

2021 Annual Report

EFFECTIVENESS MONITORING AND TARGETED STUDIES IN SUPPORT OF FIMI CONSERVATION MEASURES

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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

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



Cite this report as: Wails, C. N., S. G. Robinson, K. M. Walker, S. S. Dorsey, G. D. Hevia, S. J. Ritter, D. H. Catlin, S. M. Karpanty, and J. D. Fraser. 2021. Effectiveness monitoring and targeted studies in support of FIMI conservation measures. Annual report prepared by the Virginia Tech Shorebird Project. Virginia Tech, Blacksburg, VA.

Cover photo: Least Tern chick hatching. Photo provided by Katie Walker



Table of Contents

Background.....	5
Introduction	6
Study Objectives	8
Study Area.....	8
Piping plover monitoring on Fire Island	17
Piping plover general field methods	18
Monitoring effort.....	18
Habitat and tide mapping	18
Piping plover adult survey	19
Piping plover nest locations.....	20
Piping plover nest checks.....	21
Piping plover trapping and banding.....	21
Piping plover banding schemes	22
Piping plover brood observations.....	23
Invertebrate sampling.....	24
Piping plover nest cameras.....	25
Piping plover numbers, density, and distribution on Fire Island	26
Introduction	26
Methods.....	26
Results.....	28
Discussion	37
Piping plover nest and chick survival rates in various habitats across Fire Island	41
Introduction	41
Methods.....	41
Results.....	42
Discussion	48
Red fox monitoring on Fire Island	50
Red fox general field methods	51
Red fox GPS-collaring and ear-tagging	51

Mammal sign transects.....	51
Red fox den surveys.....	54
Camera trapping surveys.....	55
Red fox numbers, density, and distribution on Fire Island	57
Introduction	57
Methods.....	57
Results.....	59
Discussion	63
Dietary ecology of red foxes	66
Introduction	66
Methods.....	67
Results.....	67
Discussion	69
Community interactions on Fire Island	71
Investigation of mesopredator changes and their predicted effects to piping plover	72
Introduction	72
Methods.....	73
Results.....	74
Discussion	79
Literature Cited.....	81
Appendices	88
Appendix 1. General geospatial processing	89
Appendix 2. Recent research products	90

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) and 2020 (Walker et al. 2020) 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 Breach Contingency Plan 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. Virginia Tech researchers were able to conduct fieldwork in a reduced capacity due to university- and state-level health and safety considerations. As a result, some field methodology was modified and these changes are explicitly detailed in this report.

In 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: one in the Otis Pike Wilderness of Fire Island National Seashore, one in Smith Point County Park on Fire Island, and one within Cupsogue Beach County Park on Westhampton 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 as well as the threatened red knot (*Calidris canutus rufa*; USFWS 1995, 2014). Anthropogenic alterations to habitat, including dredging, building dunes and intensive human beach use, can reduce plover habitat availability (Cohen et al. 2009, 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 USACE in approximately 33.7 ha (83.2 ac) of ocean-side habitat at Great Gun restoration area to mitigate adverse effects of dune and beach construction. To create possible nesting and

foraging habitat for piping plovers, USACE also created New Made restoration area, 6 ha (15.8 ac) of bayside habitat south of Carter's Island.

Breaches and overwashes can result in creation of new foraging and nesting habitat for piping plover (Cohen et al. 2009, Robinson et al. 2019), but the manipulation of them by dunes built under the FIMI had the potential to disrupt natural plover breeding processes. Thus, the effect of the habitat modifications under the FIMI needed to be 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, reflected in demographic and growth rates, density, movement, and prey abundance, among others (Loefering and Fraser 1995, Elias et al. 2000, Cohen et al. 2009, Catlin et al. 2014, Catlin et al. 2015, Hunt et al. 2017, Hunt et al. 2018, Walker et al. 2019, Robinson et al. 2020a).

However, plover populations often are 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 to gain the full picture of the effects of habitat change. Red foxes (*Vulpes vulpes*) are a predator of piping plovers on Fire Island (USFWS 2014) 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 (see 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 this 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. 2021). There is a growing recognition by managers and the public on the need to monitor and quantitatively assess the success of predation management actions (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 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

Due to the novel coronavirus (SARS-CoV-2) pandemic in 2020, we have had to make some modifications to planned fieldwork, temporarily reducing the scope of our study. We aim to return to the complete list of study objectives included on the cover page of this report during the 2022 plover breeding season, pending university- and state-level health and safety considerations.

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:** Quantify the interactions of predators with shorebird nests and chicks.

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.

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). 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 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).

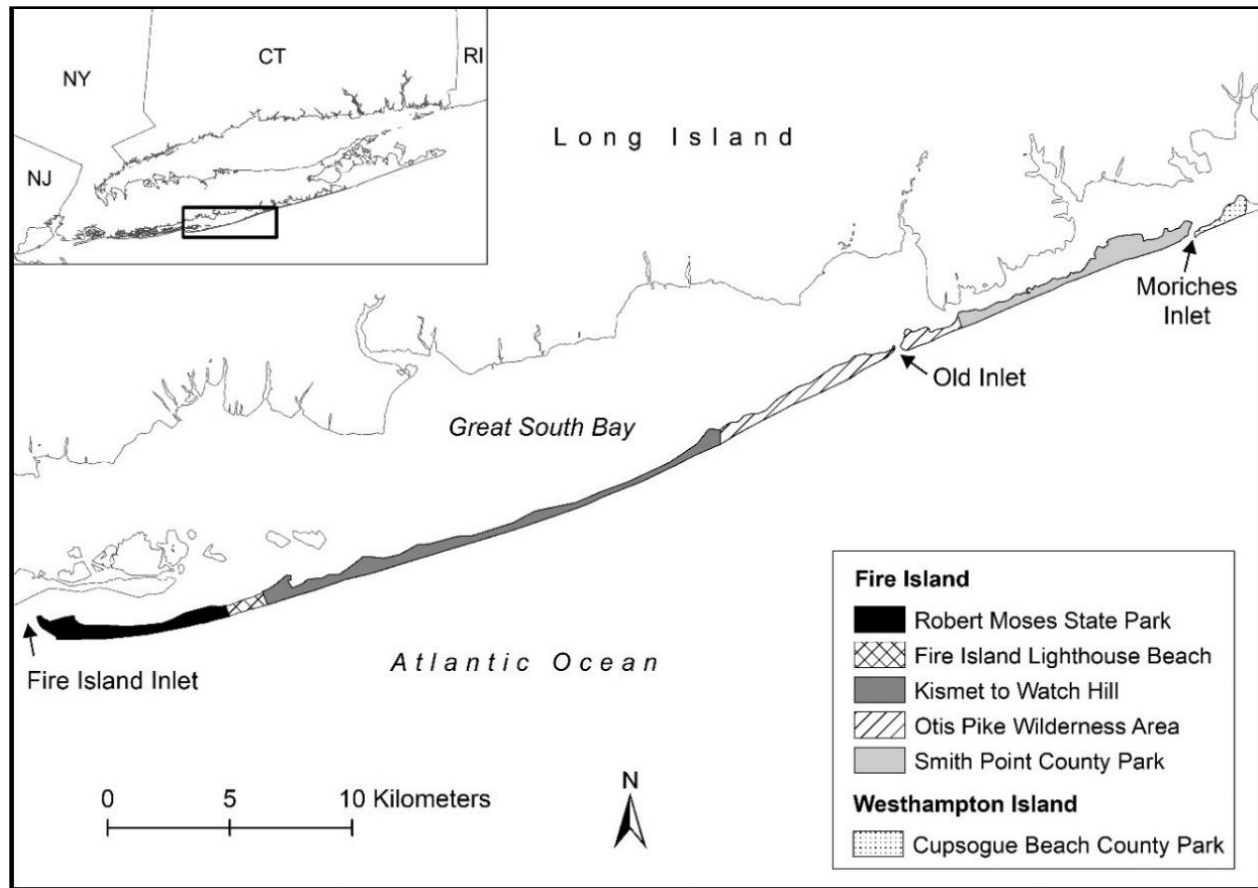


Figure 1. Map of the study area, Fire Island and Westhampton Island, NY comprising different management units.

Prior to Hurricane Sandy, the study area consisted of sandy backshores largely backed by dunes. In October 2012, Hurricane Sandy overwashed several areas of the island. The hurricane created a new breach at Old Inlet, a breach at Cupsogue Beach County Park, and a breach in the Narrow Bay West area of Smith Point County Park. The Old Inlet breach remains open. 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). 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 Great Gun restoration area, and 6 ha of flat sand bayside habitat at 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 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 sea bed to near the surface so that it is tidally exposed. They are often not static and may move seasonally and annually.
Management area	Areas defined by 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.
Sand bar	Mostly intertidal landforms that are generally composed of particles > 0.05 mm. In this study area, sand bars 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 site 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 – 2021. 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.	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 2021.

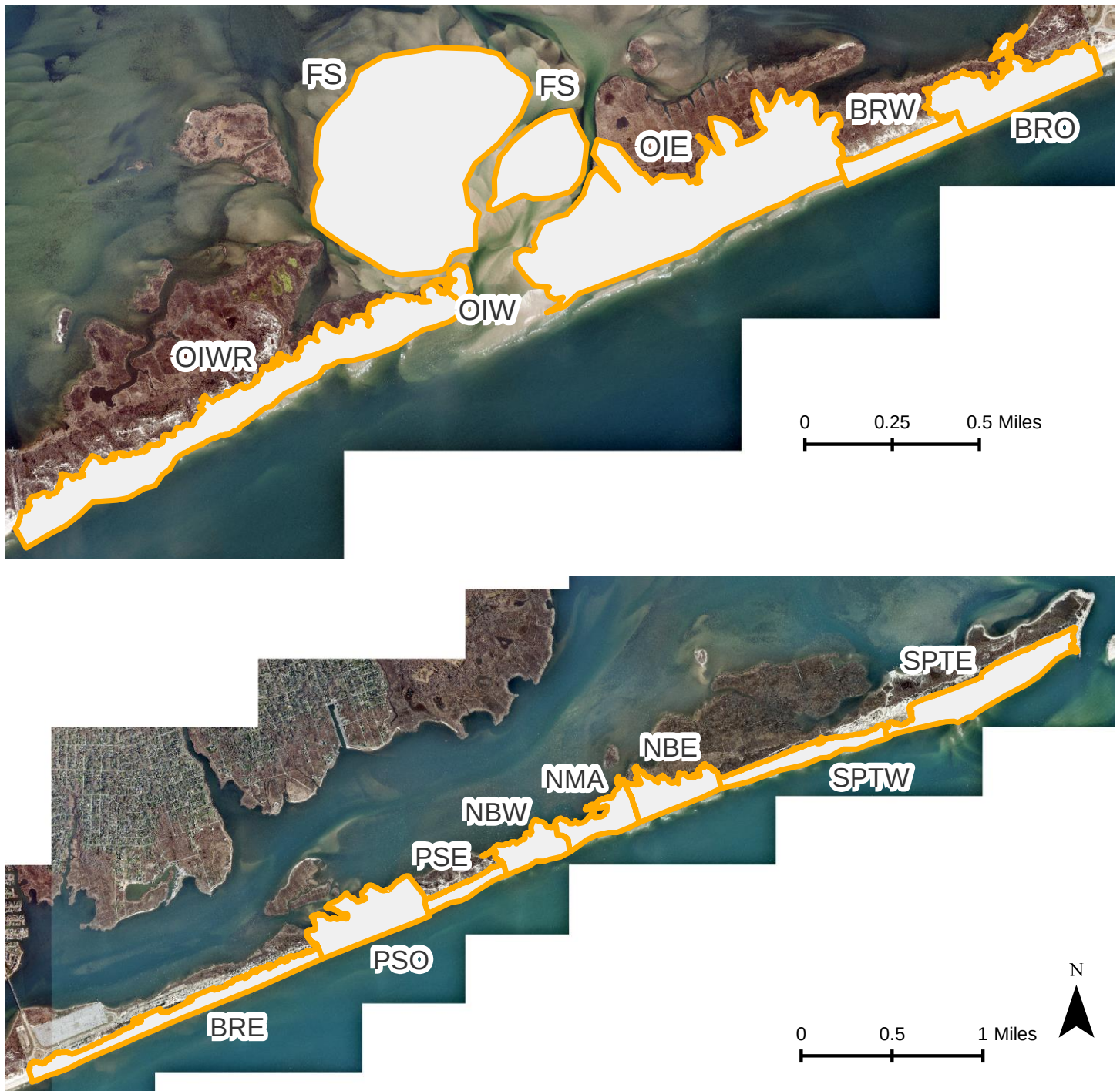


Figure 3. Study area at Otis Pike Wilderness Area Park (top) and Smith Point County Park (bottom). Subsite boundaries are outlined in orange. Refer to **Table 2** for subsite names and descriptions. Imagery captured April 2021.



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 2021.

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 subsite OSS in 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 – 2021.

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



Piping plover monitoring on Fire Island



Piping plover 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 piping plovers in 2020 and 2021 are explicitly detailed here. We aim to return to our previously used field methods during the 2022 plover breeding season, pending university- and state-level health and safety considerations.

Monitoring effort

In 2021, 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, 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 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.

Habitat and tide mapping

Previously used methods: 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.

Modified methods for 2021: Due to time constraints with a reduced field team in 2021, we opted to use LiDAR data and a digital elevation model (DEM) to identify the spring high tide line. LiDAR data collected on 05-18-19 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 in 2021.



Photo: K. M. Walker

Figure 5. Using a Trimble Geo7x to map the spring tides by walking along the high-water line, as performed in previous years.

Piping plover adult survey

Previously used methods: 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. In 2016 – 2019, 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.



Photo: K. M. Walker

Figure 6. Surveying for piping plovers using spotting scopes inside the Great Gun restoration area at SPTE.

Piping plover nest locations

Previously used methods: 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 (2015 – 2019; **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 for 2021: We resumed surveying subsites every 1 – 8 days (weather permitting) April – August, adhering to our previously used methods.



Photos: C. Robertson (left), K. M. Walker (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.

Piping plover nest checks

Previously used methods: 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 for 2021: We resumed nest checks, adhering to our previously used methods.



Photos: K. M. Walker

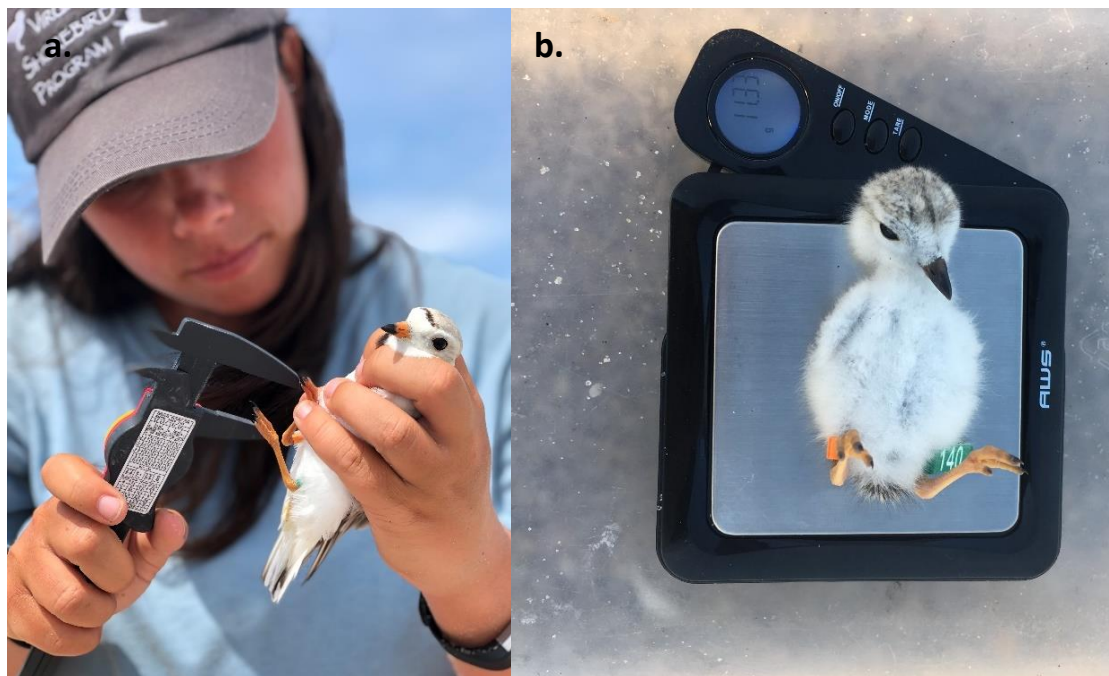
Figure 8. (a) Checking an exclosed nest for hatch using binoculars, and **(b)** checking an unexclosed nest for hatch.

Piping plover trapping and banding

Previously used methods: 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 for 2021: 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

Previously used methods: 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.



Photos: DeRose-Wilson et al. (2013)

Figure 10. Example of piping plover banding schemes, with **(a)** an adult plover with a green flag on the upper left leg and orange Darvic band on the upper right leg, and **(b)** a plover chick with four total Darvic bands.

Piping plover brood observations

Previously used methods: During 2013 – 2019, we searched for plover chicks for 30 days post-hatching, regardless of brood fate, at least once every three days (weather permitting). After locating the brood and identifying chicks, we attempted to collect behavioral observations using spotting scopes on 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 separate 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.

Modified methods for 2021: We resumed searching for plover chicks for 30 days post-hatching, though we did not record brood locations or brood behavior/disturbance observations.

Invertebrate sampling

Previously used methods: We sampled terrestrial invertebrates from 2014 – 2019 in various habitat types (i.e., 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 (in m) to the first sampling location. We 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 counted, measured (± 1 mm), and identified any arthropods to taxonomic order.

Modified methods for 2020: Invertebrate sampling was not conducted in 2020.

Modified methods for 2021: Invertebrate sampling was not conducted in 2021.



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

Previously used methods: Starting in 2016, we placed remote 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 both adults and the nest reached a 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 area.

Modified methods for 2020: No nest cameras were placed in 2020.

Modified methods for 2021: No nest cameras were placed in 2021.



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.

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 (Robinson et al. 2019). Such habitats are utilized by species or populations that are at-risk, such as the Atlantic Coast piping plover (Cohen et al. 2009, Walker et al. 2019).

Habitat changes resulting from anthropogenic and natural forces affect piping plover population dynamics (Cohen et al. 2009, Catlin et al. 2015). Many plover populations are regulated by density dependent forces on the breeding grounds (Cohen et al. 2009, Catlin et al. 2019). 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. Reduced competition for space and thus an increase in available nesting and foraging territories may lead to a population increase in density-dependent species, and has been observed in piping plover on several occasions (Wilcox 1959, Cohen et al. 2009, Catlin et al. 2015, Hunt et al. 2018, Robinson et al. 2019). Given the increase in habitat following Hurricane Sandy, a similar process is predicted to unfold in the piping plover population on Fire Island and Westhampton Island.

Disturbance and the subsequent habitat creation following Hurricane Sandy had 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 habitat available, and thus presented an opportunity to investigate these processes. 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 expected to see plover numbers quickly increase in response to the new available habitat and then eventually stabilize around carrying capacity (Wilcox 1959, Cohen et al. 2009, Robinson et al. 2019).

Methods

Using color bands and field-readable, alphanumeric ID flags, we banded adults and chicks from 2013 – 2021¹. Our goal was to capture every unbanded plover each year to help us identify individuals for a variety of analyses.

¹ Excluding 2020.

In particular, if we could capture the majority of unbanded adults and chicks in each year, we could understand the contribution of immigration to the study area from each origin class (e.g., natal recruits, immigrants).

We compared the number of pairs that used various habitats during 2013 – 2021, including backshore, restoration areas, natural overwashes, and manipulated overwashes to determine if plovers were using any of these newly 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 compared the proportion of nests that occurred in each of those habitat types. We assumed that if a male renested in the same subsite or habitat type, the location of the reneest was not independent of the first nest. We only included reneests 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 class.

We estimated annual pair numbers using the males within pairs. We used males for pair counts and productivity estimates because when pairs separate, the males often retain territories when they partner with a new female. Additionally, males are the primary territory holders and defenders (Haig and Oring 1988). Using banded adult information and nest date overlap, we used the highest number of male territories per management unit each breeding season as an estimate of the number of pairs in that 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 any nest failure. Nests were assigned a fate of successful if they hatched at least one chick and failed if they were abandoned, depredated, flooded, or suffered total 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 and chicks were present to confirm the nest fate. In cases where no chicks were present, we assigned the nest a fate based on any sign we could find near 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 sign of fate, we assigned it an unknown fate. If a nest appeared to be abandoned (e.g., cold eggs, dirty egg shells), we looked for any sign that an adult was incubating the eggs for three or more consecutive visits during a six day period. After six days, if there was no sign of an adult tending the eggs, we assigned the nest as 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 – 2021. Using the delineated subsites, extending from the ocean to the bay, we calculated the hectares of dry sand in each subsite (Appendix 1. General geospatial processing). 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 of dry sand less than 1 m² 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. 4.1.2 (R Core Team 2021).

Results

Piping plover numbers

From 2013 – 2021, we banded 303 adult piping plovers (158 females and 145 males) and 965 chicks (**Table 4**). Prior to 2021, all chicks were banded between 0 – 16 d old; due to time constraints in the field with a reduced team, chick age at banding changed to 0 – 29 d in 2021. In 2013 and 2014, eight nests were found at Watch Hill. Five adults and three chicks from those nests were banded. We have not monitored nests at Watch Hill since 2014 due to logistical limitations, though these areas were continually monitored by the National Park Service. We did not band any birds at Robert Moses State Park or the Lighthouse Tract prior to 2015. No birds were banded in the 2020 breeding season. Of the 965 total chicks banded, 3.5% were banded in the 2013 season, 4.3% in 2014, 4.9% in 2015, 10.5% in 2016, 13.5% in 2017, 17.5% in 2018, 25.2% in 2019, and 20.7% in 2021.

Table 4. Piping plover adults and chicks banded on Fire Island and Westhampton Island per management unit during the full 2013 – 2021 study period. Subsites within each management unit that were monitored are included in parentheses.

Management unit	Adults		Chicks
	Females	Males	
Robert Moses State Park (DMP, OSS, STP, FLD3, FLD4, FLD5)	47	36	207
Fire Island National Seashore – Lighthouse Tract (FIL)	5	4	22
Fire Island National Seashore – west of breach (OIWR, OIW)	4	8	59
Fire Island National Seashore – east of breach (OIE, BRO)	23	24	175
Smith Point County Park (BRE, PSO, PSE, NBW, NMA, NBE, SPTW, SPTE)	46	44	356
Cupsogue Beach County Park (CSW, CSE)	23	16	126
Offsite – Bellport to Watch Hill	10	13	20



	Total	158	145	965
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Since 2013, we have located and monitored 562 nests (373 [66.4%] surviving to hatch, 189 [33.6%] failed; **Table 5**). In 2021, we found 113 nests; 74 (65.5%) of which survived to hatching and 39 (34.5%) 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 – 2021. Nest numbers include renests of some pairs. Nests were only designated as within natural overwash if they were within the Hurricane Sandy-created overwashes.

Management unit	Habitat	Found	Hatched	Failed	Hatch %	Fail %
Robert Moses State Park ^a	Backshore	93	65	28	70	30
	Barrier interior	17	9	8	53	47
	Dredge spoil	8	6	2	75	25
	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	5	5	0	100	0
	Barrier interior	3	1	2	33	67
	Manipulated dune	7	7	0	100	0
	Swale	1	1	0	100	0
Fire Island National Seashore – west of breach	Backshore	20	12	8	60	40
	Natural overwash	31	23	8	74	26
Fire Island National Seashore – east of breach	Backshore	1	1	0	100	0
	Manipulated dune	1	0	1	0	100
	Natural overwash	84	56	28	67	33
Smith Point County Park	Backshore	33	21	12	64	36
	Barrier interior	20	15	5	75	25
	Manipulated dune	12	7	5	58	42
	Manipulated overwash	72	44	28	61	39
	Great Gun Restoration Area ^a	47	32	15	68	32
	New Made Restoration Area ^a	26	12	14	46	54
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		562	373	189	66	34

^a Not monitored 2013 – 2014

^b Not monitored in 2013 and 2020 – 2021

^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 – 2021. Fire Island National Seashore includes nests from Watch Hill in 2013 – 2014. Cupsogue Beach County Park was only monitored 2013 – 2019.

Year	Management unit	Total	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%)
Total		562	373 (66%)	64 (11%)	67 (12%)	19 (3%)	39 (7%)

Of the nests monitored in 2021 that failed, nine were abandoned, 10 were depredated, nine flooded, and 11 fates were unknown (**Table 7**). Throughout the study period, depredation and abandonment have been the two largest causes of nest failure, at 12% and 11% respectively (**Table 6**). The greatest rate of nest depredation was observed in 2015 (45.8%) and the greatest abandonment rate occurred in 2014 (20.0%).

Table 7. Piping plover nest distributions and fates on the Fire Island study area, in various habitats of each management unit, April – August 2021. Nest numbers include renests 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
		Found	Hatched	Failed	
Robert Moses State Park	Backshore	19	13	6	Abandoned (1), flooded (4), unknown (1)
	Barrier interior	2	2	0	-
	Dredge spoil	2	2	0	-
	Dune	5	5	0	-
	Swale	2	2	0	-
Fire Island National Seashore – Lighthouse Tract	Backshore	2	2	0	-
	Barrier interior	1	1	0	-
	Manipulated dune	2	2	0	-
Fire Island National Seashore – west of breach	Natural overwash	14	7	7	Depredated (1), flooded (2), unknown (4)
Fire Island National Seashore – east of breach	Natural overwash	23	13	10	Abandoned (3), depredated (1), flooded (3), unknown (3)
Smith Point County Park	Backshore	7	3	4	Abandoned (1), depredated (3)
	Barrier interior	8	6	2	Depredated (1), unknown (1)
	Breach fill	1	1	0	-
	Manipulated dune	6	5	1	Abandoned (1)
	Manipulated overwash	1	0	1	Abandoned (1)
	Restoration area	18	10	8	Abandoned (2), depredated (4), unknown (2)
Total		113	74	39	

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 92 pairs (384.2% increase since 2013), the maximum number of pairs recorded at Fire Island (**Table 8**). Across all years of the study, the majority of plovers nested in backshore



and overwash habitats. 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 renests 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%)
Total	219 (47%)	20 (4%)	165 (36%)	60 (13%)

^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

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 (e.g., 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
Robert Moses State Park	<i>12</i>	<i>10</i>	10	9	8	15	22	<i>23</i>	24
Fire Island Lighthouse	<i>1</i>	<i>1</i>	0	1	1	1	4	<i>5</i>	5
Fire Island National Seashore – west	3	1	1	1	0	1	7	<i>7</i>	11
Fire Island National Seashore – east	4	4	2	6	10	9	13	<i>11</i>	18
Smith Point County Park	6	8	13	13	20	25	31	<i>25</i>	34
Cupsogue Beach County Park	3	5	7	6	6	7	9	<i>10</i>	-
Outside study area – Bellport to Watch Hill	3	3	0	0	0	0	<i>3</i>	<i>1</i>	-
Outside study area – Davis Park to Kismet	0	0	0	0	<i>1</i>	0	0	<i>5</i>	-
All areas	32	32	33	36	46	58	89	<i>87</i>	-
Fire Island and Cupsogue sites	29	29	33	36	45	58	86	<i>81</i>	-
Fire Island sites	26	24	26	30	39	51	77	<i>71</i>	92

Piping plover densities

Density in the study area increased from 0.06 pairs/ha of dry sand habitat patches > 1m² in 2013 to 0.31 pairs/ha in 2021. Density remained relatively stable from 2013 – 2016 and showed the most increase after 2017 (Figure 13).

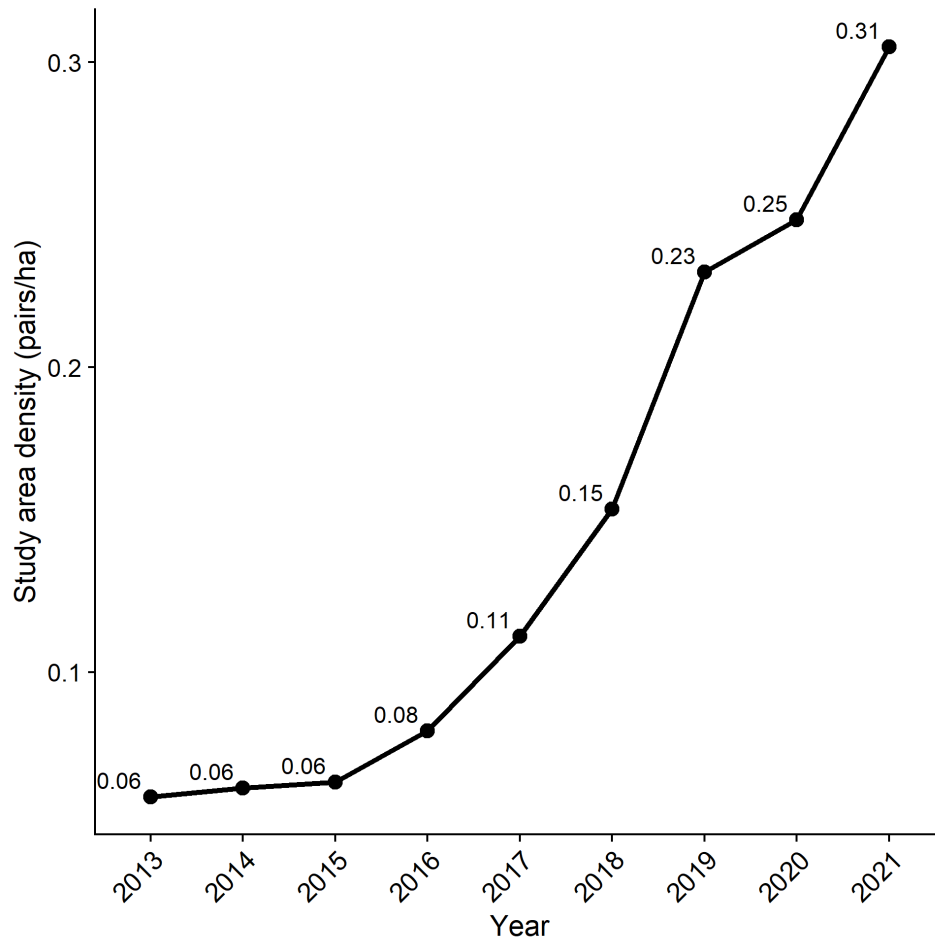


Figure 13. Estimated study area density for each year following Hurricane Sandy. Available habitat was defined as all available dry sand patches greater than 1 m² using 15 cm classified aerial imagery in each year. For 2013 and 2014, the number of pairs was a combination of the number of pairs monitored 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. Estimated densities decreased in three of the four management units 2020, and densities at all but Robert Moses increased in 2021 (Figure 14). Robert Moses State Park density decreased from 0.26 to 0.23 pairs/ha between 2020 and 2021. The greatest density

was observed at the East Wilderness Area in 2021 (0.57 pairs/ha; **Figure 14**). Density for most management units was fairly low prior to 2016 (< 0.2 pairs/ha, except East Wilderness in 2016). The West Wilderness area did not have any pairs in 2015 or 2017, and therefore density was 0.00 pairs/ha. The East Wilderness area also had a decline in density between 2014 and 2015, but increased again in 2016 (**Figure 14**).

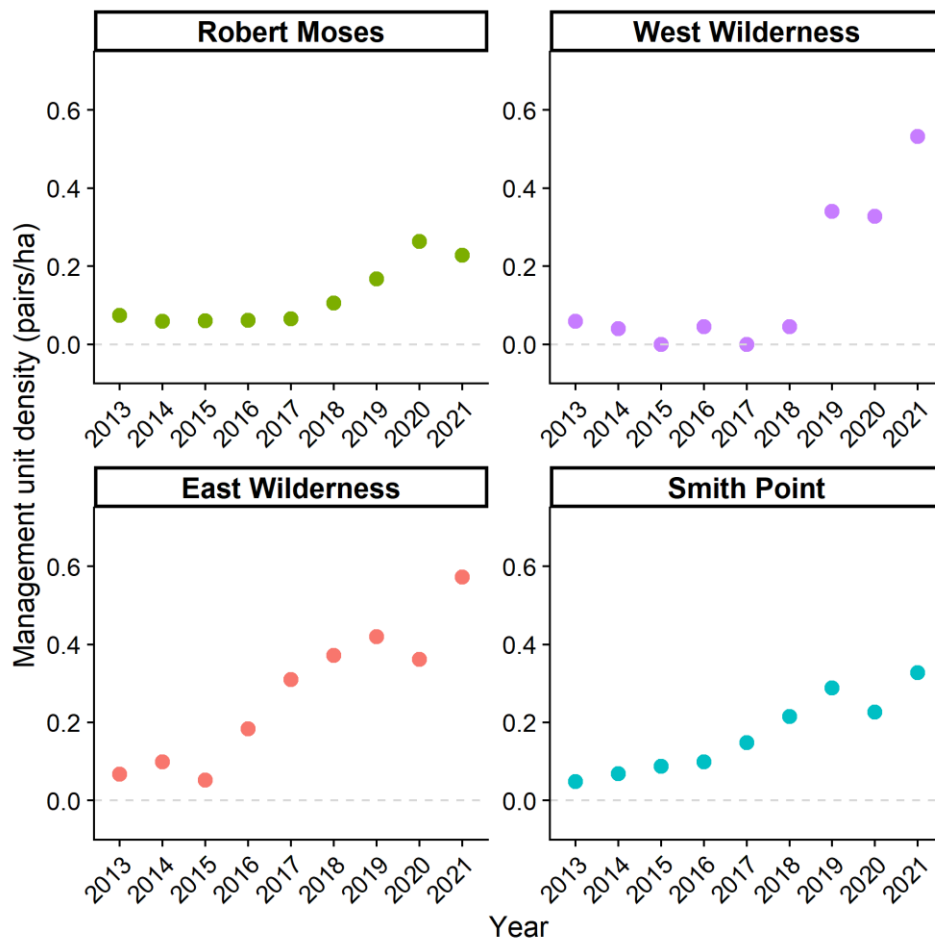


Figure 14. Estimated management unit density for each year following Hurricane Sandy. Available habitat was defined as all available dry sand patches greater than 1 m² using 15 cm classified aerial imagery for each year. For 2013 and 2014, the number of pairs was a combination of the number of pairs 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 in 2013 to 0.36 in 2019, decreased to 0.27 in 2020, and returned to 0.36 in 2021 (**Figure 15**). When separated by subsites with and without bay access, subsites with bay access had, on average, greater densities (**Figure 16**). During 2013 – 2019, subsites with and without bay access increased in density, though the increase for bay-accessible sites increased at a greater rate. Densities at subsites with and without bay access decreased slightly in 2020, and increased in 2021 (**Figure 16**).

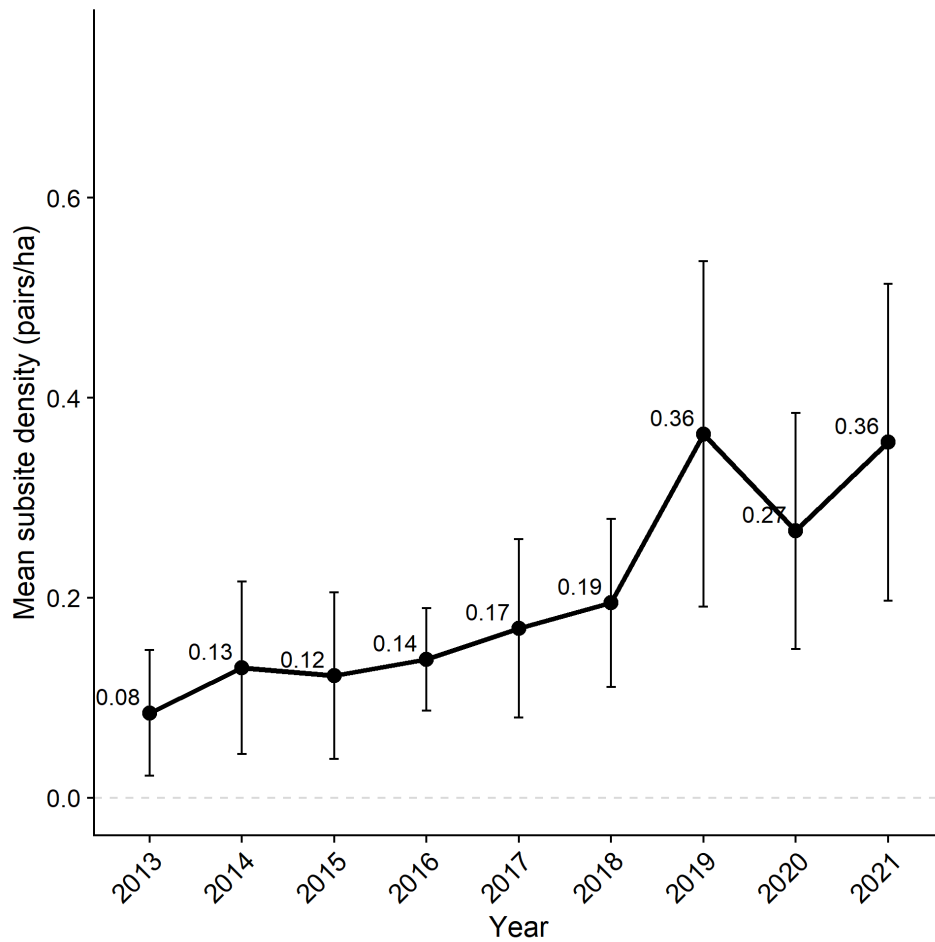


Figure 15. Mean annual subsite density. Subsite extent for density extended the entire width (north to south) of the island. Available habitat was defined as all dry sand patches greater than 1 m² using 15 cm classified aerial imagery. Error bars represent 95% confidence intervals. All annual density measures exclude observations from Cupsogue Beach County Park.

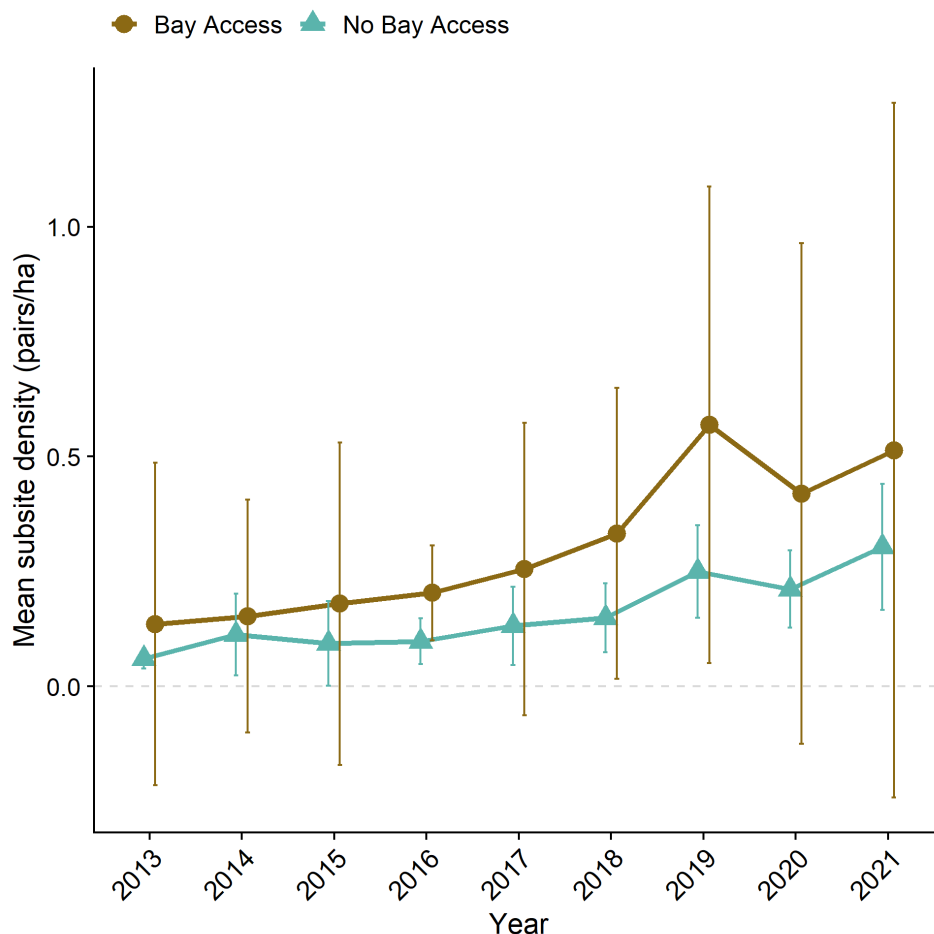


Figure 16. Average subsite density in each year for subsites with bay access (i.e., DMP, OSS/STP, PSO, NBW, CSW) compared to subsites without direct bay access. Subsite extent for density extended the entire width (north to south) of the island. Available habitat was defined as all contiguous dry sand patches (identified via an 8-cell rule; see Methods) greater than 1 m² using 15 cm classified aerial imagery. Error bars represent 95% confidence intervals. All annual density measures exclude observations from Cupsogue Beach County Park.

Discussion

Despite recent increases in habitat and relatively high immigration rates (Robertson et al. 2019, Walker et al. 2020), the increase in breeding pairs within the study area since Hurricane Sandy has been relatively modest (2013 – 2021: 187.5%) compared to some other habitat-creating events, though the observed increase falls within the range of past observations during similar time periods (42% – 745%; Robinson et al. 2019). For population growth to occur, sufficient 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 (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 the storm (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 later years, pairs have generally 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) which would 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, the pair numbers and nesting densities we recorded in 2021 were comparable to or had increased since 2019. 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.

Red fox numbers declined across most of the study area, and they were extirpated from some sites during 2015 – 2019 due to an outbreak of sarcoptic mange (*Sarcoptes scabiei*). No foxes were detected from April 2016 to March 2017 east of Old Inlet, including Smith Point County Park, and only one was believed to inhabit that area intermittently during 2017 – 2018 (Walker et al. 2020, Black 2021, this report). In the years following the fox decline, plover nest depredation simultaneously decreased. Nest abandonment was the greatest reason for nest failure from 2015 – 2021, except in 2018 when at least one red fox was present at Smith Point County Park. The plover population growth rate in the study area began to increase following the extirpation of foxes, suggesting red foxes may play a role in limiting plover populations.

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 (Smith Point County Park). During this period, the plurality of 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. An important habitat distinction of note is the difference between Hurricane Sandy-created overwashes and overwashes 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 7-fold, 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, our estimates of plover pairs nesting in overwash habitats should be interpreted as a declining use of Hurricane Sandy-created habitats through time.

Pair density quadrupled since 2013, the first breeding season following Hurricane Sandy, to 2021. Much of that increase was driven by Smith Point and Robert Moses before 2020. Interestingly, the two sites saw contrasting density changes in 2020. Robert Moses consistently increased in density each year, despite only a two pair increase between the 2019 and 2020 breeding seasons. This is likely due to limited dry sand at Democrat Point at the time the aerial imagery was collected in 2020, which lowered the classified dry sand habitat and thus continued to increase density at that site. We suspect that densities at Smith Point declined slightly from 0.29 pairs/ha in 2019 to 0.23 pairs/ha in 2020 due to vegetation removal at Great Gun in 2020, which would increase the total area of dry sand used in density calculations. The West Wilderness area and Cupsogue did not exhibit



changes in density between 2019 and 2002 due to a stabilized pair count (West Wilderness) and slight decrease (Cupsogue) and little change in available dry sand habitat between years.

We hypothesized that density would begin to decline as the piping plover population on Fire Island neared carrying capacity. In a study of Westhampton Island, Cohen et al. (2009) identified carrying capacity as 1.05 pairs/ha in two-sided habitats (i.e., ocean-side and bay-side connected by sand traversable by a walking plover) and 0.50 in one-sided habitat (i.e., no sand connection between ocean and bay), and beyond these densities, the population began to decline. Nine years after Hurricane Sandy, the density for the Fire Island population is less than half of what Cohen et al. (2009) found for two-sided habitats (mean = 0.51 pairs/ha) and less than two-thirds found in one-sided habitat (mean = 0.60 pairs/ha). However, density, like many population parameters, is highly 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 patches of habitat that would otherwise be excluded with a coarser level of habitat identification, making cross-study comparison challenging. In recent years of our study, reproductive output has not yet declined below what would be needed for stationarity (1.10 fledged chicks/pair for stationary population [Weithman et al. 2019]; 1.59 fledged chicks/pair in 2021, mean = 1.36 fledged chicks/pair, 95% CI = 0.98 – 1.74 fledged chicks/pair). Similarly, we have not 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. 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 (USFWS 2014). Since 2018, vegetation continued to increase on dry, sandy areas across the study area. This increase in vegetation is happening concurrently 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. 2009). Therefore, we expect that the amount of suitable nesting habitat will ultimately reduce and then eliminate population growth within the study area. Continued vegetation management, as was done in the Great Gun restoration area in the early spring of 2019 and 2020, may be required 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, but continual monitoring is required to understand the factors influencing this population's growth and eventual limitations. Although carrying capacity has not been reached, the near complete extirpation of red foxes on Fire Island has likely removed an important predatory effect that would otherwise reduce reproduction and survival in this population. Indeed, the local predator community has played an important role on reproductive output (Walker et al. 2020, 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 eight years and is contributing to nesting and foraging habitat loss and fragmentation. Without the 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. 2009). How the population responds to these pressures within the following few years will be crucial in determining carrying capacity in the study area and the limitations that define it.

Piping plover nest and chick survival rates in various habitats across Fire Island

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

Introduction

Reproductive output has been shown to 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 (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 inside nesting exclosures (e.g., ghost crabs, *Ocypode quadrata* [Kwon et al. 2018]; American mink, *Neovison vison* [M. Stantial and C. Weithman, pers. comm.]). Once precocial chicks have hatched, a nest exclosure is no longer useful at protecting chicks, which move about and forage for themselves soon after hatch. Therefore, it is important for the next generation of plover management and conservation to better understand the best methods to improve plover chick survival and reproductive output.

Fire Island reproductive output 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 focused on the components of reproductive output (i.e., nest survival, chick survival) on Fire Island and Westhampton Island following Hurricane Sandy and the FIMI project. We also were interested in any relationship between reproductive output time since Hurricane Sandy, and geomorphic and habitat variables. Specifically, we were interested in effects of artificially-created nesting habitats for piping plovers, which were built intentionally (i.e., restoration areas), built for island stabilization (i.e., breach fills), and were created by the hurricane (i.e., natural and manipulated overwash; **Table 2**).

Methods

Analyses were completed in the R statistical environment ver. 4.1.2 (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. When predicting from this model, hatch date was set at 34 days from 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 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.

Results

Nest survival

The probability of surviving to hatch varied across years of our study (**Table 10**). Nest survival was greatest in 2016 and lowest in 2015. Overall, nests that were exclosed during or after clutch completion were predicted to have a greater probability of surviving to hatch than unexclosed nests (**Figure 17**). Survival to hatch for exclosed nests ranged from 0.51 – 0.84, and survival to hatch for unexclosed nests ranged from 0.22 – 0.67 (**Figure 17**). However, midway through the nesting season in 2015, exclosures were removed from all of the Smith Point County Parks nests (approximately 15 days after clutch completion) as fox began to enter exclosures through the top. The nests in 2015 that had been previously exclosed had a 0% probability of surviving to hatching (Monk et al. 2015).

Table 10. Predicted probability of plover nests surviving to hatch (34 days from initiation) for all four management units in 2013 – 2021 (excluding 2020). The predicted probability comes from a logistic exposure model, estimating the interactive effect of year and management unit on daily survival, and then raised to the 33rd power.

Year	Management unit	Nests (<i>n</i>)	Survival estimate	Lower 95% CI	Upper 95% CI
2013	Fire Island National Seashore	7	0.67	0.29	0.95
	Smith Point County Park	9	0.27	0.06	0.59
	Cupsogue Beach County Park	4	0.70	0.20	0.99
2014	Fire Island National Seashore	6	0.51	0.17	0.86
	Smith Point County Park	9	0.50	0.22	0.79
	Cupsogue Beach County Park	5	0.78	0.36	0.99
2015	Robert Moses State Park	10	0.63	0.35	0.88
	Fire Island National Seashore	4	0.57	0.16	0.92
	Smith Point County Park	23	0.12	0.04	0.25
	Cupsogue Beach County Park	11	0.59	0.29	0.86
2016	Robert Moses State Park	11	0.70	0.40	0.92
	Fire Island National Seashore	9	0.77	0.46	0.97
	Smith Point County Park	14	0.84	0.60	0.98
	Cupsogue Beach County Park	6	1.00	0.95	1.00
2017	Robert Moses State Park	12	0.45	0.19	0.73
	Fire Island National Seashore	14	0.71	0.45	0.91
	Smith Point County Park	24	0.77	0.58	0.92
	Cupsogue Beach County Park	11	0.36	0.12	0.67
2018	Robert Moses State Park	19	0.71	0.48	0.89
	Fire Island National Seashore	13	0.75	0.49	0.94
	Smith Point County Park	29	0.65	0.47	0.82
	Cupsogue Beach County Park	7	0.86	0.54	1.00
2019	Robert Moses State Park	25	0.66	0.46	0.83
	Fire Island National Seashore	29	0.68	0.49	0.84
	Smith Point County Park	35	0.73	0.56	0.87
	Cupsogue Beach County Park	12	0.56	0.26	0.84
2021	Robert Moses State Park	30	0.75	0.54	0.91
	Fire Island National Seashore	42	0.66	0.44	0.84
	Smith Point County Park	41	0.72	0.52	0.88

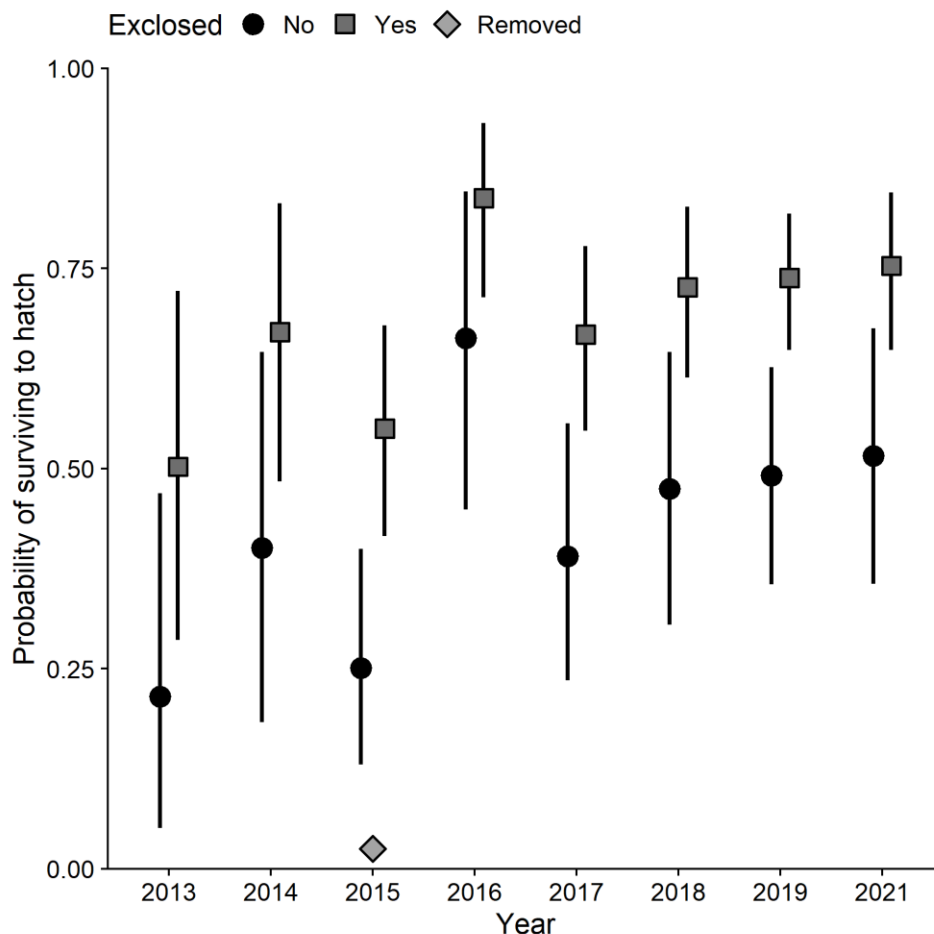


Figure 17. The probability of piping plover nests surviving to hatching with an exclosure and without an exclosure on Fire Island and Westhampton Island, 2013 – 2021 (excluding 2020). Error bars represent 95% credible intervals.

From 2013 – 2021, 56 of 543 nests were abandoned (10.3%; **Table 11**). From the 56 abandoned nests, 15 nests (27%) belonged to a pair of unbanded or unknown adults, 19 nests (34%) belonged to a pair with one banded adult, and 22 nests (39%) belonged to a pair of banded adults. Of the 22 nests with a pair of banded adults, all but one nest (95%) had at least one individual resighted after the nest was abandoned. There were two nests (9%) where both banded individuals were seen after abandonment. Of the 19 nests with one banded adult, 11 banded individuals (73%) were seen at least once after nest abandonment. In 2021, three banded individuals (one from a nest with one banded adult, two from a nest with two banded adults) were not seen after nest abandonment. In 2021, one banded male had initiated a clutch with a female but abandoned and was later observed reneesting with a new female before subsequently abandoning and was not resighted later. It is important to note that search effort for banded birds in 2021 was not consistent with that of previous years. Of the 62 banded birds from the 56 abandoned nests, 34 banded individuals (55%) were seen after their nest was abandoned. Only 12 of those 34 banded individuals reneested within the same season. Throughout the duration

of this study, we recorded four 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 three of these banded adults within 90 m of their nest.

Table 11. Abandonment rate of piping plover nests with exclosures and without exclosures on Fire Island and Westhampton Island study area 2013 – 2021.

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
Total – unexclosed		147	95	52	13	0.09
Total – exclosed		396	79	317	43	0.11

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

Chick survival

From 2013 – 2019, we monitored 233 broods. Sixty-seven additional broods were monitored by land managers in 2020 but were not all regularly observed, therefore, were not used for this analysis. In 2021, we operated under a reduced capacity and thus include 43 broods regularly observed in focal subsites (see *Piping plover general field methods*). Together, we included 276 broods across eight years in this analysis.

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 18, Table 12**).

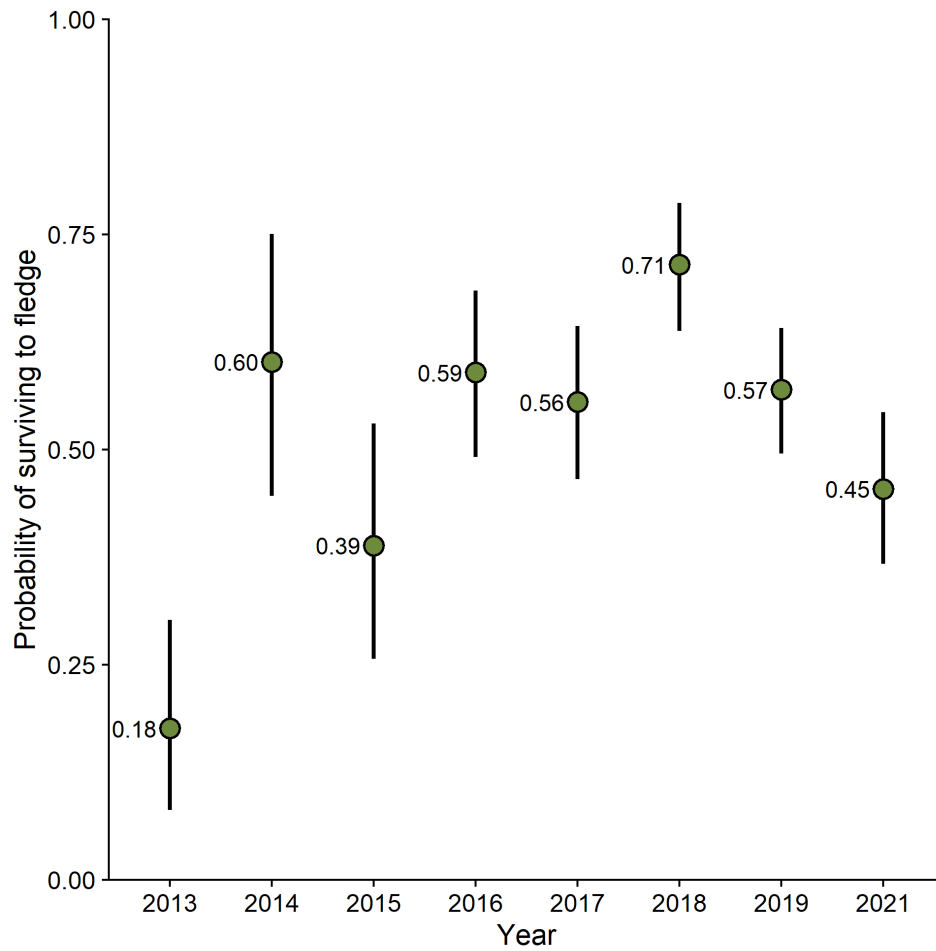


Figure 18. Probability of surviving to fledge (25 days) for chicks 2013 – 2021 (excluding 2020) on Fire Island and Cupsogue Beach County Park. Error bars represent 95% credible intervals. Note that the survival estimates for 2021 was based on observations from focal subsites only.

Table 12. Probability of surviving to fledge (25 days) for chicks in all management units 2013 – 2021 (excluding 2020), estimated from a young survival model.

Management unit	Year	Broods (n)	Survival to fledge	Lower 95% CI	Upper 95% CI
Robert Moses State Park	2015	5	0.56	0.30	0.90
	2016	8	0.42	0.24	0.61
	2017	6	0.86	0.66	1.00
	2018	14	0.75	0.61	1.00
	2019	17	0.64	0.50	0.77
	2021 ^a	17	0.46	0.32	0.60
Fire Island National Seashore	2013	5	0.17	0.05	0.34
	2014	3	0.61	0.29	1.00
	2015	2	0.99	0.91	1.00
	2016	7	0.77	0.56	1.00
	2017	10	0.49	0.33	0.66
	2018	10	0.74	0.58	1.00
	2019	21	0.77	0.64	0.92
	2021 ^a	13	0.47	0.32	0.65
Smith Point County Park	2013	3	0.37	0.11	0.72
	2014	4	0.81	0.58	1.00
	2015	2	0.58	0.21	1.00
	2016	12	0.80	0.65	1.00
	2017	19	0.55	0.42	0.69
	2018	20	0.71	0.59	0.83
	2019	27	0.52	0.41	0.64
	2021 ^a	13	0.48	0.32	0.63
Cupsogue Beach County Park	2013	3	0.10	0.01	0.32
	2014	4	0.47	0.23	0.76
	2015	7	0.13	0.03	0.29
	2016	6	0.30	0.12	0.52
	2017	4	0.47	0.23	0.75
	2018	6	0.75	0.50	1.00
	2019	8	0.32	0.17	0.49

^a Considers focal subsites only

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 13**).

Table 13. Piping plover pair counts and productivity (fledglings/pair; R_o) in four different management units on Fire Island and Westhampton Island, 2013 – 2021. Pairs is the number in each subsite, and R_o is the reproductive output (chicks fledged/pair).

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

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

Discussion

In previous years, we found direct evidence that nest exclosures improved nest survival (Walker et al. 2020), and it is likely that the same is true for 2021. 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). In 2015, for example, Smith Point County Park had high fox density, and plover nest depredation occurred after fox entered through the top of exclosures. Following these predation events, the exclosures were removed from all other nests at the county park. Once the exclosures were removed, the 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 continue to attract predators.

While nest exclosures enhance nest survival, there may be an associated risk of adult depredation (Murphy et al. 2003, Cohen et al. 2016). Depredation of adults can be difficult to immediately detect, though multi-year studies have confirmed that adult mortality leads to 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).

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 was low across the study area in 2021 (see *Red fox numbers, density, and distribution on Fire Island*), we observed tracks from feral cats (*Felis catus*) near ten plover nests, and northern raccoon (*Procyon lotor*) tracks near two plover nests (ten of the 12 nests had been exclosed). Two of the 12 nests were later abandoned, and one of which had cat tracks encircling the enclosure. The banded adult from this pair was later seen attempting to renest within the same subsite, though it is unclear if its mate from the previous nesting attempt also survived. For other instances of nest abandonment, our resight rate of banded birds after abandonment (55%) is lower than the predicted annual survival rate of 0.69 ± 0.02 (with detection probability of 0.96 ± 0.02 for adults; Walker et al. 2020). 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, we found that 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 vegetation cover. Vegetation may offer plovers cover from some avian predators (Schmidt 1999, Ellis et al. 2020). In 2021, gulls (primarily ring-billed gulls *Larus delawarensis* and herring gulls *Larus argentatus*) were often present at Democrat Point, a subsite with fairly sparse vegetation cover. While gull predation was not directly observed, we did recover a chick flag from a suspected avian pellet located in the Democrat Point subsite. In other areas with more dense vegetation, additional plant cover likely conceals terrestrial predators (Espie et al. 1996, Cohen et al. 2008). In 2018, we were able to confirm depredation of an adult plover by a feral cat within Smith Point County Park after its band had been found in a scat sample (Black 2021, Black et al. In Revision). Although we suspect that mammalian predation pressures will be from feral cat and northern raccoon given the slow recovery of fox from mange outbreaks (see *Red fox numbers, density, and distribution on Fire Island*), avian depredation rates have remained elusive. Careful consideration of vegetation cover will contribute to the outcomes of predator control programs (Hunt et al. 2019).

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. We aim to return to our previously used field methods during the 2022 plover breeding season, pending university- and state-level health and safety considerations.

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 as well as 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 since 2019 due to both the mange-related mortality of foxes (see below) as well as 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 19**). 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 possible 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 14**). 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 for 2021: We resumed transect surveys in 2021. We conducted two transect surveys (May – June and June). We collected all mammal scat visible during transect surveys, regardless of age.

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, 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).

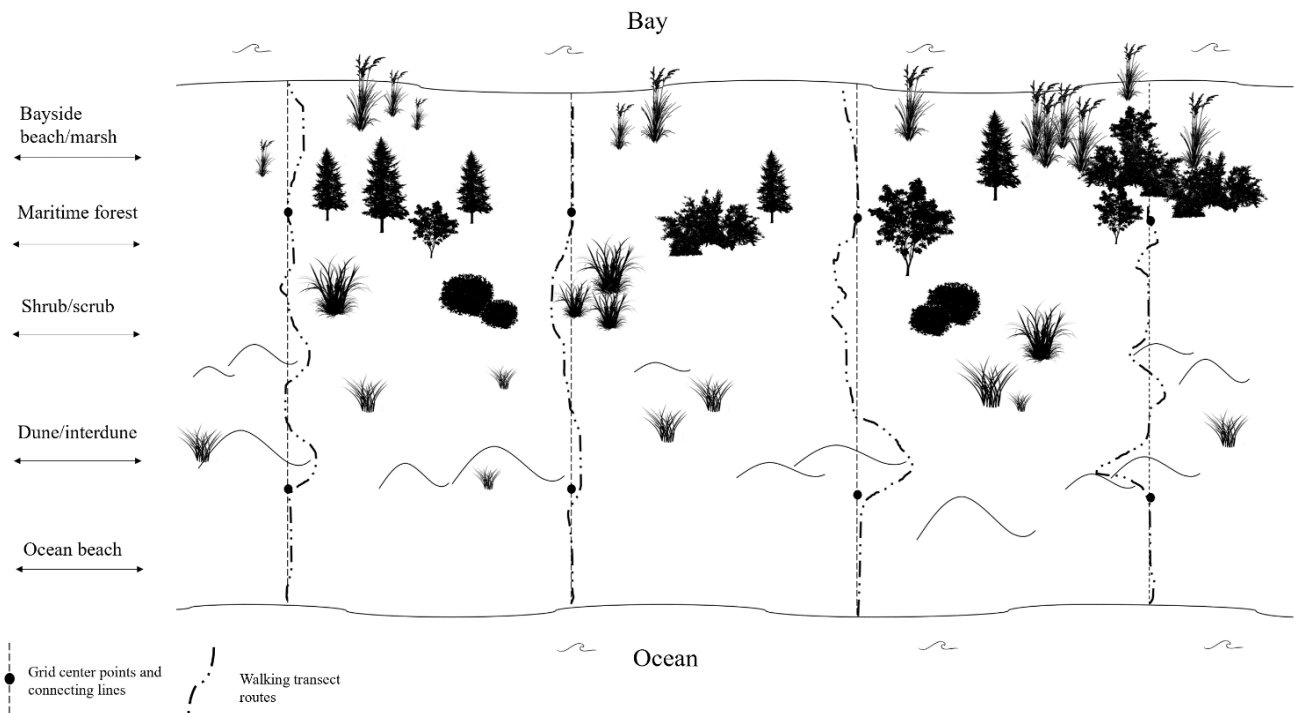


Figure 19. Diagram illustrating habitat types and transect surveys conducted on Fire Island, NY. We followed game trails, edges, and other likely wildlife travelways along general north-south transect lines based on a 300 m grid overlay of the island. These transects spanned all passable habitat types within each c. 300 m section of the study area.

Table 14. 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 ($\geq$ 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.

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 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 for 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 20**). 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 for 2020: Cameras were not deployed in 2020.

Methods for 2021: Cameras were not deployed in 2021 as they are not required to accomplish current objectives. We analyzed images from 2015 – 2019 to understand changes in the mesopredator community resulting from the sarcoptic mange outbreak within the red fox population with the goal of understanding how the mange-related decline in foxes might have impacted other predators of piping plovers.

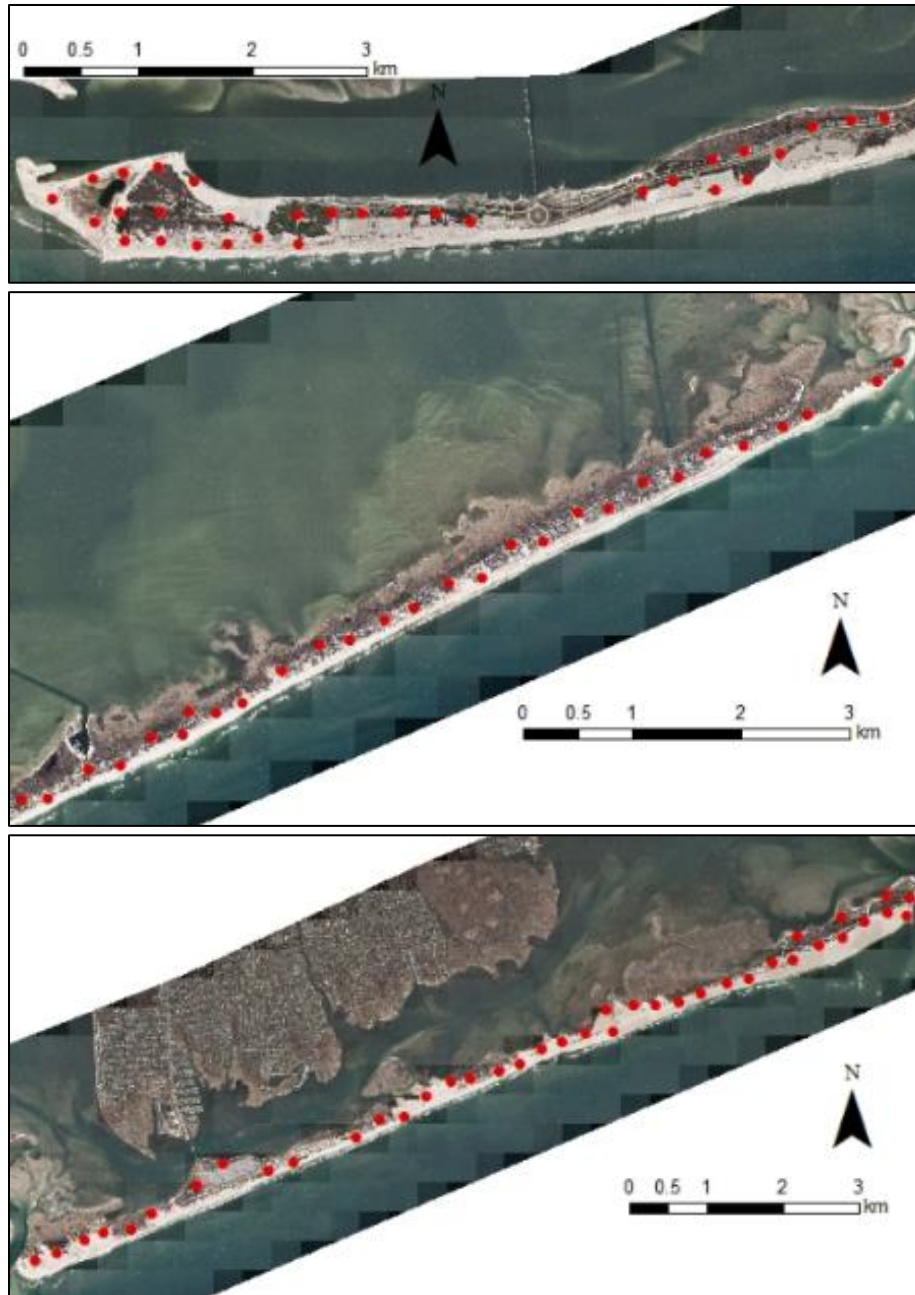


Figure 20. Approximate camera station locations (red dots) at Robert Moses State Park (top), Fire Island National Seashore (middle), and Smith Point County Park (including Old Inlet East; bottom) on Fire Island NY, 2015 – 2019. Remote camera surveys were conducted in September – February of each year, 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.

Red fox numbers, density, and distribution on Fire Island

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 both the target species and 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 (**Figure 21**).

Category and criteria	Sample photo
<u>No signs of mange:</u> No hair loss or skin crusting visible. Any visible lesions appear to be minor cuts or scars.	
<u>Mild signs of mange:</u> Hair loss, skin crusting, and/or lesions limited to one or two small areas of the body.	
<u>Moderate signs of mange:</u> Hair loss, skin crusting, and/or lesions on multiple areas of the body. Hair loss on tail often results in a “whip-like” appearance.	
<u>Severe signs of mange:</u> Hair loss, skin crusting, and/or lesions on large areas of the body. Hair loss on tail often results in a “whip-like” appearance.	

Figure 21. Visual classification scale used to describe the severity of sarcoptic mange symptoms in red foxes during monitoring activities conducted in 2015 – 2019 on Fire Island, NY.

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). 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. 4.1.2 (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 (**Figure 21**). 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

Between 2015 – 2018, we captured and ear-tagged 41 red foxes. We outfitted 33 of these red foxes with GPS collars (**Table 15**), 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 15, Table 16**) and other evidence of fox presence (e.g., sightings, scat) have been sparse (

).

Our analyses demonstrated that frequency of fox scat sampled from transects is capable of indexing fox presence ($F = 10.82$, $P = 0.02$). Based on the association of known fox scat and previously known fox populations

(2015 – 2018), we cautiously estimate three adult red fox were in the study area in 2021 (**Table 15**). Based on scat observed during transect surveys, most fox activity was likely in Robert Moses and the West Wilderness. We did have one sighting of a red fox within Smith Point on 16 June and found three scat that may be from fox (out of 54 total scat samples from Smith Point, c. 6%).

Table 15. Red fox abundances at each study site from 2016 – 2021 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).

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
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
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

^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

We observed pronounced decreases in reproduction in the breeding seasons following the detection of mange at each site (**Table 16**). 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 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.



Table 16. Number of red fox dens and mean ($\pm$ SE) number of kits per breeding group for each site monitored. 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 on Fire Island, NY in April – August of 2016 – 2021. These data appear bold and italicized.

Site	Year											
	2016		2017		2018		2019		2020		2021	
	Dens	Kits	Dens	Kits	Dens	Kits	Dens	Kits	Dens	Kits	Dens	Kits
Robert Moses + Lighthouse Tract	4	3.5 ($\pm$ 0.5)	4 ^a	2.8 ($\pm$ 0.5) ^a	0 ^b	---	0 ^c	---	0	---	0	---
West Wilderness	4	3.6 ($\pm$ 0.9)	2	1 ($\pm$ 0)	0 ^b	---	0 ^c	---	0	---	0	---
Smith Point + Old Inlet East	0 ^c	---	0	---	0	---	0	---	0	---	0	---
Total study area	8	3.5 ($\pm$ 0.6)	6	2.2 ($\pm$ 0.5)	0	---	0	---	0	---	0	---

^aMange first detected at that site

^bDuring mange outbreak at that site

^cAfter mange outbreak at that site

Red fox distributions

Throughout 2021, we observed 312 signs of mammal activity during the island-wide transects. We had three instances where an observer saw an adult red fox (June 3 and 6 at Robert Moses, and June 16 at Smith Point headed toward SPTE). We also received confirmation from NPS partners (J. Raphael, pers. comm.) of a red fox in the vicinity of Watch Hill on the Seashore in July 2021. Suspected fox fur and a skull were also located within Robert Moses in May. Although no breeding occurred in 2021, we also located an old, inactive den at Smith Point.

Tracks ($n = 165$, 52.9% of transect observations) and scat ($n = 141$, 45.2% of transect observations) represented most evidence of mammal activity observed during transect surveys (**Table 17**). We observed three partially-decomposed cat carcasses while walking transects at Smith Point in the week of 10 – 16 May 2021. We did not find any evidence that the frequency of red fox, coyote, and feral cat tracks differed between the first and second transects (Fisher's exact test p -value ≥ 0.16), though we did detect significantly fewer red fox scat and more feral cat scat during the second transect survey at the West Wilderness (West Wilderness Fisher's exact test p -value = 0.02, all other sites p -value ≥ 0.50). Red fox tracks were observed most frequently in Robert Moses and Smith Point, and in equal frequency as feral cat in the West Wilderness (**Table 17a**). The greatest frequency of identifiable cat scat occurred at Smith Point, representing 83.3% of scat collected in this area ($n = 45$ of the 54 samples; **Table 17b**). Feral cats also represented the plurality of field-identifiable scat samples collected at Robert Moses and West Wilderness; however, these sites also had a large percentage of unidentifiable scat that may be either felid or canid in origin.



Table 17. Total detections of canid and felid **(a)** tracks and **(b)** scat observed during transect surveys conducted in 2021 across Fire Island, NY.

(a) Tracks							
Site	Red fox	Coyote	Domestic dog	Feral cat	Unknown canid	Unknown	Total
Robert Moses + Lighthouse Tract	39 (52.7%)	7 (9.5%)	3 (4.1%)	11 (14.9%)	14 (18.9%)	0 (0.0%)	74
West Wilderness	4 (50.0%)	0 (0.0%)	0 (0.0%)	4 (50.0%)	0 (0.0%)	0 (0.0%)	8
Smith Point + Old Inlet East	67 (80.7%)	0 (0.0%)	0 (0.0%)	16 (19.3%)	0 (0.0%)	0 (0.0%)	83
Total	110 (66.7%)	7 (4.2%)	3 (1.8%)	31 (18.8%)	14 (8.5%)	0 (0.0%)	165

(b) Scat							
Site	Red fox	Coyote	Domestic dog	Feral cat	Unknown canid	Unknown	Total
Robert Moses + Lighthouse Tract	5 (10.9%)	6 (13.0%)	0 (0.0%)	18 (39.1%)	3 (6.5%)	14 (30.4%)	46
West Wilderness	6 (14.6%)	4 (9.8%)	0 (0.0%)	13 (31.7%)	4 (9.8%)	14 (34.1%)	41
Smith Point + Old Inlet East	1 (1.9%)	3 (5.6%)	1 (1.9%)	45 (83.3%)	2 (3.7%)	2 (3.7%)	54
Total	12 (8.5%)	13 (9.2%)	1 (0.7%)	76 (53.9%)	9 (6.4%)	30 (21.3%)	141

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 22a, Table 15**). 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 22b, Table 15**). 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 22c, Table 15**). 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.

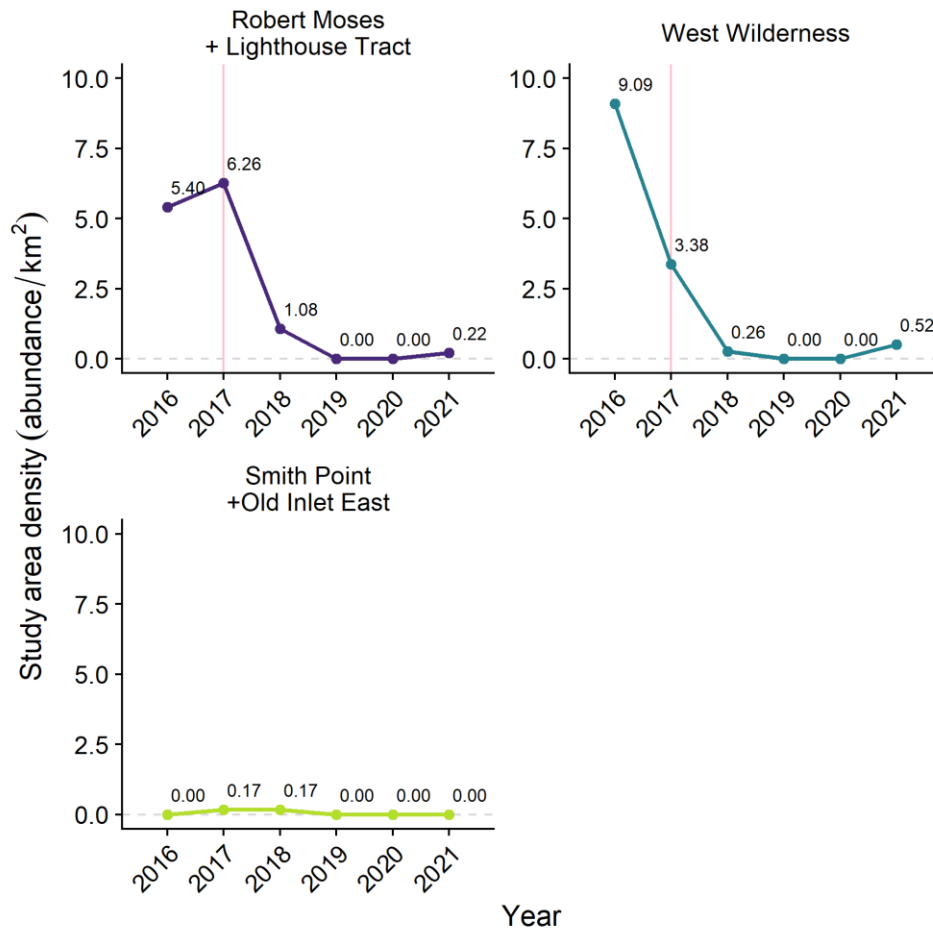


Figure 22. Estimated study area density of red fox for each year of fox monitoring at Fire Island, NY. Red fox density was calculated as the minimum red fox abundance (**Table 15**) per study site area (4.63 km² for Robert Moses State Park + Lighthouse Tract, 3.85 km² for the West Wilderness, and 5.94 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) 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).

Our transect survey results for 2021 suggest that a slight increase in red fox abundance and density though the potential trajectory for population recovery of red fox (and its future impact on nesting plovers) remains unclear. Despite the observed low abundance, reproduction, and density, there is typically high occupancy at sites when fox 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 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 2006).

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.

We found that 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.

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 island's 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 island's 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

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, grey 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).

In 2021, 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

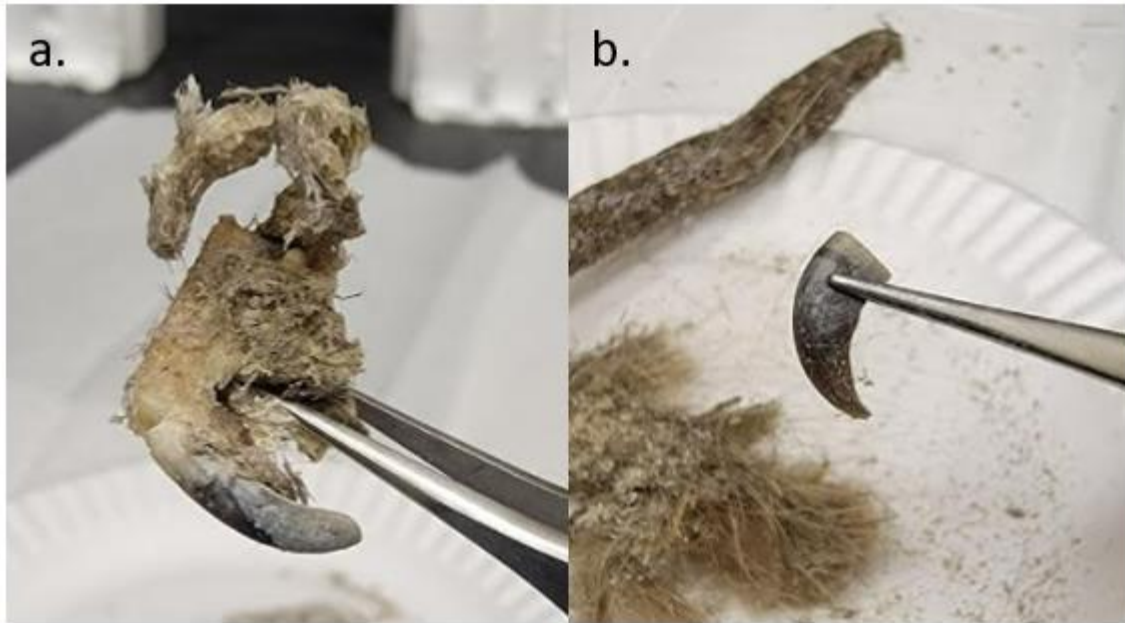
We collected 260 potential red fox scat samples while monitoring dens ($n = 53$), during transect surveys ($n = 181$), and opportunistically ($n = 26$), representing 62.7% of the 415 scat samples collected since 2016 (**Table 18**). We previously collected DNA subsamples from 267 of the scat samples for genetic verification of source species, and preserved 137 subsamples from 2021 for future genetic testing. The remaining samples were not suitable for genetic analyses due to age, biodegradation, or damage sustained during field transport.

Table 18. All scat samples and suspected contributor species collected at Fire Island, NY.

Species	Sampling method			Total
	Den survey	Transect	Opportunistic	
Coyote	0 (0.00%)	10 (76.92%)	3 (23.08%)	13 (3.12%)
Dog	0 (0.00%)	2 (100.00%)	0 (0.00%)	2 (0.48%)
Red fox	53 (20.38%)	181 (69.62%)	26 (10.00%)	260 (62.35%)
Cat	0 (0.00%)	63 (79.75%)	16 (20.25%)	79 (18.94%)
White-tailed deer	0 (0.00%)	2 (100.00%)	0 (0.00%)	2 (0.48%)
Unknown canid	0 (0.00%)	8 (88.89%)	1 (11.11%)	9 (2.16%)
Unknown	1 (1.92%)	46 (88.46%)	5 (9.62%)	52 (12.47%)
Total	54 (12.95%)	310 (74.82%)	51 (12.23%)	417

Fecal dietary synopsis

We did not find any identifiable piping plover remains (e.g., feathers, bands) in any scat samples collected in 2021. We suspect scavenging of a subadult mute swan (*Cygnus olor*; **Figure 23a**) and northern raccoon (*Procyon lotor*; **Figure 23b**) by canids at Fire Island.



Photos: K. Lapenta

Figure 23. Suspected **(a)** subadult mute swan (*Cygnus olor*) claw and **(b)** northern raccoon claw from canid scat samples collected in 2021.

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. 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 – 2019.

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 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 exclosures in 2018 (Hermanns et al. 2018), we have not found any clearly identifiable piping plover remains in 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). We found items resembling egg shells in three scat samples, but could only confidently identify them as bird egg shell remains in one instance. 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 and the western portion of Westhampton Island nearly tripled since Peterson et al. (2021) and the conclusion of this study, with 32 breeding pairs in 2013 and 92 breeding pairs in 2021. 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 were limited in scope, the past discovery of adult piping plover leg bands in a domestic cat scat sample highlights the potential impacts of free-ranging cats in shorebird nesting areas and the potential need for including domestic cats in predator management considerations (Walker et al. 2020, Black 2021). Both nest abandonment and loss of chicks have been attributed to feral cats previously on Fire Island (Winter and Wallace 2006). Moreover, on adjacent Westhampton Island, feline predation of piping plovers and nest abandonment have been reported (Winter and Wallace 2006, Cohen et al. 2009).



Community interactions on Fire Island



Investigation of mesopredator changes and their predicted effects to piping plover²

Objective P3: Quantify the interactions of predators with shorebird nests and chicks

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

Introduction

Globally, prototypical top predators are in decline as a result of human persecution. In systems where top predators are extirpated, the complete loss of top-down consumptive (i.e., predation) and non-consumptive effects (i.e., perceived predation risk) can reverberate through food webs (Ripple et al. 2016). An iconic example of the cascading impacts stemming from the loss of a top predator occurred at Yellowstone National Park, WY where trembling aspen (*Populus tremuloides*) recruitment was lost in the absence of gray wolves (*Canis lupus*; Ripple and Larsen 2000, Fortin et al. 2005).

Following the disappearance of a top predator, smaller predator populations within the ecological community can expand in the absence of consumptive (i.e., predation) and non-consumptive (e.g., competition) top-down pressures, a process referred to as mesopredator release (Soulé et al. 1988, Crooks and Soulé 1999). This may result in top-down limitations of prey populations. In New Jersey, for example, the removal of red fox enabled the expansion of American mink which then depredated nesting piping plovers (Stantial et al. 2021). In other more extreme cases, an increase in predation pressures by multiple mesopredators that share prey may result in species extirpations (Crooks and Soulé 1999). Managing predator-prey interactions can become especially complex in systems with non-native mammalian predators (Courchamp et al. 1999); asymmetric management of predators may enable generalist omnivores to expand their dietary niche (Rayner et al. 2007), while a reduction of generalist omnivores as prey will lead to increased predation pressures elsewhere in the system (Murphy and Bradfield 1992). Thus, it is vital to understand the complex nature of predator guilds for holistic management.

Barrier islands, like those along the U.S. Atlantic Coast, offer important breeding and nesting habitat for threatened and endangered shorebirds. These same areas tend to be popular vacation destinations, and anthropogenic food subsidies attract mammals such as red fox, northern raccoon, Virginia opossum (*Didelphis virginiana*), and feral cats (Prange et al. 2003, Shapira et al. 2008, Newsome et al. 2015), all of which are known to depredate nesting shorebirds (Patterson et al. 1991, Houghton 2005, Black 2021, Black et al. In Revision). As a relatively large and generalist predator, red fox likely play a key role in regulating other mammal populations within this system.

In September of 2015, a red fox with widespread hair loss and skin lesions characteristic of sarcoptic mange was observed at Smith Point County Park on Fire Island, NY. The mange outbreak eventually spread to the larger,

² **Cite chapter as:** Wails, C. N., H. A. Bellman, K. M. Black, S. G. Robinson, K.M. Walker, S.M. Karpanty, and J.D. Fraser. 2021. Investigation of mesopredator changes and their predicted effects to piping plover. Pp. 72 – 80. 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.

western portion of Fire Island, suppressing though not eliminating the Fire Island fox population. Here, we evaluate evidence that the reduction of red fox altered the mesopredator community in the years after the mange outbreak. We hypothesized that red foxes are capable of exerting predatory or competitive pressures that suppress activity of other mesopredators. We predicted that smaller mesopredators would be detected more frequently after the sarcoptic mange outbreak. We also predicted that diel activity of smaller mesopredators would increase during timeframes fox had been active prior to the mange outbreak.

Methods

Mesopredator presence and detection rates

Using remote camera data collected from 2015 – 2019 (see Red fox general field methods), we quantified detection rates and activity periods of red fox, northern raccoon, feral cat, and Virginia opossum per 100 trap nights using independent detections of each species across the three main study areas. We considered detections to be independent when individuals of the same species were detected > 30 min apart (Brook et al. 2012, Walker et al. 2020, Black 2021), and did not attempt to consider species abundance here. We present detection rates relative to time (in years) to or since the mange outbreak at that site.

We then analyzed species' presence and detection rates separately using generalized linear mixed-effects models. As we had few observations of Virginia opossum, we excluded this species from presence and detection rate analyses. To quantify presence, we recorded whether each of the four mesopredators of interest was detected in a 30 min period at each camera station. For detection rates, we reduced our observations to only include instances when at least one individual of the species was detected during the 30 min period. We fit a generalized linear mixed-effect models via the *glmmTMB* R package (Brooks et al. 2017). Our analysis of species presence had a binomial error distribution (as the dependent variable, presence, was a binary response) with a logit link function while our analysis of detection rate had a Gamma error distribution (as the dependent variable, rate, was a non-zero positive response) with a log link function. For both models, we included the mesopredator species, mange outbreak period (i.e., years before outbreak, years during outbreak, and years after outbreak), and their interaction as model predictors. Because we deployed camera stations at approximately the same locations in multiple years, we included site and camera station as a nested random effect. We checked all standard model diagnostics, including tests for temporal and spatial autocorrelation. We then used likelihood ratio tests (Bolker et al. 2009) to test the significance of the interaction term, and we performed post-hoc Tukey's HSD when significant effects were detected to identify which species' presence and detection rates had changed. Statistical significance was assessed at $\alpha = 0.05$.

Mesopredator activity periods

We tested whether species' activity periods changed relative to the mange outbreak. Because our study spanned autumn and winter months with differing day lengths, we converted clock times to 'sun times,' wherein time was converted to radians and sunrise was set to $\pi/2$ and sunset was $3\pi/2$ (Nouvellet et al. 2012). We visualized activity periods as circular kernel density estimates via the *overlap* R package (Meredith and Ridout 2021). We tested for significant shifts within species' activity periods before, during, and after the mange outbreak using Kolmogorov-Smirnov tests. We also quantified the temporal overlap of activity periods (i.e., the

coefficient of overlap, Δ), which ranges from 0 (i.e., no activity overlap) to 1 (i.e., complete activity overlap; Ridout and Linkie 2009, Meredith and Ridout 2021).

Results

Mesopredator presence and detection rates

From late October 2015 to early February 2019, we had 13,491 independent detections of mesopredators. Across the four mesopredator species of interest, red foxes were detected the most frequently (**Figure 24**).

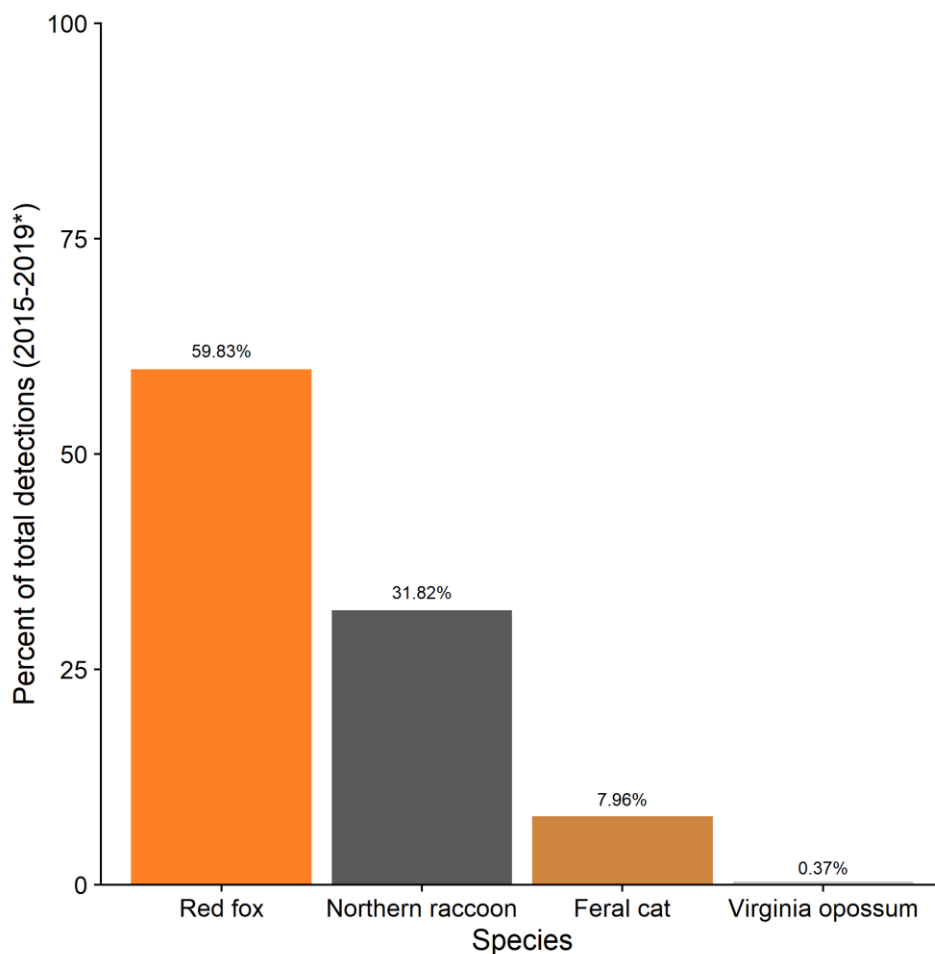


Figure 24. Percent of total detections for the four target mesopredators at Fire Island from 2015 – 2019* (*camera trapping ended by February of 2019).

Camera trapping results were reflective of known declines of red fox following outbreaks of sarcoptic mange (Black 2021; **Figure 25**). Red fox detection at camera stations plummeted in the years after mange outbreaks were observed. Mange was not observed in northern raccoon, feral cats, or Virginia opossum.

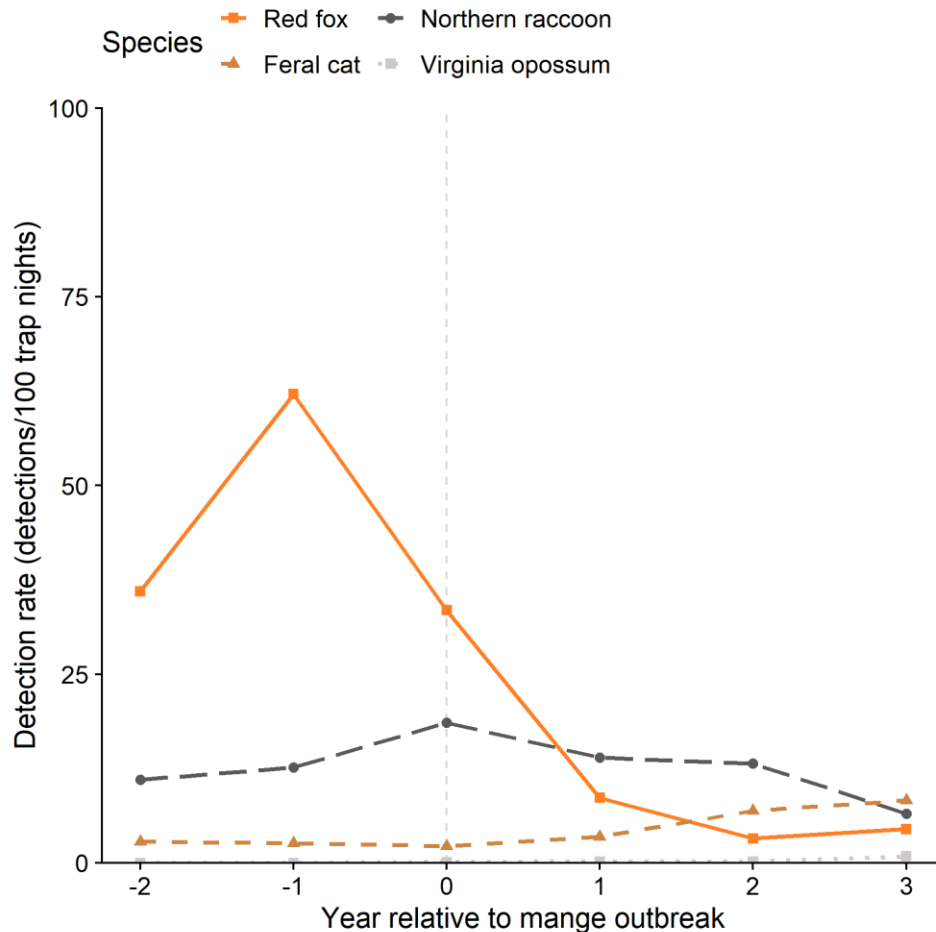


Figure 25. Detection rates (expressed as the number of independent detections per 100 trap nights) across all study sites at Fire Island during the study period. Detection rates are presented relative to the year of the mange outbreak at the site; negative values indicate years prior to reports of sarcoptic mange and positive values indicate years after mange was first observed.

We found significant changes in mesopredator presence and detection rates relative to the mange outbreaks across Fire Island (**Table 19**). Prior to mange outbreaks, fox presence and detection rates were significantly greater than all other mesopredators across Fire Island, followed by raccoon and cat (**Figure 26, Figure 27**). During the mange outbreak, both presence and detection rates of raccoons exceeded that of cats (**Figure 26**) though only detection rates were significantly greater in raccoon than fox (**Figure 27**). While raccoon presence and detection rates have shown a slight decrease since the mange outbreaks (**Figure 25**), these declines are non-significant (**Figure 26, Figure 27**). Conversely, cat presence and detection rates have significantly increased throughout the study period (**Figure 26, Figure 27**), and was showing signs of exceeding that of other mesopredators when camera trapping stopped in 2019 (**Figure 25**).

Table 19. Results of generalized linear mixed-effects modeling for **(a)** presence and **(b)** detection rates. ‘×’ indicates an interaction.

(a) Presence			
Predictor	Wald’s χ^2	df	P-value
Species	185.83	2	< 0.001
Outbreak period	17.53	2	< 0.001
Species × outbreak period	92.47	4	< 0.001

(b) Detection rate			
Predictor	Wald’s χ^2	df	P-value
Species	202.19	2	< 0.001
Outbreak period	100.05	2	< 0.001
Species × outbreak period	180.96	4	< 0.001

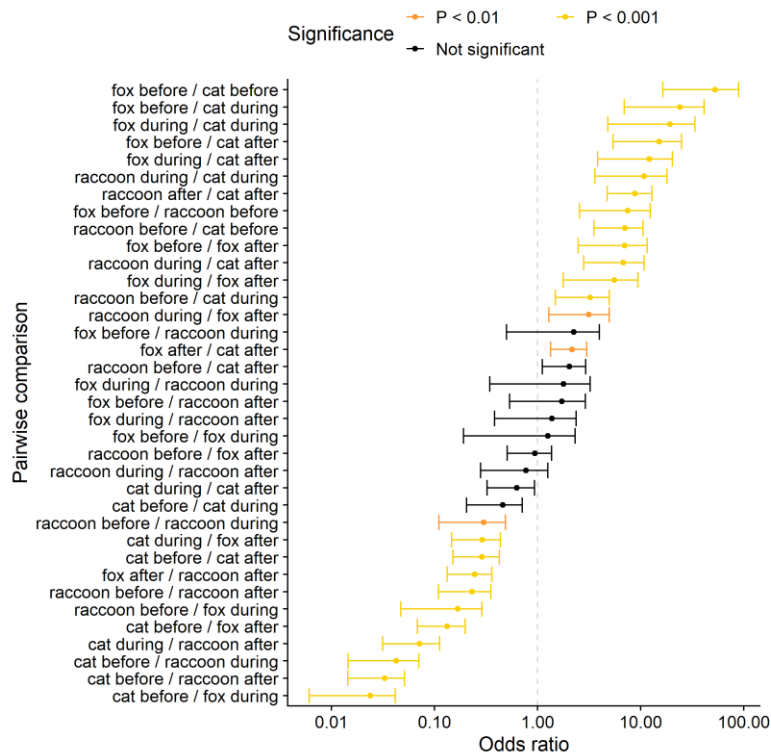


Figure 26. Odds ratios of pairwise comparisons illustrating the likelihood of mesopredator presence before, during, and after mange outbreaks from post-hoc Tukey’s HSD (log₁₀ scale). Odds ratios express the probability of mesopredator presence in the pairwise comparison. Odds ratios > 1 demonstrate that the presence of the first mesopredator during the given timeframe of the pair is more likely to occur than the second; odds ratios < 1 demonstrate that the presence of the second mesopredator of the pair is more likely. Error bars depict 95% confidence intervals.

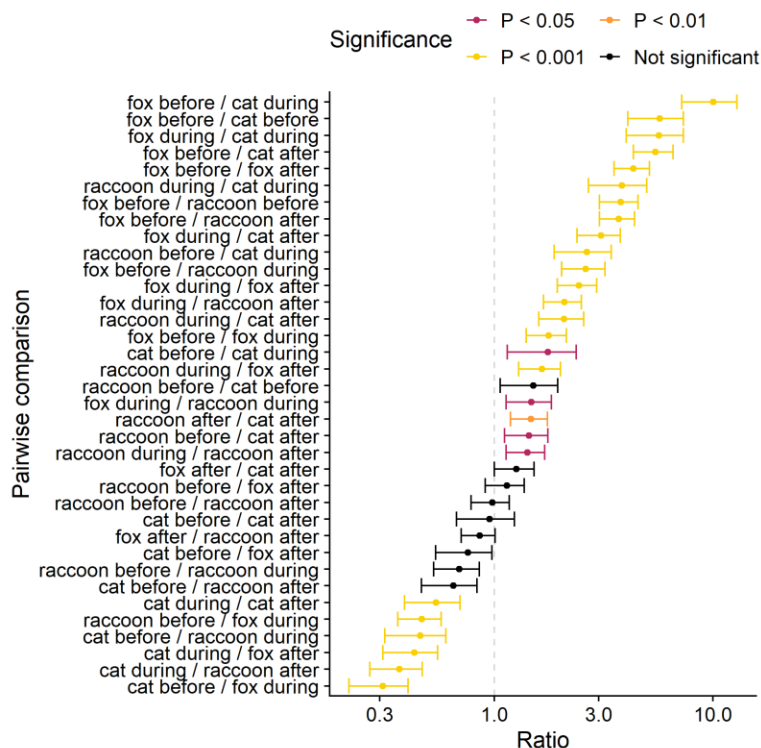


Figure 27. Ratios of pairwise comparisons of mesopredator detection rates observed before, during, and after mange outbreaks from post-hoc Tukey's HSD ($\log_{10}$ scale). Ratios express the relative magnitude difference of mesopredator detections in the pairwise comparison. Ratios > 1 demonstrate that the detection rates of the first mesopredator during the given timeframe are x-fold greater than detection rates of the second; ratios < 1 demonstrate that the detection rates of the second mesopredator of the pair is greater. Error bars depict 95% confidence intervals.

Mesopredator activity periods

We found significant shifts in red fox activity before, during, and after the mange outbreaks with a more uniform distribution of activity across the diel cycle during and after the outbreak than before (Kolmogorov-Smirnov $D = 0.06 - 0.11$, $p < 0.005$ for all comparisons). Raccoon activity changed to become more active near dawn and at night, rather than just at night, at the onset of the mange outbreak ($D = 0.07$, $p = 0.005$), but did not change after ($D = 0.02$, $p = 0.81$). Shifts in cat activity were subtle, only showing a significant difference before and after the outbreak, again with activity becoming more distributed throughout the diel cycle rather than at night ($D = 0.13$, $p = 0.005$), but not when comparing activity before and during ($D = 0.08$, $p = 0.57$), nor during and after ($D = 0.07$, $p = 0.50$).

We also detected fairly high overlap in activity between species (Δ range: $0.63 - 0.92$). The extent of overlap between red fox and northern raccoon, as well as feral cat and northern raccoon, generally decreased through the study period, while the activity overlap of red fox and feral cat was somewhat more variable (**Figure 28**).

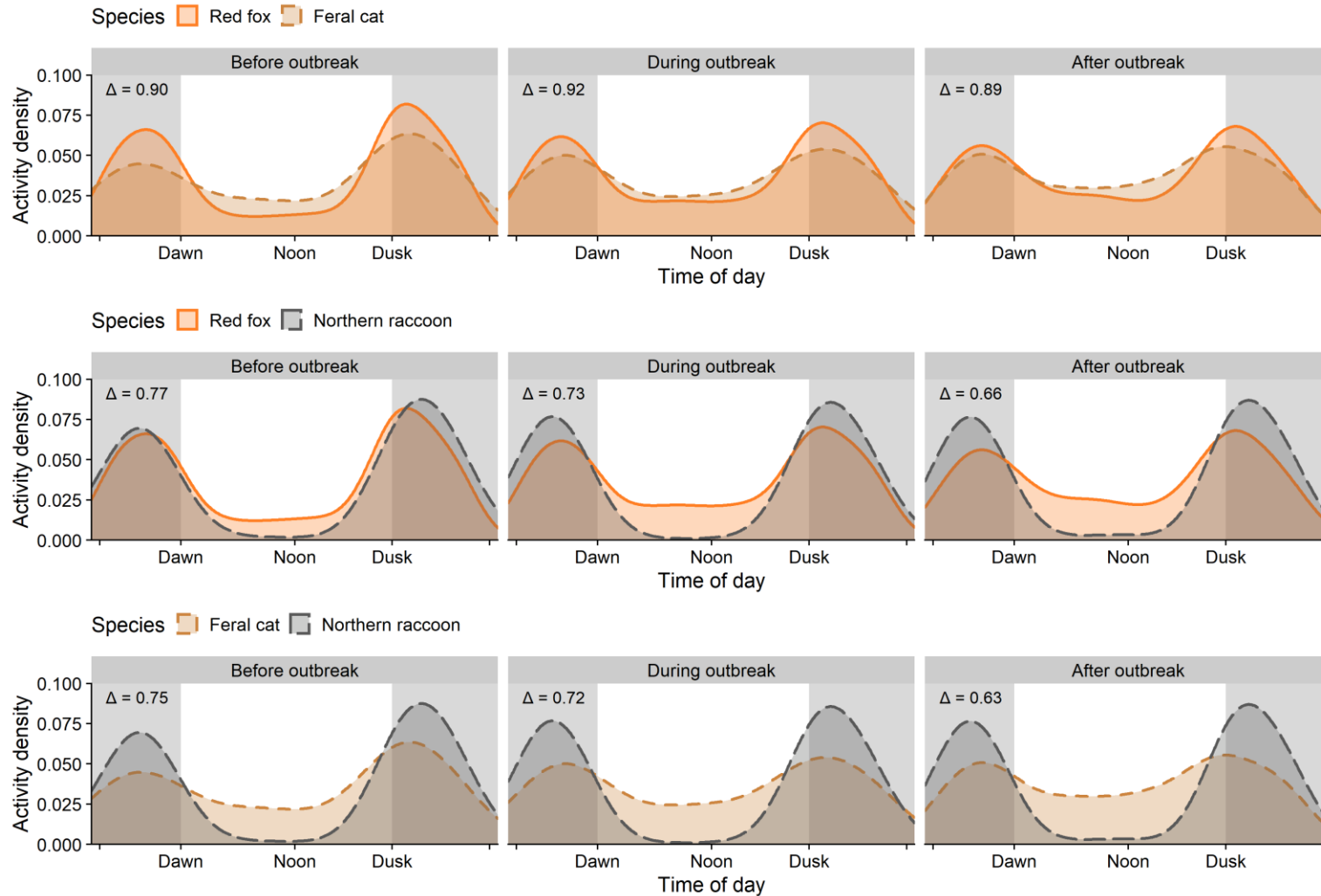


Figure 28. Activity periods of red fox, northern raccoon, and feral cat before, during, and after the mange outbreak at Fire Island during the 2015 – 2019 camera trapping study.

Discussion

While we did find significant shifts in detections and activity periods relative to mange outbreaks, the rate at which changes occurred was not as rapid as observed in other studies (e.g., Rayner et al. 2007, Stantial et al. 201). Given that the mange outbreak did not occur at the same time across the study island and that fox were observed to expand their range into areas experiencing mange outbreaks (Walker et al. 2020, Black 2021), even periodic presence of fox may have been sufficient to prevent a pronounced release of raccoon and feral cat. Alternatively, it may be that red fox simply do not exert top-down pressure to raccoons, and rather both fox and raccoon presence work in concert to modulate feral cat populations. Indirect evidence from other predator-control studies have not documented an increase in raccoon following decreases in fox within the same area (Sovada et al. 1995, Frey and Conover 2007). However, there is mixed evidence that populations of these species change dramatically in the presence of coyotes (*Canis latrans*; Crooks and Soulé 1999, cf. Gehrt and Clark 2003).

In February of 2020, a coyote was observed at Robert Moses and likely has had, at least intermittent presence throughout 2021. Coyotes have been shown to suppress both fox and feral cats, and may influence raccoon populations (Crooks and Soulé 1999, Gehrt and Clark 2003, Levi and Wilmers 2012). Yet, coyotes are also important predators of white-tailed deer (Decker et al. 1992, Saalfeld and Ditchkoff 2007, Kilgo et al. 2010) which are abundant on Fire Island (Kilheffer et al. 2019a, Kilheffer et al. 2019b). It remains unclear whether coyotes will have a direct (consumptive) or indirect (competitive or via landscape of fear) effect on the other mesopredators at Fire Island, or no effect at all.

Implications for nesting plovers

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 (Roche et al. 2010). Disturbance also can result in a change in behavior due to the perceived risk of predation (Frid and Dill 2002). In the case of plovers and other shorebirds, disturbance can influence the incubating or brooding behavior of adults and foraging behavior in chicks, both of which can reduce chick survival (Weston and Elgar 2005, Hillman et al. 2015, DeRose-Wilson et al. 2018).

The detected predator community varied annually within the study area and understanding how to adapt management strategies to account for such a dynamic system is vital. There was a significant avian predator presence in the study area, and managers should consider the implications of these findings when deciding to exclose nests and for how long to leave exclosures up following hatch. Raptors have been known to adapt their hunting strategies with the introduction of artificial perches or by cuing in on nests (Reinert 1984, Murphy et al. 2003, Niehaus et al. 2004), and if exclosures are being used as eating or hunting perches in the study area, leaving them up after hatch may be detrimental to plovers in the area. Additionally, there is a potential for the red fox population to recover in the study area and recommence exerting predatory pressures on the plover population dynamics (Robinson et al. 2020a, Walker et al. 2020). It is crucial to continue monitoring predators via nest cameras as the community changes in the following years and, ultimately, how these changes may influence the plover population in the future. Finally, using cameras to assess disturbance to plovers by linking behavioral changes of the incubating adult to capture events could lead to a better understanding of the indirect effects that predators, non-predators, and humans have on the plovers in the study area (Hillman et al. 2015, DeRose-Wilson et al. 2018).



Future directions

In 2022, we plan on returning to remote camera trapping (at nests) to monitor mesopredator presence and activity. We are also working on identifying cats based on their uniquely-identifying coat patterns to estimate abundance (Sparkes et al. 2021) relative to mange outbreaks and with the arrival of coyote to Fire Island.

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Appendices



Appendix 1. 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 – 4.11 ft in elevation, based on the Astronomical High Water at NOAA Moriches Inlet tidal station. The bayside intertidal range was 0 – 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. Recent research products

Manuscripts

Black, K. M., S. M. Karpanty, J. D. Fraser, B. N. Sacks, and D. H. Catlin. In Revision. Red fox dietary ecology in a developed coastal ecosystem. *Journal of Mammalogy*.

- *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.*

Robinson, S., H. Bellman, K. Walker, D. Catlin, S. Karpanty, S. Ritter, and J. Fraser. In Press. Adult Piping Plover habitat selection varies by behavior. *Ecosphere*.

- *Compare changes in numbers, density, and distribution of Piping Plovers in various habitats on Fire Island.*
- *Compare Piping Plover nest and chick survival rates in various habitats across Fire Island.*
- *Compare the relative contribution of local growth (recruitment) vs. immigration in population growth associated with new habitats. Determine habitat correlates of population change.*

Kwon, E., S. Robinson, C. E. Weithman, D. H. Catlin, S. M. Karpanty, J. Altman, T. R. Simons, and J. D. Fraser. 2021. Contrasting long-term population trends of beach-nesting shorebirds under shared environmental pressures. *Biological Conservation*. 261:109178.

- *Compare changes in numbers, density, and distribution of Piping Plovers in various habitats on Fire Island.*
- *Compare Piping Plover nest and chick survival rates in various habitats across Fire Island.*
- *Compare the relative contribution of local growth (recruitment) vs. immigration in population growth associated with new habitats. Determine habitat correlates of population change.*

Conference presentations

**indicates presenting author(s)*

Dorsey, S. *, S. Ritter, S. Robinson, K. Walker, C. Wails, D. Catlin, S. Karpanty, and J. Fraser. "Plover-view": The role of viewshed in Piping Plover nest-site selection. *The Wildlife Society 28th Annual Conference*. November 1-5, 2021.

- *Compare changes in numbers, density, and distribution of Piping Plovers in various habitats on Fire Island.*

Theses and dissertations

Black, K. 2021. *Red fox ecology and interactions with Piping Plovers on Fire Island, New York* [Ph.D. dissertation]. Virginia Polytechnic Institute and State University, Blacksburg, VA.

- *Produce an index of fox abundance to understand the recovery rate of the fox population.*
- *Estimate red fox reproductive success and relate it to population growth during the recovery from the mange outbreak.*
- *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.*



- *Use information collected in objectives 1 – 3 to make recommendations on utility of, and recommended methodologies for, management of red fox populations in the study area and in other coastal habitats.*