for the maximum diameter of an animal's home range (Dillon and Kelly 2008, Karanth 1995, Karanth and Nichols 1998, Karanth and Nichols 2002, Lynman et al. 2009, Maffei et al. 2005, Parmenter et al. 2003). Unfortunately, most camera studies lack any local information on home range size (Dillon 2005). Recent papers have argued that although MMDM has performed well in simulations, it is theoretically unsubstantiated (Royle and Gardner 2009). MMDM has been shown to vary widely among surveys on a single population and on small study sites where animals range widely (Maffei et al. 2009) and can thus give inconsistent results across different studies. In this study we both explore sources of bias in abundance estimation using camera traps and use radio-telemetry data to test the accuracy of the MMDM method for calculating density.

Most current studies using camera-traps identify individuals using natural markings (Karanth 1995, Karanth and Nichols 1998, Kelly 2001, Azlan and Sharma 2003, Jackson et al. 2006, Goswami et al. 2007, Kelly et al. 2008, Sarmiento et al. 2009). The use of natural markings is advantageous because it eliminates the need to handle animals, thus reducing trap response however it is rarely evaluated for sources of error. We developed a system for identifying raccoons based on their natural markings and tested its accuracy using a double-tagging system. We looked at the accuracy of the system as well as specific variables that influenced misidentification.

Our species of interest is the raccoon (*Procyon lotor*). Raccoons were ideal for this study because they are abundant at the study site and relatively easy to live-trap and mark. They respond well to a variety of baits which can increase recapture rates at camera traps (Yasuda 2004). Raccoons can occur at high density where natural predators are uncommon and hunting pressure is low and where densities are high, camera capture rates are also likely to be high. Studies have shown that raccoon populations do exist at high densities along the southeastern coast of the United States (Ratnaswamy et al. 1997, Smith and Engeman 2002, Barton and Roth 2007) especially in the National Parks where they remain unharvested and unmanaged.

MATERIALS AND METHODS

Study site.— The Outer Banks is a ~320 km stretch of barrier islands off the coast of North Carolina extending from Currituck to Cape Fear. Cape Lookout National Seashore is a chain of four barrier islands at the southern end of the Outer Banks off the coast of Beaufort, North Carolina. South Core Banks is the southernmost island of Cape Lookout National Seashore, it is 35 km in length and its width varies from 0.26 km to 2.35 km, averaging approximately 0.5 km wide. South Core Banks has a northeast-southwest orientation, and exhibits a low profile landscape of wide, bare beaches with low dunes covered by scattered grasses, flat grasslands bordered by dense vegetation, and large expanses of salt marsh along Core Sound which separates the island from the mainland.

Live trapping.— We captured and marked a total of 131 raccoons over three sampling periods between May 2007 and June 2008. Animals were anaesthetized using Telazol[®] (100mg/ml) (Elkin-Sinn, Incorporated, Cherry Hill, NJ) at a dosage of 10mg/kg. All captured animals were fitted with either dog collars made of 2.5cm wide polyurethane coated nylon or radio collars equipped with mortality sensors (n=37) (Advanced Telemetry Systems Inc. (ATS), Asanti, MN). All collars were equipped with individually numbered livestock ear tags (National Band and Tag Company, Newport, KY). All handling procedures for live animals were approved by the Institutional Animal Care and Use Committee of North Carolina State University (Protocol# 07- 0120-0).

Camera trapping.— We used infrared trail monitors (TrailMaster[™]) with 35mm

cameras (400 ASA) (hereafter called camera traps) to recapture marked individuals for population estimation. Twenty cameras, spaced approximately 1.8 km apart, were set up in May 2007 along the entire length of the island. Each camera trap was armed from 21:00 to 5:00 hours for six periods ranging in length from 21 to 48 days between 9 May 2007 and 27 August 2008. We baited camera traps with a variety of food and a scent lure (Cronk's Outdoor Supplies, Wiscasset, ME). Although trap baiting can influence animal behavior, bait was required to increase capture rates (Thorn et al. 2009, Yasuda 2004) because we placed cameras uniformly along the length of the island and not along animal trails. We collected the film from cameras at the end of each sampling period and photographs were analyzed to determine how many marked and unmarked animals were captured. The ratio of marked to unmarked animals was then used to generate a Lincoln-Petersen population estimate for each sampling period (Williams et al. 2002 pp.290-296). We used methods different from the typical Lincoln-Petersen model. Typically when using the two-sample Lincoln-Petersen estimator, animals are captured, marked and released during the first sample and the number of marked animals from the first sample recaptured in the second sample is counted. In our study animals were live-trapped and marked outside of the camera sampling periods and unmarked animals captured during the camera sampling periods were not marked when captured. The number of marks in the population during each period was not derived from how many animals were captured and marked in the first sample but instead the number of marks in the population for each

period was estimated as the number of marked animals released from the live-trapping sessions corrected for period mortality derived from the Kaplan-Meier survival curve. We estimated density by calculating the area of the polygon created by the camera locations in ArcMap (Version 9.2, ESRI Redlands, CA) and then either added a buffer width calculated from MMDM or home range diameter. We calculated MMDM based on the mean of the maximum distances moved by each animal between two cameras, excluding animals only captured on one camera (Dillon 2005). We calculated home range diameter using the average length of home ranges along the island length. These boundary strips were each added separately to the camera polygon layer and the resulting polygon areas were measured in ArcMap. Densities were then calculated using the average population estimate over all periods divided by the area of the polygon framed by camera locations plus the boundary strip.

Radio telemetry.— We collected location and movement data from radio-collared animals using a portable receiver and three-element hand-held Yagi antenna. We obtained 100 locations on each animal during the study period. Locations were equally divided among four time intervals: 00:00-6:00, 6:00-12:00, 12:00-18:00 and 18:00-00:00. Consecutive locations were taken a minimum of eight hours apart. Due to the island's narrow width (<2.35 km at any point), we determined that triangulation was not necessary to delineate a home range, assuming that animals use the entire width of the island. We recorded telemetry locations by traversing an access road and recording

locations with a geographic positioning system where the signal vector was approximately perpendicular to the nearshore beach. Locations for all animals were plotted in ArcMap and home ranges were drawn within the longitudinal bounds given by the telemetry locations. The resulting polygon areas were calculated using Hawth's Tools in ArcMap. We calculated the difference in home range size for animals recaptured and not recaptured by the camera traps. We also compared camera abundance and camera orientation within the home ranges of recaptured and non-recaptured individuals. We tested the legitimacy of assuming animals use the full width of the island in our home range calculations by triangulating home ranges for a subset of four radio-collared individuals (three males and one female) based on 40 locations accumulated for each individual from spring to summer 2008.

Survival estimation.-We used the mark-recapture data from the cameras to calculate survival and to test for the presence of animals that only rarely came into the sampling area (hereafter called transients). We tested six different models using a protocol similar to that used by Braun-McNeill et al. (2007):

$$\phi(2 \text{ age})p(\cdot)$$

$$\phi(2 \text{ age})p(t)$$

$$\phi(t)p(t)$$

$$\phi(t)p(\cdot)$$

$$\phi(\cdot)p(t)$$

$\varphi(\cdot)p(\cdot)$

Where φ_i is the probability an animal alive at time i survives to time $i+1$ and p_i is the probability an animal that is alive and available for capture at time i is captured at time i . Models with survival as a two-age parameter incorporate transients. Age one is the period after the first capture (transients and residents), and age two is the annual survival for each individual captured at least twice (residents) (Pradel et al. 1997). We used model selection and AIC criteria to choose the model that best fit the data with the minimum number of fixed parameters (Burnham and Anderson 2002).

Identification Using Natural Marks.— We based our natural marking system on 10 characteristics (Table 2.1, Figure 2.1). Because raccoons were individually identifiable from their artificial marks, we were able to create a catalog of 15 known animals which observers could use to identify unknown animals. Using Photoshop we erased all artificial marks so observers could only use natural marks for identification. We obtained pictures of raccoons from three positions: front, right side and left side. For each catalog animal we chose the highest quality photographs available and represented each animal on one page of the catalog with pictures of the animal in each of the three positions. We then assembled four folders containing five photographs each. Three folders represented each of the three positions while the fourth contained photographs of animals in a variety of positions. Sixteen observers were asked to identify animals in one folder at a time by matching them to animals from the catalog or identifying them as not present in the

catalog. Fifteen observers had no experience identifying raccoons and received basic training in distinguishing the 10 characteristics used for identification and were instructed to ignore other less reliable characteristics which could change over time (i.e. coat color, coat length, animal size and condition). One additional observer had extensive experience in raccoon identification, was considered highly trained and was also instructed to use the 10 characteristics in Table 2.1 for identification. Average success rate as well as success based on animal orientation to the camera was recorded for each observer.

RESULTS

Abundance Estimate.— Abundance estimates increased steadily over time, from 136.80 animals in period one to 287.21 animals in period six (Table 2.2). Detection probabilities were generally low and they varied widely among sampling periods (Table 2.3).

Density Estimate: We calculated home range diameter (1.74 km) as the average home range length of the radio-collared animals. In contrast, MMDM analysis resulted in a boundary strip width of 3.1 km, almost two times the average home range diameter.

Effective sampling area using the telemetry data was 41.26 km² in contrast to 73.51 km² using MMDM. Interestingly, the entire area of the study site is only 28.59 km².

Using the average of all population estimates (199.87 (SE=22.88)) we obtained a density estimate of 4.84 animals/ km² using radio-telemetry and 2.72 animals/km² using MMDM.

Home Range: Of 37 total animals known to be alive over the course of the study, only 41% (n=15) were recaptured on the cameras and 59% (n=22) were never recaptured on the cameras. Average home range size of radio-collared animals recaptured on the cameras differed significantly from those not recaptured over the study period (t=1.76, df=44, p=0.043, $\alpha=0.05$) (Table 2.4). We also found that recaptured animals had significantly more cameras located in their home ranges than non-recaptured animals (t=2.02, df=43.97, p=0.025) (Table 2.4). Recaptured animals had more cameras located in the center of their home range than non-recaptured animals while non-recaptured animals had the majority of cameras located along the edges of their home range with some home ranges not containing any cameras at all. We noticed differences in the habitat composition of home ranges and the proportion of animals recaptured. A higher proportion of non-recaptured animals than recaptured animals had home ranges located in areas of extensive marsh (Table 2.4). We investigated this phenomenon further by triangulating home ranges on a subset of four animals. Triangulated home ranges were substantially smaller than assumed home ranges (Figure 2.2). These observations suggest that some animals rarely used the upland habitats where camera traps were located. For example, although animal 164.193 (a female) was initially trapped near the camera trap transect she spent almost all of her time in the adjacent marsh and was never recaptured.

In contrast, animal 164.014 (a male) spent almost all of his time in beach and upland habitats and was never recorded in the adjacent marsh. Since the average width of the island is only 500m the differences in camera trapping success we report reflect very fine scale differences in habitat use by individual animals.

Survival Estimate: Model selection results show that model $\phi(2 \text{ age}), p(t)$ is the preferred model based on AIC criteria (Table 5). It has the lowest AIC value and the highest model weight and model likelihood. This model includes $\phi(2\text{-age})$, the parameter that incorporates transients.

Identification Using Natural Marks: The average rate of overall success in identification using natural markings was only 28% (n=300) for observers who received the basic training. We found that observers who received the basic training correctly identified animals from the catalog 38% of the time but only correctly identified animals not in the catalog 15% of the time. Although the rate of successful identifications was low, we found 25% agreement between observers when the identification made was incorrect. In contrast the one highly trained observer who participated showed 65% success in identification. We found that several individuals were highly identifiable and all observers correctly distinguished those animals at least 60% of the time. Finally, we found that the front-facing orientation (15% correct) was not as useful as right side (39% correct) or left side (29% correct) orientations.

DISCUSSION

This study provided several unique tests of camera trapping methodologies. The island study site was well isolated from the mainland and other islands and therefore considered geographically closed. The average width of Core Sound is 4.4 km and is 2 km wide at its narrowest point. We saw no evidence over the course of the study that raccoons moved from the island to the mainland or to other islands. The relatively small site with flat, homogeneous terrain, simple vegetative structure, and a high density of raccoons provided optimal conditions for comparing parameter estimates based on camera traps with those from radio-telemetry. The sources of bias evident in this study are likely smaller than would be expected on study sites with more complex terrain, more heterogeneous habitat, or with species that occur in lower densities.

Mark-recapture models for closed populations make several strong assumptions. First, that the population is closed both geographically and demographically (Pollock et al. 1990). Second, that animals retain their individual marks and all marks are noted and recorded correctly. The Lincoln-Petersen model also assumes that there is no variation in capture probabilities over time. Meeting these assumptions in the field can be difficult (DeMaster et al. 1980, Garshelis and Noyce 2006). If assumption violations are not addressed, severely biased estimates may result (Boulanger and McClellan 2001, Evans et al. 2002). In this study, variations in abundance estimates over time probably reflect temporal heterogeneity in detection probabilities caused by changes in animal behavior or

habitat use. This heterogeneity violates key assumptions of the Lincoln-Petersen estimator and it makes the interpretation of camera trap data problematic (Pollock et al. 1990). Out of 37 radio-tagged animals in the population known to be alive, only 41% were recaptured by camera traps during the study. Of those 41%, some animals were caught many times and others only once. This suggests that animals are exhibiting behavioral responses; some are seeking out the camera traps (trap-happy) while others may be avoiding them (trap-shy). A behavioral response to the cameras could be the result of the use of scent baits or a response to the cameras themselves. We used the same scent lure for both the live traps and the camera traps which may have caused animals to avoid the cameras just as they would a live trap. Baited cameras have however been shown to elicit a trap-happy response in other animals (Yasuda 2004). Response to bait can also change with seasonal food availability and seasonal activity levels which could bias estimates between different trapping periods. The cameras themselves can stimulate a behavioral response in some animals. Studies on tigers have demonstrated a trap-shy response due to the camera flash (Wegge et al. 2004) and studies on coyotes have shown avoidance of novel stimuli in an animal's home range (Harris and Knowlton 2001). Séquin et al. (2003) showed that coyotes exhibit a different response to novel stimuli based on an animal's social status; they found that more dominant individuals were less wary around novel objects within their home ranges than subordinate individuals, which were more likely to avoid those objects. The trap-happy

responses we observed could be more dominant individuals defending areas around the cameras, perhaps reducing detection rates for subordinate individuals.

Trap response is not the only way in which an animal's behavior influences detection. Individual variation in home range size and movement can increase or decrease detection rates (Dillon and Kelly 2008, Noyce et al. 2001). We found that animals with larger home ranges were more likely to be captured by cameras, likely because these individuals encountered a higher number of cameras. In addition, we found that animals were more likely to be captured if a camera was located in the center of their home range as opposed to the edge. Rarely is home range variation considered in camera trap studies and researchers assume constant detection probability regardless of potential home range differences due to age or sex, an unreasonable assumption for mammals. We suggest that camera density should be a factor of local home range information gathered in a pilot study wherever possible.

Habitat can affect detection in several ways. Dense habitats are often avoided in camera trapping studies because cameras in dense habitats have a lower effective trapping area than cameras in more open habitats. The habitat preferences of individual animals can also affect detection (Kelly and Holub 2008) but have rarely been addressed in the camera trapping literature. We found evidence that some raccoons preferred marsh habitats to the grassland habitats where our cameras were deployed. Our survival models indicated that some animals only rarely visited the grassy upland sampling areas and they

thus had a lower probability of detection than the rest of the population. Telemetry data confirmed that radio-collared animals not recaptured by camera traps were more likely to have home ranges located in areas of extensive marsh. Triangulated home ranges further established that some animals never came back into the sampling area where they were initially live-trapped. These changes in habitat preference essentially created “holes” in the sampling area where animals remained unavailable for sampling after their initial capture. These holes are likely typical of camera trapping studies because researchers tend to place cameras at locations that are chosen for human convenience and tolerance and not necessarily in the territories of the target species.

The behavioral and habitat-influenced sources of bias we discovered likely contributed to the increase in our population estimates over time. It is unlikely that this increase was due to recruitment because raccoons generally breed only in the spring (Sanderson 1999), so we would have expected a pulse of new animals each spring as cubs began to disperse. We believe the increasing abundance estimates reflect declining trends in detection probability.

One way in which camera-trapping studies attempt to mitigate a behavioral effect on detection is to avoid handling the animals at all. This is achieved by using natural markings to identify individuals rather than applying artificial marks as done in more traditional mark-recapture. We evaluated a natural marking system for raccoons using observers who received only minimal training but showed that a higher level of observer

training would likely improve the accuracy of this system. Any natural marking system requires that most individual animals be highly distinctive and although we found evidence of this in our population there was a subset of individuals that could not be distinguished from each other. Non-distinctive animals are common in any population but this proportion is rarely estimated in camera trap studies but must be accounted for statistically (Forcada and Aguilar 2000). We acknowledge that accuracy of identification is not only positively correlated with individual distinctiveness, but also photograph quality (Stevick et al. 2001, Kelly 2001). There is often a range of photograph qualities in camera trap studies based on the equipment and the environment (Stevick et al. 2001) and a substantial amount of photographs taken will need to be eliminated from the analysis because of this. In addition to photograph quality, we found that the degree to which animals are identifiable depends greatly on their orientation to the camera. When animals face the camera the tail is often obscured and this limits the number of identifiable characters available to the observer. If this type of system for raccoons were to be pursued, researchers would need to set up their cameras such that the entire side of the animals is captured (see Karanth and Nichols 1998).

With rare exceptions (Forcada and Aguilar 2000, Stevick et al. 2001, Reisser et al. 2008) most camera-trapping studies do not use a double-tagging system to validate their natural marking scheme but instead use a validation system that quantifies the degree to which a group of experts agree. This system is not as robust as the double-tagging

system and is fraught with potential observer error and bias due to lack of individual distinctiveness. In our study, we found that observers agreed 25% of the time even when identification was incorrect. Misidentification is a large problem in mark-recapture studies and is rarely addressed (see Yoshizaki et al. 2009 for a discussion on this topic). The lack of sufficient validation in most studies using natural markings could make inferences based on those results suspect and we encourage researchers to use caution when using this type of identification system.

Radio-telemetry allowed us to explore another contentious topic in camera trapping: density estimation. Our data show an almost two-fold difference in the density estimate derived through use of MMDM compared to the estimate derived through the use of home range data from radio-telemetry. Dillon and Kelly (2008) questioned the reliability of MMDM or 1/2MMDM as a proxy for home range size. Dillon (2005) noted that if camera spacing is large compared to the home range of the target animal, MMDM will artificially inflate home range estimates. This could account for some of the inflation in the MMDM home range estimate at our study site. A lack of information on true home-range size could also result in an overestimate of effective trapping area and an underestimate of density (Soisalo & Cavalcanti 2006). Telemetry data showed that many of our animals had home ranges smaller than the 1.8 km spacing between cameras. In fact several animals did not have any cameras within their home ranges. Radio-collared animals with small home ranges will be factored into the telemetry buffer width,

making it smaller. Such animals with small home ranges are ignored when calculating MMDM because they are never captured on the cameras, inflating the buffer width. Animals captured one or more times by a single camera were eliminated from the analyses because movement distances could not be calculated. This loss of information would further bias MMDM estimates (Dillon 2005). These findings illustrate the importance of radio-telemetry to support camera trap study design.

Camera traps have become a popular tool for sampling wildlife populations and an excellent tool for collecting data on rare or elusive species. However, camera traps used without accurate local information can lead to invalid inferences, especially in the case of density estimation, even in areas where animal movement is limited. We recommend that practitioners use supplemental information from radio telemetry studies to test the assumptions and aid in the design of camera trapping studies whenever possible. Where local information is unavailable, spatially explicit models may be useful. Royle and Gardner 2009 outline a method to estimate density which contains explicit models for the spatial point process governing the distribution of individuals and their exposure to and detection by traps. Using this model approach, each individual in the population possesses an encounter history matrix which accounts for the spatial information of the traps and eliminates the need for an *ad hoc* estimation of effective trapping area. This approach also addresses problems such as individual heterogeneity in

capture probabilities and the presence of potential `holes' in the array which make it an attractive alternative to traditional methods of analysis.

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TABLES AND FIGURES

Table 2.1: Physical characteristics used for visual identification of raccoons from photographs

Characteristics	Category	Definition
Number of tail stripes	Open Category	Number of light and dark stripes from tip to rump
Pattern of tail striping	Open category	Distinct pattern of thick vs. thin, light vs. dark stripes
Tail to body length ratio	Open category	
Ear shape	Rounded or pointed	Round tip or pointed tip
Presence of ear notches	Present or absent	
Face mask thickness	Thick or narrow	Thin is just covering eyes (top to bottom), thick is covering eyes and extending outward from eyes
Face mask shape	Pointing upward, downward or straight across	The direction that the top edge of the mask curves
Blaze length	Forehead, between eyes, to nose, absent	How far down face blaze extends
Blaze shape	Diamond or straight	Diamond points out at widest point and comes in, straight is same width down entire length
Presence of “stockings”	Present, absent	Black lines above rear knees

Table 2.2: Population estimates for each camera-sampling period using Lincoln-Petersen model

Sampling Period	Marked Caught	Unmarked Caught	Total Caught	Total Marked in Population	LP Estimate	SE
18/5-30/06/07	85	112	197	57	136.8	11.6
1/7-15/8/07	150	105	255	94	159.8	8.3
16/8-1/10/07	141	104	245	96	166.8	9.1
2/10-17/11/07	74	91	165	96	214.1	18.4
18/11-17/2/08	26	41	67	96	234.5	35.3
21/6-25/8/08	38	69	107	102	287.2	36.9

Table 2.3: Detection probabilities for each camera-sampling period

Sampling Period	Individual Marks	Total	Detection	SE
	Detected	Marked	Probability	
18/5-30/06/07	12	46	0.26	0.06
1/7-15/8/07	30	76	0.39	0.06
16/8-1/10/07	39	78	0.50	0.06
2/10-17/11/07	25	83	0.30	0.05
18/11-17/2/08	11	79	0.14	0.04
21/6-25/8/08	21	97	0.22	0.04

Table 2.4: Home range results for animals recaptured and not recaptured on camera traps.

Variable	Recaptured Animals (n=22)	Non-Recaptured Animals (n=15)
Home range size	200.5 ha (SE=20.2 ha)	159.2 ha (SE=16.4 ha)
Cameras in home range	1.7 (SE=0.72)	1.2 (SE=0.88)
Camera orientation:		
Center	86% (n=19)	36% (n=5)
Edge Only	14% (n=3)	52% (n=8)
Absent	0	12% (n=2)
Home ranges in extensive marsh	23% (n=5)	59% (n=9)

Table 2.5: AIC model selection results for survival models based on camera trapping data

Model	AIC _c	Delta AIC _c	Weight	Likelihood	Number Parameters	Deviance
$\varphi(2\text{-age}), p(t)$	573.315	0	0.95819	1	8	102.811
$\varphi(t), p(t)$	580.218	6.9	0.03036	0.0317	11	103.145
$\varphi(\cdot), p(t)$	582.334	9.02	0.01054	0.011	7	113.98
$\varphi(2\text{-age}), p(\cdot)$	588.332	15.02	0.00053	0.0006	3	128.384
$\varphi(t), p(\cdot)$	589.491	16.18	0.00029	0.0003	7	121.137
$\varphi(\cdot), p(\cdot)$	592.095	18.78	0.00008	0.0001	2	134.201



Figure 2.1: Individual identification of raccoons at Cape Lookout National Seashore, North Carolina. (A and C) Two different individuals showing the differences in mask shape, A is straight and C is upward. Both A and C show the presence of black “stockings” on the back feet. (B and D) Two different individuals showing the differences in blaze length, B is to nose, D is between eyes.



Figure 2.2: Assumed home ranges in white, triangulated home ranges in black of four radio-collared raccoons. 164.223, 164.285 and 164.014 are males and 164.193 is a female.

CHAPTER 3

Assessing the effects of population reduction on raccoon population dynamics and prey productivity

ABSTRACT

Predator removal can be an effective tool for increasing the productivity of prey species but few studies assess the effects removal actions can have on the remaining predator population. Predator removals have had only mixed success and have become increasingly controversial. Where native generalist predators have become overabundant, predator control may be an appropriate and necessary management action but prior knowledge of impacts on the predators themselves, the target prey, or the scope of the unintended consequences to other species in the system need to be researched. We used 14 years of nesting data to examine the effect of removal of 50% of the raccoon population on predation rates and productivity of nesting species on an island in the Outer Banks of North Carolina. We also examined how reducing the number of raccoons affected the ecology of remaining raccoons. Following the removal of raccoons, predation rates decreased for piping plover and American oystercatcher and several sea turtle species and productivity increased for the piping plover and American oystercatcher, but changes were not statistically significant compared to pre-removal estimates. Although a population projection model showed that this raccoon population

was decreasing before the removal we found few changes in raccoon survival, home range size, location or overlap in the first year following removal. A sensitivity analysis revealed that changes in juvenile survival had the greatest effect on population growth and we recommend that managers focus on juveniles if future removal actions are warranted.

INTRODUCTION

The reduction of predators to protect prey species has sparked considerable debate in recent years. Some studies have shown predator removal be effective (Engmann et al. 2002, Garmestani and Percival 2005, Stancyk 1982) but others question its merit and long-term effectiveness (Cote and Sutherland 2004, Meckstroth and Miles 2005, Parr 1993, Patry 2001). Removal of mammalian predators on islands is frequently attempted (Donlan et al. 2003, Campbell et al. 2004, Cruz et al. 2005, Jones et al. 2006, Keitt and Tershy 2003, Mulder et al. 2009) because population of prey species are often more vulnerable (e.g. smaller numbers) than on the mainland and eradication is easier where predators are confined to a limited space. Removal of predators on islands has historically focused on small islands (<60,000 ha) and exotic species and several of these programs have been successful (Knowlton et al. 2007, Campbell et al. 2004, Cruz et al. 2005) but examples of successful predator control on large islands also exist although

these cases are in the minority (Nogales et al. 2004, Clout and Russell 2006). Although exotic predators are often thought the most dangerous because native endemics on islands are naïve to predation by introduced species (Salo et al. 2007), overabundant native predators, especially where they are subsidized by anthropogenic resources, can be equally detrimental (Garrott et al. 1993). Therefore, the development of effective predator management techniques for both native and exotic species is often crucial to the persistence of rare endemics on islands (Whitehead et al. 2008, Engeman et al. 2005, Jones et al. 2006). Furthermore, islands are ideal sites for experimental manipulations to evaluate the efficacy of predator management techniques because they mimic closed systems.

Raccoon (*Procyon lotor*) populations can reach high densities in many coastal areas and have been implicated in high levels of nest predation of piping plovers (*Charadrius melodus*), American oystercatchers (*Haematopus palliatus*), and green (*Chelonia mydas*), leatherback (*Dermochelys coriacea*) and loggerhead (*Caretta caretta*) sea turtles in the southeastern U. S (Erwin et al. 2001, Davis et al. 2001, Ratnaswamy and Warren 1998, Schulte and Simons *in review*). On Cape Lookout National Seashore, high levels of raccoon nest predation have prompted the National Park Service to consider predator control as a management option. The National Park Service has historically been reluctant to manage native species (Wright 1999) however protected areas such as national parks are increasingly becoming the last refuges for the persistence of

threatened, rare and endangered species. Managing native predators may be a necessary step in the conservation of species of special concern but an understanding of the role of native predators in the ecosystem is necessary. Our objectives were to assess whether raccoon reduction was effective for increasing the productivity of nesting shorebirds and sea turtles on an island along the coast of North Carolina while at the same time determining the effect of the reduction on the raccoon population itself. Thus, our first objective was to assess the dynamics of the raccoon population by estimating abundance and gathering baseline demographic data to evaluate the response of the raccoon population to our experimental reduction. Assessing the effect of management programs on native predator populations requires an understanding of the mechanisms and behaviors involved in regulating the population's demographic processes (Caut et al. 2008, Conner et al. 1998, Harding et al. 2001, Knowlton et al. 1999). The second objective was to observe any change in nest predation and prey productivity following raccoon removal. Observational studies done on large scales are limited by the lack of suitable control sites or the opportunity for replication due to logistical and monetary constraints. Gathering extensive before and after data at one site often presents the best option for drawing conclusions about the effect of a treatment at that site. Changes based on one year of after data must be interpreted with caution. Monitoring will continue at this site over the next two years to draw stronger conclusions about the effects of predator removal.

STUDY AREA

The Outer Banks is a ~320 km stretch of barrier islands off the coast of North Carolina extending from Currituck to Cape Fear. Cape Lookout National Seashore (CLNS), a unit of the National Park Service (NPS), is comprised of a chain of four barrier islands at the southern end of the Outer Banks off the coast of Beaufort, North Carolina. South Core Banks (SCB) is the southernmost island of Cape Lookout National Seashore, it is 35 km in length and its width varies from 0.26 km to 2.35 km, averaging approximately 0.5 km wide. South Core Banks has a northeast-southwest orientation, and exhibits a low profile landscape of wide, bare beaches with low dunes covered by scattered grasses, flat grasslands bordered by dense vegetation, and large expanses of salt marsh along Core Sound which separates the island from the mainland.

METHODS

We captured and marked a total of 131 raccoons over three sampling periods between May 2007 and June 2008 using box traps (Tomahawk and Havahart). Animals were anaesthetized using Telazol[®] (100mg/ml) (Elkin-Sinn, Incorporated, Cherry Hill, NJ) at a dosage of 10mg/kg (Gehrt et al. 2001). We weighed and measured all animals, took a premolar tooth for aging which was sent to Matson's Lab (Milltown, MT) for cementum

analysis. All captured animals were fitted with either collars made of 2.5cm wide polyurethane coated nylon or radio collars equipped with mortality sensors (n=37) (Advanced Telemetry Systems Inc. (ATS), Asanti, MN). All collars were equipped with individually numbered ear tags (National Band and Tag Company, Newport, KY). All handling procedures for live animals were approved by the Institutional Animal Care and Use Committee of North Carolina State University (Protocol# 07- 0120-0).

We used infrared trail monitors (TrailMaster™) with 35mm cameras (400 ASA film) (hereafter called camera traps) to recapture marked individuals for population estimation. Twenty cameras were set up in May 2007 along the length of South Core Banks, spaced approximately 1.8 km apart. Each camera trap was armed from 9 PM to 5 AM every day from May 18th to October 21st 2007. From November 17th 2007 until January 25th 2008 and from May to August 2008 cameras were only armed for one week a month. Camera traps were baited with dry cat food, fish sauce and a scent lure (Cronk's Outdoor Supplies, Wiscasset, ME). Film was collected from cameras once a week and photographs were analyzed to determine how many marked and unmarked animals were present during each sampling period. The ratio of marked to unmarked animals was used to generate a Lincoln-Peterson population estimate for each sampling period (Williams et al. 2002: 290-296). We used methods different from the typical Lincoln-Petersen model. Typically when using the two-sample Lincoln-Petersen estimator, animals are captured, marked and released during the first sample and the

number of marked animals from the first sample recaptured in the second sample is counted. In our study animals were live-trapped and marked outside of the camera sampling periods and unmarked animals captured during the camera sampling periods were not marked when captured. The number of marks in the population during each period was not derived from how many animals were captured and marked in the first sample but instead the number of marks in the population for each period was estimated as the number of marked animals released from the live-trapping sessions corrected for period mortality derived from a Kaplan-Meier survival curve (Waldstein et al. *in prep*).

We collected location and movement data from radio-collared animals using a portable receiver and three-element hand-held Yagi antenna. We obtained 100 locations on each animal during the study period. Locations were equally divided among four time intervals: 00:00-6:00, 6:00-12:00, 12:00-18:00 and 18:00-00:00. Consecutive locations were taken a minimum of eight hours apart. Due to the island's narrow width (mean=0.05 km), we determined that triangulation was not necessary to delineate a home range, assuming that animals use the entire width of the island. We recorded telemetry locations by traversing an access road and recording locations with a geographic positioning system (GPS) where the signal vector was approximately perpendicular to the nearshore beach. Locations for all animals were plotted in ArcMap (Version 9.2, ESRI Redlands, CA) and home ranges were drawn within the longitudinal bounds given by the telemetry locations. The resulting polygon areas were calculated using Hawth's Tools in

ArcMap. We maintained 15 radio-collared individuals in the population to allow for comparison of home range area, location and overlap before and after raccoons were removed. Comparisons were made using JMP[®] statistical software (SAS Institute, Cary, NC).

We assessed raccoon activity on beach and non-beach habitats because shorebirds and turtles nested almost exclusively on beaches. We analyzed the activity of raccoons on the beach between 6pm to 6am, the time most animals were foraging. To determine if the frequency of beach activity differed significantly between pre- and post-removal periods, we performed a contingency analysis using Pearson's Chi-squared test statistic (Chernoff and Lehmann 1954).

We used box traps (Tomahawk and Havart) and leghold traps (EGG, Duffer and Lil' Grizz) to capture and humanly euthanize 149 raccoons (~50% estimated population size) from SCB between August 2008 and April 2009. We used an IV injection of Beuthanasia solution (pentobarbital sodium and phenytoin sodium) (Schering-Plough Animal Health Corp., Omaha, NE) following complete anesthesia with 100 mg/ml Telazol, as approved by the American Veterinary Medical Association (AVMA, 2001) protocol for animal euthanasia and by the IACUC at North Carolina State University (Protocol# 07- 0120-0). Euthanized animals were transported to the NCSU College of Veterinary Medicine where they were necropsied in January and May 2009 to gather information about age and litter size. We aged euthanized animals in the same way as

live-trapped animals. We estimated litter size by counting fetuses or placental scars in the uteri of necropsied females.

The NPS has collected productivity data on piping plover and several sea turtles species (loggerhead, green and leatherback) on SCB since 1998 and 2000, respectively. Data on the American oystercatcher have been collected on SCB since 1995 (Schulte and Simons *in review*). Beginning in early April, shorebird nesting areas were searched at least three times per week for territorial pairs and nests. The locations of nests were recorded, and the nests were monitored daily until they hatched or were lost. For each nesting species information on predation was also collected (LaRivière 1999). We also evaluated the effect of predator removal on predation rates and nest success of piping plover, American oystercatcher and green, loggerhead and leatherback sea turtles by looking at time plots of mean productivity for these species on SCB.

We constructed a population projection model using a Leslie matrix (Leslie 1945) to analyze which parameters would be the most efficient targets for future control efforts. We used data on females in three different age-classes: juvenile (age 0 years), subadult (age 1-2 years) and adult (age 3+ years). We calculated the pre-removal age vector by taking the total number of females in the population (143 (SE=11.44)) (Waldstein et al. *in prep*), assuming a 1:1 sex ratio, and divided the total based on the proportion of total females captured during the study that were in each age class (8% juveniles, 25% subadults, 67% adults) resulting in an initial age vector of 11 juveniles, 36 subadults and

96 adults. We calculated fecundity and survival for each age class using data from a previous study (Waldstein et al. *in prep*). Pre-removal survival was 0.83 (SE=0.03) for adults and subadults but because so few juveniles were captured we used juvenile survival (0.65) for a nonharvested raccoon population in Texas reported by Gehrt and Fritzell (1999). Litter size was found to be 2.67 (SE=0.17) which we divided by two and corrected for juvenile survival to represent the number of female offspring each female produces. We assumed juveniles did not breed and a 3% decrease in the number of offspring produced for subadults (Fritzell et al. 1985). The resulting model was projected over 20 years. We calculated population growth rate (λ) and performed a sensitivity analysis to determine the sensitivity of λ to changes in different vital rates. The variance of each parameter estimate was used to calculate the standard error of λ using the delta method.

RESULTS

We removed 50% (n=149) of the estimated raccoon population size. Population monitoring for one year following removal did not reveal any changes in home range size or overlap in those raccoons that remained on SCB (Table 3.1, Table 3.2). Only 38% (n=26) of individuals monitored throughout the study demonstrated a home range shift. We found a significant difference in the frequency of beach foraging between the pre-

(n=1309, 6%) and post-removal (n=671, 3%) periods ($\chi^2=8.148$, $p=0.0043$, $\alpha=0.05$).

Beach foraging was more frequent during the pre-removal period.

Nest predation rates dropped for piping plover, American oystercatcher and sea turtle species following raccoon removal (Fig. 3.2). Piping plover nest predation rates dropped from 27.6% (n=22) in 2008 to 5.6% (n=18) in 2009. American oystercatcher nest predation rates dropped from 62.1% (n=44) in 2008 to 23.0% (n=30) in 2009. Sea turtle nest predation dropped from 41.8% in 2008 (n=55) to 15.9% (n=69) in 2009.

Productivity increased for piping plover from 0.23 fledglings/pair (n=22, SE =0.09) in 2008 to 0.64 fledglings/pair (n=18, SE=0.25) in 2009 (Fig. 3.3) but the difference was not statistically significant ($t=1.089$, $df=24$, $p=0.287$, $\alpha=0.05$). Productivity increased for the American oystercatcher from 0.208 fledglings/pair (n=44, SE=0.12) in 2008 to 0.5 fledglings/pair (n=30, SE=0.17) in 2009 (Fig. 3.3) but it was not statistically significant ($t=1.471$, $df=40$, $p=0.149$, $\alpha=0.05$). Productivity decreased for the sea turtles from 67.49 hatchlings/nest (n=55, SE=6.89) in 2008 to 66.0 hatchlings/nest (n=69, SE=6.32) (Fig. 3.3) but this difference was not statistically significant ($t=0.005$, $df=112$, $p=0.996$, $\alpha=0.05$).

Our population model (Fig. 3.1) showed a population growth rate of 0.95 (SE=0.021) which indicates that this population is decreasing by 0.05% each year with a 95% confidence interval of (0.90, 0.99). The highest value in the sensitivity matrix was

for juvenile survival, indicating that changing juvenile survival will have a greater effect than any other vital rate on changing the population growth rate.

DISCUSSION

Recently, a number of studies have eradicated predators from islands (Campbell et al. 2004, Cruz et al. 2005, Donlan et al. 2003, Jones et al. 2006, Keitt and Tershy 2003). Insular environments can offer insights into population dynamics by eliminating or reducing the risk of immigration and emigration, whereby plausible assumptions can be evaluated. In addition, predator reduction programs rarely consider the changes that the predator population undergoes. Changes or lack of changes in population dynamics following predator removal can give managers insight into how predator populations are regulated and better inform future management strategies.

We found no differences in home range size or overlap in the raccoon population following removal however these data are preliminary and could represent between-year variation. Frey and Conover (2007) found that a reduction of raccoon density on a refuge in Utah did not result in increased home range size or decreased overlap. They concluded that because food was abundant in the study area, predators did not need to expand their home ranges, even if the space were to become available. Similarly, if food or shelter is clumped either spatially or temporally, animals will overlap their home ranges to exploit

those resources and that overlap will not change in response to a reduction in population density. In their study, Frey and Conover (2007) actually found that raccoon home range overlap increased because raccoons were exploiting a small number of pools containing carp and competition was diminished by the reduction in population density allowing the remaining raccoons to tolerate each other in smaller numbers. Rosatte et al. (2007) found that a density reduction in a raccoon population in Canada did not result in dispersal of raccoons into newly unoccupied areas but that a high reproductive rate in the remaining raccoons quickly allowed those areas to become recolonized.

We found a drop in predation rates on piping plover, American oystercatcher and sea turtles in the year following raccoon removal (2009) compared to the previous year. Productivity increased for the piping plover and American oystercatcher from 2008 to 2009 but decreased for the sea turtle species for the same period. Indeed beach foraging of raccoons was more frequent in the pre-removal period; however, this could simply represent between year variation. Although raccoon removal has been effective for reducing depredation of ground nests in some areas (Garmestani and Percival 2005), removing an insufficient number of raccoons usually offers little benefit to prey (Côte and Sutherland 1997). To address this concern we used abundance estimates derived from mark-recapture sampling to determine our removal goal of 50% of the population (~150 animals); however, heterogeneity in detection probabilities can cause biased population estimates resulting in a small proportion of raccoons being removed than

previously thought. We found several sources of bias associated with our mark-recapture sampling techniques that could result in inaccurate estimates of abundance and density (Waldstein et al. *in prep*) and result in too few animals being removed. Other researchers have suggested that only a select few individuals in a population are responsible for the bulk of ground nest predation on beaches (Conner et al. 1998, Ratnaswamy and Warren 1998, Engeman et al. 2003, Ratnaswamy et al. 1997) and that those individuals should be specifically targeted whenever possible. Ratnaswamy et al. (1997) suggest burying leg-hold traps in the sand in nesting areas to specifically target individuals that have learned to exploit ground nests found on the beaches. They argue that if those individuals are not removed, predation rates will not significantly change. Assuming that we removed at least some of the individual raccoons responsible for beach nest depredation, vacant home ranges in the uplands adjacent to beach nesting areas could be filled by dispersing raccoons from the salt marshes (Goodrich and Buskirk 1995, McDonough et al. 2007). Under these circumstances the risk of predation continues as new individuals learn foraging strategies in the beach environment. In the case of SCB, only 38% of radio-collared animals demonstrated a home-range shift following removal suggesting that animals were not quick to fill vacant home ranges. Finally, the effect of raccoon removal may take more than one year to manifest via increased productivity of prey species. Schulte and Simons (*in prep*) suggest that hurricane activity on the islands of Cape

Lookout National Seashore increased American oystercatcher productivity and on some islands productivity continued to increase 1-2 years after the hurricane event.

Our population projection model showed that this population was slowly decreasing pre-removal. A sensitivity analysis revealed that juvenile survival is the most sensitive vital rate and if this can be reduced it will have the greatest impact on the overall population growth rate. The major caveat to this statement is that the juvenile survival estimate was not derived from local data and could be different than the true local value. Nevertheless, this analysis gives managers a place to start for future management of this population.

There remains a good deal of uncertainty about how this population will respond to removal. Many studies of predator removal discuss the likelihood of a predator population rebounding quickly due to density-dependant effects (Goodrich and Buskirk 1995) or recovering to numbers higher than those before control (Van Vuren and Smallwood 1996). It is likely that the raccoon population on South Core Banks will require ongoing management. Martin et al. (*in press*) analyzed different management strategies for this population. They suggest that if initial management of this population were undertaken and then abandoned that this could endanger prey species through an ecological trap (Schlaepfer et al. 2002) where the oystercatcher breeding population increased on SCB following raccoon removal due to a decrease in predation rates. If management were then abandoned, predation rates would increase as the raccoon

population rapidly rebounded and more oystercatcher nests would then be at risk. Further field work on the recovery of the raccoon population over the next several years will help enhance the understanding of how this population will recover and how management will need to respond.

In their simulations, Martin et al. (*in press*) were able to identify optimal policies that minimized the removal of raccoons on South Core Banks while maintaining American oystercatcher productivity above a level necessary for the recovery of the species. They used local parameters and estimates from the literature to model both the raccoon and oystercatcher populations and found that maintaining the raccoon population at a threshold value of 25 females (assuming environmental stochasticity) will result in oystercatcher productivity above the minimum level for species recovery. They found that the best optimal management strategy was to initially remove enough raccoons to reduce the population size to the threshold level, requiring the removal of fewer raccoons each subsequent year to maintain the population near that threshold. This strategy resulted in lower cumulative removals of raccoons over time thus was the most efficient strategy simulated. This type of simulation using local data on vital rates has the potential to be an excellent tool to help guide management decisions. It demonstrates the utility of monitoring predator populations before and after a removal effort.

Further monitoring of this population will be essential to determining what effect removal has on the raccoon population as well their prey. It is important to note that all

of the data presented here are preliminary and more years of data need to be incorporated to draw any conclusions. Continued data collection over the next two years will help us detect any lag in productivity increase for the piping plover, American oystercatcher and sea turtles. Collection of raccoon vital rate data for two years after removal will help develop a post-removal population projection and further inform future management strategies.

MANAGEMENT IMPLICATIONS

Raccoon predation on sea turtle and shorebird nests is widely accepted as a conservation concern, and predator control can be an important tool for conservation on some beaches (Barton and Roth 2007, Côte and Sutherland 1997, Engmann et al. 2002, Ratnaswamy and Warren 1998). Our research demonstrates that predator removal programs require a good understanding of both predator and prey population dynamics to be effective.

Predator removal programs may not always be successful, especially where too few predators are removed too infrequently, where the overall distribution and behavior of remaining predators remains the same and where source-sink dynamics come into play (Pulliam 1988). Further research at our study site will help advance understanding of the long-term effects of predator removal on prey productivity.

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TABLES AND FIGURES

Table 3.1: Home range size in hectares of raccoons before and after a reduction in raccoon population density on South Core Banks, Cape Lookout National Seashore.

Stage	N	Mean	SE
Pre-Removal	46	176.00	13.05
Post-Removal	31	161.11	12.0

No significant difference ($t=0.839$, $df=74$, $p=0.202$).

Table 3.2: Degree of home range overlap in hectares of raccoons before and after a reduction in raccoon population density on South Core Banks, CLNS.

Stage	n	Mean	SE
Pre-Removal	244	0.45	0.02
Post-Removal	126	0.49	0.03

No significant difference ($t= -1.14$, $df=236$, $p=0.812$).

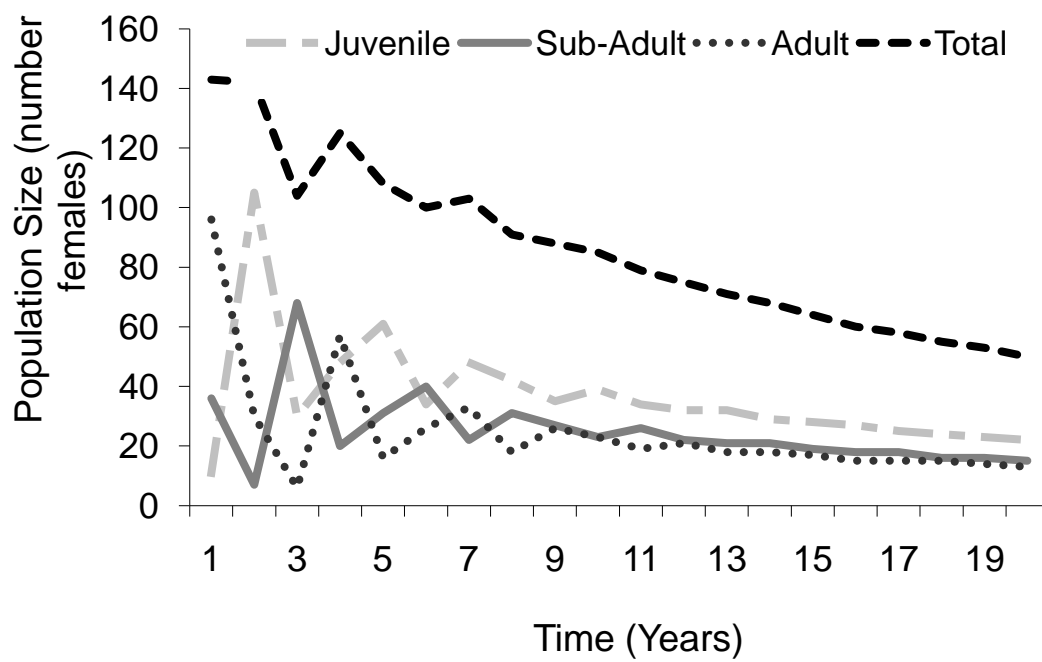


Figure 3.1: Age-specific Leslie Matrix population model for the raccoon population pre-removal projected over 20 years.

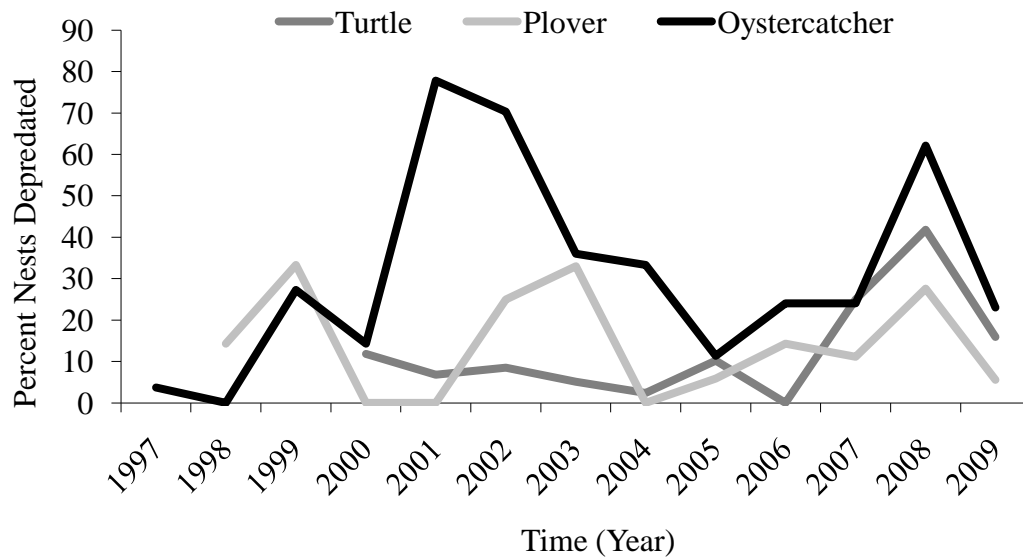


Figure 3.2: Percent nests depredated on South Core Banks, Cape Lookout National Seashore for three prey groups: sea turtles, piping plover and American oystercatcher. All years up to 2008 are pre-removal years and 2009 is post-removal.

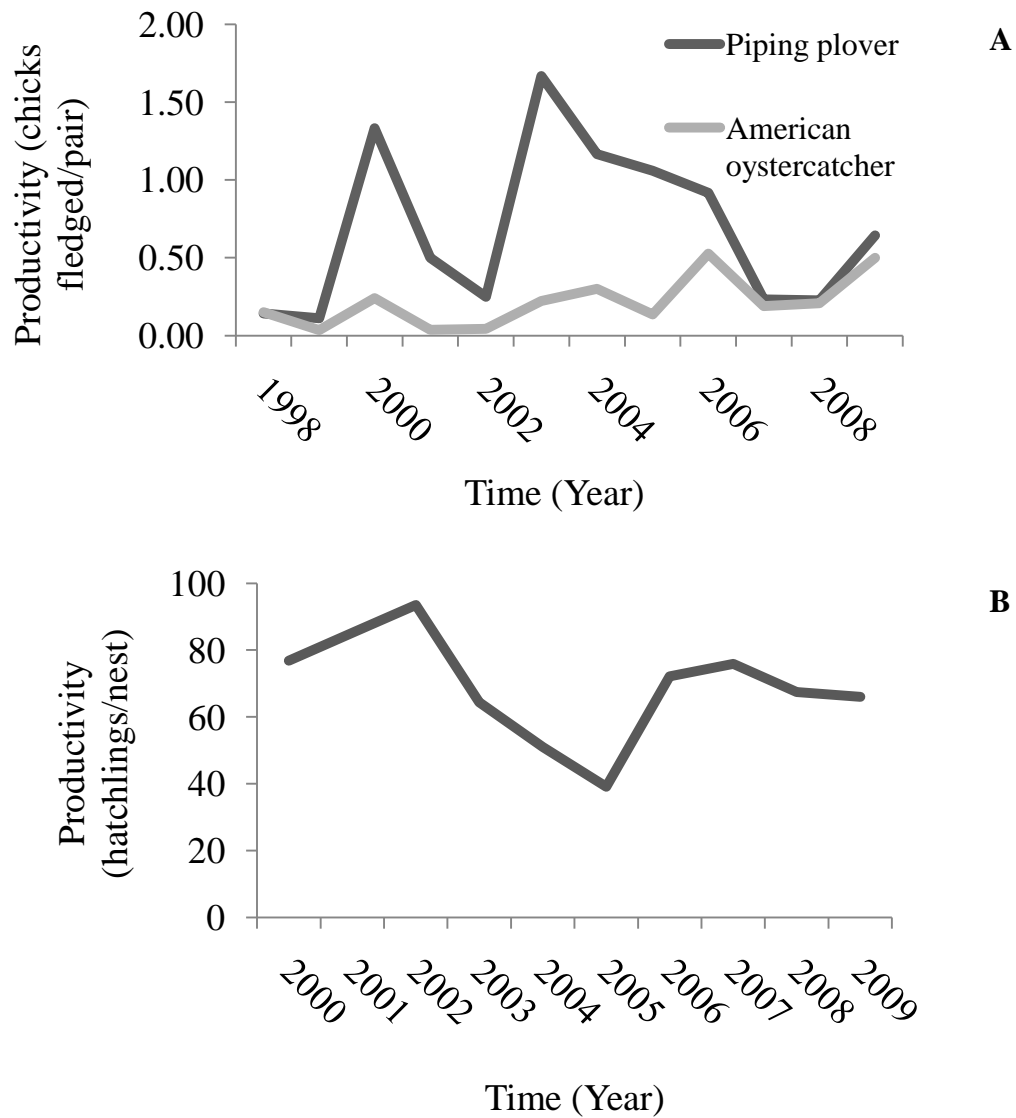


Figure 3.3: Productivity for three groups of nesting species on South Core Banks, Cape Lookout National Seashore from 1998 to 2009. Years 2000-2008 are pre-raccoon removal and year 2009 is post-raccoon removal. (A) is piping plover and American oystercatcher (B) is sea turtles.

APPENDIX

Appendix 1: Raccoons live-captured on South Core Banks from 2007 to 2009

Year Captured	ID	Sex	Age (Yr)	Wt (kg)	Length (cm)	Hind foot length (mm)	Hind foot width (mm)	Shder ht (cm)	Tail length (cm)	Year Removed
5/10/2007	1y	M	3	4.0	84.1	99.5	25.8	30.4	23.6	
5/9/2007	2y	M		3.8	78.2	94.5	28.5	29.5	24.0	
5/9/2007	3y	M	6	3.6	22.2	106.0	35.0	29.2	22.2	
5/9/2007	4y	M		5.4	84.5	103.0	39.0			
5/15/2007	5y	F	3	3.5	82.2	90.6	24.0	31.4	24.9	
6/27/2007	6y	F	2	3.4	73.6	98.0	21.0	27.4	24.6	
5/10/2007	7y	M	1	3.4	81.8	100.1	20.0	31.8	25.3	
5/17/2007	8y	M	1	2.8	79.1	101.0	21.0	30.3	25.5	4/25/2009
5/16/2007	9y	M	2	4.7	85.0	103.0	27.0	32.5	25.0	12/8/2008
5/10/2007	10y	M	2	2.8	78.2	96.8	23.2	27.5	21.4	4/25/2009
5/16/2007	11y	M	9	2.9	83.5	93.0	21.0	29.0	23.0	
5/16/2007	12y	M	1	3.0	78.0	100.0	23.0	30.2	25.5	12/3/2008
5/10/2007	13y	M	3	3.6	80.6	99.5	22.0	30.3	31.4	
5/15/2007	14y	M	2	3.7	84.5	102.5	22.7	30.0	29.0	12/9/2008
5/10/2007	15y	M	4	4.4	84.7	109.0	24.0	32.2	24.5	4/29/2009
5/16/2007	16y	M	2	3.7	48.5	97.0	26.6	31.5	31.0	
6/5/2007	17y	M	5	4.0	76.5	100.1	20.2	30.4	20.9	4/25/2009
6/6/2007	18y	M	2	3.5	79.8	97.0	22.0	26.9	23.1	4/28/2009
6/5/2007	19y	M	10	4.0	84.9	104.0	23.0	32.9	25.5	
6/5/2007	20y	M	2	3.7	79.9	98.0	22.0	29.3	23.5	4/26/2009
5/15/2007	21y	M	4	4.7	84.2	102.0	23.0	31.2	22.6	4/25/2009
5/16/2007	22y	M	5	5.2	88.4	107.0	28.0	32.0	29.0	
5/15/2007	23y	M	2	3.7	81.5	101.0	26.0	33.1	27.0	
6/26/2007	24y	M	4	4.0	85.4	103.0	22.0	31.9	27.4	4/27/2009
5/15/2007	25y	M	4	4.0	79.4	90.0	24.0	30.0	22.5	
5/9/2007	26b	M	5	3.6	78.2	106.0	31.0	30.8	22.6	12/3/2008
5/9/2007	27b	M	4	4.2	80.4	101.0	24.0	31.7	27.7	4/25/2009
5/9/2007	28b	F	2	3.5	79.5	95.0	25.0	31.5	24.4	
5/9/2007	30b	F	4	3.4	81.0	96.0	36.0	30.5	25.4	
5/9/2007	31b	M		4.2	80.3	102.0	34.0		22.4	

5/10/2007	32b	M		3.9	83.9	104.0	37.0	24.7	28.1	
5/9/2007	33b	M	7	5.0	80.1	103.0	35.0	28.7	22.5	
5/10/2007	34b	M	3	4.2	86.7	104.0	21.0	32.4	25.9	
5/15/2007	35b	M	1	2.7	73.8	101.0	20.0	22.7	22.2	
5/15/2007	36b	M	3	4.9	84.9	101.1	26.5	11.6	23.6	4/25/2009
5/15/2007	37b	M	7	4.0	88.0	101.0	23.0	32.2	24.0	12/4/2008
7/3/2007	38b	M	1	3.9	81.6	98.0	22.0	31.3	24.6	12/3/2008
7/3/2007	39b	M	6	4.2	81.8	103.0	29.0	31.5	28.1	4/26/2009
7/3/2007	40b	M	6	4.0	83.6	10.1	20.1	26.0	26.0	12/6/2008
7/3/2007	41b	M	1	2.5	76.7	103.0	21.0	30.5	29.8	12/6/2008
7/14/2007	42b	M	3	5.0	84.9	103.0	30.0	34.1	26.1	12/6/2008
7/3/2007	43b	F	3	3.8	83.2	9.8	23.0	26.0	26.0	
7/14/2007	44b	F	7	3.4	73.6	98.0	20.0	30.3	23.1	4/25/2009
7/14/2007	45b	M	5	4.1	77.6	100.0	22.0	31.2	22.1	12/6/2008
5/17/2007	46b	F	1	2.4	78.0	99.2	24.7	29.8	24.0	
5/21/2007	47b	F	5	2.9	81.2	102.0	25.0	28.5	26.4	
7/14/2007	48b	M	1	3.9	78.3	10.2	22.0	29.5	24.5	
7/15/2007	49b	F	3	3.3	76.6	96.0	22.0	31.4	24.7	12/5/2008
7/15/2007	50b	F	2	2.9	78.5	9.5	18.0	30.0	24.0	
5/17/2007	51y	M	8	4.1	83.4	104.0	23.0	30.4	28.2	12/4/2008
5/17/2007	52y	M	1	3.4	78.4	98.0	23.0	29.0	21.9	
5/21/2007	53y	M	5	4.0	77.3	102.0	24.0	26.9	25.0	12/4/2008
5/21/2007	54y	M	2	3.6	77.5	103.0	26.0	30.0	23.9	12/6/2008
5/31/2007	55y	F	2	3.1	79.9	94.5	17.0	29.8	23.8	12/5/2008
5/31/2007	56y	F	3	3.0	74.5	91.0	20.0	30.9	23.3	
6/26/2007	57y	M	4	3.8	80.5	9.5	24.0		26.7	
6/27/2007	58y	M		3.7	76.1	101.0	23.0	29.4	24.1	
6/27/2007	59y	M	2	3.4	74.3	102.0	21.0	29.8	24.2	
6/27/2007	60y	M	4	3.7	80.5	10.2	24.0	33.0	27.5	
6/27/2007	61y	M	3	3.2	73.1	9.5	23.0	28.4	23.5	12/3/2008
6/27/2007	62y	M	6	3.6	74.9	97.0	22.0	29.2	24.4	
6/28/2007	63y	M		4.0	80.3	104.0	22.0	32.0	28.8	
6/29/2007	64y	F		3.5	75.6	94.0	20.0	29.3	21.1	
6/29/2007	65y	M		3.5	77.8	100.0	23.0	30.1	22.4	
6/29/2007	66y	M		3.9	75.9	99.0	17.0	28.3	24.1	

6/29/2007	67y	M	2	4.0	81.5	96.0	22.0	30.9	28.1	4/25/2009
7/2/2007	68y	M		5.5	81.4	105.0	27.0	31.7	23.5	8/26/2008
7/2/2007	69y	M		4.2	76.1	100.0	23.0	31.4	24.5	
7/2/2007	70y	M	3	4.0	76.4	99.0	23.0	31.1	26.2	4/27/2009
6/29/2007	71y	F	2	3.3	70.8	91.0	22.0	30.7	25.3	4/28/2009
6/29/2007	72y	F		3.0	77.5	9.6	20.0	29.5	24.5	
7/2/2007	73y	F		3.7	83.1	101.0	21.0	28.7	22.3	8/26/2008
7/2/2007	74y	F	7	3.4	75.1	9.3	20.0	30.4	24.0	8/28/2008
7/2/2007	75y	F								
7/17/2007	76y	F	8	3.5	76.0	9.8	19.0	30.4	24.0	
7/17/2007	77y	M	0	4.0	80.7	109.0	23.0	32.3	29.2	
7/24/2007	78y	M	2	3.4	74.3	99.0	23.0	30.0	25.2	12/7/2008
7/17/2007	79y	F		2.7	70.2	95.0	18.0	31.3	24.9	
7/24/2007	80y	F		2.9	74.9	93.0	19.0	28.7	24.1	12/9/2008
7/24/2007	81y	F	2	2.9	70.9	9.1	23.0	28.0	23.8	12/9/2008
7/26/2007	82y	M		4.5	79.1	103.0	23.0	30.1	24.9	
7/26/2007	83y	M		3.8	75.4	93.0	23.0	30.4	25.8	
7/26/2007	84y	M	2	5.9	78.3	9.9	22.0	32.0	22.4	12/1/2008
7/26/2007	85y	M	6	3.5	79.6	99.0	22.0	29.7	23.7	4/25/2009
7/27/2007	86y	M	6	4.8	80.6	98.0	24.0	30.8	22.3	4/28/2009
7/27/2007	87y	M		4.8	83.6	105.0	23.0	32.0	27.1	
7/27/2007	88y	M	5	6.3	80.7	97.0	24.0	31.6	21.9	8/25/2008
7/27/2007	89y	M	5	6.9	80.5	109.0	264.0	34.7	22.8	8/25/2008
7/27/2007	90y	M	7	7.7	79.6	96.0	24.0	31.1	17.0	4/25/2009
7/28/2007	91y	M	3	4.5	81.2	10.4	24.0	32.4	26.8	
7/30/2007	92y	F	7	2.6	71.3	92.0	21.0	27.6	25.1	12/2/2008
7/30/2007	93y	M	2	3.4	77.9	10.1	21.0	31.0	24.9	
7/30/2007	94y	M	4	4.1	77.9	9.9	23.0	31.6	22.3	4/25/2009
7/31/2007	95y	M	1	4.7	82.7	104.0	23.0	32.1	27.6	12/4/2008
7/31/2007	96y	M	3	8.7	86.8	106.0	26.0	35.4	24.1	12/1/2008
8/9/2007	97y	M	6	3.8	83.3	10.6	24.0	32.4	24.4	
8/9/2007	98y	M	1	2.9	77.4	9.8	20.5	29.4	22.3	
8/10/2007	99y	M		4.2	72.5	103.0	26.0	30.1	13.1	
8/10/2007	100y	F	3	3.4	80.3	10.2	22.0	28.9	24.8	12/9/2008

8/10/2007	101 y	M	8	3.8	80.8	102.0	22.0	30.1	27.9	4/28/2009
8/14/2007	102 y	M		4.2	83.1	108.0	23.0	33.8	24.2	
8/15/2007	103 y	M		3.9	65.7	104.0	24.0	32.3	4.5	
8/16/2007	104 y	M		4.2	83.7	104.0	23.0	32.8	23.0	
8/16/2007	105 y	M	2	3.5	80.1	104.0	23.0	31.7	23.4	12/6/2008
10/23/2007	106 y	M	3	4.5	71.0	9.5	20.0	26.9	23.2	12/9/2008
10/23/2007	107 y	M		4.3	81.2	10.3	23.0	29.7	26.0	4/27/2009
10/23/2007	108 y	M								
10/23/2007	109 y	F	5	3.6	78.5	10.0	22.0	30.5	21.0	
10/23/2007	110 y	M	4	4.5	82.5	10.3	21.0	32.5	24.5	12/3/2008
10/24/2007	111 y	M	8	6.2	86.4	10.5	26.0	36.9	22.3	
10/25/2007	112 y	M	1	3.9	82.0	109.0	24.0	33.0	24.9	12/4/2008
5/3/2008	113 y	M	5	3.1	80.5	90.8	27.0	29.6	23.7	
5/4/2008	114 y	M	4	3.2	80.5	97.6	27.0	27.5	25.5	
5/4/2008	115 y	M	2	3.3	81.5	95.0	30.0	29.5	24.5	
5/5/2008	116 y	M	4	3.5	80.2	99.0	25.0	30.9	26.0	
5/6/2008	117 y	M		3.7	80.9	102.0	25.0	31.7	26.6	
5/6/2008	118 y	M	3	3.7	85.1	102.0	26.0	29.3	26.0	
5/6/2008	118 y2	M	3	2.7	78.2	99.0	25.0	28.0	27.1	
5/6/2008	119 y	F	4	3.8	85.0	98.7	27.9	30.5	29.5	12/5/2008
5/6/2008	120 y	M	3	3.2	93.0	100.0	29.0	32.8	28.5	
5/6/2008	121 y	M	3	3.6	77.0	101.0	28.0	30.9	26.7	
5/7/2008	122 y	M	4	4.1	80.9	100.0	27.0	28.5	24.2	4/27/2009
5/7/2008	123	M	8	3.8	86.0	97.8	29.8	33.0	24.5	

	y									
	124									
5/7/2008	y	M	6	4.4	86.0	107.5	30.0	31.5	25.0	
	125									
6/25/2008	y	F	7	2.8	78.2	95.0	26.0	29.7	26.0	
	126									
6/25/2008	y	F	8	3.8	82.5	95.3	24.0	30.5	24.0	4/25/2009
	127									
6/25/2008	y	F	4	3.3	77.9	96.9	24.5	30.9	25.5	
	128									
6/26/2008	y	M		4.5	85.8	101.9	29.5	29.5	26.2	
	129									
6/27/2008	y	M	3	2.9	79.5	96.9	26.2	30.5	24.5	
	130									
6/27/2008	y	M	9	3.6	79.9	97.1	26.1	30.5	24.0	12/8/2008
4/28/2009	1	M	3	3.3	75.9	102.0	26.0	33.4	22.4	4/28/2009
4/28/2009	5	M	2	4.4	80.4	114.0	31.0	33.8	23.0	4/28/2009
4/25/2009	11	M	3	3.5	80.7	104.0	27.0	34.3	23.1	4/25/2009
12/9/2008	12	M	3	5.0	74.2	102.0	27.0	34.8	25.0	12/9/2008
12/8/2008	14	M	3	3.8	75.3	102.0	28.0	30.9	27.0	12/8/2008
12/3/2008	15	M	10	8.9	77.5	105.0	29.0	33.5	24.7	12/3/2008
4/25/2009	17	F	2	3.2	74.7	89.0	25.0	26.7	24.0	4/25/2009
4/26/2009	19	M	3	4.3	82.0	105.0	29.0	33.2	25.7	4/26/2009
12/5/2008	20	M	3	6.8	79.1	106.0	28.0	31.8	27.5	12/5/2008
12/6/2008	24	M	2	4.3	77.1	104.0	29.0	27.9	24.4	12/6/2008
12/6/2008	26	F	2	4.4	73.0	98.0	28.0	31.3	23.7	12/6/2008
4/25/2009	30	M	4	3.3	83.0	106.0	29.0	30.9	25.7	4/25/2009
4/26/2009	32	M	5	4.1	83.8	102.0	28.0	31.7	28.1	4/26/2009
4/26/2009	33	F	4	3.3	75.6	98.0	26.0	31.9	23.9	4/26/2009
4/27/2009	34	F	8	3.2	81.8	105.0	29.0	32.4	28.0	4/27/2009
4/27/2009	36	F	4	2.5	74.0	89.0	26.0	28.8	23.7	4/27/2009
4/28/2009	43	M	1	2.5	76.0	99.0	28.0	28.8	24.9	4/28/2009
4/27/2009	46	M	0	3.9	78.3	106.0	27.0	32.5	23.6	4/27/2009
12/3/2008	50	M	1	3.2	66.5	91.0	24.0	28.4	22.5	12/3/2008
12/3/2008	52	M	0	1.6	63.7	97.0	24.0	26.5	23.1	12/3/2008
12/3/2008	53	M	0	2.2	66.0	91.0	28.0	25.1	23.0	12/3/2008
4/25/2009	56	M	3	5.7	89.1	107.0	29.0	32.5	26.3	4/25/2009
12/4/2008	58	M	2	3.7	70.9	97.0	28.0	29.2	23.1	12/4/2008

12/4/2008	59	M	4	5.2	74.5	99.0	29.0	31.6	24.0	12/4/2008
12/9/2008	62	F	0	2.3	62.2	93.0	27.0	24.9	21.8	12/9/2008
4/28/2009	67	F	4	2.8	72.0	97.0	25.0	26.5	24.0	4/28/2009
4/25/2009	71	F	4	2.8	75.2	98.0	25.0	28.5	22.9	4/25/2009
12/4/2008	72	M	1	5.4	72.4	95.0	26.0	33.9	23.3	12/4/2008
4/25/2009	73	F	3	2.5	75.6	102.0	27.0	31.7	28.0	4/25/2009
12/4/2008	77	F	2	4.8	76.0	96.0	26.0	31.0	26.3	12/4/2008
12/6/2008	79	M	3	5.9	80.0	105.0	31.0	32.2	27.8	12/6/2008
12/7/2008	80	M		2.6	64.2	90.0	29.0	26.3	23.3	12/7/2008
12/9/2008	81	M	2	6.0	74.2	97.0	29.0	32.5	24.5	12/9/2008
4/25/2009	83	M	1	4.0	77.7	101.0	25.0	30.9	26.0	4/25/2009
12/6/2008	84	F	4	3.1	73.0	97.0	28.0	32.4	22.4	12/6/2008
4/26/2009	87	M	3	3.9	76.1	98.0	27.0	31.4	26.5	4/26/2009
4/28/2009	88	M	8	3.1	74.9	100.0	26.0	32.1	24.0	4/28/2009
4/27/2009	89	M	6	4.5	79.0	102.0	28.0	32.9	21.5	4/27/2009
4/26/2009	91	M	1	3.1	78.4	104.0	30.0	30.3	24.2	4/26/2009
4/25/2009	94	M	2	2.9	79.3	104.0	27.0	31.5	26.9	4/25/2009
12/9/2008	98	F	2	2.6	74.4	101.0	25.0	30.8	24.4	12/9/2008
4/25/2009	99	M	2	3.2	72.9	99.0	26.0	30.6	22.1	4/25/2009
12/1/2008	126	F	1	3.8	65.2	92.0	32.0	24.3	24.4	12/1/2008
12/3/2008	201	M	8	6.6	76.7	104.0	32.0	33.6	25.3	12/3/2008
12/4/2008	202	M	1	4.1	74.5	91.0	27.0	28.3	24.3	12/4/2008
12/5/2008	203	M	0	2.5	62.7	93.0	30.0	26.2	24.8	12/5/2008
12/5/2008	204	M	2	6.3	79.1	105.0	30.0	34.0	21.9	12/5/2008
12/5/2008	205	F	0	2.9	63.9	89.0	27.0	29.5	23.3	12/5/2008
12/5/2008	206	F	8	4.0	71.5	95.0	26.0	28.1	21.5	12/5/2008
12/6/2008	207	F	4	3.4	72.0	96.0	29.0	30.4	26.3	12/6/2008
12/6/2008	208	M	2	3.8	73.4	96.0	29.0	30.7	23.5	12/6/2008
12/6/2008	209	F	5	3.2	69.4	100.0	28.0	29.6	26.4	12/6/2008
12/7/2008	210	F	0	2.4	62.1	88.0	25.0	27.2	21.1	12/7/2008
12/9/2008	211	F	4	3.7	74.0	96.0	26.0	30.2	24.0	12/9/2008
12/9/2008	212	M	3	5.0	72.0	107.0	28.0	32.0	25.1	12/9/2008
12/8/2008	213	F	3	4.3	72.8	95.0	26.0	31.9	22.2	12/8/2008
12/8/2008	214	F	3	4.0	69.6	95.0	28.0	29.8	22.7	12/8/2008
12/9/2008	215	M	1	4.5	68.0	91.0	31.0	25.8	21.0	12/9/2008

4/25/2009	217	M	1	3.1	77.6	95.0	26.0	31.4	22.6	4/25/2009
4/25/2009	218	F	1	1.7	67.6	87.0	20.0	25.7	22.5	4/25/2009
4/25/2009	219	F	4	3.7	78.0	100.0	26.0	26.2	24.7	4/25/2009
4/25/2009	220	M	10	3.3	83.1	102.0	29.0	31.3	26.2	4/25/2009
4/25/2009	222	M	3	4.1	83.9	107.0	31.0	33.3	30.0	4/25/2009
4/25/2009	223	M	1	2.2	68.4	95.0	25.0	27.2	22.5	4/25/2009
4/25/2009	224	M	1	3.5	82.3	103.0	30.0	31.7	28.0	4/25/2009
12/1/2008	226	F	9	4.9	69.9	95.0	27.0	30.5	21.5	12/1/2008
4/25/2009	227	F	4	2.1	70.9	96.0	25.0	29.5	20.3	4/25/2009
4/25/2009	228	M	4	3.5	80.3	109.0	29.0	32.7	25.4	4/25/2009
4/26/2009	230	F	1	2.3	69.2	97.0	27.0	28.9	21.2	4/26/2009
4/26/2009	232	F	2	3.1	75.8	97.0	27.0	33.4	23.5	4/26/2009
4/28/2009	233	M	1	2.3	69.2	95.0	26.0	28.5	23.8	4/28/2009
4/28/2009	234	F	2	3.2	78.0	100.0	27.0	30.5	23.5	4/28/2009
4/27/2009	236	F	2	2.7	71.4	95.0	25.0	30.3	24.0	4/27/2009
4/27/2009	242	F	1	1.7	68.2	98.0	25.0	28.9	21.2	4/27/2009
4/27/2009	243	F	7	2.6	81.3	96.0	29.0	30.2	25.6	4/27/2009
12/2/2008	250	M	3	5.3	72.6	101.0	32.0	31.8	24.3	12/2/2008
12/3/2008	251	F	0	3.2	56.3	86.0	22.0	25.0	20.3	12/3/2008
12/3/2008	252	F	3	3.6	73.5	92.0	27.0	27.2	23.5	12/3/2008
4/27/2009	289	F	3	3.1	69.9	93.0	26.0	29.9	19.4	4/27/2009
4/27/2009	292	M	2	3.4	72.7	100.0	28.0	30.7	21.0	4/27/2009
4/29/2009	295	M	4	4.0	78.6	104.0	29.0	32.4	22.2	4/29/2009
12/2/2008	300	M	2	4.9	75.0	102.0	30.0	29.9	24.0	12/2/2008
8/25/2008	UM 1	M	1	3.0	75.5	101.0	30.0	30.2	28.3	8/25/2008
8/25/2008	UM 2	F		2.6	70.9	95.0	24.0	25.3	26.0	8/25/2008
8/25/2008	UM 3	F	3	3.9	72.5	103.0	37.0	26.2	23.0	8/25/2008
8/25/2008	UM 4	M	3	4.1	79.3	107.0	27.0	30.9	24.9	8/25/2008
8/25/2008	UM 5	M	3	3.4	79.3	107.0	31.0	32.1	27.2	8/25/2008
8/26/2008	UM 6	F	3	3.4	73.0	95.0	28.0	31.0	23.8	8/26/2008
8/27/2008	UM 7	F	4	3.0	69.8	93.0	29.0	28.5	24.6	8/27/2008

8/28/2008	UM										
	8	F	7	2.9	64.0	89.0	24.0	27.9	10.9	8/28/2008	

Appendix 2: Raccoon home range area by season from 2007 to 2009

Season	n Female	n Male	Female Home Range Area (ha)	Male Home Range Area (ha)	Female SE	Male SE
Summer 2007	8	16	72.42	134.33	10.24	11.27
Fall 2007	8	16	178.00	249.27	40.21	68.67
Winter 2007- 2008	8	16	106.97	167.22	18.12	46.97
Spring 2008	10	14	81.67	134.03	13.02	14.32
Summer 2008	9	16	81.34	103.54	14.60	10.11
Fall 2008	9	22	134.07	114.22	27.88	11.66
Winter 2008- 2009	4	18	96.94	123.13	15.04	14.49
Spring 2009	4	13	93.17	123.87	11.94	15.99
Summer 2009	2	12	73.98	114.00	0.87	10.12

Appendix 3: Raccoon home range length by season from 2007 to 2009

Season	n Female	n Male	Female Home Range Length (km)	Male Home Range Length (km)	Female SE	Male SE
Summer 2007	2	12	1.15	1.47	0.52	0.29
Fall 2007	15	22	1.61	1.84	0.38	0.20
Winter 2007- 2008	10	18	1.07	1.51	0.11	0.21
Spring 2008	9	15	0.83	1.68	0.21	0.31
Summer 2008	9	16	0.65	1.56	0.16	0.17
Fall 2008	8	20	1.46	1.54	0.41	0.17
Winter 2008- 2009	4	13	0.64	1.53	0.15	0.18
Spring 2009	4	12	0.72	1.50	0.17	0.21
Summer 2009	2	10	0.60	1.77	0.06	0.19

Appendix 4: Live-trapping data for raccoons on South Core Banks from 2007-2009

Period	Trap Nights	New Captures	Recaptures	Total Captures
5/8/07-6/30/07	167	56	4	60
7/1/07-8/15/07	300	48	33	81
10/22/07-10/24/07	48	7	6	13
5/03/08-6/26/08	103	18	10	28
8/25/08-8/28/08	40	8	5	13
12/1/08-12/9/08	832	40	39	79
4/25/09-4/29/09	446	43	35	78

Appendix 5: Camera-trapping data for raccoons on South Core Banks from 2007-2009

Period	Trap Nights
5/18/07-6/30/07	860
7/1/07-8/15/07	900
8/16/07-10/1/07	760
10/1/07-11/17/07	840
11/17/07-2/17/08	500
6/21/08-8/26/08	420
5/12/09-7/19/09	420

Appendix 6: MMDM buffer width and home range diameter for raccoons recaptured on cameras on South Core Banks

Animal ID	MMDM	Home range diameter (Radio collars only)
1y	0	
9y	3.6	2.01697198
10y	0	
11y	0	2.14463338
12y	0	1.60606796
14y	1.8	
15y	0	
16y	0	2.2393508
17y	0	
18y	0	
19y	5.4	
20y	1.8	
21y	3.6	
25y	3.6	2.13804984
26b	0	
27b	0	
32b	0	1.34638024
34b	1.8	
35b	0	
37b	1.8	
38b	0	
40b	0	
42b	0	
43b	0	0.50214776
45b	0	
48b	1.8	
49b	0	2.42544874
50b	0	2.46744762
51y	0	
52y	0	2.02458858

53y	3.6	2.59516246
55y	0	1.00638358
57y	1.8	1.81832908
58y	1.8	
60y	5.4	
61y	0	
63y	0	
70y	0	
72y	1.8	
75y	0	
76y	1.8	
77y	1.8	
79y	0	2.66885232
80y	0	0.65269718
81y	0	1.08476118
82y	0	
84y	0	
85y	0	
88y	0	
90y	0	
91y	0	
94y	5.4	
95y	0	
96y	0	
101y	0	
102y	7.2	
104y	0	
105y	0	
112y	0	
113y	0	1.4940516
115y	0	1.30372974
117y	0	1.63323832

Appendix 7: Raccoon sexual maturity and litter size on South Core Banks

Animal ID	Age	Sex	Testes Weight	Baculum Length	Baculum Weight	Uterine Scars	Fetuses	Mature
1	3	M	15.3	95.5	3.8	N/A	N/A	Y
5	2	M	17.75	102.9	4.1	N/A	N/A	Y
11	3	M	16.1	102.3	4.75	N/A	N/A	Y
12	3	M	14.2	94.8	4.2	N/A	N/A	Y
14	3	M	9.8	94	2.35	N/A	N/A	N
15	10	M	17.85	92.7	3.15	N/A	N/A	Y
17	2	F	N/A	N/A	N/A	N/A	3	Y
19	3	M	15.2	108	4.75	N/A	N/A	Y
20	3	M	13.2	102.8	3.4	N/A	N/A	Y
24	2	M	11.65	90	4.3	N/A	N/A	Y
26	2	F	N/A	N/A	N/A	2	N/A	Y
30	4	M	14.1	96.3	4.7	N/A	N/A	Y
32	5	M	19.6	103	3.05	N/A	N/A	Y
33	4	F	N/A	N/A	N/A	N/A	3	Y
34	8	F	N/A	N/A	N/A	0	N/A	N
36	4	F	N/A	N/A	N/A	2	N/A	Y
43	1	M	1.4	63.5	1.25	N/A	N/A	N
46	3	M	16.95	96.1	3.85	N/A	N/A	Y
50	1	M	1.8	57.9	0.85	N/A	N/A	N
52	0	M	0.85	59.7	0.75	N/A	N/A	N
53	0	M	1.25	48.3	0.5	N/A	N/A	N
56	3	M	14.9	99.7	5.35	N/A	N/A	Y
58	2	M	12.3	87.4	2.45	N/A	N/A	N
59	4	M	13.15	94.5	3.5	N/A	N/A	Y
62	0	F	N/A	N/A	N/A	0	N/A	N
67	4	F	N/A	N/A	N/A	2	N/A	Y
71	4	F	N/A	N/A	N/A	1	N/A	Y
72	1	M	13	96.8	2.5	N/A	N/A	Y
73	3	F	N/A	N/A	N/A	2	N/A	Y
77	2	F	N/A	N/A	N/A	3	N/A	Y
79	3	M	17.9	96.5	4.65	N/A	N/A	Y
81	2	M	16.95	100	3.85	N/A	N/A	Y
83	1	M	13.8	87.4	3.5	N/A	N/A	N

84	4	F	N/A	N/A	N/A	4	N/A	Y
86	2	F	N/A	N/A	N/A	3	N/A	Y
87	3	M	16.75	99.9	4.6	N/A	N/A	Y
88	8	M	12.25	99.5	4.65	N/A	N/A	Y
89	6	M	21.8	95.1	3.9	N/A	N/A	Y
91	1	M	2.05	70.8	1.3	N/A	N/A	N
94	2	M	11.6	100.4	4.15	N/A	N/A	Y
99	2	M	12.5	91	3.5	N/A	N/A	Y
201	8	M	17.95	94.1	5.25	N/A	N/A	Y
202	1	M	14.5	92	3.2	N/A	N/A	Y
204	2	M	14.75	101.5	3.9	N/A	N/A	Y
205	0	F	N/A	N/A	N/A	0	N/A	N
206	8	F	N/A	N/A	N/A	3	N/A	Y
207	4	F	N/A	N/A	N/A	0	N/A	N
209	5	F	N/A	N/A	N/A	2	N/A	Y
212	3	M	14.5	91	3.15	N/A	N/A	Y
213	3	F	N/A	N/A	N/A	5	N/A	Y
214	3	F	N/A	N/A	N/A	2	N/A	Y
215	1	M	13.25	95.3	3.25	N/A	N/A	Y
216	10	M	16.7	98.6	5.4	N/A	N/A	Y
217	1	M	12.7	95.6	2.7	N/A	N/A	Y
218	1	F	N/A	N/A	N/A	0	N/A	N
219	4	F	N/A	N/A	N/A	N/A	3	Y
220	10	M	14.65	100.3	5.6	N/A	N/A	Y
222	3	M	15.5	106.8	4.05	N/A	N/A	Y
223	1	M	1.15	63	0.9	N/A	N/A	N
224	1	M	16.35	101.2	4.1	N/A	N/A	Y
226	9	F	N/A	N/A	N/A	2	N/A	Y
227	4	F	N/A	N/A	N/A	2	N/A	Y
228	4	M	13.4	102.3	4.3	N/A	N/A	Y
230	1	F	N/A	N/A	N/A	0	N/A	N
232	2	F	N/A	N/A	N/A	N/A	3	Y
233	1	M	1.6	67	1	N/A	N/A	N
234	2	F	N/A	N/A	N/A	N/A	2	Y
236	2	F	N/A	N/A	N/A	0	N/A	N
242	1	F	N/A	N/A	N/A	0	N/A	N

250	3	M	15.2	96.9	2.95	N/A	N/A	Y
251	0	F	N/A	N/A	N/A	0	N/A	N
252	3	F	N/A	N/A	N/A	0	N/A	N
289	3	F	N/A	N/A	N/A	N/A	3	Y
292	2	M	17.8	97.3	4.05	N/A	N/A	Y
295	4	M	15.75	96.8	4.3	N/A	N/A	Y
300	2	M	11.2	87.4	3.15	N/A	N/A	N
8y	2	M	16.2	101.6	4.7	N/A	N/A	Y
9y	3	M	20.25	97.2	4.5	N/A	N/A	Y
10y/196	4	M	9.05	96.7	3.35	N/A	N/A	Y
12y	2	M	14.6	100.6	3.35	N/A	N/A	Y
14y	2	M	13.55	96.3	3.95	N/A	N/A	Y
15y	4	M	17.7	102.3	4.45	N/A	N/A	Y
17y	7	M	12.7	98.3	5.15	N/A	N/A	Y
18y	4	M	16.9	97.2	4.3	N/A	N/A	Y
20y	3	M	16.55	98.6	3.95	N/A	N/A	Y
21y	6	M	16.3	98.3	4.7	N/A	N/A	Y
26b	7	M	12.2	95.1	4	N/A	N/A	Y
27b/92	7	M	15.65	96	4.8	N/A	N/A	Y
36b	4	M	18.2	105	5.45	N/A	N/A	Y
37b	7	M	15.35	102.5	5.2	N/A	N/A	Y
38b	2	M	16.85	101.9	4.45	N/A	N/A	Y
39b	7	M	14.25	99.9	5.4	N/A	N/A	Y
40b	6	M	11.55	99.6	4.55	N/A	N/A	Y
41b	2	M	14.1	100	4.55	N/A	N/A	Y
42b/57	4	M	14.2	104.5	4.9	N/A	N/A	Y
44b/#93	9	F	N/A	N/A	N/A	N/A	3	Y
45b/22	6	M	6.7	100.6	4.35	N/A	N/A	Y
49b	4	F	N/A	N/A	N/A	3	N/A	Y
51y	9	M	14.7	100	4.25	N/A	N/A	Y
55y	3	F	N/A	N/A	N/A	1	N/A	Y
61y	4	M	9.95	104.3	4.95	N/A	N/A	Y
70y/37	5	M	16.9	96.4	4.5	N/A	N/A	Y
71y/#60	2	F	N/A	N/A	N/A	N/A	2	Y
73y	2	F	N/A	N/A	N/A	3	N/A	Y
74y	8	F	N/A	N/A	N/A	N/A	3	Y

78y/82	3	M	14.85	103.4	5.15	N/A	N/A	Y
81y/#64	3	F	N/A	N/A	N/A	0	N/A	N
85y	8	M	16.15	92	4.35	N/A	N/A	Y
86y/10	8	M	14.7	98.1	4.1	N/A	N/A	Y
88y	5	M	5.15	101.4	5.45	N/A	N/A	Y
89y	6	M	4.5	96.9	5	N/A	N/A	Y
90y/65	8	M	15.7	92.9	4.5	N/A	N/A	Y
92y	8	F	N/A	N/A	N/A	3	N/A	Y
94y/95	7	M	10.8	100.4	5.15	N/A	N/A	Y
96y	4	M	14.1	98.9	3.3	N/A	N/A	Y
100y/#16	5	F	N/A	N/A	N/A	3	N/A	Y
101y	10	M	17.5	103.6	5.7	N/A	N/A	Y
105y/28	3	M	13.6	81.9	2.95	N/A	N/A	N
106y/97	4	M	11.15	99	5.5	N/A	N/A	Y
110y	5	M	14.6	107	5.2	N/A	N/A	Y
122y/35	4	M	11.95	94.6	4.2	N/A	N/A	Y
126y	8	F	N/A	N/A	N/A	3	N/A	Y
UM1	1	M	2.6	83.3	2.1	N/A	N/A	N
UM3	3	F	N/A	N/A	N/A	3	N/A	Y
UM4	3	M	3.15	95.7	2.75	N/A	N/A	Y
UM5	3	M	5.05	113.1	4.85	N/A	N/A	Y
UM6	3	F	N/A	N/A	N/A	1	N/A	Y
UM7	4	F	N/A	N/A	N/A	4	N/A	Y
UM8	7	F	N/A	N/A	N/A	2	N/A	Y
