

ABSTRACT

STOCKING, JESSICA J. Effects of Predator Control and Habitat Type on American Oystercatcher (*Haematopus palliatus*) Reproductive Success. (Under the direction of Ted Simons).

Fifty-two percent of the nation's population now lives in coastal counties. Increasing human use of coastal areas limits habitat for beach-nesting species, such as shorebirds and sea turtles, rendering undeveloped and protected spaces disproportionately important for these species. American Oystercatchers (*Haematopus palliatus*) are at risk of reduced reproductive success across their range, due to a combination of human presence and predation. Long-term monitoring of sites in North Carolina shows that overall oystercatcher productivity is improving, but surveys indicate a shift in nesting habitat use away from traditional barrier beach sites. I address questions about improving reproductive success through alternate habitat availability and predator control. Dredge spoil islands provide attractive nesting habitat for coastal birds; they are free of mammalian predators and the levels of human activity found in outer beach environments. In contrast to these benefits, they often lack the foraging resources available to birds nesting in outer beach habitats. I compared reproductive success and chick growth on traditional barrier islands and novel islands formed within the last century. There was an apparent tradeoff between success in the incubation and chick-rearing stages. Chick growth on the two island types was equal. Barrier islands had lower hatching success but higher fledging success than spoil islands. The number of chicks fledged per pair was equal, however, suggesting an ideal despotic distribution. On the barrier beaches, recent management initiatives have focused on improving oystercatcher nesting success. I evaluated oystercatcher reproductive response to an experimental reduction of a raccoon population on a barrier island. I analyze success within a Before-After-Control-Impact framework and discuss the difficulties of inference in large-scale observational studies. Considerable annual variation in the birds' reproductive success can disguise longer term trends. However, the response to a one-time disturbance in the raccoon population may be short-lived. Based on this study, it seems that both barrier and dredge spoil sites are

critical to the persistence of oystercatchers in North Carolina, and management efforts should continue to address needs in both habitats.

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Effects of Predator Control and Habitat Type on American Oystercatcher
(*Haematopus palliatus*) Reproductive Success

by
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DEDICATION

To Jo - your inspiration improves everything I do.

BIOGRAPHY

I was born in Baraboo, Wisconsin, home of the Circus World Museum and the International Crane Foundation (could have gone either way, really). We moved early and often during my childhood. That became a continuing trend for me; many lifetimes have led me to this place where I now stand. The Biology portion began formally in Asheville, North Carolina when I re-enrolled in my undergraduate education. Zoology was my most challenging class, so I declared it my major. After graduation and a long drive to find Alaska, I returned to the International Crane Foundation as an intern in Aviculture. Handling birds bigger than I was hooked me. I moved on to field studies and never looked back: Mississippi, Washington, Oregon and California. In North Carolina I found work in the Simons Lab with oystercatchers. That developed into graduate study and has led to a marvelously rewarding three[-plus] years in Raleigh. Now I return to Alaska to build the next adventure.

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CHAPTER 1: STATE OF AMERICAN OYSTERCATCHER POPULATION AND RESEARCH IN NORTH CAROLINA

INTRODUCTION

Habitat loss and sea level rise are imperiling biodiversity across the globe. Coastal areas are extremely vulnerable, as the human population grows faster along coastlines than inland areas (Crossett et al. 2004) and sea level rise will inevitably impact low, coastal areas first and most severely. North America's beaches have changed dramatically over the last several decades, with fifty-two percent of the nation's population living in coastal counties (U.S. Census Bureau 2011). Increasing human use of coastal areas limits usable habitat for beach-nesting species, such as shorebirds and sea turtles. This renders undeveloped and protected spaces disproportionately important for these species.

There is ample evidence of human encroachment on Atlantic barrier island beaches. Cape Hatteras National Seashore received half a million visitors in 2010, more than eight times as many as in 1955 (www.nps.gov). Cape Lookout National Seashore has increased its visitation twenty-five-fold since 1976. The American Oystercatcher (*Haematopus palliatus*) is an important indicator of ecological conditions on Atlantic beaches. Because of its conspicuousness and site fidelity, the oystercatcher is an ideal study species for monitoring factors affecting the conservation and management of beach-nesting birds. American Oystercatchers are listed as a "species of special concern" in North Carolina (North Carolina Wildlife Resources Commission 2008) and as a high priority species in the US Shorebird Conservation Plan (Brown et al. 2001), in large part because of threats associated with development and increasing recreational use of coastal breeding and wintering sites.

A typical oystercatcher nest is comprised of three eggs laid on the ground in a shallow scrape, often in sand but also in shell, grass, wrack, and gravel (American Oystercatcher Working Group et al. 2012). This leaves the nest exposed to many types of terrestrial and avian predation, as well as destruction by beach vehicle traffic. Eggs are incubated approximately 27 days. Chicks are semi-precocial, able to leave the nest within hours of hatching but dependent on the adults for food until after fledging. Oystercatchers eat marine bivalves and other intertidal invertebrates (American Oystercatcher Working Group

2012 et al.). Their long, specialized bill allows them to break open clams and oysters and probe into the sand. Adults are highly territorial and will often return year after year to the same site with the same mate. Annual reproductive success exhibits great variability but is generally low (American Oystercatcher Working Group et al. 2012, Schulte 2012).

Proximate causes for oystercatcher nest and chick loss include predation, overwash, and abandonment. These are often caused by or confounded with the ultimate causes for population decline which include increased human development and stabilization of temperate beaches, disturbance/displacement from recreational use, and rising sea level. Indeed, often these threats multiply, exacerbating one another (McGowan and Simons 2006, St. Clair et al. 2010).

Oystercatcher populations declined late last century in the mid-Atlantic states, despite rising numbers and an expansion of the breeding range to the north (Mawhinney and Benedict 1999; Nol et al. 2000; Davis et al. 2001). These overall declines triggered a large-scale, multi-state research effort to understand the bird's ecology and conservation needs (Schulte et al. 2007).

OYSTERCATCHER RESEARCH IN NORTH CAROLINA

Studies of oystercatcher reproductive success in North Carolina began in 1995 and found that birds nesting on the barrier beaches on South Core Banks, Cape Lookout National Seashore had extremely low reproductive success (Novick 1996, Appendix A). Davis et al. (2001) documented causes of low productivity, such as predation, overwash and weather events. An estimate of 1624 pairs along the Atlantic coast of the U.S. sparked the oystercatcher's designation as a Species of High Concern (Brown et al. 2001). Monitoring efforts in North Carolina expanded to include all of the islands of Cape Lookout and Cape Hatteras National Seashores, as well as additional sites for specific research questions. McGowan (2004) further examined spatial and temporal factors affecting nest success, citing predation as the primary cause of known nest failure. McGowan and Simons (2006) found an inverse relationship between the number of visits an oystercatcher made to the nest and the nest survival rate, suggesting that the more often nests were disturbed the more likely they were to be found by predators. Schulte (2012) illuminated causes of chick loss and modeled

demographic and hurricane effects on oystercatcher production. Audubon North Carolina initiated nest monitoring on islands in the mouth of the Cape Fear River and found higher nest success than on barrier islands but lower chick survival (McGowan et al. 2005b).

Study Sites

We currently monitor American Oystercatcher productivity at several locations along the North Carolina coast (Appendix A) in cooperation with staff from the National Park Service, the North Carolina Wildlife Resources Commission, and Audubon North Carolina. Habitat consists of a combination of natural and man-made islands: some provide public access and human habitation, while others are closed to public use.

The Cape Hatteras National Seashore, at the north end of the study area, is approximately 107 km long and consists of three barrier islands: Bodie, Hatteras, and Ocracoke (north to south). These islands are accessible by a bridge on the north end and ferry transport from two southern sites and experience heavy recreational use. Mammalian predators and human disturbance/destruction have historically been the dominant threats to oystercatcher success on these islands.

Oregon Inlet, between Bodie Island on the north and Hatteras Island on the south, supports nesting oystercatchers on dredge spoil islands (created by strategic deposition of dredged material) and two naturally formed marsh islands. One of the natural islands is owned and monitored by the National Park Service, and the NCWRC manages the remaining islands; public access is prohibited during the breeding season. These islands are closed to recreational use during the nesting period. Mammalian predators are not established on these islands, although nesting colonies of predatory gulls are present.

Ocracoke Inlet, between Ocracoke Island on the north and North Core Banks on the south, consists primarily of shell islands. Audubon N.C. monitors and manages these islands. Other avian species such as gulls (*Larus spp.*), terns (*Sterna spp.*) and Brown Pelican (*Pelecanus occidentalis*) also nest on the shell islands, and there are no resident mammals. Oystercatcher nests on these low islands are extremely vulnerable to high tides and storm overwash. Access to these islands is prohibited during the nesting season.

Cape Lookout National Seashore extends from Ocracoke Inlet to Beaufort Inlet and consists of three islands. North Core Banks and South Core Banks have a general northeast-southwest orientation and are 37 and 40 km long, respectively. Shackleford Island is 15 km long, lies to the southwest of these islands, and is oriented east-west. The islands are accessible only by boat, and commercial ferry services regularly run tourists and vehicles to the islands. Primary threats to oystercatcher nests and chicks include raccoon (*Procyon lotor*), storms/flooding, human disturbance, feral cats, and ghost crabs (Altman 2010, Schulte 2012).

In 2002 Audubon North Carolina began monitoring nests in the vicinity of the Cape Fear River, and that area has received increased effort during this study. Lea-Hutaff is an Atlantic barrier island similar in composition to the islands in the National Seashores, but it is privately owned and offers limited public access. In the Cape Fear River, Ferry Slip and South Pelican are dredge-spoil islands; Battery and Shellbed are natural islands.

Monitoring

We begin surveys in mid-March as oystercatchers establish breeding territories. Nest searching is conducted on foot and from vehicles (trucks, ATVs, boats). Pairs that appear to be active and defending a territory are monitored closely to locate nests and identify dates of nest initiation. Nests are then marked with a natural artifact for efficient relocation. Nests on the barrier islands are checked from a distance every 1-2 days to determine activity and approached only to document hatching or causes of nest loss. Sites that require boat access are checked as frequently as possible, usually every 1-3 days unless access is precluded by low tides or storms. Nests are visited daily just prior to hatching to determine exact hatching dates.

It is often possible to determine whether a pair of adult birds had chicks by observing adult behavior, even in the absence of visual verification. In most cases chicks are located by observing adults from a distance using a spotting scope. We monitor chicks every 1-5 days after hatching until fledging, or until all chicks die or disappear. On the rare occasion that a chick is found dead, we attempt to determine the cause of death, although it is often not possible to determine the cause or exact timing of chick mortality.

Nest survival - the probability that a nest survives the 27-day incubation period - is estimated using the intercept model in the nest survival module of Program MARK 6.2 (White and Burnham 1999). Nests are considered successful if at least one egg hatched. In the absence of evidence indicating the exact date of hatch or failure of a nest, we use the midpoint between two consecutive checks (Johnson 1979). Nests that are found after hatching are not included in the analysis, and nests that disappear around the anticipated hatch date but in which no chicks were seen are considered failed. Productivity is calculated as the number of chicks that successfully fledge (survive to sustained flight or 35 days after hatch) per breeding pair.

In 2010, the N. C. Wildlife Resources Commission estimated the North Carolina oystercatcher population at 369 breeding pairs (American Oystercatcher Working Group et al. 2012). We currently monitor approximately one third of known nesting pairs (Appendix A). Those nests have an estimated mean nest survival of 0.331 (SE 0.010) but exhibit high levels of spatial and temporal variability. Generally, nest survival and productivity are improving in the study area (Appendix A), in part due to increased awareness and protection.

Mark-resight

Mark-resight studies are important for understanding demographics, movements and distributions. Banding oystercatchers in North Carolina began in 2000 (McGowan et al. 2005a). In 2004, a standardized marking system was adopted by research and management entities throughout the oystercatcher's range. This allows state-level identification upon initial observation of a band and encourages communication about observations. Audubon North Carolina has recently developed an open access online database that will archive auxiliary banding data for oystercatchers across their range. This will be available through the American Oystercatcher Working Group website (www.amoywg.org) and searchable by partners.

DISCUSSION

American Oystercatchers are at risk of reduced reproductive success across their range, due to a combination of human presence and predation (Davis 1999, Brown et al. 2005). In North Carolina sites with long-term monitoring, overall oystercatcher productivity

is improving (Appendix A). This is likely due to increased protection at several key sites. Many years of data are required for drawing conclusions, as considerable annual variation can disguise longer term trends. The oystercatcher population along the Atlantic and Gulf coasts of the United States was estimated to be 11,000 individuals in winter of 2002-3 (Brown et al. 2005). Currently there are plans for an updated estimate based on winter (2012-2013) and nesting season (2014) surveys and organized resight efforts of marked birds throughout the Atlantic range. The new estimate will help researchers and conservation practitioners in the American Oystercatcher Working Group (www.amoywg.org) understand whether and where oystercatcher efforts have been successful or need reevaluation.

This research expands upon 15 years of studies describing trends in oystercatcher nest success in managed areas of North Carolina. Due to the increasing human presence in coastal areas, protected areas are critical to the success of obligate coastal species such as the American Oystercatcher. In this study we address questions about improving reproductive success through alternate habitat availability and predator control. Chapter Two compares reproductive success and chick growth on traditional barrier nest sites and novel spoil island sites formed by dredging within the last century. Chapter Three evaluates oystercatcher reproductive response to an experimental reduction of a raccoon population on a barrier island.

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CHAPTER 2: COMPARISON OF AMERICAN OYSTERCATCHER REPRODUCTIVE SUCCESS IN TRADITIONAL AND NOVEL HABITATS

ABSTRACT

Human activity, in the forms of development and recreation, is changing the availability and integrity of coastal habitats for breeding shorebirds. Historically, American Oystercatchers (*Haematopus palliatus*) nested predominantly on outer beaches and between primary and secondary dunes. In recent decades, oystercatchers and other beach-nesting birds are breeding in increasing numbers on alternate habitats such as man-made dredge spoil islands. Created during the 20th century as a byproduct of dredging to maintain navigation channels, spoil islands provide attractive nesting habitat for coastal birds because they are often free of mammalian predators and the levels of human activity now characteristic of outer beach environments. In contrast to these benefits, dredge spoil islands often lack the foraging resources available to birds nesting on traditional outer beach habitats. This study aimed to better understand the life history trade-offs American Oystercatchers face when they nest in novel dredge spoil habitats in North Carolina. We compared chick growth, nest survival, and chick survival of pairs breeding on barrier islands and dredge spoil islands to evaluate differences in the breeding biology of oystercatchers using these traditional and non-traditional nesting habitats. Growth rates of individual chicks in the two habitat types were not different. Barrier island pairs had reduced daily and cumulative rates of nest survival compared to pairs nesting on dredge spoil sites, although the two populations had the same total productivity (chicks fledged per breeding pair). Successful nests on dredge spoil islands only produced one chick for every two fledged from a barrier island nest. The distribution of oystercatchers in North Carolina appears to reflect an ideal despotic distribution. More information on the timing and causes of nest and chick loss on dredge spoil islands would greatly benefit the management of these increasingly important nesting habitats.

INTRODUCTION

Human populations in coastal zones are expanding at a rate disproportionate to inland areas (Crossett et al. 2004). This expansion pinches ecosystems, impacting both availability and quality of the space that remains. American Oystercatchers are closely tied to coastal environments throughout the year (American Oystercatcher Working Group et al. 2012). Historically, the species nested on outer beaches along the Atlantic and Gulf coasts (Bent 1929, American Oystercatcher Working Group et al. 2012). With the growth of human populations in coastal environments, oystercatchers, along with many other beach nesting species are increasing their use of nesting habitats away from the outer beaches (Post and Raynor 1964, Frohling 1965, Fisk 1978, McGowan 1969, Shields and Parnell 1990, Toland 1992, Traut et al. 2006). Recent surveys have indicated that oystercatchers are using alternate habitats more than had previously been acknowledged (Wilke et al. 2007). Although these non-traditional nesting sites could provide valuable alternative nesting habitat as outer beach sites become less suitable for oystercatchers, the quality of these habitats for nesting oystercatchers is largely unknown. Ens et al. (1992) have shown that European Oystercatchers with territories adjacent to food resources have higher nesting success than pairs nesting farther from food resources. Success in non-traditional nesting habitats along the Atlantic coast appears to be geographically and nest-stage dependent. In New Jersey, hatching success is an order of magnitude higher on interior marsh islands than on the outer barrier islands (Virzi 2008). In North Carolina, McGowan et al. (2005) found that hatching success was higher but chick survival to fledging was lower on interior islands, whereas pairs on Nantucket, Massachusetts had extremely high nest success with low chick survival (Murphy 2010).

Oystercatchers in North Carolina nest in a variety of habitats. Barrier islands stretch the entire length of the coast and their outer beaches provide the core habitat for the state's nesting population. Other naturally formed low islands such as sandbars and shell rake islands also support high densities of nesting oystercatchers, although they are subject to frequent overwash due to fluctuations in tides and storms (personal observation). Oystercatchers are nesting with increasing frequency in novel environments such as dredge

spoil islands created as a byproduct of dredging to maintain navigation channels, and even on rooftops in coastal towns (Cameron 2008).

Barrier islands on the Atlantic coast of the United States are characterized by sandy beaches continually altered and shaped by the movement of wind and current. Moving inland from the open beach, there is a line of primary dunes periodically interrupted by sand flats created where the ocean has breached the dunes during high tides or storm events. Behind the dunes, shrub thickets gradually transition to salt marshes on the back sides of the islands. Oystercatchers predominantly nest between the primary dunes and the high tide line, but they will also nest in overwash flats, interdunal areas, and occasionally in sound-side marshes. In 2010, 42 percent of the territorial pairs located during the North Carolina Wildlife Resources Commission statewide nesting survey were found on barrier islands (NCWRC unpubl. data).

Dredge spoil islands were originally created as byproducts dredging operations conducted to maintain navigation channels. Only later was their importance as habitat for nesting coastal birds recognized (Parnell and Soots 1974). American Oystercatchers were first recorded nesting on dredge spoil islands in North Carolina in 1986 (Parnell et al. 1986). Dredge spoil islands supported 30 percent of oystercatchers located in North Carolina during the 2010 nesting season (NCWRC unpubl. data).

Compared to traditional habitats, dredge spoil islands offer clear ecological tradeoffs for nesting oystercatchers. These islands are attractive to oystercatchers because unlike barrier island sites they generally do not support established mammalian predator populations. Predation by mesopredators like raccoons is the primary source of nest failure in North Carolina and elsewhere (Davis et al. 2001, Sabine et al. 2006, Virzi 2008, Schulte 2012). On the other hand, dredge spoil islands generally do not provide suitable foraging habitat in close proximity to oystercatcher nest sites. These islands are built efficiently, such that they release little sand back into adjacent navigation channels. This has two relevant results for oystercatchers: the channels only infrequently require dredging and thus infrequent dumping of new material, and the sides of the islands can be steeply sloped (Erwin et al. 2003). Neither of these characteristics benefits oystercatchers. Revegetation often occurs between deposition events, encouraging gulls and other potential avian predators to

nest on the islands, and the steep outer slopes prevent establishment of the oystercatchers' invertebrate prey. Therefore oystercatchers nesting on dredge spoil islands must commute to foraging habitats often a kilometer or more from their nesting territories.

Heppleston (1972) found that oystercatchers with distant foraging territories were at a disadvantage to those pairs that had adjacent nesting and foraging territories. Non-adjacent nesting and foraging territories require the extended absence of one adult during foraging trips. Male and female oystercatchers in a nesting pair take turns incubating their eggs. The two birds often cooperate in displays to defend their territory, nest, or chicks. In the absence of one bird during prolonged foraging trips, the other member of the pair is left to defend the nesting territory alone. In our study area, birds nesting on dredge spoil islands were regularly observed flying from their territory toward neighboring marsh habitats and returning 30 or more minutes later with food for their chicks (S. Thompson, pers. obs.).

Oystercatcher chicks depend on their parents for food for at least 60 days after hatching (Palmer 1967), and some chicks are fed for up to several months after fledging (American Oystercatcher Working Group et al. 2012). If not provided with adequate nutrition, chicks will starve to death or, in more moderate cases, grow more slowly (Ens et al. 1992, Kersten and Brenninkmeijer 1994). Oystercatcher chick growth rates therefore provide an index of both parental and territory quality (e.g. Hazlitt et al. 2002; Ens et al. 1992). Previous studies have found that rates of oystercatcher chick provisioning varied with distance to food source (Hartwick 1976; Nol 1989). Foraging outside of an established territory may increase reproductive costs, especially for pairs raising multiple chicks (Ens et al. 1992). Chicks with larger growth rates have a higher chance of fledging successfully (Tjorve and Underhill 2009).

Our study compares oystercatcher breeding biology on traditional barrier island outer beach habitats with that on evolutionarily novel dredge spoil islands in an attempt to clarify the role that dredge spoil islands play in sustaining breeding populations. We hypothesized that reduced nest attendance and longer foraging trips would reduce productivity of pairs nesting on dredge spoil islands relative to their barrier island counterparts. We also hypothesized that chicks raised in territories farther from feeding areas (dredge spoil islands)

would grow at a slower rate than chicks raised in territories with closer feeding areas (barrier islands).

METHODS

Study sites

We studied the breeding biology of oystercatchers on barrier islands in Cape Hatteras National Seashore, Cape Lookout National Seashore, and islands northeast of the mouth of the Cape Fear River (Figure 2.1). Cape Hatteras National Seashore extends over 112 km from north to south and consists of three barrier islands (Figure 2.1c). The Seashore is accessible by a bridge on the north end and by ferry on the southern end. These islands are partially developed and get extensive recreational use, particularly during the summer. Cape Lookout National Seashore is comprised of three main islands and is 90 km in length (Figure 2.1c). Off-road vehicles are permitted on both National Seashores. Lea-Hutaff Island, Masonboro Island, and Wrightsville Beach at Mason Inlet are barrier islands in the vicinity of the Cape Fear River (Figure 2.1d). These islands receive varying degrees of human recreational activity.

Dredge spoil island study sites were located at two main locations. Oregon Inlet is located between Bodie Island on the north and Pea and Hatteras Islands on the south (Figure 2.1a). These islands are owned by the N. C. Wildlife Resources Commission and monitored by the Commission and Audubon N. C. Nine Oregon Inlet islands supported nesting oystercatcher pairs during our study. Ferry Slip and Pelican Island are dredge spoil islands in the Cape Fear River, on the edge of New Hanover and Brunswick Counties (Figure 2.1d).

Nest and chick survival

Daily survival rates (DSR) were calculated according to Mayfield (1961, 1975), and cumulative survival rates (CSR) were calculated by raising the DSR to the 27th power based on a 27-day incubation period of the oystercatcher (American Oystercatcher Working Group et al. 2012). We tested pairwise comparisons between DSRs of the two nesting habitat types using CONTRAST (Hines and Sauer 1989). The 95% confidence interval for each CSR was calculated by raising the upper and lower bounds for the DSR to the 27th power (*sensu*

Johnson 1979). We concluded that two CSR values were significantly different from one another if their confidence intervals did not overlap. We calculated overall breeding success (productivity) as chicks fledged per breeding pair by dividing the number of chicks that survived to fledging by the number of breeding pairs for each year in each location. Nests with more than one week between visits at the time of nest failure and nests with undetermined fates were omitted from the analysis. Nests were considered to have failed if chicks were never observed.

When a nest failed, we attempted to determine the cause of failure by examining the nest cup and area surrounding the nest. We looked for evidence of tracks, eggshell fragments and spilled yolk. Due to the difficulty of distinguishing predator tracks in the sand, shell, and grass substrates used by oystercatchers in our study area, predation events were combined for this analysis.

Assumptions in our model of nest survival were; accurate determination of nest fate, a constant daily nest survival probability, and that our monitoring activities did not affect survival probability. Constant daily survival across the nesting period is likely unrealistic (Dinsmore et al. 2002, Murphy 2010, Schulte 2012), but a violation of this assumption should not impact the inference of this comparison.

Growth rates

In 2011, we measured chick growth and development at barrier island sites in Cape Lookout National Seashore, and dredge islands (South Pelican and Ferry Slip) in the Cape Fear River. We captured chicks every 4 - 7 days for measurements. We measured culmen length along the central line of exposed culmen from the end of the feathers to the tip of the mandible (Baldwin et al. 1931) to the nearest tenth of a millimeter with calipers. We measured the tarsus with calipers by bending both tarsal and metatarsal joints to 90-degree angles and measuring from the middle point of the joint between the tibia and metatarsus to the end of the first joint on the middle toe. This is not a true measurement of the tarsus bone, but we determined it to be more repeatable (Richner 1989). We measured the wing chord to the nearest millimeter from the wrist of the unflattened, closed wing to the tip of the longest primary (Baldwin et al. 1931) using calipers for very small chicks and a metal wing ruler as

the primaries started to emerge. Oystercatchers fledge around 35 days after hatching (American Oystercatcher Working Group et al. 2012); we truncated our analyses at 40 days of age due to limited data from older birds. Due to the difficulty of marking chicks for individual identification, we treated all measurements as independent observations and did not account for individual or brood-level effects.

Growth parameters were estimated by Gompertz curves (Ricklefs 1968, Nol 1989, Tjorve and Underhill 2009), allowing parameterization of the upper asymptote [a] and descriptors for the x-axis displacement [b] and rate [c]. Growth curves were fit to the mixed longitudinal data in a Bayesian framework using the R2OpenBUGS package to call OpenBUGS statistical software (Lunn et al. 2000) through R (Sturtz et al. 2005). Simulations ran for 50,000 iterations with a burn-in period of 5,000 iterations. We checked convergence visually and using the Gelman-Rubin diagnostic; values smaller than 1.1 were considered converged. We discarded values generated prior to convergence as burn-in, using the remaining values to generate parameter estimates.

RESULTS

Nest survival

We calculated nest survival estimates for 423 barrier island and 143 dredge spoil island nests between 2009 and 2011. Apparent hatching success was 0.3112 and 0.5874, respectively. Barrier island nests in our study had a smaller probability of daily and cumulative survival than dredge spoil island nests (n=566; Table 2.2).

The probability that a nest would survive from one day to the next (DSR) was lower for nests on barrier islands (0.9556 ± 0.0026) than for nests on dredge spoil islands (0.9719 ± 0.0035 ; Table 2.2). The CSR, or probability that a nest will survive the entire incubation period, for nests on barrier islands was also lower than for nests on dredge spoil islands (Table 2.2). This pattern varies geographically: the northern sites (Oregon Inlet and Cape Hatteras) showed no difference in either DSR or CSR, whereas the southern sites (Cape Fear River area) showed lower nest survival rates for barrier islands than for dredge spoil islands (Figures 2.2 and 2.3).

Thirty-eight percent of barrier island nest losses were attributed to raccoons, while abandonment and overwash caused 0.07% and 0.05% of nest failures, respectively (Table 2.6). Dredge spoil island nest losses were attributed to overwash (28%) predation (0.08%) and abandonment (0.04%). Causes of nest loss for a majority of both barrier island (50.8%) and dredge spoil (60.0%) island nests were not determined.

Productivity

We measured productivity for 580 nests that fledged 205 chicks during the study period (Table 2.3). Barrier islands fledged 0.5185 (SE 0.060) chicks per pair, and dredge spoil islands fledged 0.5050 (SE 0.065) chicks per pair. Productivity was not different between barrier and dredge spoil island nests ($\chi^2=0.0233$, $df=1$, $p=0.8787$).

Growth rates

We obtained growth measurements from 227 barrier island and 80 dredge spoil island chicks (Figures 2.4 and 2.5). Average culmen growth increased from 14% of adult culmen length at 2 days to 72% at 37 days; 20% of adult tarsus length at hatching to 94% at 35 days; 7% of adult wing chord length at hatching to 83% at 50 days; and 3% of adult body mass at hatching to 59% on day 38 (Table 2.4). We did not find support for differences in chick growth rates on dredge spoil and barrier island sites for any parameter-measurement combination (Table 2.5).

DISCUSSION

Differences in daily and cumulative survival estimates indicate that oystercatcher nest survival rates are lower at barrier island sites than at dredge spoil island sites in North Carolina. This difference is driven by the southern Cape Fear River area and is not detected in an equivalent comparison of the northern study sites. The latter comparison, however, suffers from a limited number of dredge island nests ($n=26$, Appendix A). Overall productivity (chicks fledged per breeding pair) was the same on dredge spoil and barrier island sites, although successful nests on barrier islands fledged nearly twice as many chicks as those on dredge spoil islands. Therefore, although a higher proportion of pairs is productive (i.e. fledged at least one chick) on dredge spoil islands, these pairs were less

successful on average than pairs that fledged chicks on the barrier islands. We found no difference between the growth rates of chicks in the two island types.

Nest and chick success

Although birds nesting on dredge spoil islands face lower predation pressure these benefits appear to be offset by costs associated with chick rearing (McGowan et al. 2005). Mammalian predation is a major cause of nest loss for oystercatchers and it is the dominant source on barrier beaches in North Carolina (Davis et al. 2001, Schulte 2012). Causes of nest loss on dredge spoil islands are difficult to determine due to less frequent monitoring combined and less impressionable substrates (such as wrack or grass). Of known causes, overwash accounted for the most nest losses on dredge spoil islands in our study, while predation was the major cause of loss at barrier island sites. Other sources of loss included abandonment and direct human destruction.

Chicks on North Carolina's barrier islands are lost to predators, environmental factors and vehicles (Schulte 2012). Despite the absence of mammalian predators and vehicle traffic on dredge spoil sites, several potential causes of chick loss are possible including starvation, attacks from neighboring oystercatcher pairs and avian predators. Unfortunately, we have little information about the causes of chick loss on the dredge spoil islands. Nearly all of the adults are unbanded, and broods are largely indistinguishable from one another in dense nesting situations.

Growth rates

Dredge spoil islands in our study area provide no immediate foraging habitat for nesting oystercatchers that therefore must commute to foraging sites on adjacent marshes and barrier beaches. We hypothesized that this added energy expenditure would manifest itself in reduced chick growth rates on dredge spoil islands. However, chicks on dredge spoil islands did not grow more slowly than those from barrier island territories. This suggests three possible explanations: 1) chicks are starving to death but not being found; 2) chicks weakened by lack of food are more vulnerable to predators; 3) time spent commuting to foraging areas reduces the time available for adults to attend their otherwise healthy chicks

increasing chick mortality caused by avian predators or exposure; and 4) adults are absorbing the cost of decreased nutrition through reduced fitness. It appears that chicks from dredge spoil islands are getting adequate food for growth rates equal to those of chicks on barrier islands.

Equal growth rates and large differences in the fledging rates of successful pairs suggests breeding pairs on dredge spoil islands compensate for the energy lost commuting to feeding territories raising a single chick for every two fledged from successful broods on barrier islands.

Habitat use and distribution

Equal annual productivity in the two types of nesting habitat in North Carolina suggests a despotic free distribution of breeding birds (Fretwell and Lucas 1969, Pulliam and Danielson 1991, Ens et al. 1995). This theory predicts that individuals of a territorial species will fill the best available breeding sites first, resulting in an equilibrium distribution with respect to nesting success. This idea is intuitive: an animal that has the ability to sample multiple available locations will presumably choose the site with the greatest perceived reproductive potential. Our results suggest that habitat quality varies from the northern to the southern portions of our study area. Birds in the northern portion of our study area may judge the barrier islands of Cape Hatteras to be more suitable than the dredge islands of Oregon Inlet. This could be a function of recent increased protective measures by the National Park Service at Cape Hatteras (National Park Service 2012). The decline in the number of pairs at Oregon Inlet and the recent increase in productivity on the outer beaches of Cape Hatteras National Seashore are consistent with this hypothesis (Appendix A). The dredge spoil islands in the Cape Fear River support relatively high and increasing densities of nesting oystercatchers. The dredge islands in Oregon Inlet have approximately one breeding pair per island, despite available open sand habitat. South Pelican and Ferry Slip, in the southern end of our study area, have between 15 and 25 pairs each (Appendix A). A formal test of the ideal free distribution hypothesis would require quantification of both nesting habitat and foraging habitat, which is beyond the scope of this study.

Site dominance is a prerequisite for territory acquisition in oystercatchers, and dominance is often site-specific (Heg et al. 2000). Therefore we would expect that oystercatchers face a clear tradeoff between sampling more prospective breeding sites and being more likely to obtain dominance at a particular site. Waiting for a territory to become available on the barrier islands of Cape Hatteras might be more beneficial to birds than settling on a lower-quality territory on the nearby Oregon Inlet dredge spoil islands. Alternatively, breeding in lower-quality territory might serve as a “stepping stone” for young birds while they wait or fight for a higher-quality territory, allowing them to get some experience in the meantime (van de Pol et al. 2007). In a long-term study of European Oystercatchers (*Haematopus ostralegus*), Ens et al. (1995) found that non-breeders born into territories of low quality (distant food source) rarely settled in high-quality territories (adjacent food source). These trends could be examined in North Carolina by continued monitoring and analysis of resight data.

Long-term mark-resight efforts are underway throughout the Atlantic range that will allow us to test specific predictions about habitat quality (American Oystercatcher Working Group et al. 2012, Schulte 2012). If barrier island territories are superior, we might expect to see birds of dredge spoil island origin searching/competing for barrier island territories early in the season or alternately, prior to first breeding attempts, and only settling in a dredge spoil island territory due to an inability to acquire one in the preferred habitat (Harris 1970, Ens et al. 1995). There is no evidence at present to suggest that they are searching for beach-side nest sites early in the season and then settling for the dredge islands. There is, however, anecdotal evidence of quick replacement in the cases when a member of a breeding barrier island pair is lost, and one case in which a replacement mate “adopted” a brood that belonged to the deceased bird (T. Borneman, pers. obs.). The degree to which young birds are attempting to move into barrier island habitat is not systematically addressed in this study but is an avenue for further exploration.

Flexibility in nest site choice can be critical to a species’ success in changing habitat structure and quality, enabling better responses to changing flood, predation or foraging conditions. Coastal nesting species, particularly those of shifting barrier islands, would be at

an evolutionary advantage if able to respond to dramatic changes in habitats by changing sites. While this strategy might be appropriate at the species level, this does not appear to be the case with individual oystercatchers. Once settled in a territory, even if an inferior one, they usually return in consecutive years (American Oystercatcher Working Group et al. 2012, Schulte 2012). This has been shown to be more true for birds that were hatched and raised in a high quality habitat than birds of low quality natal origin, the latter being more likely to “upgrade” (Heg et al. 2000; my term in quotations).

American Oystercatchers commonly live to at least ten years of age, and it is likely that birds of 20 years or more are not uncommon (American Oystercatcher Working Group et al. 2012). This study is extremely short relative to the life expectancy of an oystercatcher, and we have used annual productivity as a measure of reproductive success. Projected reproductive success over the course of a bird’s lifetime might serve as a more appropriate population-level basis for comparing two island types (Ens et al. 1995).

We found American Oystercatcher annual reproductive success did not differ in traditional barrier and non-traditional dredge spoil habitats. Growth rates and chicks fledged per pair were equal in the two site types. Daily and cumulative nest survival estimates were equal when all nests were combined, but we did find a difference in the southern island types when examined separately. Based on our study, it seems that both barrier and dredge spoil sites are critical to the persistence of oystercatchers in North Carolina, and management efforts should continue to address needs in both habitats.

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FIGURES

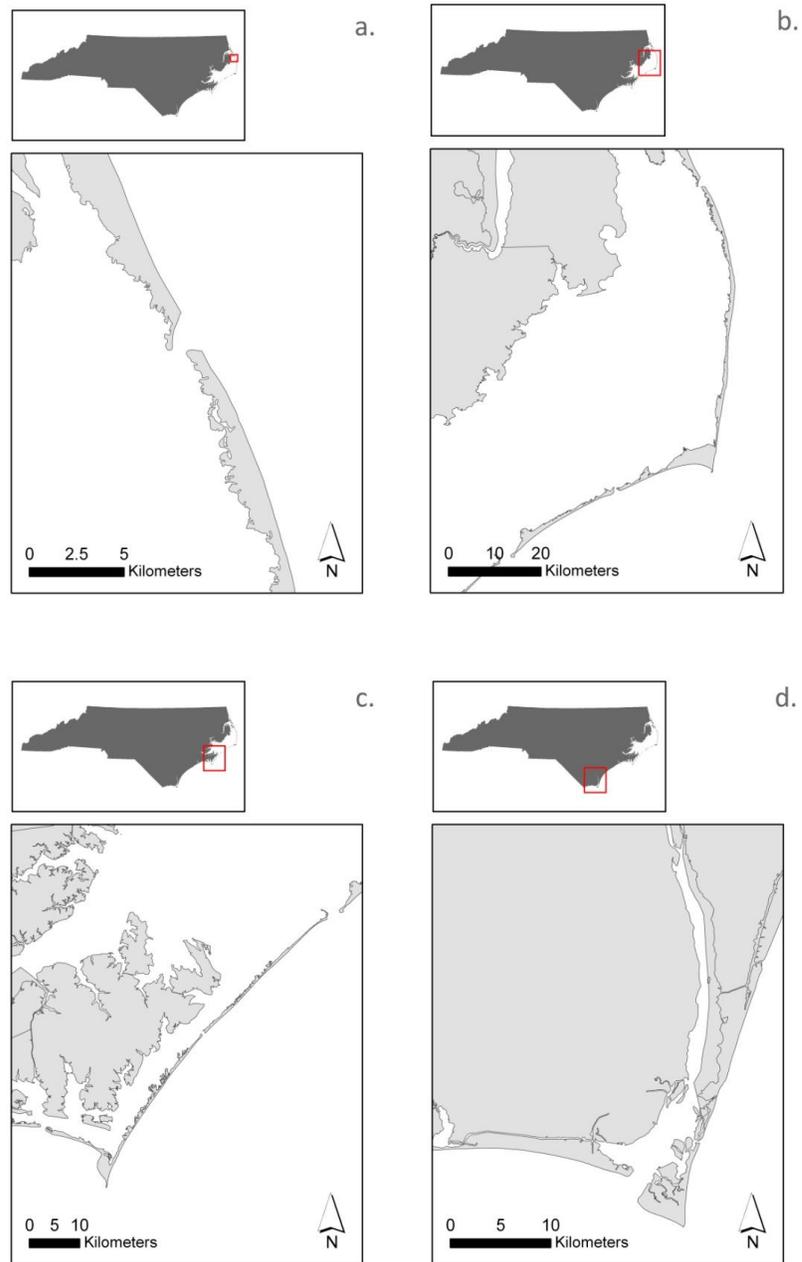


Figure 2.1. Regions of American Oystercatcher monitoring in North Carolina: a) Oregon Inlet, b) Cape Hatteras National Seashore, c) Cape Lookout National Seashore, d) Cape Fear River area.

Table 2.1. Nest-stage success metrics for oystercatcher nests in North Carolina by island type, 2009-2011.

	nests	pairs	eggs	nests hatched	eggs hatched	% eggs hatched	naïve nest success
barrier	437	297	1094	136	279	0.2550	0.3112
dredge	143	101	358	84	144	0.4022	0.5874
TOTAL	580	398	1452	220	423	0.2913	0.3793

Table 2.2. Estimated daily survival rates (DSR) for oystercatchers on two island types in North Carolina, 2009 - 2011. Barrier island nests had lower daily survival than nests on dredge spoil islands in our study ($\chi^2=13.9763$, $df=1$, $p=0.0002$). Standard errors of rates in parentheses.

	n	Exposure days	Nests failed	DSR
barrier	423	6507	289	0.9556 (0.0026)
dredge	143	2208	62	0.9719 (0.0035)

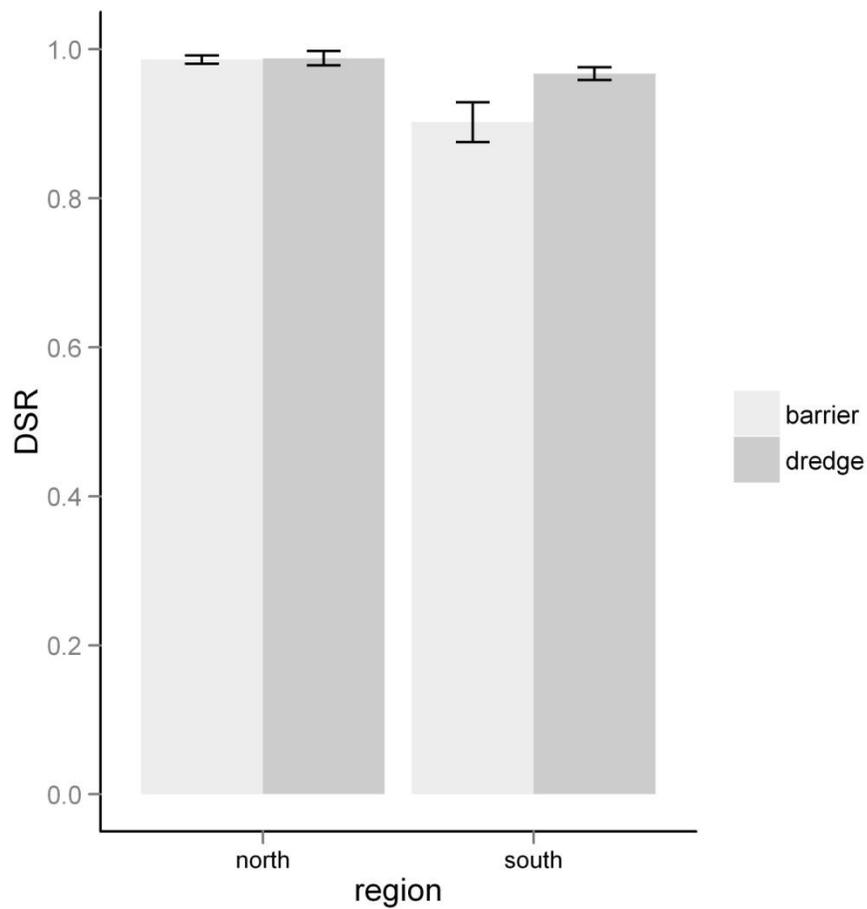


Figure 2.2. Estimated daily survival rates (DSR) for oystercatcher nests in two regions of North Carolina from 2009 to 2011, emphasizing the geographical pairing of sites. Dredge spoil island nests in the Cape Fear River (south) had higher survival than the nearest barrier island nests ($\chi^2=21.0123$, $df=1$, $p<0.0001$); Oregon Inlet dredge spoil island nests (north) showed no difference in survival from barrier island nests at Cape Hatteras National Seashore ($\chi^2=0.1192$, $df=1$, $p=0.7299$). Error bars indicate upper and lower bounds on 95% confidence intervals.

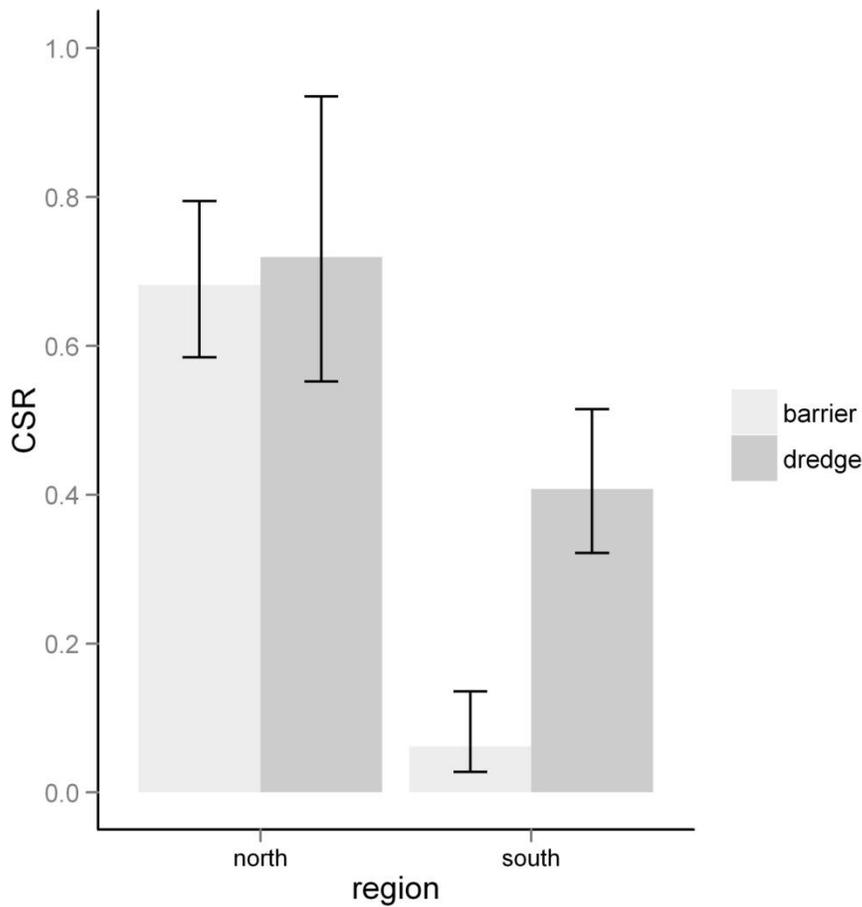


Figure 2.3. Cumulative survival rates (CSR) for oystercatcher nests in two regions of North Carolina, emphasizing the geographic pairing of sites. Survival was significantly lower on the Cape Fear (south) barrier island nests than on any of the other sites; in the north, nests on Oregon Inlet dredge spoil islands and Cape Hatteras National Seashore barrier islands did not differ in nest survival. Error bars indicate lower and upper bounds on 95% confidence intervals.

Table 2.3. Brood-stage success metrics for oystercatchers on two island types in North Carolina, 2009-2011. Productivity (chicks fledged per pair) is equal, but successful barrier island nests fledged nearly twice as many chicks as successful dredge spoil island nests. Standard errors given in parentheses.

	nests	pairs	fledged	fledged/ pair	fledged/ successful nest
barrier	437	297	154	0.5185 (0.060)	1.1324 (0.076)
dredge	143	101	51	0.5050 (0.065)	0.6071 (0.059)

Table 2.4. Measurements for adult oystercatchers captured in North Carolina, 2004-2010 and minimum and maximum chick measurements in 2011, expressed as percent of adult mean.

		n	mean (SE)
adult	culmen	37	87.94 (0.79)
	tarsus	25	65.98 (1.26)
	wing chord	24	258.21 (1.15)
	mass	20	571.70 (19.75)
		min %	max %
chick	culmen	14	72
	tarsus	20	94
	wing chord	7	83
	mass	3	59

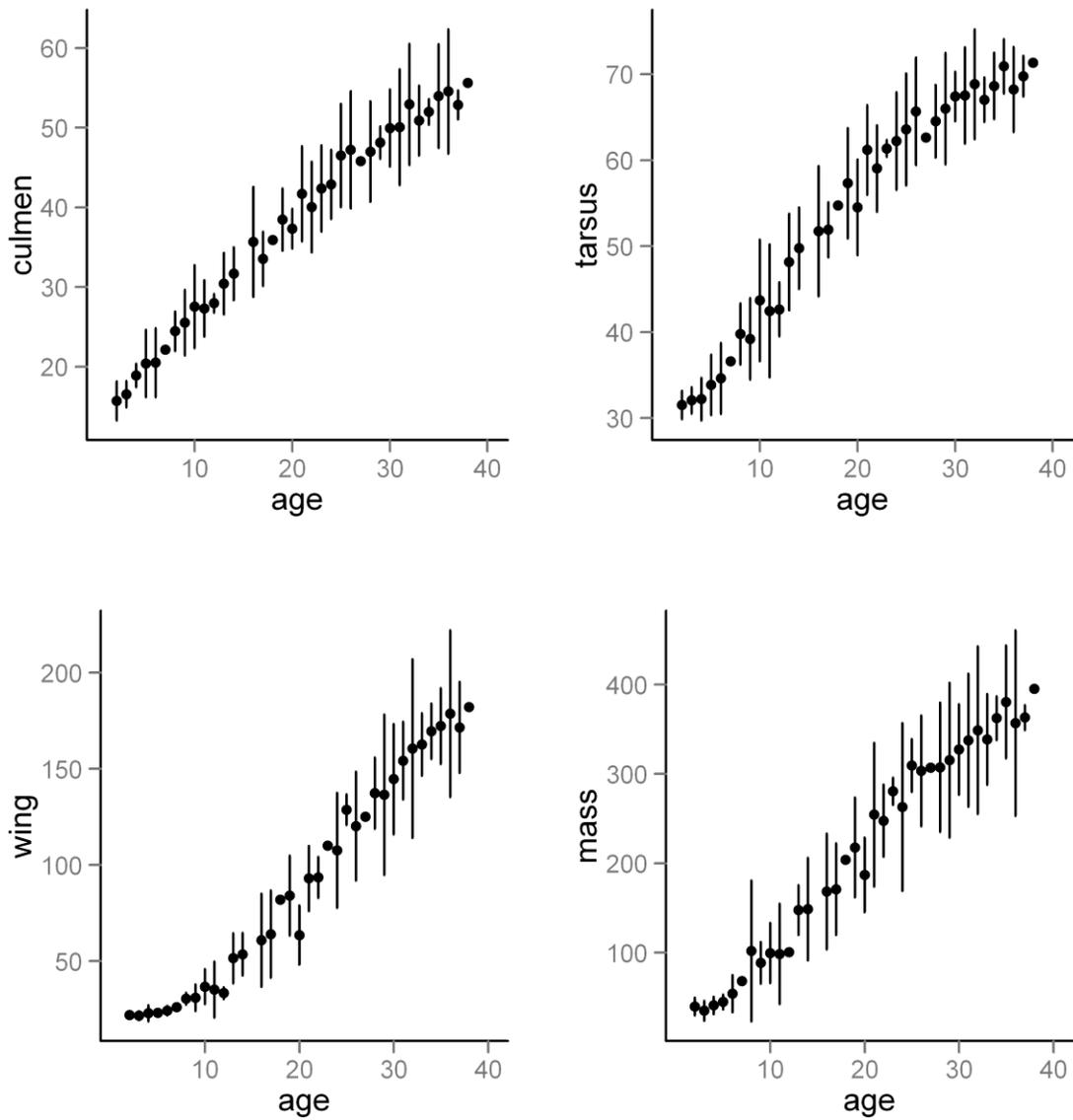


Figure 2.4. Measurements of culmen, tarsus, wing and mass of oystercatcher chicks on two barrier islands of Cape Lookout National Seashore, North Carolina, 2011. Points with no error bars indicate a single measurement for the corresponding age.

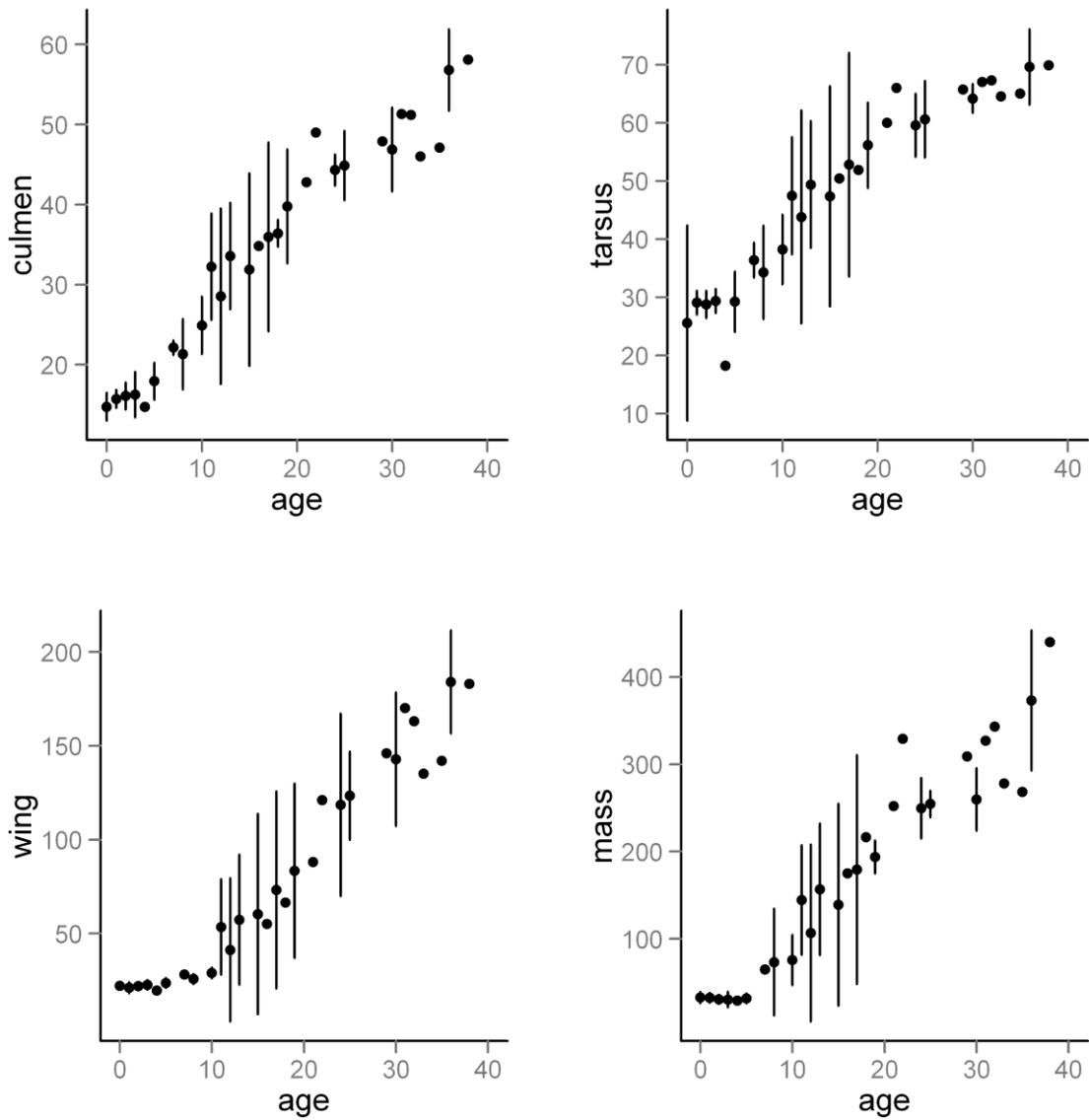


Figure 2.5. Measurements of culmen, tarsus, wing and mass of oystercatcher chicks on dredge spoil islands in North Carolina, 2011. Points with no error bars indicate a single measurement for the corresponding age.

Table 2.5. Growth parameter estimates based on the Gompertz curve (parameterized by a, b and c) for American Oystercatchers in two habitat types in North Carolina, 2011. No difference was found between chicks raised in barrier and spoil island territories, as illustrated by broad overlap in 95% credible intervals (lower and upper bounds indicated).

	a				b				c			
	mean	sd	lower	upper	mean	sd	lower	upper	mean	sd	lower	upper
culmen - barrier	66.780	3.097	61.550	73.710	-1.626	0.044	-1.715	-1.540	-0.061	0.005	-0.072	-0.050
culmen - spoil	65.230	3.728	59.030	73.690	-1.607	0.048	-1.704	-1.516	-0.059	0.006	-0.071	-0.048
tarsus - barrier	78.690	2.237	74.860	83.600	-1.111	0.029	-1.170	-1.055	-0.066	0.005	-0.077	-0.056
tarsus - spoil	79.430	4.131	72.630	88.830	-1.194	0.046	-1.287	-1.108	-0.061	0.007	-0.075	-0.048
wing - barrier	358.300	55.330	279.200	489.200	-3.436	0.100	-3.639	-3.247	-0.046	0.005	-0.056	-0.035
wing - spoil	374.200	64.770	282.600	529.200	-3.330	0.110	-3.566	-3.132	-0.043	0.005	-0.053	-0.033

Table 2.6. Causes of oystercatcher nest loss in North Carolina, 2009 - 2011.

Year	Island Type	n	Hatched	Overwash	Predation	Abandon	Unknown
2009	barrier	137	38	5	37	8	49
2009	dredge	34	25	2	0	0	7
2010	barrier	161	47	9	42	7	56
2010	dredge	49	27	7	1	2	12
2011	barrier	155	51	2	40	6	56
2011	dredge	68	24	12	5	1	26

CHAPTER 3: MANAGING NATIVE PREDATORS TO PROTECT SHOREBIRDS: EVIDENCE FROM AN EXPERIMENTAL REMOVAL OF RACCOONS ON THE OUTER BANKS OF NORTH CAROLINA

ABSTRACT

Predator control is a contentious issue. There is support for removing introduced predators to protect native prey species, but removal of native predators can be more complicated. Mammalian predation has been a primary cause of nest and chick loss for sensitive bird species on North Carolina's barrier islands. We removed half of the raccoon population on a barrier island to determine the effectiveness of a one-time partial removal for conservation of prey species of concern. Oystercatcher productivity suggests that the removal had a similar impact to predator populations as recent hurricanes. There is some evidence of a short-term, local signal to the removal, but it is unequivocal. A BACI analysis of results using the nearest island indicates no response. Because of the scale of the study area, there is no true control and no replication. This could result from an inappropriate control, too few raccoons removed, compensatory reproduction following removal, or other predators "released by the raccoons removal. Recent years of productivity data support that protection of nests and chicks, combined with these occasional disturbances in predator populations, could increase the oystercatcher population. However, performing true experiments at this scale is extremely difficult, and inference is limited. There are many interacting and potentially confounding factors influencing oystercatcher reproductive success. Predator management could be used as one tool for aiding sensitive species but would need to be part of an adaptive, multi-faceted approach.

INTRODUCTION

Predation can dramatically influence the distribution and trajectory of bird populations (Ricklefs 1969, Martin 1993, Patterson et al. 1990, Cresswell 2011). Predator removal has been justified for species that are under immediate threat of local extinction (see Côté and Sutherland 1997, Smith et al. 2010) and for enhancing populations of game species (see Reynolds and Tapper 1996). Removal of introduced mammalian predator species has been particularly effective in boosting success of native prey on isolated islands (Byrd et al. 1997, Nordström et al. 2003, Donlan et al. 2007, Medina et al. 2011, Nogales et al. 2011), where native prey species have evolved in the absence of strong predation pressure (Beggs and Wilson 1991, Wilson et al. 1998). Isolated islands are closed to recolonization, making complete extirpation of a predator species possible in some cases (e.g. Parkes et al. 2010). Closed systems are the exception to the rule, however, and recolonization can quickly counteract a removal effort (Frey et al. 2003). In cases when the predator has been introduced by anthropogenic activity, total extirpation of that predator is often considered a conservation priority. Removal of native predators is a more complicated situation, and results are more equivocal (Millus et al. 2007, Salo et al. 2010). Also, total removal of a native species is rarely a conservation objective, even in cases of threatened or endangered prey. Native predators have the same evolutionary trajectory as their ecosystems, and removal may have unintended consequences (Sih et al. 1985, Goodrich and Buskirk 1995, Ritchie et al. 2012). This integration makes it difficult to isolate the predator removal effect from other sources of variation (Millus et al. 2007). The effectiveness of partial removal of native species has been demonstrated theoretically (Boyce et al. 1999) and empirically (Whitehead et al. 2008). Partial removal may be desirable for native species (e.g. coyote and crow) whose populations have been inflated due to association with human activity and threaten already imperiled species (see Garrott et al. 1993).

Raccoons (*Procyon lotor*) are common throughout North America, with recent expansion into coastal marshes (Zeweloff 2002). They are opportunistic generalists that benefit from anthropogenic food (DeLap and Knight 2004) and activity (Bateman and Fleming 2012, Parsons et al. 2012). Raccoons are effective predators of ground-nesting

species, especially sea turtles and shorebirds, along much of the coast (Patterson et al. 1990, Erwin et al. 2001, Ellis et al. 2007) and have frequently been targeted for predator control efforts (Oring et al. 2000, Brown et al. 2001, Engeman et al. 2002, Schulte et al. 2007).

Ground-nesting birds are vulnerable to predation from opportunistic mammalian predators. Particularly devastating are species that benefit from anthropogenic activity (Garrott et al. 1993, Bateman and Fleming 2012), especially in resource-rich coastal areas (Rose and Polis 1998), where many beach-nesting species are of conservation concern (Brown et al. 2001). On the Outer Banks of North Carolina, mammals generally (Davis et al. 2001, McGowan et al. 2005) and raccoons specifically (Schulte 2012) were found to be the primary predator of American Oystercatcher (*Haematopus palliatus*) nests and chicks. We have been studying factors affecting reproductive success of the American Oystercatcher on the coast of North Carolina for over fifteen years (Figure 3.1). The oystercatcher is designated a “species of special concern” in North Carolina (North Carolina Wildlife Resources Commission 2008) and as a high priority species in the U.S. Shorebird Conservation Plan (Brown et al. 2001), in large part because of threats associated with development and increasing recreational use of coastal breeding and wintering sites.

Oystercatchers, like most shorebirds, have generally low annual fecundity characteristic of long-lived species (American Oystercatcher Working Group et al. 2012). After Hurricane Isabel passed over the Outer Banks of North Carolina in 2003, oystercatcher nest survival on North Core Banks increased dramatically (Schulte 2012; Figure 3.2; Appendix A). Schulte (2012) attributed the change to habitat creation and reduction of mammalian predators resulting from the storm. Population models indicated that occasional disturbances, such as those provided by hurricane activity on the Outer Banks, might provide a compensatory source for overall low productivity through “boom” years of above-average productivity (Schulte 2012). These findings prompted the National Park Service to initiate a study of raccoon ecology and abundance at Cape Lookout that included an experimental reduction on one barrier island (Waldstein 2010, Parsons et al. 2012). In the winter of 2008 we removed 149 raccoons (half of the island’s estimated raccoon population) between two oystercatcher nesting seasons, simulating a single, dramatic disturbance similar to a

hurricane. We hypothesized that oystercatcher reproductive success would increase following the reduction in the raccoon population.

METHODS

Study site

Cape Lookout National Seashore (hereafter Cape Lookout) extends from Ocracoke Inlet in the northeast to Beaufort Inlet in the southwest (Figure 3.1) and is managed by the National Park Service (henceforth Park Service). It is composed of three main islands, North Core Banks, South Core Banks and Shackleford Banks, although inlets open and close with storm activity. North and South Core Banks are oriented primarily northeast and extend 37 km (23 mi) and 40 km (25 mi), respectively. The islands average 0.5 km in width and are characterized by a sandy outer beach on the east, backed by sandy dunes that transition into grasses and shrubs and finally salt marsh that borders Core Sound on the west. No residences exist on the islands, and access is by commercial ferries or private boats. Vehicular traffic is limited to four-wheel-drive vehicles that traverse the island on the outer beach or a primitive access road behind the dunes. On North Core Banks, an historic village attracts some visitors, primarily foot traffic on a ferry from more heavily visited Ocracoke Island. Much of the remaining traffic can be attributed to recreational fishing. Toward the south end of South Core Banks a lighthouse and historic village attract foot and vehicle traffic. The Park Service maintains and rents cabins for public use on both islands between mid-March and early December.

Cape Lookout supports other imperiled species that also nest on the outer beaches and sand flats. Loggerhead (*Caretta caretta*), green (*Chelonia mydas*) and leatherback (*Dermochelys coriacea*) sea turtles are considered endangered by the IUCN (Baillie et al. 2004). Listed avian species within the Seashore include Least Terns (*Sterna antillarum*), Gull-billed Tern (*Gelochelidon nilotica*), Wilson's Plover (*Charadrius wilsonia*), and Piping Plover (*Charadrius melodus*). All of these are susceptible to raccoon predation.

Vegetation on Cape Lookout is described in Godfrey and Godfrey (1976). Raccoons frequently den along the marshy edges of these islands where they forage for fiddler and blue

crabs and yaupon berries (Waldstein 2010). Other potential predators of eggs and young include mink (*Mustela vison*), ghost crab (*Ocypode quadrata*), Fish Crow (*Corvus ossifragus*), gulls (*Larus spp.*), Great-horned Owl (*Bubo virginianus*), and Peregrine Falcon (*Falco peregrinus*).

Nest monitoring and reproductive metrics

Oystercatchers were monitored beginning in mid-April, as they began to establish and defend nesting territories. Many oystercatchers on Cape Lookout have been marked with uniquely identifiable leg bands, enabling us to recognize and track individual pairs through multiple nesting attempts. Nests on the outer beach were marked with signs that rerouted traffic and prevented vehicles from stopping within 300 feet of the nest. When chicks were present, vehicles were routed to a road behind the dunes when available. Nests were monitored until lost or hatched, and chicks were monitored until lost or fledged.

Cause of nest loss was determined by evidence left at the nest site, such as predator tracks, eggshells, or yolk. Predation was categorized as raccoon, mink, unknown mammal, and unknown predator. Total predation events include the above as well as the few nests that were recorded as avian or cat predation. A high percentage of nests are lost to unknown causes due to the shifting nature of the sandy substrate where oystercatchers place their nests on the islands. Predation rate was calculated as total nests depredated divided by total nests monitored. Causes of chick loss were so rarely identifiable that no comparison was possible.

Statistical analyses

Nest survival - the probability that a nest survives the 27-day incubation period - was estimated using the intercept model in the nest survival module of Program MARK 6.2 (White and Burnham 1999). MARK uses logistic regression to calculate the maximum likelihood estimate (MLE) of nest survival, incorporating the length of the monitoring interval to account for nests that were not found before they failed (Mayfield 1961, 1975). Nests were considered successful if at least one egg hatched. In the absence of evidence indicating the exact date of hatch or failure of a nest, we used the midpoint between two consecutive checks (Johnson 1979). Nests that were found after hatching were not included

in the analysis, and nests that disappeared around the anticipated hatch date but in which no chicks were seen were considered failed. Productivity is calculated as the number of chicks that successfully fledge (survive to sustained flight or 35 days after hatch) per breeding pair. American Oystercatchers raise a single brood per season, and productivity represents the maximum contribution to future breeding populations in a single year.

We compared before and after nest survival and productivity on South Core Banks as independent samples using a Welch t-test for potentially unequal variances in the two samples. This treatment is somewhat unsatisfactory due to the lack of replication or a control. Therefore we also used adjacent North Core Banks (Figure 3.1) in a Before-After-Control-Impact design (Eberhardt 1976, Stewart-Oaten et al. 1986) to provide some context for assessing changes in oystercatcher nest survival and productivity on South Core Banks following the raccoon removal. Three comparisons are made: after-before, impact-control, and the interaction. The parameter of interest is the interaction of the location and timing of the treatment (raccoon removal), as it would indicate significantly different survival after the raccoon removal on South Core Banks. The two islands are of similar size, shape and orientation, and raccoons are the main predator of ground-nesting birds on both islands. We checked for correlation between the two sets of nest survival and productivity estimates on the islands prior to the removal. After is reported as the three years following the removal (2009-2011) and as 2009 separately. Comparisons and correlations were tested using the statistical software R, version 2.15.0 (R Development Core Team 2012).

Oystercatcher reproductive success on North Core Banks was anomalous in 2004 (Figures 3.2 and 3.3) with significantly higher nest survival and productivity following Hurricane Isabel in 2003 (Schulte 2012). Therefore we performed all BACI analyses a second time, omitting 2004, to look for an effect of the raccoon removal without the anomalous year.

RESULTS

Oystercatcher nest survival

We monitored 124 oystercatcher nests on South Core Banks and 152 nests on North Core Banks between 2009 and 2011. These data were compared to nests before the removal (1999 - 2008, Appendix A). There was a correlation between nest survival in the pre-removal years for the two islands ($t=2.8701$, $df=7$, $p=0.02399$). Cumulative nest survival estimates over the periods before and after the removal, respectively, were 0.2359 ($n=349$, SE 0.040) and 0.2893 ($n=152$, SE 0.056) for North Core Banks and 0.1598 ($n=380$, SE 0.032) and 0.2797 ($n=124$, SE 0.052) for South Core Banks. Nest survival estimates on South Core Banks were not different in the before and after periods ($t = -1.9749$, $df = 3.68$, $p = 0.1256$). However, nest survival in the year following the raccoon removal was the highest on record for South Core Banks (Appendix A). Nest survival on South Core Banks in 2009 surpassed that of North Core Banks for the first time in nine years (Figure 3.2). BACI interaction results do not indicate a response of nest survival to the removal during 2009 or the entire post-removal interval (Table 3.3).

The mean predation rate of oystercatcher nests on South Core Banks during the study was 26.77% ($n=548$, SE=3.09; range 16 - 46%; Table 3.5). Raccoon predation of oystercatcher nests was high the year prior to the study (2008) and lower in years following the removal.

Oystercatcher productivity

We monitored chicks from 117 and 132 broods on South Core Banks and North Core Banks, respectively (Appendix A). We did not find support for interdependence of productivity estimates from the two islands before the removal ($t = 0.697$, $df = 8$, $p = 0.5056$). Productivity estimates before and after the removal, respectively, were 0.2388 ($n=222$, SE 0.080) and 0.4930 ($n=92$, SE 0.13) for North Core Banks and 0.1788 ($n=236$, SE 0.048) and 0.5797 ($n=69$, SE 0.080) for South Core Banks. Oystercatcher productivity increased on South Core Banks after the removal ($t = -4.3624$, $df = 3.456$, $p = 0.01654$), with the highest productivity documented in 2010 (Figure 3.2; Appendix A).

Both after-removal scenarios showed significant effects of the after-before comparison across the two islands (Table 3.4). We found no difference between the two islands across the two time intervals and no significant interaction effect.

Predation rates

Both raccoon and overall predation rates on oystercatcher nests were high in 2008 relative to years before the removal. Both rates dropped following the removal. The overall predation rate after the removal was lowest in 2009 and then began to rise, while raccoon predation rates continued to fall following the removal.

DISCUSSION

Removal of predators is a controversial issue (see Garrott et al. 1993, Goodrich and Buskirk 1995). In some systems it can be a cost-effective way of improving the reproductive success of imperiled species (Shwiff et al. 2005, Engeman et al. 2010). However, inference can be difficult due to the lack of replication in large-scale observational studies (Osaken 2001). We used correlative spatial and temporal evidence to provide context for understanding the effect of a partial raccoon removal on the reproductive success of a shorebird prey species.

Barrier islands are inherently dynamic, with weather and wave action regularly altering their form and location. Oystercatchers appear to be adapted to periodic bursts of high fecundity following disturbance of their barrier island nesting habitat (Schulte 2012). Like many long-lived species oystercatchers have generally low annual recruitment rates (American Oystercatcher Working Group et al. 2012). There is a tremendous amount of variability in reproductive success from year to year, however, caused by a combination of factors. Looking at a single year after major disturbances, it is tempting to infer an effect. However, with the added perspective of multiple years, we can begin to see the difficulty in attributing the variation to any particular factor.

The overall productivity of breeding oystercatchers is comprised of both the survival of eggs to hatching (nest survival) and the number of chicks that survive to fledging. Both nest survival and the number of chicks fledged per breeding pair increased for oystercatchers

on South Core Banks from the year before to the year after the raccoon removal. Including all years of monitoring on South Core Banks alone, we see a significant response in chicks fledged per breeding pair but not in nest survival. We know that raccoons are a primary cause of nest loss during incubation but it seems unlikely that they are effective predators of the highly precocial chicks which are defended by their parents and very mobile shortly after hatching. Ivan and Murphy (2005) experimentally showed that mammals were the primary predators of Piping Plover (*Charadrius melodus*) nests in the Great Plains, while avian predators were more likely to predate chicks.

A Before-After-Control-Impact analysis does not indicate that the raccoon removal on South Core Banks had a significant effect on nest survival or productivity. One key assumption in the BACI design is additivity - any variation outside of the treatment will affect the two islands in the same way to the same degree. Therefore, in the absence of the treatment (predator removal), we would expect no difference in success on the two islands. This approach should provide an unbiased comparison of the different predator management strategies on the two islands. However, Schulte (2012) described how weather - in the form of hurricanes - impacts the two islands separately through differing degrees of habitat creation for oystercatchers. It is therefore possible that less severe weather events also impact the two islands differently. South Core Banks receives approximately twice as many visitors per year as North Core Banks (National Park Service, unpublished data). We have anecdotal evidence that the full predator compositions of the two islands are not the same. Any of these differences violates a critical assumption necessary for inference from the BACI comparison. Additionally, while nest survival was correlated on the two islands prior to the removal event, we did not find a correlation between overall oystercatcher productivity on the two islands. While North Core Banks provides the only plausible control for a study at this scale, it may not meet the assumptions necessary for a true BACI comparison.

Low-level predator control has been sufficient in protecting a whoio population in New Zealand (Whitehead et al. 2008), but it is quite possible that removing 50% of the estimated raccoon population on South Core Banks was simply not sufficient to significantly reduce predation on oystercatcher nests and chicks. Although we know that raccoons are a major

shorebird predator in this system (Davis et al. 2001, Schulte 2012), we do not know the functional relationship between raccoon density and shorebird nesting success. A recent reanalysis of the initial raccoon capture-recapture data (Sollmann et al., in revision) indicates that the initial population may have been reduced by as much as 80%, suggesting that the threshold necessary for effective management may be substantially higher than 50%. A similar study in Florida found no response in sea turtle hatching success after a 50% reduction of a barrier island raccoon population (Ratnaswamy et al. 1997). Predator-prey models developed by Martin et al. (2010) indicate that a single, large reduction in predator numbers may not be an optimal removal strategy for increasing oystercatcher productivity on South Core Banks. The authors cautioned that temporary low predator abundance following a large removal might attract breeding oystercatchers to South Core Banks potentially creating an ecological trap as the raccoon population recovered (Harris and Wanless 1997, Martin et al. 2010). We have found no evidence for this over the past several years but it is clear the effects of the raccoon removal were short lived.

If raccoon reproduction is density dependent, we would expect increased reproductive output in response to a raccoon population decline. Rosatte et al. (2007) removed raccoons and found that densities one year post-removal were the same as the year before. In our study, the pulse of high reproductive success followed by declining success over the next several years following both a major hurricane and the raccoon removal suggests that the raccoon population recovered quickly.

Individual raccoons could be dramatically heterogeneous in their impact on beach nesting birds, i.e. there could exist in the population only a small number of “problem individuals” that regularly travel to the beachfront and are responsible for the majority of nest depredation (Sanz-Aguilar et al. 2009). In the case that these individuals were not removed, depredation of beach nests would not be reduced. Waldstein (2010) found that 6% of telemetry locations of raccoons on South Core Banks in 2008 were on the beach, while only 3% were on the beach in 2009. One interpretation of this is that fewer animals were searching the beach for food.

It is also possible that the raccoon removal on South Core Banks “released” other nest predators (Palomares et al. 1995, Crooks and Soulé 1999, Ginger et al. 2003, Rayner et al. 2007, Barton and Roth 2008, Bodey et al. 2009) resulting in lower rates of raccoon predation but no change in nest survival or productivity. As Cresswell (2011) points out, the lack of apparent effect of predator removal does not mean that the removed predator does not impact the prey species unless all other predators were removed or their effects were quantified. Other predators of ground-nesting birds may have benefited from the reduction in the raccoon population and thus hidden the effect (Ellis-Felege et al. 2012). Mink (*Mustela vison*) are present on the island and during this study were implicated in destruction of both oystercatcher and plover nests (Altman 2011). Predation at eight Piping Plover nests on South Core Banks in 2010 was attributed to mink which unlike raccoons are not excluded by enclosure fencing installed by the National Park Service. Oystercatcher nests were lost to mink predation in 2011 for the first time on record (Table 3.5). Ghost crabs predate both eggs and chicks (Schulte 2012) and Barton and Roth (2008) suggested that ghost crab populations might be limited by raccoon predation (Barton and Roth 2008). Waldstein (2010) found no evidence of ghost crabs in stomachs of the raccoons collected in 2008, although most sampling took place outside of the oystercatcher nesting season (Parsons et al. 2012). Although we cannot rule out predator release as an explanation for the observed response of oystercatcher productivity to the raccoon removal, we found no direct evidence to support this hypothesis. Other management actions, such as reducing human trash and fishing waste that serve as a supplemental food source for raccoons, may provide cost effective alternatives to direct predator control (Prange et al 2003). Expanded management of disturbance from vehicles and recreation in recent years may also explain increasing oystercatcher nest success (Altman 2011a, Appendix A).

Although American Oystercatcher vital rates are most sensitive to adult survival (Schulte 2012), management actions on the breeding grounds are rarely able to influence adult survival and focus instead on nest survival and chick production. Evidence that hurricanes improve oystercatcher productivity by reducing predator populations and creating new habitat implies that management actions that mimic these effects could help sustain

populations in the face of changing and disappearing natural habitat. For these reasons, predator control to increase oystercatcher nesting success has been identified as a priority management strategy by the American Oystercatcher Working Group (Schulte et al. 2007).

However, evidence that removing predators from our study site improved oystercatcher reproductive success is equivocal. This is not surprising given the many interacting and potentially confounding factors that influence spatial and temporal patterns of oystercatcher productivity. For instance, human disturbance can change the attendance of adult oystercatchers (Verhulst et al. 2001, McGowan and Simons 2006), resulting in higher predation rates (Tjorve and Underhill 2008; but see Verboven et al. 2001). The effects of spring storms and fall hurricanes, food availability, and predator populations can vary widely from year to year. Therefore predator management should be viewed as one of many tools available to promote shorebird conservation. We believe that effective conservation will require adaptive strategies that monitor multiple factors affecting the fecundity and survival of sensitive shorebird populations.

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FIGURES



Figure 3.1. Map and aerial photo of study area, Cape Lookout National Seashore, North Carolina. The southernmost point in the Seashore is the cape of South Core Banks (lower left), which extends northeast to Ophelia Inlet (center of image). North Core Banks continues to the northeast and ends at Ocracoke Inlet (upper right).

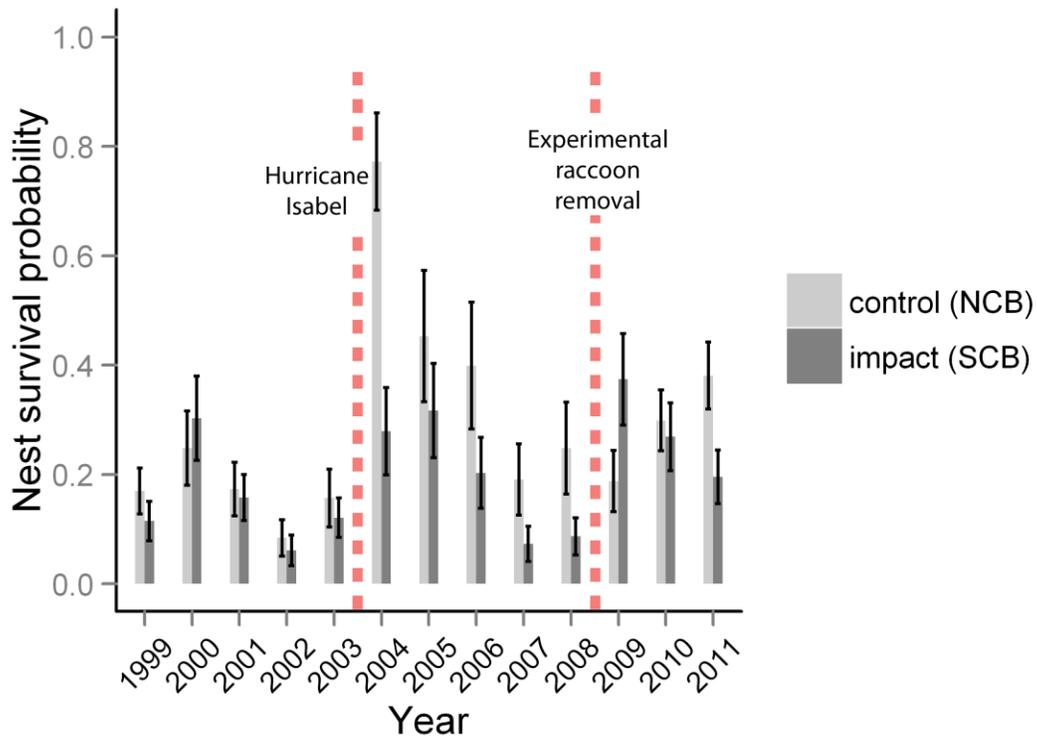


Figure 3.2. Nest survival probability of American Oystercatchers (*Haematopus palliatus*) on South Core Banks (SCB) and North Core Banks (NCB) from 1998-2011. The dashed lines indicate Hurricane Isabel, which naturally reduced predator abundance on North Core Banks, and an experimental 50% reduction in the raccoon population on South Core Banks. Bars represent one standard error.

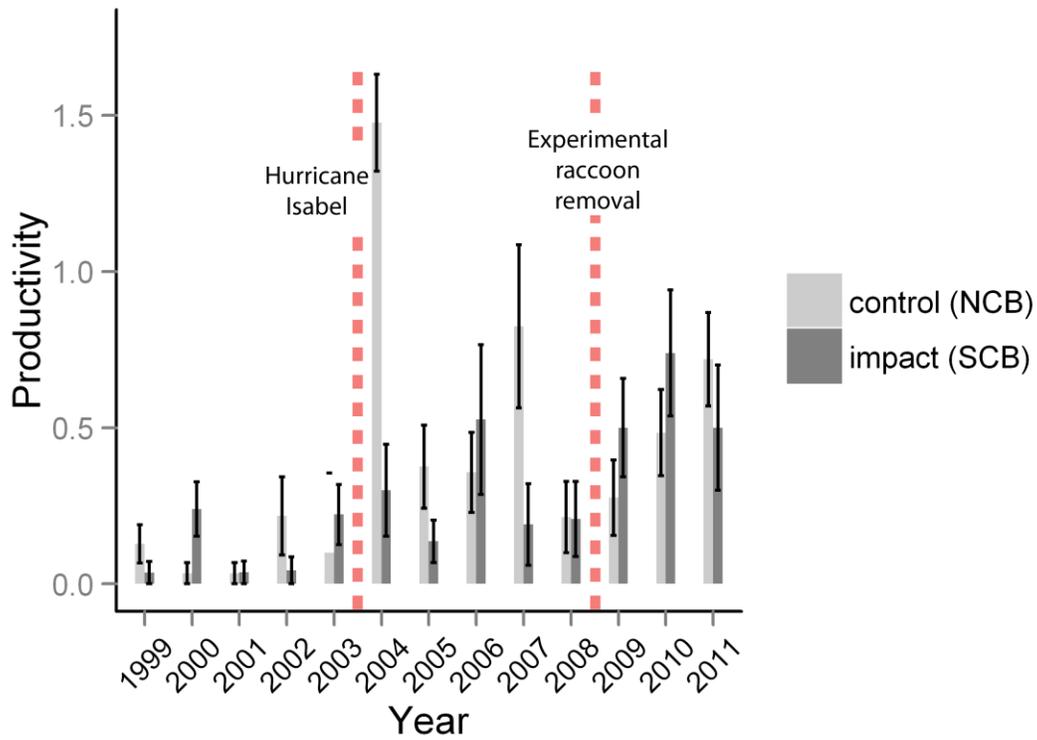


Figure 3.3. Productivity of American Oystercatchers (*Haematopus palliatus*) on South Core Banks (SCB) and North Core Banks (NCB) from 1998-2011. The dashed lines indicate Hurricane Isabel, which naturally reduced predator abundance on North Core Banks, and an experimental 50% reduction in the raccoon population on South Core Banks. Bars represent one standard error.

Table 3.1. Nest survival rates for the American Oystercatcher (*Haematopus palliatus*) in North Carolina before and after a 50% reduction in the raccoon population on a control island, North Core Banks, and the impact/treatment island, South Core Banks, Cape Lookout National Seashore. Standard error estimates in parentheses.

YEAR	BA	Control	Impact
1999	B	0.170 (0.042)	0.115 (0.036)
2000	B	0.248 (0.068)	0.303 (0.077)
2001	B	0.173 (0.049)	0.158 (0.042)
2002	B	0.084 (0.033)	0.061 (0.028)
2003	B	0.157 (0.053)	0.121 (0.036)
2004	B	0.772 (0.089)	0.279 (0.080)
2005	B	0.453 (0.120)	0.317 (0.086)
2006	B	0.399 (0.116)	0.203 (0.065)
2007	B	0.191 (0.065)	0.073 (0.032)
2008	B	0.248 (0.084)	0.087 (0.034)
2009	A	0.188 (0.056)	0.374 (0.084)
2010	A	0.299 (0.056)	0.269 (0.062)
2011	A	0.381 (0.061)	0.196 (0.049)

Table 3.2. Productivity (chicks fledged per breeding pair) for the American Oystercatcher (*Haematopus palliatus*) in North Carolina before and after a 50% reduction in the raccoon population on the impact island, South Core Banks, Cape Lookout National Seashore. Standard error estimates in parentheses.

YEAR	BA	Control	Impact
1999	B	0.128 (0.061)	0.036 (0.036)
2000	B	0.034 (0.034)	0.240 (0.087)
2001	B	0.034 (0.034)	0.037 (0.036)
2002	B	0.217 (0.125)	0.043 (0.043)
2003	B	0.100 (0.255)	0.222 (0.096)
2004	B	1.476 (0.155)	0.300 (0.147)
2005	B	0.375 (0.133)	0.136 (0.068)
2006	B	0.357 (0.128)	0.526 (0.240)
2007	B	0.824 (0.261)	0.190 (0.131)
2008	B	0.214 (0.114)	0.208 (0.120)
2009	A	0.276 (0.121)	0.500 (0.158)
2010	A	0.484 (0.138)	0.739 (0.202)
2011	A	0.719 (0.150)	0.500 (0.200)

Table 3.3. Results from BACI Analysis of Variance (ANOVA) for oystercatcher nest survival before (B) and after (A) 50% reduction in raccoon population on the impact/treatment island, South Core Banks (I), and the control island, North Core Banks (C), Cape Lookout National Seashore, North Carolina.

1999-2008 as before (without 2004) and **2009-2011** as after

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
BA	1	0.005	0.005	0.433	0.518
CI	1	0.00647	0.00647	0.559	0.463
BA:CI	1	0.03371	0.03371	2.916	0.103
Residuals	20	0.23121	0.01156		

1999-2008 as before (without 2004) and **2009 only** as after

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
BA	1	0.01136	0.01136	1.008	0.329
CI	1	0.01427	0.01427	1.267	0.275
BA:CI	1	0.0181	0.0181	1.606	0.221
Residuals	18	0.20285	0.01127		

Table 3.4. Results from BACI Analysis of Variance (ANOVA) for oystercatcher productivity before and after 50% reduction in raccoon population on the experimental/impact island, South Core Banks (I), and the control island, North Core Banks (C), Cape Lookout National Seashore, North Carolina.

1999-2008 as before (without 2004) and **2009-2011** as after

	df	Sum Sq	Mean Sq	F value	pr(>F)	
BA	1	0.4561	0.4561	11.216	0.0032	**
CI	1	0.0061	0.0061	0.15	0.7022	
BA:CI	1	0.0282	0.0282	0.694	0.4146	
Residuals	20	0.8134	0.0407			

1999-2008 as before (without 2004) and **2009 only** as after

	df	Sum Sq	Mean Sq	F value	pr(>F)	
BA	1	0.2506	0.25062	5.819	0.0267	*
CI	1	0.0191	0.01908	0.443	0.5141	
BA:CI	1	0.004	0.00398	0.092	0.7645	
Residuals	18	0.7752	0.04307			

Table 3.5. American Oystercatcher (*Haematopus palliatus*) apparent predation rates on South Core Banks, Cape Lookout National Seashore, North Carolina. Fifty percent of the island’s raccoon population was experimentally removed between the 2008 and 2009 nesting seasons.

Measure	Year												
	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
# nests	52	38	56	43	59	33	27	31	41	44	30	43	51
# predated (total)	9	6	26	18	12	11	4	6	6	18	7	12	17
raccoon	8	3	15	10	4	4	1	3	5	11	5	5	6
mammal	0	0	1	0	0	0	0	0	0	1	1	5	3
mink	0	0	0	0	0	0	0	0	0	0	0	1	7
ghost crab	0	0	0	0	0	0	0	0	0	2	1	1	1
raccoon predation rate	15%	8%	27%	23%	7%	12%	4%	10%	12%	25%	17%	12%	12%
overall predation rate	17%	16%	46%	42%	20%	33%	15%	19%	15%	41%	23%	28%	33%

APPENDIX

Appendix A– American Oystercatcher productivity in North Carolina from 1995-2012

Year and Location	Breeding pairs	Nests	Nests hatched	Nest survival observed (SE)	Nest survival adjusted (SE)	Chicks fledged	Chick Survival (SE)	Chicks fledged/ breeding pair (SE)
CAPE LOOKOUT								
North Core Banks								
1998	38	72	5	0.069 (0.030)	NA	4	NA	0.105 (0.062)
1999	39	61	11	0.177 (0.049)	0.170 (0.042)	5	0.208 (0.083)	0.128 (0.061)
2000	29	36	7	0.194 (0.066)	0.248 (0.068)	1	0.059 (0.057)	0.034 (0.034)
2001	29	53	12	0.226 (0.057)	0.173 (0.049)	1	0.091 (0.061)	0.034 (0.034)
2002	23	46	4	0.087 (0.042)	0.084 (0.033)	5	0.455 (0.150)	0.217 (0.125)
2003	20	36	7	0.194 (0.066)	0.157 (0.053)	2	0.118 (0.078)	0.100 (0.069)
2004	21	25	20	0.800 (0.080)	0.772 (0.089)	31	0.608 (0.068)	1.476 (0.255)
2005	16	20	11	0.550 (0.111)	0.453 (0.120)	6	0.286 (0.099)	0.375 (0.155)
2006	14	18	8	0.444 (0.117)	0.399 (0.116)	5	0.263 (0.101)	0.357 (0.133)
2007	17	32	8	0.250 (0.077)	0.191 (0.065)	14	0.778 (0.098)	0.824 (0.261)
2008	14	22	4	0.182 (0.082)	0.248 (0.084)	3	0.429 (0.187)	0.214 (0.114)
2009	29	40	7	0.175 (0.060)	0.188 (0.056)	8	0.533 (0.129)	0.276 (0.121)
2010	31	58	15	0.259 (0.059)	0.299 (0.056)	15	0.500 (0.091)	0.484 (0.130)
2011	32	54	18	0.333 (0.064)	0.381 (0.061)	24	0.649 (0.078)	0.750 (0.149)
2012	15	26	9	0.346 (0.093)	0.351 (0.092)	14	0.636 (0.111)	0.933 (0.284)

Middle Core Banks								
2004	5	5	4	0.800 (0.179)	NA	7	0.875 (0.117)	1.400 (0.510)
2005	7	9	5	0.556 (0.166)	0.511 (0.172)	9	0.643 (0.128)	1.286 (0.474)
2006	8	9	7	0.778 (0.139)	0.745 (0.155)	8	0.500 (0.125)	1.000 (0.267)
2007	11	11	7	0.636 (0.145)	0.570 (0.160)	10	0.833 (0.108)	0.909 (0.315)
2008	6	6	4	0.667 (0.192)	NA	7	0.875 (0.117)	1.167 (0.477)
2012	13	18	7	0.389 (0.115)	0.218 (0.106)	12	0.706 (0.111)	0.923 (0.288)
Ophelia Banks								
2007	2	3	2	0.667 (0.272)	NA	3	0.750 (0.217)	1.500 (0.500)
2008	2	2	1	0.500 (0.354)	NA	0	0.000 (0.000)	0.000 (0.000)
South Core Banks								
1995	20	36	12	0.333 (0.079)	NA	7	NA	0.350 (0.131)
1997	23	34	4	0.118 (0.055)	0.036 (0.022)	2	0.286 (0.171)	0.087 (0.060)
1998	20	26	7	0.269 (0.087)	0.135 (0.062)	3	0.214 (0.110)	0.150 (0.082)
1999	28	52	5	0.096 (0.041)	0.115 (0.036)	1	0.125 (0.117)	0.036 (0.036)
2000	25	38	17	0.474 (0.081)	0.303 (0.077)	6	0.120 (0.046)	0.240 (0.087)
2001	27	56	8	0.143 (0.047)	0.158 (0.042)	1	0.050 (0.049)	0.037 (0.036)
2002	23	43	4	0.093 (0.044)	0.061 (0.028)	1	0.143 (0.132)	0.043 (0.043)
2003	27	59	9	0.153 (0.047)	0.121 (0.036)	6	0.273 (0.095)	0.222 (0.096)
2004	20	33	13	0.394 (0.085)	0.279 (0.080)	6	0.231 (0.083)	0.300 (0.147)
2005	22	27	9	0.333 (0.091)	0.317 (0.086)	3	0.188 (0.098)	0.136 (0.068)

2006	19	31	6	0.194 (0.071)	0.203 (0.065)	10	0.769 (0.117)	0.526 (0.246)
2007	21	41	4	0.098 (0.046)	0.073 (0.032)	4	0.571 (0.187)	0.190 (0.131)
2008	24	44	5	0.114 (0.048)	0.087 (0.034)	5	0.625 (0.171)	0.208 (0.120)
2009	22	30	11	0.367 (0.088)	0.374 (0.084)	11	0.500 (0.107)	0.500 (0.170)
2010	23	43	10	0.233 (0.064)	0.269 (0.062)	17	0.680 (0.093)	0.739 (0.237)
2011	24	51	9	0.176 (0.053)	0.196 (0.049)	12	0.545 (0.106)	0.500 (0.200)
2012	21	41	14	0.341 (0.074)	0.249 (0.071)	16	0.727 (0.095)	0.762 (0.193)
Shackleford Banks								
2003	7	10	1	0.100 (0.095)	NA	0	0.000 (0.000)	0.000 (0.000)
2004	6	8	1	0.125 (0.117)	NA	1	1.000 (0.000)	0.167 (0.408)
2005	9	10	1	0.100 (0.095)	NA	0	0.000 (0.000)	0.000 (0.000)
2006	9	11	1	0.091 (0.087)	0.071 (0.061)	1	1.000 (0.000)	0.111 (0.111)
2007	10	12	0	0.000 (0.000)	0.110 (0.088)	0	NA	0.000 (0.000)
2008	11	17	3	0.176 (0.092)	0.059 (0.046)	0	0.000 (0.000)	0.000 (0.000)
2009	10	13	2	0.154 (0.100)	0.119 (0.078)	2	0.667 (0.272)	0.200 (0.200)
2010	8	12	2	0.167 (0.108)	0.186 (0.100)	2	0.667 (0.272)	0.250 (0.250)
2011	7	9	2	0.222 (0.139)	0.203 (0.123)	1	0.500 (0.354)	0.143 (0.143)
2012	8	13	0	0.000 (0.000)	0.117 (0.073)	0	NA	0.000 (0.000)
CAPE HATTERAS								
Ocracoke Island								
1999	15	17	7	0.412 (0.119)	0.321 (0.105)	2	0.182 (0.116)	0.133 (0.091)

2000	12	17	6	0.353 (0.116)	0.270 (0.107)	7	0.778 (0.139)	0.583 (0.260)
2001	13	15	11	0.733 (0.114)	0.624 (0.132)	12	0.600 (0.110)	0.923 (0.265)
2002	12	18	6	0.333 (0.111)	0.266 (0.102)	3	0.250 (0.125)	0.250 (0.131)
2003	8	12	4	0.333 (0.136)	0.255 (0.117)	1	0.250 (0.217)	0.125 (0.125)
2004	9	11	6	0.545 (0.150)	0.566 (0.144)	8	0.727 (0.134)	0.889 (0.309)
2005	5	10	3	0.300 (0.145)	0.295 (0.136)	1	0.167 (0.152)	0.200 (0.200)
2006	5	8	4	0.500 (0.177)	0.492 (0.202)	2	0.182 (0.116)	0.400 (0.400)
2007	5	12	3	0.250 (0.125)	0.102 (0.078)	1	0.250 (0.217)	0.200 (0.200)
2008	3	3	1	0.333 (0.272)	0.347 (0.260)	2	1.000 (0.000)	0.667 (0.667)
2009	4	6	2	0.333 (0.192)	0.400 (0.212)	0	0.000 (0.000)	0.000 (0.000)
2010	4	6	5	0.833 (0.152)	0.849 (0.139)	3	0.333 (0.147)	0.750 (0.250)
2011	5	5	5	1.000 (0.000)	0.825 (0.159)	7	0.500 (0.134)	1.400 (0.400)
2012	5	6	1	0.167 (0.152)	0.377 (0.164)	0	0.000 (0.000)	0.000 (0.000)

Hatteras Island

1999	24	31	7	0.226 (0.075)	0.287 (0.087)	3	0.273 (0.134)	0.125 (0.069)
2000	23	29	10	0.345 (0.088)	0.270 (0.081)	2	0.087 (0.059)	0.087 (0.060)
2001	24	28	10	0.357 (0.091)	0.259 (0.083)	7	0.389 (0.115)	0.292 (0.112)
2002	17	25	3	0.120 (0.065)	0.030 (0.023)	4	0.800 (0.179)	0.235 (0.136)
2003	16	23	10	0.435 (0.103)	0.372 (0.106)	6	0.286 (0.099)	0.375 (0.155)
2004	15	18	13	0.722 (0.106)	0.706 (0.110)	9	0.360 (0.096)	0.600 (0.235)
2005	17	24	13	0.542 (0.102)	0.501 (0.110)	10	0.417 (0.101)	0.588 (0.196)

2006	14	19	11	0.579 (0.113)	0.525 (0.120)	6	0.316 (0.107)	0.429 (0.202)
2007	15	21	10	0.476 (0.109)	0.477 (0.102)	9	0.450 (0.111)	0.600 (0.235)
2008	15	20	9	0.450 (0.111)	0.565 (0.102)	11	0.611 (0.115)	0.733 (0.267)
2009	13	19	11	0.579 (0.113)	0.555 (0.109)	9	0.429 (0.108)	0.692 (0.263)
2010	15	17	13	0.765 (0.103)	0.763 (0.103)	23	0.719 (0.079)	1.533 (0.336)
2011	14	17	13	0.765 (0.103)	0.789 (0.093)	16	0.571 (0.094)	1.143 (0.294)
2012	12	21	16	0.762 (0.093)	0.818 (0.082)	15	0.405 (0.080)	1.25 (0.218)

Bodie Island

1999	2	3	0	0.000 (0.030)	0.030 (0.035)	0	NA	0.000 (0.000)
2000	2	3	0	0.000 (0.081)	0.081 (0.081)	0	NA	0.000 (0.000)
2001	2	3	1	0.333 (0.272)	0.285 (0.253)	1	0.500 (0.354)	0.500 (0.500)
2002	2	5	1	0.200 (0.179)	0.138 (0.137)	2	1.000 (0.000)	1.000 (1.000)
2003	5	5	1	0.200 (0.179)	0.311 (0.182)	0	0.000 (0.000)	0.000 (0.000)
2004	3	6	0	0.000 (0.000)	0.091 (0.089)	0	NA	0.000 (0.000)
2005	2	3	1	0.333 (0.272)	0.390 (0.260)	0	0.000 (0.000)	0.000 (0.000)
2006	2	2	1	0.500 (0.354)	0.400 (0.367)	0	0.000 (0.000)	0.000 (0.000)
2007	2	2	1	0.500 (0.354)	0.545 (0.331)	0	0.000 (0.000)	0.000 (0.000)
2008	3	5	2	0.400 (0.219)	0.361 (0.212)	2	0.100 (0.000)	0.667 (0.333)
2009	4	4	1	0.250 (0.217)	0.274 (0.205)	1	0.500 (0.354)	0.250 (0.250)
2010	1	2	1	0.500 (0.354)	0.477 (0.353)	0	0.000 (0.000)	0.000 (0.000)
2011	1	1	1	1.000 (0.000)	1.000 (0.000)	1	1.000 (0.000)	1.000 (0.000)

2012	1	2	0	0.000 (0.000)	0.267 (0.250)	0	NA	0.000 (0.000)
Green Island								
2004	2	3	2	0.667 (0.272)	NA	2	0.500 (0.250)	1.000 (1.000)
2005	2	3	2	0.667 (0.272)	NA	0	0.000 (0.000)	0.000 (0.000)
2006	2	2	2	1.000 (0.000)	NA	2	1.000 (0.000)	1.000 (0.000)
2007	2	2	1	0.500 (0.354)	NA	2	0.667 (0.272)	1.000 (1.000)
2008	2	4	1	0.150 (0.217)	NA	2	1.000 (0.000)	1.000 (1.000)
2009	2	2	1	0.500 (0.354)	NA	3	1.000 (0.000)	1.500 (0.882)
2010	3	3	2	0.667 (0.272)	0.529 (0.337)	4	0.667 (0.192)	1.333 (0.667)
2011	3	3	3	1.000 (0.000)	1.000 (0.000)	4	0.667 (0.192)	1.333 (0.333)
CAPE FEAR REGION								
Cape Fear River Islands								
2002	32	47	26	0.553 (0.073)	0.534 (0.073)	7	0.149 (0.052)	0.219 (0.074)
2003	34	50	15	0.300 (0.065)	0.367 (0.064)	7	0.333 (0.103)	0.206 (0.066)
2009	57	62	42	0.677 (0.059)	0.509 (0.075)	27	0.435 (0.063)	0.474 (0.094)
2010	50	63	39	0.619 (0.061)	0.570 (0.071)	37	0.514 (0.059)	0.740 (0.237)
2011	65	122	48	0.393 (0.044)	0.371 (0.044)	28	0.354 (0.054)	0.431 (0.076)
2012	21	128	36	0.281 (0.040)	0.296 (0.039)	21	0.420 (0.070)	1.000 (0.108)
Lea-Hutaff Island								
2003	16	16	11	0.688 (0.116)	0.617 (0.133)	9	0.391 (0.102)	0.563 (0.203)
2009	18	22	4	0.182 (0.082)	0.085 (0.050)	1	0.143 (0.132)	0.056 (0.056)
2010	14	18	0	0.000 (0.000)	0.006 (0.008)	0	NA	0.000 (0.000)

2011	15	17	3	0.176 (0.092)	0.039 (0.035)	2	0.667 (0.272)	0.133 (0.091)
2012	8	11	0	0.000 (0.000)	0.000 (0.000)	0	NA	0.000 (0.000)
INLET ISLANDS								
Ocracoke Inlet Islands								
2009	15	23	7	0.304 (0.096)	0.358 (0.102)	2	0.167 (0.108)	0.133 (0.091)
2010	16	19	15	0.789 (0.094)	0.859 (0.092)	21	0.677 (0.084)	1.313 (0.235)
2011	15	16	13	0.813 (0.071)	0.781 (0.136)	16	0.615 (0.095)	1.067 (0.211)
Oregon Inlet Islands								
2009	11	12	10	0.833 (0.108)	0.806 (0.123)	7	0.350 (0.107)	0.636 (0.279)
2010	10	11	6	0.545 (0.150)	0.537 (0.167)	4	0.400 (0.155)	0.400 (0.163)
2011	8	8	7	0.875 (0.117)	0.758 (0.148)	4	0.286 (0.121)	0.500 (0.189)
SUMMARY	1689	2633	863	0.328 (0.009)	0.331 (0.010)	710	0.480 (0.013)	0.399 (0.018)