

Evaluating Past and Present Management of Beach-Nesting Wildlife Species at the Cape Hatteras National Seashore



Photo by Shiloh Schulte



Photo by Matthew Godfrey



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Introduction

North Carolina's barrier islands are among the most popular recreational destinations within the mid-Atlantic region of the United States. A large portion of the barrier islands is included in the Cape Hatteras National Seashore, stretching over 70 miles from the southern portion of Bodie Island south to Ocracoke Inlet and including all of Hatteras and Ocracoke Islands (Figure 1). Tourists flock to the Seashore in the summer to enjoy the ocean and sand beaches, anglers line the shore in the fall, and birdwatchers search for waterfowl, shorebirds and rarities of all sorts in the winter. The beaches, marshes and other aquatic habitats within the Seashore are home to an abundance of wildlife at all seasons of the year, notably migrating and wintering birds, and beach-nesting birds and sea turtles.



Figure 1. Satellite image showing the location of towns, major geographic features, and federal properties on North Carolina's Outer Banks. Cape Hatteras National Seashore is outlined in yellow, and Cape Lookout National Seashore in black. From Riggs and Ames (2007).

Accommodating both people and wildlife in the same space can be challenging. At Cape Hatteras National Seashore (CAHA) this is particularly true of its beaches, which are popular areas for recreational use and the primary habitat used by a number of wildlife species. Managers of the Seashore strive to protect wildlife while enabling recreational uses of the beach, but in some cases, impose restrictions on the latter in order to achieve the former. Particularly controversial is regulation of pedestrian and vehicle traffic on the beach, including in some cases closure of portions of the beach to human use, to protect nesting birds and sea turtles.

Successful integration of recreational uses of CAHA with wildlife management depends on accurate knowledge of the needs of wildlife species and the potential impact of human activity on meeting those needs. Recognizing this, The National Park Service (NPS) contacted the American Ornithological Society (AOS) in June 2016 to request that the AOS convene an independent panel to review and synthesize current scientific knowledge about the biology of the Seashore's beach-nesting species of conservation concern, and to assess the appropriateness of the Seashore's beach management plan in light of this knowledge. The AOS agreed to conduct the review and convened a panel of scientists with expertise appropriate to the task, including the biology of Piping Plovers (*Charadrius melodus*) (Cheri Gratto-Trevor), American Oystercatchers (*Haematopus palliatus*), (Erica Nol), other shorebirds and colonial waterbirds (Stephen Dinsmore) and sea turtles (Matthew Godfrey). The Panel also included experts in the human dimensions of wildlife management (Ashley Dayer), barrier island geomorphology and dynamics (Stanley Riggs), and the natural and human history of North Carolina's Outer Banks (Lou Browning). This report presents the findings of the Panel.

This review falls within the purview of the AOS Subcommittee on Science Arbitration, which is a subcommittee of the Conservation Committee. This Subcommittee is charged with developing and overseeing panel reviews of science relevant to issues in avian conservation. Science arbitration involves evaluating the science relevant to an issue and providing the results of the evaluation to decision-makers without taking a position on the outcome (Pielke 2007). The objective of science arbitration is to provide scientific information to decision-makers in an objective form, without engaging in decision-making or formulation of policy. The chair of the Panel (Jeffrey Walters) is also chair of the AOS Subcommittee on Science Arbitration. Thus, the chair's responsibilities included not only overseeing and coordinating the activities of the Panel, but also ensuring that the report meets standards of scholarship established by the AOS for science arbitration projects, and adheres to the central tenets of objectivity, credibility and absence of advocacy. A manuscript based on the report will be submitted to a peer-review process established by the AOS for considering panel reports for publication in one of the Society's journals. The National Park Service provided funding for the panel review, and the firm Kearns & West acted as facilitators for the panel review. Facilitators play a key role in science arbitration by ensuring that both the panel and the review sponsors respect boundaries between science and policy. In this case, this role was well executed by Abby Arnold and Kara Naves, who in addition handled many of the logistics of conducting the review.

The Panel's specific charge, as negotiated between AOS and NPS, was as follows: Review current knowledge of the ecology, population dynamics, and habitat needs of beach-nesting bird and turtle species at Cape Hatteras National Seashore and prepare a written report that:

- Synthesizes the relevant scientific knowledge about the abiotic and biotic factors that may affect the species' use of Seashore habitats and their productivity;
- Assesses the role and relative importance of these abiotic and biotic factors in determining the species' use of Seashore habitat and their productivity;
- Assesses the reasonableness of the Seashore's management targets, a.k.a. desired future conditions for the species; and
- Provides conclusions about key uncertainties and scientific monitoring and research needs that would assist the Seashore in reaching management objectives through adaptive management.

To accomplish this task the Panel reviewed relevant published literature and unpublished reports, including key reports produced by CAHA, and held an information-gathering workshop. The

workshop was held in Nags Head, North Carolina on September 26-28, 2016, during which the Panel heard presentations from experts about the relevant species and environments, human activities, relevant history and the current issues faced by CAHA.

We first provide relevant background information, and then present our assessments and findings, organized by wildlife species. We conclude by summarizing and integrating our findings to provide an overall assessment relative to the Panel's charge.

Background

The Barrier Island System

The northeastern portion of North Carolina contains one of the world's most extensive and diverse coastal systems. A complex network of rivers, streams, and creeks moves the surface water off the uplands of the Appalachian, Piedmont, and Coastal Plain provinces towards the Atlantic Ocean. These freshwater drainages flow through their self-eroded valleys to sea level, where they intermingle with the salt waters of the Atlantic Ocean. As the channels drop below sea level there is a broad, low-sloping transition zone that forms the vast Albemarle-Pamlico drowned-river estuarine system, a great mixing basin of fresh and salt waters that connects the rivers to the ocean (Figure 1).

Fronting the estuarine zone is a narrow strip of barrier islands, referred to as the Outer Banks that act as a dam between the estuaries and ocean (Figure 1). The ribbon of sand islands that include CAHA are produced at sea level by the interaction of high-energy ocean storms with the paleo-topography of the low-sloping Coastal Plain. The sand dam is broken by a series of small openings called inlets that allow the mixing of salty ocean water with the fresh riverine water and are essential for ultimately discharging the riverine water into the sea. Only a small portion of barrier islands rise above the sea surface; the greatest portion is hidden below sea level. The surface of a barrier island is perched at the top of the shore-face, a steep ramp that slopes seaward to between 8 to 20 meters below sea level, where it flattens out onto the inner continental shelf. The beach and upper portion of this shore-face ramp is the portion of a barrier island that functions as a critical energy-absorbing surface for wave, tide, and current energy along the margin of the western Atlantic Ocean.

CAHA contains both simple and complex barrier island segments (Appendix 1). Simple barrier islands, which constitute most of the Seashore, are dominated by overwash dynamics that occur in response to storm-tide events (Figure 2). Overwash results in the deposition of sand ramps on top of the barrier, and occasionally occurs all the way across the barrier and into the back-barrier estuary. The simple barrier islands have narrowed over time as a result of dune stabilization, which minimizes the overwash process (Appendix 1). The various villages within the Seashore (Figure 1) occur on complex barrier islands, which are characterized by a younger, overwash-dominated component welded onto an older barrier island (Appendix 1). The habitats on these high and wide islands, which include maritime forest, are older and considerably more stable and consequently of less importance for the beach-nesting species considered in this report.



Figure 2. An oblique aerial photograph of an overwash event from a small tropical storm on a simple barrier island without the human interference of dune dikes, structures, or roads. Notice the abundant new shorebird and turtle nesting habitat on the upper overwash plain.

Inlets are critical components of healthy, high energy and dynamic coastal barrier island systems. Most of the simple barrier islands of the Outer Banks have had one or more inlets periodically in the recent past (Fisher 1962; Moslow and Heron 1981; Fitzgerald and Hayes 1980; Everts et al. 1983; Heron et al. 1984; Pilkey et al. 1998; Hayes and Michel 2008; Mallinson et al. 2010a; Riggs et al. 1995; 2011). Inlet dynamics are critical for barrier island building and create vast habitats of coastal flats and ponds that range from shallow subaerial grass flats to supra-tidal microbial flats, inter-tidal marshes, and sub-tidal sand shoals (Appendix 1). Inlets widen and narrow with the throat channel migrating back and forth in response to individual storms like the “wagging of a dog’s tail”. In addition, inlets migrate laterally through time, often opening, closing, and re-opening in response to changing storm dynamics and supplying new sand flat habitat for shorebirds, colonial water birds and turtles. In addition, the continuously changing pattern of inlet channels through time result in tremendous human conflicts (Fitzgerald and Hayes 1980; Cleary et al. 2001; Hayes and Michel 2008; Riggs et al. 2009; Riggs and Ames 2011).

On the simple barrier islands of CAHA, the two processes of inlet dynamics and overwash interact through time to produce a stable barrier island that is in equilibrium with both the storm dynamics and rising sea level (Riggs and Ames 2007; Riggs et al. 2009; 2011). A narrowing island segment collapses with development of an inlet. The inlet builds a back-barrier flood-tide delta (FTD), then the inlet closes, and overwash dynamics bury the FTD, fill the inlet channels, and build island elevation. Figure 3 is a summary diagram that uses examples of different aerial images from different parts of the Outer Banks to demonstrate the successive island stages within a full cycle of simple barrier island evolution in response to rising sea level and storm activity. The full cycle starts with erosion causing substantial narrowing of the island segment (Figure 3A). An inlet breaches the weak segment via a storm surge (Figure 3B). The shallow flood-tide delta shoals of the inlet build a back-barrier base that substantially increases island width (Figure 3C). When the inlet closes, the shallow FTD inter-tidal shoals develop platform marshes (Figure 3D). As the shoreline continues to recede, overwash events associated with smaller storm surges build island elevation, forming “molar-tooth” structures (Figure 3E). As the island segment continues to narrow and increase in elevation, island vegetation communities evolve (Figure 3F)

and ultimately the narrow island becomes vulnerable to inlet formation once again (Figure 3A), but now the island segment is located substantially landward and slightly higher than during the previous cycle.

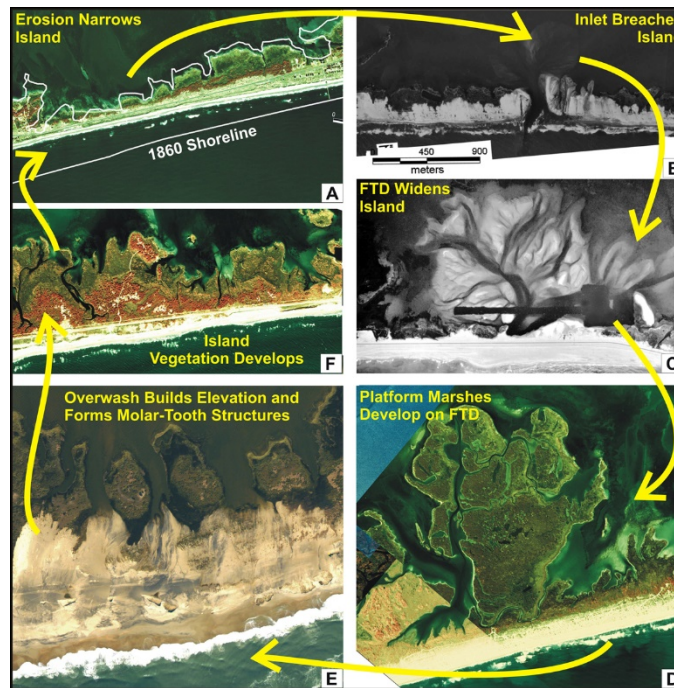


Figure 3. Summary diagram shows the successive stages of the ongoing cycle of simple barrier island evolution for the North Carolina Outer Banks in response to rising sea level and storm activity. From Riggs and Ames (2007). See text for description of stages.

Inlets may open and overwash occur, creating new and critical habitats for beach-nesting birds and turtles, in response to any given storm event wherever the barrier island is low and narrow within the Seashore. However, the immediate human reaction is to attempt to stop these processes either during the storm, or as shortly thereafter as possible. Consequently, bulldozers are brought in to “hold the status quo”. From a geomorphological and habitat point-of-view, filling an inlet or mining the newly deposited overwash sand does not change any aspect of the weak condition of that island segment. Thus, it remains just as vulnerable for yet another inlet to open or overwash fan to be deposited whenever the next “right storm” impacts that coastal segment.

The Impact of Sea-level Rise and Climate Change. Major human modifications disrupt the natural processes of island building (inlet formation and overwash dynamics) in order to “stabilize” the barrier in its present position (Figure 4). Ocean front construction (constructed barrier-dune-ridges; dense rows of motels and houses; sandbag bulkheads; highways; increased vegetative cover) minimizes or eliminates both inlet and overwash activity. However, in the current situation of a rising sea level the barrier’s ocean shoreline continues to recede. Without the ability to naturally sustain island building processes, the barrier cannot maintain equilibrium and the islands will experience long-term island narrowing, ultimately leading to island collapse into vast shallow shoal systems.



Figure 4. A. Bulldozers reconstructing a dune-ridge in the Avon to Buxton island segment after Hurricane Isabel, a small category 2 storm that came ashore in September 2003. B. Newly constructed dune-ridge adjacent to Highway 12 shortly after planting with sea oats to help stabilize it. C. Severely eroding, wave- scaped ocean side of the newly constructed dune-ridge a few months later. From Riggs and Ames (2007).

Storms are the drivers of change that erode shorelines, move barrier islands, create habitats, and cause ecosystems to migrate (Riggs and Ames 2007; Riggs et al. 2011). Storms produce the energy to do the “work” of building and moving coastal systems, to create inlets and overwash. Two kinds of storms are dominant within the North Carolina coastal system (Dolan et al. 1988; Robinson 2005). Tropical cyclones range from frequent tropical depressions to tropical storms to infrequent hurricanes; they occur primarily in summer and fall. Extra-tropical storms include ‘nor’easters’ and ‘sou’westers’ and occur primarily in fall through spring and summer, respectively. The consequences of any one storm or set of storms are highly variable and generally unpredictable (Appendix 1), but generally increase with storm size, strength and duration, all of which are components of storm intensity.

Since coastal systems form at the intersection of the land and sea, if sea level is rising, then the coastal system will migrate upward and landward with the interface. Sea-level rise by itself is a fairly slow and gentle process that is like filling a bathtub. However, climate change is expected to increase the intensity of coastal storms. An increased intensity of coastal storms superimposed upon rising sea level likely will exacerbate the issue of narrowing of simple barrier islands. The resulting habitat loss is a long-term threat common to all beach-nesting species.

Population Regulation of Beach-nesting Species

The effects of abiotic factors, biotic factors and management activities on animal populations can be divided into two categories; those impacting carrying capacity and those impacting vital rates. Carrying capacity is the maximum population size of a species that the environment can sustain. Carrying capacity depends on the quantity and quality of habitat available, and is reduced by habitat loss and increased by creation of new habitat. Vital rates include fecundity (reproductive rate), mortality (survival rate) and recruitment. Vital rates can be impacted by a variety of factors, for example increased predation on nests (reduced reproductive rate) or adults (reduced survival). Effects of habitat fragmentation such as high rates of nest loss (and thus reduced reproductive rate) at the edges of forest fragments (Robinson et al. 1995) and

disruption of dispersal (and thus reduced recruitment) by open areas between forest fragments (Cooper and Walters 2002) in forest-dwelling birds are other examples of adverse impacts expressed through changes in vital rates. Effects of habitat degradation can be conceived in terms of impacts on both carrying capacity and vital rates. Habitat degradation refers to a species-specific decline in habitat quality and as such typically results in reduced survival or reproductive rate. However, population density may be lower in degraded habitat and thus result in reduced carrying capacity as well.

Conservation issues that involve vital rates are easier to address than those that involve carrying capacity. The effects of the insecticide DDT on bird species such as Bald Eagles (*Haliaeetus leucocephalus*), Osprey (*Pandion haliaetus*), Peregrine Falcons (*Falco peregrinus*) and Brown Pelicans (*Pelecanus occidentalis*) are a classic example of a vital rates issue. DDT exposure resulted in sufficient thinning of the eggshells of these species that eggs cracked when adults sat on them during incubation. This reduced reproductive rates sufficiently that populations declined, so much so that in some cases the species was declared endangered or threatened. The solution to a vital rates problem is to identify the factor causing the adverse effect on the vital rate and address it (Caughley 1994). Since carrying capacity is unaffected, once the threat is removed and the vital rate returns to its previous level the species will in theory increase to fill the available habitat. This is precisely what happened in this case: once DDT was removed from the environment, reproductive rates increased and populations recovered. In contrast, the solution to carrying capacity issues, increasing available habitat, is difficult to achieve in practice (Reed 1995) in a world where generally habitat is declining due to anthropogenic activities resulting from an increasing, and increasingly consumptive, human population (Wilting et al. 2017). Where the issue is habitat degradation rather than habitat loss chances of success are better as the solution is habitat restoration rather than habitat creation. The recovery of the endangered Red-cockaded Woodpecker (*Picoides borealis*) achieved through application of prescribed fire to restore degraded pine savanna habitat in the southeastern United States is an example of success in addressing habitat degradation. In this case, carrying capacity was clearly the issue: population densities were greatly reduced in degraded habitat, but vital rates remained normal (Walters 1991).

Our assessment focuses primarily on the beach-nesting species of greatest management concern, namely Piping Plovers, American Oystercatchers and sea turtles. We also examine beach-nesting colonial waterbirds, albeit in less detail, focusing especially on tern species and Black Skimmers (*Rynchops niger*). Finally, we examine, again in less detail, shorebird species (i.e., Charadriiformes), including Piping Plovers and American Oystercatchers, inhabiting CAHA during migration and winter. Effects on carrying capacity through both habitat loss and habitat degradation are relevant in all these cases, as are effects on the vital rate of survival. For beach-nesting species, effects on reproductive rate are also relevant. It is through effects on this vital rate that impacts of beach closures are most likely; we therefore assess this vital rate especially thoroughly.

Conflict over Uses of Beach Habitat

Coastal shorelines and their geomorphic features (e.g., barrier islands and associated wetlands) are directly built and maintained by the region's storm dynamics. These features are essential for coastal systems to migrate in response to regional changes in sea level and tectonics. Extensive coastal development in the twentieth century on a storm dependent coastal system has created challenges with immense long-term consequences for the coastal-based economy.

Maintaining human structures built at or near sea level is often at odds with the natural dynamics of the barrier island system. In particular, the measures taken to protect the highway, buildings and other human structures interfere with the processes of inlet formation and overwash that produce habitat for beach-nesting wildlife species within the Seashore. Addressing this issue is beyond the authority of the National Park Service, and beyond the charge of the Panel, but we note that interference with these processes is a root cause of habitat limitation of beach-nesting species (see below), and of conflict over uses of the habitat that remains.

Conflicts at Cape Hatteras National Seashore. The immediate challenge that the managers of Cape Hatteras National Seashore face is to integrate provision of high-quality habitat for beach-nesting birds and sea turtles with recreational uses of beach habitats. The Seashore runs from Bodie Island to Ocracoke (Figure 1), encompassing 24,470 acres, and thus contains considerable amounts of relatively undeveloped barrier island habitats, including beach. It is the nation's first national seashore, authorized by Congress in 1937 and established in 1953. The 1937 enabling legislation of CAHA provides specific instructions and guidance on how this park must be managed. According to the enabling legislation, "except for certain portions of the area, deemed to be especially adaptable for recreational uses, particularly swimming, boating, sailing, fishing, and other recreational activities of similar nature, which shall be developed for such uses as needed, the said area shall be permanently reserved as a primitive wilderness and no development of the project or plan for the convenience of visitors shall be undertaken which would be incompatible with the preservation of the unique flora and fauna or the physiographic conditions now prevailing in this area". Thus, from the outset, the mandate of managers of CAHA has been to protect habitat (i.e., "primitive wilderness") for the area's native fauna and flora while providing for recreational activities. For protected species such as those that are the focus of this report, this mandate is reinforced for by other by laws and policies, such as the Endangered Species Act and the Migratory Bird Treaty Act. The challenge to managers is to integrate these uses of habitat over space and time, and conflicts involve how this integration is accomplished to achieve both objectives.

Managers of CAHA have enjoyed considerable success in meeting these dual objectives. According to the State of the Park Report for Cape Hatteras National Seashore (NPS 2016a), CAHA visitation is over 2 million/year, and an impressive 99% of visitors are satisfied with their visit to the Seashore, which is an increase from the average for the previous ten years (93%). CAHA offers 1600 visitor programs with over 32,000 visitors participating, and is known for both its wildlife and the opportunities for recreational activities such as swimming and fishing that it offers. The Seashore is the foundation of a large tourist industry that is critical to the local economy. Nevertheless, disagreements among stakeholder groups over uses of beach habitat exist. In particular, there is disagreement over whether restrictions on pedestrian and off-road vehicle use of beach habitats benefit beach-nesting species sufficiently to justify negative impacts on some recreational activities. This review in a sense is an assessment of whether current management is an appropriate and effective means of accomplishing and integrating the objectives of protecting habitat for flora and fauna while providing for recreational uses of beach habitat.

Piping Plovers

Piping Plovers (Figure 5) are small shorebirds that breed on sparsely vegetated shorelines of the Atlantic Coast of Canada and the United States, the Great Lakes, and the U.S. Great Plains

and Canadian Prairies (Elliott-Smith and Haig 2004; Elliott-Smith et al. 2015). Piping Plovers winter along the Atlantic and Gulf coasts of the United States and into Mexico, as well as in the Caribbean (Haig and Oring 1988a; Elliott-Smith and Haig 2004). Recent analyses based on mitochondrial DNA sequences and nuclear microsatellite loci provide genetic evidence for two subspecies: *C. melodus melodus* breeding in Eastern Canada and the Atlantic Coast of the United States, and *C. melodus circumcinctus* breeding in the Great Lakes, Prairie Canada and the U.S. Great Plains (Miller et al. 2013). In the United States, the Great Lakes breeding population was listed as Endangered, and all other populations (including the birds breeding at CAHA) as Threatened, in 1985 (USFWS 1986; 2009). In Canada, both subspecies have been listed as Endangered since 1985 (COSEWIC 2003). The IUCN Red List initially described the species as Threatened in 1988, but this has been downgraded to Near Threatened since 2005 (www.iucnredlist.org, accessed 13 Dec 2016).



Figure 5. Piping Plover at Cape Hatteras National Seashore. Photo by Katie Walker.

The species was listed due to declines in population numbers, a relatively small global population (<10,000), and loss or degradation of habitat, broadly defined to include decreased productivity due to increases in avian and mammalian predators and increased human disturbance. There are currently about 4000 birds in the Atlantic Coast breeding population (www.fws.gov/northeast/pipingplover, accessed 14 Dec 2016). Since management efforts, designed primarily to increase nest success and protect nesting habitat, were put in place, numbers have increased significantly on the Atlantic Coast of the United States (Hecht and Melvin 2009a; www.fws.gov/northeast/pipingplover, accessed 14 Dec 2016). Increases have not been equal in all regions. For example, in New England pairs increased by almost five times from 184 in 1986 to 918 in 2015, while in the Southern Recovery Unit (Delaware, Maryland, Virginia, North Carolina, South Carolina) the increase was more modest (158 to 362 pairs in the same time period, or slightly more than double). Numbers more recently have increased at CAHA as well, from 2-3 pairs in 2001-2005 to 15 pairs in 2011-2012, averaging 10 pairs from 2006 through 2017. The current population is slightly smaller than the population that existed from 1989-1996 (averaging 13 pairs) before it declined to the levels observed in 2001-2005 (Kwon et al. 2017). The CAHA population constitutes (between 2005 and 2015) 8-24% of all pairs counted in North Carolina, and 1-5% of all Southern Recovery Unit breeders. Population

goals for the Southern Recovery Unit are a consistent population of 400 pairs (USFWS 1996), while goals for CAHA are 15 pairs (maximum known to breed there) for the short-term, and 30 pairs for the long-term (NPS 2010a).

Biology and Management

Piping Plovers arrive on the breeding grounds during March to April, initiating clutches as early as mid-April on the Atlantic Coast, and usually by the first week in May. Renests are common unless the eggs are lost late in the breeding season. Eggs hatch after 26-28 days. Chicks are precocial and can walk and feed themselves from the day of hatching, but initially are brooded and protected by both parents. Chicks fledge in 21-35 days (Elliott-Smith and Haig 2004). On the Atlantic Coast, bayside habitat appears important to chick foraging and perhaps survival (Cohen et al. 2009). Length of time for chicks to fledge, and so become less vulnerable to predators, may be dependent on food availability in this species (Catlin et al. 2013; Gratto-Trevor unpublished data). Many Piping Plovers, at least in some populations, do not breed as yearlings (Gratto-Trevor et al. 2010). Piping Plovers are normally monogamous within a year, although birds may pair with a different mate when renesting, and there are rare reports of polyandry (Haig and Oring 1988b; Amirault et al. 2004). Members of a pair do not normally migrate or overwinter together, and most pairs do not reunite the following year (Haig and Oring 1988b; Wiens and Cuthbert 1988; Friedrich et al. 2014).

Piping Plovers continue to face a number of threats on the breeding grounds, particularly habitat loss and degradation due to coastal development, anthropogenic influences on natural processes that create habitat (i.e., inlet formation and overwash), and oil or contaminant spills. Human disturbance, including high levels of human activity (pedestrians, walking dogs off leash) and vehicle use on nesting, staging, and wintering beaches, is considered an important threat (DeRose-Wilson et al. 2018; Gibson et al. 2018). Greater than historical predation rates, likely resulting from changes in human activity and land-use practices that influence the abundance and behavior of predators, are also thought to have a negative impact on Piping Plover populations (Haig et al. 2005).

Management activities undertaken in the Atlantic Coast in general since the mid-1980s to mitigate threats to Piping Plover on the breeding grounds include activities to reduce human-related disturbance on nesting beaches, such as public education and outreach, signage, symbolic fencing, guardian programs (including enforcing dog leash regulations), and beach closures (Hecht and Melvin 2009b). Activities to reduce predation include nest exclosures and predator removal. Rarely used habitat improvement and mitigation strategies include translocation of nests liable to flooding in severe weather, and captive rearing of eggs/chicks from deserted or flooded nests (Gratto-Trevor and Abbott 2011).

Piping Plovers at Cape Hatteras National Seashore

Current Monitoring and Management. Piping Plovers were first recorded nesting in the Outer Banks in 1901, and have been consistently documented nesting at Cape Hatteras National Seashore since 1960 (Golder 1986; JRW, personal observation). Consistent monitoring of breeding Piping Plovers by National Park Service staff began in 1989. Monitoring includes surveys for nests and breeding plovers beginning in early March, and monitoring of all nests found until the breeding attempt fails or chicks fledge. Beginning in 2008 when vehicle buffer areas around nests became obligatory (see below), nests and broods have been monitored daily and additional data on disturbance of nesting birds have been collected (Kwon et al. 2017).

Breeding season monitoring results in annual estimates of the number of breeding pairs, the average number of fledglings produced per pair, hatching success and fledging success, and other aspects of breeding biology. Most information about the Piping Plover population on CAHA comes from the Park's monitoring program, in contrast to the situation with the American Oystercatcher (see below), and little research into the factors that might drive the variation in demography and population dynamics observed has been conducted. Current ongoing research promises to address this need.

Management for Piping Plovers includes protecting nests and chicks from disturbance from off-road vehicles (ORVs), pedestrians, dogs and other anthropogenic sources. Protective measures changed significantly in 2008 because of legal action directed towards the National Park Service. Prior to that, nesting areas were protected from vehicular and human traffic, but with a variety of closure techniques. Since 2008 protective measures have followed the Cape Hatteras National Seashore Off-Road Vehicle Management Plan, enacted in 2012 (NPS 2010a). The ORV Management Plan requires establishment of protective buffers (i.e., areas temporarily closed to public entry) for pre-nesting areas, nesting birds and chicks, delineated by symbolic fencing and signage. Boundaries for pre-nesting closures are determined before nesting territories are established (i.e., before 15 March), based on historic nesting locations and available suitable habitat. Pre-nesting closures are intended to reduce disturbance that might cause birds to desert a potential nesting territory. These buffers are modified as necessary during the breeding season to reflect locations of nests and chicks. A review required by The National Defense Authorization Act of 2014 (signed in 2015) resulted in a modification of the size of nesting buffers for Piping Plovers. Closures around nests were reduced in diameter from 75 m to 50 m, and those around chicks from 1000 m to 200-500 m for vehicles and 300 m to 100 m for pedestrians. The new policy stated that the 200-500 m buffer for vehicles would apply only if staff could monitor chicks continuously while beaches were open; otherwise, the buffer would revert to the previous level of 1000 m (NPS 2015a).

Since 1994, predator exclosures have been employed on every Piping Plover nest on CAHA to increase nest success. These are placed around nests once three or more eggs are present. Exclosures are approximately ten feet in diameter and made of two x two inch welded-wire fence, anchored by steel rebar (Melvin et al. 1992). The exclosure is topped with three-quarter inch mesh netting to prevent access by avian predators. Nests are monitored daily from a distance, and approached once weekly to inspect the exclosure, determine the number of eggs, and check for signs of predators and abandonment. More intense monitoring is initiated 5-7 days before the expected hatch date, and broods are monitored at least twice a day until they fledge or are lost. A trapping program for potential mammalian predators is implemented as an additional means to increase nesting success when deemed necessary, if staff are available (NPS 2015b).

Population Trends: A Comparison with Cape Lookout National Seashore. Cape Lookout National Seashore (CALO) is close to Cape Hatteras National Seashore but has much less human activity: the former experiences about a quarter the number of visitors as the latter (Mansfield et al. 2011). Thus, it is instructive to compare population trends and productivity between the two areas. Population trends are highly synchronous between the two Seashores. On both, populations declined from the mid-1990s to very small sizes in the early 2000s, and then rebounded to a similar population size as existed in the early 1990s in the case of CAHA, and an even larger population size than existed in the early 1990s in the case of CALO (Figure 6). The two populations have continued to fluctuate in size synchronously in recent years: there is a

significant positive correlation between the number of pairs in the two Seashores during 2005-2016 (Pearson Correlation 0.60, $P = 0.04$). Note that the short-term population goal for CAHA of 15 pairs was achieved twice in recent years, in 2011 and 2012, but the population remains far below the long-term goal of 30 pairs.

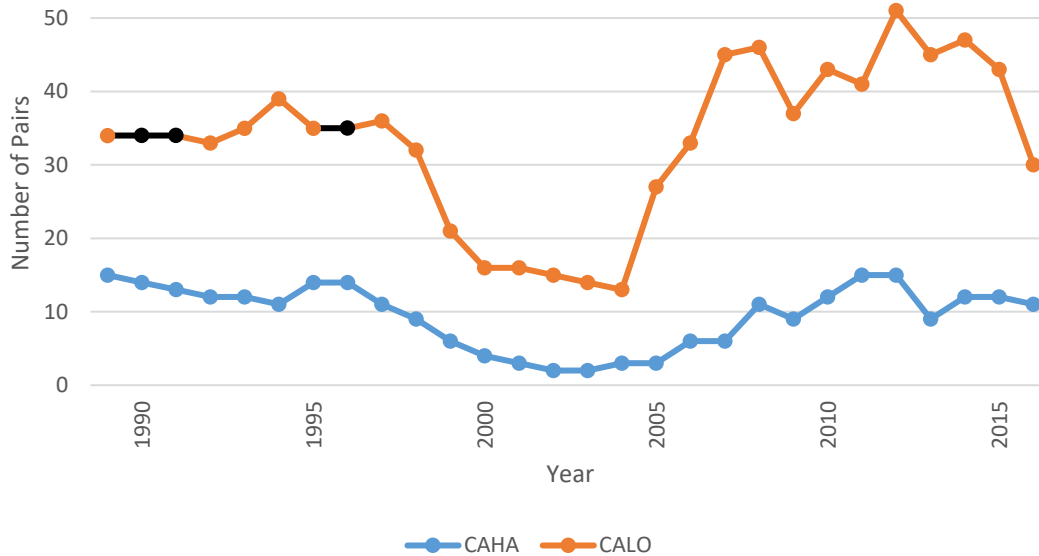


Figure 6. Number of breeding pairs of Piping Plovers at Cape Hatteras (CAHA) and Cape Lookout (CALO) National Seashores, 1989-2016. Adapted from Kwon et al. (2017). The previous year's value is used as an estimate of the number of pairs when no data were available (marked in black).

Over this same period (2005-2016), there is also a significant positive correlation (Pearson Correlation 0.71, $P = 0.01$) between the two Seashores in annual variation in percentage of nests that hatched at least one chick (Figure 7). Since virtually all nests at CAHA are exclosed, whereas almost half (on average) at CALO are not, this result is surprising, and suggests that variation in nest loss is driven more by abiotic factors such as storms rather than predation. The average percentage of nests hatched was reasonably high for a temperate, ground-nesting shorebird, 71% at CAHA and 60% at CALO.

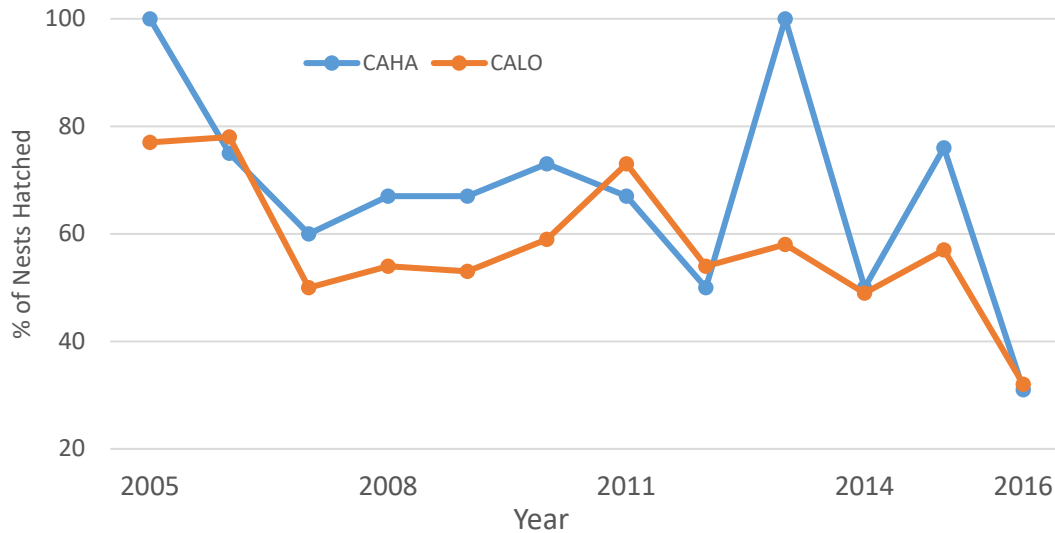


Figure 7. Annual variation in the percent of nests of Piping Plovers hatched at Cape Hatteras (CAHA) and Cape Lookout (CALO) National Seashores, 2005-2016.

Productivity, measured as fledglings per pair, was not correlated between the two Seashores (Pearson Correlation 0.32, $P = 0.30$), nor was there a significant difference between the two in average productivity during 2005-2015 (CAHA mean 0.77 fledglings/pair, $SD = 0.48$; CALO mean 0.64 fledglings/pair, $SD = 0.30$; t -test 0.74, $P = 0.46$) (Figure 8). Average productivity for those years was similar to that observed across North Carolina (0.64 fledglings/pair, $SD = 0.27$), but was far less than in the Southern Recovery Unit overall (1.09 fledglings/pair, $SD = 0.19$) and the entire U.S. Atlantic Coast population (1.16 fledglings/pair, $SD = 0.17$). Productivity goals for CAHA are 1.0 chicks fledged per pair in the short-term, and 1.5 chicks per pair in the long-term (NPS 2010a), well above current levels of productivity. However, evaluating productivity rates and population trends for each Recovery Unit in the Atlantic Coast population, Hecht and Melvin (2009a) concluded that in the Southern Recovery Unit, an average of only 0.93 chicks fledged per pair was needed to maintain a stationary population. Still, the rate at CAHA surpassed even that target only twice during 2005-2016 (Figure 8).

It appears that fledging success rather than nest success is responsible for low productivity of Piping Plovers at Cape Hatteras National Seashore. At CAHA, only an average of 26% of hatched chicks fledged from 2010-2015. This percentage is lower than fledging rates observed in other areas during the same time period (e.g., 49%, Chincoteague National Wildlife Refuge VA, 2011-2015, USFWS 2015; 39%, Assateague Island National Seashore, MD, 2010-2015, J. Kumer unpublished data; 64%, Maine, Zitske et al. 2016; 64%, Eastern Canada, J. Rock unpublished data), suggesting that productivity of Piping Plovers on the Seashore is limited by survival of chicks to fledgling.

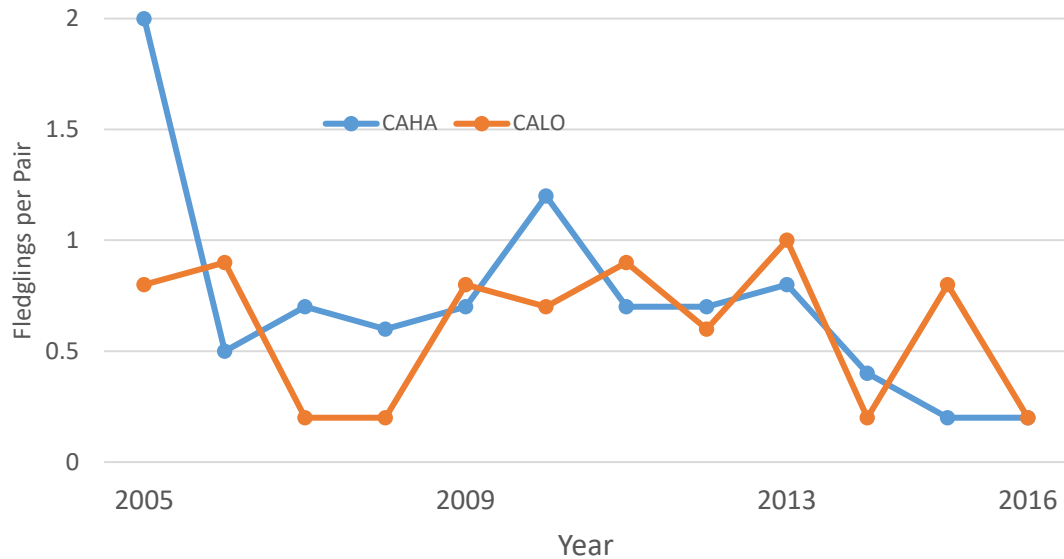


Figure 8. Productivity, measured in fledglings produced per pair, of Piping Plovers at Cape Hatteras (CAHA) and Cape Lookout (CALO) National Seashores, 2005-2016.

Factors Impacting the Cape Hatteras National Seashore Population

The two key questions about Piping Plovers on CAHA are what is limiting breeding population size, and what is limiting productivity. Does the current population size represent the maximum achievable on the Seashore, or might the population be further increased through management? Alternatively, is the current population size sustainable, or might it decline to an even smaller size? Why is productivity consistently much lower on CAHA than elsewhere within the Southern Recovery Unit? Can productivity be increased through management, and if so, by what means? The major declines of the CAHA and CALO populations from the mid-1990s through the early 2000s occurred during a period when the regional population was stable. In contrast, and the subsequent population increases that returned the CAHA population to pre-decline levels and the CALO population to above pre-decline levels by the late 2000s (Figure 6) paralleled a regional population increase (Kwon et al. 2017). Productivity during the period of decline was no lower than it is now on CAHA, but it was lower on CALO (Kwon et al. 2017). In recent years, annual variation in hatching success has been highly correlated between the two Seashores, but annual variation in productivity has not been. In both cases, values are slightly, but not significantly, higher on CAHA. This information provides a context for evaluating the factors that may be affecting the population dynamics of Piping Plovers on CAHA. We will now evaluate these factors and discuss the implications of our findings for the questions articulated above.

Predation. Although exclosures generally are effective in increasing nest success of Piping Plovers (Cohen et al. 2009), this may not be the case at CAHA. The correlation between the proportion of nests hatched at CAHA versus CALO, and the relatively slight difference between the two in average proportion of nests hatched, despite the large difference between the two in the proportion of nests exclosed, suggests that most nest loss is due to factors other than predation by mammals and birds. However, to be confident in this conclusion one must rule out the possibilities that some predation of exclosed nests by mammals and birds occurs, and/or that reduction in predation achieved through nest exclosures is countered by abandonment of

exclosed nests. Desertion of nests after exclosures have been put in place has been a problem in most areas (e.g. Vaske et al. 1994), and exclosures have sometimes resulted in increased rates of predation of incubating adults (e.g., Johnson and Oring 2002; Murphy et al. 2003). Any gain in the population due to increased productivity from exclosures thus can be negated by loss of adults (e.g., Calvert and Taylor 2011). Roche et al. (2010) determined that in most instances of nest desertion one of the adults is dead. At CAHA, exclosures are only left on nests if birds return to incubate within an hour, and thereafter nests are checked often for evidence of desertion or predation. Therefore, it is unlikely that nest desertion is a major issue. Nevertheless, it would be useful to confirm that later desertions are due to death of an adult rather than disturbance caused by the exclosure. This will be easier to determine now that birds are banded (banding was initiated in 2016).

Several authors have recommended using exclosures only at sites where egg predation has been documented as a significant limiting factor, chick fledging success is relatively high, and the species of nest predators have been identified (Melvin et al. 1992; Mabee and Estelle 2000). CAHA does not meet these criteria. Further, we conclude that the available evidence suggests that productivity of Piping Plovers on CAHA is not limited by predation at the egg stage. It follows that exclosures are not an effective tool for meeting management goals.

Causes of nest loss at the egg stage common to both CAHA and CALO include storms and overwash that flood nests, and predation by ghost crabs (*Ocypode quadrata*), which can take eggs inside exclosures. While a study in Virginia concluded that ghost crabs very rarely depredate plover eggs or young (Wolcott and Wolcott 1999), recent analysis of plover nesting success at CAHA concluded that 10% of exclosed nests were depredated by the crabs, and several other nests were abandoned after crab visits (Kwon et al. 2018). As suggested by Wolcott and Wolcott (1999), even if crabs do not often take plover chicks, chick survival may be affected by avoidance of good foraging areas due to the presence of crabs, and attraction of other predators by behavior of plover parents avoiding crabs. This needs further study.

Rather than discontinue use of exclosures altogether, it would be prudent to first test whether rates of nest loss are higher in non-exclosed nests in case similar rates of nest loss on the two Seashores do not in fact reflect similar causes of loss. For example, it could be that CALO experiences more losses to flooding but fewer losses to mammalian and avian predation compared to CAHA. This is a reasonable hypothesis given that greater anthropogenic activity on the latter could support larger populations of species such as raccoons (*Procyon lotor*) and crows (*Corvus* species).

Predator control often is viewed as a simple and obvious method to increase productivity, not only by reducing nest predation, but also by increasing adult and chick survival (Newton 2004; Summers et al. 2004; Bolton et al. 2007). However, predator control often shows little or no productivity or survival benefit for the target avian species, and is much more labor-intensive than it first appears (Gratto-Trevor et al. 2013). At CAHA, predator control has been carried out in some years, dependent on availability of staff, since 2002. One might not expect to see a relationship between predator control and rates of nest loss given the apparent ineffectiveness of nest exclosures (see above), but there might potentially be a relationship to survival of chicks to fledging. Removal of mammalian predators has proven effective in increasing survival of Piping Plover chicks at other locations (Cohen et al. 2009). At CAHA, predator control efforts do indeed appear to be reflected in changes in productivity in American Oystercatchers (see below), but no such pattern is apparent for Piping Plovers. Although the increase in population size on the Seashore corresponds to initiation of predator control, the population increase is not

accompanied by an increase in productivity, and the simultaneous increase in population size on CALO occurred in the absence of predator control. There is actually more linkage between population change, productivity and predator control on CALO than on CAHA, as on the former the population decline occurred during a period of low productivity, and elevated productivity in 2009 and beyond followed a major effort to control predators on CALO in 2008-2009 (Figure 8). Although it is likely that chick survival limits productivity of Piping Plovers on CAHA, there is no compelling evidence that mammalian predation is the cause of low chick survival. Furthermore, even if predation is limiting chick survival and hence productivity, it is clear that removal of mammalian predators as currently conducted is ineffective in alleviating the problem.

Vehicle and Human Disturbance. Little information is available about the impact of anthropogenic disturbance on Piping Plovers at Cape Hatteras National Seashore specifically. Therefore, we must rely on information about such impacts available from other Piping Plover populations, and from another species nesting on CAHA beaches, the American Oystercatcher. Elsewhere, incidences of Piping Plover chicks being killed by vehicles have been documented, and human disturbance has been shown to have detrimental effects on breeding success and chick behavior. Flemming et al. (1988) found that disturbance reduced foraging rates of chicks in Nova Scotia, and that the number of chicks surviving to 17 days of age was negatively correlated with disturbance levels. In a recent study in New York, daily survival of chicks was lower on weekends and related to level of recreational use: chicks in high use areas had a 19% probability of fledging, compared to 82% in low use areas (Derose-Wilson et al. 2018). In addition, 22-day-old chicks were smaller in high use areas (19 g vs 35-37 g) and took four days longer to fledge. This was apparently due to chicks spending less time foraging, attempting to capture prey at lower rates, and being driven away from areas with high prey abundance on weekends when disturbance was high.

Both direct mortalities of chicks due to vehicles and detrimental effects of disturbance on chick foraging behavior have been documented in American Oystercatchers on CAHA (see below). In addition, there are indications that the institution of protective measures beginning in 2007-2008 resulted in improved productivity of this species on CAHA and CALO, possibly contributing to improved population behavior (see below). Modeling of population trends by Kwon et al. (2017) suggests that employment of protective buffers has had a positive effect on population size of Piping Plovers as well, and that higher annual visitation rates are associated with lower population sizes. At CALO, visitation rate was not correlated with population size after protective buffers were employed, whereas it still was at CAHA (Kwon et al. 2017). As with predator control, there is no indication that employment of protective buffers has resulted in higher productivity of Piping Plovers on CAHA, nor is productivity correlated with visitation rate. In contrast, use of buffers is associated with higher productivity in recent years on CALO.

Given these observations, and the adverse effects of human disturbance on breeding Piping Plovers documented elsewhere, it is likely that without protective buffers the population size and productivity of Piping Plovers on CAHA would be lower than they already are. However, just as we concluded for mammalian predation, it appears that something other than human disturbance is limiting chick survival and hence productivity of Piping Plovers on the Seashore.

Habitat. There is evidence from several locations indicating that Piping Plovers populations may be limited by availability of breeding habitat, including at CALO (Robinson et al. 2019). This is suggested by large increases in local populations after hurricanes, other major storms, or

flooding created extensive new nesting and/or foraging habitat, as well as decreases in local populations when habitat was lost to development (Cohen et al. 2009; Catlin et al. 2015; Fraser and Catlin 2018; Robinson et al. 2019). We note that creation of new habitat could result in increases in habitat quality as well as quantity. Population increases resulted from greater productivity in some cases (i.e., due to better foraging habitat), and probably by increased immigration to newly created habitat in other instances (Robinson et al. 2019). Hurricanes and other overwash events produce new plover habitat at CAHA and CALO as elsewhere (Schupp et al. 2013; Robinson et al. 2019), and the population increases on both Seashores began 1-2 years after Hurricane Isabel created substantial new plover habitat (Figure 6) (Kwon et al. 2017). Thus, it is quite possible that habitat limits the size of the Piping Plover population on Cape Hatteras National Seashore, and hence providing more habitat could increase population size. However, management options to do so are severely limited. Anthropogenic attempts to create nesting habitat are expensive and challenging to maintain due to vegetation regrowth; these issues increase with the scale of restoration. Altering barrier island dynamics to promote more frequent overwash events likely would be effective, but would require changes in anthropogenic activities that are well beyond the authority of the National Park Service. Hence, although the short-term population goal of 15 pairs is reasonable and is in fact occasionally achieved, the long-term goal of 30 pairs likely is unattainable under current conditions. It may become more difficult to attain these goals in the future if sea level rise and increased impacts of storms due to climate change reduce beach habitat.

Geographic Range Limits. The North Carolina barrier islands represent the southern limit of the breeding range of the Piping Plover. Range limits may be determined by availability of habitat, or ability to reproduce at replacement rates required to sustain populations due to factors affecting adult survival and/or productivity. Seemingly similar barrier island systems occur south of the current range, so the distribution of the Piping Plover does not appear to be limited by availability of habitat at the southern end of the range. Neither does there appear to be a predator of Piping Plovers that can account for the range limit by virtue of being present, or present in greater numbers, at the southern end of the range and beyond. The northern breeding range limit of another shorebird, the Wilson's Plover (*Charadrius wilsonia*), coincides roughly with the southern range limit of the Piping Plover, a pattern that suggests the possibility that competition between them limits their distributions. The foraging ecology of the two species is sufficiently different that it is unlikely that competition for food with Wilson's Plover is a factor limiting the range of the Piping Plover. The two species do overlap considerably in nesting habitat, such that competition for this resource is conceivable, but there is no evidence supporting this possibility.

There is a geographical gradient in productivity of Piping Plovers, with lowest productivity at the southern end of the range where CAHA is located. Low productivity at the southern end of the range is due specifically to poor survival of chicks from hatching to fledgling, at least at CAHA and CALO (see above). A climate gradient affecting food availability for chicks is a reasonable hypothesis to explain the geographic gradient in productivity and thus low productivity in North Carolina. Kuklinski et al. (1996) found that food for chicks at CAHA appeared to be lower than in other areas of the range, and suggested that poor foraging conditions likely affected chick survival. Chick foraging rate in the best habitat appears to be lower in North Carolina than in New York, further suggesting that there may be more food for chicks farther north (J. Fraser, personal communication). We conclude that poor foraging for chicks is a viable hypothesis for explaining the low productivity of Piping Plovers on the

Seashores specifically, and the southern range limit of the species more generally. Further study of the impact of food availability on chick survival is a critical research need.

Another intriguing hypothesis is that low productivity at the southern end of the range is due to heat stress on chicks (Grémillet et al. 2012). The open beaches of the Outer Banks are a challenging thermal environment, particularly for young birds with limited thermoregulatory and osmoregulatory capabilities. This possibility has been only minimally investigated, but the one study that addressed temperature effects on chick behavior at CAHA found no support for this hypothesis (Kuklinski et al. 1996).

In some temperate species, life history varies with latitude, with more northern populations exhibiting higher productivity and lower adult survival compared to more southern populations (Conner et al. 2001). If Piping Plovers exhibit such a pattern, comparatively lower productivity of Piping Plovers in North Carolina should be accompanied by comparatively higher survival. The limited evidence available indicates this is not the case. For 2015-2016 and 2016-2017, true annual adult survival of plovers breeding in North Carolina (mostly CAHA, Pea Island National Refuge and parts of CALO) was slightly, although not significantly, lower than that of birds breeding in New York (C. Weithman and S. Robinson, personal communication). Since the results for North Carolina are from a small number of years, and numbers of birds marked are relatively low, it would be useful to mark and resight birds in subsequent years to determine if survival is consistently lower in North Carolina than in other segments of the Atlantic Coast population. Nevertheless, clearly it is unlikely that survival of adult Piping Plovers on CAHA is sufficiently high to offset the lower productivity of these birds compared to other populations.

The implication of the survival data, combined with the possibility that foraging conditions for chicks limit productivity, is that the birds on Cape Hatteras National Seashore represent a marginal population at the limits of the species' range. If so, options to increase productivity sufficiently through management to produce population growth may be limited, and both short-term and long-term goals for productivity may be unattainable except in occasional years when conditions are exceptionally good for reproduction. Furthermore, it is conceivable that climate change is exacerbating the causes of low productivity: the ranges of many avian species are contracting at the southern end while expanding at the northern end in response to climate change, due to declining conditions in the former and improving conditions in the latter. Whether Piping Plovers at CAHA are being similarly impacted by climate change is a possibility worthy of investigation.

Conclusions

The Piping Plover population on Cape Hatteras National Seashore experienced a large decline, paralleled by a similar decline in Cape Lookout National Seashore, from the mid-1990s through the early 2000s that was specific to these areas rather than part of any broader, regional decline. Both populations subsequently recovered to previous levels. Population recovery coincided with implementation of new management in the form of protective buffers around plover nests and chicks and removal of mammalian predators, as well as a decline in the number of visitors to the Seashores and creation of new habitat due to the impacts of Hurricane Isabel. It is challenging to separate the impacts of these factors, not only because they are confounded, but also because the CAHA population is sufficiently small for demographic stochasticity to be a major source of variation in demographic parameters and population size. Thus, it is difficult to separate variation that is deterministic from that which is stochastic.

Nevertheless, several conclusions are possible. First, although there is some indication that implementation of predator removal and protective buffers may have resulted in improved productivity of Piping Plovers on CALO, this is not the case for CAHA. It may be that productivity is limited by factors other than predation and human disturbance at CAHA, or that it is more difficult to reduce predation and disturbance to minimal levels due to the number of predators and/or visitors at CAHA compared to CALO. Whatever is the case, it is clear that management in the form of predator removal and protective buffers does not consistently result in achievement of short-term (1 chick per breeding pair) or long-term (1.5 chicks per breeding pair) objectives for productivity of Piping Plovers on CAHA. However, it is also clear that without this management productivity likely would be even lower, and perhaps almost nil.

Second, low productivity on CAHA is due to low chick survival rather than low hatching success. Exclosures used to protect nests on the Seashore may not be cost effective, given that hatching success on CAHA is only slightly higher than on CALO where exclosures are employed much less, and hatching success fluctuates in synchrony between the two. The available evidence does not support mammalian predation or human disturbance limiting chick survival under the current management regime. That foraging conditions for chicks are poor is a viable hypothesis for explaining the low productivity of Piping Plovers on the Seashores specifically, and the fact that these populations represent the southern range limit of the species more generally. If this is the case, current objectives for productivity are unrealistic.

Third, given that preliminary analyses indicate that low productivity of Piping Plovers on the Seashores is not accompanied by relatively high survival, the Piping Plover population on CAHA likely is sustained by immigration of birds from other areas. Thus, the Seashore appears to be a sink for Piping Plovers, even with current management efforts. This is not surprising, given that the birds breeding on the Seashore constitute a peripheral population at the edge of the species' geographic range.

Fourth, it is likely that the size of the Piping Plover population on CAHA is limited by habitat, although it is unclear whether it is at carrying capacity currently. Options for increasing habitat through management are limited, however, as anthropogenic methods to create nesting habitat have only modest potential due to issues of scale, and altering barrier island dynamics to promote creation of more habitat through natural processes would require changes in anthropogenic activities that are well beyond the authority of the National Park Service. Under these conditions, the long-term objective of establishing a population of 30 pairs likely is unrealistic. Over the long term, habitat limitation could be exacerbated by sea level rise and increased intensity of storms due to climate change.

Key Uncertainties and Monitoring and Research Needs. Our conclusions about the current state of the Piping Plover population on Cape Hatteras National Seashore suggest the following key uncertainties and research and monitoring needs:

- To determine whether nest exclosures are cost effective, test whether rates of nest loss are higher in non-exclosures on CAHA in case similar rates of nest loss on the two Seashores do not in fact reflect similar causes of loss.
- Further study of the impact of food availability on chick survival is a critical research need. Comparing areas where productivity differs consistently is one possible approach to investigating availability of food for chicks that might enable further assessment of the roles predation and human disturbance play in chick survival as well. Such studies might suggest more realistic productivity objectives.

- Confirm through marking and resighting adults for additional years that adult survival is consistently lower in North Carolina than in other segments of the Atlantic Coast population and that the CAHA population is sustained by immigration.
- Whether the Piping Plover population on CAHA is limited by habitat, and whether it is at or near carrying capacity, are key uncertainties. Resolving these uncertainties might enable setting more realistic long term population objectives.

American Oystercatcher

The American Oystercatcher, although not listed as Endangered or Threatened, has received substantial attention over the last 15 years due to concerns about its conservation. Large, brightly colored, with a stunning red orange bill, highly visible on coastal beaches and iconic (Figure 9), the American Oystercatcher is one of the first bird species for which a ‘working group’ was created (Simons 2017). The inception of the working group came in 2001, when the U.S. Conservation Plan listed the American Oystercatcher as a species of special concern due to its presumed population size of less than 10,000 individuals (Simons 2017). The American Oystercatcher Working Group, made up of academics, state and federal government scientists as well as scientists from the non-profit sector, was responsible for updating the Birds of North America account for this species in 2013 (Simons 2017). In that updated account, virtually everything known about this species was summarized. Additionally, as a follow-up to a successful targeted symposium at the 2015 Waterbird Society annual meeting, a Special Publication of the scientific journal *Waterbirds* featured 12 papers detailing results of studies examining a wide variety of management issues of the American Oystercatcher both in the continental United States and Mexico. Four of these papers were published specifically on the biology of the American Oystercatcher in North Carolina (Addison et al. 2017; Felton et al. 2017; Schweitzer et al. 2017; Stocking et al. 2017). Much of the research reported in that special symposium was funded by managers hoping to understand the effects of park or protected area management practices on the reproductive success of this species.



Figure 9. American Oystercatcher entering a closed area. Photo by Shiloh Schulte.

In addition to the working group, the American Oystercatcher may be the only bird species for which a Focal Species Business Plan has been created (Johnston 2008). The oystercatcher is also one of very few coastal species for which active management has apparently resulted in an increase in population across its mostly eastern North American range (NABCI 2014). Unlike many shorebird species, for which there are large uncertainties in designation of population change (Elliott et al. 2010; Ross et al. 2012), there is a high degree of confidence in the data supporting this increase. American Oystercatchers can be surveyed efficiently during the non-breeding season using aerial survey techniques, as the species concentrates on open coastal habitats like oyster shell rakes and beaches (Brown et al. 2005). The species, with its highly recognizable white wing stripe, is easily observed and unambiguously identified during overhead flights, so counts from simultaneous ground and aerial surveys are highly consistent (e.g., $r > 0.91$, Brown et al. 2005). As American Oystercatchers feed on bivalves and crustaceans (Clay et al. 2014), they can be considered indicators of the condition of coastal saltmarsh and beach environments (Ogden et al. 2014). Their absence from some coastal environments within their historical range, given available nesting or roosting habitat, suggests potential problems with underlying prey populations.

The American Oystercatcher does not have a federal legal status in the United States. However, five U.S. states with American Oystercatchers have assigned it status with varying degrees of concern (NC, FL: Species of Special Concern; SC: Species of Conservation Concern; GA: Rare; CT: Threatened). There is no official status in Virginia, Maryland, New Jersey, New York, Massachusetts, Maine, or Texas, all states with breeding populations, nor in Nova Scotia, Canada, where there is a very small breeding population. Despite the varying status among state and provincial jurisdictions, the American Oystercatcher Conservation Plan authored by the Working Group (Schulte et al. 2010) lists it as a high priority species due to potential threats from development, recreational activities and changes to predator communities, and threats to coastal environments from rising sea levels. Internationally, the American Oystercatcher is endangered in El Salvador and Mexico, Near Threatened in Guatemala, and Vulnerable in the state of Sao Paulo, Brazil. Globally, the IUCN considers the American Oystercatcher a Species of Least Concern.

American Oystercatchers at Cape Hatteras National Seashore

American Oystercatchers have a sizable breeding population in the Outer Banks region of coastal North Carolina, with about one-third (range 29-49%) of the regional pool of about 95 breeding pairs on CAHA, and the remaining and majority of breeding pairs on CALO (Simons et al. 2017). The number of pairs and nests on CAHA have been recorded since 1999, and total nest numbers as well as hatching and fledgling success have been recorded separately for Ocracoke, Hatteras, Bodie and Green Islands. Similarly, monitoring has occurred separately on the four islands of Cape Lookout National Seashore. Within CAHA, Hatteras Island hosts the largest number of breeding pairs of oystercatchers (9-24), followed by Ocracoke (3-15), Bodie and Green (1-5 pairs each). Within CALO, North Core Banks has 14-39 pairs and South Core Banks 19-33 pairs, while Ophelia, Shackelford and Middle Core Banks host a combined 2-13 pairs (Simons et al. 2017).

Population Trends: A Comparison with Cape Lookout National Seashore. Between the years 2003 and 2013, American Oystercatcher nesting populations appeared to be stable in the state of North Carolina (Schweitzer et al. 2017). Of nesting territories counted on barrier islands

during that statewide survey (Schweitzer et al. 2017), most were within the two Seashores. Pairs nesting on barrier islands made up between 41% (2007) and 51% (2004) of the total state population (Schweitzer et al. 2017). In contrast to the trends across the state, the CAHA population of American Oystercatchers has declined significantly (measured over a slightly longer period, 1999-2016, Figure 10) (Simons et al. 2017). The declines in the CAHA population have not been linear over the study period. During an initial linear phase, which occurred between 1999 and 2006 (breakpoint in the line at 2006 determined via breakpoint regression, Toms and Lesperance 2003), the decline corresponded to an annual loss of 2.9 pairs per year (SE = 0.26). Since 2006 (2007-2016), there has been no significant change in numbers of pairs, suggesting that numbers have stabilized. The Seashore's population goals of 30 pairs in the short term and 45 pairs in the long term (NPS 2010a) correspond to pre-decline levels (Figure 10).

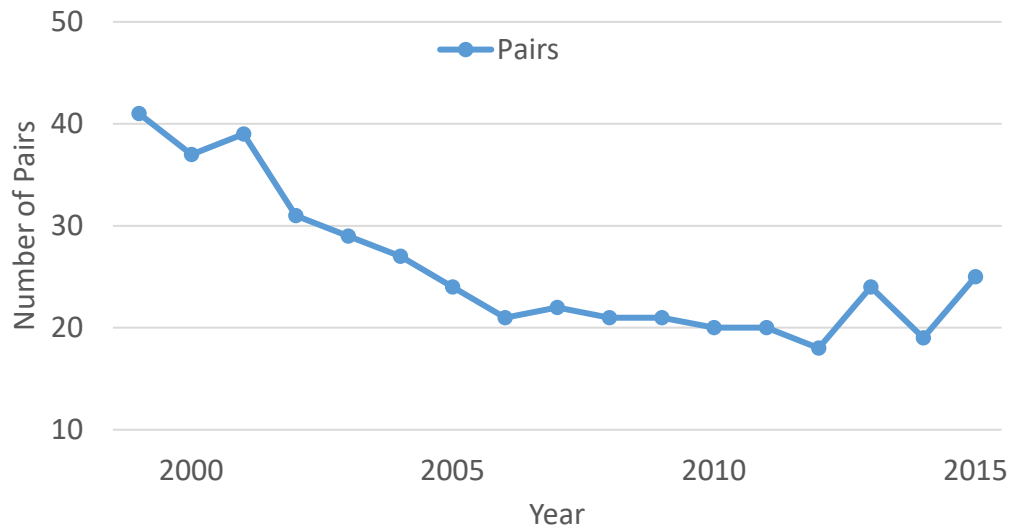


Figure 10. The number of pairs of American Oystercatchers on Cape Hatteras National Seashore (Ocracoke, Hatteras and Bodie Islands only), as a function of year.

Total numbers for CALO have been more stable and are nearly three times greater (annual average of 60 pairs) than on CAHA (annual average of 21.6 pairs) (Simons et al. 2017), despite CALO having less linear extent of shoreline (CALO: 90 km; CAHA: 110 km). This comparison of approximate densities assumes that oystercatcher territories are mostly distributed linearly along the beach, from north to south, rather than east to west (see below). That the numbers are stable across the state (and possibly on CALO) suggests that reasons for decline from 1999 to 2006 were specific to activities and conditions on CAHA.

Current Monitoring and Management. The existing program for managing American Oystercatcher on Cape Hatteras National Seashore consists of annual surveys of the locations of breeding pairs, as well as the collection of a variety of other demographic data, the nature and extent of which has varied since 1994. The research has been a collaborative effort of the North Carolina Wildlife Resources Commission, the National Park Service, USFWS and North Carolina Audubon. Most research on American Oystercatchers on the Outer Banks has been conducted under the direction of Dr. Ted Simons from North Carolina State University. The goals of the research have primarily been to understand the causes of nest failure and chick

mortality, rather than to understand the causes of adult mortality, disappearance of breeding adults, or annual population fluctuations per se.

The basic techniques of study have been to capture breeding oystercatchers using a decoy and noose carpets or whoosh nets, and to band individuals with individual-specific alphanumeric labelled leg bands. An identical readable band is placed on both legs above the tarso-metatarsal joint. The purpose of the identical bands is that a researcher or member of the public who sees the bird, needs to see only one leg to record the identity correctly. This protocol was established by the American Oystercatcher Working Group (Simons 2017) and is used throughout the range of the species within the United States. In addition to monitoring nests for successful hatching, in some years small radio-transmitters have been affixed to chicks to determine both movements and survival. Chicks have also been banded at the nest shortly after hatching and juveniles have been captured prior to fledging, in their territories. Movements of fledglings and adults have been determined using radio-transmitters glued to the feathers just above the tail or affixed to the leg bands. Monitoring of nests has occurred at intervals that vary from 1-14 days depending on researcher objectives.

One of the greatest management efforts for American Oystercatchers on CAHA is protecting nests from human disturbance. Until 1977, there was no attempt by Seashore managers to protect nests and chicks from off-road vehicles. From then through 2007, nesting areas were protected from vehicular traffic, but with a variety of closure techniques. Management of vehicles on the Seashore changed significantly in 2008 because of legal action directed towards the National Park Service. Specifically, beginning in 2008, vehicle buffer areas, delineated by symbolic fencing, were obligatory around known nest sites. The most recent management modification (NPS 2015a) for American Oystercatcher *nests* is that a vehicle corridor may be designated such that ORV's may continue to drive along the water's edge in the vicinity of a nest, but only if there is no other option, and there is preferably 50 m but no less than 25 m between the nest and the vehicle corridor. Such corridors are established for only a small minority of nests (e.g., 5% of 41 nests in 2016, 9% of 53 nests in 2017), and have not resulted in incidences of nest loss to disturbance (e.g., all 6 nest failures near vehicle corridors in 2016 and 2017 were due to flooding or predation). For *chicks*, the beach is closed within a 200 m radius area after hatching (NPS 2015a). Violations of restrictions protecting nests occur frequently. For example, in 2015, 210 pedestrian, 13 ORV, and 29 dog, boat or horse intrusions into closures were reported, a number that is conservative, given that monitoring of nests or closed areas is not continuous (NPS 2015c). On CALO, ORV's can be used to access recreational areas throughout the breeding season of American Oystercatchers (March 1 to December 31, NPS 2014a). CALO institutes a 37 m² closure around active nests and also prohibits park visitors from stopping, parking or camping on the beach within 10 m of the nest (NPS 2006). When chicks hatch, beach vehicles must return to trails behind the dunes, if available, to allow chicks access to the beach without disturbance. CALO managers implement ramp-to-ramp closures where chicks are present. These closures permit pedestrian traffic.

Management for American Oystercatchers on CAHA also includes predator control. Predator control methods were initiated after foxes colonized Hatteras island with a resulting reduction in American Oystercatcher nest success on the island from 27% in 1999-2001 to 3% in 2002 (Schulte 2012). A simulation study of the impact of what is likely the most frequent predator of oystercatcher nests, raccoons, suggested that if managers removed only a minimal number of raccoons annually, American Oystercatchers attracted to the Seashore might experience an ecological trap (Martin et al. 2010). This study suggested that managers remove an

optimal number of 50 female raccoons annually to maintain American Oystercatcher productivity of 0.35 chicks fledged per breeding pair. This value represents the reproductive rate (R_0) necessary for a stable population, derived from data in Schulte (2012). This value is lower than estimates of R_0 in Felton et al. (2017), who suggested a reproductive rate of 0.63 chicks per pair was necessary to achieve population stability. It is also lower than the Seashore's short-term and long-term objectives of 0.4 and 0.5 chicks per pair (NPS 2010a) respectively. The extent of predator control on CAHA has been variable. For example, during 2002-2004 and 2006, 26-38 predators (including 2-21 raccoons) were removed, whereas 94 (including 54 raccoons) were removed in 2005, and 1227 (including 394 raccoons) during 2007-2009, with a peak of 701 (including 150 raccoons) in 2008. Similarly, no predators were trapped during 2015-2017, but 28 raccoons were removed in 2018. At CALO, approximately half the raccoon population was removed from one of the two large islands in the winter of 2008-2009 (Parsons et al. 2013).

Factors Impacting the Cape Hatteras National Seashore Population

The key features of the population dynamics of American Oystercatchers on Cape Hatteras National Seashore over the last two decades are the population decline from 1999 to 2006 and the stability of the population since that time (Figure 10). The period of population decline corresponds to a time of lower productivity (Figure 11), suggesting that the decline was caused by a change in vital rates (i.e., nest loss and/or chick survival), and that the subsequent stability of the population was due to improved conditions for reproduction, brought about through management and/or environmental changes. Indeed, if Martin et al. (2010) are correct that 0.35 chicks fledged per breeding pair is the reproductive rate (R_0) required for population stability (see above), productivity was below that level during the period of population decline and has been above that level during the period of stability (Figure 11). However, although there are signs of population increase in the past few years, clearly the population has not rebounded to the levels observed prior to the decline (Figure 10). Thus, it may be that productivity (or some other vital rate) is still suppressed sufficiently to prevent the population from increasing to fill the available habitat. In this context, it is relevant that the reproductive rate has reached the level necessary for population stability estimated by Felton et al. (2017) in only some years during the period of stability. It is also possible that the population decline was accompanied by, or caused by, habitat loss, and that the current, stable population size represents a new, lower carrying capacity. With these possibilities in mind, we discuss the various factors that may be limiting productivity and habitat availability for oystercatchers on the Seashore.

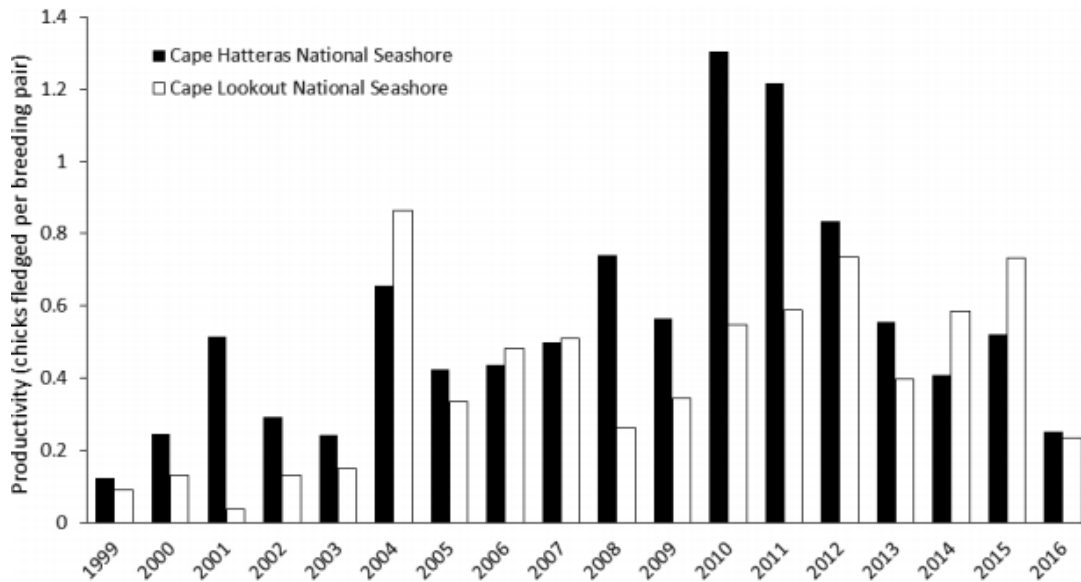


Figure 11. Annual productivity (1999-2016) of American Oystercatcher on Cape Hatteras and Cape Lookout National Seashores measured in fledglings per pair (from Simons et al. 2017).

Predation. Predation is the major cause of nest loss for nesting American Oystercatchers, as it is for most bird species (Martin 1988). Sources of oystercatcher egg loss on the Outer Banks were identified either from tracks in the sand, or in some cases, from cameras set up near the nests (Schulte and Simons 2015; Borneman et al. 2016). For a sample of over 400 nests, over 50% of egg loss was attributable to mammalian predation (probably primarily raccoon, Martin et al. 2010), and another 30% was due to overwash from storms (Schulte and Simons 2015). Hatching rates (that is the number of eggs that hatch from nests where at least one egg hatches) averaged 75% over a 15-year period. This relatively low rate suggests that partial loss of individual eggs from nests is relatively common. The cause of these partial losses is unknown, but disruption of incubation due to disturbance is a proposed mechanism (see below).

Chick survival to fledging, defined as the probability of survival from hatch to age 35 d, averaged 48% from 1999 to 2011 (Simons and Stocking 2011). Sources of chick loss are, in 50% of the cases, unknown. Of those 37 chicks for which the cause of loss was determined (through a radio-telemetry study conducted during 2005-2007, Schulte 2012), predators (including feral cats) were the most important factor but human activities were responsible for 16% (6) of known losses. Overall, the rate of chick production, measured as chicks fledged per breeding pair, was very low in the years prior to 2004 and has been, on average, higher since 2008 (Figure 11). This same pattern has been observed on Cape Lookout National Seashore. Since 2008, productivity has been higher on average on CAHA than on CALO (Figure 11), although the total number of chicks fledged per seashore has been higher on CALO than on CAHA in 14 of the last 18 years (Figure 12). There is a rough correspondence between predator control and subsequent productivity on CAHA. Specifically, the elevation in productivity beginning in 2004 follows the initiation of predator control in 2002-2003, the massive predator trapping effort in 2009 is followed by two years of exceptional productivity, and the very low productivity in 2016 occurs during a period when predator control was discontinued (Figure 11). Thus, that productivity of American Oystercatchers on CAHA varies with predation rate and can be positively affected by predator control are viable hypotheses.

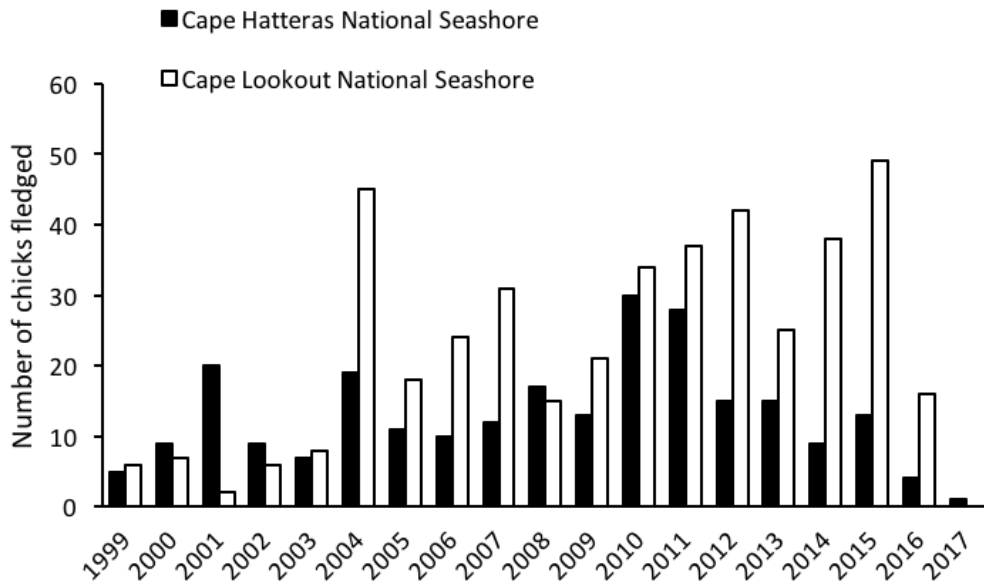


Figure 12. Annual number of American Oystercatcher fledglings produced (1999-2016) on Cape Hatteras and Cape Lookout National Seashores (from Simons et al. 2017).

Unfortunately, with respect to predation, a steady state can never be assumed. In 2017, an invasion of coyotes (*Canis latrans*) resulted in nesting productivity plummeting to zero on both Seashores. The control of these predators is clearly essential for managing American Oystercatchers. Ten coyotes were removed from CAHA in the winter of 2017-2018 but whether this effort was sufficient to reduce the effects of this emerging predator is unknown.

Vehicle and Human Disturbance. Observational studies in which the number of vehicles passing by American Oystercatcher nests are counted, show that as more vehicles pass by nests, the daily survival rate of nests declines by between 2 and 10% per day (Bornemann et al. 2016). Disturbance from vehicles also resulted in (sometimes much) less time on the nest, which is hypothesized to be the mechanism resulting in lower hatching success. Single-passenger, all-terrain vehicles, which are permitted on CALO but not on CAHA, were particularly disruptive. An experimental study showed that passing vehicles caused incubating birds to leave their nests more often than pairs without experimental vehicle passes (Felton et al. 2018). Incubating birds also spent more time in an alert posture during experimental vehicles passes by nests than while undisturbed. Driving within 150 m of the nest influenced the heart rate of incubating adults in one year but not in a second (Felton et al. 2018). Pedestrians also impact behavior of incubating adults, whereas aircraft (including military jets) passing overhead have a negligible impact on nesting behavior or adult stress as measured by heart rate monitors (Borneman et al. 2014). In a study in Georgia, pedestrian activity had a larger effect on the behavior of nesting American Oystercatchers than passing vehicles (Sabine et al. 2008), and caused some nests to fail (Sabine et al. 2006). Borneman et al. (2016) observed similar impacts of pedestrians on oystercatcher behavior on CAHA, but did not have a sufficient sample of pedestrian disturbance events to assess their impact on reproductive success.

Vehicles can be a direct cause of mortality of chicks: vehicles killed a minimum of thirteen chicks between 2003 and 2007 on CAHA and CALO. At CALO, areas around chicks were

closed to vehicles beginning in 2005, and no mortality from vehicles has been documented since that time. The survival probability of chicks doubled when beaches were closed as opposed to vehicle corridors being present (Schulte 2012). These changes in survival likely are partially related to reducing direct mortality, but also closures allow access of oystercatcher parents and their chick to critical feeding areas along the beach and in the intertidal zone (Schulte 2012).

Prior to 2007 when beach closures were implemented to protect nesting oystercatchers at CAHA, vehicles and pedestrians likely affected nest productivity, habitat quality, and presumably, adult stress levels. The impact of vehicles and pedestrians likely contributed to the repeated poor reproductive success associated with the period of population decline, along with high rates of nest and chick predation. Seashore visitation peaked during years when the population was declining rapidly, although vehicle data suggest that vehicle visits were lower during those years (Mansfield et al. 2011). Reduction of impacts of human disturbance likely contributed to improved productivity relative to the period of population decline. Direct mortality from vehicles has been greatly reduced, although appears to still occur in those few areas with vehicle corridors rather than full beach closures. Impacts of human disturbance on behavior that could affect productivity still exist. However, this impact likely is small relative to the impact of predation on productivity. A full population viability analysis is required to determine whether the low productivity prior to 2004 balanced adult mortality during those years and hence whether the combined effects of high predation rates and human disturbance on productivity played a role in the decline of American Oystercatchers on CAHA. A similar analysis is needed to determine whether productivity, despite improvement, remains sufficiently low to account for the failure of the population to recover to pre-decline levels. Suppressed productivity due to impacts of human disturbance alone is an unlikely explanation of the failure of the population to increase, given the effectiveness of beach closures in mediating chick mortality from vehicles, but continued suppression of productivity due to elevated levels of nest and chick predation remains a viable explanation of the lack of population increase.

Habitat. American Oystercatchers on Cape Hatteras and Cape Lookout National Seashores nest primarily on the open ocean beaches, with nests on CAHA clustered around the inlets on Bodie Island, and on the narrow portions of northern Hatteras Island (Figure 13). Few analyses of oystercatcher breeding habitat have been conducted on either Seashore. However, even though the islands have narrowed compared to 2000 and earlier due to interruption of the natural process of dune formation (Gares and White 2005; see above), oystercatchers continue to nest primarily on the narrow beachfronts. Hurricanes, which increase the amount of overwash fans and mostly occur outside the nesting season, have a strong positive impact on nest success that persists for several years, by temporarily diminishing predator populations and/or providing more chick foraging habitat away from human disturbance (Schulte and Simons 2016). In contrast, overwash from storms during the nesting season destroys nests, accounting for 29% of nest loss during 1999-2008 (Schulte and Simons 2015). Oystercatchers presumably are adapted to occasional loss of nests to overwash, and it is unlikely that changes in vulnerability of nests to flooding were a factor in the decline of the population during 1999-2006, or the subsequent improvement in productivity. However, that sea level rise, and perhaps increased intensity of storms during the nesting season due to climate change, could increase the frequency of such losses and thereby impose a new limit on productivity is a concern. Continuing anthropogenic beach stabilization efforts exacerbate this risk.

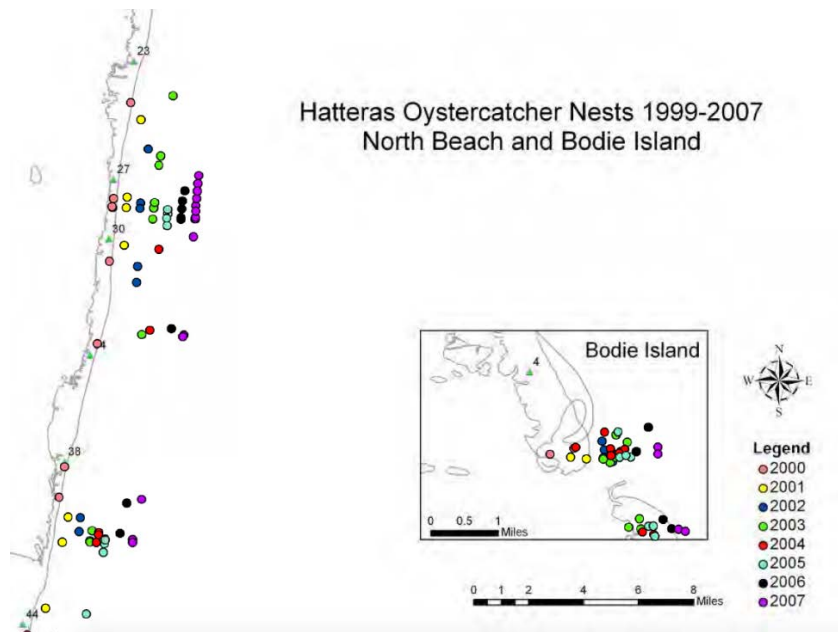


Figure 13. Locations of American Oystercatcher nests from 1999 to 2007 on North Beach and Bodie Island within Cape Hatteras National Seashore (from Simons 2016).

Losses of nests during the population decline on CAHA appear to be concentrated at the Bodie Island inlet where productivity also was especially poor, but this needs to be confirmed by a formal spatial analysis to look at patterns of nest loss across all islands. That there was a negative relationship between the number of nest attempts and nest survival on the Seashore (data from 1998-2008, Schulte 2012) strongly suggests that there was spatial variation in nest success related to habitat, with territories where breeding pairs consistently lost nests exhibiting the lowest productivity. Whether this is true for data since 2008 when management, including beach closures, was more intensive and there were fewer pairs across the island, is unknown. Whether there is a relationship between distance to foraging areas and the probability of disappearance of nesting pairs or individuals (e.g., habitat-specific adult survival estimates, sensu Nol et al. 2012) also is unknown.

In 2015 and 2016, new pairs began to nest on Cape Hatteras National Seashore. Given that there is strong site fidelity and territoriality among breeding American Oystercatchers, this colonization suggests that the population is not at carrying capacity, even though there has been a significant loss of island area (Gare and White 2005). There has not been any significant new habitat formed on CAHA since Hurricane Isabel in 2003 and Hurricane Ophelia in 2005 (T. Simons, personal communication). Thus, the influx of new pairs has occurred in the presence of habitat loss, which suggests that currently habitat is not a limiting factor for breeding oystercatcher populations on the Seashore. Whether reduced carrying capacity due to habitat loss will prevent the population from recovering to pre-decline levels remains to be seen. The current, stable population size is very close to the short-term goal of 30 pairs (NPS 2017). Whether the long-term goal of 45 pairs is attainable and therefore appropriate depends on whether carrying capacity is reduced relative to pre-decline levels. There has not been a full analysis of habitat quality (i.e., habitat factors independent of predation and effects of human disturbance affecting productivity and/or territory size) for American Oystercatchers on CAHA. Reductions in habitat

quality could limit both carrying capacity (through effects of habitat degradation on territory size) and productivity.

Population Dynamics of American Oystercatchers on the Outer Banks

Adult Survival, Productivity and Recruitment. As a result of a massive banding effort (Simons 2016), there are now detailed demographic data with which to determine whether the reproductive success experienced by American Oystercatchers on the Outer Banks is sufficient to result in a stable population (e.g., Felton et al. 2017, Wilke et al 2017). According to a population viability analysis using data from a sample of 700 nests from CALO (Felton et al. 2017), given an observed annual adult survival of 0.92 (Schulte 2012), a combination of nest success*chick survival > 0.42 is the level of success necessary for a stable population. This estimate of productivity translates to be 0.63 chicks fledged per pair per year (Felton et al. 2017). It appears feasible to achieve these levels of nest success and chick survival through management, and indeed this level of productivity has been achieved and exceeded on Cape Hatteras National Seashore in some years since the population stabilized. However, this level of productivity has not been achieved since 2012 (Figure 11).

Productivity generally is influenced by local factors, and indeed, there are consistent differences in productivity between localities. Reproductive success has been extremely low on Bodie Island (e.g., Bodie Point, Appendix A in Schulte 2012). Green Island pairs, by contrast, produced one chick per pair (above the critical level for population stability) in four of five years (Schulte 2012). It may be significant that Green Island is located within the sound and thus is relatively protected from predators and overwash, as well as rarely being visited. On CALO, Middle Core Banks, which is frequently visited, has had consistently high productivity. By contrast, on Shackleford Banks, despite its isolation and small visitor populations American Oystercatchers have experienced very high nest loss, suggesting egg predators are severely limiting reproductive success. Thus, while one may be tempted to conclude that islands that are inaccessible to humans have the highest productivity, this is only sometimes true. The islands with high productivity on the two Seashores are small, with a small number of pairs, which, therefore, limits their ability to contribute new recruits to the larger Outer Banks population. A broader analysis of nest loss and chick survival across a larger sample of barrier islands of the southeastern U.S. coast may help to better understand the range of factors influencing reproductive success of American Oystercatchers.

American Oystercatcher chicks have been banded since 2002. Not surprisingly, given low productivity, a small number of chicks banded, and an age of maturity of nearly 4 years (Felton et al. 2017), the number of banded chicks recruited as breeders was small through 2006, but increased thereafter (Figures 14-15). Chicks banded on the Seashores have been recorded breeding as close to their natal nest site as 2.9 km, and as distant as 90.7 km (Schulte 2012). Many chicks banded on the two Seashores returned to breed on the Seashores (Figures 14-15), thus much (but not all) recruitment appears to be internal. Recruitment of new breeders to CALO has been much greater and more consistent than recruitment to CAHA (Figures 14-15). Correspondingly, chicks that hatched from CALO have recruited at higher rates than chicks hatched at CAHA. That some individuals hatched from locations outside the two Seashores have recruited to the Seashores suggests that the Outer Banks birds are part of a meta-population.

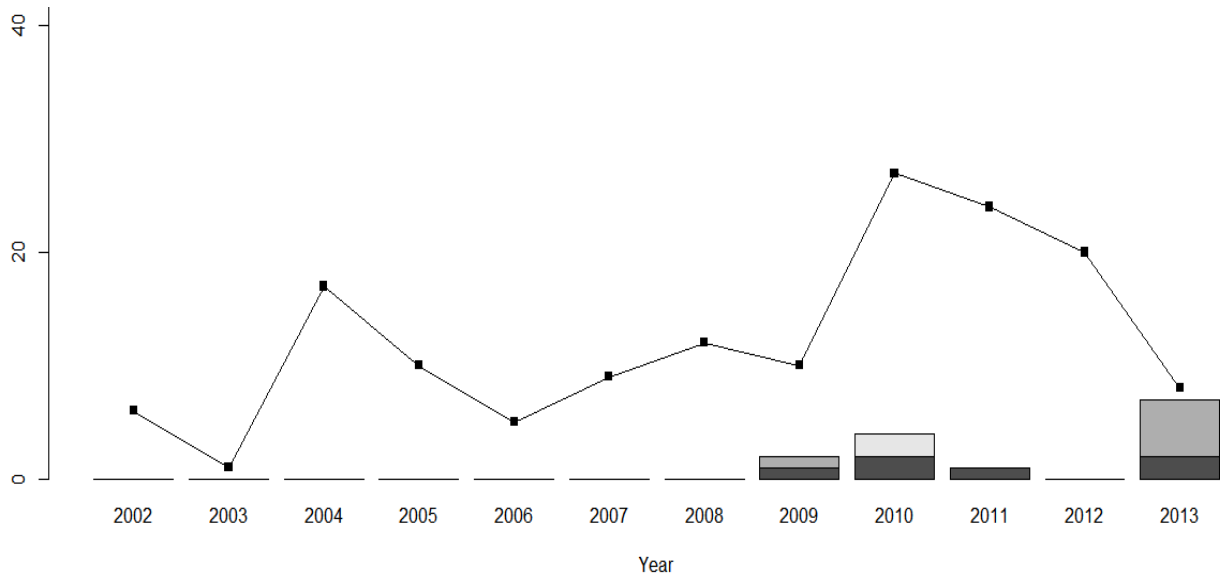


Figure 14. Number of American Oystercatcher chicks banded at Cape Hatteras National Seashore and their breeding locations. Solid line indicates the number of chicks banded each year and the bars indicate the number of those chicks subsequently observed breeding, by location (black = CAHA, dark gray = CALO, light gray = other) (from S. Felton unpublished).

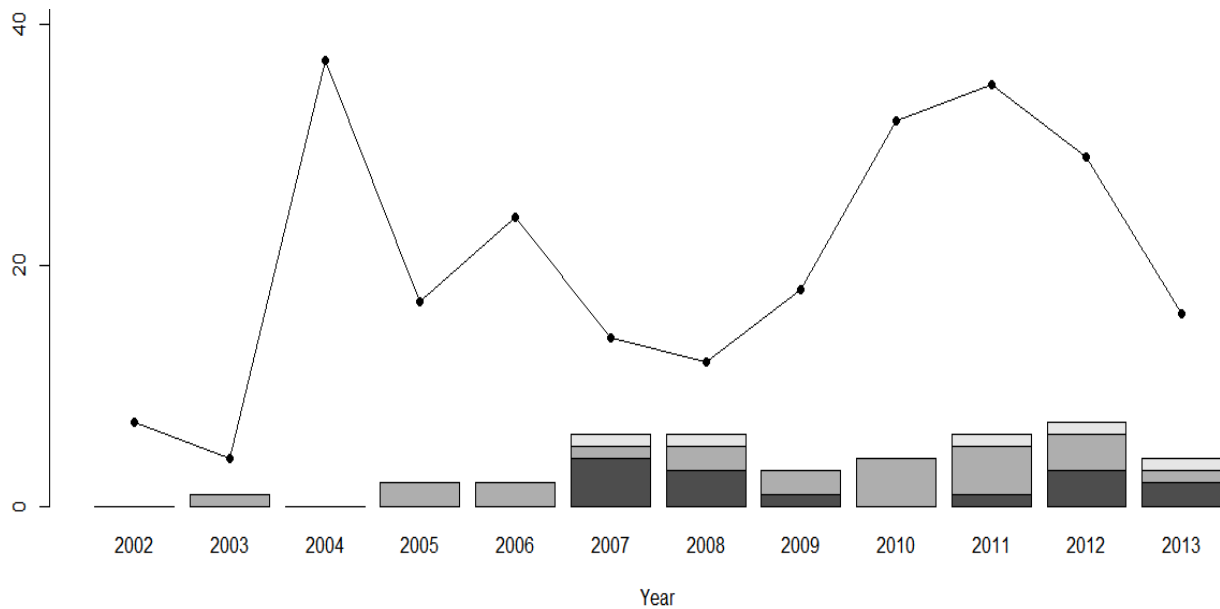


Figure 15. Number of American Oystercatcher chicks banded at Cape Lookout National Seashore and their breeding locations. Solid line indicates the number of chicks banded each year and the bars indicate the number of those chicks subsequently observed breeding, by location (black = CAHA, dark gray = CALO, light gray = other) (from S. Felton unpublished).

American Oystercatchers have high site fidelity so return rates corrected for re-sighting probabilities using mark-recapture techniques (e.g., Nol et al. 2012; Schulte 2012; Murphy et al. 2017) can provide relatively accurate measures of adult survival. Survival estimates of migratory and non-migratory populations, and using two different analytical techniques, are all very similar (0.81 to 0.94, Murphy et al. 2017). Of the two techniques, the Barker method (Murphy et al. 2017) has advantages over the CJS method (Nol et al. 2012) in estimating survival of oystercatchers as the Barker method incorporates movement data. For example, if a bird has not appeared at its traditional nesting site, but is observed somewhere in a flock during the winter months, the Barker method will allow inclusion of the probability of movement and re-sighting of that individual in the final survival estimate. This is important, as many nesting pairs have disappeared from the CAHA population. The Barker method enables one to account for the probability of movement and thus estimate the true mortality rate. The CJS method does not distinguish between permanent emigration and mortality and thus all birds that disappear are counted as deaths (Sandercock 2003).

In American Oystercatchers, population growth rate is most sensitive to changes in adult survival (Wilke et al 2017). Thus, understanding why and whether adults leave the breeding population (either through death or dispersal) is important to understanding the underlying problem of sustaining American Oystercatcher populations on the Outer Banks. A study of transition probabilities between Virginia barrier islands, making use of re-sightings from across that state, indicates virtually no immigration or emigration of barrier island adults (Wilke et al 2017). This finding supports the assertion that the loss of nesting adult birds during the 1999-2006 population decline on CAHA and CALO was due to adult mortality rather than permanent emigration out of the study area. Similarly, Schulte (2012) never recorded movement of a breeding individual more than two territories away from the original breeding site, but acknowledges that it would be naïve to assume that there is absolutely no movement (Schulte 2012, p.121). Wilke et al. (2017) report that numbers of breeding pairs in seaside lagoons in Virginia declined while barrier island populations increased, suggesting possible movements of breeding adults. A careful scouring of the database of winter re-sightings, of which there are thousands on the AMOY Working Group web site (Simons 2017), may determine whether any of the adults that have been lost from the two Seashores have been seen elsewhere subsequently.

The observed rate of loss of breeding pairs on Cape Hatteras National Seashore of 2.9 pairs per year during the 1999-2016 decline represents a loss of nearly 35 breeding adults from a starting population of about 82 birds (41 pairs). This rate of loss is equivalent to an annual survival probability of 0.93%, well within the range of survival estimates measured through various mark-recapture analyses (Nol et al. 2012; Murphy et al. 2017). Therefore, these data are consistent with the hypothesis that the decline was due to lack of local recruitment of new breeders during a period of low productivity rather than elevated adult mortality. However, what is unusual about these losses is that both members of the pairs appear to have disappeared together. As other species of shorebirds show evidence for increased dispersal of pairs that have been unsuccessful (Flynn et al. 1999), an alternative explanation is that both members of the pair dispersed due to repeatedly low reproductive success. Supporting this hypothesis is the unlikelihood that both members of a pair would die due to natural or age-related mortality at the same time. Whether unsuccessful pairs move could be tested, as suggested above, by examining the existing database for records of movements of breeding adults. Determining whether particular breeding locations result in particularly poor survival (e.g., habitat-specific survival

estimates, Nol et al. 2012) would be useful in evaluating the possibility that “normal” (albeit spatially variable) survival can account for the observed population decline.

Stochasticity and Demographic Rescue. It appears that statewide, American Oystercatchers have relatively stable populations in North Carolina (Schweitzer et al. 2017). Regionally, the Outer Banks population of American Oystercatchers is very important to the population in the State of North Carolina, comprising up to 50% of the total. However, dispersal of adults between the Outer Banks and other areas in North Carolina is low, and recruits to the Cape Hatteras and Cape Lookout National Seashore populations are coming primarily from within the two Seashores (Figures 14-15), although a small number of recruits from outside the area will probably continue to supplement the Outer Banks populations. Nonetheless, if conditions on the Outer Banks could be made more conducive to promoting nest success and chick survival, this might increase the probability of adding new pairs to the population, especially to CAHA, reducing the vulnerability of the population to stochastic events. Reproductive success has been very low on the smallest islands within the Seashores and, not surprisingly, many pairs have been lost (e.g., Bodie, Green Islands have only two pairs in total as of 2016, Simons et al. 2017), suggesting that demographic and environment stochasticity are impacting local populations. That the total population on CAHA declined to < 20 pairs in 2009 (Simons et al. 2017) suggests that the CAHA population is at extreme risk from demographic or (more likely) environmental events (e.g., disease outbreaks, major predation events) that might reduce this population to the point of no return. The CALO population is larger and is producing more potential recruits than the CAHA population. Potentially, immigrants from CALO could rescue the smaller populations on any of the CAHA islands from declines or local extinction. This assumes that the amount of suitable habitat, and thus the carrying capacity of the population, remains stable. Given that the Seashore populations do not yet appear to be at carrying capacity (see above), the population goals for the two Seashores need not simply be to maintain the population sizes that currently exist, but to plan for future increases and allow enough protected, suitable habitat to support recruits into the breeding population and thereby enable population increase. The CAHA short-term and long-term population goals of 30 and 45 pairs respectively capture this well in that the former represents the current population and the latter the desired, future, larger population.

Population Viability Analysis. The research conducted on the Outer Banks on American Oystercatchers has been both extensive and intensive, sufficiently so that there are enough data to conduct a Population Viability Analysis (PVA) and put together scenarios indicating whether reproduction is sufficient to balance adult mortality (Felton et al. 2017). Although data on the rate of transition from sub-adult to adult in this excellent modeling exercise were taken from studies of the Eurasian Oystercatcher, a more recent unpublished analysis suggests that management per se does not affect the rate of transition from sub-adults to breeding individuals (S. Felton unpublished data). Additionally, the Eurasian Oystercatcher numbers used in the PVA for the rate of transition from sub-adult to adult (Felton et al. 2017) are close to those obtained from recent data on American Oystercatcher (between 0.17 and 0.22, S. Felton unpublished data). Results regarding the level of productivity needed to produce a sustainable population are unlikely to change markedly with small differences in these transition rates (although see Wilke et al. 2017 for caution on using fixed threshold levels without consideration of rates of immigration and emigration). The results of this population viability analysis indicate that the management actions associated with the transition from a declining to a stable population,

although necessary, do not appear to be sufficient to maintain the level of productivity required for population growth, ($R_0 > 0.63$, Felton et al. 2017). The required level of productivity has been achieved only in 4 years (2008, 2010-2012) on CAHA and 2 years (2004, 2012) on CALO (Figure 11).

Conclusions

The causes of the steep decline of the American Oystercatcher population and associated lack of recruitment during 1999-2006 on Cape Hatteras National Seashore are not entirely clear. However, given the evidence for direct vehicle impacts and extensive predation, we conclude that the impacts of human disturbance and predation on productivity likely drove this decline. Furthermore, improvements in use of pedestrian and vehicle closures to protect nesting birds from human disturbance, as well as implementation of predator removal, likely are responsible for subsequent improved productivity and associated population stability. Although the population has been stable since 2006, it has not recovered to pre-decline levels. The most likely explanation of this, revealed by the recent population viability analysis (Felton et al. 2017), is that productivity is not consistently high enough to support population growth. If this viability analysis is accurate, then the Seashore's long-term goal for productivity (0.5 chicks per pair) is too low to support the population growth required to meet the long-term goal for population size (45 pairs).

It is unlikely that human disturbance is limiting productivity to levels below those required for population growth, given the effectiveness of management in limiting direct mortality of nests or chicks, unless there are a significant number of violations of closures. It is much more likely that predation is limiting productivity to levels below those required for population growth, and predator removal appears to be sufficiently effective that its sustained employment could conceivably raise productivity to the required level. The possibility that predation is limiting productivity could be further evaluated through comparison of rates of nest loss and chick survival to those in other populations with higher productivity. Finally, that loss of nests to overwash, already the second leading cause of nest loss, may increase as seas levels rise and storms become more intense, and thus emerge as an important limitation on productivity is a possibility that should be monitored.

Another possibility to consider is that habitat quality is limiting productivity, especially the possibility that foraging conditions for chicks are poor. That the impacts of human structures on the processes of inlet formation and overwash have resulted in degradation of nesting habitat for American Oystercatchers on CAHA is feasible, for example by affecting the distance between nest locations and feeding areas for chicks. A landscape analysis of variation in nest loss and chick survival in relation to change in beach habitat could be used to examine this hypothesis. Still, the lack of difference between the two Seashores in productivity, given the substantial difference in number of human structures between them, speaks against this hypothesis. One expects whatever factor limits productivity to be shared between the two Seashores.

Understanding how adult survival varies across the two Seashores or across habitats is a key uncertainty, and the data with which to test this appear to readily available. Adult survival does not appear to be limiting oystercatcher populations on the two Seashores, but such a landscape analysis of variation in adult survival might provide insight into variation in habitat quality, and into the phenomenon of the disappearance of breeding pairs.

That carrying capacity rather than productivity is currently constraining the growth of the oystercatcher population on CAHA is unlikely, given the increase in population size (Figure 10)

that followed a period of high productivity (Figure 11). However, it is quite possible that reduced carrying capacity due to habitat change will eventually preclude an increase to pre-decline levels. The effects of human structures on barrier island dynamics appear to have impacted the quantity, as well as quality, of habitat for American Oystercatchers as there has been no significant increase in habitat on CAHA since 2003 (Hurricane Isabel) and 2005 (Hurricane Ophelia) and most new habitat is lost to road and dune building (T. Simons, personal communication). If habitat will indeed ultimately be limiting, increasing habitat, which might occur with a reduction in annual road and artificial dune maintenance on CAHA (Schulte and Simons 2016), may increase the probability of attracting recruits and thereby stimulate population growth.

We conclude that current policy on beach closures is sufficient and likely necessary to avoid significant adverse impacts of recreational use of beach habitat on productivity of American Oystercatchers on Cape Hatteras National Seashore, provided that vehicle corridors continue to be only rarely permitted. We further conclude that predation and/or habitat quality limiting productivity and habitat quantity limiting carrying capacity are hypotheses worthy of further investigation. Specific lines of further inquiry are detailed below.

Key Uncertainties and Monitoring and Research Needs. Our conclusions about the current state of the American Oystercatcher population on Cape Hatteras National Seashore suggest the following key uncertainties and research and monitoring needs:

- While population growth has relatively low sensitivity to juvenile survival (Wilke et al. 2017), factors influencing it are poorly known. Using the AMOY Working Group data set, these factors can be explored to determine whether management has any capacity to improve juvenile survival.
- Test, using spatially explicit methods, whether losses of nesting pairs, losses of nests or low chick survival are concentrated on specific habitats or islands with particular features (e.g., distance to inlets, width of beaches, distance to foraging areas, size of island, management). These analyses probably can be achieved with existing data. Using data from across barrier islands in North and South Carolina, Virginia and Georgia would provide more statistical power to detect major influences.
- Determine whether violations of closures contribute to nest loss and/or chick mortality.
- Determine whether there is any evidence of movement of breeding pairs due to repeated nest failure.
- Calculate juvenile and immature survival rates for the CAHA and CALO populations using data collected under the auspices of the AMOY Working Group, to replace data derived from European Oystercatchers in demographic models (Felton et al. 2017). To assist this effort, record annually the presence and identity, if known, of all non-breeding birds on or near territorial individuals.
- As advocated by Martin et al. (2010), employ adaptive management for predator removals. Reporting on the annual level of predator removals would be desirable for understanding variation in productivity, and whether impacts of predators on productivity may be limiting population growth. We acknowledge that there is some sensitivity associated with this information.
- Determine if there is any evidence that any island in either Seashore has reached carrying capacity for American Oystercatchers (e.g., Middle Core Banks), thus limiting population growth.

- Report annual productivity consistently as number of chicks fledged per adult pair in addition to reporting total number of fledged chicks and rate of nest loss. Develop a range of values of R_0 that are necessary to achieve stable American Oystercatcher populations under various estimates of adult survival and rates of immigration and emigration (Wilke et al. 2017).
- Regularly re-assess vital rates of American Oystercatcher on the Seashores. This will require continued monitoring as well as banding of newly recruited, unbanded breeding birds.

Shorebirds in the Non-breeding Season

Although much of the focus of shorebird management has been on protecting beach-nesting species, the beaches of Cape Hatteras National Seashore provide important habitat for shorebirds during all seasons. The Outer Banks are an important migratory stopover for many migrating shorebirds and as a result have been designated a Globally Important Bird Area by the American Bird Conservancy (National Audubon Society 2017). The Seashore hosts a diverse shorebird community across the entire annual cycle and supports globally important numbers of several species (Dinsmore et al. 1998; Manning 2004). In addition to the Piping Plover and American Oystercatcher, the federally Threatened Red Knot (*Calidris canutus*) (U.S. Department of the Interior 2014), the Wilson's Plover, a Species of Special Concern in North Carolina (North Carolina Wildlife Resources Commission 2014), and several species with heightened conservation status under the U.S. Shorebird Conservation Plan (Brown et al. 2001) use the Seashore.

There are no detailed studies of breeding shorebirds other than Piping Plovers and American Oystercatchers on CAHA, although DeRose-Wilson et al. (2013) studied breeding Wilson's Plovers at nearby Cape Lookout National Seashore. Current management efforts have unknown impacts on this species, which has always been a rare breeder at CAHA. The other breeding shorebirds of possible interest include the Eastern Willet (*Tringa semipalmata semipalmata*), which is uncommon as a breeding species in summer and as a wintering species, but occurs more commonly as a spring and fall migrant, and Black-necked Stilt (*Himantopus mexicanus*) and Killdeer (*Charadrius vociferus*), both of which are rare breeders (Dinsmore et al. 1998). Thus, the Seashore is not an important breeding location for shorebirds other than Piping Plovers and American Oystercatchers; hence, management is appropriately focused on these two species during the breeding season. Oystercatcher numbers are greatly reduced in the nonbreeding season as breeding birds depart. For Piping Plovers, on the other hand, CAHA serves as an important migratory stopover, and to a lesser extent wintering site, for birds breeding elsewhere as well as supporting a breeding population. The Seashore serves as a migratory stopover and wintering site for a variety of other shorebirds as well.

Piping Plovers

A large number of Piping Plovers use Cape Hatteras National Seashore during the nonbreeding season. Numbers appear to be greatest in late summer and early fall: at Ocracoke South Point in 2016, 94 to 188 birds were counted in single surveys between 22 July and 30 September, with a peak (188 birds) on 26 August (Weithman et al. 2018). Fifty different previously banded birds were observed. These banded birds were primarily from the Atlantic Coast and Great Lakes breeding populations, including 14% of all Eastern Canada marked birds known to be alive in 2016. Weithman et al. (2018) estimated that 569 Piping Plovers,

representing 10% of the global population and 15% of the Atlantic Coast population of the species, used South Point during fall migration in 2016. Lower numbers of Piping Plovers were counted at other sites, including Ocracoke North Point (12-20 birds), and Green Island (3-17 birds). Similar numbers of plovers have been observed using the Seashore during fall migration in subsequent years. Clearly, CAHA is an important migratory stopover for Piping Plovers, and in particular, Ocracoke South Point may be the single most important migratory stopover for the species (Weithman et al. 2018).

Numbers of Piping Plovers overwintering (1 November through 28 February) at CAHA are lower, but are a substantial portion of all North Carolina wintering plovers (23 and 22 counted during the 2006 and 2011 International Piping Plover Winter Census, compared to 84 and 43 for all of North Carolina; Elliott-Smith et al. 2009; 2015). Thus, CAHA plays a role in the conservation of Piping Plovers as a wintering area comparable to its role as a breeding area, providing for a small portion of the Atlantic Coast population. Both roles are considerably less than the Seashore's role as a stopover site during fall migration.

Surveys of nonbreeding Piping Plovers on CAHA began in 2000, and were modified to follow the Nonbreeding Piping Plover survey protocols established by the Southeast Coast Inventory Monitoring Network's Migratory and Wintering Shorebird Monitoring Study (Byrne et al. 2009) beginning in 2006 (Kwon et al. 2017). According to the protocol, surveys are to be carried out weekly from April through March, excluding the breeding season. Surveys consist of high (carried out weekly) and low (carried out monthly) intensity transects along the ocean shoreline. Most transects established on CAHA are one mile in length, and the species counted included Piping Plovers, American Oystercatchers, Wilson's Plovers, Red Knots, Black-necked Stilts, Whimbrels (*Numenius phaeopus*), and Sanderling (*Calidris alba*). Transects were limited to spits and inlets where shorebirds are concentrated until 2011 when the entire shoreline was incorporated into the sampling scheme. In some years some transects were not surveyed, or not all scheduled surveys of some transects were conducted, due to lack of available staff (Kwon et al. 2017). These surveys document a similar distribution of Piping Plovers in the nonbreeding season as that described by Weithman et al. (2018) with the largest concentration at Ocracoke South Point, smaller concentrations at Bodie Island spit, Cape Point and North Ocracoke, and few birds along the shoreline between inlets (NPS 2015a).

Recent research indicates that condition and survival of migrating and wintering Piping Plovers is reduced in areas of high anthropogenic activity, including both shoreline modification and disturbance by humans, dogs and vehicles (Gibson et al. 2018). Although the areas used by migrating and wintering plovers on CAHA are subject to relatively little shoreline modification, they may be subject to disturbance. To protect nonbreeding sites at CAHA, winter closures are established in primary plover foraging habitat, and permanent vehicle free areas also provide some protection. Vehicle-free areas exist in all four locations where Piping Plovers are concentrated in the nonbreeding season, although seasonal ORV routes reduce their extent at the end of the migratory period (i.e., starting in mid-September) at Bodie Island and Ocracoke South Point. Determining whether current protections are adequate requires an assessment of the extent to which closures and vehicle-free areas are effective in reducing human disturbance to low levels in key areas, especially Ocracoke South Point, during plover migration. An assessment of whether closures in areas other than the four in which the birds concentrate that consistently have very low numbers of birds are cost effective, at least with respect to protecting Piping Plovers, might also be in order.

With increased numbers of marked Piping Plovers on the Atlantic coast and CAHA, band-reading surveys at sites of high abundance would provide considerable information on use of the Seashore by local and nonlocal plovers. Including band reading in the nonbreeding season Piping Plover surveys could produce data of great value to the Piping Plover recovery program.

Other Shorebirds

Much has been learned about the ecology of the two focal shorebird species (Piping Plover and American Oystercatcher) on CAHA. However, the rest of the shorebird community has received less attention, and for many species, even a basic understanding of seasonal patterns of distribution and abundance is lacking. A study conducted in the 1990s on CAHA and CALO was the first attempt to document these patterns (Dinsmore et al. 1998). This study highlighted the importance of the two Seashores to >20 species of shorebirds and documented that the Seashores are a globally important stopover site for migrating Whimbrel and Sanderlings. Burger et al. (2012) studied the stopover ecology of Red Knots (subspecies *rufa*) along the entire Atlantic coast and concluded that the two Seashores represent one of seven key stopover sites for this subspecies. Other shorebird species found in significant numbers on the Seashores during migration include Willets, Black-bellied Plovers (*Pluvialis squatarola*), and Ruddy Turnstones (*Arenaria interpres*).

The detailed shorebird surveys conducted by Dinsmore et al. (1998) in 1992-1993 showed clear seasonal patterns of occurrence for all shorebirds at CAHA. Shorebird numbers are lowest in January to March, rapidly build to a peak in May representing spring migration, taper off in June, and then peak again during fall migration in July and August, followed by a steady decline in numbers through December. Sanderling and Black-bellied Plovers drive this pattern (Dinsmore et al. 1998). American Oystercatchers exhibit a similar decline in numbers from August through December, and absence in January and February, but in this case, the pattern represents the departure of breeding birds. Red Knots exhibit a similar pattern as well, but in this species the fall migration peak was earlier in June, and a relatively constant rather than declining number of birds was observed July – December. Thus, migrating birds of these species can linger into December, whereas those of other species move through much more quickly, as all species do in spring migration. True wintering birds are present only in January to mid-March, although labeling Sanderlings, Black-bellied Plovers and Red Knots migrants does not adequately reflect the dependence of these species on the Seashore, considering that some individuals may spend six of the nine months between the breeding season and the subsequent spring migration using the habitats the Seashore provides. Stopover duration for Sanderlings in spring (mean = 5 days; Dinsmore and Collazo 2003) and fall (mean = 20 days; S. Dinsmore unpublished data) on Portsmouth Island within CALO represent the only available information on how a particular species uses the two Seashores during migration.

Many thousands of shorebirds visit the Outer Banks in both spring and fall (Dinsmore et al. 1998). Why the two Seashores are heavily used by migrating shorebirds is not well understood. Migratory shorebirds use stopover sites for foraging and loafing, and there may be added benefits with respect to minimizing disturbance and predation (Myers et al. 1987; Skagen and Knopf 1994; Forgues 2010). Linking shorebird use of CAHA to specific attributes such as food resources or disturbance would enable land managers to integrate the needs of migratory shorebirds with other land uses. To date, the stopover ecology of migratory shorebirds on the Outer Banks has been little studied. Dinsmore and Collazo (2003) studied Sanderlings on Portsmouth Island within CALO during spring migration and were unable to link local survival

rates to changes in body mass or body condition. Residency probabilities were high through late May and then dropped as the bird departed north.

Clearly, CAHA is an important migratory stopover site for a number of shorebird species. Its importance as a wintering site is less clear. Most shorebird species are absent or nearly so during January – March. However, the Red Knot may be an exception. Although Dinsmore et al. (1998) did not find any indication of a wintering population of knots in the early 1990s, in more recent surveys by the Park Service (see above), more knots were counted during November – March than during July – October (NPS 2015a). These wintering birds were concentrated on the south-facing beaches near Cape Point and on Ocracoke Island. Dinsmore et al. (1998) also noted that Ocracoke was the most important site for knots within CAHA. North Core Banks on CALO is another important site for Red Knots (Dinsmore et al. 1998).

Other non-breeding shorebirds are distributed throughout the Seashores, although some locations are used more than others are. Dinsmore et al. (1998) noted that migratory shorebirds were most numerous on the open beach between Cape Hatteras and Rodanthe and least abundant on beaches between Cape Hatteras and Hatteras Inlet and on Bodie Island. Patterns for individual species varied. Sanderlings, Black-bellied Plovers and Willets all were most numerous on the open beach between Cape Hatteras and Rodanthe (41%, 32% and 29% of total species counts respectively) and on North Core Banks on CALO (20%, 27% and 26% respectively). This information suggests that inlets and the associated mudflats do not host the greatest number of non-breeding shorebirds, although some species (e.g., Red Knot) seemed to favor these sites. Instead, the open, east-facing beaches of Cape Hatteras National Seashore from Cape Hatteras north to Rodanthe hosted the most shorebirds overall, and the greatest numbers for three of the most numerous species (Black-bellied Plover, Willet, and Sanderling).

Management of Non-breeding Shorebirds. Other shorebirds likely benefit from management to protect American Oystercatchers and particularly Piping Plovers, but are not a management focus themselves. In the absence of data relating the distribution, foraging success and survival of these species to factors such as habitat availability, disturbance and predator pressure, there is little basis for designing a management plan for them. Research aimed at understanding the ecology of non-focal shorebird species during migration (and possibly other seasons) would be useful given the global importance of the Outer Banks to several migratory shorebirds. Lacking such information, one management strategy might be to protect areas where nonbreeding shorebirds are concentrated. This may not be a viable approach, however, given how widespread migrating and wintering shorebirds are within the Seashore, and the variation among species in the locations where they are most abundant. The primary concern with respect to migrating and wintering shorebirds is disturbance due to human activities. Shorebirds are susceptible to disturbance from pedestrians, ORVs and off-leash pets (Barbee 1994; Tarr et al. 2010). A detailed study of shorebird disturbance on CAHA and CALO documented the importance of closed (undisturbed) areas to Sanderlings and other shorebirds, which spent less time roosting and more time being vigilant in areas subject to frequent disturbance (Barbee 1994). The effects of human disturbance on foraging behavior are less understood with some studies finding no effect (Barbee 1994; Morton 1996) and others finding a negative effect (Burger and Gochfeld 1991; Thomas et al. 2003; Murchison et al. 2016).

Closures designed to protect Piping Plovers may serve the purpose of reducing disturbance of other migrating shorebirds, especially closures that encompass beach foraging areas or roost sites. Specifically, the most important area for Piping Plovers in fall migration (South Point on

Ocracoke) is also an important wintering site for Red Knots. Still, specific benefits to other species are unknown and surveys for non-target use of closed areas are lacking. The current beach management plan in the nonbreeding season at CAHA does not address the importance of the Seashore and South Point in particular as a migratory stopover site for Piping Plovers and the importance of the Seashore as a migratory stopover and wintering site for other shorebird species. In particular, several species might benefit from extending closures in key areas farther into the fall months.

Beach-nesting Colonial Waterbirds

Beach-nesting colonial waterbirds have received considerable attention in the southeastern U.S. due to widespread threats to breeding colonies and resulting population declines in many areas. Cape Hatteras National Seashore is a regionally important area for several species of beach-nesting colonial waterbirds. Use of the Seashore by colonial nesting waterbirds prior to extensive development in the twentieth century is not well known. Parnell et al. (1997) summarized changes during most of the twentieth century and noted that species diversity and numbers increased during that period. In the early 1900s Least (*Sternula antillarum*) and Gull-billed Terns (*Gelochelidon nilotica*) were rare, and all beach-nesting colonial waterbirds used natural beaches on barrier islands (Parnell et al. 1997). By the 1970s, the pattern of nesting had shifted with 50% of colonies occurring on man-made or man-altered substrates, especially dredge spoil islands in the sounds (Parnell et al. 1997; Erwin et al. 2003). Thus, the pattern of occurrence of this group of birds has shifted dramatically during the last century from nesting entirely on natural beaches to the present split between natural beaches and man-made habitats on the sound side of the barrier islands.

Beach-nesting Colonial Waterbirds at Cape Hatteras National Seashore

The colonial waterbird community at Cape Hatteras National Seashore is comprised of two bird groups, the herons (Family Ardeidae) and gulls, terns and skimmers (Family Laridae). The former were monitored in early surveys prior to 2010, but surveys since 2010 have only monitored the latter (e.g., NPS 2010b; 2011; 2012; 2013; 2014b; 2015d). Current monitoring focuses on five focal species of beach-nesting colonial waterbirds: Least Tern, Gull-billed Tern, Common Tern (*Sterna hirundo*), Forster's Tern (*Sterna forsteri*), and Black Skimmer (*Rynchops niger*). Other colonial waterbirds such as herons continue to occur in the area, but are not classified as beach-nesting and therefore are not covered by current monitoring efforts. None of the monitored species are federally listed, but the state of North Carolina lists Least Tern and Black Skimmer as species of Special Concern, Gull-billed Tern as Threatened, Common Tern as Endangered, and Forster's Tern as a Watch List species (LeGrand et al. 2018). Thus, all of these species have special protection in North Carolina.

Population Trends. Cape Hatteras National Seashore contributes to conservation of beach-nesting colonial waterbirds within a regional context. This context includes regional population goals set by the North Carolina Waterbird Committee in 2000 that were derived from the work of the South Atlantic Migratory Bird Initiative, which based goals on abundance estimates from the 1970s and early 1980s. Within this context, the Seashore in 2010 set its own population goals for four of its focal species, expressed as desired future conditions (NPS 2010a). Desired future conditions for the short term are five-year averages of 482 nests of Least Terns, 292 of Common Terns, 21 of Gull-billed Terns and 132 of Black Skimmers. For the long-term desired future

conditions are five-year averages of 577 nests of Least Terns, 533 of Common Terns, 40 of Gull-billed Terns and 244 of Black Skimmers. These goals apply to colonies located in beach habitat and thus do not include colonies on dredge spoil islands in the area.

Monitoring data for the five focal species of beach-nesting colonial waterbirds are available beginning in 2010, but there is only anecdotal information on trends prior to that time. Monitoring is restricted to counts of nests, eggs, and chicks, although the latter are not believed to be indicative of true productivity (NPS 2015d). These efforts occur during the peak nesting season, typically in June but varying in response to colony activity. As expected, during the period the Committee examined (2011-2017) numbers of the focal species fluctuated considerably (Table 1). That there is high annual variation in the nesting effort of these species is the reason that population objectives are expressed, appropriately, as five-year averages rather than single-year values. Least Tern, the most numerous species, showed a steady decline from over a thousand nests in 2011 to 200-300 in 2015-2017. Nests of Common Tern, Black Skimmer and Gull-billed Tern also seemed to be declining, but there was a resurgence in the number of their nests in 2016 and 2017. Forster’s Terns rarely nest on the beaches within CAHA: nearly all the nests of this species observed during 2011-2017 occurred in a single year (Table 1). Collectively, the number of nests of beach-nesting colonial waterbirds counted annually during surveys has declined from over a thousand during 2011-2013 to 400-600 in 2014-2017. The survey timing in June is probably sufficient to capture the majority of nesting activity, although the small number of visits (1-2 per colony per year) increases the chances of missing nests that are depredated quickly.

Table 1. Nest counts of beach-nesting colonial waterbirds at Cape Hatteras National Seashore, North Carolina, 2011-2017.

Species	2011	2012	2013	2014	2015	2016	2017
Least Tern	1,063	833	802	469	291	295	189
Gull-billed Tern	15	43	6	1	3	23	14
Common Tern	112	218	34	38	16	91	90
Forster’s Tern	0	0	42	4	2	0	0
Black Skimmer	99	221	119	95	85	169	214
TOTAL	1,289	1,315	1,003	607	397	578	507

Declining numbers of nests have precluded reaching population goals that seemed achievable when they were set in 2010. Both the short-term and the long-term goals for Least Terns were achieved during the five-year period 2011-2015, but only the short-term goal was achieved during 2012-2016 and neither goal was during 2013-2017. Goals for Common and Gull-billed Terns have not been reached, although the number of nests of Gull-billed Terns (but not Common Terns) in a single year has exceeded the desired five-year averages in a couple of years. The resurgence of nesting by Black Skimmers in 2016 and 2017 has resulted in attainment of the short-term goal for this species during 2012-2016 and 2013-2017.

Accurate counts of adults, when possible, might increase the value of survey efforts. Nest counts are not a robust monitoring technique for tracking population trends because not all nests can be counted (some are missed for many reasons), and the factors affecting nest detectability (e.g., different observers or substrates) are not considered in the counts (see Thompson et al.

1998). Thus, current nest counts are probably subject to an unknown level of bias with respect to true population trends of some or all of the species nesting on Seashore beaches.

Current monitoring and management. The monitoring program for colonial waterbirds at CAHA began in 1977, and in the intervening years, there have been 12 complete surveys of the area (S. Schweitzer, pers. comm.). Survey results have been collated for the years 2010-2017 (NPS 2010b; 2011; 2012; 2013; 2014b; 2015d; 2016b; 2017); the earlier surveys were less complete and have not been included in recent park analyses of colonial waterbirds. Except in recent years, surveys have lacked a single standardized approach. All colonial waterbird surveys use “active nests” as the metric for surveys. Surveys are conducted in May and June at a time when nesting activity peaks for the focal species. Methods for a “survey” include a mix of field approaches: walking surveys, observations of the colony from a distance to minimize disturbance, and occasionally estimates generated from counts of adults. Basic information collected during colony surveys includes the number of nests, number of eggs, and, when possible, minimum counts of the number of chicks. Chick counts are done during nest surveys, and thus are not really timed to provide a robust estimate of productivity. Similarly, there are no attempts to estimate nest survival, as individual nests are not monitored.

Recommended methods for colonial waterbird conservation include habitat protection and restoration, predator management, and protection from human disturbance (Burger et al. 2004). The management program on CAHA Seashore focuses on protecting colonies from disturbance via closures, and also includes some modest efforts to reduce predation. A lawsuit in 2007 resulted in a Consent Decree on 30 April 2008 that provided specific guidelines for closures to protect beach-nesting colonial waterbirds. This agreement was contingent on the later development of an ORV management plan. On 23 January 2012 the Cape Hatteras National Seashore Off-Road Vehicle Management Plan and Special Regulation (hereafter ORV Management Plan) was enacted and includes continuing provisions for protecting beach-nesting colonial waterbirds. Those provisions are (1) for the Least Tern a 100 m buffer for breeding behavior, scrapes, and nests and a 200 m buffer for chicks, and (2) a 200 m buffer for all breeding and nesting activity of other beach-nesting colonial waterbirds (NPS 2010a). Closures can be modified during the nesting season in responses to colony changes, and closed areas buffer the greatest extent of multi-species colonies. Deliberate closure violations can result in an expansion of the closed area, from a 50 m expansion for the first offense to a 500 m expansion on the third offense (NPS 2010a).

Factors Impacting Beach-nesting Colonial Waterbirds at Cape Hatteras National Seashore

Habitat. As a group, beach-nesting colonial waterbirds tend to select open sites such as recent overwash areas that they reuse for several years before habitat degradation (e.g., encroaching vegetation) or other factors causes them to shift to an alternative site (Erwin et al. 1998; Wyman et al. 2014). For example, Figures 16 and 17 illustrate how colonies within CAHA shifted during the 6-year period from 2010-2015 at Bodie Spit and Cape Hatteras respectively. The specific habitat requirements of each species in this group differ subtly, which when combined with disturbances such as overwash events may play a role in nest-site selection. The Least Tern (Downing 1973), Common Tern (Nisbet et al. 2017), and Black Skimmer (Erwin 1980) prefer open sandy, gravel, or shell substrates with sparse vegetation in coastal areas. The Gull-billed Tern prefers sandier sites (Molina et al. 2014) and the Forster’s Tern prefers marshy edges or sites with mats of dead vegetation (Parnell et al. 1997; McNicholl et al. 2001).



Figure 16. Map showing the location of beach-nesting waterbird colonies at Cape Hatteras Point, Cape Hatteras National Seashore, North Carolina, 2010-2015.

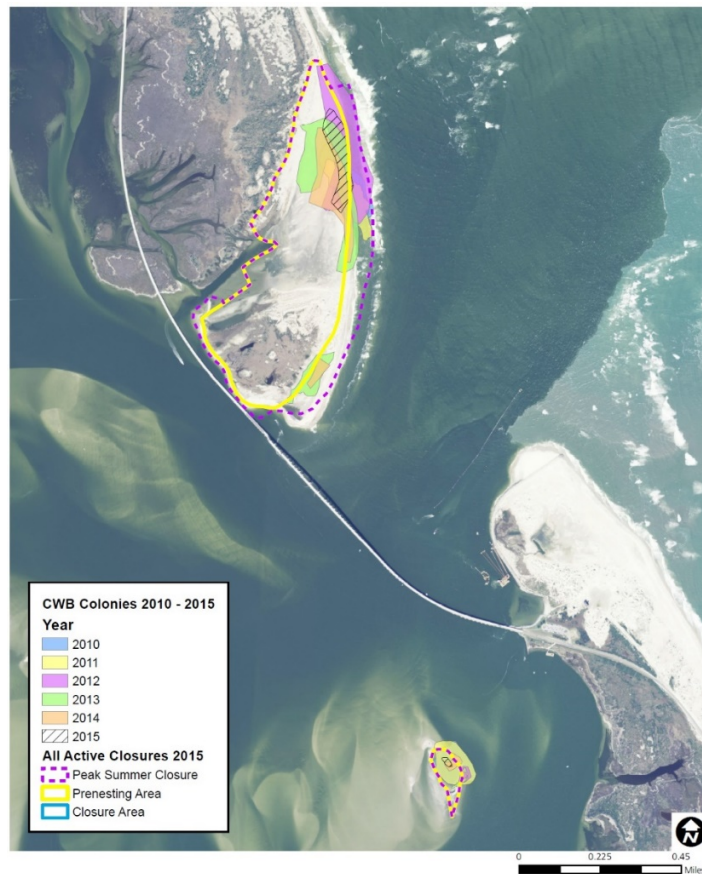


Figure 17. Map showing the location of beach-nesting waterbird colonies at Bodie Island and Green Island, Cape Hatteras National Seashore, North Carolina, 2010-2015.

Overwash events create and maintain nesting habitat for Least Terns, Common Terns and Black Skimmers, and thus nesting colonies are often located near inlets and on spits. Hence the same concerns about the potential for anthropogenic activities such as beach stabilization that disrupt barrier island dynamics to limit carrying capacity discussed for other beach-nesting species potentially could also apply to beach-nesting colonial waterbirds. However, this possibility seems unlikely in species that nest in dense aggregations. Colonial species generally are not limited by availability of nesting habitat, but are sensitive to factors such as predation and disturbance that affect the quality of a site. Declines in numbers at a particular site thus do not necessarily reflect a decline in the regional population size, but instead may be a result of individuals abandoning that site for another, more productive nesting site. Therefore, factors such as predation, human disturbance and foraging conditions (and flooding, see below), coupled with availability of nesting sites in other locations where those factors are more favorable, are better possibilities for explaining the recent declines in nesting of colonial waterbirds on CAHA than changes in availability of nesting habitat.

Although habitat for beach-nesting colonial waterbirds does not appear to be limited currently, the effect of sea level rise on nesting habitat for these species is a looming concern (Hunter et al. 2015). Indeed, significant tidal events have flooded nests and caused nesting colonies to be abandoned in recent years (e.g., NPS 2017). Accommodating predicted rises in sea level into colony management (e.g., closure size, vegetation management) will be necessary if the current management approach is to remain effective, although engineered solutions to providing habitat for colonial waterbirds might also be possible (see Hartman et al. 2016).

Predation. Nests and chicks of beach-nesting colonial waterbirds are vulnerable to predation by mammals and birds during the nesting season. Laughing Gulls (*Leucophaeus atricilla*) are thought to be a predator of eggs and small chicks (see Donehower et al. 2007). This species has always been a fairly common breeding bird in the region (Parnell et al. 1997) and annual colonial waterbird reports from CAHA sometimes mention concerns about this species. Other birds can also be predators, including non-breeding gulls and even shorebirds such as the Ruddy Turnstone (Rounds et al. 2004). Mammalian carnivores are the other primary group of predators and include red fox (*Vulpes vulpes*), raccoon, Virginia opossum (*Didelphis virginiana*), and others (Erwin et al. 2001). CAHA has implemented methods of mammalian predator control to improve colony reproductive success, although this program has varied annually. In 2010, there was a pilot project to place electric fence around a single colony at Bodie Spit to deter mammalian predators (NPS 2010b). This project had inconclusive results – some canids apparently were deterred from entering the colony while other mammals were able to breach the fence. In 2011, the predator control program again included employing the electric fence at the Bodie Spit colony, as well as targeted mammal trapping at colonies where there was evidence of predators (NPS 2011). In 2011, ten fence breaches and ten fence contacts were noted; 46 eggs were lost as a result of the breaches, although colony success was deemed high overall. In 2012 the electric fencing program continued at Bodie Spit, and an additional electric fence was deployed at a second colony at Cape Hatteras; targeted trapping of mammalian predators continued as in 2011 (NPS 2012). In 2012, 15 fence breaches and 9 fence contacts were noted; impacts on colony success were not noted. Predator control efforts were reduced after 2012 to include only targeted trapping of mammals, which has been conducted in some years, but not others. Removal of mammalian predators to protect other species, such as the removal of 10 coyotes in the winter of 2017-2018 to benefit oystercatchers may benefit beach-nesting colonial waterbirds as well.

Reductions in numbers of nests of beach-nesting colonial waterbirds within the Seashore to some extent parallel reductions in efforts to control mammalian predators. Without data on nesting success it is impossible to determine how high predation rates are, or how effective electric fencing and targeted trapping are. Clearly mammalian predators are attacking colonies of beach-nesting waterbirds on CAHA. We concluded above, based on nest success data, that mammalian nest predators likely were limiting productivity of American Oystercatchers on the Seashore. It is reasonable to infer that beach-nesting colonial waterbirds, which are highly vulnerable to mammalian predators, are also experiencing elevated mortality of eggs and chicks due to these same predators. Thus, that numbers of colonial waterbirds nesting on beaches are declining because individuals are perceiving nesting sites on the Seashore as unsafe and moving elsewhere is a valid hypothesis. In particular, birds could be moving to dredge spoil islands in the area. Most Forster’s Terns and Common Terns nest on dredge spoil islands, and more Black Skimmers nest there than on the beaches. Testing the hypothesis that declines in the number of colonial waterbirds nesting on beaches within CAHA are due to birds moving elsewhere to avoid high levels of predation will require collecting data on nest success, ideally in an experimental setting that includes controlling predators at some colonies.

Disturbance and closures. Beach-nesting colonial waterbirds are vulnerable to many types of disturbance during the nesting season, and appropriate conservation measures are often necessary to protect colonies (Burger and Niles 2013). Colonies are susceptible to disturbance from pedestrians (Hillman et al. 2015), ORVs (Hillman et al. 2015), off-leash pets, human development (Erwin 1980), flooding mediated by sea-level rise (Erwin et al. 2011), and modification to nesting habitats (Molina and Erwin 2006). Perceptions about beach closures to protect waterbirds from human disturbance are not necessarily negative, although a process to designate a closure that includes stakeholder input is critical (Burger and Niles 2013). The closures at CAHA appear to be quite effective. Annual reports for colonial waterbird monitoring document known unlawful entries into closed areas. Most (87%) are due to pedestrians, followed by dogs and ORVs (Table 2). Known violations declined greatly from 2010 through 2015. Although these observations do not result from a formal sampling protocol, they nevertheless suggest that levels of disturbance are not excessive and are declining. Hence, it is unlikely that declines in the numbers of colonial waterbirds nesting on beaches are due to birds increasingly abandoning these sites due to disturbance.

Table 2. Documented violations of closed areas protecting beach-nesting colonial waterbirds at Cape Hatteras National Seashore, North Carolina, 2010-2015.

Year	Type of violation			
	Pedestrian	Dog	ORV	Other
2010	231	18	4	5
2011	176	14	5	7
2012	84	15 ¹	3	0
2013	78	4 ¹	5	0
2014	31	0	3	0
2015	18	3	1	2
Total	618	54	21	14

¹This was a combined total for dogs, boats, and horses.

Regional perspective. Population trends of beach-nesting colonial waterbirds at CAHA must be viewed in a regional context to fully assess the role of the Seashore in the conservation of these species and to interpret population trends. Nest counts on the Seashore must be compared with counts in other locations such as dredge spoil islands and beaches on barrier islands to the north and south to evaluate whether declines in nesting within CAHA represent shifts to other breeding locations or true population declines.

The importance of CAHA to regional conservation depends on the position of the Seashore within a species' range. CAHA is in the core of the breeding range of Least Terns and Black Skimmers along the Atlantic Coast and hence is an important breeding location for these species. Of the other species, Gull-billed Tern (near northern edge), Common Terns (near southern edge) and Forster's terns (near southern edge) may be nesting near the limits of their natural breeding ranges. Peripheral populations often occupy less optimal habitats than those at the core of a species' range, and as such may suffer lower breeding success and survival (Golawski et al. 2016). It is possible that this pattern exists for some species at CAHA, although the lack of robust productivity data makes this difficult to discern and reinforces the need for such data in the future. Thus, population goals for Least Terns and Black Skimmers at CAHA may be appropriate, whereas those for Common Terns and Gull-billed Terns may be overly ambitious.

In addition, in many temperate species populations at the southern edge of the range are declining, often to extinction, as the climate warms (Hampe and Petit 2005; Chandler et al. 2018). Thus, maintaining breeding populations of Common and Forster's Terns may not be feasible in the long term due to factors that cannot be addressed through management. Thus, the nesting population objectives set for the Common Tern in particular appear to be overly ambitious and possibly unachievable. In fact, the Common Tern is declining not just locally at CAHA but also regionally. Factors such as impacts of changing climate on the food supply, or on thermoregulatory and osmoregulatory stress on chicks (Grémillet et al. 2012), may thwart any attempt to significantly increase the number of nesting Common Terns on CAHA. On the other hand, Least Terns are increasing regionally and hence the reduction of nesting on CAHA in a part of their range where they should be doing well is of concern. A focus on increasing numbers of this species would be well justified. Finally, one might anticipate that the Seashore may become a more important breeding site for the Gull-billed Terns in the future due to its location at the northern edge of this species' breeding range, such that population goals for this species are realistic.

Conclusions

The colonial waterbird monitoring program at Cape Hatteras National Seashore emphasizes population and habitat goals, which then feed into management actions, identify future research needs, and lead to greater collaboration across the region. Recent reductions in numbers of nesting birds make it difficult to reach population goals established when numbers were higher. Comparison of nest counts on the Seashore to comparable data from other, nearby locations, particularly dredge spoil islands, is necessary to determine whether declines on CAHA represent true population declines or shifts of birds to other nesting locations. In either case, high levels of predation by mammals seems the most likely cause of reduced nesting as closures appear to be effective in reducing human disturbance to reasonable levels, and nesting habitat does not appear limited. Increasing flooding of nesting sites due to sea level rise, is another possible cause of reduced nesting worth investigating. Efforts to assess these factors are limited by lack of reproductive metrics such as hatching and fledging success. The decline in nests of Least Terns

is of most concern as this species has increased regionally and CAHA should be a prime breeding site due to its location in the middle of the species' breeding range. Population goals for Common Terns appear overly ambitious for a population of this regionally declining species at the southern edge of its breeding range.

Key uncertainties, research and monitoring needs. Much could be learned about beach-nesting colonial waterbirds at Cape Hatteras National Seashore through collecting additional information that complements basic colony surveys. Colony surveys that gather basic information about colony occupancy, numbers of adults and nests, and perceived threats should continue, but these data are of limited use in understanding population trends without adequate productivity estimates. Hence, measuring annual productivity is the highest priority research need. Productivity could be assessed in several ways, from simple estimates of hatching success (calculated by monitoring a sample of nests within each colony; Eyer et al. 1999) or standardized nest and chick counts at the appropriate season (Corace et al. 2017), to detailed mark-recapture work with individual chicks to estimate survival during the fledgling period (Brooks et al. 2013; Gach et al. 2018). Careful consideration of a marking scheme, number of individuals marked, and protocols for re-encountering birds is critical because colonial waterbirds are often long-lived and require long-term monitoring schemes (Horswell et al. 2018).

Measures of productivity would be useful in addressing two other key uncertainties, the impacts of predation and sea level rise on beach-nesting colonies. Ultimately, it would be useful to relate attributes of individual colonies (size, susceptibility to disturbance and other metrics of interest) to a colony-specific measure of productivity, which in turn could be used to guide future management decisions such as need for closures and predator control. Finally, the extent to which changes in numbers of nesting birds on CAHA represent true population declines or shifts to nesting locations outside the Seashore such as dredge spoil islands is another key uncertainty.

Sea Turtles

Five of the seven extant species of sea turtles, loggerhead (*Caretta caretta*), green (*Chelonia mydas*), leatherback (*Dermochelys coriacea*), Kemp's ridleys (*Lepidochelys kempii*) and hawksbill (*Eretmochelys imbricata*), occur in North Carolina waters. The life of a sea turtle begins with a dangerous dash to the sea upon hatching (Figure 18), and those that survive spend most, or all in the case of males, of the rest of their life in the ocean, moving and migrating great distances between foraging and nesting areas. Juveniles spend many years in the open oceans, whereas adults spend more time in near-shore waters. Adult females emerge from the sea to nest on beaches, completing the life cycle.

The significance of Cape Hatteras National Seashore to sea turtle conservation is as a nesting area. Sea turtles are a focus of major conservation efforts worldwide. Of the species occurring in North Carolina, Kemp's ridley and Hawksbill turtles are critically endangered globally, whereas green sea turtles are endangered; leatherbacks and loggerheads are classified as vulnerable. Hawksbills and leatherbacks are globally distributed species that nest in tropical areas, such that CAHA is north of their regular nesting range. Kemp's ridley sea turtles are similar in being tropical nesters whose regular nesting areas are south of CAHA, but they are a much more narrowly distributed species whose nesting range is limited to the Gulf of Mexico. Green and loggerhead sea turtles have a broad nesting distribution that includes temperate areas. The southeastern coast of the U.S., including the Seashore, represents a minor nesting area for green sea turtles and a major nesting area for loggerheads.



Figure 18. Loggerhead sea turtle hatchlings head to the sea upon hatching. Photo by Matthew Godfrey.

Sea turtle populations have been adversely affected by a variety of factors impacting different stages of their life history. These include, among others, alterations to marine ecosystems and oil spills that reduce the carrying capacity of the foraging habitat of adults, harvest of eggs and adults by humans, mortality of adults and subadults due to fisheries bycatch, loss of nesting habitat to coastal development, hatchling mortality due to light pollution, and loss of nests due to human activities and predation (Bolten et al. 2011; Caillouet et al. 2018). Due to the life history of sea turtles as a long-lived species that produces a large number of offspring with little investment in each, loss of an adult or especially a subadult has a much larger impact on a population than loss of an egg or a hatchling (Crouse et al. 1987). Considerable mortality at the egg and hatchling stages is a normal part of sea turtle life history. Nevertheless, excessive losses of eggs and hatchlings due to anthropogenic impacts on beach habitat and nests can cause populations to decline. The appropriate management goal for sea turtles at CAHA is to provide sufficient beach habitat for the populations of loggerhead and green sea turtles for which the Seashore is a significant nesting area, and to enable a level of productivity within that habitat sufficient to support stable or increasing populations. The western Atlantic populations of both green and loggerhead sea turtles are in fact increasing currently (Mazaris et al. 2017).

Sea Turtle Nesting at Cape Hatteras National Seashore

Except for Kemp's ridleys, reproductive female sea turtles in the southeastern U.S. emerge from the sea to deposit their eggs on the beach between dusk and dawn. Generally, there are several basic requirements for a suitable nesting beach for sea turtles. There must be accessibility to the beach from the sea, and the beach must be high enough that the sand is not constantly inundated by high tides or the water table below. The beach sand must be conducive to nest cavity construction by the female and to gas diffusion, and the incubation temperature must be within the range necessary for successful embryonic development (Mortimer 1990; Miller et al. 2003). Other conditions may render nesting beaches unsuitable or unused, including excessive

nighttime lighting visible on the beach (Witherington 1992) or the presence of artificial structures placed directly on the beach (Bouchard et al. 1998). Reproductive females may return to the ocean without laying eggs if they are disturbed by lighting or people. Even when conditions are thought to be perfectly suitable, reproductive females that have already emerged from the sea onto the beach may abandon their nesting attempt (these are often referred to as “false crawls” or “non-nesting emergences”). Generally, the average ratio of nesting to non-nesting emergences for loggerhead sea turtles is 1:1 (Dodd 1988).

Ocean-facing beaches within Cape Hatteras National Seashore constitute suitable nesting habitat for sea turtles. Sea turtles have used the ocean-facing sandy beaches at the Seashore as nesting habitat at least since the late 1970s, when the earliest nest records were documented, although observations by Catesby (1731-1743) suggest loggerhead sea turtles may have been nesting in the Carolinas in the 1700s. Similar to other sea turtle nesting beaches in North Carolina, the majority (>90%) of sea turtle nests found in CAHA are laid by loggerheads (Table 3). In addition, in order of decreasing abundance, nests laid by green turtles, leatherbacks, and Kemp’s ridleys turtles occur at CAHA (Table 3). In 2015, two nests laid by a hawksbill turtle (*Eretmochelys imbricata*) were documented at CAHA, representing the northernmost documented nesting record for this species in the Atlantic (Finn et al. 2016). Reproductive females of all sea turtle species nesting in North Carolina generally lay multiple clutches of eggs in a single season, and commonly skip 1-4 years between nesting seasons.

Table 3. Number of sea turtle nests documented on Cape Hatteras National Seashore beaches, and proportion (%) relative to the total number of nests laid in North Carolina.

Year	Loggerhead	Green	Leatherback	Kemp’s ridley	Hawksbill	% of sea turtle nests in NC
2002	95	3	1	0	0	14%
2003	83	4	0	0	0	10%
2004	40	3	1	0	0	13%
2005	69	9	0	0	0	12%
2006	80	4	0	0	0	11%
2007	73	8	1	0	0	15%
2008	109	3	0	0	0	13%
2009	101	2	1	0	0	17%
2010	148	5	0	0	0	17%
2011	138	8	0	1	0	15%
2012	220	1	1	0	0	20%
2013	231	23	0	0	0	19%
2014	122	2	0	0	0	22%
2015	277	10	0	1	2	22%
2016	314	10	0	1	0	20%

The number of loggerhead sea turtle nests on CAHA beaches is steadily increasing, sufficiently so that the Seashore’s population objectives for the species (94 nests annually in the short-term and 115 in the long-term, representing a 2% annual increase and 10% of the nests in North Carolina, NPS 2010a) have all been exceeded (Table 3). Almost certainly, goals that are more ambitious could be set and achieved if desired.

Sea turtles nest on beaches at CAHA between May and early September of each year. In 2000 and 2007, a freshly laid leatherback nest was found in the third week of April, but sea turtle nesting before May is rare in North Carolina generally. Similarly, nesting after 15 September is rare in North Carolina (a loggerhead nest was laid in the Seashore on 7 October 2009). Based on sea turtle nesting data from 2009-2016, 4% of nests were laid in May, 12.5% were laid in August, and <1% were laid in September. In contrast, 37% were laid in June and 45% were laid in July (Figure 19). The five species have overlapping nesting seasons, although leatherbacks tend to nest in late spring and early summer, while green turtles tend to nest after June.

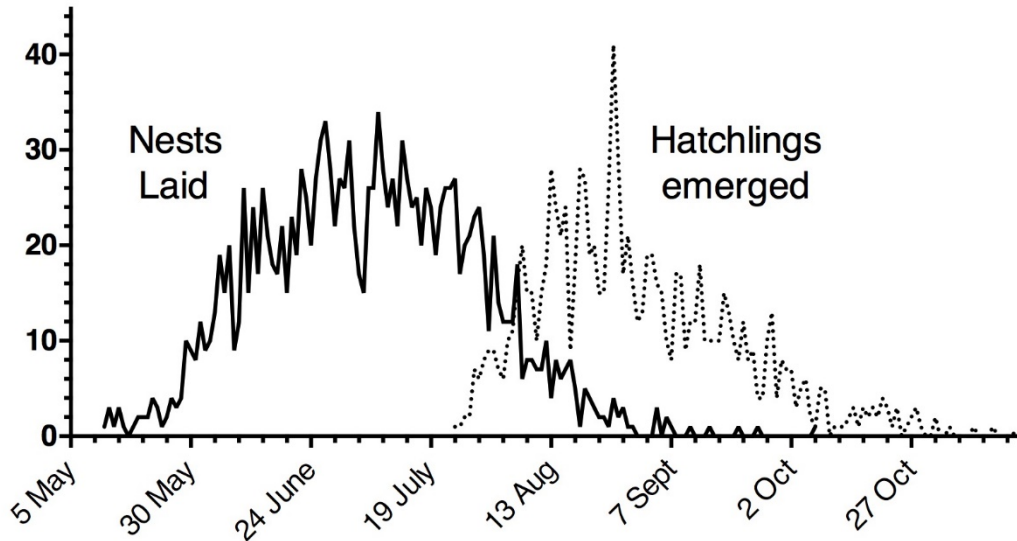


Figure 19. Dates when sea turtle nests were laid, and when hatchlings emerged from nests, at Cape Hatteras National Seashore, 2009-2016.

Each nest contains 60-160 yolked eggs, depending on species and individual turtle. The eggs require 50-75+ days of incubation before they hatch, depending on the temperature of the surrounding sand. Cooler incubation temperatures that generally occur earlier and later in the season result in longer incubation periods, while warmer incubation temperatures that occur in the middle of the nesting season tend to result in shorter incubation periods (Godfrey and Mrosovsky 1997). After hatching from the eggs, the hatchlings take a few days to move to the top of the nest cavity, near the surface of the sand, and will emerge from nests and scramble to the sea during nighttime hours. The majority of hatchlings emerge from nests in August (55%) and September (33%), with only 4% emerging in July, 8% in October, and 1% in November (Figure 19).

Management of Sea Turtles at Cape Hatteras National Seashore

All known sea turtle nests at Cape Hatteras National Seashore are marked and protected during their incubation period. At least 72 hours after the majority of sea turtle hatchlings have emerged from the nest cavity (based on hatchling tracks emerging from the nest area), or after 70-100+ days of incubation, depending on time of year and status of the eggs, the nests are inventoried to quantify hatching success (the relative number of hatchlings that leave the eggs). Hatching success can be negatively affected by external factors such as inundation (from either

overwash or excessive rainfall; Eckert and Eckert 1990; Kraemer and Bell 1980), predation (Stancyk et al. 1980), temperature (Matsuzawa et al. 2002; Bentivegna et al. 2010), and anthropogenic activities such as beach cleaning, driving or even sunbathing that may lead to inadvertent piercing or crushing of eggs (Witherington 2003). A primary driver affecting hatching success of sea turtle clutches in the Southeast is excessive inundation and erosion associated with tropical storms and hurricanes that affect the region in late summer and early fall (Pike and Stiner 2007). From 2009-2016, hatching success of sea turtle nests incubating at CAHA varied by the month the nests were laid, with 60% or higher average success of nests laid in May, June and July, compared to only 36% for August nests. Eggs that are laid in September and October are exposed during the incubation period to sand temperatures that are near or below the minimum for fully successful embryonic development (Bentivegna et al. 2010). No nests laid in September or October at CAHA produced viable hatchlings during 2009-2016.

Sometimes, sea turtle females place nests in areas prone to repeated, daily inundation that would result in no embryonic development, or in areas that are known to erode during the summer months, resulting in loss of all eggs. In such cases, staff at CAHA move the eggs to a location that is safe from daily overwash or predicted erosion, generally further away from the water line but close to the original location. This procedure is in accordance with the NMFS/USFWS 2nd Revision of the Recovery Plan for the Northwest Atlantic Population of the Loggerhead Sea Turtle (NMFS 2008). Relocation of sea turtle eggs is considered a tool to be used as a last resort, because of potential negative impacts on embryonic development (Limpus et al. 1979), alteration of the thermal environment of the eggs during incubation and concomitant impact on hatchling sex ratios (DeGregorio and Williard 2011; Pintus et al. 2009), and possible effects on hatchling behavior or fitness (Adam et al. 2007; Read et al. 2013).

Although relocation rates of sea turtle nest at CAHA were high in the 1990s, they have been much lower, roughly 30% overall, for each season since the late 2000s (Figure 20). The Seashore’s goal of relocating < 30% of nests (NPS 2010a) has been achieved every year since 2006 except one (2010). When nests are grouped by month laid, and hatching success for relocated versus non-relocated nests are compared, relocated nests always had a somewhat higher hatching success, but were not significantly different from non-relocated nests except for nests laid in August, when hatching success for both relocated and non-relocated sea turtle clutches was lowest (Table 4).

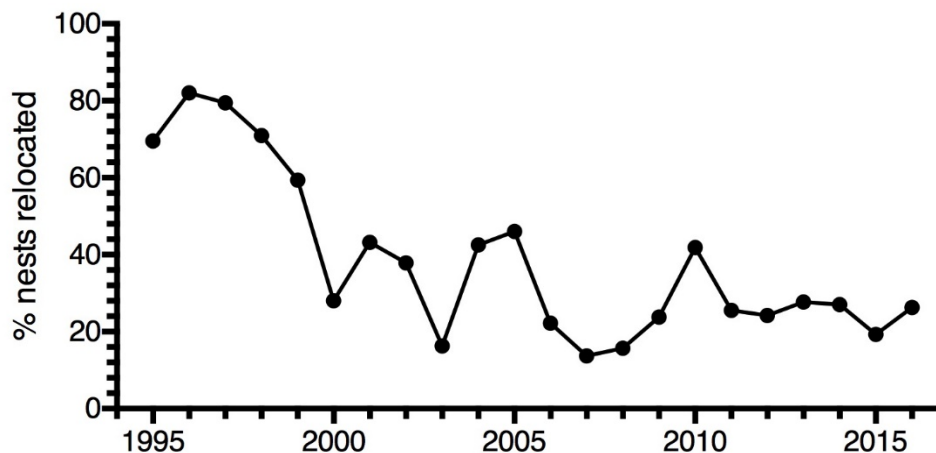


Figure 20. Percentage of sea turtle nests relocated each year since 1995 at Cape Hatteras National Seashore.

Table 4. Average hatching success (and standard error, SEM) of sea turtle nests by month and by treatment (No = not relocated, Yes = relocated), for Cape Hatteras National Seashore, 2009-2014, assessed from data provided by the Seashore. P-values are for nonparametric Mann-Whitney tests between pairs within months.

	May No	May Yes	Jun No	Jun Yes	Jul No	Jul Yes	Aug No	Aug Yes
Hatching success	64%	77%	68%	70%	56%	68%	33%	43%
SEM	4.1	4.5	1.6	2.2	1.8	2.3	3.4	5.0
N	61	10	440	165	542	194	138	58
P value	>0.05		>0.05		>0.05		0.04	

Factors Impacting Sea Turtle Populations at Cape Hatteras National Seashore

One of the primary factors affecting hatching success of sea turtle clutches in the Southeast is excessive inundation and erosion associated with tropical storms and hurricanes that affect the region in late summer and early fall (Pike and Stiner 2007; Van Houtan and Bass 2007). This factor likely accounts for lower success rates of nests in August, along with deteriorating sand temperatures during the incubation period, which precludes successful nesting in September or later. Losses to storms are inevitable and have always been part of the life cycle of sea turtles. However, such losses may become excessive due to changes in beach structure resulting from human activities, coupled with sea level rise and possibly increasing intensity of storms due to climate change. The potential for this likely exists at CAHA. CAHA managers appear to be effectively mitigating this issue through their nest relocation program thus far.

Predation of sea turtle eggs and hatchlings on the beach is not a major issue on Cape Hatteras National Seashore. Occasional predation events by mink, fire ants, and birds were documented during 2009-2016, but not in large numbers. The most common predator of eggs and hatchlings during this period was the ghost crab. These crabs may take more than a dozen eggs and/or hatchlings from a single nest, and can be identified either by cuts on eggshells or from hatchling tracks terminating at ghost crab holes, if not observed directly. Similar partial brood loss to ghost crabs in Florida, where they also are the most common nest predator, is estimated to reduce overall hatching success by about 10% (Brost et al. 2015). Ghost crab predation is not considered a serious threat to productivity, whereas mammalian predation is (Brost et al. 2015). Fortunately, mammalian predation of sea turtle nests has been negligible at CAHA. However, in 2016 a coyote depredated incubating eggs from a nest located on Bodie Island. Coyotes are a new threat to beach-nesting species, and have potential to have a significant impact on nesting success of sea turtles, as evidenced by their impact on nesting success of American Oystercatchers in 2017 (see above). The removal of ten coyotes in the winter of 2017-2018 to protect oystercatcher nests may benefit sea turtles as well. In addition, CAHA staff have started placing wire mesh in the sand above incubating eggs, to deter possible coyote predation. The wire mesh is large enough to allow hatchlings to emerge while not allowing coyotes to successfully dig into the nest cavity. As with losses to storms, current management appears effective in limiting losses to nest predation to levels appropriate to the life history of sea turtles.

Nocturnal activities by sea turtles include mature females laying eggs on the beach and hatchling emergence from the nest cavity on the surface of the sand. While these activities may occasionally occur during daylight hours, they most often occur during hours of darkness. Artificial lighting can affect both nesting females and emerging hatchlings on beaches. Early

work on artificial lighting by Witherington (1992) showed that when artificial lights were turned on at night to brighten an otherwise darkened sea turtle nesting beach, the number of females that nested was greatly reduced. Thus, high levels of artificial light on beaches dissuade female sea turtles from nesting there. Less intense but still visible sources of artificial light may not dissuade adult female sea turtles from nesting, but may interfere with their ability to successfully return to the sea (Deem et al. 2007). Sea turtle hatchlings generally emerge from their nests at night, and rely on visual cues to orient their movement toward the sea (e.g., Ehrenfeld 1968; Mrosovsky and Shettleworth 1969). When exposed to sources of artificial light, sea-finding behavior of hatchlings will become disrupted, and often hatchlings will travel away from the sea (e.g., Peters and Verhoeven 1994; Philibosian 1976; Salmon et al. 1995). This increases the time they are exposed to terrestrial predators, reduces the amount of residual internalized yolk available to hatchlings for their initial swim offshore, and could result in desiccation and death if the hatchlings remain on land after sunrise.

There are reports on CAHA of both nesting females and hatchlings that have been confused by artificial lights visible at night, and were unable to find the ocean, including one adult female in 2016 that crossed Highway 12 and risked being hit by vehicular traffic. Starting in 2016, CAHA staff began collecting standardized data on emergent hatchlings that displayed disrupted sea-finding. Although data were not collected for all nests in 2016, for the 115 nests for which data were collected, 28 had evidence of disrupted sea-finding affecting a total of >500 hatchlings. More efforts could be made to document this issue and develop plans to reduce or eliminate the presence of artificial lighting during months when turtles are laying eggs and hatchlings are emerging from nests. Fortunately, much of the beach within the Seashore has no or little artificial lighting, making this a localized rather than pervasive problem.

Conclusions

Overall, management of sea turtle nests and nesting habitat on Cape Hatteras National Seashore is appropriate and effective. Hatching success of relocated nests (67% in the data in Table 4) is comparable to values recorded elsewhere, which are typically around 70% (Dodd 1988; Foley et al. 2006). Success of non-relocated nests (58% in the data in Table 4) is lower than typical values, although higher than some reported values (e.g., 52% for non-relocated loggerhead nests in Florida, Brost et al. 2015). Hatching success on Cape Lookout National Seashore is similar. Higher levels of hatching success have been achieved on other North Carolina beaches south of the two Seashores. Although some of the difference likely can be attributed to warmer temperatures to the south and to the higher energy of the beaches on the Seashores compared to those to the south, there is little doubt that hatching success on CAHA could be increased by relocating more nests. However, this would come at the cost of reduced fitness of hatchlings (see above). There is no recognized standard for a required level of nest success necessary for population stability. Reproduction of sea turtles is characterized by high levels of mortality at the egg and hatchling stages, such that population size is insensitive to variation in nesting success (Crouse et al. 1987; Crowder et al. 1994; Heppell et al. 1996). It is therefore unlikely that increasing nest success by relocating more nests would have an impact on the size of sea turtle populations. We conclude that the current criteria for nest relocation and resulting frequency of nest relocation that matches the Seashore's goal for such activity (< 30% of nests relocated) are appropriate. The level of productivity resulting from the nest management program likely is sufficient to maintain sea turtle populations.

Populations of the only two sea turtle species for which CAHA provides significant nesting habitat, green and loggerhead sea turtles, are both increasing (Mazaris et al. 2017). The Seashore is contributing an increasing share of the growing number of sea turtle nests in North Carolina (Table 1), and exceeding all population goals for sea turtles. In these circumstances, there is no compelling reason to alter the management program for sea turtles, and it might be appropriate to set new, more ambitious goals for the sea turtle management program.

Key Uncertainties and Monitoring and Research Needs. Currently, we see just one key uncertainty and research need that merits attention, artificial light at night on the nesting beaches. More data on this issue are needed to determine the frequency with which artificial light is disrupting sea-finding of hatchlings and nesting of females and assess the extent to which such disruption might be reduced.

Although the sea turtle management program has been successful, it is important to continue monitoring population size and hatching success because circumstances could change in the future in ways that would necessitate a reevaluation of management. First, if hatching success begins to decline, a reexamination of nest management would be in order. One can imagine increasing sea level and severity of storms due to climate change resulting in more loss of nests, given continuing human alteration of barrier island dynamics. This eventuality might compel changes in the nest relocation program. Second, a reexamination of sea turtle management might be in order if the number of nesting turtles begin to decline. If sea turtle populations begin to decline it will likely be due to factors other than productivity, but loss of nesting habitat due to sea level rise, increased severity of storms and human activity conceivably could be a factor. Third, population growth might begin to asymptote indicating an approach to carrying capacity (Caillouet et al. 2018). In this case, too, availability of nesting habitat should be examined as one of the possible factors limiting sea turtle populations.

Impacts of Barrier Dynamics on Beach-nesting Species

A recurring theme in this report is the possible role that habitat limitation plays in the population dynamics of beach-nesting birds and sea turtles on Cape Hatteras National Seashore. The role that habitat availability plays in determining population sizes and productivity receives far less attention than the roles of factors such as predation and human disturbance, yet habitat may be just as important as these more highly publicized and more readily observed influences. Habitat availability is tied to barrier island dynamics, specifically the dynamics of inlet formation and overwash, which create new habitat and affect existing habitat (see above). Here we integrate our findings across species to provide a synthesis of both short-term and long-term effects of how anthropogenic influences on barrier island dynamics may be influencing populations of beach-nesting species through impacts on habitat availability.

Short-term Impacts

There are both positive and negative short-term effects of barrier island dynamics on beach-nesting species. Storms and overwash events can have direct, negative impacts on reproduction. Inundation is a significant cause of nest failure in both beach-nesting birds and sea turtles, and beach erosion causes some sea turtle nests to fail as well. However, recurring overwash is essential to maintaining the relatively vegetation-free, high quality sand flat habitat on which many beach-nesting species depend. This is particularly important for Piping Plovers. Overwash and inlet formation can create new habitat in the form of new sand flats as well, thus increasing

the carrying capacity for species that depend on such habitat. Disruption of these processes can result in development of vegetation and ultimately dune formation in sand flats and thus loss of high quality habitat.

We conclude that with sea level rising and the simple barrier islands that provide most of the habitat used by beach-nesting species becoming increasingly narrower, nest loss due to inundation likely will become more frequent. Increasing frequency of strong storms due to climate change may contribute to increasing vulnerability to nest inundation, especially for sea turtles. On the other hand, because tropical cyclones generally occur outside their breeding season, beach-nesting birds might benefit from more frequent strong storms due to their positive impact on habitat availability, and possibly their capacity to reduce predator populations (Schulte and Simons 2016), despite some loss of nests to storms that occur during the breeding season.

Long-term Impacts

Long-term impacts on beach-nesting species take the form of alteration of barrier island dynamics by anthropogenic activities that result in reduction of available habitat. The changes to the natural system that affect Cape Hatteras National Seashore today go back to actions taken in the 1930s to 1950s. Three major actions in particular have resulted in the loss of habitat over the long term and constrain current management:

- The U.S. Congress directed the National Park Service to build and vegetate dune dikes on the North Carolina barrier islands from the Virginia line to Ocracoke during the 1930s.
- The dune dikes were built on Federal land on the ocean side of all villages so that the National Park Service could “protect the villages through time”.
- The transfer of land deeds from the state to the federal government gave the rights to the North Carolina Department of Transportation to build and maintain a highway corridor from Nags Head to Ocracoke for what was to become NC Highway 12.

These decisions made in the absence of an understanding of the natural barrier island dynamics resulted in changes that impede the natural cycle of creation of habitat for beach-nesting species. The consequence is a loss of critical habitats and narrowing and increased flooding of the simple island segments within which those habitats occur. The attempt to engineer and stabilize the barrier islands in order to protect the built environment and maintain the local economy continues, but becomes more difficult as the simple island segments continue to narrow and sea level continues to rise.

Although the conflict between human engineering of the system and barrier island dynamics has existed for decades, it is not inevitable that it must persist in the future. In theory, changes to the built environment could alleviate the conflict and allow natural habitats to redevelop within CAHA within open areas between village segments, in the same manner as has occurred on CALO since the 1970s. The result would be increased carrying capacities for beach-nesting species that might lead to larger population sizes. However, this would require major changes in the attitudes and character of the coastal economy, changes that are well beyond the management authority of the National Park Service. The following exemplify the kinds of changes that would be required to achieve this result:

- Take the highway off the barrier island segments between the villages by either raising the road on a system of causeways and bridges or moving the paved road onto a back-barrier causeway.

- Let the beaches move in response to storms and allow overwash sands to build island elevation with a natural and open, rather than anthropogenically reinforced, dune system.
- Whenever and wherever ephemeral inlets open or overwash fans develop, allow them to build critical island width and associated habitats with active flood-tide delta deposits behind the barrier, and elevation on top of the barrier.
- In the villages, establish a land-based buffer zone along the moving oceanfront and sound-front shorelines as common space for public, water-based tourism. This would require rewriting land-ownership laws to accommodate a moving oceanfront.

If such actions were taken, most of the island segments within the Seashore, which are the simple barrier islands that occur between the villages, would revert to natural barrier islands similar to CALO with vast areas of critical habitats that would be restored and maintained for the long term. Continuing on the current management path will almost certainly lead to loss of island beaches and net island height and width. Thus, the carrying capacity for beach-nesting species within CAHA likely will decrease further. The impact this will have on population sizes depends on whether these species are habitat limited. For any species currently at carrying capacity, neither management actions nor changes in abiotic or biotic factors positively affecting vital rates will result in an increase in population. However, we consider it unlikely that any of the species we examined have reached their carrying capacity on CAHA, with the notable exception of the Piping Plover. More likely is that reduced carrying capacity will prevent populations from returning to historic levels once the issues with vital rates that caused populations to decline are addressed. This possibility is particularly applicable to American Oystercatchers.

Human Dimensions of Conservation of Beach-nesting Species

The human dimensions of wildlife is “a field of study that **applies the social sciences** to examine human-wildlife relationships, and, in doing so, provides information that contributes to effective wildlife conservation efforts” (Manfredo 2008). The National Park Service has a long history of focusing on the human dimensions of natural resource management associated with visitor use and satisfaction, and a shorter history related to human dimensions of wildlife management. The Social Science Program of the National Park Service was created in 2002, as part of the Director’s Order #78 (<https://www.nps.gov/policy/DOrders/DO78.htm>) which called for establishing NPS social science. Previous to that Director’s Order, in 1996, NPS originally approved a plan for social science, “Usable Knowledge: A Plan for Furthering Social Science and the Parks”, which created a small Washington, DC office to coordinate social science research for the Park Service. The 2002 order stated:

“proper management of the National Park System requires accurate, science-based understanding of the relationships between people and parks in order to protect park resources unimpaired and provide for public enjoyment. Social science research in support of the National Park Service (NPS) mission is an important function that provides new and helpful information upon which to base management decisions.”

The Program was described as conducting economics, geography, psychology, political science, sociology, and interdisciplinary research. As of 2016, the Social Science Program was conducting activities related to socioeconomic technical support, socioeconomic project

management, economic analyses, information collections review, park visitor-use statistics, and visitor satisfaction (<https://www.nature.nps.gov/socialscience/docs/ProgramBriefSSP01-28-16.pdf>). Additionally, NPS is contracting a group of researchers called the RCG through an indefinite delivery, indefinite quantity contract for 5 years from 2015 to 2020 to conduct visitor use and social science research. According to their FY 2015 Annual Report, the RCG was conducting 19 projects in 15 park units across six of the seven NPS regions. These projects relate to visitor use assessment, economic impact/welfare, and transportation studies. Further, from 2007- 2016, NPS employed a Human Dimensions Program Manager in their Biological Resource Management (BRM) Program. This Human Dimensions Program in BRM was created to address the “critical interface between the human and ecological dimensions of biological resource management.” This program focused on supporting park units with human dimensions of wildlife research, focusing on “values and desires for biological resources and associated management actions” (https://www.nature.nps.gov/biology/human_dimensions/). Their initiatives (https://www.nature.nps.gov/biology/human_dimensions/Program_Initiative.cfm) included stakeholder engagement and public participation especially related to human-wildlife conflict; understanding risk communication, education, and persuasion associated with wildlife disease and human-wildlife interaction; recreation management to reduce threats to resources, including wildlife disturbance; and citizen science and stewardship. Additionally, recently NPS units, including Yellowstone National Park (<http://yellowstoneinsider.com/2016/06/22/an-interview-with-yellowstone-social-scientist-ryan-atwell/>) and Golden Gate National Park (<https://www.usajobs.gov/GetJob/PrintPreview/458471600>), have begun to hire social scientists. Still, with 400-some units and just a few social scientists on staff, NPS has yet to employ social science to its full potential.

Most NPS units, such as Cape Hatteras National Seashore, have made limited use of social science research. CAHA’s human dimensions emphasis thus far has been on characterizing visitor use generally, rather than the human dimensions of wildlife management per se. In 2003, Vogel song published the *Cape Hatteras National Seashore Visitor Use Study*, which was sponsored by the National Park Service. That same year Neal wrote *The Lower Outer Banks of North Carolina: Results of a survey of residents, nonresident property owners, and visitors*, which was sponsored by the Outer Banks Preservation Association. A peer-review panel with five academics specializing in visitor use surveys, survey design and methods, economic impact assessment, and human dimensions of fisheries provided an independent review of these two publications for NPS. According to their 2008 report (Gramann 2008), the panel concluded that the technical reports did not “provide a social scientific basis for estimating ORV use at Cape Hatteras National Seashore or the economic impact of visitor spending associated with ORV use” (p. 2). Their report provides extensive detail on the numerous limitations and issues of these studies. Additionally, the NPS Visitor Services Project hired the University of Idaho to produce *Outer Banks Group Parks Visitor Study, Cape Hatteras National Seashore Visitor* (2003) and *Cape Hatteras National Seashore 2008 Visitor Survey Card Data Report*. Information from these two studies was used in the Visitor Experience section of an Environmental Assessment of ORV use within the Seashore issued in 2016 (see below). In 2010-2011 RTI International published the reports *Visitor Intercept Survey: Off-Road Vehicle Management, Cape Hatteras National Seashore; Ramp Counts: Off-Road Vehicle Management, Cape Hatteras National Seashore; Business Survey: Off-Road Vehicle Management, Cape Hatteras National Seashore; and Benefit-Cost Analysis of Proposed ORV Use Regulations in the Cape Hatteras National*

Seashore (Mansfield et al. 2011). These reports were used to inform the Environmental Assessment.

The 2016 Consideration of Modifications to the Final Rule for ORV Management Environmental Assessment considered morning opening of beaches that are closed to ORV use at night, dates for seasonal ORV routes, and size and location of vehicle areas through three action alternatives. It also examined impacts on visitor use and experience and socioeconomics. The document noted that visitor experience could be impacted by the modification, enhancing recreation experiences for some and diminishing them for others. Further, the local economy could be impacted by these decisions.

Additionally, social science researchers have explored the conflict surrounding the participatory processes applied at CAHA to address the issue of ORV use of beach habitat. In his 2009 dissertation, Merritt evaluated the Negotiated Rule-making process to create the Off Road Vehicle Management Rule at Cape Hatteras National Seashore. Merritt (2009) described that the Seashore was only the fourth NPS unit to use this public involvement approach to manage conflict between recreational use and biological resources, such as beach-nesting wildlife. Through interviews with park staff and negotiated rulemaking participants and observation of the process, he found that the process increased the knowledge of participants about democratic process and of Seashore staff of public engagement, and enhanced the relationship between participants and park management. Yet, stakeholders were frustrated with the rulemaking process and became more negative about it over time, likely because they were aiming for consensus, rather than information sharing.

Human dimensions research conducted elsewhere has explored topics that could have important insights for CAHA in managing potential human disturbance of wildlife due to activities of visitors. Multiple studies indicate beachgoers have generally positive attitudes towards shorebirds (e.g., Comber and Dayer 2019a; Glover et al. 2011). Also, research has illustrated the importance recreationists place on protecting shorebirds, ranking protecting endangered species and the environment as well as closing areas to protect shorebirds higher than recreational opportunities such as jogging and dog walking (Burger et al. 2017). Yet, there is often a disconnect between beach recreationists' perceived impacts of their activity and the actual activity (e.g. Priskin 2003 related to beach driving, boating, and surfing). For example, those who participate in beach driving believe there is less impact from their activity than those who do not participate in driving believe. Similarly, Tharpa (2010) found that those who drive on beaches do not believe that driving has significant negative impacts. Beyond just beach driving, recreationists tend to believe other user groups cause more disturbance to wildlife than their own user group (Taylor and Knight 2003).

As for support for restrictions to minimize disturbance to shorebirds, Comber and Dayer (2019b) surveyed land managers at over 100 sites within Important Bird Areas across the Atlantic Flyway, finding issues with public compliance with restrictions, especially those related to dog walking and flying unmanned and wind-powered aircraft. Researchers have found that the underlying beliefs of individuals influence whether they follow restrictions (e.g., Blouin et al 2015). Additionally, norms play a role in compliance (Comber and Dayer 2019 a). For example, observing others not leashing their dogs, paired with a lack of enforcement can limit compliance of beach recreationists with rules (e.g., Bowes et al. 2015). Birdwatchers and anglers have been found to be more compliant with beach closures for shorebirds than dog walkers, joggers, and off-road vehicle users (Burger and Niles 2013). Research has also explored demographic

differences of individuals who support shorebird conservation measures, finding women and out-of-state visitors to be more tolerant of restrictions (Priskin 2003; Rutter 2016).

Mengak et al. (2019) provide a more complete review of the social science related to human disturbance of shorebirds in coastal areas. This document provides guidance to managers and describes best practices that can be used to manage shorebird disturbance. One of the best practices emphasizes the value in using decision support frameworks that integrate social and ecological knowledge. Through such decision-making approaches stakeholders are engaged in defining the problem and objectives, as well as associated social and ecological information needs. Such approaches could be useful to managers of Cape Hatteras National Seashore in reducing conflict over integrating provision of high-quality habitat for beach-nesting birds and sea turtles with recreational uses of beach habitats.

Conclusions

While Cape Hatteras National Seashore has addressed human dimensions research in terms of visitor use and experience, this is just a first step toward having adequate social science information to inform decisions. Human dimensions research information that would be useful to the park includes information on people's response, in terms of behavior, attitudes, motivations, and satisfaction, to management changes. Of particular importance is how visitors respond (in terms of compliance, conflict, or satisfaction, depending on park goals) to management actions taken to protect beach-nesting birds and migrating shorebirds. More generally, the Seashore would benefit from information on the importance recreationists place on protecting beach-nesting wildlife, visitor attitudes toward wildlife, and how experiences viewing wildlife (or simply awareness of their existence at the park) enhances park visitation experiences. This information could better inform future management, as well as interpretive, activities.

Summary and Conclusions

Populations of beach-nesting species on Cape Hatteras National Seashore have exhibited varying trajectories in recent years. The number of sea turtles nesting on the Seashore's beaches is increasing, accounting for an increasingly larger share of an increasing regional population. In contrast, the number of terns nesting in colonies on the beaches is declining, and species-specific patterns bear no consistent relationship to regional trends. The Piping Plover population on CAHA experienced a large decline, paralleled by a similar decline at Cape Lookout National Seashore, from the mid-1990s through the early 2000s that was specific to the two Seashores rather than part of any broader, regional decline. Both populations subsequently recovered to previous levels. The American Oystercatcher population experienced a decline that was specific to CAHA during a period, 1999-2006, when the regional population, including that on CALO, was stable. The population subsequently increased, but not to pre-decline levels. For all these species, the factors that are most relevant to their population dynamics are carrying capacity (i.e., habitat availability), and fecundity (i.e., offspring produced per nest).

Anthropogenic structures and activities within CAHA disrupt barrier island dynamics, specifically the processes of inlet formation and overwash that create habitat for beach-nesting species. This no doubt has reduced the carrying capacity for these species on CAHA. However, populations of the species we examined do not appear to be limited by availability of nesting habitat currently, with the notable exception of the Piping Plover. Nesting habitat availability is unlikely to limit populations of sea turtles or beach-nesting colonial waterbirds in the foreseeable future. In contrast, carrying capacity could become limiting for American Oystercatchers should

the CAHA population increase sufficiently, and may well prevent the population from reaching pre-decline levels. Investigating the carrying capacity of nesting habitat, as well as spatial variation in productivity, are the most important research needs for oystercatchers. Current populations of Piping Plovers and American Oystercatchers are very close to the Seashore's short-term population objectives for these species (15 and 30 pairs respectively), but long-term population objectives (30 and 45 pairs respectively) may be unrealistic due to lack of sufficient carrying capacity. In contrast, for sea turtles, for which carrying capacity is not an issue, all population objectives have been surpassed and more ambitious objectives are likely attainable.

Management options for increasing carrying capacity for Piping Plovers or American Oystercatchers are severely limited. Anthropogenic attempts to create nesting habitat are of little use, due to vegetation regrowth and limitations in scale. Reducing anthropogenic impacts on barrier island dynamics to promote more frequent overwash events, for example by reducing artificial dune maintenance or relocating Highway 12, likely would be effective and are possible in theory, but implementing such measures is well beyond the authority of the National Park Service. In the future, sea level rise is expected to exacerbate habitat loss, but climate change driven increases in the frequency of especially powerful hurricanes and other storms promise to increase the rate of formation of new habitat.

Productivity is the most important vital rate for beach-nesting species on Cape Hatteras National Seashore, and is more amenable to management. The primary factors affecting productivity are flooding, predation and human disturbance. Inundation is a significant cause of nest failure in all beach-nesting species, but does not appear to be a source of variation in productivity that is tied to population declines, unlike the other two factors. With sea level rising and the simple barrier islands that provide most of the habitat used by beach-nesting species becoming increasingly narrower, nest loss due to inundation likely will become more frequent in the future. Increasing frequency of strong storms may increase nest inundation for sea turtles as well, but because hurricanes occur outside their breeding season, beach-nesting birds are not as vulnerable to increasing storm intensity as sea turtles.

There is considerable evidence that human disturbance and predation have had important impacts on productivity of beach-nesting species on Cape Hatteras National Seashore. Impacts of human disturbance and predation on productivity likely drove the decline of the American Oystercatcher population on CAHA during 1999-2006, and implementation of pedestrian and vehicle closures and predator removal likely are responsible for subsequent improved productivity and associated population stability. Productivity is strongly linked to population dynamics in this species because recruitment is largely internal. An alternative to lack of carrying capacity to explain the failure of the oystercatcher population to recover to pre-decline levels is that productivity is high enough to maintain the population, but not high enough to support population growth. It is unlikely that human disturbance is limiting productivity to levels below those required for population growth, given the effectiveness of management in limiting direct mortality of nests or chicks. It is much more likely that predation is limiting productivity. Predator removal appears to be effective in increasing productivity, but is employed at varying levels from year to year. Its sustained employment at an appropriate level could conceivably raise productivity to the level required for population growth. The best estimate of the level of annual productivity required for population growth (> 0.63 chicks per pair) is higher than the Seashore's short-term and long-term objectives (0.4 and 0.5 chicks per pair respectively).

Predation emerged as the leading concern for beach-nesting colonial waterbirds as well, but with much less certainty than with oystercatchers. In this case, it is not clear whether recent

reductions in numbers of beach-nesting birds represent population declines or shifts to other, safer nesting locations such as dredge spoil islands. In either case, high levels of predation by mammals rather than human disturbance seems the most likely cause of reduced beach-nesting as closures appear to be effective in reducing human disturbance to reasonable levels. However, ability to identify the cause of declining numbers is limited by lack of measures of reproductive success such as hatching and fledging success. Obtaining such data is the greatest research need for beach-nesting colonial waterbirds.

The recent declines in numbers of beach-nesting colonial waterbirds make it difficult to reach population goals established for CAHA when numbers were higher. The decline in nests of Least Terns is of most concern as this species has increased regionally and CAHA should be a prime breeding site due to its location in the middle of the species' breeding range. Goals for Common Terns appear overly ambitious for a population of this regionally declining species at the southern edge of its breeding range.

In contrast to American Oystercatchers, recruitment of Piping Plovers appears to be largely external, and although population size tracks productivity somewhat on CALO, it clearly does not on CAHA. Variation in productivity cannot account for the decline of the CAHA plover population nor its subsequent recovery, and productivity remains well below short-term (1.0 chicks per breeding pair) and long-term (1.5 chicks per breeding pair) objectives established by CAHA. Productivity of Piping Plovers on CAHA is limited not by hatching success but by survival of chicks to fledging. Management in the form of predator removal and protective buffers has not been effective in raising chick survival to desired levels, and the available evidence does not support the hypothesis that mammalian predation or human disturbance is limiting chick survival. However, it is clear that without this management productivity likely would be even lower, and perhaps almost nil. That foraging conditions for chicks are poor is a viable hypothesis for explaining the low productivity of Piping Plovers on CAHA. The Seashore is at the southern limit of the species' range, and conditions for Piping Plovers may be marginal there, such that the productivity objectives for this species may be unachievable. Examining foraging conditions for chicks and obtaining more data on survival of adults are the most critical research needs for Piping Plovers.

Cape Hatteras National Seashore is most important to Piping Plovers not as a breeding site, but as a migratory stopover. South Point on Ocracoke Island in particular is a critical site for migrating Piping Plovers: an estimated 10% of the global population and 15% of the Atlantic Coast population of the species use South Point during fall migration. The Seashore is an important migratory stopover and wintering site for other shorebird species as well, most notably the federally Threatened Red Knot, for which South Point is an important wintering site. The primary concern with wintering and especially migrating shorebirds is protecting them from human disturbance, to which they are susceptible. It is not clear whether existing closures at South Point during fall and winter protect Piping Plovers and Red Knots from human disturbance sufficiently. No other areas meriting special protection are evident as migrating and wintering shorebirds are widespread within CAHA, and species vary in the locations where they are most abundant.

Overall, closures protecting beach-nesting birds from human disturbance appear to be necessary to maintain the productivity of those species, but not sufficient to raise productivity to desired levels. For some but not all species, high rates of predation likely limit productivity. Populations of many predators such as raccoons and crows likely are elevated by food subsidies resulting from human activity, increasing predation rates above historical levels. It is also

possible that visitation rates at CAHA are such that violations of closures continue to result in occasional detrimental impacts on productivity of some species. Still, controlling predator populations is by far the most promising avenue to increase productivity of beach-nesting birds. One can imagine positive impacts on all species with the likely exception of the Piping Plover, whose productivity appears to be limited by factors other than human disturbance and predation.

Cape Hatteras National Seashore is a major nesting area for loggerhead sea turtles and a minor one for green sea turtles. The appropriate management goals for sea turtles at CAHA are to provide sufficient beach habitat for nesting for these two species, and to enable a level of productivity within that habitat sufficient to support stable or increasing populations. Management of sea turtle nests and nesting habitat on CAHA is appropriate and effective for meeting these objectives. There is little doubt that hatching success could be increased by relocating more nests, but this likely would come at the cost of reduced fitness of hatchlings. Given the life history of sea turtles, which results in population size being relatively insensitive to variation in nesting success, it is unlikely that increasing nest success by relocating more nests would have an impact on the size of sea turtle populations. We therefore conclude that the current criteria for nest relocation and resulting frequency of nest relocation, which match the Seashore's objective for nest relocation, are appropriate. We further conclude that the level of productivity resulting from the nest management program is sufficient to maintain sea turtle populations. Regional populations of loggerhead and green sea turtles are increasing, and the Seashore is contributing an increasing share of the growing number of sea turtle nests in North Carolina. In these circumstances, there is no reason to alter the existing management program. However, we perceive one issue that merits attention, artificial light at night on the nesting beaches. More data on this issue are needed to determine the frequency with which artificial light is disrupting nesting of females and sea-finding of hatchlings, and to assess the extent to which such disruption might be reduced.

The challenge to managers of CAHA is to integrate protecting the Seashore's native fauna and flora with providing recreational activities for visitors. Conflicts typically involve how this integration is accomplished to achieve both objectives. In particular, there is disagreement over whether restrictions on pedestrian and ORV use of beach habitats benefit beach-nesting species sufficiently to justify negative impacts on some recreational activities. We have concluded that there is ample evidence of those benefits. There is much less information about impacts on recreational activities. While CAHA has addressed human dimensions research in terms of visitor use and experience, this is just a first step toward having adequate social science information. Additional human dimensions research could provide useful information about how visitors respond (in terms of compliance, conflict, or satisfaction) to management actions taken to protect beach-nesting birds and turtles. More generally, the Seashore would benefit from information on the importance recreationists place on protecting beach-nesting wildlife, visitor attitudes toward wildlife, and how experiences viewing wildlife (or simply awareness of their existence at the park) enhances park visitation experiences.

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Appendix 1: Barrier Island Geomorphology and Dynamics

Barrier island segments within Cape Hatteras National Seashore can be divided into two basic types: simple and complex. This determination is based upon the geomorphology of the barriers, which in turn is a product of the evolutionary history, available sediment supply, and physical dynamics operating upon the islands. Riggs and Ames (2003; 2007), Riggs et al. (2011),

and Schupp (2015) defined simple barrier island segments as low and narrow barriers dominated by inlet and overwash dynamics (Figure A1A and A1B). Simple barriers are sediment poor with little ‘new’ sand being supplied to the beach through time. Consequently, they tend to be extremely dynamic with common and extensive modern and ancient inlet flood-tide deltas extending into the back-barrier estuary and overwash fans building island elevation. These barriers tend to be relatively young in that the modern topography and its present habitats have formed within the last 500 years of the Quaternary and continuously evolve with every storm and human modification. Examples of simple barrier islands include the northeast half of Ocracoke Island, as well as island segments from the villages of Hatteras to Frisco, Buxton to Avon, Avon to Salvo, Rodanthe to Oregon Inlet, and Oregon Inlet to Whalebone Junction (Smith 2004; Ricardo 2005; Rosenberger 2006; Smith 2006; Twamley 2006; Culver et al. 2005; 2007; Hale 2008; Smith et al. 2008). The general distribution of simple barrier islands within the Seashore is summarized in Figure A2.

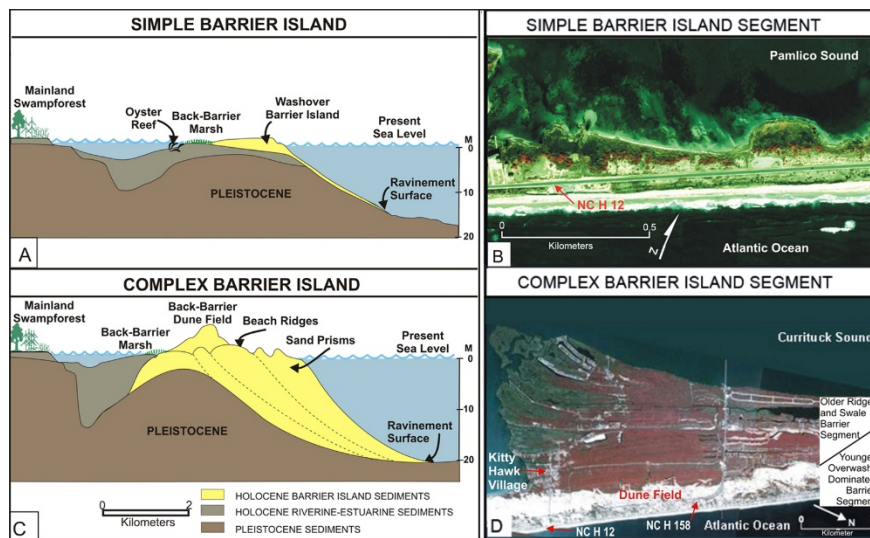


Figure A1. Panels A and C. Schematic cross-sectional diagrams show a simple, inlet and overwash dominated barrier island and complex barrier island, respectively. Panels B and D show aerial photographs of examples of Panels A and C and include the Frisco to Hatteras Village and Kitty Hawk barrier island segments, respectively. Figure is modified from Riggs and Ames (2007).

Riggs and Ames (2003; 2007), Riggs et al. (2011), and Schupp (2015) defined complex barrier islands as substantially older islands that are high and wide due to major inputs of new sand into the coastal system at various times in their evolutionary history (Figure A1C and A1D). Complex islands generally are characterized by a younger, overwash-dominated component that has migrated into and welded onto an older barrier island segment composed of different sets of geomorphic features (Mallinson et al. 2005; 2008; 2010b; Riggs et al. 2011). Consequently, these sediment-rich islands have a large volume of sand that occurs in beach ridges and dune fields on the mid- and back-side of the island. Due to this large volume of sand, normal storm surges have little to no potential for opening new inlets through the island and oceanic overwash only occurs along the modern, ocean side of the barrier. Thus, salt spray is minimal allowing extensive maritime forests to develop towards the sound side of these island segments. These high and wide islands tend to be much older and consequently, their habitats are considerably

more stable and of less importance for the animal species discussed in this report. For example, the northern-most beach ridges on Buxton Woods began forming about 1,700 years ago. Beach ridges get younger to the south where the modern beach ridge is forming today (Mallinson et al. 2018).

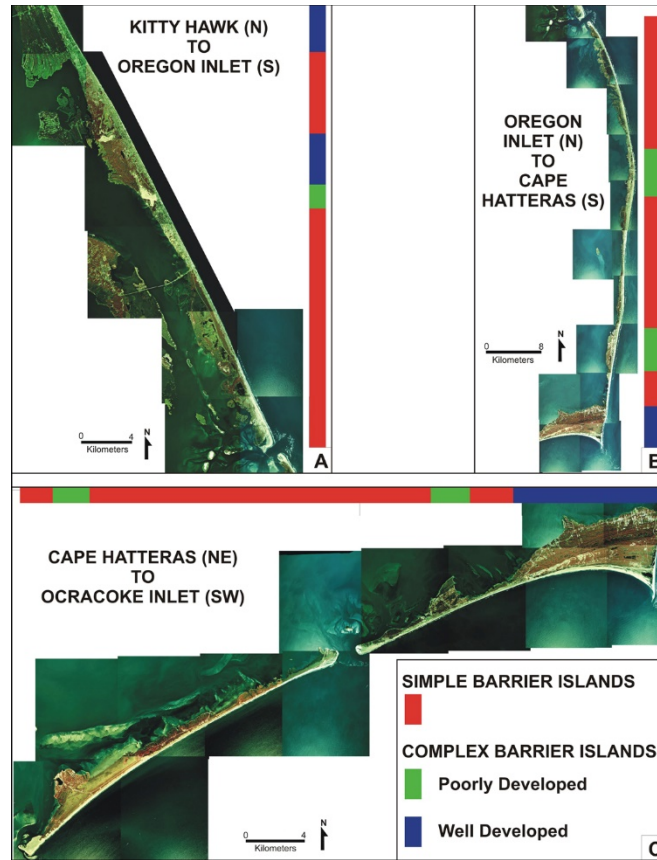


Figure A2. This 1998 aerial photograph mosaic shows the general distribution of simple and complex barrier island systems from Kitty Hawk on the north to Ocracoke Inlet on the south. Figure is from Riggs and Ames (2007).

Complex barriers are further subdivided along a continuum between two end members that range from well-developed to poorly-developed complex barriers (Figure A2). Buxton Woods is an example of a well-developed complex island, while Ocracoke and Hatteras villages are moderately-developed complex islands, and the villages of Rodanthe, Waves, Salvo, and Avon are poorly-developed complex islands. It is interesting to note that all of the early and ongoing urban development occurs on the more stable complex barrier islands within the Seashore while the bulk of the Seashore (with the exception of Buxton Woods) is located on the highly dynamic simple barrier islands (Figure A1). Consequently, it is the simple barrier islands that in the past contained vast areas of the shallow shorebird and turtle habitat that has suffered major losses due to creation of a highway corridor protected by sand-bagged dune dikes. However, there is potential for increasing critical habitat in the near future if there are major changes to our management approach to the Seashore's barrier island system.

Inlet and Overwash Dynamics and Resulting Habitats

Major portions of the Cape Hatteras National Seashore barrier segments (e.g., Bodie and Pea islands, barrier segments from the villages of Salvo to Avon, Avon to Buxton, and Frisco to Hatteras, as well as the northeastern portion of Ocracoke Island) are inlet and overwash dominated barrier island segments. However, they have all experienced island narrowing for many decades as a direct result of the continuous rebuilding of the constructed dune-ridges to protect Highway 12 (Culver et al. 2005; Riggs et al. 2009; 2011; Smith et al. 2008; Riggs and Ames 2011). Since the late 1930s, the practice of constructing dune-ridges has prevented inlets from forming and minimized the overwash processes leading directly to island narrowing, as well as island lowering relative to rising sea level. The latter situation results in more frequent island flooding in response to wind tides and abnormal spring tides on Pamlico Sound.

In spite of the constructed dune-ridges and almost continuous efforts of bulldozer fleets, ongoing ocean shoreline recession has led to numerous relocations of Highway 12 in successive landward positions (Riggs and Ames 2003; 2011; Riggs et al. 2009; 2011). Because simple barrier island segments are being actively eroded from both sides, many narrow portions of the barriers have Highway 12 located at the edge of the estuarine marsh or adjacent to the Pamlico Sound shoreline. In order for these barrier segments to maintain themselves over the short-term (months to years) and evolve over the long-term (decades to centuries) in response to ongoing sea-level rise, inlets and overwash dynamics need to occur. Human interventions that prevent these processes in order to maintain the status quo only accentuate the rate of barrier island narrowing and eventual collapse into intertidal shoal systems.

Since barrier islands form a sand dam on the ocean side of the drowned river estuarine system, inlets in the barrier sand dam are essential to let the fresh water streams and rivers of the many drainage systems flow into the sea. In addition, when storm surges impact the barrier sand dam, inlets become important conduits for large portions of the flooding storm surge to flow into the estuaries and most importantly, back out again as the storm passes. Thus, inlets are “self-adjusting safety valves” in the barrier island sand dam. The size of an inlet opening increases during storms to let increased water volumes to flow in and/or out. When the storm and river floods abate, then inlets close back to an equilibrium point that is dictated by the volume of normal freshwater input into the estuaries by the riverine system and the natural hydrodynamics of the estuarine-oceanic system. Thus, inlets “breathe” during storm events in direct response to changing storm and discharge conditions.

Inlet dynamics create vast habitats of coastal flats and ponds that range from shallow subaerial grass flats to supra-tidal microbial flats, inter-tidal marshes, and sub-tidal sand shoals (Figure A3). This includes building an extensive sequence of shallow sand shoals (flood-tide deltas) on the estuarine side of the barrier islands. These sand shoals form an essential base for the barrier island to migrate over in response to ongoing rise in sea level. Flood-tide deltas (FTDs) form on the sound side, and ebb-tide deltas (ETDs) form on the ocean side. These tidal deltas are important sand deposits that store sediment necessary for the long-term evolution of the barrier islands. The deltaic sand bodies are important parts of the coastal sediment budget, and during storm events they bypass sand up and down the ocean shoreline, as well as in and out through the inlet. The multiple channels on both the FTD and ETD flow into a common central inlet throat that occurs in line with the barrier islands. This inlet throat is the narrowest and deepest portion of an inlet system with the highest current energy. This energy is dissipated into the larger standing water bodies on either side of the inlet throat.

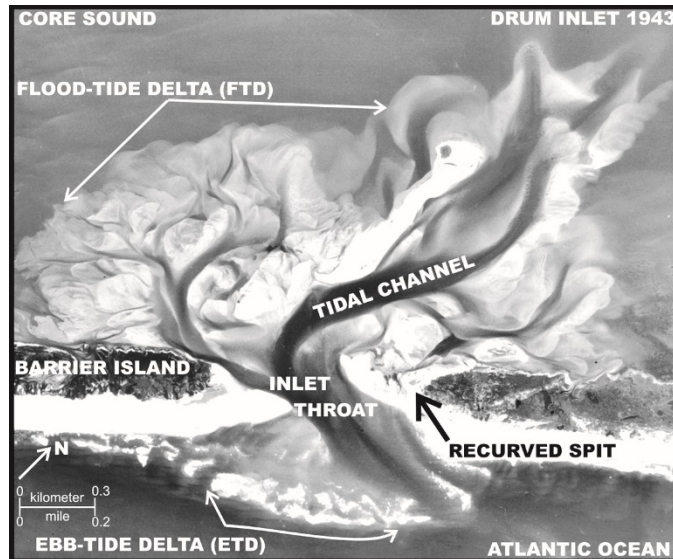


Figure A3. A 1943 aerial photograph of Drum Inlet on Core Banks shows development of a small ebb-tide delta on the ocean side of the barrier island. An extensive flood-tide delta, with its vast sand shoal habitats and associated tidal channels, produces the base for a two- to threefold increase in island width on the estuarine side of the barrier island. Figure is from Riggs and Ames (2007).

Inlet features are high energy coastal systems that respond differentially to changing weather patterns, storms, sediment supply, and underlying geology. Modern channels of Oregon, Hatteras, and Ocracoke inlets, as well as the many paleo-inlets demonstrate extremely dynamic responses at both the scale of individual storms and the decadal scale (Moslow and Heron 1981; Everts et al. 1983; Riggs et al. 1992; 1995; Smith 2006; Riggs and Ames 2007; Mallinson et al. 2010a; Riggs et al. 2008; Riggs et al. 2011). It is not possible to predict the absolute location of a future inlet or series of inlets through the barrier islands due to three interacting sources of variability: 1) the pattern and characteristic of each storm or sets of storms; 2) amount and status of human modification along each barrier island segment; and 3) the geologic underpinnings and geographic variability within any barrier segment. However, Figure A4 presents an Inlet Vulnerability Index (IVI) for the Seashore that puts a realistic likelihood on the potential for inlet development if and when the right storm or series of storms impact that coastal segment (Mallinson et al. 2010a; Riggs et al. 2009; Riggs and Ames 2011).

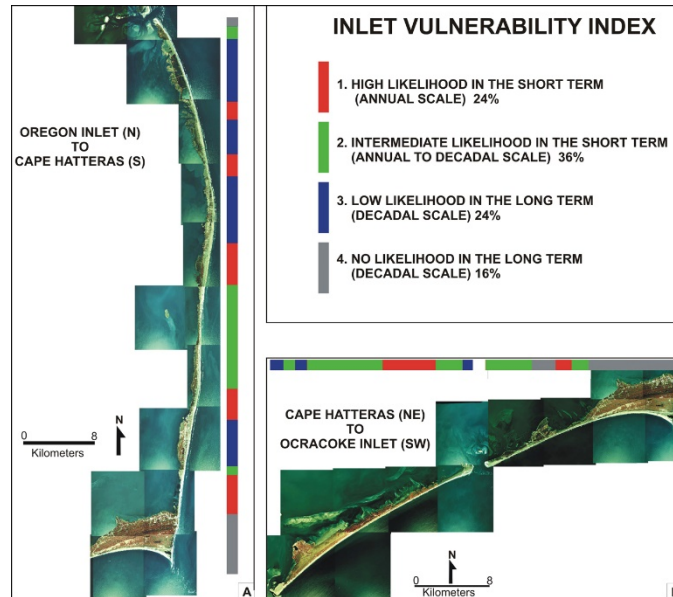


Figure A4. The Inlet Vulnerability Index summarizes the likelihood for inlets to open between Oregon Inlet and Ocracoke Inlet, assuming that the “right storm or set of storms” directly impact this coastal region. Panel A is from Oregon Inlet on the north to Cape Hatteras on the south, whereas Panel B extends from Cape Hatteras on the northeast to Ocracoke Inlet on the southwest. The percentages indicate the fraction of barrier islands with specific likelihoods of inlet formation. Figure is modified from Riggs et al. (2009; 2011).

Historic Impacts of Sea Level and Storms

The last period of geologic time, the Quaternary (~2.6 million years ago to the present), is known as the Ice Ages (Imbrie and Imbrie 1979; Williams et al. 1998; Bradley 1999; Ruddiman 2008). The Ice Ages were times of severe global climate fluctuations that alternated between “hot-house” and “ice-box” conditions. These global oscillations of warm and cold climates led to the alternating interglacial and glacial episodes characterized by high sea levels and low sea levels, respectively. The last glacial maximum occurred between 25,000 and 18,000 years ago with the North American ice sheet covering most of Canada and the northern United States southward to the Missouri and Ohio rivers (Ruddiman and Wright 1987; Lamb 1988; Pielou 1991; Anderson and Borns 1994). This glaciation formed the Great Lakes, Cape Cod, Long Island, and the great “bread-basket soils” of the midwestern United States. North Carolina had a boreal zone climate that was semi-arid and very stormy, with large braided (i.e., multi-channeled and sediment rich) river valleys and abundant dune fields. Vegetation was scattered and dominated by spruce and fir trees and inhabited by a population of large mammals such as woolly mammoths, mastodons, rhinoceros, tigers, bison, and others. Sea level in North Carolina was about 410 feet below the present ocean level. This placed the coastal system off the edge of the continental shelf and seaward about 40 miles east of Nags Head and 15 miles east of Cape Hatteras (Horton et al. 2009; Riggs et al. 2011; Thieler et al. 2014). Thus, the Coastal Plain was about double its present size with rivers flowing across the present Continental Margin.

After 18,000 years ago the climate warmed, sea level and its coastal system moved upward and landward across the continental shelf to form the modern North Carolina coastal system (Horton et al. 2009). Rising sea level flooded up the river valleys to form the vast estuarine

system and portions of the modern barrier islands began to form by about 3,000 years ago. Since then, the estuaries have continued to expand and flood vast portions of those land areas with lower elevations. The barrier islands continue to erode on the front side and migrate upward and landward. Some of the sand-poor barriers have collapsed into inlets and shallow shoals in response to major storm events and reformed into sub-aerial barriers, in many cases within the last 500 years.

For many millennia before the first English explorers landed on Roanoke Island in 1584, extra-tropical storms, tropical storms, and hurricanes played substantial roles in the processes of changing and rearranging the natural coastal system, as well as impacting the lives of native peoples living within the coastal system. Records of North Carolina's tropical cyclones began to be kept in the early 1850s (Stick 1958; 1983; Barnes 2001). Since then there have been many hundreds of tropical depressions, tropical storms, and hurricanes that passed into the North Carolina coastal system. Figure A5 shows the number of tropical storms and hurricanes that occurred within a 200-mile zone of Cape Lookout between 1853 and 2010. Of these 252 tropical storms, 93 were category 1 through 5 hurricanes that represent an average of about six hurricanes per decade. In addition, North Carolina gets about 30 extra-tropical coastal storms per year. The geometry of North Carolina's coastline, together with its proximity to the confluence of the warm-water Gulf Stream and cold-water Labrador Current off Cape Hatteras, makes the North Carolina coast a serious storm-driven system (Figure A5).

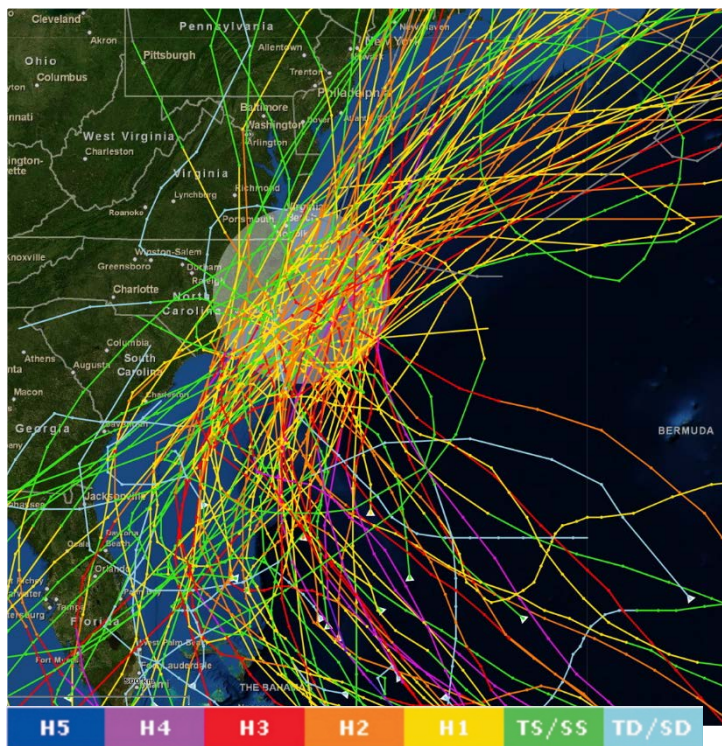


Figure A5. The map shows all of the 252 tropical storm tracks, including 93 hurricanes that occurred between 1853 and 2010 within a 200-mile radius of Cape Lookout (gray circle). The colors for H5, H4, H3, H2, and H1 represent hurricane categories 5 through 1; TS/SS represents tropical to sub-tropical storms; and TD/SD represents tropical to sub-tropical depressions. Data are from NOAA's National Hurricane Center (2013).

The impact of any one storm or set of storms upon the coastal system depends on the storm type, size, strength, duration, and amount of precipitation (Riggs and Ames 2003; 2007). Additional variables include the forward speed of the storm, height of the storm surge, storm track relative to the orientation of the coast, state of the astronomical tides, geometry of the adjacent continental shelf, and the frequency and pattern of subsequent storms.