

2022 Annual Report

EFFECTIVENESS MONITORING AND TARGETED STUDIES IN SUPPORT OF FIMI CONSERVATION MEASURES

Christy N. Wails, Henrietta A. Bellman, Kat M. Black, Audrey DeRose-Wilson, Sharon S. Dorsey, Kristen E. Johnson, Shannon J. Ritter, Sam G. Robinson, Hope L. VanDerwater, Katie M. Walker, Daniel H. Catlin, Sarah M. Karpanty, and James D. Fraser



This report summarizes Virginia Tech's Effectiveness Monitoring of certain conservation measures required under the Biological Opinion for the Fire Island Inlet to Moriches Inlet Stabilization Project

With significant contributions from:

Robert Smith, Peter Wepler, Steve Papa, Mike Bilecki, Jordan Raphael, Nick Gibbons, Diana Lynch, Annie McIntyre, Timothy Byrne, George Kiesel, Kevin Boone, Robert Moses State Park staff, National Park Service staff, and Suffolk County plover stewards

With 2022 field support from:

Kat Black, Samuel Denenberg, Don Fraser, Claire Helmke, Kristen Johnson, Jessica Klassert, Daniel Reisch, Sam Robinson, Alison Schroeder, Samantha Smith, Eric Swan, Jennifer Velasquez, Katie Walker, and Winnie Wei

With past field and analytical support from:

M. Aroff, K. Attinger, S. Ball-Douglas, R. Bechtold, B. Bedwell, H. Bellman, E. Berge, S. Bilodeau, K. Black, N. Bosco, E. Brown, M. Carroll, D. Crawford, J. Deans, L. Deets, A. DeRose-Wilson, J. Domino, S. Flanagan, D. Fraser, E. Gardner, J. Gardner, N. Giordano, J. Girgente, L. Granger, J. Green, J. Hall, B. Hardt, C. Helmke, L. Hermanns, C. Howell, T. Humbert, J. Ialeggio, K. Jackson, C. Kupferman, M. Kynoch, J. LeClaire, B. Lin, A. Miller, R. Moylan, T. Ostrander, C. Park, E. Peyton, C. Robertson, S. Robinson, K. Rosamond, T. Russell, V. Schabert, K. Sloan, P. Sonnega, M. Spears, F. Stetler, S. Sullivan, E. Swiecki, H. Taylor, K. Todd, K. Walker, J. Whaley, A. Wiese, R. Winters, M. Zak, E. Zawatski, and A. Zeller

This work took place on the ancestral lands of the Massapeguas, Secatogue, Unkechaug, and Shinnecock Nations



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Cover photo: GF(2J7) broods their chicks at the Fire Island lighthouse tract. Photo provided by Hope L. VanDerwater



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Executive summary

This report summarizes Virginia Tech's Effectiveness Monitoring of certain conservation measures in 2013 – 2022, required under the Biological Opinion for the Fire Island Inlet to Moriches Inlet Stabilization Project. When appropriate and possible, we provide 2022-specific results in addition to cumulative results to facilitate the inspection of trends across the entire study period. From April – August of 2013 – 2022 (but excluding 2020), we conducted field monitoring of federally-threatened piping plovers (*Charadrius melodus*). Red fox (*Vulpes vulpes*) monitoring began in June 2015 and continued year-round until March 2019. Since 2021, we conducted field surveys for mammalian predators (including red foxes) April – August.

Overall piping plover monitoring findings

- Since 2013, the piping plover population has grown 346.2%. Approximately half of the plover pairs monitored (51.7%) nested in backshore and barrier interior habitats, and an approximate third of the pairs monitored (31.8%) nested in overwash habitats. The most pronounced increases in nesting density are at the NPS East and West Wilderness areas. Human recreation and vegetation growth are believed to explain the comparatively slower increases or stabilization in nesting density in other areas.
- Since the outbreak of sarcoptic mange in red fox, the near-ubiquitous use of nest exclosures has resulted in consistently high piping plover nest survival. The greatest causes of nest failure are abandonment (12.5%) and predation (10.8%). In contrast, chick survival to fledging has declined since 2019 across much of the study area, concurrent with an increase in the relative abundance of feral cats (*Felis catus*), vegetative succession, and increased plover density.
- From 2013 – 2022, the annual breeding pairs were comprised of an average of 16.0% (SE: $\pm 4.2\%$) natal recruits and 35.4% (SE: $\pm 4.8\%$) immigrants. Most (65%) returning adult residents monitored in this study have nested in habitats created by Hurricane Sandy or the subsequent habitat alterations on Fire Island and at Cupsogue Beach County Park. Natal recruits and immigrants nested in Hurricane Sandy-created habitats in slightly lower frequencies (57% and 44%, respectively). Adult (AHY) survival has remained high throughout the study (0.70 ± 0.01), whereas hatch-year survival has fluctuated (range: 0.17 – 0.55).
- Low-energy moist habitats, such as bay intertidal habitat and ephemeral pools have greater invertebrate abundance relative to interior and ocean intertidal habitats. The recent restoration at Democrat Point is believed to have increased accessibility of a large tidal pool to piping plovers.

Overall predator monitoring findings

- Since 2019, red foxes have at least intermittent presence across the study area, but have yet to recover from the outbreak of sarcoptic mange. There is no evidence of foxes breeding in the study area since the mange outbreak, though anecdotal evidence from 2022 suggests foxes may be breeding in the interior of Fire Island near residential villages.
- There is no recorded evidence of red foxes taking piping plovers, their young, or nests since 2019. Fox tracks were observed outside two nest exclosures in 2022, though was apparently not sufficient to cause nest abandonment nor appeared to cause adult or chick mortality. We have not found any clearly identifiable piping plover remains in fox scat throughout this study.
- Since 2019, there has been a 50% increase in cat detections during mammalian predator transect surveys. In 2019, plover bands were recovered from cat scat. In 2022, four piping plover nests, two plover broods, a plover chick, and an adult plover were lost to feral cats. Feral cats are likely now the primary mammalian predator of piping plover on Fire Island, and their increasing relative abundance is a management concern.



Management recommendations

- **Continue use of nest exclosures.** Nest exclosures should continue to be used unless predation dynamics change dramatically, and then altered only after careful consideration, as they are a strong predictor of plover nest success. There is no evidence that exclosures resulted in increased adult mortality across the study period.
- **Management of vegetation to increase available nesting habitat.** Vegetation management should be used to maintain early-successional habitat for nesting piping plovers. Vegetation management will also reduce cover for mammalian predators.
- **Consider initiation of a feral cat control program to reduce plover take.** Feral cats are an invasive species, and effective predators. There is a marked increase in the feral cat abundance, and their increase coincides with a decline in piping plover chick survival, and observed instances of taking adult plovers. While taking plovers is documented, it is not clear whether feral cats are a limiting factor. Feeding and providing shelter to feral cats should be prohibited within all management areas to reduce the population growth of feral cats.



Background



Introduction

The work described in this report was funded under the Breach Contingency Plan (BCP; USACE 1996), the Fire Island Inlet to Moriches Inlet Stabilization Coastal Storm Risk Reduction Project (FIMI; USACE 2014), the U.S. Fish and Wildlife Service (USFWS) North Atlantic Landscape Conservation Cooperative Beach Resiliency Project (Resiliency Project), and by Virginia Polytechnic Institute and State University (Virginia Tech). Both the FIMI project and the Resiliency Project were funded under the Disaster Relief Appropriations Act of 2013 (Public Law 113-2). Because these projects were so closely coordinated, we produced cumulative reports in 2019 (Robertson et al. 2019), 2020 (Walker et al. 2020), and 2021 (Wails et al. 2021) to provide the best overview of conditions of the piping plover (*Charadrius melodus*) population, the habitat that supports it, and the red fox predator population and its interactions with piping plovers. The U.S. Army Corps of Engineers' (USACE) contributions to this project until June of 2015 were made under the BCP and were restricted to plover studies in the Corps Project Areas. The Resiliency Project funding was intended to expand the plover project to include areas of Smith Point County Park in 2013 – 2016.

As continuation of the FIMI funding, our monitoring efforts resumed in 2021 after a pause in 2020 due to the novel coronavirus (SARS-CoV-2) pandemic. Since 2020, Virginia Tech researchers were able to conduct fieldwork compliant with university- and state-level health and safety considerations.

In October 2012, Hurricane Sandy made landfall on the barrier islands of New York and New Jersey and created multiple breaches and overwashes. Three breaches were created within the study area in Fire Island and Westhampton Island (from east to west): one within Cupsogue Beach County Park on Westhampton Island, one in Smith Point County Park on Fire Island, and one in the Otis Pike Wilderness of Fire Island National Seashore, one in Smith Point County Park on Fire Island.

The breaches in the County Parks were filled soon after the hurricane, in accordance with the BCP. The FIMI project entailed sections of beach and dune construction along the 30.5 km of Fire Island and was initiated by USACE in the winter of 2014. The USACE formally consulted with the USFWS pursuant to the Endangered Species Act of 1973, as amended, due to the potential direct and indirect adverse impacts of these projects on the federally listed species. The USFWS's Biological Opinion (BO) for the BCP covered the threatened piping plover and threatened seabeach amaranth (*Amaranthus pumilus*); the BO for the FIMI included these species and the threatened red knot (*Calidris canutus rufa*; USFWS 1995, 2014). Anthropogenic alterations to habitat, including dredging, dune construction, and intensive human beach use, can reduce plover habitat availability (Cohen et al. 2009a, Catlin et al. 2015).

The potential adverse effects of the FIMI on piping plovers listed in the BO included habitat loss and degradation, recreational disturbance, and exacerbation of predation. As a result, the FIMI BO included a number of conservation measures to address these potential effects and reduce the effect of potential incidental take authorized by the USFWS, including inlet-to-inlet monitoring, habitat maintenance and restoration, predator management, and reduction of human disturbance. Vegetation was cleared and the topography was modified by the USACE in approximately 33.7 ha (83.2 ac) of ocean-side habitat at the Great Gun restoration area to mitigate adverse effects of dune and beach construction. To create possible nesting and foraging habitat for piping plovers, the USACE also created the New Made restoration area, 6 ha (15.8 ac) of bayside habitat south of Carter's Island.

Breaches and overwashes can create new foraging and nesting habitat for piping plover (Cohen et al. 2009a, Robinson et al. 2019), though the manipulation of these same areas during dune construction under the FIMI had the potential to disrupt natural plover breeding processes. Thus, the effect of the habitat modifications under the FIMI were monitored and evaluated. Understanding quality of nesting and foraging habitats can lead



to a better understanding of the plover response to Hurricane Sandy and subsequent habitat alterations. The response to habitat change can manifest in myriad changes in demographic and growth rates, density, movement, and prey abundance, among others (Loefering and Fraser 1995, Elias et al. 2000, Cohen et al. 2009a, Catlin et al. 2014, Catlin et al. 2015, Hunt et al. 2017, Hunt et al. 2018, Walker et al. 2019, Robinson et al. 2020a).

Additionally, plover populations may be affected by predators (Ivan and Murphy 2005, Hunt et al. 2019). Understanding how the local predator community responds to storm and anthropogenic habitat alterations, and how the predators are affecting piping plovers during the same period, is necessary for a holistic understanding of habitat change. Red foxes (*Vulpes vulpes*) are a predator of piping plovers in barrier island ecosystems (Rimmer and Deblinger 1990, Patterson et al. 1991, Cohen and Fraser 2010, Gieder 2015, Stantial et al. 2020, Black 2021). Aspects of red fox population ecology have been studied in a variety of ecosystems (Lloyd 1980, Ables 1983). Yet changes in red fox site occupancy (i.e., presence, accounting for detection) have been studied only twice on the U.S. Atlantic Coast prior to our long-term study: once on Cape Cod, MA (O'Connell et al. 2006) and once on Assateague Island, MD (Gieder 2015). Prior to our work (e.g., Black 2021), no previous monitoring has specifically examined red fox population responses to storm and anthropogenic habitat alterations, nor detailed fox population ecology in a barrier island ecosystem. Land managers frequently undertake both non-lethal (e.g., exclosures) and lethal (e.g., targeted or systematic removal) management of red foxes along the U.S. Atlantic Coast to benefit piping plovers and other imperiled shorebirds and sea turtles (Hunt et al. 2019). Rarely are the efficiency and efficacy of predation management efforts assessed (but see Stantial et al. 2021a). There is a growing recognition by managers and the public that monitoring and quantitative assessments of the success of predation management actions are needed (Hunt et al. 2019). Thus, the red fox monitoring in the report quantifies occupancy, home range dynamics, numbers, vital rates, habitat selection, movement ecology, and diet of red foxes in plover habitats to best guide predator management efforts.

This report seeks to summarize targeted monitoring programs created to evaluate the efficacy of the BO conservation measures, developed by the USFWS. The monitoring effort seeks to provide a scientific evaluation of the effectiveness of the management and habitat maintenance and restoration efforts within the FIMI project area and Fire Island, as a whole. Both Virginia Tech and the USFWS worked closely with an interagency team consisting of the National Park Service (NPS), New York State Office of Parks, Recreation, and Historic Preservation, Suffolk County Parks, and the USACE to obtain the necessary permits to undertake this work.

The goal of this project is to provide a broad ecological understanding of how overwashes, breaches, breach fills, and the USACE projects affect piping plover populations, their red fox predators, and their invertebrate prey communities. We compared the dynamics of bird use and invertebrate densities in an open breach area, two filled breach areas, two restoration areas, nine overwash areas, and other pre-Hurricane Sandy habitats within the study area. Ultimately, these results will help to refine the collective understanding of the timing of and manner in which piping plover habitat develops and persists, and how piping plovers respond to habitat creation in the context of predation by red fox.

Study Objectives

The report of piping plover monitoring includes the following objectives:

- **Objective P1:** Compare changes in numbers, density, and distribution of piping plovers in various habitats on Fire Island.
- **Objective P2:** Compare piping plover nest and chick survival rates in various habitats across Fire Island.
- **Objective P3:** Compare the relative contribution of local growth (recruitment) vs. immigration in population growth associated with new habitats. Determine habitat correlates of population change.
- **Objective P4:** Compare prey densities at several habitat types.
- **Objective P5:** Quantify the interactions of predators with shorebird nests and chicks.
- **Objective P6:** Investigate the potential for applying population models developed by Fire Island to other known and potential plover habitats in Army Corps' Project Areas.

The report of red fox monitoring includes the following objectives:

- **Objective F1:** Produce an index of fox abundance to understand the recovery rate of the fox population.
- **Objective F2:** Estimate red fox reproductive success and relate it to population growth during the recovery from the mange outbreak.
- **Objective F3:** Quantify the interactions of red fox with shorebird nests and chicks during the recovery from the mange outbreak. Monitor the impacts of foxes vs. other predators on plover nests and chicks.
- **Objective F4:** Use information collected in Objectives F1 – F3 to make recommendations on utility of, and recommended methodologies for, management of red fox populations in the study area and in other coastal habitats.

Study Area

The complete study area consisted of 36.6 km of Fire Island and Westhampton Island (27.1 km of beachfront monitored for plovers, with fox surveying through an additional 9.5 km stretch of Fire Island; Figure 1). The study area falls within the administrative boundaries (referred to as management units in this report) of the National Park Service (Otis Pike Wilderness Area and the Fire Island Lighthouse Tract), Suffolk County Parks (Cupsogue Beach County Park and Smith Point County Park), and the New York State Office of Parks, Recreation, and Historic Preservation (Robert Moses State Park). Some research activities were limited to subsites within the complete study area, changed between years, or were used to meet a particular study objective. From 2013 to June 2015, the study area only consisted of Otis Pike Wilderness Area, Smith Point County Park, and Cupsogue Beach County Park (17.3 km of beachfront). In June 2015, when the FIMI funding became available, the study area was expanded to include Robert Moses State Park and the NPS Fire Island Lighthouse Tract (9.8 km; Figure 1). Although some research activities took place in Watch Hill, the communities between Kismet to Watch Hill were not monitored for plovers after 2014. Since 2020, the study area spanned from Robert Moses State Park to Smith Point County Park (total length: 34.3 km, length monitored for plovers: 24.8 km), no longer including Cupsogue Beach County Park (2.3 km).

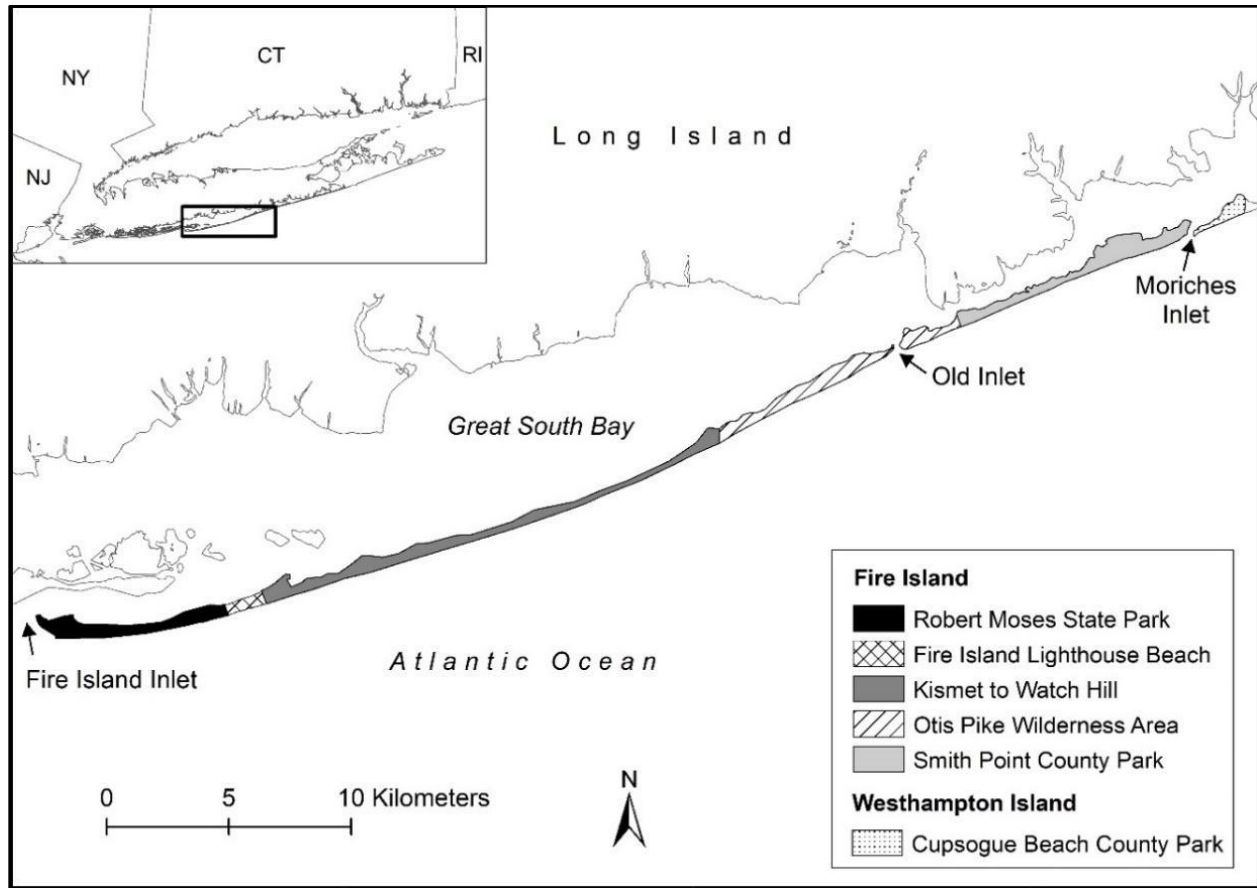


Figure 1. Map of the study area at Fire Island and Westhampton Island, NY with the corresponding management units.

Prior to Hurricane Sandy, the study area consisted of sandy backshores largely backed by dunes. In October 2012, Hurricane Sandy overwashed several areas of Fire Island and Westhampton Island. The hurricane created breaches at Cupsogue Beach County Park, in the Narrow Bay West area of Smith Point County Park, and opened Old Inlet in the Otis Pike Wilderness Area. The Cupsogue breach was filled with a berm ≥ 9 ft National Geodetic Vertical Datum (NGVD) in elevation, 150 ft wide and 2,000 ft long (Bocamazo 2013). The Narrow Bay West breach was filled with a berm about 12 ft NGVD in elevation (USACE 2014). Old Inlet remained open and subject to natural erosion and accretion processes. Prior to the 2015 piping plover breeding season, the USACE constructed two restoration areas to provide potential plover nesting and foraging habitat. This construction consisted of 33.7 ha of flat sand oceanside habitat at the Great Gun restoration area, and 6 ha of flat sand bayside habitat at the New Made restoration area (hereafter, 'New Made').

Habitats were defined based on USACE polygons of restoration and management areas, post-Sandy delineation of overwash, and Light Detection and Radar (LiDAR) elevation data (Table 1). When possible, we describe landcover characteristics using terminology in Zeigler et al. (2017).

Table 1. Habitat categories used within this report refer to geographic areas on Fire Island and Cupsogue Beach County Park.

Habitat	Definition
Backshore	The dry sand area between the high-water line and the toe of the dune.
Barrier interior	The area inland of the toe of the dune and interior of Fire Island.
Beach	The area that is between the high- and low-water line. Can be further defined as ocean, bay, or inlet depending on the geologic location. Synonymous with the intertidal zone.
Breach fill	The areas of Fire Island and Cupsogue Beach County Park that breached during Hurricane Sandy, but were filled shortly after by the USACE. Both breach fills fall adjacent to manipulated overwashes.
Dredge spoil	Stockpiles and graded areas of dredged sand.
Flood shoal	A submerged ridge or bar associated with inlets consisting of or covered by sand; rising from the seabed to near the surface so that it is tidally exposed. Rarely static and may move seasonally and annually.
Management area	Areas defined by the USACE to manage under the FIMI.
Manipulated dune	Any dune or berm created by other than natural sources. This includes snow fencing, Christmas trees, planting, and USACE/other engineering.
Manipulated overwash	Areas of Fire Island that overwashed during Hurricane Sandy, but were manipulated through dune manipulation (including snow fencing and Christmas tree placement), or vegetation management.
Mud flat	Mostly intertidal land forms where particles are generally < 0.05 mm in diameter and often mixed with bits of organic material. Areas where fine-grained sediments have been deposited in the low-energy shallows behind the island.
Natural overwash	Areas of Fire Island that overwashed during Hurricane Sandy, but have not been further modified.
Nourishment	Beach widening by the USACE in Smith Point County Park from 2014 – 2015 and 2018 – 2019.
Overwash	Any area that overwashed during Hurricane Sandy. This is a broad category that is subdivided into manipulated or natural overwash.
Restoration area	New Made and Great Gun restoration areas, created between the 2014 and 2015 piping plover breeding seasons.
Sandbar	Mostly intertidal landforms that are generally composed of particles > 0.05 mm. In this study area, sandbars occur north of the inlets.
Swale	A depression or blow-out behind the primary dunes, between ridges; either vegetated or unvegetated.
Other	A broad category used when the area does not fall into one of the overarching categories above. All following categories could further be defined under the ‘other’ category, depending on the specificity of the question.

We divided the study area into smaller subsites intended to be relatively homogeneous in geomorphological and ecological structure (e.g., areas where overwash occurred were distinct from highly vegetated tall dunes; Figure 2 – Figure 4, Table 2). These subsites allowed for general location referencing when GPS accuracy was not required. Monitoring surveys were performed by subsite.



Table 2. Subsite names, location descriptions, and habitat on Fire Island and Westhampton Island, 2013 – 2022. Subsites are generally listed from west to east. Table continues to next page.

Subsite name	Subsite code	Description	Habitat type
Democrat Point	DMP	Begins at federal navigation channel stone jetty on bay and ocean side at the west ends of OSS and STP, and extends west to the inlet.	Beach, backshore, and interior
Ocean South Side	OSS	Begins at blue hut at the west end of Field 3 and extends west to the western stone jetty on the ocean side. Includes Field 2 and the ORV area.	Beach, backshore, interior, and swale
Stockpile	STP	Sand stock-pile area on bay side of island; begins at west end of golf course and extends west to the northernmost point of the rock jetty.	Dredge spoil, beach, interior
The Golf Course	GOLF	Begins at west side of the Field 2 parking lot behind the primary dune line and extends west to the beginning of the sand stock-pile area and north to the major roadway. Not plover habitat.	Golf course
Field 3	FLD3	Begins at blue hut to the west end of Field 4 and extends west to the next blue hut in front of the Field 2 parking lot.	Beach, backshore, and interior
Field 4	FLD4	Begins at blue hut to the west end of Field 5 and extends west to the next blue hut in front of the Field 3 parking lot.	Beach, backshore, and interior
Field 5	FLD5	Begins at signage at the west end of Fire Island Lighthouse and extends west to the blue hut at the boundary of Field 4.	Beach, backshore, and interior
Fire Island Lighthouse	FIL	Fire Island Lighthouse Tract of NPS; begins at eastern edge of sand ORV track running inland before Kismet community housing begins and extends west to signage indicating the boundary with Robert Moses State Park.	Beach, backshore, interior, swale, and manipulated dune
Watch Hill	WHL	Begins at Watch Hill and extends east to Bellport Beach.	Beach, backshore, interior, and natural overwash
Old Inlet West Reference	OIWR	Stretch of beach between the east end of Old Inlet West and Bellport Beach.	Beach, backshore, interior, and natural overwash
Old Inlet West	OIW	Begins at west side of Old Inlet and extends to the point where large primary dunes begin.	Beach, backshore, interior, and natural overwash
Flood Shoals	FS	Flood shoals associated with Old Inlet.	Flood shoals
Old Inlet East	OIE	Begins where the primary dune line ends and extends west to Old Inlet.	Beach, natural overwash, and interior
Bridge West	BRW	Stretch of backshore with primary dune line between Bridge Overwash and Old Inlet East.	Beach, backshore, and interior
Bridge Overwash	BRO	Overwash in the Otis Pike Wilderness, just west of the NPS Visitor Center.	Beach, natural overwash, and interior

Subsite name	Subsite code	Description	Habitat type
Bridge East	BRE	Begins at the primary dune line at the west end of Pattersquash Overwash and extends to the Otis Pike Wilderness at NPS Visitor Center.	Beach, backshore, and interior
Pattersquash Overwash	PSO	Overwash west of Pattersquash East.	Beach, manipulated overwash, interior, and manipulated dunes
Pattersquash East	PSE	Stretch of backshore with primary dune line between Narrow Bay West and Pattersquash Overwash.	Beach, backshore, and interior
Narrow Bay West	NBW	Includes the overwash and small breach fill west of the hazardous area.	Beach, manipulated overwash, interior, manipulated dunes, and breach fill
New Made	NMA	Begins at the east end of the filled dredge spoil and extends west to the east edge of the small breach fill.	Beach, backshore, manipulated dune, interior, and restoration area
Narrow Bay East	NBE	Begins at the westernmost end of the large dune and extends to the westernmost end of the Hazardous Area.	Beach, manipulated overwash, interior, and manipulated dunes
Smith Point West	SPTW	Begins at Cut 3 on Burma Road and extends to the end of the large dune and beginning of the Narrow Bay East manipulated overwash area.	Beach, backshore, and interior
Smith Point East	SPTE	Begins on west side of Moriches Inlet and extends west to Cut 3 on Burma Road.	Beach, backshore, manipulated dune, interior, and restoration area
Cupsogue West	CSW	Breach fill and Moriches Inlet area of the Cupsogue Beach County Park. Begins at the west edge of the breach fill and extends west to Moriches Inlet.	Beach, manipulated overwash, manipulated dunes, interior, and breach fill
Cupsogue Sandbar	CSB	Large sandbar/shoal north of Cupsogue West.	Sand bar
Cupsogue East	CSE	Beach and narrow strip of bayside intertidal zone at east end of the Cupsogue Beach County Park. Begins at the west end of the parking lot and extends west to the east edge of the breach fill.	Beach, backshore, manipulated dune, and interior
Cupsogue Mud Flats	CMF	Mud flats north of the Cupsogue parking lot, and south of the high bare Island in Moriches Bay.	Mud flat



Figure 2. Study area at Robert Moses State Park (top and bottom) and Fire Island Lighthouse tract (bottom). Subsite boundaries are outlined in orange. Refer to Table 2 for subsite names and descriptions. Imagery captured April 2022.

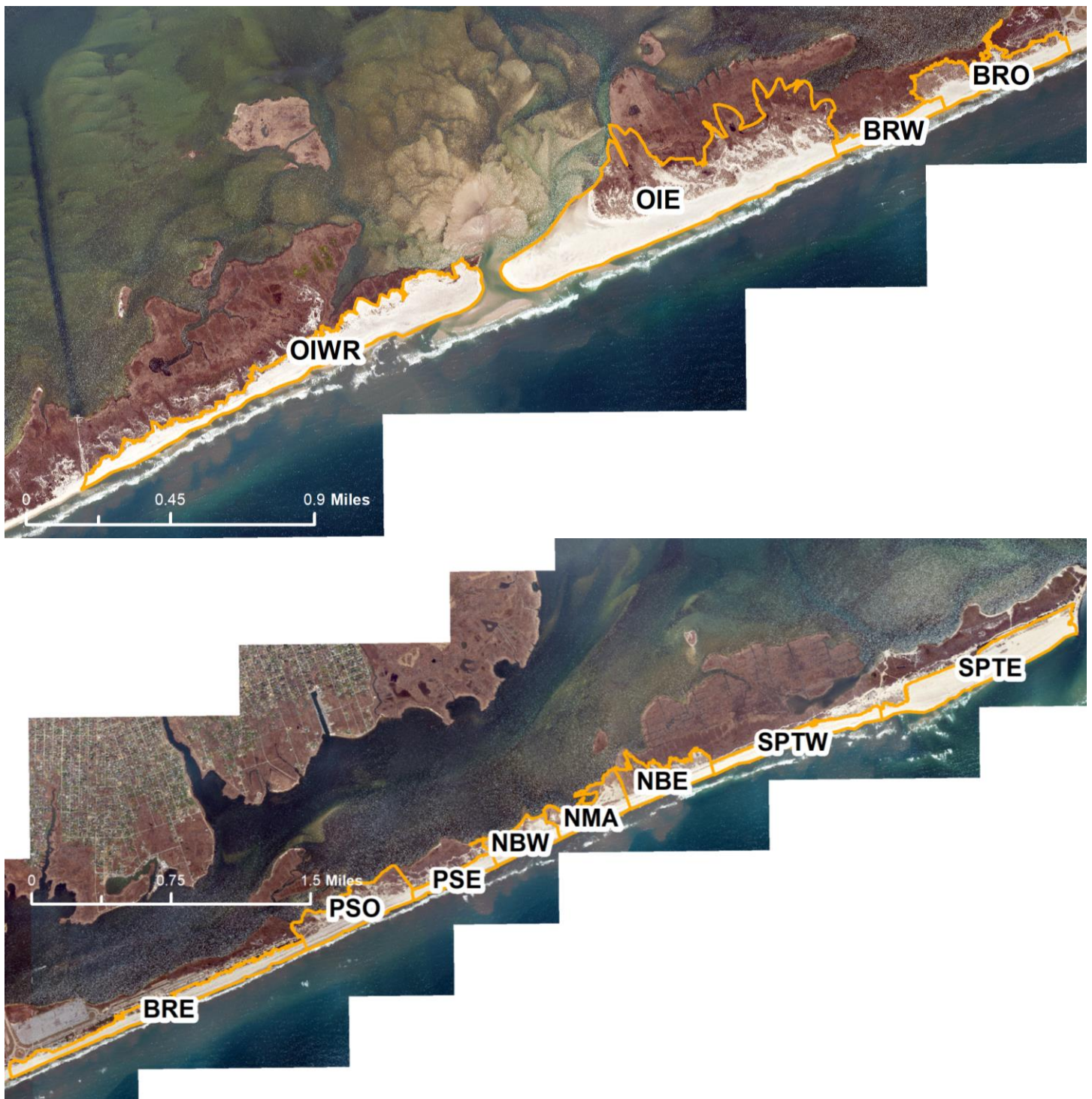


Figure 3. Study area at Otis Pike Wilderness Area Park (top) and Smith Point County Park (bottom). Subsite boundaries are outlined in orange. Refer to Table 2 for subsite names and descriptions. Imagery captured April 2022.

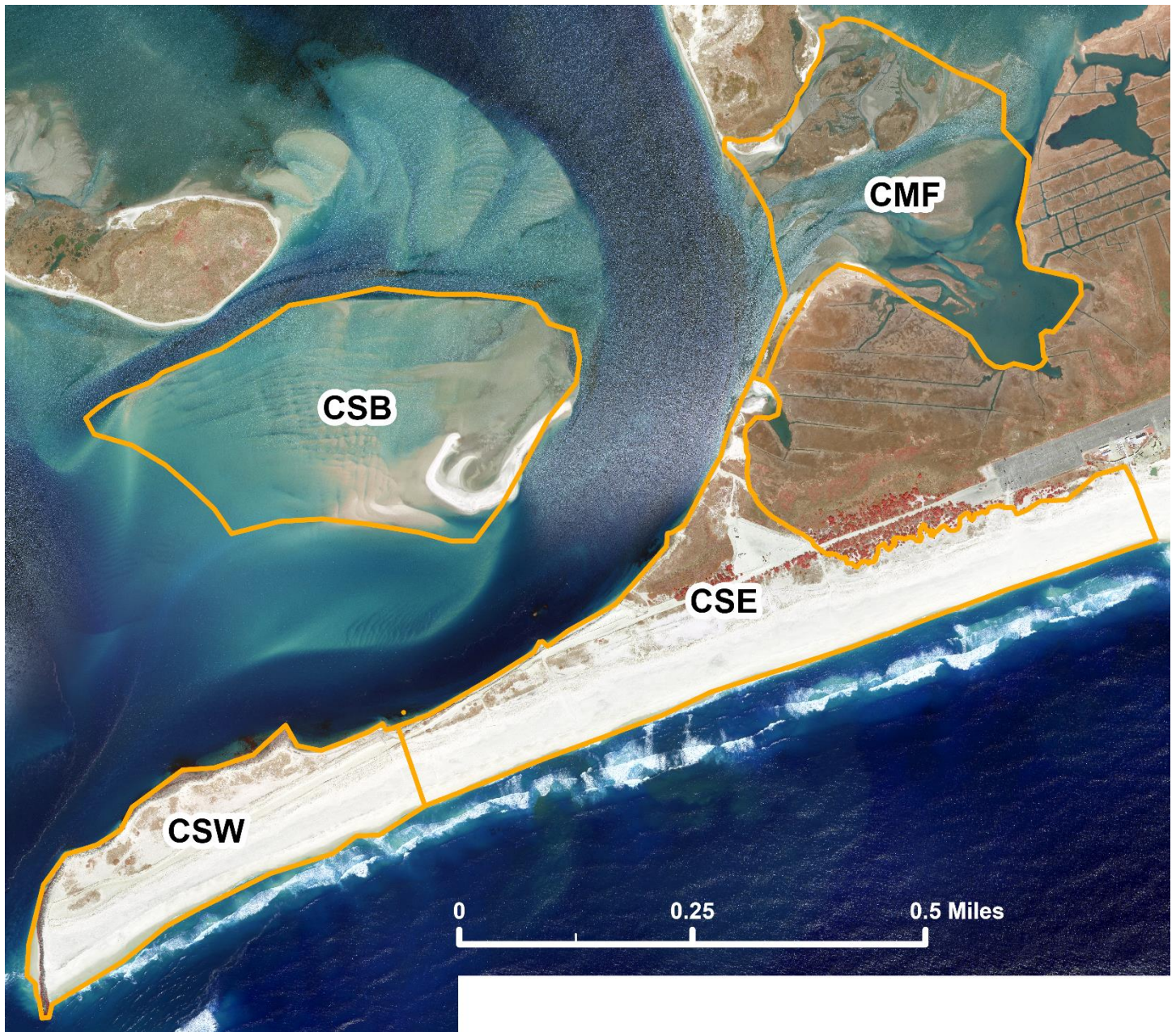


Figure 4. Formerly monitored areas at Cupsogue Beach County Park. Subsite boundaries are outlined in orange. Refer to Table 2 for subsite names and descriptions. Imagery captured April 2022.

Throughout the plover breeding seasons, all beaches were open to pedestrians. Smith Point County Park and Robert Moses State Park also allowed recreation vehicles to drive onto the beaches. Every year, these beaches were closed to vehicles once chicks hatched and were re-opened once chicks fledged (Table 3). However, Smith Point County Park in 2013, and the ORV area (OSS subsite) within Robert Moses State Park - South in 2017 – 2019, remained open to vehicle access following piping plover hatch dates. Official dates of closures for 2013 are uncertain. Despite vehicle closure to the general public, law enforcement, park staff, and plover monitors were still permitted to drive through closed areas, depending on the circumstance.



Table 3. Beach closure dates for vehicle use at Smith Point County Park and Robert Moses State Park 2013 – 2022.

Year	Management Area		
	Smith Point County Park	Robert Moses State Park – North ¹	Robert Moses State Park – South ²
2013	Closed, dates unknown	-	-
2014	09 Jun – 01 Aug	-	-
2015	09 Jul – 13 Aug	06 Jun – 04 Aug	10 Jul – 04 Aug
2016	22 May – 25 Aug	06 Jul – 22 Jul	25 Jun – 05 Aug
2017	27 May – 05 Aug	07 Jun – 16 Jul	24 May – 10 Aug
2018	11 Jun – 21 Aug	02 Jun – 03 Aug	29 Jun – 03 Aug
2019	23 May – 09 Aug	06 Jun – 13 Jul	04 Apr – end of season
2020	21 May – 27 Aug	29 May – 22 Jul	Apr – end of season
2021	22 May – 04 Aug	Closed, dates unknown	23 Jun – 28 Jul
2022	23 May – 09 Aug	20 Jun – 19 Jul	20 Jun – 19 Jul

¹Defined as the sandy roads north of the ORV area and golf course

²Defined as the ORV area west of Field 2

Piping plover monitoring on Fire Island



Piping plover general field methods

Due to the novel coronavirus (SARS-CoV-2) pandemic in 2020, we had to make modifications to planned fieldwork. In 2020, Virginia Tech researchers were unable to conduct fieldwork and instead relied on data from land managers. In 2021, we resumed fieldwork in a limited capacity. We made an effort to monitor the entire study area, prioritizing several subsites (DMP, OIE, BRW, BRO, PSO, PSE, SPTE; Table 2). These subsites were selected as they overlap with an array of habitat types that are related to our study objectives. As plover nesting activities began in 2021, we monitored these focal subsites sites twice a week (typically every 1 – 3 days) and focused much of our banding efforts in these areas. In non-focal subsites, we opportunistically searched for nests, banded adults and chicks, and surveyed at all the other areas. Towards the end of the 2021 breeding season in July, we were able to fully cover the entire study area and band many of the broods outside of the focal subsites. In 2022, we fully resumed fieldwork. Here, we explicitly detail our field methods used to monitor piping plovers and describe modifications due to the pandemic.

Habitat and tide mapping

Methods prior to 2020: In 2013 and 2014, we collected all mapping data with a Garmin 78 (Garmin International, Olathe, KS). In 2015 – 2019, we used a Trimble Geo7x (Trimble Navigation, Ltd., Sunnyvale, CA) to map the shoreline during spring high and low tides along the bay and ocean beach (Figure 5). Tides were mapped at the shoreline throughout the entire season, although no low tide data was collected in 2014. In 2015, we mapped spring high tide and low tides during mid-May and early June, and mapped various habitats and subsite boundaries during late July and early August. In 2016, we mapped shorelines at spring high tide and low tides May – September, and completed snow and symbolic fencing mapping during late-July and August. In 2017, we mapped the shorelines during spring high and low tides in May, and the snow and symbolic fencing during late June – August. In 2018, we mapped shorelines at spring high and low tides in May and June, and mapped snow and symbolic fencing May – August. In 2019, we mapped shorelines at spring high and low tides in May and June, and mapped snow and symbolic fencing April – July.

Modified methods for 2020: No tide lines were mapped in 2020.

Methods post 2020: We opted to use LiDAR data and a digital elevation model (DEM) to identify the spring high tide line. LiDAR data and spatial imagery was collected each spring by Axis Geospatial (Easton, MD) and a raster of water and intertidal sand classified from DEM data in ArcGIS 10.6 (West Redlands, CA). We used an oceanfront intertidal range of 0.00 – 4.11 ft elevation based on the astronomical high water at NOAA Moriches Inlet tidal station.



Photo: K. M. Walker

Figure 5. Using a Trimble Geo7x to map the spring tides by walking along the high-water line prior to 2020.



Piping plover adult survey

Methods prior to 2020: We surveyed subsites in the study area (Table 2) for adult piping plovers every 1 – 3 days (weather permitting), April – August. We surveyed subsites by UTV and on foot. Some subsites that did not have bayside habitat and consisted only of narrow ocean beach were surveyed by driving a UTV slowly along the ocean beach and along the backshore. UTVs were only driven outside of areas protected by symbolic fencing established by land managers. At least twice per week during the nesting period (April – July), we searched subsites on foot for plover tracks, scrapes, and nests. When conducting foot surveys, observers walked parallel tracks 50 – 100 m apart (dependent upon the size of the subsite), moving longitudinally across the subsite. Transect width varied depending on topography, vegetation, weather conditions, and other factors affecting visibility. If the entire width of the subsite could not be surveyed in one pass, observers conducted multiple passes of the subsite walking parallel transects. Monitors carried spotting scopes (20 – 60x, 80 mm) to resight plovers (Figure 6). Upon sighting a plover, we recorded its band combination (if banded), behavior, and habitat. In 2013 – 2015, we recorded the location of individuals with a Garmin GPS unit. Since 2015, we used a Nikon laser rangefinder and a Trimble GPS unit to record the location of plovers (Robinson et al. 2020b).

Modified methods for 2020: In 2020, land managers from the National Park Service, New York State Parks, and Suffolk County Parks surveyed nesting areas and resighted banded adults as often as possible, though resight effort was not consistent with that of 2013 – 2019.

Modified methods for 2021: We resumed surveying subsites in the study area every 1 – 5 days (weather permitting) April – August, adhering to our previously used methods. Adult locations were not recorded.

Methods post 2021: We surveyed subsites every 1 – 3 days (weather permitting) April – August, adhering to our previously used methods and recorded adult locations.



Photo: C. N. Wails

Figure 6. Surveying for piping plovers using spotting scopes at Robert Moses State Park.

Piping plover nest locations

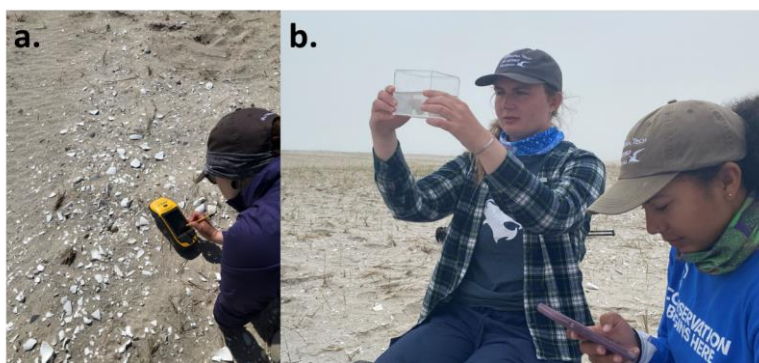
Method prior to 2020: We searched all potential nesting areas within subsites, including backshore, overwashes, and sparsely vegetated habitats every 1 – 3 days (weather permitting). Nest locations were recorded using a Garmin GPS unit (2013 – 2014) or a Trimble GPS unit (after 2014; Figure 7a). If a nest was found at full clutch (i.e., all eggs had been laid), we used egg flotation to estimate the date of clutch initiation and



hatch date (Alberico 1995, Liebezeit et al. 2007). We floated eggs individually in warm water (Figure 7b). Local land managers installed exclosures around nests and typically not at nests with fewer than four eggs.

Modified methods for 2020: Local land managers conducted nest searches and collected nest locations through NestStory (National Park Service), handheld GPS (Suffolk County Parks), and by digitizing them into Google Earth based on geography and land markers (New York State Parks).

Methods post 2020: We resumed surveying subsites (2021: every 1 – 8 days, weather permitting; post 2021: every 1 – 3 days) April – August, adhering to our previously used methods.



Photos: S. S. Dorsey (left), C. N. Wails (right)

Figure 7. (a.) Using a Trimble Geo7x to map the location of a piping plover nest, and **(b.)** floating an egg from a four-egg nest in SPTE and consulting with digital copies of an egg floatation chart.

Piping plover nest checks

Methods prior to 2020: We checked nests every 1 – 3 days throughout the incubation period (weather permitting; Figure 8). As the hatching date approached, we attempted to check nests every day. Nest fate was determined from observations of the nest and surrounding area.

Modified methods for 2020: In 2020, land managers checked nests as frequently as possible, typically every 1 – 3 days in ideal conditions.

Methods post 2020: We resumed nest checks, adhering to our previously used methods.



Photos: K. M. Walker

Figure 8. Checking for hatch at **(a.)** an exclosed nest using binoculars, and **(b.)** an unexclosed nest.

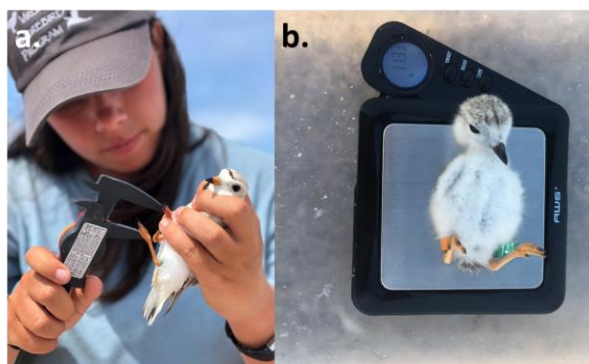


Piping plover trapping and banding

Methods prior to 2020: We attempted to trap both banded and unbanded adults during the incubation period, over a full clutch, using a drop-box trap or shoo-in funnel trap. Once captured, plovers were removed from the trap, banded, weighed, and measured (Figure 9a). If the second adult of a particular nest had not yet been trapped that year, we reset the trap while processing the first adult. We attempted to capture banded plovers to assess the condition of bands, flags, and leg health. If no adults entered a trap within approximately 20 minutes of trap deployment, we removed the trap and did not return to the nest that day, nor attempt to trap until at least 48 hours had passed. We captured, banded, and processed plover chicks on the initial nest visit post-hatching (Figure 9b) and attempted to recapture them at a minimum of 20 days post-hatch to weigh and measure them a second time.

Modified methods for 2020: Piping plovers were not captured nor banded in 2020.

Methods post 2020: We resumed trapping and banding, adhering to our previously used methods.



Photos: K. M. Walker

Figure 9. (a.) Measuring the tarsus of an adult piping plover, and **(b.)** recording the mass of a 10-day old chick.

Piping plover banding schemes

Methods prior to 2020: In 2013, most plovers were banded with a green flag with a field-readable alphanumeric code applied to the upper left leg and a numbered orange Darvic plastic band applied to the upper right leg (Figure 10a); a smaller portion of plovers received two color bands on each upper leg or green flags with alphanumeric combinations with a yellow band on the opposite leg (DeRose-Wilson et al. 2013, 2014; Figure 10b). During 2014 – 2019, we banded all adults and chicks with dark green flags and an orange Darvic band on the opposite upper leg (Figure 10a).

Modified methods for 2020: Piping plovers were not captured nor banded in 2020.

Methods for 2021: We resumed trapping and banding, adhering to our previously used methods.

Methods post 2021: Adult plovers were banded with a green flag with a field-readable alphanumeric code applied to the upper left leg and a numbered orange Darvic plastic band applied to the upper right leg (Figure 10a). Following permit changes, chicks received a unique color combination using two color bands placed on each upper leg (Figure 10b).



Photos: C. N. Wails

Figure 10. Example of piping plover banding schemes, with **(a.)** an adult plover with a green flag on the upper left leg and a numbered orange Darvic band on the upper right leg, and **(b.)** a plover chick with four total Darvic bands. Note that in 2022, banders wore gloves and masks to reduce risk of the highly pathogenic avian influenza (Caliendo et al. 2022).

Piping plover brood observations

Methods prior to 2020: During 2013 – 2019, we searched for plover chicks for 30 days post-hatching, regardless of brood fate, every 1 – 3 days (weather permitting). After locating the brood and identifying chicks, we used spotting scopes and attempted to collect behavioral observations of at least one randomly selected chick in each brood during every visit. To determine the start time of the behavioral sample after locating a brood and to randomly choose a chick, we used a random number generator. For five minutes, we recorded forage rate (i.e., peck, pull, probe) continuously and behavioral state (i.e., forage, sit, run, walk, chase, flee, preen, encounter with another individual) and habitat (i.e., moist sand, moist vegetation, dry sand, dry vegetation, and wrack) every 10 s (Altmann 1974). We conducted a disturbance point count for which we recorded the presence of anything other than the brood (e.g., humans, vehicles, predators, competing piping plovers) within 100 m of the observer for five minutes, noting their location relative to the brood. If more than one observer was present, we conducted the behavioral sample and the disturbance point count separately but simultaneously; if only one observer was present during the brood observation, we conducted the disturbance point count immediately after the behavioral sample. If the brood moved more than 100 m during the observation, we randomly selected another chick and conducted a behavioral sample at the new location.

Modified methods for 2020: During 2020, broods were visited by local land managers as often as possible until chicks were confirmed as fledged. No brood observations or disturbance samples were taken in 2020.

Methods post 2020: We resumed searching for plover chicks for 30 days post-hatching. In 2021, we did not conduct brood behavior/disturbance observations. Brood behavior/disturbance observations resumed in 2022.

Invertebrate sampling

Methods prior to 2020: We sampled terrestrial invertebrates from 2014 – 2019 in various habitat types (backshore, overwash, breach fill, restoration area, beach) from the time that the first chicks hatched until the time the last chicks fledged. We collected four samples during each invertebrate sampling set. Two of these samples were taken on the beach and two were taken at randomly selected locations within the subsite. For the beach samples, we followed a transect parallel to the beach just above the intertidal zone. Using a Garmin GPS unit, we navigated to the last data point collected on the beach during the previous sampling set in the subsite. We then used a random number generator to determine the distance (m) to the first sampling location. We



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walked 50 m along the beach for the subsequent sample. The third and fourth samples for the set were chosen from a predetermined, random selection of points within 100 m of the previous sample. We sampled invertebrates using two paint sticks coated with Tanglefoot® Tree Insect Barrier (Tanglefoot Company, Grand Rapids, MI). One paint stick was coated on one side with Tanglefoot and placed flat on the sand, Tanglefoot side up. The second paint stick was coated on both sides with Tanglefoot and positioned vertically in the sand. The sticky area on the horizontal paint stirrer was 64.5 cm² and 129 cm² on the vertical. We placed a cage over the trapping station to prevent any birds from getting stuck in the Tanglefoot (Figure 11). We returned to each sample location after 30 min and identified, counted, and measured (± 1 mm) any arthropods to taxonomic Order.

Modified methods for 2020 and 2021: Invertebrate sampling was not conducted.

Methods post 2021: We resumed sampling for terrestrial invertebrates, adhering to our previously used methods.



Photo: K. M. Walker

Figure 11. Invertebrate trapping station with a vertical and horizontal paint stirrer coated in Tanglefoot®, placed in a wire cage.

Piping plover nest cameras

Methods prior to 2020: Starting in 2016, we placed remote, motion-triggered cameras (Reconyx Rapidfire, RC60 Covert IR, Holmen, WI) at plover nests to detect anything that visited the nest enclosure. All cameras were placed in a camouflaged box for theft protection. We programmed the cameras to have a 5 min picture interval, and take 3 pictures per trigger with a 5 s delay after each trigger. To reduce the chance that a camera would cause a decrease in nest survival, we only placed cameras at exclosed nests. Cameras were installed once we successfully trapped adults and the nest reached full clutch. Before installing the camera box, we made observations of piping plover tracks surrounding the nest enclosure, and chose an area that did not appear to be heavily used by adults entering or exiting the enclosure. When installing the cameras, our first priority was to place the boxes in low-traffic area to reduce camera-caused disturbance. We then attempted to face the boxes north or south to avoid having them face directly towards the sun, thereby preventing overexposure of photographs. We placed the camera box against the outside of the enclosure and aimed the camera toward the nest. We attached the camera box to the enclosure with a bungee cord to secure it in place (Figure 12). We performed any maintenance needed to cameras, such as replacing memory cards and batteries, while we conducted daily nest checks to reduce disturbance. The cameras were removed when the nest failed or the brood permanently left the enclosure.

Modified methods post 2019: No nest cameras were deployed. We intend to deploy cameras at nests in 2023.





Photo: VTSP 2018 field team

Figure 12. A remote nest camera installed outside a nest enclosure with a Snowy Owl (*Bubo scandiacus*) to the left of the enclosure.

Chapter 1. Piping plover numbers, density, and distribution on Fire Island

Objective P1: Compare changes in numbers, density, and distribution of piping plovers in various habitats on Fire Island

Introduction

Hurricanes, wildfires, and other disturbance events are important drivers of ecological processes and creators of habitat heterogeneity (Turner 2010). On barrier islands, process such as overwashing and breaching which occur because of high surges during storms like Hurricane Sandy, often result in the creation of new, early successional habitats by removing or burying vegetation and creating bayside sand flats (Feagin et al. 2015, Robinson et al. 2019). Such habitats are used by species or populations that are at-risk, such as the Atlantic Coast piping plover (Cohen et al. 2009a, Walker et al. 2019).

Habitat changes resulting from anthropogenic and natural forces affect piping plover population dynamics (Cohen et al. 2009a, Catlin et al. 2015). Many plover populations are regulated by density-dependent forces on the breeding grounds (Cohen et al. 2009a, Catlin et al. 2019a). There is evidence that adult survival, breeding propensity, nest survival, and chick survival are reduced when nesting densities are high (Hunt et al. 2018, Weithman et al. 2019). Lower densities may reduce intra- and inter-specific competition among individuals for nesting and foraging space (Ashmole 1963). Reduced competition for space, and thus an increase in available nesting and foraging territories, may lead to a population increase in density-dependent species, as has been observed in piping plover on several occasions (Wilcox 1959, Cohen et al. 2009a, Catlin et al. 2015, Hunt et al. 2018, Robinson et al. 2019). Given the increase in early successional habitat following Hurricane Sandy, a similar process is predicted to unfold in the piping plover population on Fire Island and Westhampton Island.

The disturbance from Hurricane Sandy has the potential to lessen the negative feedback from density-dependent processes acting on the Fire Island and Westhampton Island plover population by increasing the amount of early successional habitat available. With the addition of the breach fills and restoration areas, we were able to assess how plovers used natural and human-created habitats and how plover abundance changed following their creation. Specifically, we investigated how these processes interacted with the changes in piping plover abundance, density, and distribution following Hurricane Sandy. To test whether plover populations on Fire Island and Westhampton Island were density-dependent, we monitored changes in population size, availability habitat, and density following Hurricane Sandy. If plovers are regulated by density-dependent factors, we predict plover numbers will quickly increase in response to the new available habitat and eventually stabilize around carrying capacity (Wilcox 1959, Cohen et al. 2009a, Robinson et al. 2019).

Methods

Using color bands and field-readable, alphanumeric ID flags, we banded adults and chicks from 2013 – 2022¹. Our goal was to capture every unbanded plover each year to help us identify individuals for a variety of analyses. By capturing the majority of unbanded adults and chicks in each year, we could understand the contribution of immigration to the study population.

We compared the number of pairs that used various habitats during 2013 – 2022, including backshore, restoration areas, natural overwashes, and manipulated overwashes to determine if plovers were using any of these created habitats more often than habitats that were available before Hurricane Sandy (Table 1). Using the GPS locations recorded at every nest, we assigned nests a discrete habitat type and describe changes to the

¹ Excluding 2020.

proportion of nests per habitat type. We assumed that if a male renested in the same subsite or habitat type, the location of the renest was not independent of the first nest. Thus, we only included renest attempts of faithful pairs (i.e., same male and female) in the habitat-specific pair counts if their new nest was in a different subsite or different habitat type.

We estimated annual pair numbers, and used males to assign pair numbers. We used male-defined pairs because as males tend to defend the territory and will retain territories after pair divorce (Haig and Oring 1988, Friedrich et al. 2015). Using banded adult information and nest date overlap, we used the greatest number of male territories per management unit each breeding season as an estimate of the number of pairs within a given management unit. To account for pairs that included unbanded individuals, we used the initiation and failure dates to determine if the pairs were independent from individuals banded later in the season. We used this estimate to avoid double counting any male that nested more than once in a management unit, resulting in a conservative pair estimate.

We monitored nests each year for success and to determine the cause of nest failures. Nests were considered successful if they hatched at least one chick and were considered failed if abandoned, depredated, flooded, or experienced complete egg loss that was not due to hatch. Any time an egg was missing during a routine nest check, we attempted to determine the cause of egg loss. In most cases, egg loss could be attributed to hatch, as evidenced by chicks in the nest cup. In cases where chicks were not present, we assigned fate based on evidence within 2 m of the nest (e.g., predation: mammal or avian tracks or visible egg yolk, flooding: sand erosion from high water). If a nest disappeared with no discernable evidence, we assigned it an unknown fate. If a nest appeared to be abandoned (e.g., cold, dirty eggs), we looked for any sign that an adult was incubating the eggs for three or more consecutive visits during a six-day period. If there was no sign of an adult tending the eggs after six days, we considered the nest to be abandoned.

Nests also were given a habitat designation depending on their habitat type and geographic location on the island. The habitats assigned included swale, natural overwash, manipulated overwash, dune, manipulated dune, barrier interior, backshore, or restoration area. Nests and their associated breeding pairs were then regrouped into overwash, breach fill, restoration area, or backshore/barrier interior to compare how many pairs nested in newly created habitat versus previously available habitat. Overwashes, breach fills, and restoration areas all represent new habitats created after Hurricane Sandy. The USACE provided polygons delineating restoration areas (USACE 2014), and we generated polygons to delineate all additional post-Hurricane Sandy habitats. Backshore/barrier interior was habitat available to plovers before the overwashes and breaches occurred in 2012. Natural overwash was only assigned to a nest if it was within the original, Hurricane Sandy-created overwashes. Nests found in overwashes created after Hurricane Sandy were given a habitat designation other than natural overwash.

We estimated nesting plover pair density by subsite, management unit, and total study area for all years 2013 – 2022. Using the delineated subsites, extending from the ocean to the bay, we calculated the hectares of dry sand in each subsite (Appendix 1. Supplemental methods). Using the Region Group tool in ArcGIS, we calculated size of each dry sand patch in the study area, applying an 8-cell neighbor rule wherein both orthogonally- and diagonally-adjacent cells are used to quantify contiguous patches (i.e., Moore neighborhood; Kari 2005). We then removed all patches that consisted of $< 1 \text{ m}^2$ of contiguous dry sand. We assessed density using the total number of pairs that nested in the study area, management unit, or subsite. Pair counts were obtained from Robert Moses and Fire Island National Seashore staff for 2013 and 2014 in Robert Moses State Park and the Fire Island Lighthouse. Pair counts for 2020 were provided by Robert Moses State Park, the National Park Service, and Suffolk County Parks. For subsite density, we grouped OSS and STP to allow for comparisons of island-cross



sections. For management unit density, we grouped the Fire Island Lighthouse Tract with Robert Moses State Park, as no perceivable barrier to piping plover movement was detectable between the two management units. We compared the density between subsites that had direct bay access (DMP, OSS-STP, PSO, NBW, and CSW) and subsites that did not have direct bay access. We used the estimated densities to test for a relationship between reproductive output and density later in this report.

Analyses were conducted in the R statistical environment ver. 4.1.3 (R Core Team 2021).

Results

Piping plover numbers

From 2013 – 2022, we banded 348 adult piping plovers (183 females and 165 males) and 1,218 chicks (Table 4). In 2013 – 2019 and 2022, all chicks were banded between 0 – 16 d old. Due to time constraints associated with a reduced team, chicks were banded between 0 – 29 d old in 2021. In 2013 and 2014, eight nests were found at Watch Hill. Five adults and three chicks from those nests were banded. Of the 1,218 total chicks banded, 2.8% were banded in 2013, 3.4% in 2014, 3.9% in 2015, 8.3% in 2016, 10.7% in 2017, 13.9% in 2018, 20.0% in 2019, 16.4% in 2021, and 20.8% in 2022.

Table 4. Piping plover adults and chicks banded on Fire Island and Westhampton Island per management unit during the full 2013 – 2022 study period. Subsites within each management unit that were monitored are included in parentheses. See Table 2 for subsite names and delineations.

Management unit	Adults		Chicks
	Females	Males	
Robert Moses State Park (DMP, OSS, STP, FLD3, FLD4, FLD5)	57	43	292
Fire Island National Seashore – Lighthouse Tract (FIL)	6	4	42
Fire Island National Seashore – west of breach (OIWR, OIW)	5	12	95
Fire Island National Seashore – east of breach (OIE, BRO)	25	26	216
Smith Point County Park (BRE, PSO, PSE, NBW, NMA, NBE, SPTW, SPTE)	57	51	427
Cupsogue Beach County Park (CSW, CSE)	23	16	126
Offsite – Bellport to Watch Hill	10	13	20
Total	183	165	1,218

Since 2013, we located and monitored 695 nests (461 [66.3%] surviving to hatch and 234 [33.7%] failed; Table 5). In 2022, we found 133 nests, 88 (66.2%) of which survived to hatching and 45 (33.8%) failed (Table 6).

Table 5. Piping plover nest distributions and status within Fire Island and Westhampton Island study area, in various habitats of each management unit, 2013 – 2022. Nest numbers include re-nest attempts of some pairs. Nests were only designated as within natural overwash if they were within the Hurricane Sandy-created overwashes.

Management unit	Habitat	Total found	Hatched	Failed	Hatch %	Fail %
Robert Moses State Park ^a	Backshore	104	73	31	70	30
	Barrier interior	38	24	14	63	37
	Dredge spoil	13	10	3	77	23
	Dune	9	7	2	78	22
	Management area	1	1	0	100	0
	Swale	8	6	2	75	25
Fire Island National Seashore – Lighthouse Tract ^a	Backshore	8	8	0	100	0
	Barrier interior	6	3	3	50	50
	Manipulated dune	7	7	0	100	0
	Swale	1	1	0	100	0
Fire Island National Seashore – west of breach	Backshore	32	22	10	69	31
	Barrier interior	3	3	0	100	0
	Natural overwash	31	23	8	74	26
Fire Island National Seashore – east of breach	Backshore	8	7	1	88	13
	Manipulated dune	1	0	1	0	100
	Natural overwash	107	66	41	62	38
Smith Point County Park	Backshore	53	32	21	60	40
	Barrier interior	24	18	6	75	25
	Manipulated dune	15	8	7	53	47
	Manipulated overwash	72	44	28	61	39
	Great Gun restoration area ^a	64	43	21	67	33
	New Made restoration area ^a	27	13	14	48	52
Cupsogue Beach County Park ^b	Backshore	6	4	2	67	33
	Barrier interior	11	7	4	64	36
	Manipulated dune	3	3	0	100	0
	Manipulated overwash	35	24	11	69	31
Offsite – Bellport to Watch Hill ^c	Backshore	8	4	4	50	50
Total		695	461	234	66.3%	33.7%

^a Not monitored 2013 – 2014

^b Not monitored in 2013 and 2020 – current

^c Bellport Beach to Watch Hill only monitored 2013 – 2014



Table 6. Annual nest fates on the Fire Island and Westhampton Island study area within each management unit, 2013 – 2022. Fire Island National Seashore includes nests from Watch Hill in 2013 – 2014. Cupsogue Beach County Park was only monitored 2013 – 2019. Nests not monitored by Virginia Tech in 2020 were obtained from USFWS, NPS, New York State Parks, and Suffolk County Parks correspondence and annual reports, and appear as bold italicized text within the table.

Year	Management unit	Total found	Successful	Failed			
				Abandoned	Depredated	Flooded	Unknown/ other
2013	Fire Island National Seashore	11	8 (73%)	1 (9%)	-	1 (9%)	1 (9%)
	Smith Point County Park	9	3 (33%)	-	5 (56%)	-	1 (11%)
	Cupsogue Beach County Park	4	3 (75%)	-	-	-	1 (25%)
2014	Fire Island National Seashore	11	4 (36%)	5 (45%)	2 (18%)	-	-
	Smith Point County Park	9	4 (44%)	-	5 (56%)	-	-
	Cupsogue Beach County Park	5	4 (80%)	-	1 (20%)	-	-
2015	Robert Moses State Park	10	5 (50%)	5 (50%)	-	-	-
	Fire Island National Seashore	4	2 (50%)	-	2 (50%)	-	-
	Smith Point County Park	23	2 (9%)	1 (4%)	18 (78%)	-	2 (9%)
	Cupsogue Beach County Park	11	7 (64%)	-	2 (18%)	1 (9%)	1 (9%)
2016	Robert Moses State Park	12	8 (67%)	4 (33%)	-	-	-
	Fire Island National Seashore	9	7 (78%)	1 (11%)	1 (11%)	-	-
	Smith Point County Park	16	12 (75%)	1 (6%)	1 (6%)	-	2 (13%)
	Cupsogue Beach County Park	6	6 (100%)	-	-	-	-
2017	Robert Moses State Park	13	6 (46%)	3 (23%)	-	1 (8%)	3 (23%)
	Fire Island National Seashore	14	10 (71%)	2 (14%)	-	1 (7%)	1 (7%)
	Smith Point County Park	24	19 (79%)	2 (8%)	2 (8%)	-	1 (4%)
	Cupsogue Beach County Park	11	5 (45%)	2 (18%)	2 (18%)	-	2 (18%)
2018	Robert Moses State Park	19	14 (74%)	2 (11%)	1 (5%)	2 (11%)	-
	Fire Island National Seashore	13	10 (77%)	1 (8%)	1 (8%)	1 (8%)	-
	Smith Point County Park	31	20 (65%)	4 (13%)	7 (23%)	-	-
	Cupsogue Beach County Park	7	6 (86%)	1 (14%)	-	-	-
2019	Robert Moses State Park	26	17 (65%)	4 (15%)	2 (8%)	2 (8%)	1 (4%)
	Fire Island National Seashore	29	21 (72%)	3 (10%)	1 (3%)	-	4 (14%)
	Smith Point County Park	37	27 (73%)	6 (22%)	2 (5%)	-	2 (5%)
	Cupsogue Beach County Park	11	7 (64%)	-	2 (17%)	-	2 (17%)
2020	Robert Moses State Park	26	20 (77%)	5 (19%)	-	1 (4%)	-
	Fire Island National Seashore	21	20 (95%)	-	-	-	1 (5%)
	Smith Point County Park	27	22 (82%)	2 (7%)	2 (7%)	-	1 (4%)
2021	Robert Moses State Park	30	24 (80%)	1 (3%)	-	4 (13%)	1 (3%)
	Fire Island National Seashore	42	25 (60%)	3 (7%)	2 (5%)	5 (12%)	7 (17%)
	Smith Point County Park	41	25 (61%)	5 (12%)	8 (20%)	-	3 (7%)
2022	Robert Moses State Park	37	27 (73%)	5 (14%)	-	3 (8%)	2 (5%)
	Fire Island National Seashore	51	34 (67%)	11 (22%)	-	-	6 (12%)
	Smith Point County Park	45	27 (60%)	7 (16%)	6 (13%)	-	5 (11%)
Total		695	461 (66.3%)	87 (12.5%)	75 (10.8%)	22 (3.2%)	50 (7.2%)



Of the nests monitored in 2022 that failed, 23 were abandoned, six were depredated, three flooded, two were attributed to human vandalism², and 11 fates were unknown (Table 7). Of the six nests that were depredated, four were the result of feral cats, one was depredated by an unknown mammal, and one was believed to be depredated by an unknown bird; five of the six depredated nests were not exclosed (Table S2). Nest abandonment was the greatest cause of nest loss in 2022, representing 51.1% of all nest failures (Figure 13). The greatest rate of nest depredation was observed in 2015 (45.8%; Figure 13).

Table 7. Piping plover nest distributions and fates on the Fire Island study area, in various habitats of each management unit, April – August 2022. Nest numbers include renest attempts of some pairs. Nests were only designated as within natural overwash if they were within original Hurricane Sandy-created overwashes.

Management unit	Habitat	Nests			Reason for loss
		Total found	Hatched	Failed	
Robert Moses State Park	Backshore	11	8	3	Abandoned (1), flooded (2)
	Barrier interior	21	15	6	Abandoned (3), flooded (1), unknown/other (2)
	Dredge spoil	5	4	1	Abandoned (1)
Fire Island National Seashore – Lighthouse Tract	Backshore	3	3	0	-
	Barrier interior	3	2	1	Other (1)
Fire Island National Seashore – west of breach	Backshore	12	10	2	Abandoned (2)
	Barrier interior	3	3	0	-
Fire Island National Seashore – east of breach	Backshore	7	6	1	Abandoned (1)
	Natural overwash	23	10	13	Abandoned (8), unknown (5)
Smith Point County Park	Backshore	20	11	9	Abandoned (4), predation (4), unknown (1)
	Barrier interior	4	3	1	Abandoned (1)
	Manipulated dune	3	1	2	Abandoned (2)
	Restoration area	18	12	6	Predation (2), unknown (4)
Total		133	88	45	

² On 24 May, VTSP researchers at Robert Moses State Park observed a damaged exclosure and missing eggs from a plover nest located at the stockpile next to the Field 5 parking lot. VTSP researchers contacted Robert Moses park staff, who filed the incident report. A second vandalized nest was reported the same day by NPS staff in the lighthouse tract.

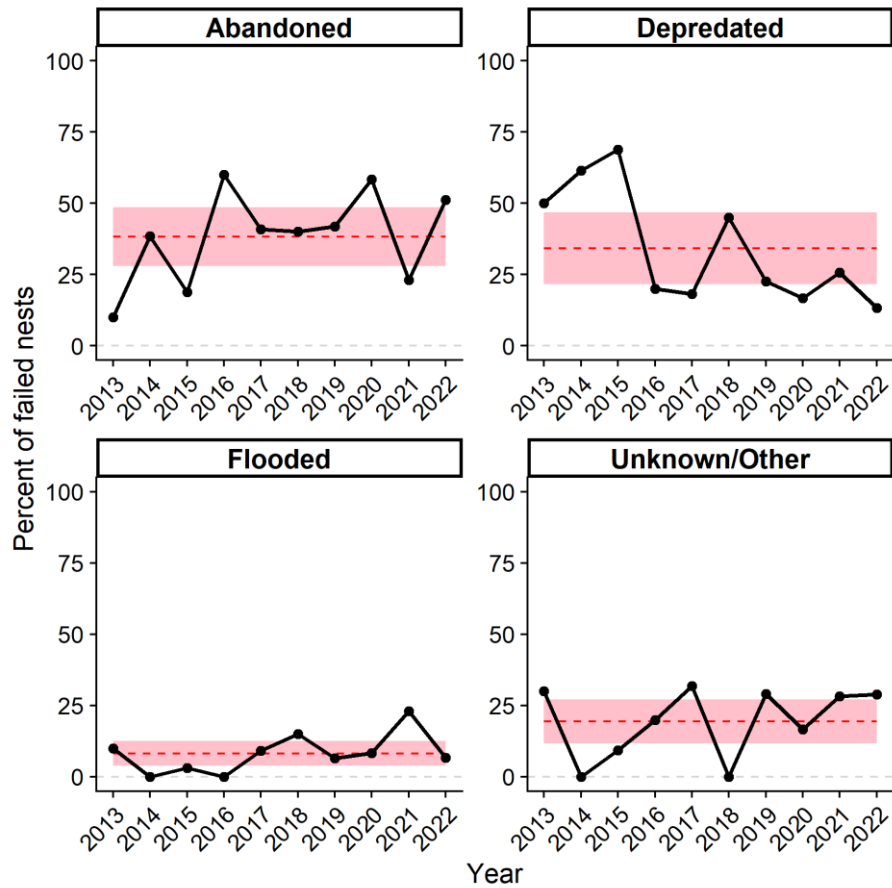


Figure 13. Annual causes of nest failure on Fire Island, NY from 2013 – 2022. Mean percent of nest failure ($\pm$ 95% CIs) is presented for each cause in red; annual observations above or below the confidence interval indicate a statistically significant increase or decrease (respectively).



Piping plover distributions

The number of pairs monitored increased from 19 in 2013 to 86 in 2019 (352.6% increase), temporarily decreasing to 71 pairs in 2020, and rebounding to 116 pairs in 2022 (63% increase since 2020 and 510.5% since 2013), the maximum number of pairs recorded at Fire Island (Table 8). Across all years of the study, approximately half of the pairs monitored (51.7%) nested in backshore and barrier interior habitats, and overwash habitats supported approximately a third of the pairs monitored (31.8%). Restoration areas were not created until after the 2014 breeding season, and they have been used by piping plovers each season thereafter (Table 8).

Table 8. Number of monitored piping plover pairs in each of the various habitats. Restoration areas were not created until after the 2014 nesting season. Some pairs had more than one nest per year, but re-nest attempts were only counted if the subsequent nest was in a different subsite and habitat class. Table footnotes indicate changes in monitoring area among years.

Year	Habitat			
	Backshore & barrier interior	Breach fill	Overwash	Restoration
2013 ^a	7 (37%)	1 (5%)	11 (58%)	-
2014 ^a	3 (14%)	1 (5%)	17 (81%)	-
2015 ^b	12 (34%)	3 (9%)	15 (43%)	5 (14%)
2016 ^b	16 (43%)	3 (8%)	14 (38%)	4 (11%)
2017 ^b	13 (29%)	3 (7%)	23 (51%)	6 (13%)
2018 ^b	24 (41%)	3 (5%)	21 (36%)	10 (17%)
2019 ^b	45 (52%)	3 (3%)	25 (29%)	13 (15%)
2020 ^{b,c}	38 (54%)	2 (3%)	21 (30%)	10 (14%)
2021 ^c	61 (66%)	1 (1%)	18 (20%)	12 (13%)
2022 ^d	79 (71%)	0 (0%)	18 (16%)	15 (13%)
Total	298 (51.7%)	20 (3.5%)	183 (31.8%)	75 (13.0%)

^a Includes Watch Hill nesting pairs, monitored 2013 – 2014 only

^b Includes Robert Moses and the Lighthouse Tract monitored after 2014

^c Does not include Cupsogue Beach County Park

^d An additional six pairs nested in habitat other than these four main categories, or had nests that were never found by VT researchers

Using information from USFWS and NPS, we estimated pairs for the entire study area (Robert Moses State Park to Cupsogue Beach County Park; Table 9). Though there was fluctuation within management units, the overall number of pairs increased nearly every year following Hurricane Sandy.

Table 9. Annual number of piping plover pairs nesting within each management unit across the study area. Pair counts for areas not monitored by Virginia Tech in a given year (i.e., Robert Moses State Park in 2013 – 2014, and all of the study area in 2020) were obtained from USFWS, NPS, New York State Parks, and Suffolk County Parks correspondence and annual reports, and appear as bold italicized text within the table.

Management unit	Year									
	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022
Robert Moses State Park	12	10	10	9	8	15	22	23	24	34
Fire Island Lighthouse	1	1	0	1	1	1	4	5	5	6
Fire Island National Seashore – west	3	1	1	1	0	1	7	7	11	16*
Fire Island National Seashore – east	4	4	2	6	10	9	13	11	18	25*
Smith Point County Park	6	8	13	13	20	25	31	25	34	36
Cupsogue Beach County Park	3	5	7	6	6	7	9	10	-	-
Outside study area – Bellport to Watch Hill	3	3	0	0	0	0	3	1	-	-
Outside study area – Davis Park to Kismet	0	0	0	0	1	0	0	5	-	-
All areas	32	32	33	36	46	58	89	87	-	-
Fire Island and Cupsogue sites	29	29	33	36	45	58	86	81	-	-
Fire Island sites	26	24	26	30	39	51	77	71	92	116*

*One pair nested in the western wilderness before relocating to the east wilderness (40 unique pairs between FNS west and east wilderness areas)

Piping plover densities

Density (pairs/ha) of nesting piping plovers across the entire study area increased from 0.06 pairs/ha of dry sand habitat patches > 1m² in 2013 to 0.37 pairs/ha in 2022. Density increased slowly from 2013 – 2016, and increased rapidly after 2017 (Figure 14).

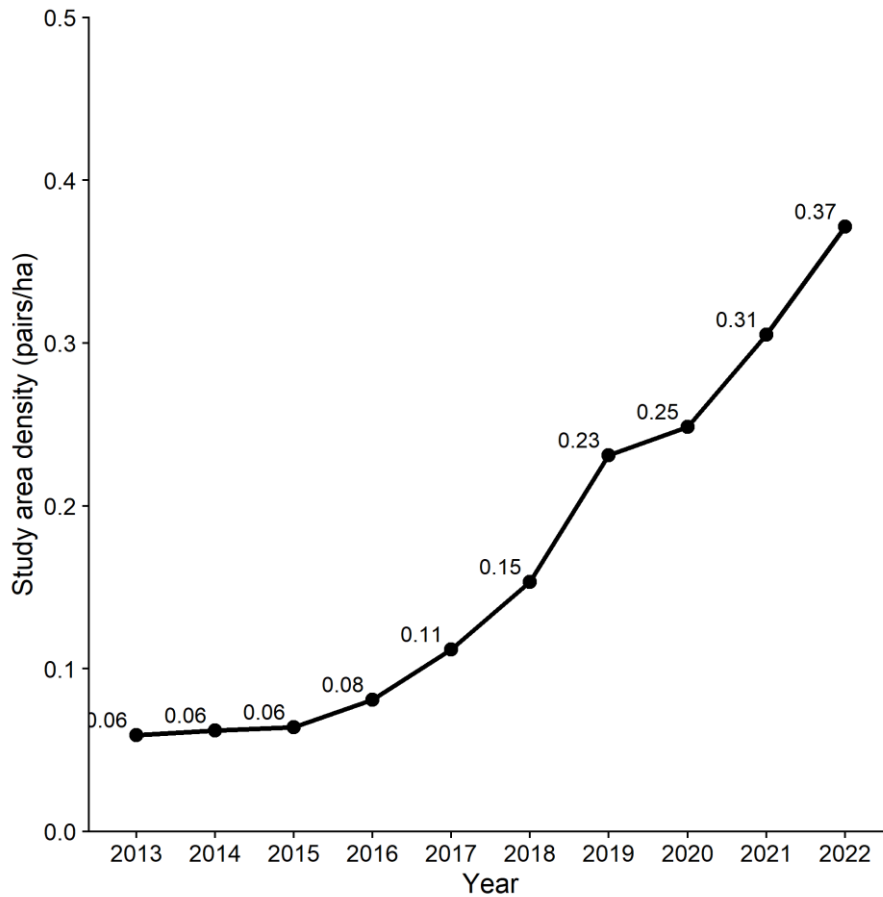


Figure 14. Estimated study area density (pairs/ha) for each year following Hurricane Sandy. Available habitat was defined as all available dry sand patches $\geq 1 \text{ m}^2$ using 15 cm classified aerial imagery in each year. For 2013 and 2014, the number of pairs was a combination of the number of pairs monitored by Virginia Tech and the number of pairs estimated at Robert Moses State Park and the Lighthouse Tract from USFWS, NPS, New York State Parks, and Suffolk County Parks correspondence and annual reports. For 2020, the number of pairs was the total estimated by land managers at the National Park Service, New York State Parks, and Suffolk County Parks. All annual density measures exclude observations from Cupsogue Beach County Park.

Density increased within all management units during 2013 – 2019. Density at Robert Moses State Park and Smith Point County Park have increased slowly since 2019 (Robert Moses mean $\pm$ SD: 0.24 ± 0.05 pairs/ha, Smith Point mean $\pm$ SD: 0.30 ± 0.05 pairs/ha; Figure 15). In contrast, density at the NPS West and East Wilderness areas has continued to increase rapidly (West Wilderness: 0.34 pairs/ha in 2019 to 0.69 pairs/ha in 2022 [102.9% increase], East Wilderness: 0.42 pairs/ha in 2019 to 0.68 pairs/ha in 2022 [61.5% increase]; Figure 15).



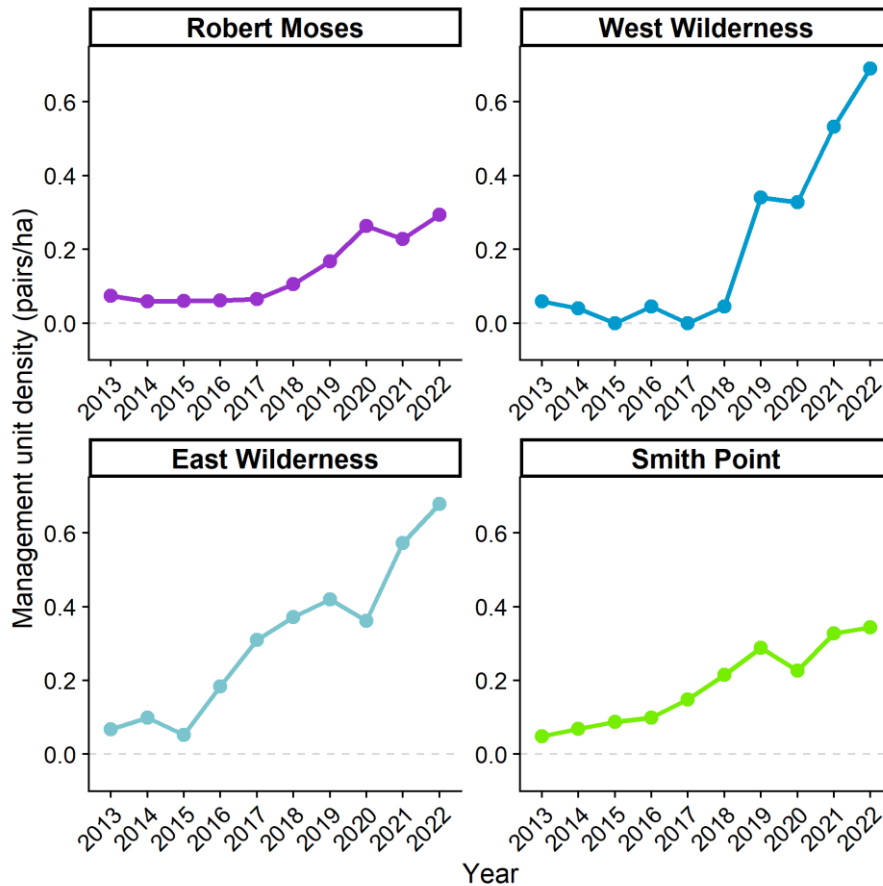


Figure 15. Estimated management unit density (pairs/ha) for each year following Hurricane Sandy. Available habitat was defined as all available dry sand patches $\geq 1 \text{ m}^2$ using 15 cm classified aerial imagery for each year. For 2013 and 2014, the number of pairs was a combination of the number of pairs monitored by Virginia Tech and the number of pairs estimated at Robert Moses State Park and the Lighthouse Tract. For 2020, the number of pairs was the total estimated by land managers at the National Park Service, New York State Parks, and Suffolk County Parks. The Lighthouse Tract was grouped in with Robert Moses State Park for all years.

Mean subsite density increased from 0.08 pairs/ha in 2013 to 0.36 pairs/ha in 2019, and has remained stable at 0.36 pairs/ha since 2019 (Figure 16). When separated by subsites with and without bay access, subsites with bay access had, on average, greater densities (Figure 17). During 2013 – 2019, subsites with bay access increased at a marginally greater rate; density across all areas have been comparatively stable since 2019 (Figure 17).

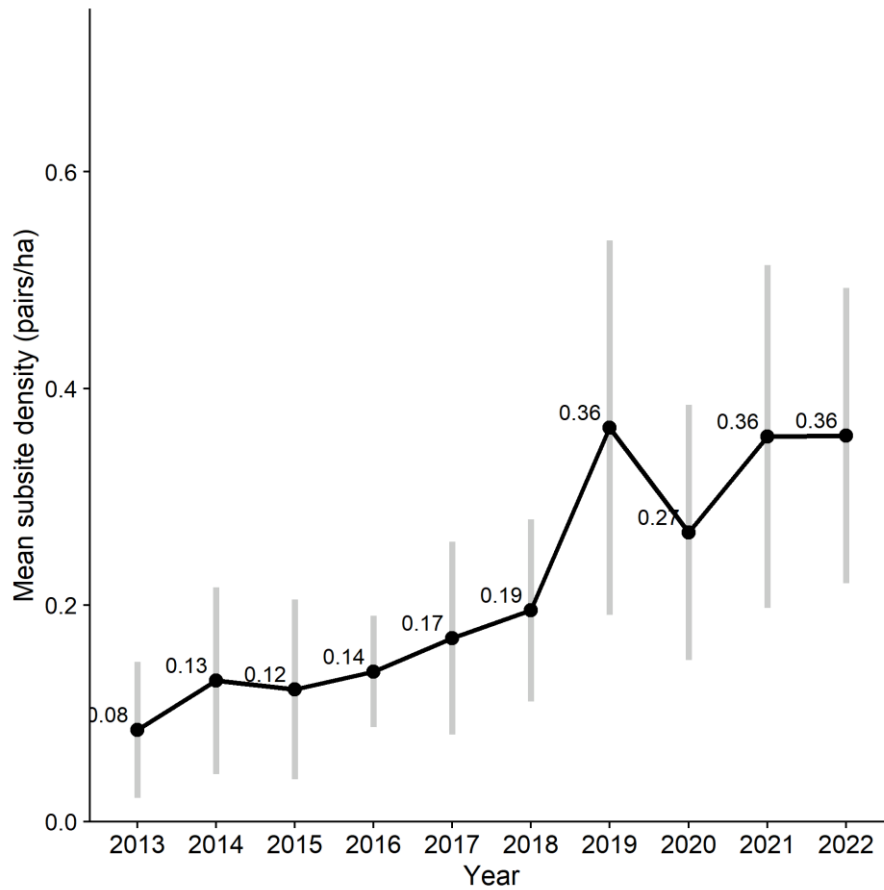


Figure 16. Mean annual subsite density (pairs/ha). Subsite extent for density extended the entire width (north to south) of the island. Available habitat was defined as all dry sand patches $\geq 1 \text{ m}^2$ using 15 cm classified aerial imagery. Error bars represent 95% confidence intervals. All annual density measures exclude observations from Cupsogue Beach County Park.



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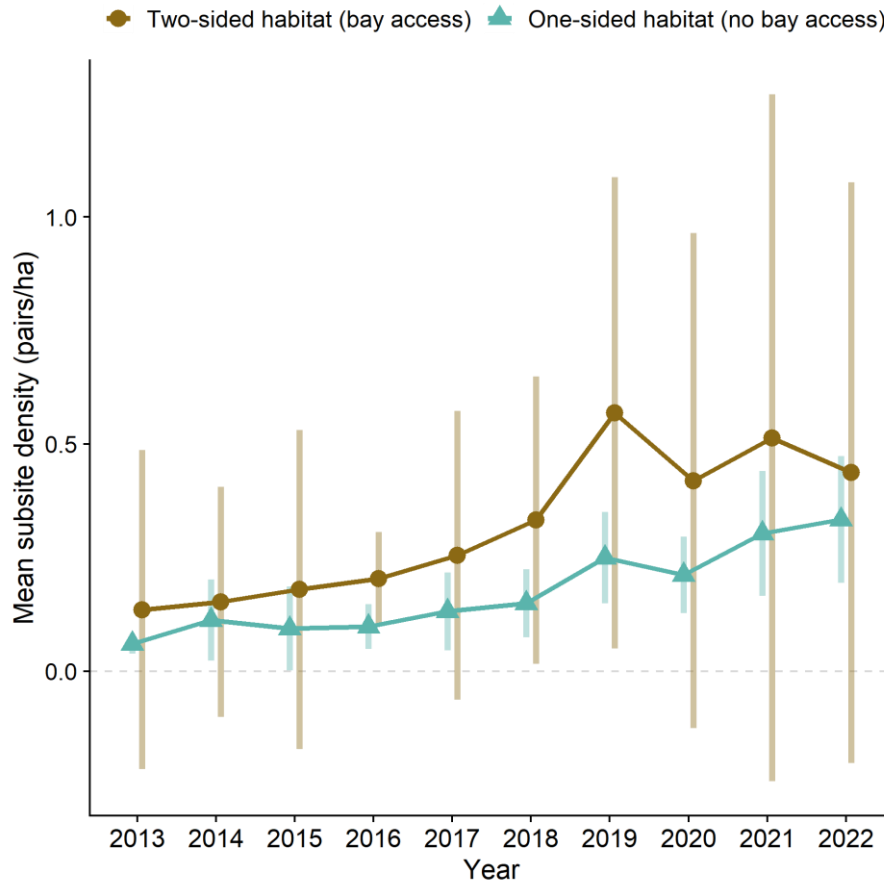


Figure 17. Average subsite density (pairs/ha) in each year for subsites with bay access (two-sided habitat; i.e., DMP, OSS/STP, PSO, NBW, CSW) compared to subsites without direct bay access (one-sided habitat; all other subsites, Table 2). Subsite extent for density extended the entire width (north to south) of the island. Available habitat was defined as all contiguous dry sand patches (identified via an 8-cell rule; see Methods) $\geq 1 \text{ m}^2$ using 15 cm classified aerial imagery. Error bars represent 95% confidence intervals. All annual density measures exclude observations from Cupsogue Beach County Park.

Discussion

Since Hurricane Sandy, the increase in breeding pairs at Fire Island has been dramatic (2013 – 2022: 346.2%), and largely driven by recent increases in available habitat and high immigration rates (Robertson et al. 2019, Walker et al. 2020, Wails et al. 2021). In comparison to other habitat creating events, the observed increase falls within the range of past observations during similar periods (42% – 745% increases across all US populations post-habitat creation; events that resulted in >350% increases observed after 1938 hurricane and 1992 nor'easter at West Hampton Island, NY, and after flooding events in 1997, 2010, and 2011 along Missouri River sites in SD and NE [Robinson et al. 2019]). For population growth to occur, sufficient immigration and annual reproduction to compensate for annual mortality is necessary, and we estimate that an average of 31% of adult plovers in this population die annually (Robinson et al. 2020a, Walker et al. 2020). Early in our study, population growth rates were low, likely a result of poor local and regional reproduction observed just before and immediately following Hurricane Sandy (USFWS 2019). Without substantial immigration, this population may have declined (Robinson et al. 2020a, Walker et al. 2020), as occurred elsewhere in the NY-NJ recovery unit in



2012 – 2013. In more recent years, pairs have increased, with some interannual fluctuations that could be attributed to nest depredation and sampling effort. Predators have been known to associate nest exclosures with a food source (Murphy et al. 2003, Niehaus et al. 2004, but see Black et al. In Press which showed no evidence that red foxes in this study area were attracted to nest exclosures), which could explain the apparent high percent of nest failure in 2015 following the removal of nest exclosures. In 2020, overall abundance and pairs seemingly decreased within all management units except Robert Moses (one pair increase) and the West Wilderness Area (seven pairs, consistent with 2019). Yet, nesting densities have remained relatively stable for several areas. While the slight decrease in 2020 could be reflective of natural population fluctuations, it may also be a consequence of reduced monitoring effort due to the SARS-CoV-2 pandemic that would affect detection of plover nesting activities.

Most plover pairs nested in natural and manipulated overwash habitats from 2013 – 2015. During 2016 – 2018, we observed fluctuations in nesting habitat selection after beach widening in 2015 at Smith Point County Park. During this period, the plover pairs alternated between the backshore and barrier interior habitats and overwashed areas. By 2019, the greatest percentage of plover pairs were in backshore and barrier interior habitats. Hurricane Sandy created overwashes, and additional overwashes were created by other storm events. Since 2012, there have been multiple occasions where a storm surge created overwash habitat on a smaller scale than Hurricane Sandy. For example, in 2019, we observed new overwash habitat west of Old Inlet on Fire Island National Seashore. Following this overwash, Old Inlet West pair numbers increased, from one pair in 2018 to seven in both 2019 and 2020. While technically overwash habitat, many of the nests in newly-created overwashes were classified as backshore and barrier interior habitat, since they were not a result of Hurricane Sandy, and our objective was primarily to understand the effect of Hurricane Sandy on the plover population. Though created at different times, these other overwashes are functionally comparable habitat to the Hurricane Sandy natural overwash. Thus, the detected decline in plover pairs nesting in overwash habitats should be interpreted as a declining use of Hurricane Sandy-created habitats through time.

Pair density increased 5-fold from 2013, the first breeding season following Hurricane Sandy, to 2022. The increase in nesting density observed this year was largely driven by changes occurring in the NPS East and West Wilderness areas. Old Inlet, one of the breaches from Hurricane Sandy, was left to undergo natural accretion and erosion processes. Compared to previous years, the area immediately east of the breach (the Old Inlet east subsite; Table 2) has increased in area from sand accretion, forming two-sided habitat (i.e., sand habitat with both ocean and bay access that is traversable by a walking plover), which supports a greater density of nesting plovers than one-sided habitat (Cohen 2005, Cohen et al. 2009, this study). We are anticipating a similar marked increase in nesting density at Robert Moses State Park, particularly at Democrat Point which recently underwent habitat restoration (Chapter 6. Assessing the effects of habitat restoration for piping plover (*Charadrius melodus*) at Democrat Point, Fire Island, NY of this report). Similar to at Smith Point (particularly at Great Gun) between 2019 – 2020, we suspect that densities remained somewhat stable 2020 – 2022 at Democrat Point due to the restoration, which would increase the total area of dry sand used in density calculations.

We hypothesize that density will begin to decline as the piping plover population on Fire Island nears carrying capacity. In a study of Westhampton Island, Cohen et al. (2009a) identified carrying capacity as 1.05 pairs/ha in two-sided habitats and 0.50 in one-sided habitats (i.e., no sand connection between ocean and bay). Beyond these densities, the population began to decline. Now a decade after Hurricane Sandy, the density for the Fire Island population is less than half of what Cohen et al. (2009a) found for two-sided habitats (Cohen et al. [2009] estimate = 1.05 pairs/ha vs. 2022 mean $\pm$ SE = 0.44 ± 0.20 pairs/ha) and about two-thirds of what was found for one-sided habitat (Cohen et al. [2009] estimate = 0.50 pairs/ha vs. 2022 mean $\pm$ SE = 0.33 ± 0.06 pairs/ha). However, density is sensitive to the boundaries in which it is defined. Here, we classified available piping plover



nesting habitat as any patch of dry sand $\geq 1 \text{ m}^2$ composed of 15 cm dry sand pixels. This fine-scale habitat delineation likely identified habitat that plovers may not use, and these areas appear to coincide with human disturbance, which has been shown to negatively influence habitat use and chick survival (Burger 1994, DeRose-Wilson et al. 2018). For example, Field 5 at Robert Moses supported two plover pairs across 16.9 ha dry sand in 2022. Field 5 is a popular recreational beach, and heavy human use may discourage more piping plovers from attempting to nest. In contrast, other, less-frequented areas (e.g., NPS wilderness areas, Democrat Point) supported a greater proportion of the nests monitored in 2022. Further, in recent years of our study, reproductive output has not yet declined below what would be needed for population stationarity (1.10 fledged chicks/pair for stationary population [Weithman et al. 2019]; 1.30 fledged chicks/pair in 2022, mean = 1.28 fledged chicks/pair, 95% CI = 1.03 – 1.53 fledged chicks/pair [Wails et al. 2022]). Similarly, we have not previously detected slowing immigration rates (Walker et al. 2020). Taken together, it appears that this population may not yet be nearing carrying capacity.

Given ongoing vegetative succession, it is possible that availability of nesting habitat may soon limit population growth within some management units. After Hurricane Sandy, vegetation increased in each of the newly created habitats from 2013 – 2017, especially in overwash areas (Bellman 2018), where it exceeded the 30% vegetation cover threshold established under the BO for vegetation removal (USFWS 2014). Since 2018, vegetation continued to increase on dry, sandy areas across the study area (Chapter 5. Assessing the role of nest site viewshed on nest site selection of this report). This increase in vegetation is concurrent with overall increases in plover density. Piping plovers generally nest in sparsely vegetated or flat, open sand due to the potential increased risk of depredation associated with dense vegetation (Espie et al. 1996, Cohen et al. 2008). Elsewhere, piping plover populations have declined when habitat was overgrown with vegetation (Wilcox 1959, Cohen et al. 2009a). Therefore, we expect that the amount of suitable nesting habitat will ultimately reduce and then slow or eliminate population growth within the study area if vegetation is left unmanaged. Continued vegetation management, as was done in the Great Gun restoration area in the early spring of 2019 and 2020 and most recently at Democrat Point in the winter of 2021/2022, will likely be needed to maintain open patches of nesting habitat in areas of dense vegetation or in sites where vegetation inhibits chick access to foraging habitat.

The implications of this study on plover population dynamics are numerous, and continual monitoring is required to understand the factors influencing this population's growth and eventual limitations. Although carrying capacity has not been reached, the combination of ongoing habitat management and turnover in the predator community will have important implications for both long- and short-term plover population growth. Indeed, the local predator community has played an important role on reproductive output (Walker et al. 2020, Wails et al. 2021, Catlin et al. In Prep, this report) and the population growth rate, but we also have observed greater conspecific disturbance and competition each year as density increases. The succession of plant communities at a natural rate and accelerated by anthropogenic activities, like dune construction and snow fence installation, also has been occurring in the overwashes and breach fill habitats for more than ten years and is contributing to nesting and foraging habitat loss and fragmentation. Without the regular creation of new habitat, either mechanically or naturally, this population may soon begin to experience negative density-dependent feedback as habitat availability decreases (Cohen et al. 2009a, Catlin et al. 2015). How the population responds to these pressures within the following few years will be crucial in determining carrying capacity and its correlates.



Chapter 2. Piping plover nest and chick survival rates

Objective P2: Compare piping plover nest and chick survival rates in various habitats across Fire Island

Introduction

Reproductive output may correlate with population change in piping plovers (Saunders et al. 2018). For Atlantic piping plovers, nest survival tends to be high due to the installation of nest exclosures (Burger 1987, Rimmer and Deblinger 1990, Melvin et al. 1992). Nest exclosures, however, do not prevent all causes of nest failure, such as flooding or depredation by predators that can fit between the wire mesh of nest exclosures (e.g., ghost crabs, *Ocypode quadrata* [Kwon et al. 2018]; American mink, *Neovison vison* [M. Stantial and C. Weithman, pers. comm.]). Once the precocial plover chicks have hatched, a nest exclosure is no longer useful at protecting chicks, which move about and forage for themselves soon after hatch (Cairns 1977). For effective adaptive management, it is important to understand fluctuations in plover reproductive output and survival.

Piping plover reproductive output at Fire Island has been assessed in terms of human disturbance (DeRose-Wilson et al. 2018) and in comparison to a more southerly site on the Atlantic Coast (Weithman et al. 2019). Here, we focus on the components of reproductive output (i.e., nest survival and chick survival) on Fire Island and Westhampton Island following Hurricane Sandy and implementation of the FIMI project. We also are interested in any relationship between reproductive output with time since Hurricane Sandy, and geomorphic and habitat variables. Specifically, we studied the effects of human-created nesting habitats for piping plovers, which were built as intentional nesting habitat (i.e., restoration areas), for island stabilization (i.e., breach fills), and those that were created by Hurricane Sandy (i.e., natural and manipulated overwash; Table 2).

Methods

Analyses were completed in the R statistical environment ver. 4.1.3 (R Core Team 2021).

Nest survival

We used a logistic exposure model (Rotella et al. 2004, Shaffer 2004) to estimate the daily survival rates of piping plover nests throughout the study period. We built models that compared survival between exclosed (i.e., large, wire cages placed around nests to reduce predation; Melvin et al. 1992) and exposed (i.e., nests without exclosures) nests and management units. A nest was considered exclosed during the period when the nest exclosure was present through hatching. When predicting from this model, hatch date was set at 34 days from the nest initiation date (Heisey et al. 2007).

Chick survival

To estimate chick survival, we used a modified Dail-Madsen model, which is a Bayesian implementation of the Lukacs young survival from marked adults model, applied in JAGS (Lukacs et al. 2004, Kellner and Meredith 2021). We modeled chick survival for each year (excluding 2020). We estimated daily survival for a 25-day period using 5-day chick detection intervals. We considered two models to derive estimates of chick survival for the whole study area. These included an annual survival model and one which estimated chick survival among management units. To estimate the study and management unit-specific probability of survival to fledge, we calculated the product of the five 5-day interval survival rates from the year-specific model.

Reproductive output

We estimated reproductive output using annual monitored pair counts and the number of fledglings observed for each year. Chicks were considered fledged if they reached 25 days of age or were observed in sustained flight.



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Results

Nest survival

The probability of surviving to hatch varied across years of our study (Figure 18, Table S1a). In general, nest survival was greatest in 2016 and lowest in 2015.

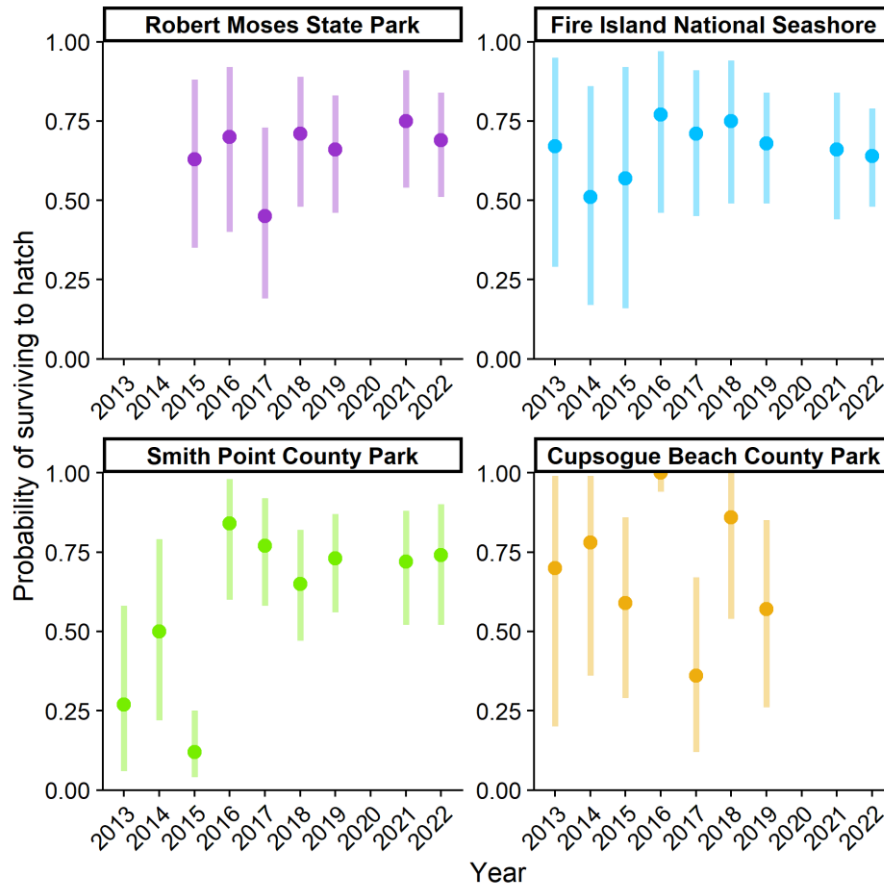


Figure 18. Probability of plover nests surviving to hatch (34 days from nest initiation) for each management unit monitored from 2013 – 2022 (excluding 2020). Survival estimates are from a logistic exposure model, estimating the interactive effect of year and management unit on daily nest survival raised to the 33rd power. Error bars represent 95% credible intervals.

Overall, nests that were exclosed during or after clutch completion were predicted to have a greater probability of surviving to hatch relative to unexclosed nests (Figure 19). Survival to hatch for exclosed nests ranged from 0.49 – 0.83, and survival to hatch for unexclosed nests ranged from 0.25 – 0.69 (Figure 19). In 2015, exclosures were removed from all of the Smith Point County Parks nests (approximately 15 days after clutch completion) after a fox entered two exclosures through the top. The nests in 2015 that had been previously exclosed had a 0.00 probability of surviving to hatching. For both exclosed and unexclosed nests, the probability of surviving to hatch has varied minimally since 2017 (Figure 19).



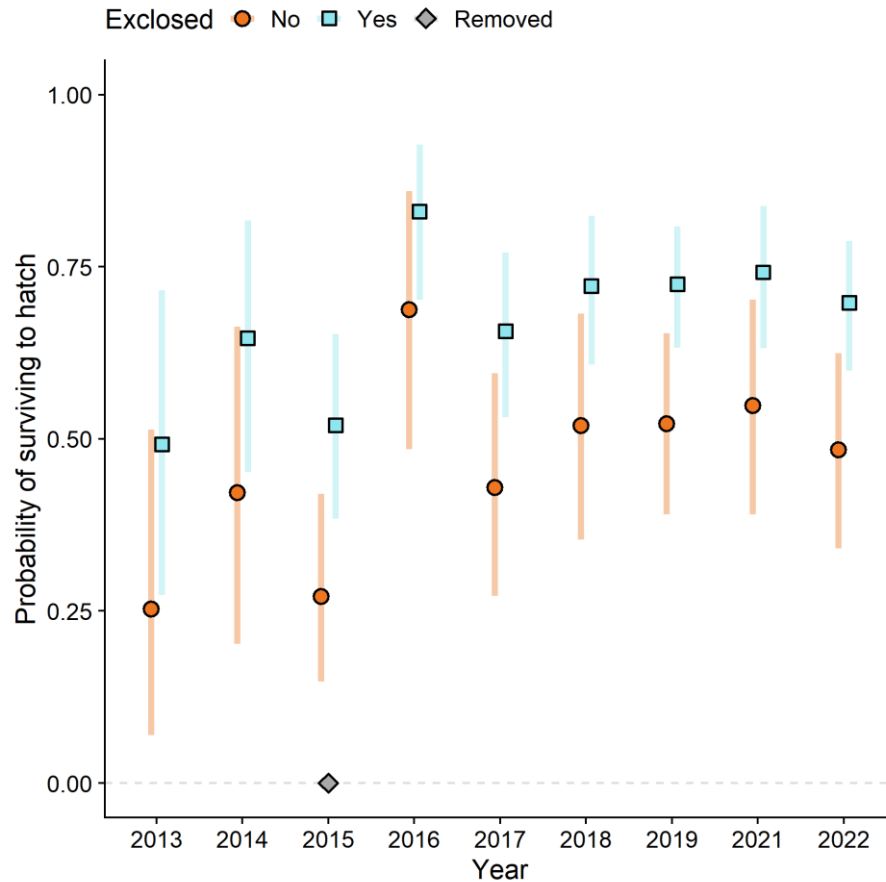


Figure 19. The probability of piping plover nests surviving to hatching with an exclosure and without an exclosure on Fire Island and Westhampton Island, 2013 – 2022 (excluding 2020). Error bars represent 95% credible intervals.

From 2013 – 2022, 78 of 675 nests were abandoned (11.6%; Table 10). Of the 78 abandoned nests, 30 nests (38%) belonged to a pair of unbanded or unknown adults, 25 nests (32%) belonged to a pair with one banded adult, and 23 nests (29%) belonged to a pair of banded adults. Of the 55 nests with at least one identifiable adult, 44 of the 74 (59.5%) known adults were seen at least once after nest abandonment. Throughout the duration of this study, we recorded five occasions in which a nest was abandoned around the same time as the mortality of one of the incubating adults. We discovered the remains of four of these banded adults within 90 m of their nest.

Table 10. Abandonment rate of piping plover nests with exclosures and without exclosures on Fire Island and Westhampton Island study area 2013 – 2022. Nests not monitored by Virginia Tech in 2020 were obtained from USFWS, NPS, New York State Parks, and Suffolk County Parks correspondence and annual reports, and appear as bold italicized text within the table.

Year	Exclosed	Nests (n)	Failed	Hatched	Abandoned	Abandonment rate ^a
2013	N	6	5	1	0	0.00
	Y	14	4	10	1	0.07
2014	N	8	5	3	0	0.00
	Y	12	4	8	2	0.17
2015	N	25	20	5	2	0.08
	Y	23	12	11	5	0.22
	Removed	5	5	0	0	0.00
2016	N	8	3	5	1	0.13
	Y	32	4	28	4	0.13
2017	N	16	14	2	4	0.25
	Y	45	7	38	4	0.09
2018	N	9	9	0	1	0.11
	Y	58	8	50	6	0.10
2019	N	22	15	7	1	0.05
	Y	78	13	65	9	0.12
2020	N	25	7	18	3	0.12
	Y	49	5	44	4	0.08
2021	N	28	17	11	1	0.04
	Y	85	22	63	8	0.09
2022	N	33	21	12	5	0.15
	Y	99	23	76	17	0.17
Total – unexclosed		180	116	64	18	0.10
Total – exclosed		495	102	393	60	0.12

^a Calculated as the proportion of abandoned nests out of all nests observed that year

Chick survival

From 2013 – 2022, we monitored 457 broods at Fire Island and West Hampton Island³, though in 2021, we could not visit all broods at regular intervals due to reduced field capacity in observance of COVID-19 mitigation protocols (piping plover general field methods of this report). Together, we included 355 broods across nine years in this analysis (77.7% of all monitored broods; Table S1b).

Chick survival to fledge was highly variable among years, with the lowest survival rate in the first year of the study, 2013, and the highest rate in 2018 (Figure 20). Likewise, chick survival varied between management units (Figure 21).

³ In 2020, 67 broods were monitored by land managers but were not regularly observed; thus, we did not consider 2020 data in analyses of chick survival to fledging.



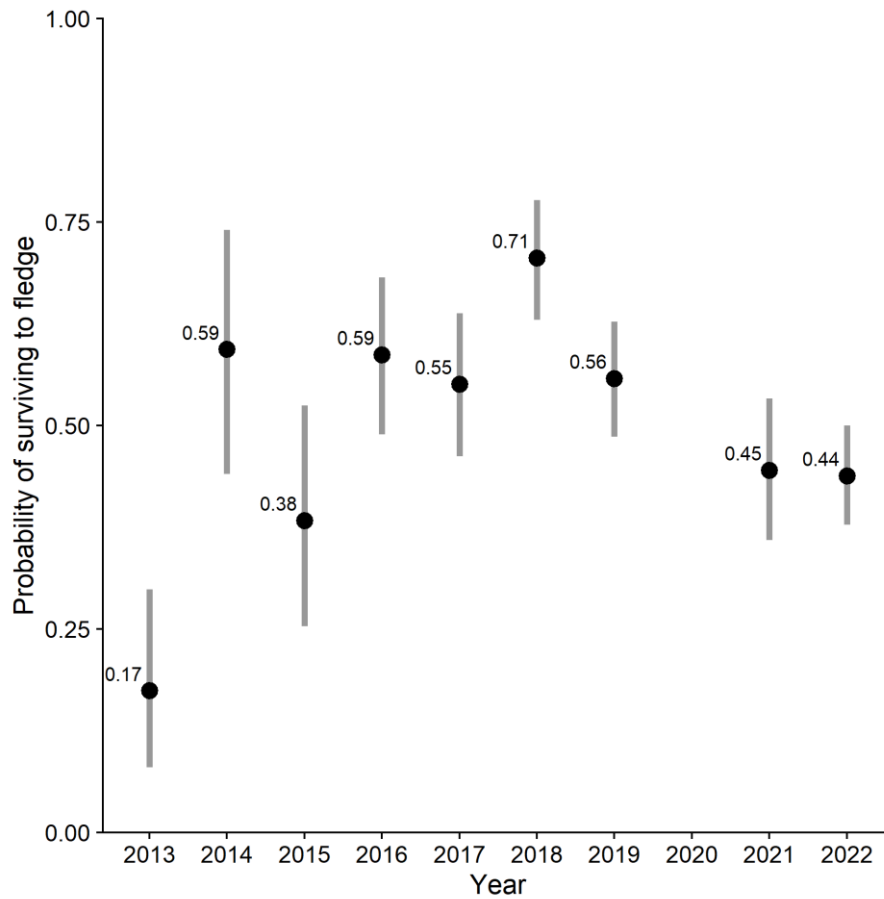


Figure 20. Probability of surviving to fledge (25 days) for chicks 2013 – 2022 (excluding 2020) on Fire Island and Cupsogue Beach County Park. Error bars represent 95% credible intervals. Annual mean probability of surviving to fledge is labeled. Note that the survival estimates for 2021 was based on observations from focal subsites only.



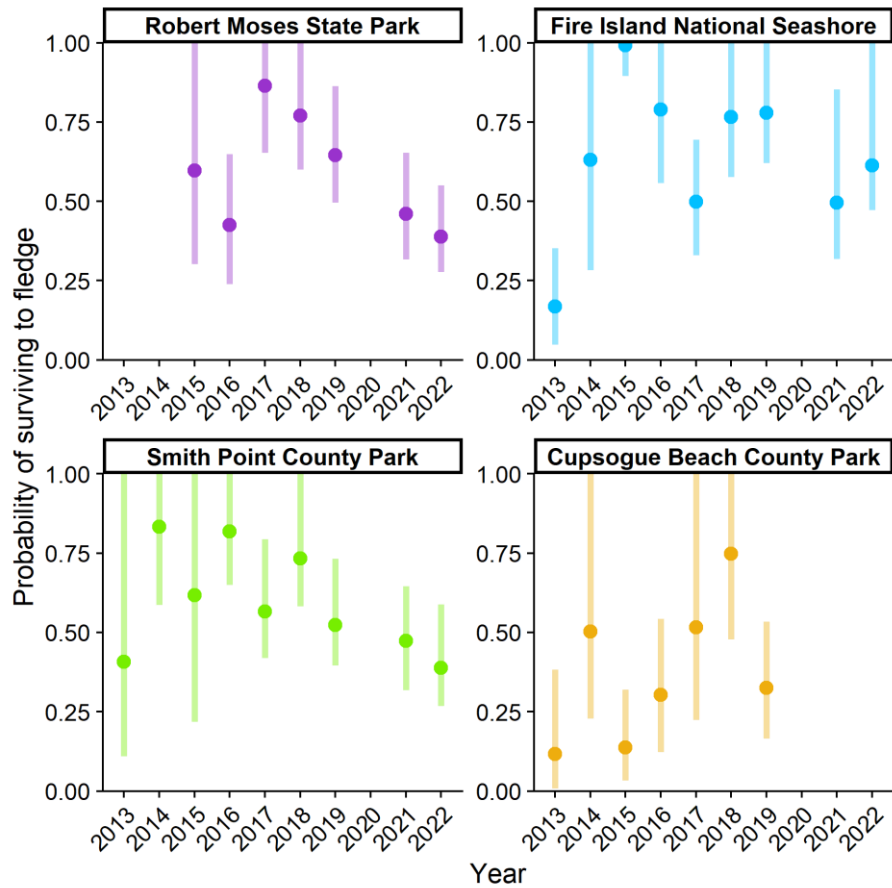


Figure 21. Probability of surviving to fledge (25 days) for chicks in all management units 2013 – 2022 (excluding 2020), estimated from a young survival model. 2021 data considers broods in focal subsites only.



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Reproductive output

Piping plover reproductive output across the study area was variable among years, with the highest study area reproductive output (2.02 fledglings/pair) in 2018 and the lowest observed reproductive output (0.39 fledglings/pair) in 2015 (Table 11).

Table 11. Piping plover pair counts and reproductive output (fledglings/pair; R_o) in four different management units on Fire Island and Westhampton Island, 2013 – 2022. Pairs is the number in each subsite, and R_o is the reproductive output (chicks fledged/pair). Pair counts and reproductive output not recorded by Virginia Tech in 2020 were obtained from USFWS, NPS, New York State Parks, and Suffolk County Parks correspondence and annual reports, and appear as bold italicized text within the table.

Year		Robert Moses State Park	Fire Island National Seashore ^a	Smith Point County Park	Cupsogue Beach County Park	Total
2013	Pairs	-	10	6	3	19
	R_o	-	0.60	0.50	0.00	0.47
2014	Pairs	-	8	8	5	21
	R_o	-	0.75	1.50	1.20	1.14
2015	Pairs	10	3	13	7	33
	R_o	0.30	1.67	0.31	0.14	0.39
2016	Pairs	9	8	13	6	36
	R_o	1.11	2.00	2.38	1.00	1.75
2017	Pairs	8	11	20	6	45
	R_o	1.75	1.64	1.70	0.67	1.56
2018	Pairs	15	11	25	7	58
	R_o	2.27	2.64	1.80	1.29	2.02
2019	Pairs	22	24	31	9	86
	R_o	1.36	1.83	1.32	1.00	1.44
2020	Pairs	23	23	25	10	81
	R_o	1.65	2.22	2.08	1.10	1.88
2021	Pairs	24	34	34	-	92
	R_o	1.71	1.41	1.68	-	1.59
2022	Pairs	34	46	36	-	116
	R_o	1.38	1.50	1.03	-	1.32

^a Does not include sites outside the study area (Bellport to Kismet)

Discussion

In previous years, we reported evidence that nest exclosures improved nest survival (Walker et al. 2020, Wails et al. 2021), and the same appears to be true for 2022. Nest exclosures are used across the study area to reduce nest predation (Melvin et al. 1992), which has largely contributed to the consistent annual nest survival for exclosed nests we have observed at Fire Island throughout our study. In some instances, predators have been known to associate exclosures as a potential food source (Murphy et al. 2003, Barber et al. 2010, Beaulieu et al. 2014). In 2015, for example, Smith Point County Park had high fox density (Chapter 8. Predator numbers, density, and distribution on Fire Island, with an emphasis on red fox of this report), and plover nest depredation occurred after a fox entered through the top of nest exclosures. Following these predation events, the exclosures were removed from all other nests at the county park. Once the exclosures were removed, the



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remaining nests were depredated. Later in 2018, a fox observed at Smith Point was digging at exclosures. After the digging incidents, exclosures were not placed on several nests throughout the eastern portion of Smith Point County Park in 2018 due to concern that exclosures would attract predators.

While nest exclosures enhance nest survival, there may be an associated risk of adult depredation (Murphy et al. 2003, Barber et al. 2010, Cohen et al. 2016, Anteau et al. 2022). Depredation of adults can be difficult to immediately detect, though multi-year studies have confirmed that adult mortality is correlated with apparent nest abandonment (Neuman et al. 2004, Barber et al. 2010, Roche et al. 2010). In the Great Lakes, for example, 70% of the instances where nest abandonment was observed coincided with the disappearance of at least one of the incubating adults (Roche et al. 2010). Predator activity should be considered and carefully monitored to limit the risk of adult depredation (Cohen et al. 2016, Darrah et al. 2020).

At Fire Island, the highest nest abandonment rates at exclosed nests (2014, 2016) occurred in the years before the widespread outbreak of mange in fox. While fox activity has been low across the island since the mange outbreak, cats are likely now the primary mammalian predator of piping plover on Fire Island (Chapter 8. Predator numbers, density, and distribution on Fire Island, with an emphasis on red fox of this report, Table S4). In previous years, our resight rate (or minimum survival rate) of banded birds after abandonment (55%) was lower than the past annual survival rate of 0.69 ± 0.02 (with detection probability of 0.96 ± 0.02 for adults; Walker et al. 2020). This year, we resighted only 50% of plovers that abandoned their nests after evidence of predators was observed around nest exclosures (Wails et al. 2022b), which was also less than the annual survival rate of 0.70 ± 0.01 (with detection probability of 0.97 ± 0.02 for adults; Chapter 3. Population growth of this report). Therefore, we suspect that adult mortality or emigration from the study area are the two most likely causes for the relatively low percentage of banded adults resighted after nest abandonment (Roche et al. 2010).

In previous years, nest survival was positively correlated with nesting plover density, though the opposite was true for chick survival (Walker et al. 2020). Plovers are territorial and compete for space during the nesting and brooding period (Elliott-Smith and Haig 2004). High plover nesting density can lead to intra-specific aggression and attract the attention of shorebird predators (Whitfield 2003, Hunt et al. 2019). Since 2018, we have detected a decline in plover chick survival to fledging which could be explained by changing vegetation cover and predator activity, and further mediated by plover nesting density. Vegetation may offer plovers cover from some avian predators (Schmidt 1999, Ellis et al. 2020). However, in other areas with more dense vegetation, additional plant cover likely conceals terrestrial predators (Espie et al. 1996, Cohen et al. 2008), and may limit plovers' detection of predators (Chapter 5. Assessing the role of nest site viewshed on nest site selection of this report). In 2018, a plover band was found in a scat sample, confirming depredation of an adult plover by a feral cat in Smith Point County Park (Black 2021, Wails et al. 2021), and have observed multiple instances of cat depredation in 2022 (Chapter 8. Predator numbers, density, and distribution on Fire Island, with an emphasis on red fox of this report). Careful consideration of vegetation cover will contribute to the outcomes of predator control programs (Hunt et al. 2019).

Chapter 3. Population growth

Objective P3: Compare the relative contribution of local growth (recruitment) vs. immigration in population growth associated with new habitats. Determine habitat correlates of population change

Introduction

Changes in population vital rates as a function of population density (i.e., density-dependence) are required for population regulation (Royama 1977, Turchin 1990, Newton 1998, Coulson et al. 2008). In such regulation, some or all vital rates (e.g., reproductive output, survival, immigration, emigration) decline at high population densities and increase at low population densities, leading the population to a maximum sustainable level, or carrying capacity (Coulson et al. 2008, Gamelon et al. 2016, Andersen et al. 2017). Density-dependent responses can occur due to site-dependence, interference, crowding, and predation, and are particularly notable in territorial species (Rodenhouse et al. 1997, Nevoux et al. 2011). An increase in habitat following a disturbance event is likely to release a population from existing negative density-dependent processes by decreasing population density.

When habitats change following disturbance events, increasing the quantity or quality of that habitat and lessening density-dependent constraints, the composition of the breeding cohort may change, altering immigration and local recruitment rates. Broadly, a breeding population can be composed of (1) adult birds that returned to the area in which they previously nested (adult residents), (2) yearling birds that return to their natal site (natal recruits), and (3) birds that immigrated from other areas (immigrants; Lampila et al. 2016, Millon et al. 2019). Newly created habitat may change the contribution from each cohort, such as by an increase in immigrants or natal recruits, which can settle due to newly opened territories.

For piping plovers, population changes often follow habitat-creating events, such as major floods or hurricanes (Cohen et al. 2009a, Hunt et al. 2018, Catlin et al. 2019a, Catlin et al. 2019b, Robinson et al. 2019). However, the demographic drivers of these changes may vary. Potential density-dependent reasons for these changes could be increased immigration into the newly available habitat, increased reproductive output leading to more natal recruits, or increased adult and post-fledgling survival (Cohen et al. 2009a, Catlin et al. 2015, Hunt et al. 2018). Immigration rates often are unknown as they are difficult to measure. Even though immigration is more often studied in birds than in mammals (Millon et al. 2019), we still lack detailed studies on the role of immigration in the population dynamics of many birds, including the piping plover. Studies on the Atlantic Coast population have indicated that immigration is influential in changing a local piping plover population (Cohen et al. 2009a, Weithman et al. 2019, Robinson et al. 2020a). To understand the relative contributions of local recruitment and immigration to population change on Fire Island and Westhampton Island, we (1) determined the composition of the nesting adult cohort following Hurricane Sandy, (2) compared use by each nesting class to available habitat to determine if any particular habitat could be driving population change, and (3) assessed adult and hatch-year survival to determine if increased survival could be leading to observed population change.

Methods

We were able to band most chicks and adults in each year from 2013 – 2019 and in 2022; therefore, we estimated immigrants using the number of newly banded adults in a year. In 2020, no banding occurred; therefore, we considered the number of unbanded birds to be the number of immigrants. While great effort is exercised to confirm individual birds to nests each year, we were not able to confirm each bird as a breeder in each year due to trap aversion, nests in denser vegetation, and limitations of site accessibility and visitation frequency while observing COVID-19 protocols. Here, we assume that our detection was not biased towards any specific nesting class. If individuals that hatched in the study population returned to breed ≥ 2 years after



hatching, we counted them as natal recruits in the year of first breeding. Although a population is likely to have floater adults that could not obtain either a mate or a breeding territory (Penteriani et al. 2011), they do not contribute to the reproductive output in a population and were not included in annual pair counts, and thus we did not consider them here.

Recruitment and immigration

Using banded and known-age birds, we estimated natal recruitment and immigration for 2014 – 2022 whenever possible. We estimated immigration rate using a Pradel (1996) model in MARK (White and Burnham 1999). The parameters estimated from the Pradel model are apparent survival, resight rate, and fecundity rate. We interpreted fecundity as an estimate of immigration rate (the number of immigrants in $t+1$ per individual in year t), which included natal recruits and immigrants. To estimate annual immigration rates, we fit a single model holding apparent survival and resight rate constant, as prior studies indicated that adult survival and detection probability were relatively constant (Catlin et al. 2015, Weithman et al. 2019). For immigration rate in 2014 and 2015, the estimate included only the eastern portion of the study area (Old Inlet West to Cupsogue). Starting in 2016, estimates of immigration rate also included the Democrat Point site, as birds were first banded there in 2015 and thus we could first estimate the breeding cohort composition in 2016. Annual local recruitment was estimated using individuals banded as chicks in years prior that returned to breed for the first time in the population in a given year. In addition to immigration rates, we assessed how many of the newly banded individuals (2014 – 2019) or unbanded individuals (2020; immigrants), local recruits, and adult residents settled into habitats created from Hurricane Sandy (i.e., breach fill, manipulated overwash, natural overwash, restoration area, and other; Table 1).

To determine if adult residents, natal recruits, and immigrants selected for or against certain habitat types, we compared percent area of each habitat type available and the percent each habitat type used by the classes of nesting plovers. To estimate amount of available habitat, we included patches of dry sand in the eastern study area in 2014 – 2022 (2014 – 2019: study area included Old Inlet West to Cupsogue; 2020 – 2022: study area included Old Inlet West to Smith Point East; Table 2) that were $\geq 1 \text{ m}^2$ (Appendix 1. Supplemental methods). We only conducted this analysis for the eastern study area because we had data for each year of the study, and the targeted habitat types were only located in this area. We then determined the amount of dry sand used the within each of the habitat areas, as defined by delineated ocean-to-bay overwash and USACE-provided shapefiles. We calculated the proportion of habitat use as the proportion of returning adult residents, natal recruits, or immigrants in each habitat type. We conducted a chi-squared test to determine if there was overall selection occurring within nesting classes and compared 95% confidence intervals on proportion of use to availability to determine selection for or against particular habitat types (Byers et al. 1984, Catlin et al. 2011b). If a breeding class did not nest in a habitat type within a year, we interpreted it as avoidance. We completed this analysis for each year 2014 – 2022 (excluding 2020 and 2021), to evaluate if selection changed as plover dynamics and island characteristics changed.

Survival and site fidelity

To estimate true survival of piping plovers in the Fire Island study area, we used the Barker live-dead encounter model (Barker 1997). The Barker model uses supplemental (i.e., off-site) resightings to distinguish permanent emigration from mortality for a more accurate estimate of survival than what is available when using return rate alone. We calculated true survival (S) for two age classes following Catlin et al. (2015): after-hatch-year (AHY; individuals ≥ 1 year post-hatch) and hatch-year (HY; hatch to the following breeding season) piping plovers. Other parameters that we estimated using this model were detection probability (p), probability of being reported dead during the supplemental period (r), the probability of being resighted during and surviving the



supplemental period (R), the probability of being resighted during the supplemental period and dying (R'), site fidelity (F), and the probability of returning to the study area after emigration (F').

Model selection occurred in three steps: (1) assessing r , R' , and F' ; (2) assessing p and R ; and (3) assessing S and F . We fixed both r , R' , and F' to 0, as we did not have any reports of dead individuals during the supplemental period, nor did we observe any previously emigrated individuals return to the study area. We modeled variation in p and R with year and COVID disturbance (pre-pandemic and since the pandemic) to account for reduced resighting in 2020 and 2021. We modeled variation in S and F with age, year, the interaction between the two (Weithman et al. 2019). We then modeled all combinations of survival and fidelity structures using parameter structures from steps 1 and 2, and compared models using second-order Akaike's Information Criterion (AIC_c ; Akaike 1974, Hurvich and Tsai 1989). We tested for overdispersion ($\hat{c}$) using a model with age and year variation in all models and corrected for overdispersion if confidence intervals $\hat{c}$ did not overlap 1.

We used annual pre-fledge chick survival estimates (i.e., survival from hatch to fledged; Chapter 2. Piping plover nest and chick survival rates of this report) and our annual estimates of hatch-year survival (survival of individuals < 1 year old) from the Barker model to derive post-fledging survival (i.e., survival from fledging until returning the following breeding season). Post-fledging survival is hatch-year survival divided by pre-fledge survival (Catlin et al. 2015, Weithman et al. 2019).

Results

Recruitment and immigration

From 2014 – 2022, the breeding population included 31 – 54% returning adult residents annually (Table 12). The remainder of the birds that nested in the study area had hatched the year before (natal recruits; 6 – 25%) and unbanded immigrants (34 – 63%). The Pradel model resulted in estimates of combined natal recruitment and immigration between 0.53 and 0.87 immigrants in year $t+1$, per individual in year t from 2014 – 2019.

Table 12. Breeding adult piping plovers divided into three population constituents: returning adult residents, local recruits, and immigrants. Numbers for 2014 and 2015 only include the eastern study area (Fire Island National Seashore Wilderness Area to Cupsogue Beach County Park). Numbers for 2016 – 2019 include the entire study area (Robert Moses State Park to Cupsogue Beach County Park). Numbers for 2020 – 2022 include Robert Moses State Park, Fire Island National Seashore, and Smith Point County Park.

Year	Adult residents	Natal recruits ^a	Immigrants ^b	Immigration Rate ($\pm$ SE) ^c
2014	17 (47%)	2 (6%)	17 (47%)	0.62 (0.20)
2015	20 (53%)	5 (13%)	13 (34%)	0.52 (0.16)
2016	21 (31%)	4 (6%)	42 (63%)	0.87 (0.18)
2017	32 (38%)	16 (19%)	36 (43%)	0.67 (0.12)
2018	46 (43%)	17 (16%)	44 (41%)	0.65 (0.11)
2019	60 (41%)	36 (25%)	50 (34%)	0.75 (0.11)
2020 ^d	65 (54%)	21 (18%)	34 (28%)	-
2021 ^e	-	-	-	-
2022 ^f	55 (43%)	27 (21%)	47 (36%)	-

^a Natal recruits refers to individuals that were hatched as chicks in the study area and returned in a following year to nest.

^b Immigrants refers to the individuals that were newly banded and any birds that were known to have hatched outside of the study area that then nested within the study area.

^c Immigration rate estimated using a Pradel (1996) model in Program MARK. Immigration rate represents the number of immigrants in year $t+1$, per individual in year t .

^d 2020 includes 85% of breeding adults. The other 15% could not be identified into any of the three population constituents.



^e We were unable to quantify individuals per category, as COVID-19 protocols prevented fieldwork and banding in 2020.

^f 2022 includes 70% of breeding adults. The remaining 30% could not be identified into any of the three population constituents.

Breach fills made up the smallest proportion of the eastern study area, ranging from 2.6% – 3.2% of the available nesting habitat during 2014 – 2019 (including Cupsogue breach) and only 0.5% – 1.0% of the available nesting habitat from 2020 – 2022 (excluding Cupsogue breach). Manipulated overwashes comprised 6.2 – 21.4%, natural overwashes 10.3% – 15.7%, and restoration areas 10.3% – 25.0% of the available nesting habitat.

Across all years, 65% of returning adult residents nested in habitats created as result of Hurricane Sandy (Table 13). Adult residents nested in manipulated overwashes (21%) and natural overwashes (23%) the most, followed by restoration areas (12%) and breach fills (9%). The remaining 35% of adult residents nested in other habitat not modified by Hurricane Sandy (Table 13). Relative to the other population constituents, returning adult residents exhibited habitat selection most frequently (Figure 22). Adult residents disproportionately favored breach fills in 2015 and 2017, manipulated overwashes in 2016 and 2018, and natural overwashes from 2017 – 2019. In 2020, adult residents did not favor any particular habitat. In 2022, adult residents disproportionately favored natural overwashes and restoration areas; it was also the first year that adults nested less frequently in any Hurricane Sandy-modified habitat. Adult residents consistently nested less frequently in other habitat not modified by Hurricane Sandy (Figure 22).

Table 13. The number of returning adult residents that nested in habitats created by Hurricane Sandy or the subsequent habitat alterations on Fire Island and at Cupsogue Beach County Park.

Year	Total adult residents (n)	Breach fill	Manipulated overwash	Natural overwash	Restoration area	Other, pre-Sandy habitats	% adult residents in Hurricane Sandy habitats
2014	17	0 (0%)	7 (41%)	4 (24%)	-	6 (35%)	65%
2015	20	5 (25%)	7 (35%)	3 (15%)	1 (5%)	4 (20%)	80%
2016	21	4 (19%)	8 (38%)	6 (29%)	0 (0%)	3 (14%)	86%
2017	32	7 (22%)	7 (22%)	8 (25%)	2 (6%)	8 (25%)	75%
2018	46	3 (7%)	12 (26%)	15 (33%)	3 (7%)	13 (28%)	72%
2019	60	4 (7%)	12 (20%)	13 (22%)	11 (18%)	20 (33%)	67%
2020 ^a	65	2 (3%)	9 (14%)	12 (18%)	11 (17%)	31 (48%)	52%
2021 ^b	-	-	-	-	-	-	-
2022 ^c	31	0 (0%)	0 (0%)	6 (19%)	7 (23%)	18 (58%)	42%
Total	292	25 (9%)	62 (21%)	67 (23%)	35 (12%)	103 (35%)	65%

^a Numbers in 2020 only consisted only of the Fire Island study area and did not include Cupsogue Beach County Park, and represent 85% of breeding adults. The other 15% were not resighted for bands, therefore, could not be identified into any of the three categories.

^b We were unable to quantify individuals per category, as COVID-19 protocols prevented fieldwork and banding in 2020.

^c Includes two individuals that renested in another habitat type.

Natal recruits were the smallest of the three population constituents (6 – 25% over the years; Table 14). Across all years, 57% of natal recruits nested in habitats created as result of Hurricane Sandy (Table 14). Natal recruits nested in restoration areas (26%) and natural overwashes (15%) the most, followed by manipulated overwashes (9%) and breach fills (8%). The remaining 43% of natal recruits nested in other habitat not modified by Hurricane Sandy (Table 14). Natal recruits nested in restoration areas every year, and more frequently than expected in



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2015, 2018, and 2019. Natal recruits avoided nesting in breach fills in 2017 and 2018. No natal recruits nested in breach fills in 2014, 2015, 2018, and 2022; manipulated overwashes in 2016, 2020, and 2022; natural overwashes in 2014, 2015, and 2019; and other habitat not modified by Hurricane Sandy in 2015. During 2016 – 2018, natal recruits selected for natural overwashes in proportion to their availability, selected against them in 2019, and selected for them in 2020 and 2022 (Figure 22).

Table 14. The number of piping plover natal recruits that nested in habitats created by Hurricane Sandy or the subsequent habitat alterations on Fire Island and at Cupsogue Beach County Park.

Year	Total natal recruits (n)	Breach fill	Manipulated overwash	Natural overwash	Restoration area	Other, pre-Sandy habitats	% natal recruits in Hurricane Sandy habitats
2014	2	0 (0%)	1 (50%)	0 (0%)	-	1 (50%)	50%
2015	5	0 (0%)	2 (40%)	0 (0%)	3 (60%)	0 (0%)	100%
2016	4	1 (25%)	0 (0%)	1 (25%)	1 (25%)	1 (25%)	75%
2017	16	3 (19%)	3 (19%)	3 (19%)	3 (19%)	4 (25%)	75%
2018	17	0 (0%)	2 (12%)	1 (6%)	7 (41%)	7 (41%)	59%
2019	36	3 (8%)	2 (6%)	1 (3%)	9 (25%)	21 (58%)	42%
2020 ^a	21	2 (10%)	0 (0%)	7 (33%)	5 (24%)	7 (33%)	67%
2021 ^b	-	-	-	-	-	-	-
2022	16	0 (0%)	0 (0%)	5 (31%)	2 (13%)	9 (56%)	44%
Total	117	9 (8%)	10 (9%)	18 (15%)	30 (26%)	50 (43%)	57%

^a Numbers in 2020 only consisted only of the Fire Island study area and did not include Cupsogue Beach County Park, and represent 85% of breeding adults. The other 15% were not resighted for bands, therefore, could not be identified into any of the three categories.

^b We were unable to quantify individuals per category, as COVID-19 protocols prevented fieldwork and banding in 2020.

Across all years, 44% of immigrants nested in habitat created as result of Hurricane Sandy (Table 15). Immigrants nested in manipulated overwashes (14%) and restoration areas (15%) the most, followed by natural overwashes (12%) and breach fills (4%). The remaining 56% of immigrants nested in other habitat not modified by Hurricane Sandy (Table 15). Based on availability (Figure 22), we would have expected more immigrants to nest in other habitat every year except 2019 and 2020. In 2019 and 2020, immigrants nested in other habitat proportional to its availability. Across Hurricane Sandy habitats, immigrants generally nested with a frequency we would expect based on availability in every habitat type across all years until 2019. Immigrants avoided breach fills in 2019 and 2020, and manipulated overwashes in 2020 and 2022 (Figure 22). 2020 was the first year in which immigrants selected for any habitat, as they nested in restoration areas more than would be expected based on the available nesting habitat and this persisted in 2022 (Figure 22).



Table 15. The number of immigrant piping plovers that nested in habitats created by Hurricane Sandy or the subsequent habitat alterations on Fire Island and at Cupsogue Beach County Park.

Year	Total immigrants (n)	Breach fill	Manipulated overwash	Natural overwash	Restoration area	Other, pre-Sandy habitats	% immigrants in Hurricane Sandy habitats
2014	17	2 (12%)	6 (35%)	3 (18%)	-	6 (35%)	65%
2015	13	1 (8%)	5 (38%)	1 (8%)	3 (23%)	3 (23%)	77%
2016	42	1 (2%)	7 (17%)	5 (12%)	4 (10%)	25 (60%)	40%
2017	36	3 (8%)	6 (17%)	8 (22%)	7 (19%)	12 (33%)	67%
2018	44	3 (7%)	7 (16%)	2 (5%)	6 (14%)	26 (59%)	41%
2019	50	0 (0%)	5 (10%)	7 (14%)	2 (4%)	36 (72%)	28%
2020 ^a	34	0 (0%)	0 (0%)	3 (8%)	10 (29%)	21 (63%)	38%
2021 ^b	-	-	-	-	-	-	-
2022	28	0 (0%)	0 (0%)	3 (11%)	7 (25%)	18 (64%)	36%
Total	264	10 (4%)	36 (14%)	32 (12%)	39 (15%)	147 (56%)	44%

^a Numbers in 2020 only consisted only of the Fire Island study area and did not include Cupsogue Beach County Park, and represent 85% of breeding adults. The other 15% were not resighted for bands, therefore, could not be identified into any of the three categories.

^b We were unable to quantify individuals per category, as COVID-19 protocols prevented fieldwork and banding in 2020.

(a.) Adult residents									
	2014	2015	2016	2017	2018	2019	2020	2021	2022
Breach fill	-	+	=	+	=	=	=		=
Manipulated overwash	=	=	+	=	+	=	=		-
Natural overwash	=	=	=	+	+	+	=		+
Restoration area		=	-	=	=	+	=		+
Other	-	-	-	-	-	-	-		-
(b.) Natal recruits									
	2014	2015	2016	2017	2018	2019	2020	2021	2022
Breach fill	-	-	=	=	-	=	=		=
Manipulated overwash	=	=	-	=	=	=	-		-
Natural overwash	-	-	=	=	=	-	+		+
Restoration area		+	=	=	+	+	=		=
Other	=	-	=	-	-	=	-		-
(c.) Immigrants									
	2014	2015	2016	2017	2018	2019	2020	2021	2022
Breach fill	=	=	=	=	=	-	-		=
Manipulated overwash	=	=	=	=	=	=	-		-
Natural overwash	=	=	=	=	=	=	=		=
Restoration area		=	=	=	=	=	+		+
Other	-	-	-	-	-	=	=		-

Figure 22. Selection by (a.) returning adult residents, (b.) natal recruits, and (c.) immigrants in each habitat type. Plus sign (+) and gold represents significantly more nesting individuals than expected, minus sign (-) and blue represents significantly fewer nesting individuals than expected, and equal sign (=) with gray represents approximately equivalent nesting individuals to expected in a habitat based on availability (see Table 13 – Table 15 for annual percentages). Habitats that did not exist in a year (i.e., restoration areas in 2014) or in instances where individuals could not be assigned into population constituencies are not symbolized.

Survival and site fidelity

The top model from the survival and fidelity analysis indicated temporal variation in survival for HY birds (i.e., survival from hatch until returning the following year), a constant AHY survival rate for all years, an age effect on site fidelity (F), and temporal variation for resighting rates (p , R ; Table 16). Estimates of hatch-year survival varied each year of the study (Figure 23). Our estimate of adult survival ($S = 0.70 \pm 0.01$) was within the range of past studies of adult piping plover survival from 2013 – 2019 (Weithman et al. 2019, Walker et al. 2020). Site fidelity for adult birds ($F = 0.87 \pm 0.01$) was higher than for hatch-year birds ($F = 0.71 \pm 0.08$). Detection probability varied annually, though did not differ with age (p range = 0.72 [2020] – 0.97 [2022]). Post-fledgling survival (i.e., survival from fledging until returning the following year), has been consistently at or above 0.75 (Figure 24). We observed the highest rate of post-fledgling survival among fledglings from 2021.

Table 16. Models for true survival (S), fidelity (F), detection probability (p), and the probability of being resighted during and surviving the supplemental period (R) for adult piping plovers on Fire Island, NY for the three top-ranked (cumulative $\omega = 1.00$) Barker (1997) survival models. The probability of being reported dead during the supplemental period (r), the probability of being resighted during the supplemental period and dying (R'), and the probability of returning to the study area after emigration (F') were fixed to 0 in the model. Models are ranked by second-order Akaike's information criterion (AIC_c) and difference in AIC_c is shown (ΔAIC_c). k is the number of model parameters and ω_i is the AIC_c weight.

S	F	p	R	k	ΔAIC_c	ω_i	Deviance
Age + year + age $\times$ year	Age	Year	Year	44	0.00	0.90	1431.58
Age + year + age $\times$ year	Age + year + age $\times$ year	Year	Year	60	4.40	0.10	1403.24
Age	Age + year + age $\times$ year	Year	Year	42	12.10	< 0.01	1447.76

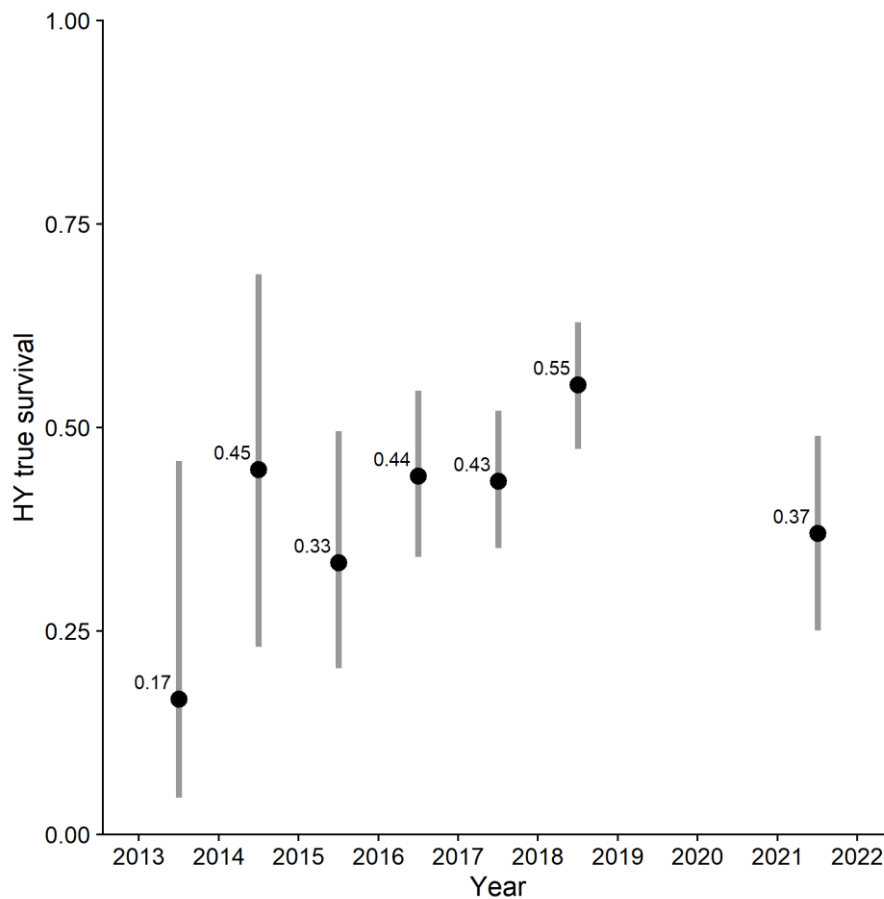


Figure 23. True survival (S) of hatch-year piping plovers on Fire Island, NY from 2013 – 2021. Hatch-year survival is the survival from hatch to the following breeding season (i.e., individuals < 1 year post-hatch). Estimates are from the top model with an interaction of year and age. Error bars represent 95% confidence intervals.

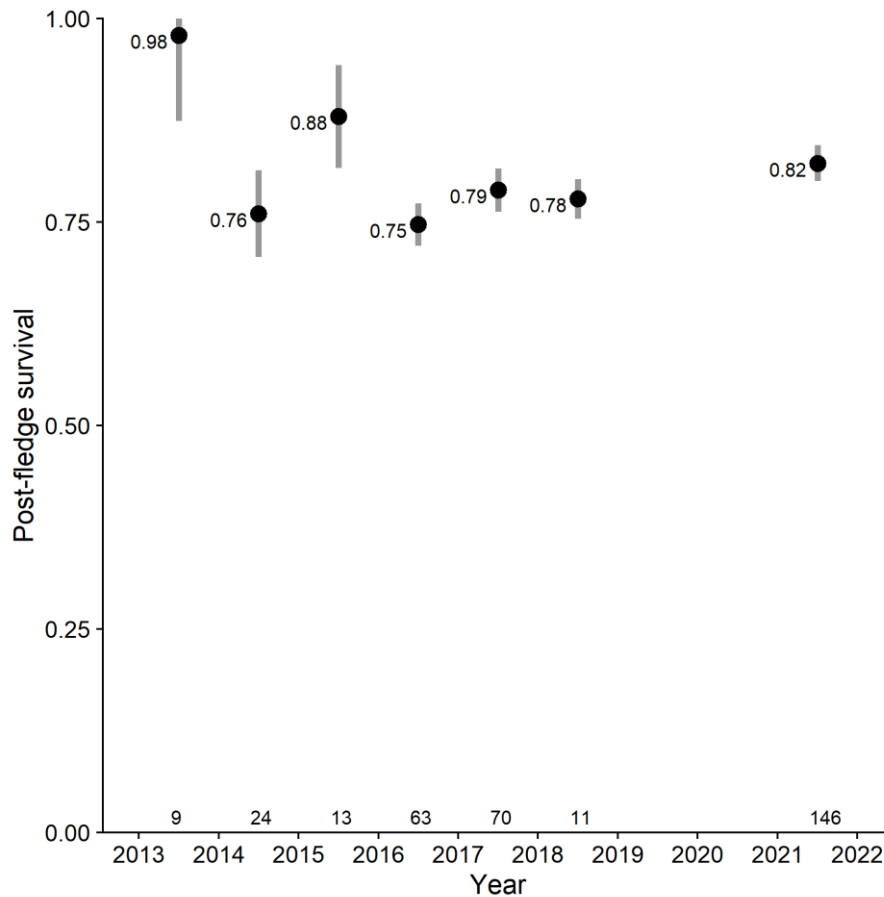


Figure 24. Post-fledge survival of piping plovers on Fire Island, NY from 2013 – 2021. Post-fledge survival is the survival from fledgling to returning the following breeding season, and is calculated by dividing hatch-year survival by pre-fledge chick survival (Figure 20). Error bars represent 95% confidence intervals. Numbers above x-axis represent number of fledglings considered in analysis.

In 2018, six mortalities of banded adult piping plovers were confirmed either by discovering carcasses of adults ($n = 5$) or by procuring bands or flags from predator scat samples collected during field surveys ($n = 1$). All mortalities were identified as predation. At least one mortality is believed to have been caused by peregrine falcon (*Falco peregrinus*)⁴ and one mortality was later confirmed to be caused by a feral cat (*Felis catus*) via mitochondrial DNA sequencing of the scat sample we collected. In 2019, we confirmed two mortalities of banded adults within the study area. The flag leg of one adult was found outside of an enclosure and this bird likely was depredated by a peregrine falcon based on field sign: we discovered evidence that at least one peregrine falcon was frequenting the area and using enclosures as perches, as several enclosures had bones, feathers, and entrails in the vicinity. The carcass of another banded adult was found in vehicle tire tracks on the driving portion of open beach in NBW. The cause of mortality was not determined, but following its discovery, a nest within 250 m was abandoned. Unfortunately, the adult was not confirmed to the nest so we cannot be certain that the two incidents are related. In 2022, we recovered a deceased banded adult, suspected to have

⁴ Incident observation report filed 11 June 2018. VTSP researchers learned from Suffolk County stewards that a peregrine falcon was briefly entangled in the netting of a nest enclosure, and a deceased plover was inside the enclosure.



been the result of gulls which were seen harassing the nest enclosure⁵. We also observed an instance of adult mortality as evidenced from a recovered plover wing, suspected to have been depredated by a cat (tracks nearby, sign of cats at nests in the area; Table S2 – Table S4).

Discussion

Backshore and barrier interior habitats that were present before Hurricane Sandy made up the greatest proportion of available plover nesting habitat, yet birds nested in pre-Hurricane Sandy areas at lower proportions when compared to the habitat created as a result of Hurricane Sandy. When combined, returning adult residents, natal recruits, and immigrants nested in habitats created since Hurricane Sandy 55% of the time and only nested in other habitat not modified by Hurricane Sandy 45% of the time, despite the fact that other habitat comprised 42 – 61% of the available nesting habitat.

All three population constituents (i.e., returning adult residents, natal recruits, and immigrants) of nesting piping plovers avoided other habitats or selected for the habitats created as a result of Hurricane Sandy, relative to what was available. Many of these areas were overwashed, or were created to appear like overwash and, thus, had wide backshores with open sand nesting habitat. Open sandy habitat, like that created by Hurricane Sandy, can assist nesting plovers in detecting predators and can play a role in the camouflage of their eggs and chicks (Fraser and Catlin 2019). Conversely, habitat fragmentation and being closer to dense vegetation may make the pre-Sandy habitats less attractive to nesting birds. These pre-Sandy areas also have less distance between the high tide line and the toe of the dune, which makes nests more susceptible to flooding compared to all of the post-Hurricane Sandy habitats. Although for this analysis, we considered every patch of dry sand > 1 m² available, some of this habitat may be unsuitable to nesting birds, however much of the unsuitable habitat is likely to be the pre-Hurricane Sandy other habitat (Walker et al. 2019).

Immigration is an important demographic component of population growth for plovers along the Atlantic Coast, and likely sustains the small and southernmost population on the Outer Banks of North Carolina (Weithman 2019, Weithman et al. 2019, Robinson et al. 2020a). On Fire Island and Westhampton Island, immigration contributed an average 40.75% (SE: ± 3.82%) annually to the overall 346% population increase that occurred between Hurricane Sandy and 2022. In 2016, we observed the highest immigration rate of all years examined, although the count of immigrants may be a slight over-estimate due to a small number of birds that did not get banded in the year prior due to early nest failure or trap aversion. This relatively high immigration rate may have been related to low nest and chick survival in 2015, causing adults to not return to nest on Fire Island and thereby increasing available habitat for new nesting birds, especially as 2016 had the lowest proportion of returning nesters and natal recruits in the population. In addition, the Great Gun restoration was completed prior to the 2016 breeding season, which further enhanced nesting habitat for piping plovers. However, the proportion of the population that was composed of immigrants, and the proportion of immigrants in new habitats has declined in recent years. This may suggest that the Fire Island and Westhampton Island piping plover population is slowly nearing the carrying capacity of its habitat, resulting in less available space for new birds.

Since 2020, immigrants generally selected storm-created and human-created habitats in proportion to their availability. Increasing nesting density likely decreases the unoccupied nesting habitat available to immigrants,

⁵ Incident observation report made on 16 June 2022 by VTSP researchers working on NPS property. A mixed flock of 12 herring and great black-backed gulls were harassing a nest enclosure, and a banded piping plover associated with the nest was found dead outside the enclosure.



and therefore it is possible that some habitat types (e.g., overwashes) have become too densely populated for immigrant birds to obtain territories while returning adults were remaining faithful to their nest sites. Similarly, on the Missouri River, immigration declined as nesting density increased (Catlin et al. 2015). Immigrants only exhibited selection for a specific habitat in 2020 and 2022 on Fire Island, in which they selected for restoration areas. The only restoration area used by plovers in these years was the Great Gun restoration area. Vegetation growth caused dry sand fragmentation within the New Made restoration area, which may deter nesting plovers through reduced habitat availability and increased cover for terrestrial predators. In all years prior to 2020, immigrants used habitat in proportion to what was available, therefore a good way to increase and maintain immigration into the population is to maintain open, wide, sparsely vegetated sand. However, selection patterns of immigrants in future years will be paramount in considering how immigrants select for habitats in a stabilizing population with locally high nesting densities.

Natal recruits, or birds that hatched within the study area and returned the following year, nested in each habitat, every year, almost as much as we would expect based on area of habitat. However, in 2020, natal recruits selected for breach fills and in 2020 and 2022, selected for natural overwashes more often than expected based on availability. One pair of second-year birds nested in Narrow Bay West, and seven nested at Old Inlet East, a surprising finding given that in 2019, natal recruits selected against natural overwashes. Natal recruitment tended to be variable and higher following years of higher chick survival. Fledglings prospect for potential nesting sites prior to departing for the wintering grounds (Davis et al. 2017), thus habitat selection among natal recruits could have been driven by observations of high reproductive success and conspecific adults selecting for post-Hurricane Sandy habitats during their first year. Similar to what we have seen on Fire Island, across avian taxa, the contribution of immigration to population growth generally is higher than the contribution of natal recruitment (Millon et al. 2019).

Piping plover adult survival was constant among years, which is generally consistent with many avian populations (Sæther et al. 2002, Lampila et al. 2016). Sufficient adult survival, which is usually found to have high sensitivity relative to population change (Calvert et al. 2006), is unlikely to be allowing the near doubling of the population in the Fire Island and Westhampton Island study area. Hatch-year survival generally tracks the trends for pre-fledging chick survival, suggesting that variation in pre-fledging chick survival is the major component of this parameter. For this report, we have used both a different model and a differing model structure from previous years to estimate chick survival, and thus derived estimates of post-fledgling survival vary from previous reports (Hermanns et al. 2018, Robertson et al. 2019). Deviations from previous estimates of post-fledgling survival are most noticeable for 2013, a year with low sample size which may result in overestimation of chick survival. We also changed this model such that we use a 5-day encounter window, and use the number of eggs that hatched rather than the number of chicks ever observed as the first encounter. Only using the chicks observed certainly contributed to the estimate of high chick survival in the past.

A population that experienced exponential growth, such as Fire Island, is likely to have very different dynamics than a stationary population (Schaub et al. 2013, Millon et al. 2019). Following Hurricane Sandy, immigration was the primary correlate with population growth, although reproductive output is positively associated with population change (Robinson et al. 2020a). While demographic estimates were not possible given the constraints of our study in 2020 and 2021, future monitoring of this population is needed to understand the variation in the contribution to population change of each demographic parameter as vegetation increases, suitable habitat declines, and population growth slows.



Chapter 4. Piping Plover prey densities at Fire Island

Objective P4: Compare prey densities at several habitat types

Introduction

Understanding how hurricanes and other environmental perturbations alter coastal food webs is an essential part of predicting and, where appropriate, mitigating the effects of these disturbances on threatened and endangered species. Barrier islands, as narrow, dry sand islands that lay parallel and close to the mainland are particularly susceptible to environmental changes. Following an environmental event to a barrier island, such as a hurricane, modifications such as dune building often are conducted to protect the mainland from flooding in future events or to improve the beach for recreational purposes after hurricane-related changes. The post-hurricane natural and engineered changes affect local wildlife in both direct (e.g., loss or creation of nesting habitat space) and indirect (e.g., changes in food availability and distribution) ways.

Storms disturb and rearrange shoreline sediments, altering habitat for benthic, marine, and terrestrial invertebrate communities. For example, breaches and overwashes, such as those resulting from Hurricane Sandy, open up new areas for invertebrate colonization, providing excellent foraging opportunities for many invertivorous shorebirds (Cohen et al. 2009a, Walker et al. 2019). Less beneficial changes to elevations, shoreline shape, and sediment composition may result in diminished shorebird foraging opportunities. Similarly, post-storm beach nourishment efforts and other shoreline engineering projects have the potential to reduce or improve foraging habitat quality (Speybroeck et al. 2006, Leewis et al. 2012). With many shorebird populations in decline, it is important to assess and, when appropriate, mitigate the effects of natural and anthropogenic environmental modifications on their invertebrate prey base.

Piping plovers forage on terrestrial and marine invertebrates along sandy shorelines. Prey abundance and location contribute to nest site selection on the breeding grounds, as plovers favor nesting close to moist substrates for availability to food sources during the nesting period (Elias et al. 2000, Cohen et al. 2009a) while natal territories with available high-quality foraging habitat can contribute fitness advantages to offspring (Catlin et al. 2014). Specifically, plover chicks with access to high densities of terrestrial invertebrates have higher foraging rates, weigh more, and are more likely to survive (Loeagering and Fraser 1995, Cohen et al. 2009a, Catlin et al. 2015).

Piping plovers display diverse foraging methods, consuming terrestrial and marine invertebrates by pecking for prey on the surface of sediment, probing shallowly into the substrate, and occasionally by gleaning invertebrates on vegetation (Shaffer and Laporte 1994, Cuthbert et al. 1999, Majka and Shaffer 2008, Cisek 2013). The variety of foraging methods piping plovers exhibit allows them to prey on a diverse range of taxa. Adult piping plovers prefer to consume benthic invertebrate prey in the pre-laying season (Cohen and Fraser 2010), while young chicks rely primarily on terrestrial arthropods to support their rapid growth (Cuthbert et al. 1999). These specialized developmental foraging preferences suggest that foraging habitat quality, the quantity, diversity, and accessibility of suitable prey, is an important element of plover nest site selection (Elias et al. 2000, Cohen et al. 2009a). Therefore, as shorelines are transformed, through natural forces and human engineering, it is likely that the subsequent changes in invertebrate prey composition and abundance will have a considerably impact on the resulting demographic response of piping plover populations (Cohen et al. 2009a, Catlin et al. 2015).

After Hurricane Sandy, the FIMI stabilization project transformed parts of Fire Island by moving sand to fill breached areas and stabilize the shoreline. We hope that an understanding of how these recent landscape changes affected the Fire Island invertebrate prey communities can inform our interpretation of post-Hurricane Sandy plover population dynamics and inform strategies on future habitat creation or modification.



SHOREBIRDS

Methods

Data processing

In some instances, invertebrate sampling was stopped before or after the target 30-min interval (e.g., sudden inclement weather). We reduced our sample size to include only samples that were within 5 min of our target sampling interval (i.e., invertebrate traps that operated for 25 – 35 minutes). Additionally, we excluded samples where coordinates were not collected on a GPS unit or phone, as we used coordinates to confirm whether sampling locations fell within USACE polygons. The taxonomic Order Thysanoptera (thrips) were excluded from our calculation of invertebrates captured, as they are likely too small to be plover prey. For analyses, we summed the total number of invertebrates captured on the horizontal and vertical trap sticks.

Statistical analysis

To assess the difference in the mean number of terrestrial invertebrates collected across sampled habitats, we performed a Kruskal-Wallis chi-squared test (Kruskal and Wallis 1952). Because we had relatively few samples in ephemeral pools, we limited our Kruskal-Wallis tests to compare invertebrate captures from bay intertidal areas, ocean intertidal areas, and the island interior using the invertebrate samples from 2014 – 2019 and 2022 alone.

We then tested for the effect of weather and habitat variables on invertebrate abundance observed from 2014 – 2022. We fit a negative binomial mixed-effects model with log link (Hilbe 2007, Bolker et al. 2009) via the R package *glmmTMB* (Brooks et al. 2017) to predict invertebrate catch. We included scaled temperature, scaled windspeed, standardized date of sampling (relative to 1 April of that year) and its quadratic term, the hour of sampling and its quadratic term, habitat (bay intertidal, ephemeral pool, ocean intertidal, and island interior), and three treatments (overwash, breach fill, restoration area) as fixed effects. Habitat type was assigned in the field based on sampling location, whereas treatment was assigned using USACE polygons. Additionally, we included year and subsite as crossed random effects. All covariates were included as additive effects in our maximal model, and we did not consider interactions. After examining model diagnostics and confirming the maximal model could meet all standard model assumptions, we performed stepwise backward elimination model selection using a likelihood ratio test, removing covariates with low explanatory power ($\alpha = 0.05$) to arrive at the most parsimonious model (Dunkler et al. 2014). We then used the resulting model to predict mean invertebrate abundance for each habitat and treatment. We extrapolated results to 3-hr periods to be comparable to past studies (Houghton 2005, Cohen et al. 2009a, Anteau and Sherfy 2010).

All statistical analyses were conducted in R, ver. 4.1.3 (R Core Team 2021).

Results

We collected 2,905 invertebrate samples from 2014 – 2022 (excluding 2020 and 2021, of which 2,608 (89.8%) were used in analysis. Not all samples considered in analysis contained invertebrates; in total, 1,327 (50.9%) of samples used in analysis contained at least one invertebrate. The number of invertebrates captured between 2014 – 2019 differed between the three habitats (bay intertidal, island interior, and ocean intertidal; K-W = 203.27, df = 2, $P < 0.001$; Table 17a). Across the whole study, bay intertidal samples had 2 – 4× as many invertebrates captured during the sampling period. However, in 2022, we did not detect a significant difference in the number of invertebrates captured between habitats (K-W = 2.38, df = 2, $P = 0.31$; Table 17b).



Table 17. Mean invertebrates captured during a 30-min sampling interval at bay intertidal, ephemeral pools, ocean intertidal, and island interior habitats on **(a.)** Fire Island and West Hampton Island, May – August 2014 – 2019, and **(b.)** on Fire Island May – August of 2022.

Island	Habitat	(a.) 2014 – 2019			(b.) 2022 only		
		Mean	SE	<i>n</i>	Mean	SE	<i>n</i>
Fire Island	Bay intertidal	4.27	0.41	412	4.25	1.87	32
	Ephemeral pool	6.46	1.89	72	29.5	25.5	2
	Ocean intertidal	1.14	0.09	1,062	1.73	0.45	73
	Island interior	1.52	0.49	431	3.43	1.18	174
Westhampton Island	Bay intertidal	1.49	0.38	55	-	-	-
	Ephemeral pool	17.70	3.28	3	-	-	-
	Ocean intertidal	0.48	0.06	187	-	-	-
	Island interior	2.24	0.32	105	-	-	-

In our analysis of weather and habitat variables on invertebrate catch, temperature and the quadratic hour did not have a significant effect on invertebrate catch. Among habitat types, ephemeral pools and bay intertidal areas had greater projected invertebrate catches than the ocean intertidal or island interior areas (Figure 25). When considering treatment types, we found significant differences though there was no consistent trend within treatment types (Figure 25). Although the invertebrate catch rate was high in ephemeral pools, ephemeral pools were rare, as indicated by the small sample size (Figure 25).

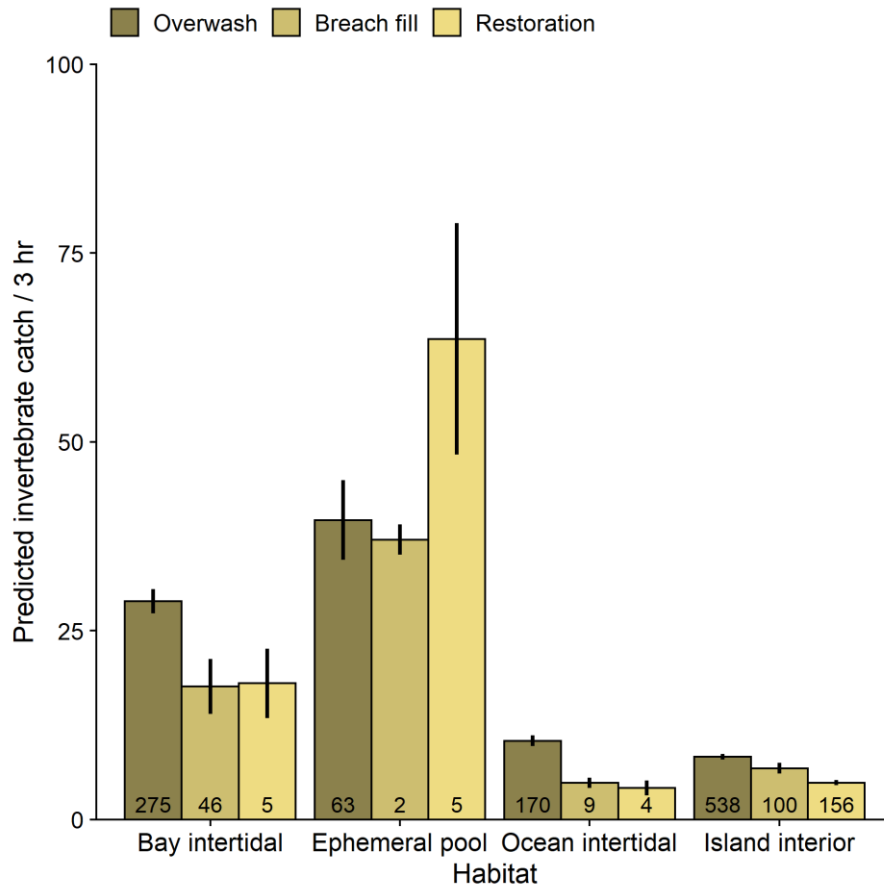


Figure 25. Predicted invertebrates catch over a projected 3-hr period in various habitats from 2014 – 2022. We predicted invertebrate catch using a negative binomial mixed-effects model. Invertebrate samples were collected over 30-min sampling intervals and projected to 3-hr periods to facilitate comparison to similar piping plover studies. Numbers within bars indicate sample sizes. Error bars represent 95% confidence intervals. Sample sizes differed among habitats, as samples were collected in proportion to habitat availability.

Discussion

To assess overall plover habitat quality, it is essential to understand the abundance, distribution, and composition of their invertebrate prey base. From terrestrial invertebrate samples on Fire Island and Cupsogue during 2014 – 2019 and on Fire Island in 2022, our results indicate what past studies also have indicated: that low-energy moist habitats, such as bay intertidal habitat and ephemeral pools have greater invertebrate abundance relative to interior and ocean intertidal habitats (Loefering and Fraser 1995, Elias et al. 2000). Contrary to some studies, we found that the ocean intertidal habitat was generally more productive than interior habitats (Loefering and Fraser 1995), but invertebrate availability in the intertidal likely varies with slope, elevation, and wrack availability, which we did not evaluate here.

Though it might be expected that disturbances, such as overwashes, would negatively impact the invertebrate community (McCabe and Gotelli 2000), we found that more terrestrial invertebrates were collected in overwashes compared to breach fills and restoration areas. Adult terrestrial invertebrates, often insects, are frequently capable of flight, allowing them to find and recolonize disturbed or overwashed areas more quickly



than benthic invertebrate taxa that spend their full life cycle in the substrate. Our collection methods are designed for the collection of terrestrial invertebrates, a primary source of food for chicks (Cuthbert et al. 1999, Elliott-Smith and Haig 2004), and are not optimal for the collection of benthic or aquatic invertebrates which are more often foraged by adult plovers (Cohen and Fraser 2010). Future studies may benefit from expanding invertebrate collection methods to target intertidal benthic invertebrates to better understand how overwashes affect prey availability for adult plovers.

Lower invertebrate abundance in restored habitat may be partly because intertidal areas were not included in restoration area design polygons, and we only included areas within the design footprint in our evaluation. Future evaluations of the restoration areas should also consider sampling adjacent, accessible intertidal zones even if not explicitly in the design criteria.

Overall, understanding which habitats support the greatest invertebrate communities is essential, as greater invertebrate abundance and densities are likely to improve foraging efficiency of both adult plovers and chicks, leading to improved growth rates and potentially higher survival rates (Catlin et al. 2014). Ideally, future habitat construction can improve on the creation of ephemeral pools in design criteria and perhaps mimic overwash dynamics, and allow access to intertidal zones to improve invertebrate abundance for plovers.

Chapter 5. Assessing the role of nest site viewshed on nest site selection⁶

Objective P1: Compare changes in numbers, density, and distribution of piping plovers in various habitats on Fire Island

Introduction

Nest site selection in piping plovers is reflective of environmental characteristics at two geospatial scales. At the landscape level, distance to intertidal zone (Maslo et al. 2011, Walker et al. 2019), distance to dunes (Burger 1987, Maslo et al. 2011, Boyne et al. 2014), distance to vegetation (Burger 1987), flat locations (Burger 1987), and beach width (Walker et al. 2019) are factors that have been shown as influential in plover nest site selection. At the microhabitat level, plovers have selected for substrate comprised of shell cover and gravel (Prindiville Gaines and Ryan 1988, Espie et al. 1996, Cohen et al. 2008, Maslo et al. 2011, Grant et al. 2019) and little to no vegetation coverage (Espie et al. 1996, Cohen 2005, Cohen et al. 2008, Maslo et al. 2011). Because of this, biologists and managers regularly describe plover nesting habitat as open and sparsely vegetated (Wilcox 1959, Elliott-Smith and Haig 2004). What has yet to be determined is the functional reason for selecting nest sites in areas of sparse vegetation.

Nesting in open areas may be a strategy that increases an incubating plover's ability to visually detect predators (Burger 1987), allowing for quick initiation of distraction behaviors to protect nest and chicks (Wilcox 1959, Cairns 1982, Elliott-Smith and Haig 2004). Known predators across the piping plover life cycle include northern raccoons (*Procyon lotor*), red foxes (*Vulpes vulpes*), feral cats (*Felis catus*), peregrine falcons (*Falco peregrinus*), and herring gulls (*Larus argentatus*). The risk of predation for a nest by terrestrial predators may vary with the size of the predator and distance to landscape features that could either conceal the predator from the incubating plover or provide protection to the nest from predators (Espie et al. 1996, Swaisgood et al. 2018).

Viewshed is the area visible from a specified place (Aben et al. 2018). Two coastal landscape elements that can affect the viewshed of an incubating plover are topography and vegetation. Nesting at the bottom of dunes and in close proximity to dense vegetation stands, such as those formed by American beachgrass (*Ammophila breviligulata*) or common reed (*Phragmites australis*), will reduce an incubating plover's viewshed. Although viewshed area is predicted to be an important component of habitat selection in animals (Aben et al. 2018), it has not yet been formally tested as a factor in nest site selection in ground-nesting species, perhaps due to the difficulty in quantifying it (Burger 1987, Swaisgood et al. 2018), often requiring on the ground equipment and tedious measurements (Yanes and Oñate 1996, Gómez-Serrano and López-López 2014).

Recent advances in remote sensing technology have enabled novel approaches to quantifying viewsheds. Here, we modernized prior methods to quantify the viewshed from a piping plover nest and evaluated this factor as a predictor of piping plover nest site selection. Our objectives were to: (1) characterize the visible area at nest sites in the study area between 2015 and 2020, (2) evaluate if nest sites possessed greater surrounding visible area than random points on the landscape by performing a use-versus-availability analysis (Johnson 1980), and (3) assess how adding viewshed as a covariate contributes to a previously created nest site selection model for the population of piping plovers in Fire Island, NY (Walker et al. 2019).

⁶ Forthcoming graduate thesis chapter in "Piping plover (*Charadrius melodus*) nest site selection following landscape and predator community changes." **Cite chapter as:** Dorsey, S. S., S. Ritter, C. N. Wails, S. G. Robinson, K. Walker, D. H. Catlin, S. M. Karpanty, and J. D. Fraser. 2022. Assess the role of nest site viewshed on nest site selection. Pp. 67 – 74. In: *Effectiveness monitoring and targeted studies in support of FIMI conservation measures* (eds. Wails et al.). Annual Report prepared by the Virginia Tech Shorebird Project. Virginia Tech, Blacksburg, VA.



Methods

Geospatial Analysis

To quantify the viewshed, we used two layers: classified imagery (Appendix 1. Supplemental methods) and a digital elevation model derived from LiDAR data. Using the classified imagery, vegetation pixels were set to a specified height annually, ranging from 15 – 46 cm based on field measurements performed in 2016, 2017 (Bellman 2018), and 2022. The ground and vegetation heights were added together using the ArcGIS ‘Plus’ tool to combine both factors that obstruct visibility in the landscape. We set the offset from the ground at 6 cm, based on an incubating plover’s height of eye. To generate the viewshed for a nest, we restricted the area to a 30-m buffer from the center point, which we refer to as the “nest site.” This distance was determined based on the range of observed flush distances in plovers when a threat is detected (Cairns 1982, Burger 1987, Loegering 1992, Jorgensen and Bomberger Brown 2016). We used the ArcGIS Pro ‘Viewshed’ tool to generate an output of either visible or not visible for each pixel within the 30-m buffer (Figure 26).

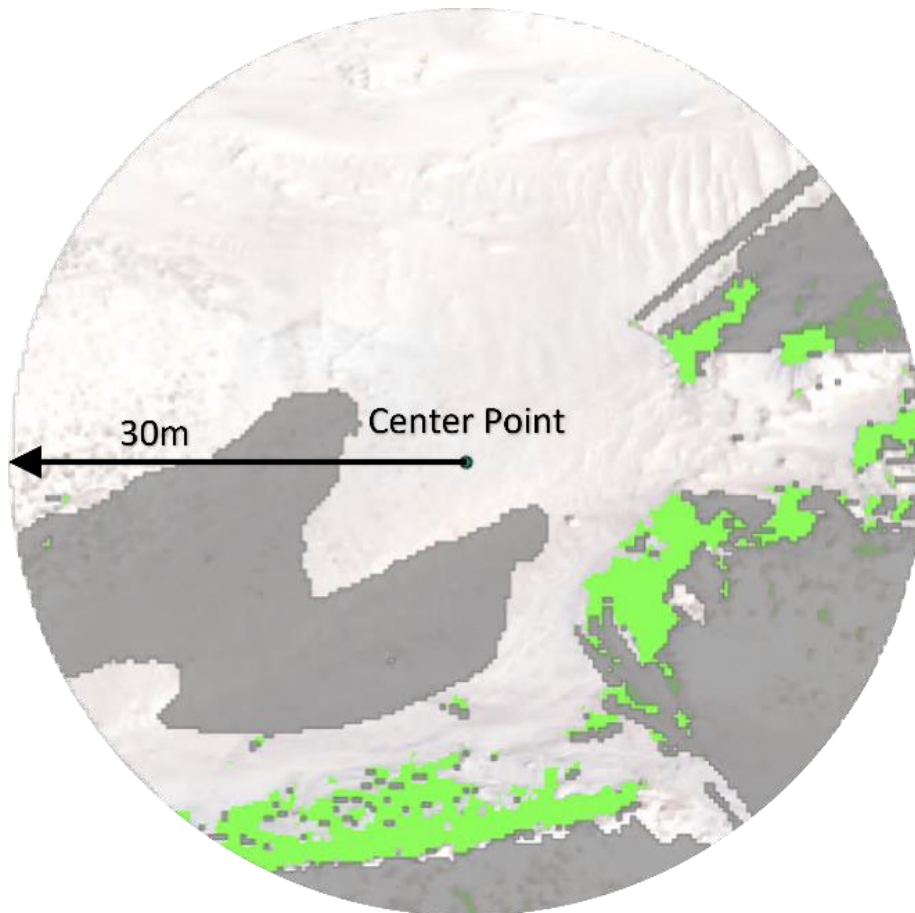


Figure 26. Example ArcGIS Pro output of viewshed analysis within 30 m of the center point showing areas not visible due to vegetation height or elevation changes in gray and vegetation coverage in green.

Statistical Analysis

We modeled nest site selection using logistic regression (Hosmer and Lemeshow 2000). Nest locations recorded across the study area from 2015 – 2020 were encoded as “used” locations. For each year, we then randomly selected points within pixels classified as dry sand to represent “available” locations.



We first tested whether viewshed influenced nest site selection in plovers. We included year, viewshed (% of area in buffer visible to an incubating plover), and their interaction in a generalized linear model with a binomial error distribution and logit link. We tested for significance of model covariates using a Wald test (McCullagh and Nelder 1989), and performed post-hoc analyses of the interaction using least-squares means (Lenth 2016).

We then examined if viewshed further explained nest site selection in plovers relative to variables previously demonstrated to influence nest site selection (Walker et al. 2019). We constructed a generalized linear mixed-effects model via the R package *lme4* (Bates et al. 2015) that included least-cost distance to bay (m), least-cost distance to ocean (m), and Euclidean distance to bay (m; Walker et al. 2019) with a binomial error distribution and logit link. Least-cost distances were calculated as the shortest distance a plover could walk from a point to the bay without passing through vegetation or water. We included year as a random effect. We compared Walker et al.'s model to the same model including viewshed using a likelihood ratio test (Lewis et al. 2011).

All analyses were conducted in R ver. 4.1.3 (R Core Team 2021).

Results

We assessed 353 nests across the study area from 2015 – 2020. Nest sites, on average, had a percentage of visible area consistently above 60% (Figure 27, Table 18). Visible area declined over time (2015: $\bar{X}$ = 84.6% vs 2020: $\bar{X}$ = 60.0%, t = 5.52, P < 0.001). The range in nest site visible area was greatest in 2020 (0.7% – 100.0%) and the next largest range in values was in 2018 (4.5% – 100.0%).

Table 18. Annual piping plover nest counts and nest site visible area (%) mean, median, and range values between 2015 – 2020. Nest numbers include renest attempts of some pairs.

Year	Nests (<i>n</i>)	Mean visible area (%)	Median visible area (%)	Minimum visible area (%)	Maximum visible area (%)
2015	36	84.6	92.0	17.4	100.0
2016	37	73.9	83.8	24.9	100.0
2017	51	64.7	64.9	9.1	97.8
2018	63	68.2	69.8	4.5	100.0
2019	92	64.5	65.4	9.8	100.0
2020	74	60.0	62.5	0.7	100.0
All years	353	67.3	69.8	0.7	100.0

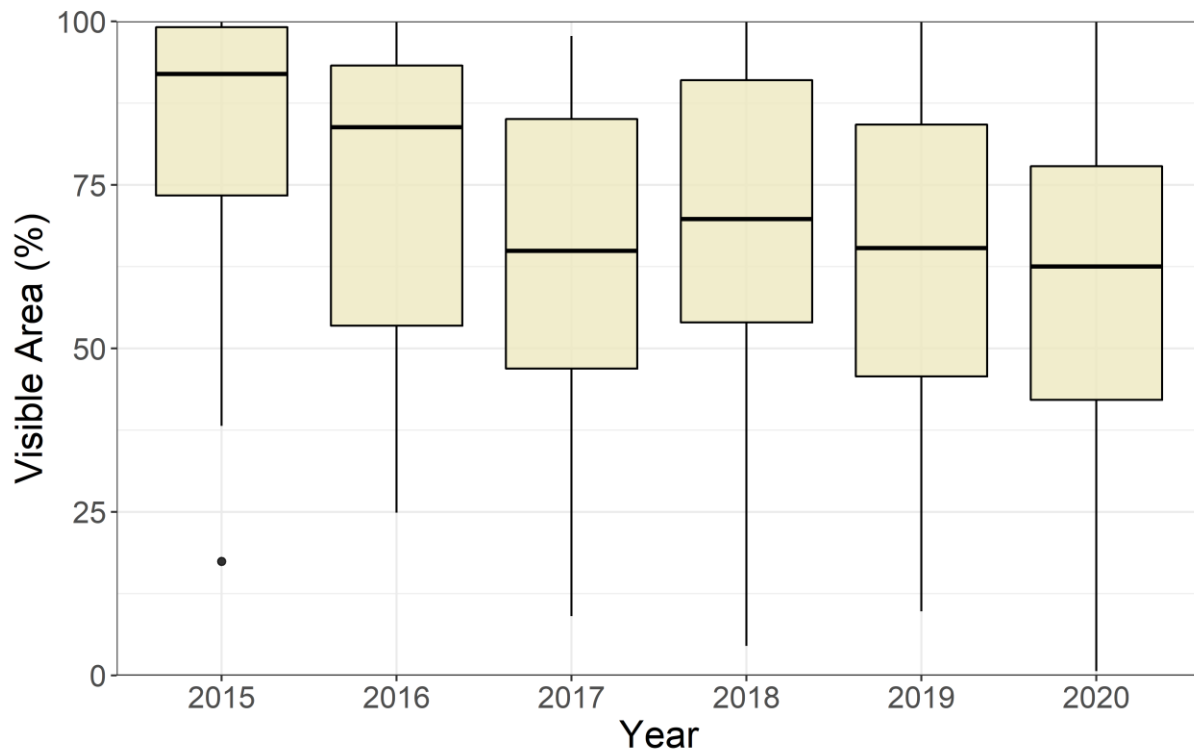


Figure 27. Annual visible area (%) within nest sites on Fire Island study area between 2015 – 2020. Boxplots in this figure illustrate the median as the central horizontal line within the box and the first and third quartile as the extents of the colored box. The maximum and minimum are illustrated by lines extending out from the interquartile range.

Nest site visible area was variable across the three management units (Figure 28, Table 19) with median values ranging from 54.7% – 96.7%. Within the Fire Island National Seashore, median visible area within nest sites showed little variation between roughly 70.0 – 100.0% in 2015 and 2016, but drastically increase to roughly 9.0 – 100.0% from 2017 – 2020. All median values remained above 60% in this management unit across the five-year span. The nests at Robert Moses State Park showed a relatively stable amount of nest site visible area from 2015 – 2019 until a sudden decrease in 2020. Similar to Fire Island National Seashore, nest site visible area values in Robert Moses State Park also had the smallest range in 2015. Nests in Smith Point County Park had the greatest visible area in 2015 (median = 96.7%, range = 60.7% – 100.0%), though nest site visible area decreased dramatically after that when median values fell between 54.7% – 67.0%.

Table 19. Annual nest site visible area (%) median and ranges for each management unit between 2015 – 2020. Nests located within the Fire Island National Seashore Lighthouse Tract were so few that they were included with the Robert Moses State Park nests.

	Fire Island National Seashore		Robert Moses State Park		Smith Point County Park	
Year	Median visible area (%)	Range	Median visible area (%)	Range	Median visible area (%)	Range
2015	94.6	79.0 – 99.0	76.2	17.4 – 96.9	96.7	60.7 – 100.0
2016	92.3	69.4 – 98.9	87.1	24.9 – 97.5	59.4	30.0 – 100.0
2017	70.3	27.1 – 96.0	76.9	9.1 – 97.2	58.0	28.4 – 97.8
2018	80.0	38.4 – 99.7	80.1	31.0 – 99.8	67.0	4.5 – 100
2019	62.6	9.8 – 99.7	82.3	20.5 – 100.0	54.7	12.9 – 92.3
2020	67.5	8.9 – 100.0	56.5	0.7 – 99.7	66.2	17.6 – 99.9
All years	72.6	8.9 – 100.0	71.7	0.7 – 100	66.2	4.5 – 100.0

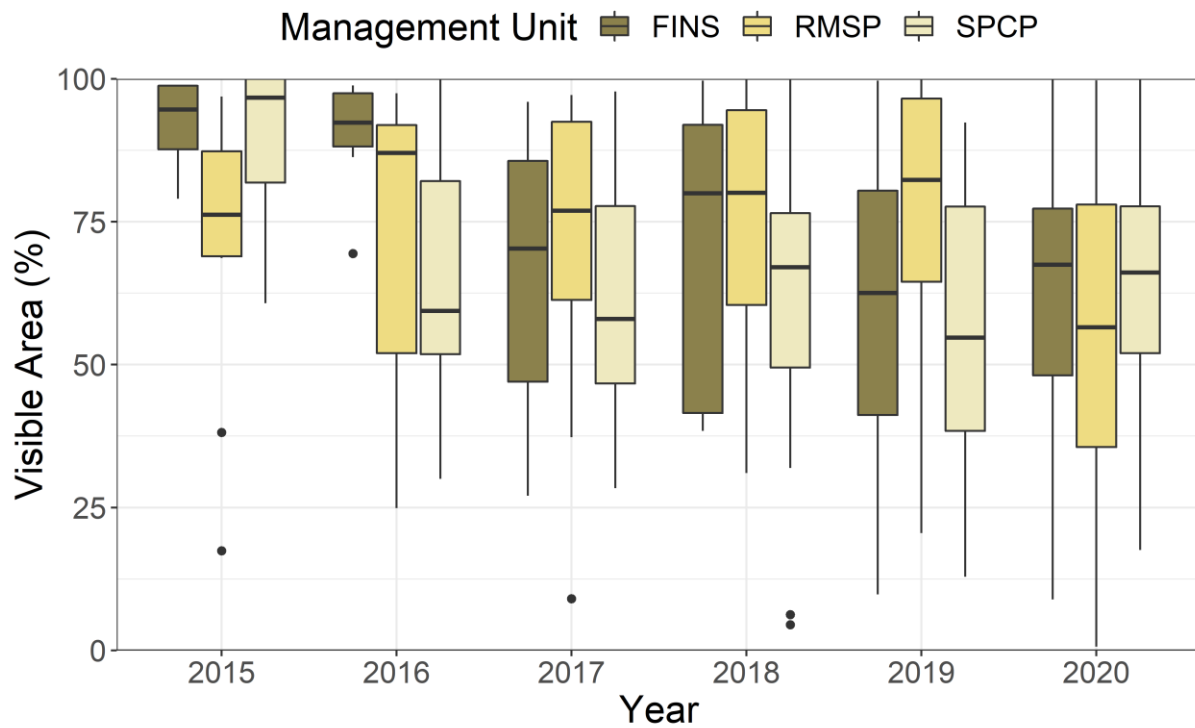


Figure 28. Visible area (%) within nest sites on Fire Island study area by management unit (Fire Island National Seashore, Robert Moses State Park, and Smith Point County Park). Nests located within the Fire Island National Seashore Lighthouse Tract were so few that they were included with the Robert Moses State Park nests. Each boxplot color represents a different year between 2015 – 2020. Boxplots in this figure illustrate the median as the central horizontal line within the box and the first and third quartile as the extents of the colored box. The maximum and minimum are illustrated by lines extending out from the interquartile range.

We modeled plover nesting habitat using 353 nests and 1,765 random points from 2015 – 2020. Piping plovers exhibited selection for nest sites with greater visible area relative to the amount of visible area at random points (logistic regression, $\beta = 1.50$, $P < 0.001$). There also was significant change in nest site visible area relative to the year (Table 20), such that nest site visible area weakens in its predictive ability in the years after 2015 (Table 21). This trend is demonstrated by the increasing overlap between nest site and random points visible area values in the years after 2015 (Figure 29).

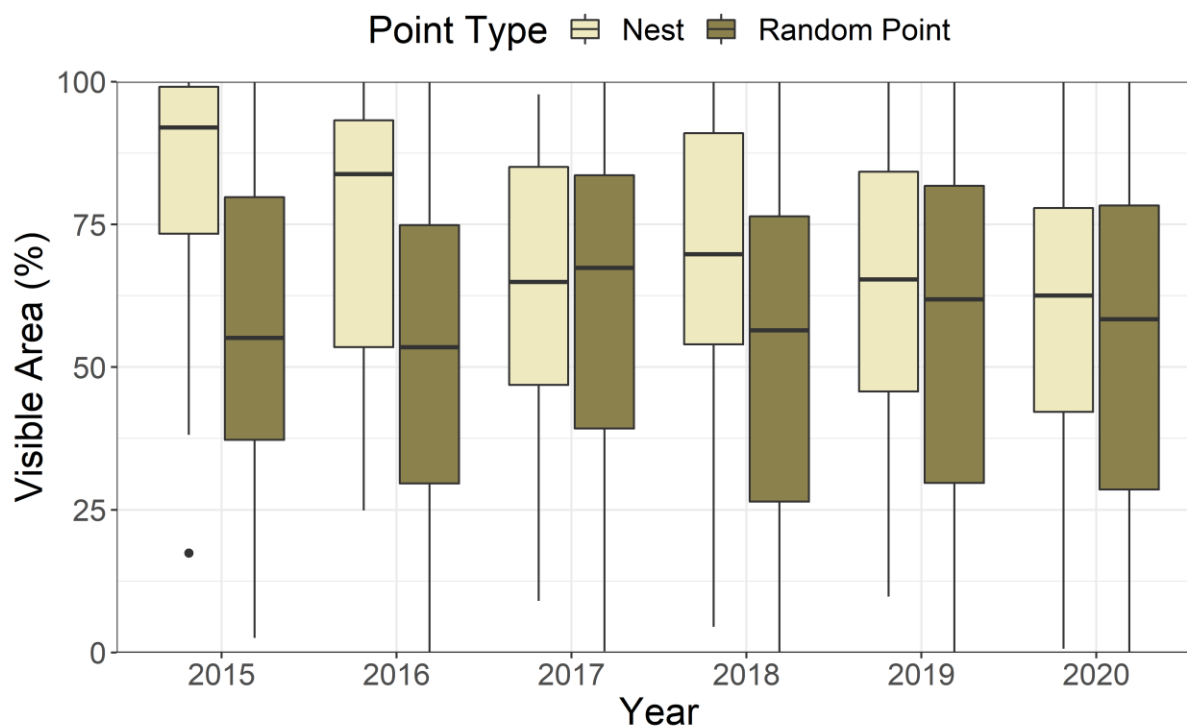


Figure 29. Annual visible area (%) within nest sites and random points from 2015 – 2020. Boxplots in this figure illustrate the median as the central horizontal line within the box and the first and third quantile as the extents of the colored box. The maximum and minimum are illustrated by lines extending out from the interquartile range.

Table 20. Variables explaining nest site selection and their significance from logistic regression. ‘x’ indicates an interaction.

Predictor	Wald’s χ^2	df	P
Year	4.73	5	0.45
Nest site visible area	51.49	1	< 0.001
Year x nest site visible area	30.37	5	< 0.001

Table 21. Post-hoc least squares means tests, illustrating marginalized means probability of nest site selection and standard errors (SE) for a given year at the median nest site visible area (Table 18). Logit-transformed confidence intervals that do not overlap with zero indicate a significant effect (denoted with *).

Year	Marginalized mean probability of nest site selection	SE	Lower 95% confidence interval	Upper 95% confidence interval	
2015 × Nest site visible area	81.78%	0.04	0.916	2.088	*
2016 × Nest site visible area	70.68%	0.04	0.439	1.321	*
2017 × Nest site visible area	52.62%	0.04	-0.222	0.432	
2018 × Nest site visible area	63.99%	0.03	0.276	0.874	*
2019 × Nest site visible area	56.90%	0.03	0.047	0.509	*
2020 × Nest site visible area	55.70%	0.03	-0.027	0.485	

A model including visible area added to the model from Walker et al. (2019) had a significantly stronger predictive ability than the Walker et al. model alone (likelihood ratio test: $\chi^2 = 42.06$, $P < 0.001$; Table 22).

Table 22. Likelihood ratio test comparing the generalized linear mixed-effects regressions tested in this analysis and the significant model variables from Walker et al. (2019). The fixed effects included least-cost distance to bay intertidal line (m), least-cost distance to ocean intertidal line (m), Euclidean distance to bay intertidal line (m), and visible area (%) within nest site. The random effect is year. The number of covariates in the model is represented by the symbol k .

Mixed-effect logistic regression	k	χ^2
Least-cost dist. to bay + least-cost dist. to ocean + Euclidean dist. to bay + % visible area	6	42.06
Least-cost dist. to bay + least-cost dist. to ocean + Euclidean dist. to bay	5	($P < 0.001$)

Discussion

Piping plovers selected nest sites that provided greater visibility from the nest than would be expected at random, consistent with past results that show plovers favor open areas (e.g., Wilcox 1959, Espie et al. 1996, Cohen 2005, Cohen et al. 2008, Maslo et al. 2011). This investigating into the role of nest site viewshed in plover nesting behavior has also allowed for novel use of geospatial technology to map the landscape from a bird's eye view; not in an aerial sense, but from a plover's height on the ground. In our analysis, we configured the geospatial parameters to best represent the landscape plovers observe upon their return to their breeding grounds in April of each year by using aerial imagery (Appendix 1. Supplemental methods). Investigating new variables to better our understanding of piping plover behavior will continue to benefit managers in their decision-making regarding habitat alterations.

The obscured vision of an incubating plover from the nest could hinder its ability to detect approaching predators (Götmark et al. 1995, Javůrková et al. 2011). Not only is the nest at risk, the risk of adult depredation may be higher (Wiebe and Martin 1998, Amat and Masero 2004, Low et al. 2010). Our ability to assess nest site visible area on nest survival is hampered by the presence of exclosures, which increase nest survival (Melvin et al. 1992), but assessment of nest site visible area effect on adult survival is a possible next step in this analysis.



Coupled with our increased detections of predators (especially feral cats; Chapter 8. Predator numbers, density, and distribution on Fire Island, with an emphasis on red fox of this report) in recent years, management of the landscape is predicted to improve adult survival of breeding plovers at Fire Island.

Patterns of visible area within possible nesting habitat varied between the management units at Fire Island. Nests within the Fire Island National Seashore were located near Old Inlet, an area that is relatively open and flat due to the lasting effect of Hurricane Sandy, which would explain the high nest site visible area values in 2015 and 2016. The number of nests around Old Inlet increased from 8 in 2016 to 13 in 2017, so it is possible that due to the increasing nesting density (Chapter 1. Piping plover numbers, density, and distribution on Fire Island of this report), plovers began selecting nest sites that are in less ideal locations (i.e., have more visual obstructions; Fretwell and Lucas 1970, Fretwell 1972). The sudden decrease in nest site visibility at Robert Moses State Park is inexplicable, as there were no known habitat alterations during the years we considered and the number of nests in the area decreased from 37 nests in 2019 to 27 in 2020 (Chapter 1. Piping plover numbers, density, and distribution on Fire Island of this report). It may be that human beach recreation and subsequent disturbance resulted in piping plovers using habitat in less suitable places (Burger 1987;1994). Further investigation is necessary to explain this trend. Lastly, the high nest site visible area in Smith Point County Park in 2015 could reflect the restoration efforts completed where large portions of the landscape (i.e., New Made restoration area and Great Gun restoration area) experienced vegetation removal that we assumed would correspond to greater visible area at nest sites. In the years following restoration, the visible area for nest sites at Smith Point became more similar to other management units than what had been observed at Smith Point in 2015. Overall year-to-year fluctuations in nest site visible area could correspond to the variability in topographic landscape characteristics from sand deposition and erosion.

Using a use-versus-availability analysis, we also determined that the visible area was greater at nest sites compared to sites selected at random. This association may be caused by plovers prioritizing other habitat variables in a way that compromises the amount of visible area achieved at a given nest site. It also could reflect a switch toward the role of cover and camouflage in nest site selection (Swaigood et al. 2018), where predator avoidance becomes the strategy to avoid nest depredation instead of advanced predator detection.

By adding nest site visible area to the variables found to be significant from Walker et al. (2019), we see that the inclusion of nest site visible area significantly improves the predictive ability of the model in characterizing nest site selection behavior of this population of piping plovers. Although greater nest site visible area was selected for in this study, it has been noted that nest site selection can change over time or across regions (Walker et al. 2019, Zeigler et al. 2021). Barrier islands are dynamic coastal landscapes that can change rapidly during egg and incubation (April – June) and our novel geospatial analysis represents only a short period of the landscape in each year. Future research should expand upon our limited temporal scale by considering visible area at later stages in incubation, conditions experienced during renest attempts, and how rapid changes to visible area may influence site fidelity.

Chapter 6. Assessing the effects of habitat restoration for piping plover (*Charadrius melodus*) at Democrat Point, Fire Island, NY⁷

Objective P2: Compare piping plover nest and chick survival rates in various habitats across Fire Island

Introduction

Coastal populations of piping plovers rely predominately on sandy Atlantic beaches to provide suitable nesting habitat and abundant food resources for raising young. Like many other ground nesting shorebirds, piping plovers are particularly vulnerable to shoreline changes (Seavey et al. 2011, Zeigler et al. 2022). When choosing nest sites on these sandy breeding grounds, plovers select for territories that feature sparse vegetation (Burger 1987, Espie et al. 1996, Cohen 2005, Cohen et al. 2008, Zeigler et al. 2021), shell and pebble cover (Prindiville Gaines and Ryan 1988, Espie et al. 1996, Cohen et al. 2008, Maslo et al. 2011, Grant et al. 2019, Zeigler et al. 2021), and enable easy access to rich foraging grounds (Cohen et al. 2009a, Maslo et al. 2011, Walker et al. 2019). The same sandy beaches are preferred by humans, who displace breeding shorebirds with commercial and residential developments and beach nourishment projects (USFWS 1996, 2009; Hecker 2008). In areas with minimal human involvement, natural vegetative succession can reduce optimal nesting habitat while gradual erosion and sea level rise can increase susceptibility to flooding by storms or high tides.

Although shoreline disturbances contribute to the loss of shorebird habitat, other environmental processes create new nesting habitat, through both natural and anthropogenic mechanisms. For example, the physical changes left by hurricanes and other storms, while often harmful to human communities, may generate new habitat for colonization by nesting shorebirds (Cohen et al. 2009a, Convertino et al. 2011, Robinson et al. 2019, Robinson et al. 2020a). As storm surges overwash beaches, especially those on low elevation barrier islands, they remove vegetation and flatten dunes, depositing sediments inland and on lower elevation areas. The newly leveled and devegetated beaches provide high-quality piping plover nesting habitat (Walker et al. 2019, Zeigler et al. 2019a) while storm-created tidal flats and wetlands support the invertebrate prey base. After Hurricane Sandy in 2012, piping plover population on Fire Island, NY increased by 346.2% (Chapter 1. Piping plover numbers, density, and distribution on Fire Island of this report), with areas overwashed by the hurricane containing the most suitable plover habitat (Walker et al. 2019). Other studies have found the same type of population irruptions after these habitat creation events (Robinson et al. 2019), suggesting that these piping plover populations were at or near carrying capacity and were habitat limited until an increase in suitable breeding habitat allowed substantial population growth.

Indeed, the loss and degradation of nesting habitat, by both natural and anthropogenic means, has been identified as a major threat to the success of many declining shorebird populations, including piping plovers (USFWS 1996, Sutherland et al. 2012). Conservationists and land managers have therefore prioritized habitat creation and restoration as a conservation strategy, particularly for vulnerable or displaced species. However, due to our often-incomplete understanding of ecosystem dynamics and habitat selection, efforts to engineer and restore habitats frequently have unexpected outcomes (Suding 2011, Miller et al. 2017, Jones et al. 2018). Moreover, there may be a temporal lag between habitat creation events and the desired conservation outcome, and so multi-year studies are necessary to fully understand the outcomes of management actions.

⁷ **Cite chapter as:** VanDerwater, H.L., C.N. Wails, S.S. Dorsey, H.A. Bellman, S.G. Robinson, K.M. Walker, S.M. Karpanty, and J.D. Fraser. 2022. Assessing the effects of habitat restoration for piping plover (*Charadrius melodus*) at Democrat Point, Fire Island, NY. Pp. 75 – 84. In: *Effectiveness monitoring and targeted studies in support of FIMI conservation measures* (eds. Wails et al.). Annual Report prepared by the Virginia Tech Shorebird Project. Virginia Tech, Blacksburg, VA.



Densely concentrated human activity and development on sandy shorelines is a major contributor to piping plover habitat loss (Burger 1986;1994, DeRose-Wilson et al. 2018), and population growth in this species is density dependent (Robinson et al. 2019). Therefore, creating new nesting habitat and protecting naturally created habitat are important conservation strategies for this federally threatened species. The dune flattening and vegetation removal undertaken at Democrat Point, on the western end of Fire Island, during the winter of 2021/2022 provides a unique opportunity to study effects of restoration on a plover population that has been monitored for several years before this landscape modification. Here, we quantify the immediate effects of habitat restoration by comparing nesting pairs, hatching and fledging success, and abundance of non-breeding plovers resighted before and after the restoration. Given that past work has identified a temporal lag between habitat creation events and population growth (e.g., Robinson et al. 2020a), we predicted stationary breeding pairs and reproductive output immediately before and after the restoration, and that the new access between bay and ocean intertidal zones would promote an abundance of foraging piping plovers in years following (Cohen et al. 2009a).

Methods

Study area

Located within Robert Moses State Park, Democrat Point is a protected shorebird nesting area with minimal disturbance from foot or vehicle traffic during the summer breeding season, relative to the other recreation-focused beaches in the same park. In the fall and winter of 2021, the U.S. Army Corps of Engineers and U.S. Fish and Wildlife Service initiated a beach nourishment and habitat restoration project (Figure 30) with the goal of increasing quality plover nesting habitat. Before restoration, the flatter, more open parts of Democrat Point hosted nesting plovers, though much of the varied terrain and inconsistent elevation – including mudflats, patches of dense vegetation (Figure 30a), and large dunes – was not representative of optimal plover habitat. During the restoration, USACE and USFWS removed vegetation (Figure 30b), flattened dunes, and reallocated sediments to lower elevation areas. The 2022 landscape of Democrat Point has been leveled and reduced to an early state of vegetative succession, visibly similar to the aftermath of a hurricane overwash event.



Figure 30. Photos of the winter 2021/2022 construction at Democrat Point. Photographs from USACE.

Landscape changes

We quantified the transformation of the Democrat Point landscape using aerial imagery and LiDAR data in ArcGIS (Appendix 1. Supplemental methods). Using classified imagery from April 2021 and April 2022, we quantified the loss of vegetation during the restoration project. The digital elevation model (DEM), derived from LiDAR data, provided information on how restoration-related sand redistribution, in addition to natural

accretion and erosion, changed the topography of Democrat Point. We subtracted the April 2021, pre-restoration DEM layer from the April 2022, post-restoration DEM layer to visualize changes in elevation.

Demographic effects

Since 2015⁸, we monitored piping plovers nesting at Democrat Point. Nesting density across Robert Moses has been generally increasing since 2016 (Figure 15), and we sought to understand if post-restoration changes in the local breeding cohort (hereafter, “local population”) exceeded the observed trends. Using data from 2015 – 2021, we developed generalized linear models to predict 2022 nesting pairs and density, reproductive success (observed hatching and fledging rates), and abundance of nonbreeding piping plovers based on pre-restoration trends and compared observed post-restoration success to the values predicted by the models. In this analysis, we considered “nonbreeding” individuals to be all adult plovers that did not nest in Democrat Point in the same year that they were observed (e.g., a piping plover known to have a nest at Robert Moses Field 5 would be considered “nonbreeding” at Democrat Point).

We fit separate generalized linear models for each, and included year as the predictor variable. We used pair density to evaluate the relative volume of breeding pairs, where density was quantified as nesting pairs per hectare of dry sand habitat (negative binomial error distribution with an offset for hectares of dry sand). Hatching and fledging success are measured as the proportion of total nests that hatch and chicks that fledge, respectively (each with binomial error distributions). Nonbreeding individuals were measured as abundance per survey hour (negative binomial error distribution with an offset for survey hours; see Piping plover general field methods for detail on survey frequency). Generalized linear models were fit using the R package *glmmTMB* (Brooks et al. 2017), and we checked standard model diagnostics before using models to predict expected values.

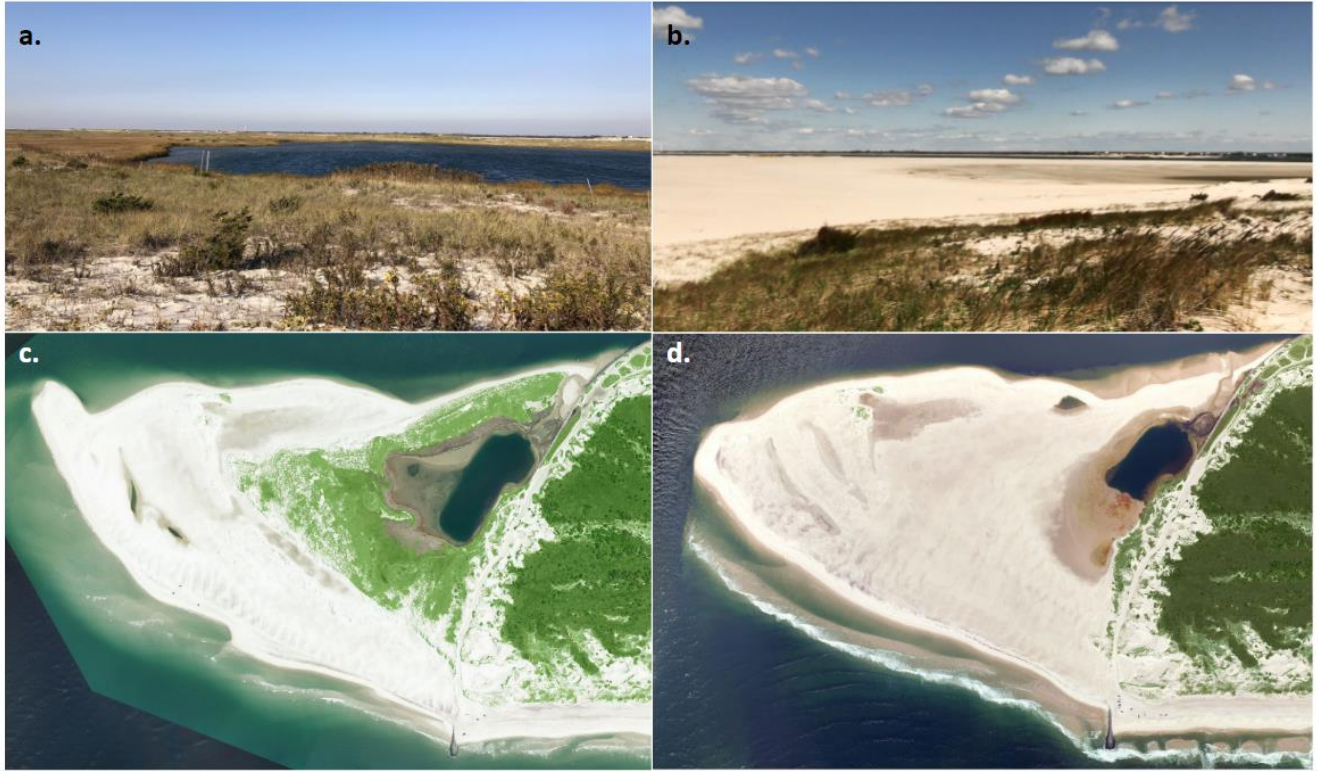
Results

Landscape changes

Overall, the restoration project increased the proportion of Democrat Point classified as unvegetated dry sand (i.e., potential nesting habitat) from approximately 21 ha (31% of Democrat Point; Figure 31a, c) in 2021 to 44 ha (68% increase; Figure 31b, d) in 2022. The most pronounced elevation changes (Figure 32) were generally on the eastern side of Democrat Point. Moderately sized dunes toward the northern shore of Democrat Point were reduced in elevation by > 8 ft and sand was reallocated to low elevation patches closer to the ocean, generally increasing their elevation by 4 – 8 ft.

⁸ But excluding 2020.





Photos: (a.) and (b.) Sierra Boucher/USFWS, (c.) and (d.) VTSP aerial imagery

Figure 31. Vegetation loss at Democrat Point. Photos taken from the same location on the eastern edge of Democrat Point **(a.)** in November 2021 **(b.)** and in April 2022 show extensive vegetation loss. Vegetation is colored bright green in these aerial images captured in **(c.)** April 2021 and **(d.)** April 2022.

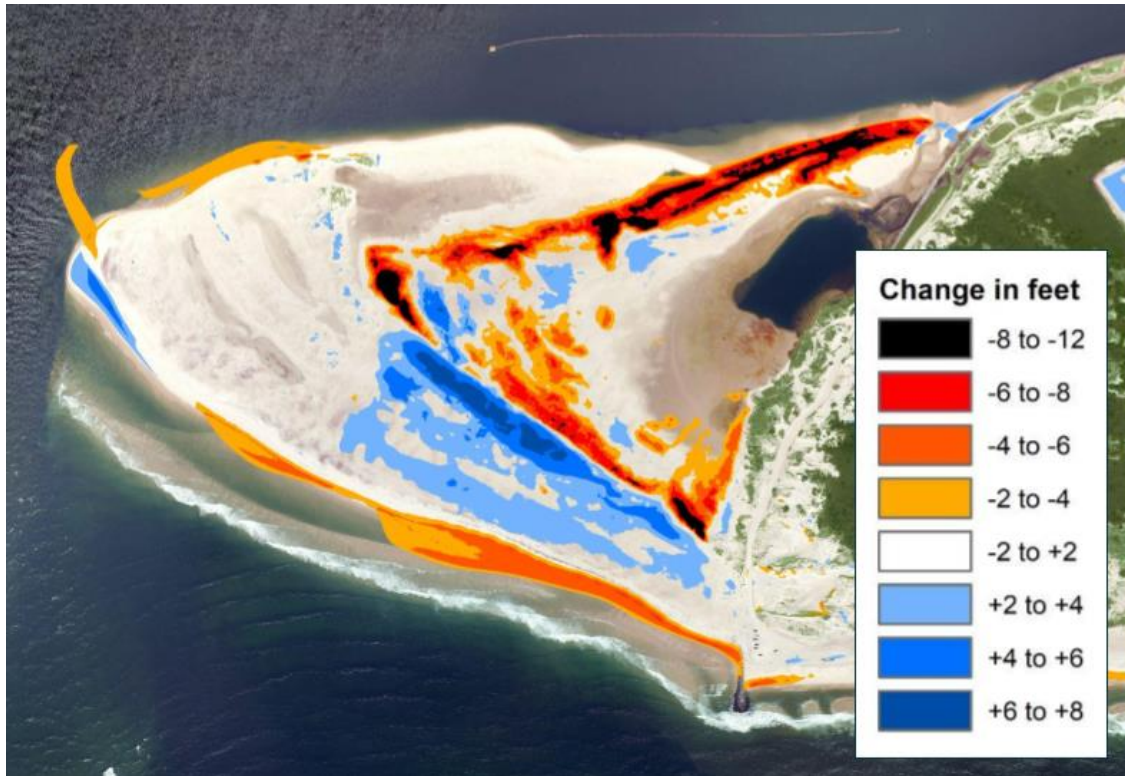


Figure 32. Elevation changes superimposed over April 2022 aerial imagery of Democrat Point. Colors indicate changes in elevation between April 2021 and April 2022; warm colors indicate dune removal, while cool colors indicate depressions that were leveled. The tidal pool was largely unchanged, and its extent fluctuated with tides and precipitation.

Demographic effects

The number of breeding pairs nesting at Democrat Point has grown annually since 2016 and post-restoration, we observed a substantial increase in pairs at this site, from 17 pairs in 2021 to 26 in 2022 (52.9% increase from 2021 – 2022, and 550% increase since 2016; Figure 33a). This post-restoration value is significantly greater than the value predicted by our model. With the 2022 expansion of available nesting habitat, the pair density in Democrat Point decreased from about 0.8 pairs/ha of dry sand in 2021 to about 0.6 pairs/ha in 2022 (27% decrease from 2021 – 2022; Figure 33b).

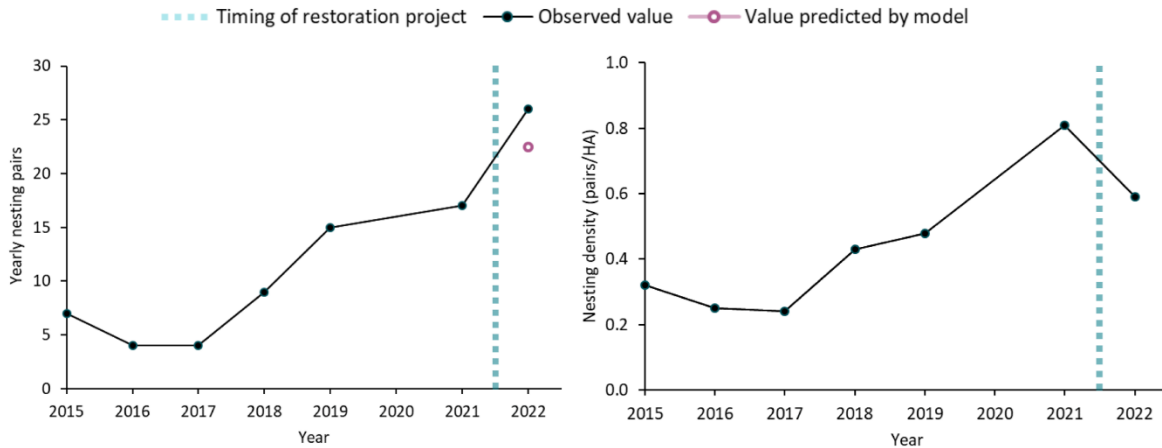


Figure 33. Nesting plover pairs at Democrat Point 2015 – 2022, expressed as **(a.)** total number of nesting pairs and **(b.)** density of nesting pairs, measured as pairs per hectare of available habitat. Available habitat was defined as dry sand patches using classified aerial imagery. In **(a.)**, the 95% confidence interval for the model-predicted estimate of nesting pairs is 21.97 – 22.87 pairs.

The post-restoration hatching success rate was 75% (21 of the total 28 nests hatched) and was within the margin of error of the model-predicted hatch rate (Figure 34a). Fledging success, the proportion of chicks observed in flight or survived to 25 days after hatch, has varied considerably at this site, and decreased since 2017. Though the 43% fledge rate in 2022 marked a slight increase from 36% fledged in 2021, the observed fledging success was also within the model-predicted range (Figure 34b).

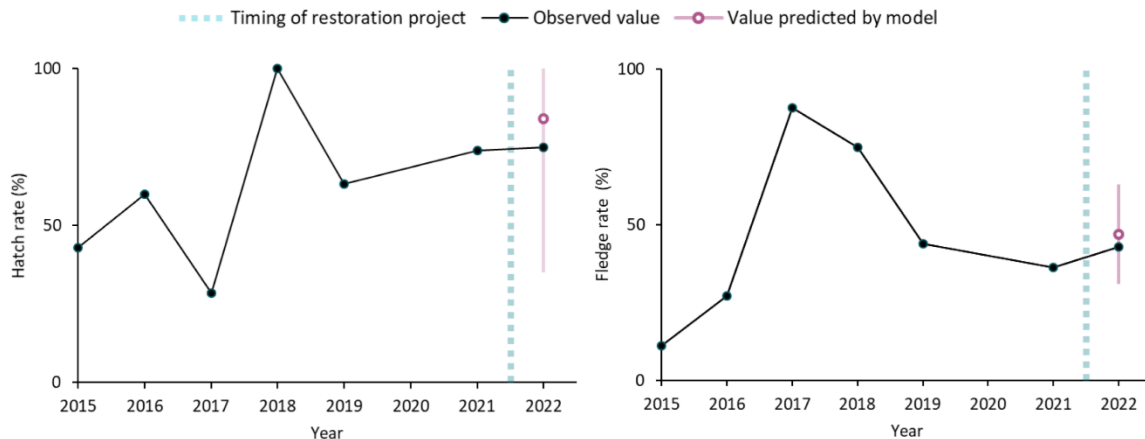


Figure 34. Reproductive success at Democrat point during 2016 - 2022 measured by **(a.)** hatching success and **(b.)** fledging success. Vertical dashed lines indicate the timing of the 2021 restoration project and gray square points indicate the 2022 value predicted by the model.

The observed post-restoration increase in non-breeding plovers was marginally greater than the mean estimated daily non-breeding birds, and within the margin of error of the model-predicted hatch rate (Figure 35). During the average 2022 survey at Democrat Point, we observed 13.5 non-breeding plovers, or adult birds that had not nested within the subsite, which is greater than the 2021 average (7.4 non-breeding birds; 82.4% increase from 2021 – 2022).



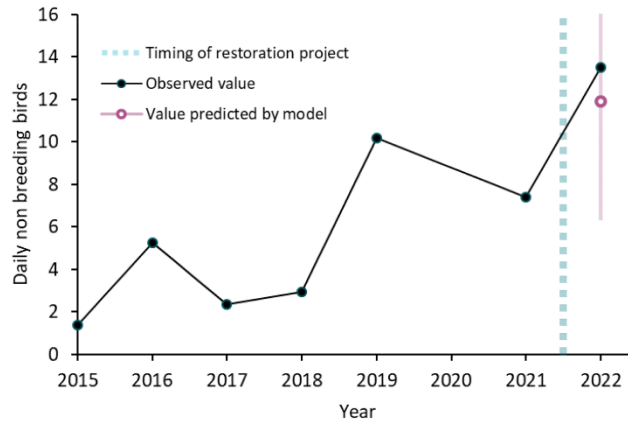
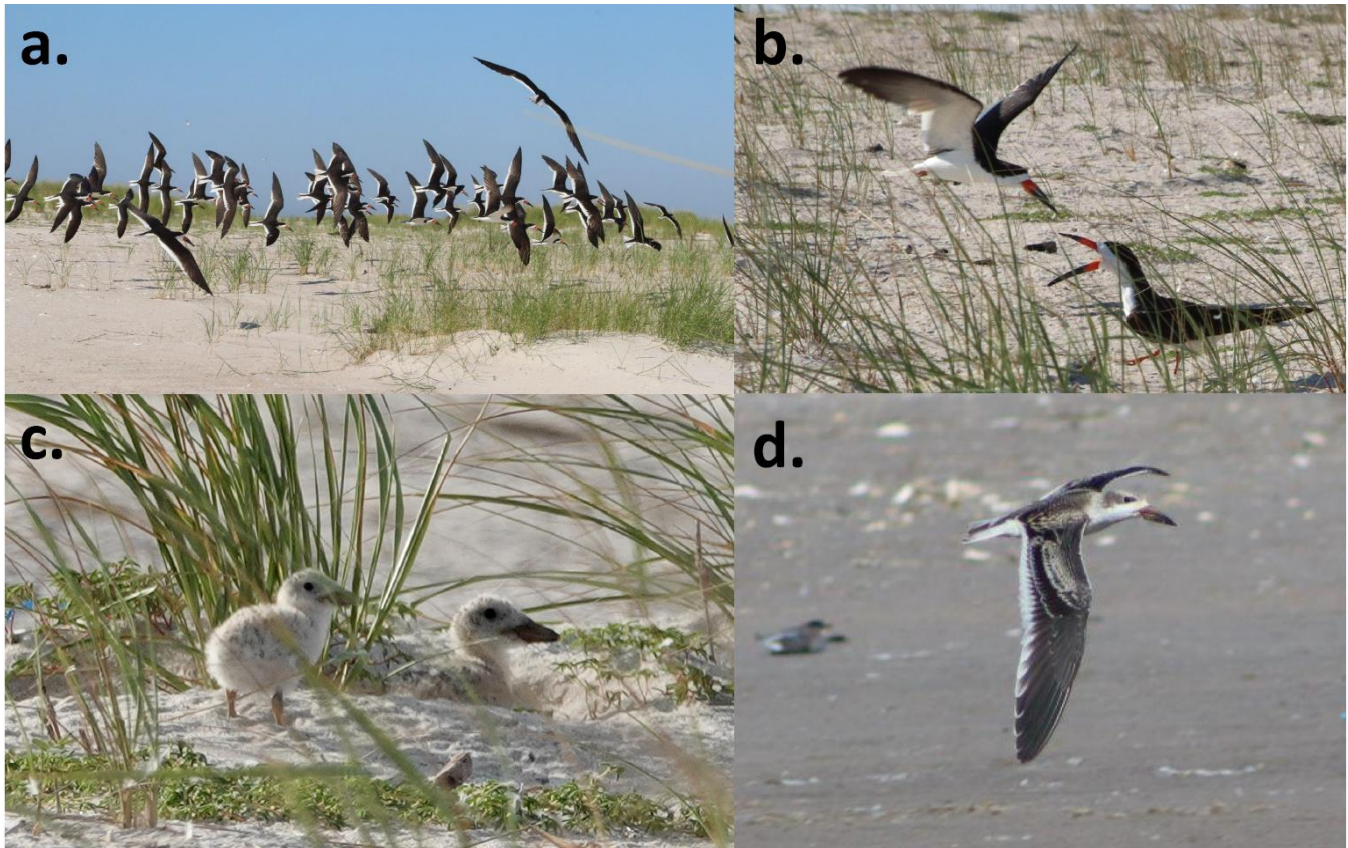


Figure 35. Non-breeding adult plovers per survey day at Democrat Point 2015-2022. Vertical dashed line indicates the timing of the 2021 restoration project and gray square point indicates the 2022 value predicted by the model.

The transformed landscape also provided nesting habitat to American oystercatchers (*Haematopus palliatus*), willets (*Tringa semipalmata*), and existing least (*Sternula antillarum*) and common tern (*Sterna hirundo*) colonies. In mid-June, black skimmers (*Rynchops niger*) arrived and established a new breeding colony with >85 adults in the northwestern part of Democrat Point (Figure 36a, b). Despite the colony’s late arrival and some observed skimmer chick mortality (including two observed instances of skimmer chick depredation by great black-backed gulls; H. L. VanDerwater, S. J. Denenberg, K. E. Johnson, and D. A. Resich, *pers. comm.*), the skimmer pairs successfully hatched (Figure 36c) and fledged chicks (Figure 36d).



Photos: (a.), (b.), and (d.) Hope VanDerwater and (c.) Eric Swan

Figure 36. Photographs of the newly established black skimmer (*Rynchops niger*) colony following the restoration at Democrat Point, Fire Island, NY in 2022, showing (a.) adults in flight, (b.) adults at a nest, (c.) pre-fledged skimmer chicks, and (d.) a recently fledged skimmer in flight.

Discussion

In the breeding season immediately following the Democrat Point habitat restoration project, we did not find a significant difference between post-restoration plover success and pre-restoration success. Generally, plover reproductive output was slightly lower than but within the pre-restoration values, whereas our estimate of the number of breeding pairs and non-breeding individuals may have been attracted to the rich and accessible foraging habitat in the newly restored areas. The removal of many dunes and dense vegetation by beach regrading throughout the site likely enhanced access between bay and ocean intertidal zones. We expect that the removal of these physical obstacles has reduced broods' least-cost distance to intertidal foraging areas on all three sides of the point and we hope that the improved accessibility of food resources will have a positive effect on chick survival (Cohen et al. 2009a). At the easternmost edge of Democrat Point, tidal pools and mudflats provided rich foraging habitat (Elias et al. 2000), with many parents permanently relocating their broods to this area. Later in the season, Democrat Point hosted many adults and fledglings from nests from further parts of Fire Island (e.g., Smith Point), apparently foraging before migration.

We attribute the lack of an immediate post-restoration reproductive response, in part, to two specific post-restoration circumstances: predation by peregrine falcons (*Falco peregrinus*, Table S2 and Chapter 7. An investigation of the avian predators at Robert Moses State Park and the NPS Lighthouse of this report) and a



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mid-season flood (Figure 37). This year, we observed frequent peregrine falcon activity, recording their presence on 10 of our 19 visits to Democrat Point in July and August, and identifying at least three unique falcons, two juvenile and one adult, from their leg bands and plumage. Examination of peregrine pellets – which contained bone, feathers, and, once, bands – confirmed the depredation of many least terns and at least two plover chicks from unidentified broods. Plover parents generally prefer habitats with sparse vegetation, selecting for 5 – 40% vegetation density (Cohen et al. 2009b, Swaisgood et al. 2018), which conceals chicks from visual predators (Troschianko et al. 2016), like raptors, on the otherwise open beaches. Hence, the complete loss of vegetation during the Democrat Point restoration enabled hunting peregrine falcons to more easily locate their plover prey, which may have contributed to the lower-than-expected fledging success we observed. Despite the recent large-scale habitat change on Democrat Point, we observed moderate plant growth over the course of the 2022 season, indicating that natural vegetation succession will proceed and presumably protect chicks from persistent aerial predation. All found nests were within the area protected from human disturbance by the symbolic fencing and each nest was individually exclosed by land managers. Consequently, we did not observe any instances of nest disturbance or egg or chick depredation by terrestrial predators (Table S2, Table S4).

In mid-July, an exceptionally high tide pushed by strong winds flooded much of the western extent and part of the interior of Democrat Point. On 12 July, the overnight high tide at Democrat Point was 3.53 ft (maximum high tide in 2022 prior to July was 15 June at 3.57 ft) with northerly wind gusts that were in excess of 20 mph (vs. approximately 10 mph northerly gusts on 15 June), and the following evening, the high tide was 3.60 ft with overnight northerly gusts in excess of 30 mph (NOAA 2022). Four of the six remaining plover nests were lost to flooding, and floodwaters submerged the exclosures of eight nests that hatched earlier (Figure 37a, b). Because the flood occurred late in the season and the floodwater receded slowly, the pairs associated with the four flooded nests (to the best of our knowledge) did not renest at Democrat Point. This flooding had a negative impact on hatching success, causing four out of seven nest failures for the remaining unhatched nests, and we believe that the flood, and the resulting habitat disturbance and fragmentation, also decreased chick survival. We observed one plover chick stranded in a pool on a small raised patch of grass > 10 m away from dry ground (Figure 37c). Though flooding is not uncommon on barrier islands, the loss of stabilizing vegetation and protective berms and dunes by regrading projects may put coastal ecosystems at increased risk of erosion, storm damage, and flooding (Withers 2002). However, we expect the normal deposition process and continuing westerly expansion of Democrat Point (USACE 1996, 2002) will shape natural landscape features that limit the extent of flooding.



Photos: Hope VanDerwater

Figure 37. Images of the high tide floodwaters at Democrat Point showing (a.) a nest exclosure surrounded by water on the western end of the point, (b.) a flooded nest exclosure close to the bay, and (c.) a plover chick (red

arrow) isolated on a small patch of sand surrounded by high floodwaters. All of the water visible in these images was a result of the flooding.

Because available nesting habitat in 2022 was more than twice the area of 2021 habitat, we observed an immediate post-restoration decline in nesting density. If the local piping plover population at Democrat Point is habitat-limited, we predict the release of density will increase the local carrying capacity and allow population growth in subsequent seasons (Robinson et al. 2019, Walker et al. 2019). However, as observed in previous studies, the delayed demographic response after a large-scale disturbance is dependent on gradual physical processes (Schupp et al. 2013, Bourque et al. 2015) and intrinsic patterns of plover recruitment and population growth (Robinson et al. 2019). Since the first chicks hatched in newly created habitat cannot contribute to local population growth as breeders until their second year, the effects of post-restoration natal recruitment are inherently delayed by a year. The non-breeding plovers observed foraging, or otherwise prospecting potential nest sites at Democrat Point may forecast future recruitment rates (Cohen et al. 2009a, Davis et al. 2017), while non-local plovers may require more time to find and colonize new habitat. These intrinsic delays in demographic growth, coupled with the loss of protective habitat features at Democrat Point (e.g., dunes to prevent flooding and vegetation to conceal chicks), led to hatching and fledging success that was slightly lower than the pre-restoration trend. However, we expect that moderate vegetative growth and dune formation will have a protective effect on newly created habitat, improving retention of local breeders and supporting general reproductive success in future years (Cohen et al. 2008, Robinson et al. 2019).

Overall, our results for the first post-restoration breeding season at Democrat Point appear to align with the findings of similar studies, which have shown a temporal lag between a habitat creation event and the subsequent population response (Nagelkerke et al. 2002, Robinson et al. 2019, Walker et al. 2019, Robinson 2020, Robinson et al. 2020a). If the population at Democrat Point was previously habitat limited, we should expect to see a similar local population irruption within the next few years. However, as sediment movement and vegetative succession gradually return the site to its pre-restoration state, human intervention may be necessary to preserve shorebird habitat (Catlin et al. 2011b, Convertino et al. 2011, Zeigler et al. 2019b). Consequently, long-term monitoring of populations before and after such disturbance events (Schupp et al. 2013) is important to inform future conservation and stabilization practices.

Chapter 7. An investigation of the avian predators at Robert Moses State Park and the NPS Lighthouse⁹

Objective P5: Quantify the interactions of predators with shorebird nests and chicks

Introduction

During the breeding season, piping plovers' survival and breeding success are affected by many threats, including those posed by mammalian and avian predators. Historically, management has focused on protecting nests from these mammalian predators. The use of nest exclosures prevents most mammals from accessing nests (Burger 1987, Rimmer and Deblinger 1990, Melvin et al. 1992). Due to their design, exclosures also provide protection against larger avian predators; however, the exclosures may also increase visibility of the nest location and create perches for avian predators. Consequently, it is important to consider both the advantages and disadvantages of erecting exclosures at specific locations, in order to maximize the benefit to nesting plovers (Murphy et al. 2003, Barber et al. 2010, Cohen et al. 2016, Anteau et al. 2022).

At Fire Island, the impact of avian predation remains largely unknown; therefore, we sought to characterize the avian predator community, and the abundance and distribution of predators. In addition to characterizing the avian predator community, we were also interested in how the distribution of predators compares to the locations of plover nests and broods. Plover eggs and chicks have different predators; therefore, it is important to note how the distribution of potential predators change over the breeding season and to consider the presence of species in relation to their expected impact. We expected to observe more species of potential avian predators at Democrat Point, due to the large tern colony and therefore abundance of prey. We predicted the distribution of potential avian predators to change over the course of the breeding season, as newly-fledged juveniles and adults of many species disperse from their breeding territories to nearby areas with minimized human disturbance, such as Democrat Point. Additionally, we summarized our anecdotal evidence of interactions between potential avian predators and piping plovers. These behavioral interactions support the classification of these species as predators of plovers. This study represents the first step towards understanding how piping plover mortality from avian causes compares to other sources of mortality and can be used as a model to expand the knowledge of the effect of avian predators on other portions of Fire Island to inform management practices.

Methods

Literature review

To determine which avian species to monitor during fieldwork, we conducted a literature review to identify known and suspected predators of piping plovers. Because avian predators of plovers vary geographically, we considered only predators reported in the literature that also occur at Fire Island.

Study area

Fieldwork was conducted across Robert Moses State Park and the NPS lighthouse. Robert Moses State Park contains some of the most popular recreational beaches on Long Island. Consequently, some areas of the beach, such as Field 5, are heavily used by humans, and there is reduced availability of plover nesting habitat. In contrast, the westernmost tip of the island, Democrat Point, has restricted recreational access, and the wide-open sandy space supports large, mixed-species breeding colonies of common (*Sterna hirundo*) and least terns

⁹ Forthcoming honors thesis. **Cite chapter as:** Johnson, K. E., C. N. Wails, H. L. VanDerwater, S. S. Dorsey, S. M. Karpanty, and J. D. Fraser. 2022. An investigation of the avian predators at Robert Moses State Park and the NPS Lighthouse. Pp. 85 – 94. In: *Effectiveness monitoring and targeted studies in support of FIMI conservation measures* (eds. Wails et al.). Annual Report prepared by the Virginia Tech Shorebird Project. Virginia Tech, Blacksburg, VA.



(*Sternula antillarum*), and a newly established colony of black skimmers (*Rynchops niger*). American oystercatchers (*Haematopus palliatus*), willets (*Tringa semipalmata*), and many pairs of piping plovers also nest among these colonial breeders (Figure 38).

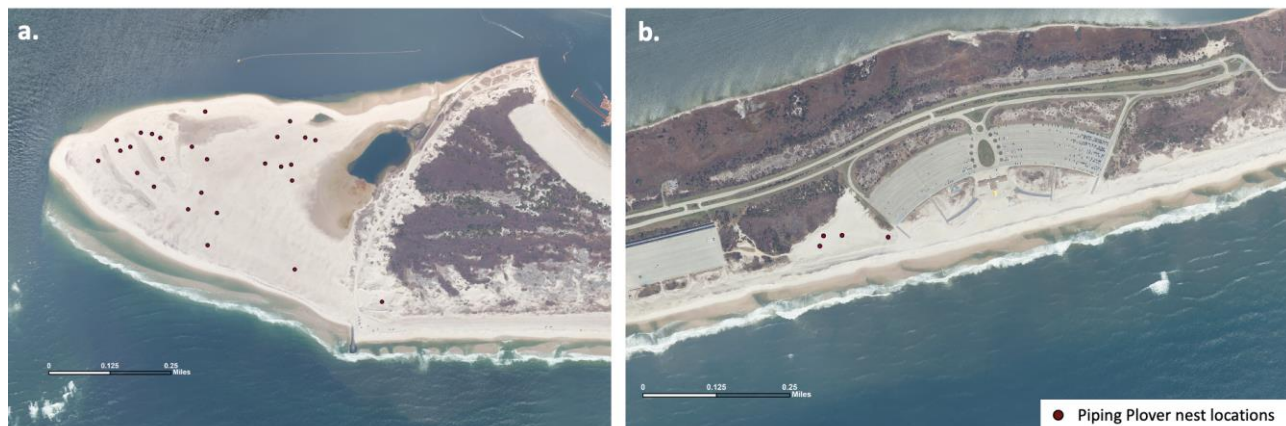


Figure 38. Aerial imagery of (a.) Democrat Point and (b.) Field 5, two sections of Robert Moses State Park, in addition to piping plover nest locations in summer 2022.

Field methods

To determine nesting and fledging success of the piping plovers nesting on Fire Island, we routinely monitored nests until hatching or failure, and regularly checked broods after hatch. At Robert Moses State Park and NPS Lighthouse, we monitored 46 nests of about 40 pairs of plovers in 2022. Thirty-one of these nests successfully hatched at least two chicks, for a total of 108 chicks. Fifty-seven of these chicks were confirmed fledged, based on observing them in sustained flight or alive after 25 days of age.

During the summer of 2022, we made instantaneous counts (Altmann 1974), recording numbers of potential avian predators while traveling through subsites, of potential avian predators while in plover nesting areas from early June to mid-August. During routine monitoring activities, we made observations in plover nesting areas and recorded the species of each potential avian predator, their habitat (intertidal, ephemeral pool, wrack line, dry sand, dry vegetation, or other), behavior (loafing, foraging, roosting, courtship, parental, incubating, flying, disturbed, or other), and flock size. We recorded abundance and behavior of raptors, gulls, corvids, blackbirds, and other shorebird species (i.e., ruddy turnstone and American oystercatcher) that are known or suspected to take piping plovers or their eggs (Kruse et al. 2001, Lauro and Tanacredi 2002, Wilke and Denmon 2014).

Data processing and statistical analyses

We conducted all statistical analyses in R ver. 4.1.3 (R Core Team 2021) and geospatial processing in ArcGIS (ver. 10.8.2; ESRI, West Redlands, CA).

We considered the density, calculated as the number of potential avian predators per observer hour per subsite (total ha in subsite), and distribution of avian predators at Robert Moses State Park and NPS lighthouse. The number of days with observations in each subsite varied (range: 8 – 31) depending on the abundance of plovers nesting in each area, and likewise, the total time spent in a subsite also varied (range: 7.60 – 99.42 hr). Thus, we standardized predator counts for observer effort and survey area (predator abundance / observer hours per subsite ha), and compared predator community composition between months. We visualized avian predator counts each month using principal component analysis (Jolliffe and Cadima 2016) and tested for significant

changes in the PCA centroid means and spread using PERMANOVA and PERMDISP (Anderson 2001) in the R package *vegan* (Oksanen et al. 2022).

We also overlaid the density and distribution of avian predators with nest and brood locations to visualize the relative locations of plovers and the density of avian predators. We calculated the average daily abundance of predators recorded in each subsite. To visualize predator density for each subsite, we summed the average daily predator abundance across all predator species in June, July, and August and divided by the subsite area (ha).

We tested whether maximum avian density (observed in July) explained the number of plover fledglings per brood. We fit a logistic regression predicting brood success (our y variable represented the frequency of chicks fledging [1] and chicks failing to fledge [0] per brood) and included avian density (average daily predator count per ha in a subsite) in July using the R package *glmmTMB* (Brooks et al. 2017) and examined significance of avian density ($\alpha = 0.05$).

Results

Literature review

Through the literature, we found 15 confirmed or suspected avian predators of piping plovers that are also present along the Atlantic Coast (Table 23), and we considered these species as predators in our analyses. In addition to known plover predators, we also considered boat-tailed grackles (*Quiscalus major*) and ruddy turnstones (*Arenaria interpres*), as they have been documented taking eggs of other shorebird species (Parkes et al. 1971, Watts et al. 2006). Likewise, we considered lesser black-backed gulls (*Larus fuscus*) which form mixed-species foraging flocks with other gull predators, osprey (*Pandion haliaetus*) which are unlikely to take plovers though may be perceived as a predator (Weithman et al. 2020), and peregrine falcons (*Falco peregrinus*) which depredate other species of shorebirds (e.g., Beauchamp 2008, Watts and Truitt 2021).

Table 23. Avian predators on the Atlantic Coast that are known or suspected to depredate piping plover eggs, chicks, and adults reported in the published literature. Avian predators known or suspected to depredate plovers but do not occur on the Atlantic Coast (e.g., Swainson's hawk, *Buteo swainsoni*; Murphy et al. [2003]) were not included. Avian predators are listed in taxonomic order following Jarvis et al. (2014). Plover populations correspond to footnotes a = US Atlantic Coast, b = Great Plains, c = Great Lakes, d = Canadian Maritimes/Canadian Atlantic Coast. Table continues on the next page.

Avian predator	Egg predation	Chick predation	Adult predation	Unspecified or suspected
American oystercatcher (<i>Haematopus palliatus</i>)	Wilke and Denmon (2014) ^a	Call et al. (In Review) ^a		
Laughing gull (<i>Leucophaeus atricilla</i>)				Lauro and Tanacredi (2002) ^a
Ring-billed gull (<i>Larus delawarensis</i>)	Ivan and Murphy (2005) ^b	Ivan and Murphy (2005) ^b , Saunders et al. (2018) ^c	Saunders et al. (2018) ^c	Whyte (1985) ^b
Great black-backed gull (<i>L. marinus</i>)				Rimmer and Deblinger (1990) ^a , Lauro and Tanacredi (2002) ^a
Herring gull (<i>L. argentatus</i>)		Saunders et al. (2018) ^c	Saunders et al. (2018) ^c	Rimmer and Deblinger (1990) ^a , Lauro and Tanacredi (2002) ^a
Northern harrier (<i>Circus hudsonius</i>)		Ivan and Murphy (2005) ^b		Whyte (1985) ^b , Murphy et al. (2003) ^b
Bald eagle (<i>Haliaeetus leucocephalus</i>)	Andes et al. (2019) ^b			
Great-horned owl (<i>Bubo virginianus</i>)	Andes et al. (2019) ^b	Kruse et al. (2001) ^b , Catlin et al. (2011a) ^b		Rimmer and Deblinger (1990) ^a , Murphy et al. (2003) ^b
Red-tailed hawk (<i>Buteo jamaicensis</i>)				Murphy et al. (2003) ^b
American kestrel (<i>Falco sparverius</i>)		Kruse et al. (2001) ^b		
Merlin (<i>F. columbarius</i>)			Murphy et al. (2003) ^b , Roche et al. (2010) ^c , Saunders et al. (2018) ^c	Barber et al. (2010) ^d
Common raven (<i>Corvus corax</i>)	Barber et al. (2010) ^d	Saunders et al. (2018) ^c	Saunders et al. (2018) ^c	

Avian predator	Egg predation	Chick predation	Adult predation	Unspecified or suspected
Fish crow (<i>C. ossifragus</i>)				Lauro and Tanacredi (2002) ^a
American crow (<i>C. brachyrhynchos</i>)	Barber et al. (2010) ^d , Cohen et al. (2009a) ^a	Ivan and Murphy (2005) ^b , Saunders et al. (2018) ^c	Saunders et al. (2018) ^c	Whyte (1985) ^b , Rimmer and Deblinger (1990) ^a , Lauro and Tanacredi (2002) ^a , Murphy et al. (2003) ^b , Ivan and Murphy (2005) ^b
Red-winged blackbird (<i>Agelaius phoeniceus</i>)	Deblinger et al. (1992) ^{a,d}			Lauro and Tanacredi (2002) ^a
Common grackle (<i>Quiscalus quisculua</i>)	Deblinger et al. (1992) ^{a,d}	Ivan and Murphy (2005) ^b		Rimmer and Deblinger (1990) ^a

Field results

We observed 14 species of potential avian predators across the study location (Table 24). Gulls were the most abundant potential avian predator and were observed daily (great black-backed gull, *Larus marinus* = 122.1 birds/day; herring gull = 58.0 birds/day; and lesser black-backed gull, *L. fuscus* = 35.5 birds/day; Table 24), though we detected no evidence of gulls taking plovers within the study area in 2022 (but see Table S2). Gulls increased in the latter half of the breeding season, which is consistent with our observations of more fledged juvenile gulls and accompanying adults in August (Figure 39).

Table 24. Average number of individuals per day per subsite for each of the 14 potential avian predator species observed.

Species	Mean individuals per day
Great black-backed gull	122.08
Herring gull	57.95
Lesser black-backed gull	35.53
American oystercatcher	28.83
Ring-billed gull	17.58
Laughing gull	9.60
Red-winged blackbird	2.45
Osprey	1.75
Common grackle	1.70
Ruddy turnstone	1.28
Peregrine falcon	0.30
American crow	0.15
Fish crow	0.13
Boat-tailed grackle	0.05

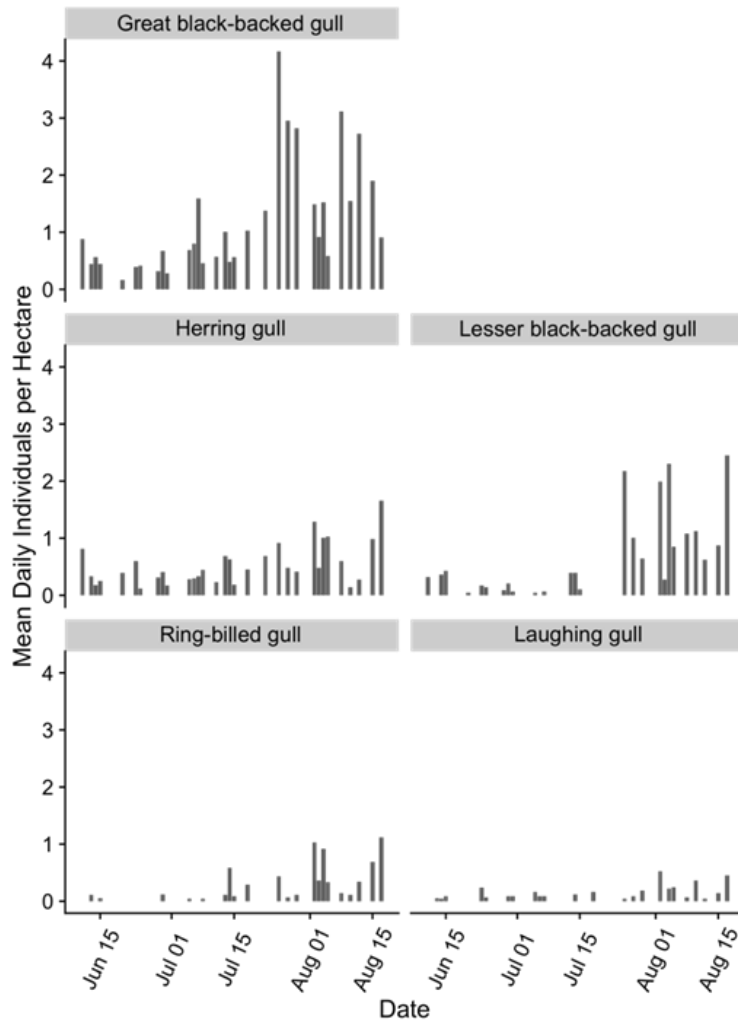


Figure 39. Changes in gull abundance (mean daily individuals per hectare) across months for all species.

Principal component analysis revealed that the potential avian predator community varied significantly across the three months surveyed (PERMANOVA $F = 6.44$, $P < 0.01$, PERMDISP $F = 8.13$, $P < 0.01$; Figure 40). The principal components accounted for 99.8% of the variance in the data, with the most influential principal component, approximately corresponding to gulls, driving the majority of that influence. In early June, the predator community was consistent across areas we surveyed (Figure 40). As the summer progressed, we saw greater abundances of shorebirds, particularly influencing the July community, and gulls which largely characterized the July and August communities (Figure 39).



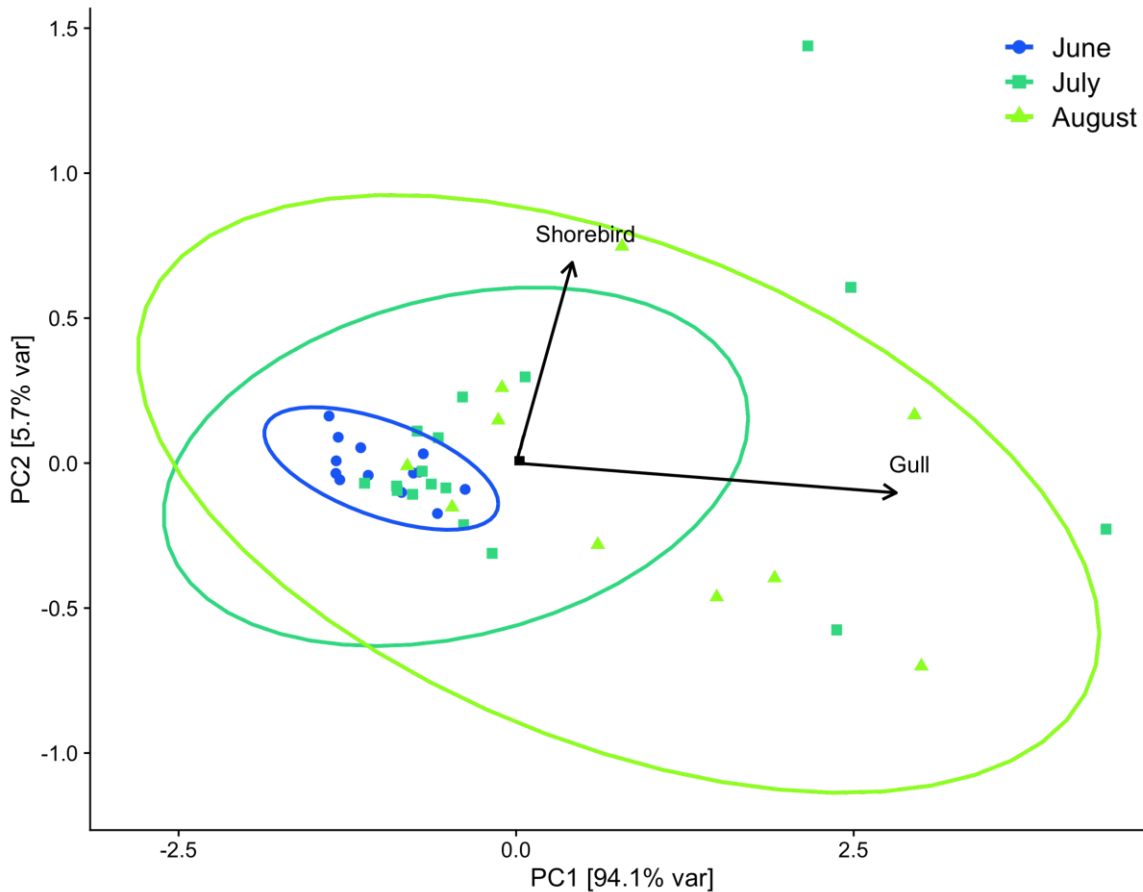


Figure 40. Principal component analysis of the potential avian predator community. Each point, color-coded by the month, represents an observation at a survey area on a given date. Points closer together are similar in their avian predator community composition while points farther apart are different. Ellipses illustrate the 95% confidence interval for the avian predator community composition for a given month. The vectors show which taxa are driving changes in the avian predator community.

Considering the densities of potential avian predators across taxa within the subsites at Robert Moses State Park and the NPS Lighthouse, we found differences between June, the early part of the breeding season, and August, late in the breeding season (Figure 41). Overall, there was a higher density of potential avian predators in August, especially in the subsites of Field 5 and Field 2/ORV area, which are among the heavily human-used beaches. The map of June also shows all of the nest locations, and the brood locations from June. The map of August only shows all brood locations from July and August.



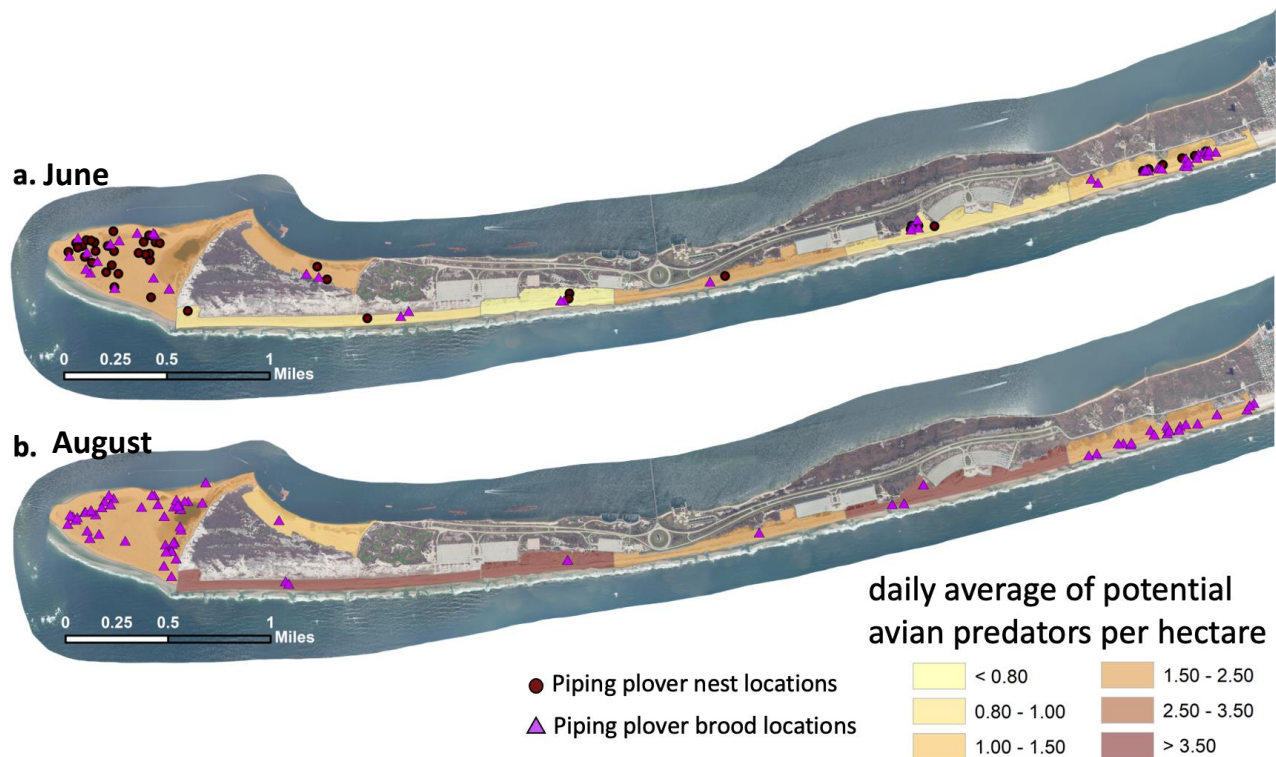


Figure 41. Potential avian predator densities (calculated as the daily average of potential avian predators per hectare) between **(a.)** June, the early part of the breeding season, and **(b.)** August, late in the breeding season. The map of June also shows all of the nest locations (red circles), and the brood locations (pink triangles) from June. The map of August only shows all brood locations (pink triangles) from July and August.

We rarely observed avian predators taking plovers or their eggs, and observed three instances of birds perching on nest enclosures in the study area (two instances of willet, one instance of a red-winged blackbird). Peregrine falcons were the only predator observed actively hunting nesting shorebirds and were seen on 10 of the 19 visits to Democrat Point (peregrine falcon abundance across entire 2022 study period: 0.3 birds/day). We confirmed the peregrines were taking plovers by recovering plover legs and color bands inside a peregrine pellet that was recovered after seeing the falcon depart from a roost¹⁰. Three individual falcons (two juveniles and an adult) were identified over the summer. One juvenile observed in August that was banded at the Met Life building in New York City in May, a distance of about 58 km. Although we observed two instances of take (Table S2), avian predator density did not explain brood success ($Z = -1.27$, $P = 0.21$).

Discussion

Across the plover breeding season, the diversity and abundance of avian predators changed, reflecting fledging of avian predators that breed on or around Long Island (gulls, peregrine falcons) and migratory patterns (other shorebirds). Despite the changing avian predator community, we did not find evidence that predator density as

¹⁰ Two confirmed instances of take by peregrine falcons occurred at Democrat Point. VTSP researchers recovered plover legs and color bands from peregrine pellets at a known feeding roost. VTSP researchers were unable to determine which broods lost chicks to falcon depredation. VTSP researchers filed incident observation reports on 8 July and 13 July.

measured here explained plover brood success, and instances of take by avian predators were infrequent at Fire Island.

Elsewhere, gulls and crows are considered to be the primary avian predators of plover nests (Lauro and Tanacredi 2002, Struthers and Ryan 2005), though the same is unlikely to be true at Fire Island. Most plover nests at Robert Moses and the NPS lighthouse are exclosed, limiting the potential for nest depredation by most of the avian predators we considered (Table 23, Table S2). Additionally, we seldom observed other birds using nest exclosures as perches, and the two species seen on nest exclosures (willets and a red-winged blackbird) were not associated with any cause of nest failure in the study area (Chapter 2. Piping plover nest and chick survival rates of this report).

Although gulls were the most abundant avian predators, we did not observe any instances of chick or adult take at Robert Moses or the NPS lighthouse (but see Table S3). However, the abundance of gulls observed in the study area was less than other locations where gulls are recognized as primary predators (e.g., at Monomoy National Wildlife Refuge, which supported > 19,000 pairs of herring and great-black backed gulls; Melvin et al. 1992, but see Keane et al. 2002). Fire Island does not currently support large colonies of breeding gulls, and based on locations where gulls were most frequently observed, we believe gulls were likely relying on human refuse on recreational beaches (ORV area/Field 2 and Field 5 at Robert Moses). It may be that gull predation at Fire Island is rare and opportunistic.

We confirmed peregrine falcons as a predator of piping plovers by recovering piping plover chick bands and legs inside peregrine pellets. We first observed a peregrine at Democrat Point on 6 July, and at least one peregrine was present on 10 of the 19 visits to Democrat Point (6 July – 19 August). We saw two individual juveniles based on plumage (one of which was banded and originated from New York City) on 10 August; all other observations were of one unbanded adult. It is possible this was a related group, with parents hunting at Fire Island to feed their young. In Colorado, some peregrine falcons traveled 79 km when foraging (Enderson and Craig 1997), and the distance to NYC is within this range. However, we were unable to confirm whether this truly was a family group, as two of the three peregrines were unbanded. While this is just one of our observed potential predators, it demonstrates that there is avian predation on piping plovers at the study site.

We did not detect strong evidence of the abundance of potential avian predators explained plover brood success. Perhaps avian predation risk for piping plover chicks is diluted in later months, as the increase in migratory shorebirds (especially those that may be larger and more conspicuous) may be at greater risk of depredation. Alternatively, it is possible that due to the limited scope of this study, we are missing factors that have a greater impact on plover chick survival. In either case, further monitoring can help to answer these questions.

We were able to characterize the potential avian predator community present at Robert Moses State Park and NPS lighthouse in the summer of 2022. We were also able to make comparisons of the avian predator community across the breeding season, for the months surveyed; however, we do not yet understand the potential for annual changes to the avian predator community. Knowing the species present the study location can inform focal species for future research, for example understanding the interactions between peregrine falcons and piping plovers. However, merely the presence of these potential predators does not guarantee an impact on plovers. More study, such as considering how often these avian predators are visiting nests, which could be quantified through the use of nest cameras, is needed. Though our current understanding of the distribution and types of potential avian predators at present is incomplete, it does increase our knowledge of the avian predator community at Fire Island and can inform future management decisions, such as vegetation control.



A close-up photograph of a sandy beach. Several sets of fox tracks are visible, appearing as dark, irregular impressions in the sand. The tracks are arranged in a series, suggesting a path taken by a fox. The sand is light-colored and has a fine, granular texture. The tracks are located in the upper half of the frame, with one set in the top left and another set further down and to the right. The overall scene is a natural, outdoor environment.

Red fox monitoring on Fire Island

Red fox general field methods

Due to the novel coronavirus (SARS-CoV-2) pandemic in 2020, we have had to make some modifications to planned fieldwork. All changes made to the field methods used to monitor red foxes in 2020 and 2021 are explicitly detailed here.

Red fox GPS-collaring and ear-tagging

Previously used methods: During August – December 2015, April – May 2016, and February – April 2017 and 2018, we attempted to capture a portion of the red fox population in each study site using padded and offset foothold traps and box traps, following the Association of Fish and Wildlife Agencies' best management practices (AFWA 2014). Upon capture, we manually restrained red foxes, determined their sex, and ear-tagged them when allowed by site managers. We placed a single tag in the animal's right ear using serially-numbered plastic livestock ear tags (custom MiniTags, Premier 1 Supplies, Washington, IA) at Robert Moses State Park and Smith Point County Park, and small metal ear tags (self-piercing small animal ear tags, size 4; National Band and Tag Company, Newport, KY) at Fire Island National Seashore. The colored plastic ear tags were intended to aid in identification of individual red foxes in remote camera photos and those encountered during transect surveys (see below). In contrast, the small metal ear tags were not intended to be visible in remote camera photos or during transect surveys, but were meant to assist in identifying any individuals recaptured in future years. We also outfitted a sample of red foxes in each site with GPS collars.

The GPS collars (Quantum 4000E, medium size; Telemetry Solutions, Concord, CA) included a VHF transmitter and remote data download and reprogramming capabilities via a UHF connection. To download GPS location data, we used VHF radio telemetry to relocate GPS-collared red foxes every 10 – 14 days until the programmed drop off date 5 – 6 months later, or until the animal's death. During these relocations, we also checked for mortality signals, and used VHF signals to locate the remains of red foxes that had died. For those foxes, we attempted to determine a cause of death through close inspection of remains and collected tissue samples, when possible, for later genetic analysis. GPS data from collared red foxes assisted in identifying den locations and in tracking a subset of individuals through time to estimate survival rates. GPS tracking periods spanned August 2015 – April 2016, and from February – August of 2016 – 2018.

Methods since 2019: We have not issued any new GPS collars, nor ear tagged any red fox due to both the mange-related mortality of foxes (see below) and the current objectives which do not necessitate collaring. We include previously used collaring and ear-tagging methods because we reference these results in later sections.

Mammal sign transects

Previously used methods: In 2015, we established 96 walking transects spanning Robert Moses State Park, Fire Island National Seashore, and Smith Point County Park (including Old Inlet East) subsites. We used a 300-m grid overlay to space transects c. 300 m from each other, following a general north-south direction across the island (Figure 42). The 300-m grid size was initially selected because Gieder (2015) found that red foxes on Assateague Island, MD, moved infrequently between camera stations spaced c. 300 m apart. Home range estimates based on concurrent GPS monitoring (Black 2021), suggest that this spacing was sufficient to ensure that each red fox home range encompassed ≥ 2 transects. These transects ran from the ocean high tide line through all passible habitats to the bayside high tide line, or the point where bayside transects became impassible due to deep marsh or dense shrub thickets. We walked each transect across the island approximately once per month in May – July and October/November – January/February in 2016 – 2018, ending in February 2019, for a minimum of three complete surveys during each spring/summer and fall/winter survey period for each year. These months



were chosen to fall within with the piping plover breeding and red fox denning and kit-rearing seasons (May, June, July) and outside of them (October – February).

During these transect surveys, we followed game trails, habitat edges, and other likely wildlife travelways along the guiding north-south transect lines and searched for red fox dens, mammal scat and tracks, and other signs of red fox use. We recorded GPS waypoints for all dens and scat found, and all red foxes encountered during surveys. We collected all potential red fox scat found that appeared to be ≤ 1 month old, based on visible biodegradation of fecal material (Table 25). We recorded the age class (i.e., adult or kit), and ear tag color and number (if one was visible) for all red foxes encountered during surveys. For all den sites, we recorded signs of recent use, such as tracks leading in and out of den entrances, scat and prey items in the vicinity, or red foxes observed nearby. In 2017 and 2018, we conducted additional transect surveys in late April/early May to locate breeding dens. During these surveys, multiple observers worked together to search all accessible areas for potential dens, in addition to searching along the game trails and transect areas typically surveyed during other months.

Near the end of the study period, from October 2018 – February 2019, we conducted monthly driving transect surveys along the main access road and outer beach between Kismet and Watch Hill. During these surveys, we recorded the GPS locations of all red fox tracks and live red foxes encountered along the survey route. We also solicited information on red fox sightings on the island, particularly in the villages in this area, from local National Park Service (NPS) staff and project collaborators.

Lab methods

Scat samples collected during transect surveys and opportunistically during den monitoring and trapping were individually bagged and stored in a 0° C freezer, then transported to a -20° C walk-in freezer for storage until DNA subsampling. For each sample, we separated a small portion (approximately 2 mL) of fecal material from the outside edges and tapered ends and submerged this portion in 95% ethanol for DNA preservation. These preserved subsamples were sent to the University of California, Davis Mammalian Ecology and Conservation Unit (MECU) for genetic analyses, including mitochondrial DNA sequencing to verify the source species (Aubry et al. 2009, Miles et al. 2015) and microsatellite genotyping for individual identification (Sacks et al. 2010, Sacks et al. 2011, Quinn et al. 2019). Tissue samples collected from dead red foxes were also sent for individual identification via microsatellite genotyping, to determine whether any GPS-collared foxes had been previously detected via scat sampling.

Modified methods for 2020: Mammal sign transects were not surveyed in 2020.

Methods since 2020: We resumed transect surveys in 2021. In 2021, we conducted two transect surveys (May – June and June). In 2022, we conducted one transect survey (August) and opportunistically recorded locations of predator sign (e.g., tracks, scat, hair, depredations, and carcasses of predators) during routine plover monitoring work as described earlier in the report. During opportunistic sampling, we record one location per set of tracks unless multiple sets of tracks are observed and tracks are ≥ 30 m apart, are oriented in different directions, or are clearly of different species or sizes. We collected all mammal scat visible during transect surveys, regardless of age (Table 25).

We dissected scat, searching for evidence of piping plover depredation (e.g., leg flags or plastic Darvic bands). For each sample, we separated a small portion (approximately 2 mL) of fecal material from the outside edges and tapered ends and submerged this portion in 95% ethanol for DNA preservation. In the event plover depredation was suspected, the preserved subsample would be sent to the University of California, Davis Mammalian Ecology and Conservation Unit (MECU) for genetic analyses as described above.



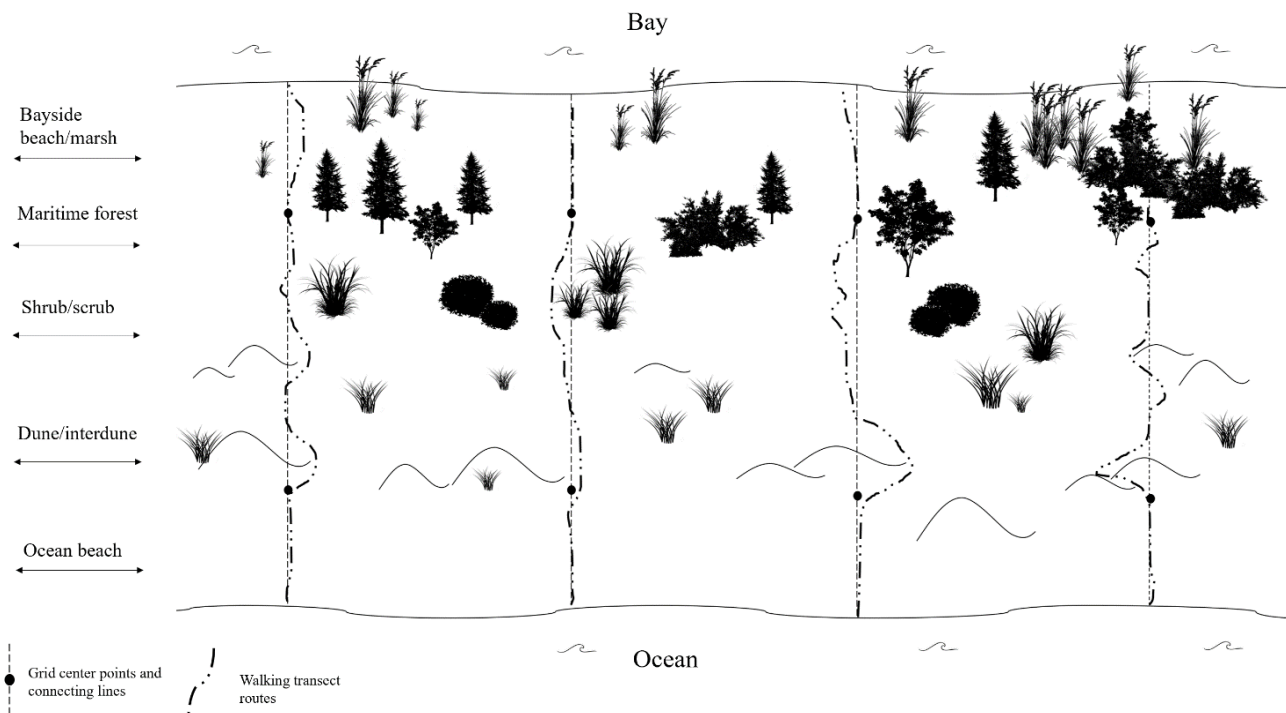


Figure 42. Habitat types and transect surveys conducted on Fire Island, NY. We followed game trails, edges, and other likely wildlife travelways along general north-south transect lines based on a 300-m grid overlay of the island. These transects spanned all passable habitat types within each c. 300 m section of the study area.



SHOREBIRDS

Table 25. Scat collection criteria used in sign, scat, and den surveys conducted on Fire Island. The use of ‘fecal matter’ refers to the unidentifiable digested waste material contained in a sample, as opposed to the identifiable prey remains.

Age category	Color	Moisture	Biodegrading	Notes
Very fresh (< 24 hours old)	Varied	Very moist	None	Observed a fox depositing the sample and/or fresh fox sign within 2 m, collect
Fresh (24 hours – a few days old)	Usually dark	Somewhat moist	None	Collect
Mid (≥ a few days old)	Usually somewhat dark, no sun bleaching	May be moist due to recent dew/humidity	None, outer fecal material intact	Collect Sample photo: 
Old (≤ 1 month old)	Usually lighter, may be sun bleached	Usually dry	Some, outer fecal material beginning to wear away but still present	Collect if outer fecal material is mostly intact Sample photo: 
Very old (> 1 month old)	Usually very light, sun bleached	Very dry	Significant, hardly any outer fecal material left	Do not collect*

*We did not collect very old scat 2015 – 2019; we collected all observed scat in 2021 and 2022.

Red fox den surveys

Previously used methods: We monitored all apparently active red fox dens from the time of discovery until at least 2 weeks after the last signs of active den use. Any newly discovered active dens were revisited within 1 week of discovery and re-assessed to determine the likelihood of current use based on tracks and other sign. Dens that appeared active during this second assessment were added to our ongoing den monitoring.

We visited active dens once every 2 weeks. During den monitoring visits, we collected scat and recorded and photographed prey remains in a c. 15 m radius around the den. Adult red foxes commonly bring prey items back to the den to provision their young; this radius encompassed the area where prey items were left. We also recorded any adult red foxes or kits in the area. To assist in determining the number of kits associated with each den, we set up one or more remote cameras (Moultrie M880i, EBSCO Industries, Calera, AL) aimed at den entrances and/or adjacent clearings, when feasible. We did not use remote cameras at dens in areas of high human use to avoid drawing attention to dens. Where cameras were used, we changed memory cards and batteries and adjusted camera orientation during den monitoring visits. Most red fox family groups used



SHOREBIRDS

multiple breeding dens throughout the breeding season. To account for these relocations in later analyses, we used remote camera photos and in-person observations of kits, and information on family group movements based on GPS-collared and ear-tagged red foxes, to classify dens as either natal dens (used early in denning season and in weeks following parturition, typically active through late May) or secondary dens (used later in the season, created or re-excavated after kits had emerged from dens, typically active starting in late May/early June).

Modified methods for 2020: Red fox den surveys were not completed in 2020.

Methods since 2021: We resumed den surveys, adhering to our previously used methods.

Camera trapping surveys

Previously used methods: In late September – January/February of 2015 – 2018, ending in February 2019, we used remote camera grids to survey for red foxes in each of the three study sites (Figure 43). We used the same 300-m grid cells and center points that we used to guide our transect surveys to guide remote camera placement, and we established an unbaited camera station within 50 m of the center point of each grid cell containing suitable sites. To minimize the risk of equipment damage and theft, we avoided setting cameras out on the open beach, in low-lying areas prone to flooding, and in areas of heavy human use. For similar reasons, we also excluded grid cells that were entirely composed of large parking lots and inundated marsh.

Each camera station consisted of two trail cameras (Moultrie M880c and M880i Gen2, EBSCO Industries, Calera, AL), each mounted on a plastic t-post and facing towards a game trail, clearing, or other potential wildlife travelway. We set cameras on opposite sides of the animal trail to photograph both sides of passing animals. Cameras were programmed to take three pictures per trigger, with a 5 s delay after each trigger. We returned every 10 – 14 days to switch memory cards, replace batteries as needed, and clear any vegetation growing directly in front of the camera. In years following the initial 2015 camera surveys, we set up each camera station in the same location used in previous years, or as close to it as possible if the previous year's location was no longer suitable. Thus, the exact locations and numbers of camera stations in each grid varied slightly among years due to habitat changes. Each grid was operational for a minimum of 1,200 trap nights each year, defined as the number of camera stations multiplied by the number of nights that at least one of the two cameras at each station were in operation.

Modified methods since 2020: Cameras have not been deployed.

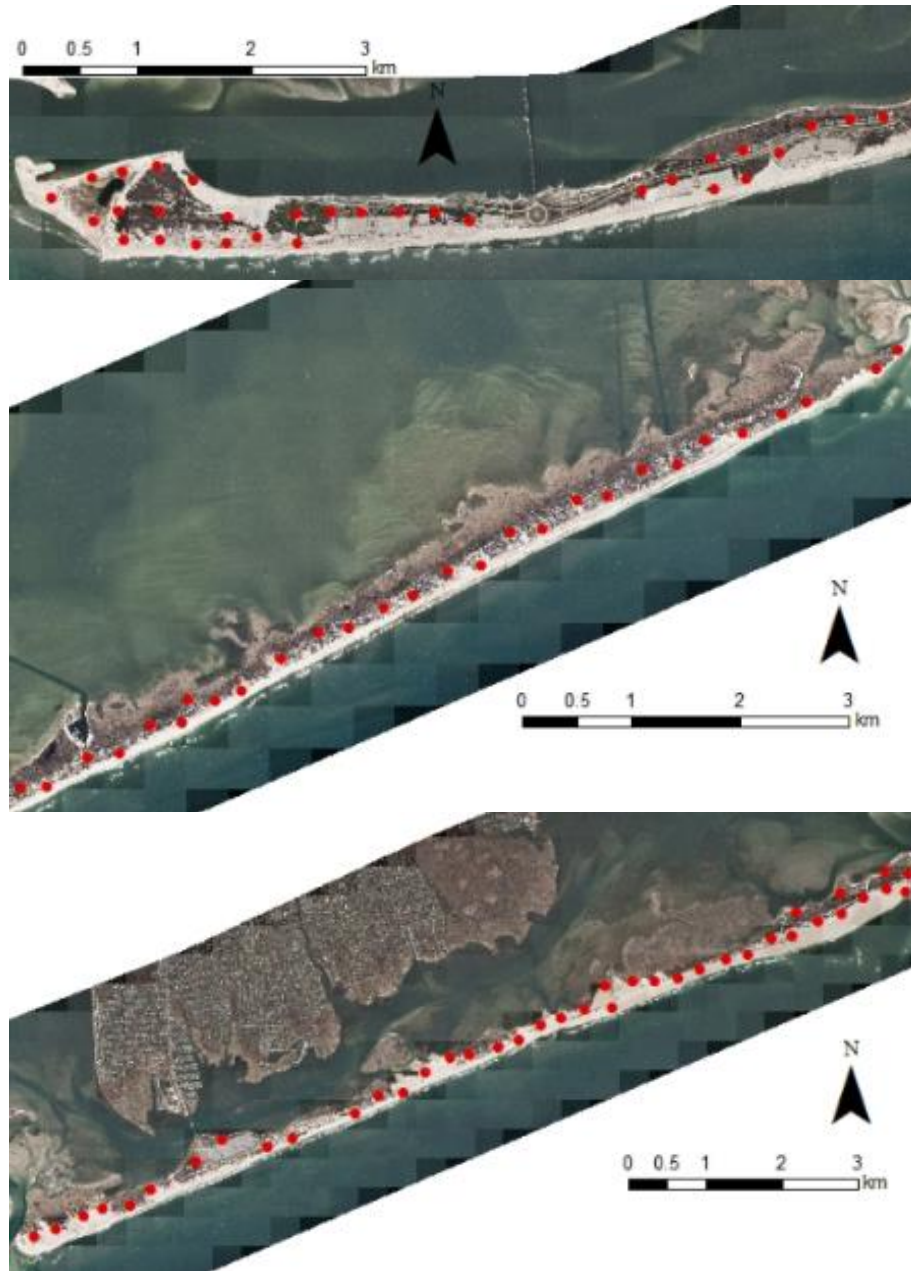


Figure 43. Approximate camera station locations (red dots) at Robert Moses State Park (top), Fire Island National Seashore (middle), and Smith Point County Park (including Old Inlet East; bottom) on Fire Island NY, 2015 – 2019. Remote camera surveys were conducted in September – February of each year, ending in February 2019. Exact locations and numbers of stations varied slightly among years due to changes in habitat that altered site suitability for camera setup.

Chapter 8. Predator numbers, density, and distribution on Fire Island, with an emphasis on red fox

Objective F1: Produce an index of fox abundance to understand the recovery rate of the fox population

Objective F2: Estimate red fox reproductive success and relate it to population growth during the recovery from the mange outbreak

Introduction

Information on population size and the factors influencing animal abundance is needed for wildlife management and conservation. For threatened and endangered species conservation, information on the population ecology of the threatened species and also on its important predators can help guide recovery efforts. In June 2015, we began monitoring a coastal red fox population as part of a study investigating red fox interactions with the federally threatened piping plover. Red foxes are key predators of the federally threatened piping plover along the U.S. Atlantic Coast, leading to frequent red fox management efforts aimed at reducing predation on these and other shorebirds (Hunt et al. 2019). Shortly after the start of data collection, a series of sarcoptic mange outbreaks began in the study area, presenting an unexpected opportunity to examine the effects of sarcoptic mange on red fox population dynamics in a North American coastal ecosystem.

Sarcoptic mange has been identified as a driver of red fox population dynamics in several parts of the world. In Bristol, UK, red fox population density decreased by > 95% following an outbreak (Baker et al. 2000). Similar declines in red fox populations following mange outbreaks were reported in Sweden (Lindström et al. 1994) and Spain (Gortázar et al. 1998). However, there is less information on the effects of sarcoptic mange in North American red fox populations, as much of the published literature about mange in red foxes coming from European studies (Niedringhaus et al. 2019). In the United States, Gosselink et al. (2007) reported lower survival of juvenile foxes during times when mange was present in Illinois red fox populations. Gosselink et al. (2007) also observed differing effects of mange in urban versus rural areas, with mange accounting for 45% of urban red fox mortalities but only 2% of rural red fox mortalities, potentially due to higher densities, increased social interactions, and fewer den sites in urban versus rural areas. Storm et al. (1976) described mange in red foxes in the midwestern United States, but reported that mange did not appear to be limiting red fox numbers in their study area, as other mortality causes were observed more frequently. Aside from these studies, detailed descriptions of mange in red fox populations in North America are relatively limited, and do not address red fox populations in coastal settings.

When considered as a proxy for lethal removal, the effects of such a disease-induced red fox population decline may help inform predator management for shorebird conservation. Here we quantify the impacts of a series of sarcoptic mange outbreaks on red fox population ecology on Fire Island, NY relative to mange outbreaks.

Methods

Using evidence collected from our remote camera grid, transect and den surveys, opportunistic resights, and reports from land managers and the public, we recorded the approximate timeline of mange outbreaks across Fire Island. Whenever possible we assigned the severity of mange based on symptoms presented (Table 26).



Table 26. Visual scale of sarcoptic mange symptoms observed in red foxes at Fire Island, expanding on Pence et al. (1983).

Rank	Description of symptoms	Example
0	No symptoms of visible mange. Assumed to be a healthy individual or an individual that is not yet exhibiting symptoms.	
1	Mild signs of mange, mostly limited to hyperkeratosis or alopecia of the face. ^a	
2	Mange spreading. Hyperkeratosis or alopecia on face and spreading to legs (and $\leq 5\%$ of visible torso). ^a	
3	Moderate infection. Hyperkeratosis, alopecia, or exposed gray skin spreading to tail (and $\leq 15\%$ of visible torso). ^b	
4	Worsening infection. Hyperkeratosis, alopecia, or exposed gray skin on c. 50% of visible body surface. ^b	
5	Severe infection. Visible mange symptoms on $\geq 50\%$ of visible body surface. ^c	

^a Approximates Class I (Pence et al. 1983): small infections (c. 1 cm²) of fore and hind limbs, ischium, and ears.

^b Approximates Class II (Pence et al. 1983): increased infection size ($\leq 50\%$ of body surface), knees and elbows.

^c Approximates Class III (Pence et al. 1983): infection size across most of the body ($> 50\%$ of body surface).

We quantified the number of adult red fox likely present at each study area on Fire Island using ear-tagging and GPS collar data (2016 – 2018 only), resighting of individuals at dens or during transect surveys, and scat evidence. In 2021, we estimated adult fox presence based on scat by developing a simple linear regression using data from 2015 – 2018 (when foxes were present). We used the number of scat in each study area confirmed to have been from fox (via genetic analysis) as our independent variable, and the number of adult fox known to reside in the study area as our dependent variable (Walker et al. 2020, Black 2021, Wails et al. 2021). We also summarize red fox breeding activity and reproductive output for each study site.

We compared the frequency of red fox and other mammal sign (specifically, tracks and scat) detected during transect surveys. As our surveys were conducted at two separate times (and thus represent mammal activity across different time periods), we compared the frequency of red fox, coyote, and feral cat tracks and scat per sampling period at each study site using Fisher's exact test (preferred when expected frequencies are ≤ 5 ; Zar 2013).

Lastly, we quantified red fox density per study area since 2016. We used our estimates of adult red fox in each study area and measured the total study area in ArcGIS (4.63 km² for Robert Moses State Park + Lighthouse Tract, 3.85 km² for the West Wilderness, and 5.94 km² for Smith Point County Park + Old Inlet East).

Analyses were conducted in the R statistical environment ver. 4.1.3 (R Core Team 2021).

Results

Red fox mange outbreaks

Sarcoptic mange was first observed in red foxes in early fall of 2015 east of Old Inlet, and first documented in a photograph that was taken by a visitor at Smith Point County Park, that was shared with us by local collaborators on 3 September 2015. We determined that the red fox in this photograph was exhibiting moderate-severe signs of mange (Table 26). Prior to that, we had not observed any red foxes with hair loss, facial crusting, skin lesions, or other signs that are characteristic of sarcoptic mange during shorebird monitoring activities conducted in the study area in 2013 and 2014 and a pilot red fox study conducted in June – August 2015 (A. L. DeRose-Wilson, *pers. comm.*). We first observed signs of mange in a red fox at Robert Moses State Park on 22 May 2017, signaling the beginning of a second outbreak on the island, this time west of Old Inlet. The first observed signs of mange at the West Wilderness study site were recorded in remote camera photos of a red fox taken there on 18 October 2017.

Red fox numbers and indices of abundance and distribution

Between 2015 – 2018, we captured and ear-tagged 41 red foxes. We outfitted 33 of these red foxes with GPS collars (Table 27), including two foxes that were recaptured and GPS-collared in multiple years for a total of 35 GPS location datasets. While we have not captured and tagged or collared new red fox since 2018, we have not found evidence of active breeding after 2018 (Table 27, Table 28) and other evidence of fox presence (e.g., sightings, scat) has been sparse (Table 29).

Our analyses demonstrated that frequency of fox scat sampled from transects was a representative index of fox presence ($F = 10.82$, $P = 0.02$). Based on the association of known fox scat and previously known fox populations (2015 – 2018), we conservatively estimate three adult red foxes were in the study area in 2022 (Table 27). There were single fox scat samples collected from Robert Moses and the West Wilderness. We observed one adult red fox in the East Wilderness, in the upper dunes near Old Inlet.



Table 27. Red fox abundances at each study site from 2016 – 2022 on Fire Island, NY. We estimated red fox abundance as the sum of GPS-collared adults, ear-tagged adults, unmarked (i.e., no ear tag nor collar) adults associated with each breeding group, kits associated with each breeding group, and any other known individuals observed during the census period (1 February – 1 August). Fox activity not monitored by Virginia Tech in a given year (i.e., 2020) was obtained from USFWS, NPS, New York State Parks, and Suffolk County Parks, and these data appear bold and italicized.

Site	Year	GPS-collared adults	Ear-tagged adults	Adults at dens	Kits	Other individuals	Estimated abundance
Robert Moses State Park + Lighthouse Tract	2016	6	0	5	14	0	25
	2017	8	2	5	13	1	29 ^a
	2018	2	1	1	0	1	5 ^b
	2019	0	0	0	0	0	0
	2020	0	0	0	0	0	0
	2021	0	0	0	0	1 ^d	1 ^d
	2022	0	0	0	0	1 ^d	1 ^d
West Wilderness	2016	5	1	11	18	0	35
	2017	4	3	4	2	0	13 ^c
	2018	0	0	0	0	1	1
	2019	0	0	0	0	0	0
	2020	0	0	0	0	0	0
	2021	0	0	0	0	2 ^d	2 ^d
	2022	0	0	0	0	1 ^d	1 ^d
Smith Point County Park + Old Inlet East	2016	0	0	0	0	0	0
	2017	0	0	0	0	1	1
	2018	0	0	0	0	1	1
	2019	0	0	0	0	0	0
	2020	0	0	0	0	0	0
	2021	0	0	0	0	0 ^d	0 ^d
	2022	0	0	0	0	1 ^e	1 ^e

^a 5 adults and 2 kits included in this count died during the census period

^b 2 adults included in this count died during the census period

^c 1 adult included in this count died during the census period

^d Estimated from analyses of scat frequencies from transect surveys

^e 1 apparently healthy adult seen by plover technicians during fieldwork

Red fox reproduction decreased in the breeding seasons following the detection of mange at each site (Table 28). At Robert Moses State Park and the Fire Island National Seashore Lighthouse Tract, the number of breeding groups ($n = 4$) stayed constant between 2016 and 2017, though the average litter size decreased by 20% (2016 mean $\pm$ SE: 3.5 ± 0.5 kits/den vs. 2017 mean $\pm$ SE: 2.8 ± 0.5 SE kits/den). Since 2017, we have not observed any successful reproduction by red fox. In May 2021, we located an inactive fox den at Smith Point. It is likely that this was an older den as it remained unused in 2021 and we did not detect further fox activity by the den during later surveys. There was no evidence of red fox breeding in 2022.

Table 28. Number of red fox dens and mean ($\pm$ SE) number of kits per breeding group for each site monitored on Fire Island, NY in April – August of 2016 – 2022. Counts include all kits that survived through the end of the kit-rearing period, using August 1 as the cut-off date. Breeding activity not monitored by Virginia Tech in a given year (i.e., 2020) was obtained from USFWS, NPS, New York State Parks, and Suffolk County Parks, and these data appear bold and italicized.

Year	Evidence of breeding	Robert Moses + FNS Lighthouse Tract	West Wilderness	NPS Old Inlet East + Smith Point	Total study area
2016	Active dens	4	4	0 ^c	8
	Kits	3.5 ($\pm$ 0.5)	3.6 ($\pm$ 0.9)	0 ^c	3.5 ($\pm$ 0.6)
2017	Active dens	4 ^a	2	0	6
	Kits	2.8 ($\pm$ 0.5) ^a	1 ($\pm$ 0.0)	0	2.2 ($\pm$ 0.5)
2018	Active dens	0 ^b	0 ^b	0	0
	Kits	0 ^b	0 ^b	0	0
2019	Active dens	0	0	0	0
	Kits	0	0	0	0
2020	Active dens	0	0	0	0
	Kits	0	0	0	0
2021	Active dens	0	0	0	0
	Kits	0	0	0	0
2022	Active dens	0	0	0	0
	Kits	0	0	0	0

^a Mange first detected at that site

^b During mange outbreak at that site

^c After mange outbreak at that site

Throughout 2022, we observed 261 signs of mammal activity during the island-wide transects. We had one instance where an observer saw an adult red fox (25 July 2022 at Old Inlet East, during routine plover monitoring work). We saw no evidence of breeding in the areas we monitored.

Tracks ($n = 215$, 82.4% of transect observations) and scat ($n = 46$, 17.6% of transect observations) represented all evidence of mammal activity observed during transect surveys; no red foxes were observed during transect surveys in 2022 (Table 29a, b). In addition to transect surveys, we opportunistically recorded 373 instances of mammal sign (tracks, scat, carcasses, and live sightings) during plover monitoring work (Table 29c, Table S4). Among both transects and opportunistic sampling, we detected evidence of feral cats most frequently.



Table 29. Total detections of canid and felid **(a.)** tracks and **(b.)** scat observed during transect surveys, and **(c.)** tracks and scat recorded opportunistically during plover monitoring work in 2022 across Fire Island, NY. Other predator includes species known or suspected to predate piping plover, but have not yet been documented depredating plovers or their nests at Fire Island (other category includes northern raccoon, Virginia opossum, American mink [*Neogale vison*], and North American river otter [*Lontra canadensis*]).

(a.) Tracks – transects

Site	Red fox	Coyote	Domestic dog	Feral cat	Other predator	Unknown	Total
Robert Moses + Lighthouse Tract	0 (0.0%)	1 (1.3%)	1 (1.3%)	23 (29.5%)	46 (59.0%)	7 (9.0%)	78
West Wilderness	1 (2.7%)	0 (0.0%)	0 (0.0%)	22 (59.5%)	14 (37.8%)	0 (0.0%)	37
Smith Point + Old Inlet East	0 (0.0%)	0 (0.0%)	0 (0.0%)	53 (98%)	44 (44.0%)	3 (3.0%)	100
Total	1 (0.5%)	1 (0.5%)	1 (0.5%)	98 (45.6%)	104 (48.4%)	10 (4.7%)	215

(b.) Scat – transects

Site	Red fox	Coyote	Domestic dog	Feral cat	Other predator	Unknown	Total
Robert Moses + Lighthouse Tract	1 (3.0%)	0 (0.0%)	0 (0.0%)	13 (39.4%)	6 (18.2%)	13 (39.4%)	33
West Wilderness	1 (12.5%)	0 (0.0%)	0 (0.0%)	4 (50.0%)	0 (0.0%)	3 (37.5%)	8
Smith Point + Old Inlet East	0 (0.0%)	0 (0.0%)	0 (0.0%)	0 (0.0%)	2 (40.0%)	3 (60%)	5
Total	2 (4.3%)	0 (0.0%)	0 (0.0%)	17 (37.0%)	8 (17.4%)	19 (41.3%)	46

(c.) Opportunistic evidence

Site	Red fox	Coyote	Domestic dog	Feral cat	Other predator	Unknown	Total
Robert Moses + Lighthouse Tract	7 (4.8%)	15 (10.2%)	14 (9.5%)	23 (15.6%)	67 (45.6%)	21 (14.3%)	147
West Wilderness	1 (1.2%)	0 (0.0%)	0 (0.0%)	33 (39.3%)	42 (50.0%)	8 (9.5%)	84
Smith Point + Old Inlet East	7 (4.9%)	2 (1.4%)	7 (4.9%)	37 (26.1%)	66 (46.5%)	23 (16.2%)	142
Total	15 (4.0%)	17 (4.6%)	21 (5.6%)	93 (24.9%)	175 (46.9%)	52 (13.9%)	373

Table 30. Total detections of canid and felid **(a.)** tracks and **(b.)** all evidence during transect surveys, across Fire Island, NY from 2016 – 2022. For a cumulative summary of scat collection, see Table 31. Other predator includes species known or suspected to predate piping plover, but have not yet been documented depredating plovers or their nests at Fire Island (other category includes northern raccoon, Virginia opossum, American mink [*Neogale vison*], and North American river otter [*Lontra canadensis*]).

(a.) Tracks – transects

Site	Red fox	Coyote	Domestic dog	Feral cat	Other predator	Unknown	Total
Robert Moses + Lighthouse Tract	182 (52.0%)	10 (2.9%)	0 (0.0%)	57 (16.3%)	50 (14.3%)	51 (14.6%)	350
West Wilderness	240 (69.8%)	8 (2.3%)	0 (0.0%)	34 (9.9%)	17 (4.9%)	45 (13.1%)	344
Smith Point + Old Inlet East	50 (12.9%)	0 (0.0%)	2 (0.5%)	118 (30.5%)	47 (12.1%)	170 (43.9%)	387
Total	472 (43.7%)	18 (1.7%)	2 (0.2%)	209 (19.3%)	114 (10.5%)	266 (24.6%)	1,081

(b.) All evidence – transects

Site	Red fox	Coyote	Domestic dog	Feral cat	Other predator	Unknown	Total
Robert Moses + Lighthouse Tract	522 (56.2%)	11 (1.2%)	0 (0.0%)	72 (7.8%)	61 (6.6%)	263 (28.3%)	929
West Wilderness	708 (66.0%)	8 (0.7%)	0 (0.0%)	36 (3.4%)	17 (1.6%)	304 (28.3%)	1,073
Smith Point + Old Inlet East	132 (22.4%)	0 (0.0%)	2 (0.3%)	119 (20.2%)	50 (8.5%)	286 (48.6%)	589
Total	1,362 (52.6%)	19 (0.7%)	2 (0.1%)	227 (8.8%)	128 (4.9%)	853 (32.9%)	2,591

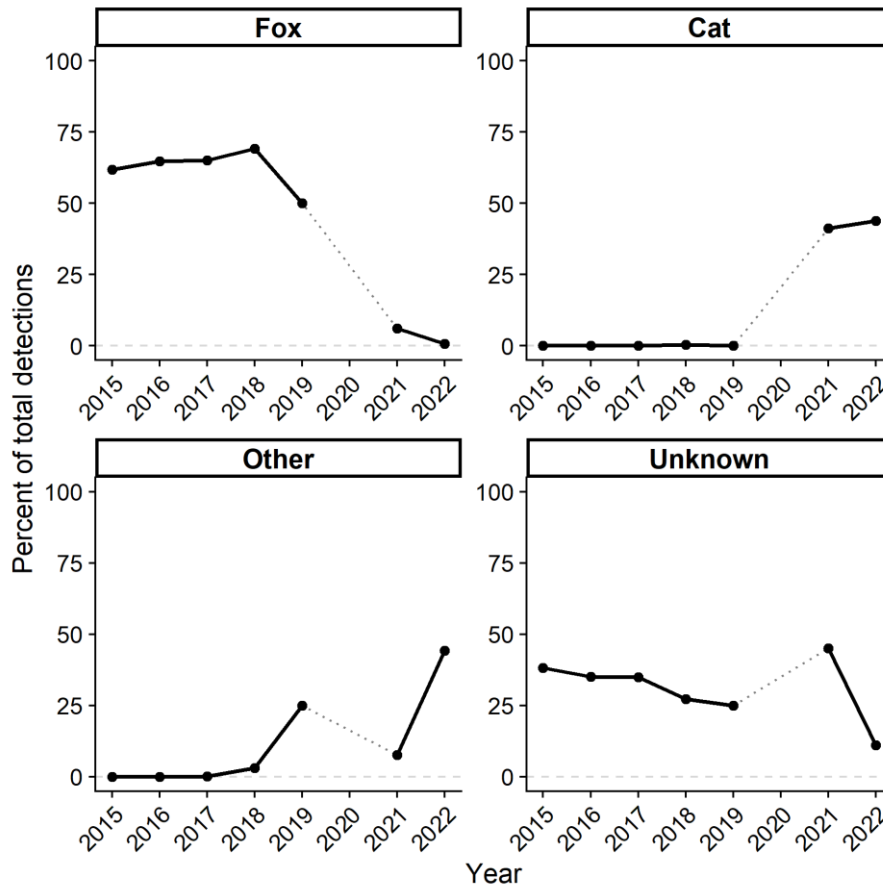


Figure 44. Change in the total annual detections per predator during transect sampling from 2015 – 2022 on Fire Island, NY. No transect surveys were conducted in 2020, thus the change between 2019 and 2021 is represented with a dashed line. Other predators include coyote, domestic dog, northern raccoon, Virginia opossum, American mink [*Neogale vison*], and North American river otter [*Lontra canadensis*].

Red fox densities

At Robert Moses State Park and the Fire Island National Seashore Lighthouse Tract, red fox abundance and density were highest in February – August 2017, with the first detected signs of mange observed roughly halfway through this census period (2017: 29 red foxes, 6.26 foxes/km², 4.63 km² total site area), and had decreased substantially by the 2018 census period (2018: 5 red foxes, 1.08 foxes/km²; Figure 45a, Table 27). Red foxes were not detected again at Robert Moses until 2021. In the West Wilderness, abundance and density were highest in February – August 2016, before the arrival of mange (2016: 35 red foxes, 9.09 foxes/km², 3.85 km² total site area), but decreased substantially afterwards (2018: 1 red fox, 0.26 foxes/km²; Figure 45b, Table 27). Similar to Robert Moses, red foxes were not observed in 2019 and 2020, though we began detecting their presence in 2021. At the start of red fox monitoring in 2016 at Smith Point County Park and Old Inlet East, we found no evidence of red fox presence (2016: 0 red foxes, 0 foxes/km², 5.94 km² total site area; Figure 45c, Table 27). Red foxes were detected in 2017 and 2018 (2017 and 2018: 1 red fox, 0.17 foxes/km²), but were rarely, if ever, detected in later years. As of 2022, we have detected some evidence that red foxes have, at minimum, transient presence across each of the three study areas. Anecdotal observations from the Sunken Forest on Fire



Island in August 2022 suggest breeding is likely occurring in the island interior (as evidenced by suspected red fox kit tracks; H. L. VanDerwater and M. L. VanDerwater, *pers. comm.*).

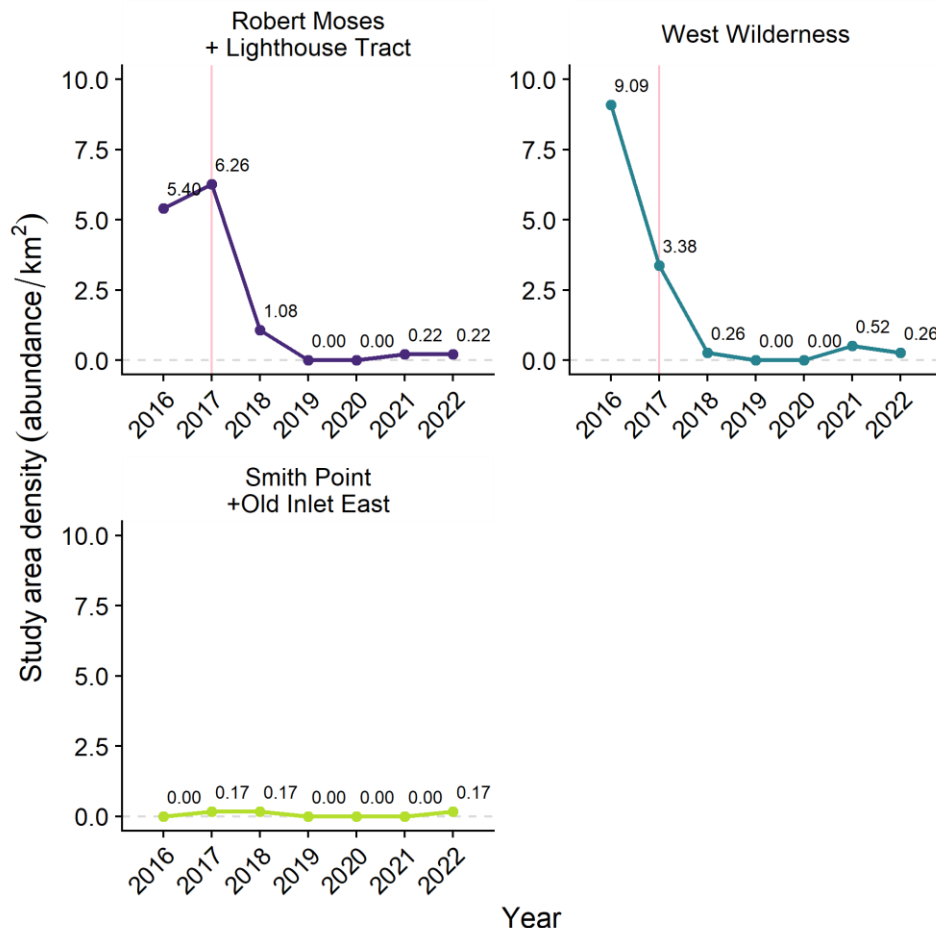


Figure 45. Estimated study area density of red fox for each year of fox monitoring at Fire Island, NY. Red fox density was calculated as the minimum red fox abundance (Table 27) per study site area (4.63 km² for Robert Moses State Park + Lighthouse Tract, 3.85 km² for the West Wilderness, and 5.94 km² for Smith Point County Park + Old Inlet East). Vertical dashed lines indicate the year mange was first observed; estimates of area density for Smith Point occurred after the onset of mange (2015) at that site.

Discussion

Our multi-year combination of monitoring GPS-collared and ear-tagged foxes, conducting transect surveys, den monitoring, remote camera surveys, and use of genetic analyses provide multiple lines of evidence indicating a dramatic decline in red fox relative abundance, annual reproduction, and minimum density related to mange outbreaks at each site. The observed proportion of fox deaths attributable to mange (57% of all known deaths) and population declines (complete mortality east of Old Inlet following the first outbreak, 83% and 97% decreases in minimum abundance and density estimates at sites west of Old Inlet following the second outbreak) previously observed at Fire Island (Walker et al. 2020, Black 2021, Wails et al. 2021) are on par with the dramatic mortality rates reported elsewhere for this species as a result of sarcoptic mange (Mörner 1992, Baker et al. 2000, Soulsbury et al. 2007).



SHOREBIRDS

The transect survey results for 2022 suggest that a slight increase in red fox sign though the potential trajectory for population recovery of red fox (and its future impact on nesting plovers) remains unclear. The recovery has been slower than we hypothesized (S. M. Karpanty, *pers. comm.*) based on known reproductive potential of red foxes. Despite the observed low abundance, reproduction, and density, there is typically high occupancy at sites when foxes are present (Walker et al. 2020, Black 2021). With the exception of the c. 1-year period where red foxes apparently were absent east of Old Inlet and in Smith Point County Park, previous occupancy estimates across all sites and years were all ≥ 0.80 , even for periods during which we believe that very few foxes were present (Walker et al. 2020, Black 2021). This suggests that foxes that persisted during and after mange outbreaks likely expanded their use areas into former territories of deceased foxes, in line with other studies (Baker et al. 2000, Potts et al. 2013).

When the fox mortality observed from mange is considered as a potential proxy for intensive lethal removal of foxes, our results suggest that even if removal activities could result in decreased fox abundance and density, they may not be effective in reducing the overall distribution of foxes unless complete eradication can be achieved. Previous studies have found that lethal removal of red foxes may not be effective at reducing occupancy in many cases. Although red fox occupancy on Assateague Island was low, it did not decrease despite annual red fox removal efforts (Gieder 2015). However, these removal efforts had been ongoing since 2009, before that study began, thus, they may not have captured any initial response to removal activities (Gieder 2015). In some cases, lethal removal may actually have the opposite effect, leading to increased occupancy via colonization of previously unoccupied sites and/or compensatory reproduction. For example, Towerton et al. (2011) found that foxes colonized previously unused sites following lethal removal. In addition, previous studies of red fox population responses to lethal removal call into question whether removal efforts would succeed in reducing the overall population size, regardless of occupancy levels, due to compensatory immigration (Greentree et al. 2000, Harding et al. 2001, Baker and Harris 2004).

Increased reproduction among remaining individuals also has been identified as a potential factor counteracting lethal removal efforts. For example, Marlow et al. (2016) observed significantly larger litter sizes following artificial decreases of c. 26% in fox density compared to litter sizes at a control site. We did not observe such increases after the mange-related population declines in the study area during the time period of the study (i.e., 0 – 2 years of post-mange data). Instead, we observed complete reproductive failure in breeding seasons following mange outbreaks at each site, likely related to the low number and poor condition of any remaining foxes. We also observed fewer breeding groups and fewer kits per litter in the West Wilderness in 2017, prior to the arrival of mange at the site, suggesting that factors other than mange likely influenced reproduction on the island as well. Mange has previously been shown to reduce the proportion of individual foxes breeding within a population. In their study of the impacts of mange on a population of foxes in Bristol, UK, Soulsbury et al. (2007) found no evidence of breeding among severely affected females, and severely affected males apparently failed to produce sperm during the breeding period. Interestingly, mange may not affect the reproductive success of animals that are still able to reproduce during and after outbreaks, as Soulsbury et al. (2007) found that average litter sizes remained comparable among pre-mange, epizootic, and enzootic periods. As we continue to monitor the recolonization and recovery of red foxes on Fire Island, we may still detect evidence of compensatory reproduction in future years, but no evidence of breeding in the study area has yet been detected.

Red fox occupancy may remain high even after substantial decreases in abundance and population density following mange outbreaks. When considered as a proxy for intensive lethal removal, these results highlight the idea that, while such removal efforts may reduce predator abundance and population density within a given area, they may not be effective in reducing the overall distribution of red foxes or eliminating the risk of predation by red foxes in a given area. However, given our finding that higher relative abundance of red foxes



pre- vs. post-mange (e.g., as measured by trap success on cameras) was negatively related to plover chick survival (Robinson et al. 2020, Catlin et al. in prep), there may be some benefit to a reduced fox population with respect to plover chick survival. However, the higher chick survival that was correlated with lower red fox relative abundance seems to have been lost due to a variety of possible other causes, as described in earlier sections of this report.

In general, predator disturbance has been shown to increase the probability of nest abandonment (Doherty and Heath 2011), thus, even in areas implementing nest exclosures, predators can cause nest failure through harassment and predation of incubating adults (Frid and Dill 2002, Roche et al. 2010). Since the mange in the red fox population, we have detected an increase in feral cat activity across Fire Island (Wails et al. 2021, this report). Globally, cats are recognized as a significant threat to island vertebrates (Nogales et al. 2004, Bonnaud et al. 2011, Medina et al. 2011), responsible for population reduction and extirpation in prey species (Crooks and Soulé 1999, Winter and Wallace 2006, Longcore et al. 2009, Herrera et al. 2022). In recent years of our study, we have recovered plover bands from cat scat (Walker et al. 2020, Wails et al. 2021) and observed instances of cats taking adult plovers from exclosed nests, chicks within reach inside of exclosures, and mobile chicks (Wails et al. 2022b, Tables S2 – S4). We believe that feral cats currently are the primary mammalian predator of piping plovers at Fire Island and their increasing relative abundance is a management concern.

Chapter 9. Dietary ecology of red foxes

Objective F3: Quantify the interactions of red foxes with shorebird nests and chicks

Introduction

The dietary ecology of red foxes has been studied extensively, spanning a variety of geographic locations and ecosystems (Ables 1975, Lloyd 1980, Henry 1986, Soulsbury et al. 2010). In general, red foxes are opportunistic omnivores that will eat a wide range of items including fruit and plant matter, insects, rodents, lagomorphs, small birds, fish, and other freshwater or marine organisms when available, and will also scavenge upon carcasses of larger animals. Relative availability and ease of attainment influence the proportions of red fox diets comprised of specific items (Ables 1975, Lloyd 1980).

Despite the wealth of information on red fox diets in the existing literature, there is relatively little information on red fox diets in North American coastal ecosystems, particularly on the U.S. Atlantic Coast and on barrier islands. Previous research suggests that small mammals and birds comprise a significant portion of red fox diets in these systems. In a study of coastal red fox diets on Assateague Island, MD, Krim et al. (1990) reported eastern cottontail rabbits (*Sylvilagus floridanus*), vole (*Microtus* sp.), mouse (*Peromyscus* sp.), and other mammal remains occurring most frequently in collected scat (87% of samples), with evidence of frequent crustacean (64.8%) and bird (46.3%) consumption as well. On Fire Island, NY, Peterson et al. (2021) found remains of birds (39%) and cricetid rodents (37.8%) most frequently in scat samples.

Red fox diets in developed coastal areas may be influenced by anthropogenic food availability. Anthropogenic foods may include scavenged or intentionally fed scraps, food left out for pets, produce from gardens and compost heaps, and waste material from unsecured trash containers; these foods have been shown to be the most common items in red fox diets in some developed areas (Soulsbury et al. 2010). Where fishing occurs, discarded bycatch also may be available, allowing red foxes to access aquatic food sources more easily. For example, Krim et al. (1990) frequently observed skate (Family Rajidae) and bluefish (*Pomatomus saltatrix*) remains outside of red fox dens on Assateague Island and hypothesized that they were scavenged from angler discards. Access to anthropogenic food sources has been shown to increase the abundance of red foxes and other mesopredators, leading to increased predation on prey and even potentially altering prey behavior (Shapira et al. 2008, Newsome et al. 2015). This increase in predation can be particularly problematic when threatened and endangered species are affected (Gompper and Vanak 2008).

Red foxes are known to depredate federally threatened piping plover nests at many sites in their Atlantic Coast breeding range (Rimmer and Deblinger 1990, Patterson et al. 1991, Melvin et al. 1992, Cohen and Fraser 2010). As a result, extensive lethal and nonlethal red fox control efforts are conducted in piping plover nesting areas along the U.S. Atlantic Coast (Hunt et al. 2019). However, it is unknown whether plovers and other shorebirds make up a large proportion of red fox diets in these areas. Although Krim et al. (1990) found bird remains in 46.3% of scat samples collected in May – August on Assateague Island, when piping plovers and other shorebirds nest on the island, the authors did not report a bird remains by taxa. Peterson et al. (2021) did not observe any piping plover remains in red fox scat collected along boardwalks on Fire Island in 2011 – 2012, but the Fire Island piping plover population has increased substantially since the years of their scat collection (Walker et al. 2019, Weithman et al. 2019, Robinson 2020, Robinson et al. 2020a), likely increasing relative availability of this species across the landscape. As many piping plover breeding areas along the Atlantic Coast experience heavy seasonal human use, it is important to consider whether anthropogenic food resources might be subsidizing populations of red foxes and other predators in these settings. If so, these anthropogenic food



resources could be indirectly influencing predation pressure on piping plovers and should be considered in management planning.

The goals of this monitoring were to investigate red fox dietary ecology on Fire Island, NY in light of increasing piping plover numbers and to explore the effects of anthropogenic food resources on the Fire Island fox population. We also sought to expand upon the work conducted by Peterson et al. (2021) by characterizing red fox diets on the island using scat collected across additional habitat types and geographic areas.

Methods

Pre-2020

We freeze-dried scat samples for a minimum of 48 hr to remove any remaining moisture, then separated out and sorted contents by hand. We identified contents to the lowest taxonomic level possible through comparison to a variety of resources, including field guides, gray literature, published reference materials (including Spiers 1973, Debelica and Thies 2009), and museum specimens. We used a dissecting microscope to assist with identification of scat contents as needed, and we used a digital compound light microscope to search for and identify hairs and feathers.

In previous years, we visually separated hairs and feathers into groups based on morphological similarity and examined 3 – 6 representative samples from each group. For hairs, we examined the internal medulla and external cuticular scales, selecting both guard hairs and underfur when present (Debelica and Thies 2009). We then compared these hairs to a hair microscopy guide for mammal species found in the study area, created using hairs from known identity specimens in the Virginia Tech Department of Fish and Wildlife Conservation's Natural History Museum (Black et al. 2020), and other pre-existing hair microscopy guides (Spiers 1973, Debelica and Thies 2009). For feathers, we focused our examinations on the barbs and barbules of plumulaceous feathers, as those sections contain the most useful diagnostic characteristics (Dove and Koch 2011).

Post-2020

Since 2020, we dissected scat and searched for evidence of depredation of banded birds (specifically, of piping plovers). We preserved samples so that future genetic or hand-sorting identification of species could be completed.

Results

In total, we observed 476 scat samples, of which we collected 463 samples (97.3% of all observed scat). We estimate that we have collected 262 potential red fox scat samples while monitoring dens ($n = 53$), during transect surveys ($n = 183$), and opportunistically ($n = 26$), representing 56.6% of the 463 scat samples collected since 2016 (Table 31). We previously collected DNA subsamples from 404 of the scat samples for genetic verification of source species, and preserved 46 subsamples from 2022 for future genetic testing. The remaining samples were not suitable for genetic analyses due to age, biodegradation, or damage sustained during field transport.



Table 31. All scat samples and suspected contributor species collected at Fire Island, NY 2016 – 2022 (excluding 2022). Other predator includes species known or suspected to predate piping plover, but have not yet been documented depredating plovers or their nests at Fire Island (other category includes northern raccoon and Virginia opossum).

Species	Sampling method			Total
	Den survey	Transect	Opportunistic	
Coyote	0 (0.0%)	10 (71.4%)	4 (28.6%)	14
Dog	0 (0.0%)	2 (100.0%)	0 (0.0%)	2
Red fox	53 (20.2%)	183 (69.8%)	26 (9.9%)	262
Unknown canid	0 (0.0%)	8 (66.7%)	4 (33.3%)	12
Cat	0 (0.0%)	80 (83.3%)	16 (16.7%)	96
White-tailed deer	0 (0.0%)	2 (100.0%)	0 (0.0%)	2
Other predator	0 (0.0%)	8 (100.0%)	0 (0.0%)	8
Unknown	1 (1.3%)	65 (81.3%)	14 (17.5%)	80
Total	54 (11.3%)	358 (75.2%)	64 (13.4%)	476

Fecal dietary synopsis

We did not find any identifiable piping plover remains (e.g., bands) in any mammalian scat samples collected in 2022.

Discussion

In previous years of this study, red fox diets on Fire Island were highly varied and included a wide range of natural and anthropogenic food items (Walker et al. 2020, Black 2021). On Assateague Island, MD, Krim et al. (1990) found mammal remains in 87% of scat samples collected in May – August, compared to our previous observation of mammal remains in 78% of scat samples collected during the Spring/Summer surveys, which were conducted in April – August (Walker et al. 2020, Black 2021). However, our observed frequencies of occurrence for species differed. At Assateague, eastern cottontail rabbit, meadow vole (*Microtus pennsylvanicus*), and white-footed mouse (*Peromyscus leucopus*) were the most frequent mammal species represented, occurring in 44.4%, 31.5%, and 24.1% of scat sample, respectively, and they did not identify any other mammal remains to species (Krim et al. 1990). For comparison, white-tailed deer (*Odocoileus virginianus*), unidentified voles (*Microtus* sp.), and cottontail rabbits were the most common mammalian prey species in we detected, occurring in 12%, 9%, and 8% of scat (Walker et al. 2020, Black 2021, Black et al. In Revision). Although we found a significant proportion of mammalian remains, particularly bone fragments and hairs, many could not be identified further (28% of samples). While cottontail rabbits, meadow voles, and white-footed mice were among the mammalian prey recorded in our study, each of these species occurred in $\leq 8\%$ of the scat samples



we collected. In their previous study of fox diets on Fire Island, Peterson et al. (2021) found voles in 32.2% of samples collected throughout 2011 and 2012, a frequency more similar to that reported by Krim et al. (1990). They also found cottontail rabbit remains in 15.6% of samples and white-footed mice remains in 4.4% of samples, roughly twice as often as we found these species in samples collected in 2016 – 2022.

A major difference between our study and the two prior studies of red fox diets on barrier islands is the lower frequency at which we found bird remains in scat samples. Krim et al. (1990) and Peterson et al. (2021) reported bird remains in 46.3% of scat samples and 43.3% of scat samples, respectively, compared to our previous observation of bird remains in only 35% of samples (Walker et al. 2020, Black 2021). Although this difference is not that large, the contrast becomes more apparent when broken down by taxonomic Order and Family. Similar to Peterson et al. (2021), we found passerine birds remains most often in scat, but at a much lower frequency, with their study reporting a 22.2% of samples containing passerine remains compared to the 6% of samples in which we observed remains from this order. As with their study, Anseriformes and Charadriiformes were the most next commonly found bird orders, including the suspected swan from 2021. Krim et al. (1990) did not report a further taxonomic breakdown in their results. The high proportion of bird remains that we were unable to identify further (24% of samples) may have contributed to these discrepancies in results between the two studies.

Despite likely red fox predation of nests at Smith Point County Park in 2015 (Monk et al. 2015) and observed red fox disturbances at nest enclosures in 2018 (Hermanns et al. 2018), we have not found any clearly identifiable piping plover remains in fox scat during our study. Moreover, we have found relatively few remains identifiable to Family Charadriidae, in our examination of red fox scat (Walker et al. 2020, Black 2021). In addition to piping plovers, two other members of Family Charadriidae (Semipalmated Plovers [*Charadrius semipalmatus*] and Black-bellied Plovers [*Pluvialis squatarola*]) are present in the study area during the month (June) in which those samples were collected (Monk et al. 2020). Previously, we found items resembling egg shells in three scat samples, but could only confidently identify them as bird egg shell remains in one instance (Walker et al. 2020, Wails et al. 2021). Those particular egg shell remains were blue in color, ruling out the possibility of them having come from piping plovers.

Peterson et al. (2021) similarly found no piping plover remains in suspected red fox scat samples collected in 2011 – 2012. This lack of remains in scat suggests that red fox predation of piping plover adults and chicks is opportunistic and variable across years, as opposed to habitual, targeted hunting. It also suggests that increases in piping plover abundance on the island in between the two studies have not translated to increases in plover predation events as detectable in scat. For context, the number of breeding piping plover pairs on Fire Island has more than tripled since Peterson et al. (2021) and the conclusion of this study, with 26 breeding pairs in 2013 and 116 breeding pairs in 2022. While direct predation by red foxes on piping plovers may not be a frequent occurrence on the island, infrequent predation still may impact the plover population. In addition, indirect effects of red fox presence and disturbance may have sub-lethal negative effects on piping plovers. For example, piping plover chick survival during this study, based on concurrent monitoring, was negatively correlated with red fox trap success (Robinson 2020). Previously, (Doherty and Heath 2011) found the number of red fox tracks counted on nearby transects to be the strongest predictor of piping plover nest abandonment at several sites across Long Island, NY, including areas we examined in our study.

While the analysis of non-fox samples was limited in scope, the past discovery of adult piping plover leg bands in a feral cat scat sample highlights the potential impacts of free-ranging cats in shorebird nesting areas and the potential need for including feral cats in predator management considerations (Walker et al. 2020, Black 2021, Wails et al. 2021). Both nest abandonment and loss of chicks have been attributed to feral cats previously on



Fire Island (Winter and Wallace 2006) and the threat appears to be ongoing (Wails et al. 2021, Wails et al. 2022b, Tables S2 – S4). Moreover, on adjacent Westhampton Island, feline predation of piping plovers and nest abandonment have been reported (Winter and Wallace 2006, Cohen et al. 2009a).

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Appendices



Appendix 1. Supplemental methods

General geospatial processing

Each year, a raster of classified habitat types was created from annual high-resolution (15 cm) aerial imagery with four bands (R, G, B, and NIR), using ArcGIS Desktop 10.6 Interactive Supervised Classification (Maximum Likelihood Classification). Training samples were selected for dry sand, vegetation, wet sand, and water. Interactive Supervised Classification is an iterative process which allows refinement of the classification results. Even with this, there was a great deal of overlap in band reflectance of vegetation, water, wet sand, dark sand, and shadows. The human brain is adept at perceiving whether a pixel classified as water is actually the shadow of a tree but numerical models, as yet, are not capable of making those distinctions. In order to use these habitat classifications for modeling, various methods in ArcGIS were used to reclassify pixels which were misclassified in the initial classification of the whole island, for example, pixels classified as vegetation on the beachfront or pixels classified as wet sand or water in dry areas of the island interior.

In order to reclassify large, uniform classes on the landscape (e.g., beachfront dry sand), we created a polygon shapefile over the area. A value field was added to the shapefile attribute table and given a numeric value equal to the desired class. The polygon was converted to a .tiff raster of the desired class by using the Polygon to Raster tool and then merged to the habitat classification. A second method to correct misclassified pixels was to clip an area of the original classified raster (Clip) and then reclassify certain pixel classes (Reclassify). This method was used to clip the interior of the island where water and wet sand are known to be absent and reclassify those pixels as vegetation.

In addition to correcting and masking misclassified pixels, a raster of water and intertidal sand was classified from DEM (digital elevation model) data on the ocean side of the island and bayside overwashes. The oceanfront intertidal range was 0.00 – 4.11 ft in elevation, based on the Astronomical High Water at NOAA Moriches Inlet tidal station. The bayside intertidal range was 0.00 – 1.46 ft in elevation, based on mean high water at NOAA Smith Point Bridge tidal station. We reclassified each DEM to select pixels within both the oceanfront and bayside intertidal and subtidal range. These pixels were reclassified with numeric values to represent intertidal wet sand and water, respectively. The refined layers from each of these methods were superimposed over the original maximum likelihood classification, masking the misclassified pixels.

Appendix 2. Supplemental tables

Table S1. Predicted probability of plover **(a.)** nest survival to hatch and **(b.)** chick survival to fledging for all four management units in 2013 – 2022 (excluding 2020). Nest survival to hatch (24 days from initiation) is estimated from a logistic exposure model, estimating the interactive effect of year and management unit on daily survival, and then raised to the 33rd power. Chick survival to fledge (25 days) is estimated from a young survival model.

Year	Management unit	(a.) Nest survival					(b.) Chick survival			
		Nests (n)	Prop. exclosed	Survival estimate	Lower 95% CI	Upper 95% CI	Broods (n)	Survival estimate	Lower 95% CI	Upper 95% CI
2013	FINS	7	0.86	0.67	0.29	0.95	5	0.17	0.05	0.35
	SPCP	9	0.44	0.27	0.06	0.58	3	0.41	0.11	1.00
	CBCP	4	1.00	0.70	0.20	0.99	3	0.12	0.01	0.38
2014	FINS	6	0.50	0.51	0.17	0.86	3	0.63	0.28	1.00
	SPCP	9	0.56	0.50	0.22	0.79	4	0.83	0.59	1.00
	CBCP	5	0.80	0.78	0.36	0.99	4	0.50	0.23	1.00
2015	RMSP	10	1.00	0.63	0.35	0.88	5	0.60	0.30	1.00
	FINS	4	0.00	0.57	0.16	0.92	2	0.99	0.90	1.00
	SPCP	23	0.30	0.12	0.04	0.25	2	0.62	0.22	1.00
	CBCP	11	0.55	0.59	0.29	0.86	7	0.14	0.03	0.32
2016	RMSP	11	0.91	0.70	0.40	0.92	8	0.43	0.24	0.65
	FINS	9	0.78	0.77	0.46	0.97	7	0.79	0.56	1.00
	SPCP	14	0.71	0.84	0.60	0.98	12	0.82	0.65	1.00
	CBCP	6	0.83	1.00	0.95	1.00	6	0.30	0.12	0.54
2017	RMSP	12	0.83	0.45	0.19	0.73	6	0.86	0.65	1.00
	FINS	14	0.86	0.71	0.45	0.91	10	0.50	0.33	0.69
	SPCP	24	0.79	0.77	0.58	0.92	19	0.57	0.42	0.79
	CBCP	11	0.36	0.36	0.12	0.67	4	0.52	0.22	1.00
2018	RMSP	19	0.95	0.71	0.48	0.89	14	0.77	0.60	1.00
	FINS	13	0.92	0.75	0.49	0.94	10	0.77	0.58	1.00
	SPCP	29	0.79	0.65	0.47	0.82	20	0.73	0.58	1.00
	CBCP	7	0.86	0.86	0.54	1.00	6	0.75	0.48	1.00
2019	RMSP	25	0.88	0.66	0.46	0.83	17	0.64	0.50	0.86
	FINS	29	0.86	0.68	0.49	0.84	21	0.78	0.62	1.00
	SPCP	35	0.74	0.73	0.56	0.87	27	0.52	0.40	0.73
	CBCP	11	0.46	0.57	0.26	0.85	8	0.32	0.17	0.53
2021	RMSP	23	0.96	0.75	0.54	0.91	17	0.46	0.32	0.65
	FINS	23	0.87	0.66	0.44	0.84	13	0.50	0.32	0.85
	SPCP	23	0.65	0.72	0.52	0.88	13	0.47	0.32	0.65
2022	RMSP	36	0.97	0.69	0.51	0.84	26	0.39	0.28	0.55
	FINS	46	0.91	0.64	0.48	0.79	30	0.61	0.47	1.00
	SPCP	23	0.96	0.74	0.52	0.90	23	0.39	0.27	0.59



Table S2. Observations of all predator and human activity at or around piping plover nests during the 2022 field season on Fire Island, NY and updated from Wails et al. (2022b). Subsites are listed west to east (Table 2). Nests that were not exclosed are denoted with an asterisk (*) next to the management nest number. Evidence includes date of observation (month/day) and a description of the predator sign. Table continues on the following page.

Management unit	Subsite	Nest number (mgmt. nest number)	Nest fate	Date and predator evidence observed
Robert Moses State Park ¹¹	STP	207 (02A)	Hatched	6/11: Raccoon tracks outside exclosure
	FLD3	202 (03A)	Failed - Abandoned	5/23: Suspected raptor depredation of nearby AMOY adult and nest
	FLD4	208 (08A)	Hatched	5/24: Adults defending from fish crow; 6/11: Raccoon tracks outside exclosure
	FLD5	210 (10A)	Failed - Other	5/24: Vandalized by humans, dog tracks present
National Park Service	FIL	203 (13A)	Failed - Other	5/26: Vandalized by humans on 5/24, dog tracks present
	FIL	205 (14A)	Hatched	6/07: Fox tracks circling exclosure, 6/11: Unknown avian tracks outside exclosure
	OIWR	20 (37A)	Hatched	6/21: Cat attacked chick ¹²
	OIWR	28 (44A)	Hatched	6/21, 6/30: Gull tracks circling exclosure
	OIWR	39 (45A)	Hatched	6/30: Gull tracks circling exclosure
	OIWR	40 (57A)	Hatched	6/30: Gull tracks circling exclosure, exclosure dented
	OIWR	43 (56A)	Hatched	6/30: Cat tracks circling exclosure
	OIE	4 (04A)	Failed - Abandoned	6/04: Raccoon tracks outside exclosure
	OIE	8 (23A)	Hatched	6/07: Fox tracks circling exclosure
	OIE	17 (29A)	Hatched	6/17: Raccoon tracks outside exclosure
	OIE	21 (36A)	Hatched	6/04: Raccoon tracks outside exclosure
	OIE	23 (53A)	Failed - Unknown	6/04: Raccoon tracks outside exclosure

¹¹ Additionally, at least two instances of take by peregrine falcons was observed at Democrat Point. VTSP researchers recovered plover legs and color bands from peregrine pellets at a known feeding roost. VTSP researchers were unable to determine which broods lost chicks to falcon depredation. VTSP researchers filed incident observation reports on 8 July and 13 July.

¹² On 21 June, VTSP researchers at the NPS Western Wilderness Area were capturing recently hatched brood 20 (37A) for routine banding. During capture, an injured plover chick was observed in the vegetation. The chick was bleeding and alive, but weak and unresponsive. The area had numerous cat tracks intermixed with adult plover tracks surrounding the injured plover chick. VTSP researchers believe they interrupted a cat attack, and reported to NPS staff and filed an incident observation report.

Management unit	Subsite	Nest number (mgmt. nest number)	Nest fate	Date and predator evidence observed
National Park Service (cont.)	OIE	29 (48A)	Failed - Abandoned	6/16: Gull depredation of adult ¹³ , gull tracks circling exclosure
	OIE	34 (52A)	Hatched	6/16, 6/24: Gulls on top of exclosure, tracks circling exclosure
	OIE	44 (59A)	Hatched	6/24: Gull tracks circling exclosure
	BRO	35 (51A)	Hatched	6/28: Ghost crab burrow inside exclosure
Smith Point County Park	NBW	119 (05B)*	Hatched	5/24: Raccoon tracks by nest
	NBE	139 (33A)*	Failed - Predated	6/13: Cat tracks by nest, four eggs depredated
	SPTW	116 (15A)	Hatched	6/13: SP reported cat depredation of hatching nest
	SPTW	126 (19A)	Hatched	6/28: Cat tracks circling exclosure
	SPTW	134 (28A)*	Failed - Predated	6/04: SP reported predation, suspected cat
	SPTW	135 (31A)	Failed - Abandoned	6/25: Suspected cat, plover wing found after adult depredation
	SPTW	142 (35A)	Failed - Abandoned	6/28: Raccoon tracks outside exclosure
	SPTE	107 (10A)	Hatched	6/13: Cat depredation
	SPTE - GG	121 (24A)*	Failed - Predated	5/29: SP reported cat predation
	SPTE - GG	122 (25A)	Hatched	6/14: Gull tracks circling exclosure
	SPTE - GG	129 (14B)*	Failed - Predated	6/06: Likely mammal, unclear which species
	SPTE - GG	130 (27A)	Failed - Predated	6/06: Avian tracks outside exclosure
	SPTE - GG	144 (27B)*	Failed - Predated	6/25: SP reported cat predation

¹³ On 16 June, VTSP researchers at the NPS Eastern Wilderness Area located a deceased and depredated piping plover outside of its nest exclosure. A mixed flock of 12 herring and great black-backed gulls were present. VTSP researchers reported to NPS staff and filed an incident observation report

Table S3. Outcomes for nests with observed predator activity at or around the nest during the 2022 field season on Fire Island, NY and updated from Wails et al. (2022b). Subsites are listed west to east (Table 2). Nests that were not exclosed are denoted with an asterisk (*) next to the management nest number. Evidence includes date of observation (month/day) and a description of the predator sign. Table continues on the following page. text

Management unit	Subsite	Nest number (mgmt. nest number)	Nest fate	Outcome
Robert Moses State Park	STP	207 (02A)	Hatched	GF(7JM) and GF(TUX) successfully fledged 4/4 chicks.
	FLD3	202 (03A)	Failed - Abandoned	GF(AP5) and GF(X80) renested and successfully fledged 2/3 chicks.
	FLD4	208 (08A)	Hatched	GF(375) and GF(226) successfully fledged 4/4 chicks.
	FLD5	210 (10A)	Failed - Other	GF(237) and unbanded renested and successfully fledged 1/4 chicks.
National Park Service	FIL	203 (13A)	Failed - Other	GF(51T) and unbanded renested, hatched 2/4 eggs, and successfully fledged 2/2 chicks.
	FIL	205 (14A)	Hatched	GF(27P) and GF(PP1) successfully fledged 3/4 chicks.
	OIWR	20 (37A)	Hatched	Two unbanded adults successfully fledged 3/4 chicks, 1 chick lost to cat depredation.
	OIWR	28 (44A)	Hatched	GF(3UX) and unbanded successfully fledged 2/3 chicks.
	OIWR	39 (45A)	Hatched	GF(5YM) and unbanded successfully fledged 2/3 chicks, 1 chick assumed depredated.
	OIWR	40 (57A)	Hatched	GF(YLE) and suspected pair member GF(JYJ) successfully fledged 3/3 chicks.
	OIWR	43 (56A)	Hatched	GF(U61) and GF(059) successfully fledged 1/4 chicks.
	OIE	4 (04A)	Failed - Abandoned	GF(1EL) not seen since 17 May. GF(YTV) renested with GF(CX8) in OIWR and successfully fledged 2/4 chicks.
	OIE	8 (23A)	Hatched	GF(TXK) and GF(PY3) successfully fledged 4/4 chicks.
	OIE	17 (29A)	Hatched	GF(62L) and GF(LKX) successfully fledged 4/4 chicks.
	OIE	21 (36A)	Hatched	GF(U54) and GF(OP9) successfully fledged 3/4 chicks.
	OIE	23 (53A)	Failed - Unknown	Unclear if eggs were lost or hatched. GF(PNJ) seen at Cape Hatteras 17 July.
	OIE	29 (48A)	Failed - Abandoned	GF(CVA) found depredated by gulls 16 June. Nest abandoned by GF(Y5A), and was not seen after 5 July.

Management unit	Subsite	Nest number (mgmt. nest number)	Nest fate	Outcome
National Park Service (cont.)	OIE	34 (52A)	Hatched	GF(5CH) and GF(Y9C) successfully fledged 1/4 chicks.
	OIE	44 (59A)	Hatched	GF(VLP) and unbanded successfully fledged 2/4 chicks.
	BRO	35 (51A)	Hatched	Two unbanded adults successfully fledged 3/3 chicks.
Smith Point County Park	NBW	119 (05B)*	Hatched	GF(XXK) and GF(KE0) fledged 0/3 chicks. GF(XXK) observed in Florida 16 August. GF(KE0) not seen since 12 July.
	NBE	139 (33A)*	Failed - Predated	GF(8XP) and unbanded lost nest. GF(8XP) seen in Florida 13 October.
	SPTW	116 (15A)	Hatched	GF(MH9) and unbanded last seen 11 June. 2 eggs and 1 chick apparently depredated on 13 June. GF(MH9) not seen since 11 June.
	SPTW	126 (19A)	Hatched	GF(522) and GF(N6T) successfully fledged 1/3 chicks.
	SPTW	134 (28A)*	Failed - Predated	GF(50Y) and GF(88K) renested and fledged 0/3 chicks.
	SPTW	135 (31A)	Failed - Abandoned	GF(21U) and GF(JH7) last seen incubating 11 June. Both pair members not seen since 11 June.
	SPTW	142 (35A)	Failed - Abandoned	GF(Y1A) and unbanded last seen incubating 28 June.
	SPTE	107 (10A)	Hatched	GF(CE9) and GF(15X) last seen with nest 10 June. 2 chicks died, 2 eggs lost. Adult wing found. GF(CE9) not seen since 10 June. GF(15X) not seen since 7 June.
	SPTE - GG	121 (24A)*	Failed - Predated	GF(M2P) and unbanded last seen at nest on 25 May. GF(M2P) renested, unbanded adult banded as GF(UL6). Successfully fledged 1/3 chicks.
	SPTE - GG	122 (25A)	Hatched	GF(140) and GF(K42) fledged 2/3 chicks.
	SPTE - GG	129 (14B)*	Failed - Predated	4/4 eggs lost 6 June. GF(AM6) and GF(E00) renested and fledged 0/4 chicks.
	SPTE - GG	130 (27A)	Failed - Predated	Suspected pair GF(736) and GF(H1M) last seen at this nest 1 June. Pair has not appeared to reneest. GF(736) last seen 11 July, GF(H1M) last seen 4 August.
	SPTE - GG	144 (27B)*	Failed - Predated	GF(OMK) last seen at this nest 21 June. 4/4 eggs lost 25 June. GF(OMK) last seen 26 July.

Table S4. Opportunistically observed mammalian predator evidence during the 2022 piping plover breeding season at Fire Island, NY and updated from Wails et al. (2022b). Subsites are listed from west to east (see Table 2 for subsite delineations). We considered all known and suspected mammalian predators at Fire Island.

Management unit	Subsite	Area (ha)	PIPL nests ^a	Prop. nests exclosed	Nests with mammal evidence ^b (A)	Mammal evidence in subsite ^b (B)	Total mammal evidence (A + B)	Observer hrs (hrs in subsite × observers)	Area effort (observer hrs / ha)	Detections / area effort
Robert Moses State Park	DMP	70.88	28	0.86	0	31	31	476.65	6.72	4.61
	STP	14.71	2	1.00	1	27	28	84.95	5.77	4.85
	OSS	24.61	2	1.00	0	5	5	119.58	4.86	1.03
	FLD3	20.52	2	1.00	0	4	4	52.97	2.58	1.55
	FLD4	34.69	1	1.00	1	15	16	92.40	2.66	6.01
	FLD5	32.08	4	1.00	1	20	21	84.95	2.65	7.93
National Park Service	FIL	31.06	8	0.75	2	45	47	206.72	6.66	7.06
	OIWR	37.09	16	0.94	2	84	86	335.02	9.03	9.52
	OIE	67.46	23	0.65	5	29	34	459.83	6.82	4.99
	BRW	5.74	1	1.00	0	5	5	47.00	8.19	0.61
	BRO	15.07	7	0.71	0	10	10	162.43	10.78	0.93
Smith Point County Park	BRE	34.38	1	0.00	0	4	4	93.65	2.72	1.47
	PSO	39.29	1	1.00	0	14	14	139.67	3.55	3.94
	PSE	8.71	2	0.00	0	4	4	93.37	10.72	0.37
	NBW	16.80	7	0.00	1	11	12	181.45	10.80	1.11
	NMA	17.72	1	0.00	0	6	6	59.97	3.38	1.77
	NBE	21.78	3	0.33	1	12	13	60.35	2.77	4.69
	SPTW	19.43	9	0.78	4	13	17	199.38	10.26	1.66
	SPTE	41.93	22	0.59	4	34	38	428.05	10.21	3.72
Total		553.96	140	0.71	22	373	395	3,378.39	6.10	64.77

^a Values here may differ from what was used in analyses (e.g., **Table 10**); this includes all nests (including those not visited by Virginia Tech researchers before nest loss) and surprise broods where nests were never located.

^b Evidence includes visual sightings, tracks, scat, hair and bones, depredations, and predator carcasses.

Appendix 3. Recent research products

Manuscripts

Robinson, S.G., K.M. Walker, H.A. Bellman, D. Gibson, D.H. Catlin, S.M. Karpanty, S.J. Ritter, and J.D. Fraser. 2022. Piping plover chick ecology following landscape-level disturbance. *Journal of Wildlife Management*. DOI: 10.1002/jwmg.22325.

Black, K.M., S.M. Karpanty, S.G. Robinson, H.A. Bellman, K.M. Walker, D.H. Catlin, and J.D. Fraser. In Press. Using GPS location data to assess red fox responses to predator exclosures setup and hatching at shorebird nests. *Wildlife Society Bulletin*.

Conference presentations

**indicates presenting author(s)*

Dorsey, S. *, S. Ritter, C. Wails, S. Robinson, K. Walker, D. Catlin, S. Karpanty, and J. Fraser. How does nest-site viewshed affect Piping Plover nest-site selection. The Waterbird Society's 46th Annual Meeting. October 31-Nov 4, 2022.

Johnson, K. *, C.N. Wails, H.L. VanDerwater, S.S. Dorsey, S.M. Karpanty, and J.D. Fraser. 2022. Potential avian predators of Piping Plover: A study at Fire Island, NY in 2022. The Waterbird Society's 46th Annual Meeting. October 31-Nov 4, 2022.

Swan, E. *, C.N. Wails, S.S. Dorsey, H.L. VanDerwater, H.A. Bellman, S.G. Robinson, K.M. Walker, S.M. Karpanty, and J.D. Fraser. 2022. Impacts of invertebrate community composition on Piping Plover (*Charadrius melodus*) survival between ocean and bay side habitats on Fire Island. The Waterbird Society's 46th Annual Meeting. October 31-Nov 4, 2022.

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