

2017 Annual Report

Piping Plover and Fox Monitoring on Fire Island, NY

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Cover photo was provided by Amanda Carey

Green flag 'TUP' was banded at Fire Island, NY in June 2016 and was photographed at Robert Moses State Park in September 2016

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Executive Summary

This is a report summarizing Virginia Tech's Effectiveness Monitoring of certain conservation measures required under the Biological Opinion for the Fire Island Inlet to Moriches Inlet Stabilization Project in 2017. Where appropriate and possible, we provide cumulative results to facilitate inspection of trends. In April–August we conducted field monitoring of piping plovers and red foxes. In January–March and September–December we conducted data Q&A, data analyses, and project support (e.g., renting houses, equipment maintenance, technician hiring, etc.).

Piping Plover Monitoring Summary

During the 2017 piping plover breeding season, we banded 36 adults and 129 chicks. Of the 62 nests detected, 40 hatched (66% of exclosed nests and 42% of unexclosed nests). Of the 135 chicks that hatched, 65 fledged (48%), resulting in a reproductive output of 1.44 fledges/pair for the entire study area. When compared by management area, Robert Moses State Park productivity was 1.50 fledges/pair, Fire Island National Seashore productivity was 1.56 fledges/pair, Smith Point County Park productivity was 1.65 fledges/pair, and Cupsogue Beach County Park productivity was 0.50 fledges/pair. Overall, chicks had a 56% predicted probability of surviving and fledging. Chick survival and reproductive output remain low in the high human use area of Cupsogue County Park, However, reproductive output was higher in all sites in good years, such as 2016 and 2017, than in other years.

Piping Plover Habitat Use, Selection and Demography

Adult and fledgling piping plovers moved large distances (max = 35km) to access overwash, flood shoal and mudflat habitats, some of which were not protected from disturbance during migration or otherwise. In 2014–2017, of birds with an active nest, those with bay access were resighted nearer their nest ($\bar{x}$ = 117.8, SE = 14.5) than those without bay access ($\bar{x}$ = 196.8, SE = 28.6). There was no difference in the clutch masses of piping plovers that had bay access compared to those that did not have bay access. Brood home range size increased in size as broods aged. For the plover broods in 2017 that had at least one chick survive to fledging, we found no difference in home range size between broods that had direct access to the bay and those that did not. Over the four years of the study, we found that broods that had direct access to bay intertidal foraging within their nesting subsite had smaller home range size than broods that did not have such access. Mean distance between successive brood locations was smaller for broods with bay access than for other broods. Invertebrate samples in the bay beach had a greater mean arthropod catch than barrier interior or ocean beach invertebrate samples. For the treatment types, overwashes had greater invertebrate catches than the restoration areas and the breach fills.

We estimated that the amount of suitable piping plover habitat increased by 36% between 2010 and 2015. Engineered habitat produced 44% of all new suitable habitat after the storm, while 56% of new suitable habitat was created during Hurricane Sandy. Only 45% of 2015 suitable habitat was within protected areas. Piping plovers quickly began using storm-created and USACE plover restoration habitat. Overall, 83% percent

of immigrants, and 80% of local recruits nested in newly created habitats (i.e., overwashes, breach fill, or restoration areas). Restoration areas and manipulated overwashes supported the most natal recruits with all years combined, and the manipulated and natural overwashes supported the most immigrants. Individuals coming into the study area did not use pre-Hurricane Sandy habitat, and recruits selected for restoration areas, providing evidence that the overwashes and restoration areas are attracting new birds relative to the rest of the study area.

Despite increased habitat and immigration, the population increase in the study area since before the hurricane (50%) has been modest compared to other habitat creating events. Historic coastal populations following habitat creating events increased by as much as 450% by the fifth year following the event. Less than anticipated growth may have been related to generally low local and regional reproduction just before and immediately following the storm. Twenty-eight percent of adult birds die annually, and there must be enough reproduction to replace those just to maintain a stationary population. Without immigration, the population may have decreased as it did elsewhere in the NY-NJ recovery unit in 2012–2013, and remained stable or increased following 2014.

Habitat restoration areas were used by plovers; but the total number of breeding pairs that can be supported on these habitats remains to be estimated. We observed increasing pair densities and nest attempts in manipulated and natural overwashes following creation. Pair density and nest numbers in the restoration areas, however, were fairly constant despite increasing male site fidelity and nest success. Site fidelity was higher for males than females (except in the natural overwash in 2017). In the restoration areas, site fidelity has been unexpectedly low given the high nest and brood success over the last two years. Nest and brood success was highest in the natural overwash in 2015 (33% nest success, 33% brood success), and in the restoration areas in 2016 and 2017 (all 100%). Although nest numbers and success varied annually between the two restoration areas (New Made and Great Gun), this habitat did support nesting piping plovers. Vegetation management through mechanical or chemical removal is strongly recommended as all areas re-vegetate and habitat is degraded.

Although not described in this report, vegetation is rapidly colonizing plover habitat. It is not clear how much habitat will be available by 2018. Still, local and regional reproduction was high in 2017, so if unoccupied habitat remains, we may see an increase in breeding plovers in 2018. Although more work needs to be done, the results to date are consistent with the longstanding understanding that natural processes of overwash and breaching create and maintain piping plover habitat.

Red Fox Monitoring Summary

By extrapolating red fox densities to the area available between Fire Island Inlet and Old Inlet, we estimate a minimum red fox population size of 39 adults and 16 sub-adults as of August 2017. We confirmed the presence of at least one adult red fox at Smith Point County Park using temporary remote camera stations in March 2017, and continued observing signs of red fox presence in all sign surveys since that date. We did not find any evidence of breeding dens in Smith Point County Park or anywhere east of Old Inlet during this 2017 breeding season.

Starting in June 2017, we observed signs of sarcoptic mange (caused by the *Sarcoptes scabiei* mite) in red foxes at Robert Moses State Park. From June to August 2017, 3 GPS collared foxes died from confirmed or strongly suspected mange-related causes, and one additional collared fox and several unmarked foxes showed signs of infection west of Old Inlet. In December 2017, we observed early signs of mange in some foxes via remote camera photos in the Wilderness Area just west of Old Inlet and near Watch Hill. Our current fall camera grids will provide important information needed to assess the overall impact of this ongoing mange outbreak on the red fox population west of Old Inlet.

Red foxes were photographed at least once at every camera station at Fire Island National Seashore west of Old Inlet in Fall-Winter 2016, resulting in an occupancy of 1 in this management area. No red foxes were photographed east of Old Inlet during this time. Final analyses for camera station data from Robert Moses State Park are almost complete, and will be added to the next draft of this report. Preliminary analyses suggest very high occupancy in this area.

When the 50% and 95% core and home range areas of all GPS-collared foxes in Spring-Summer 2017 are overlaid, the resulting contours encompass all of Robert Moses State Park and the Fire Island National Seashore Lighthouse Tract, about half of the Fire Island National Seashore Wilderness Area, including all areas east of Bellport Beach where Virginia Tech monitors piping plovers, and several large areas in the villages. This GPS collar data, combined with our sign surveys and field observations of unmarked individuals, suggests that red foxes were ubiquitous throughout all piping plover management areas west of the Inlet during Spring-Summer 2017. We observed 2 GPS-collared males making multiple long distance movements between the westernmost end of Fire Island and the central villages in Spring-Summer 2017.

Breeding red foxes at all management areas used multiple den sites within a breeding season, and moved between dens as the season progressed. We observed an average of 3.25 kits per pair of red fox at Robert Moses State Park and the Fire Island National Seashore Lighthouse Tract. At Fire Island National Seashore, between Watch Hill and Old Inlet, we observed an average of 1.0 kit per pair of red fox.

Three of the 8 foxes that were GPS collared in 2017 at Robert Moses State Park and the Fire Island National Seashore Lighthouse Tract, and 3 of the 4 foxes at Fire Island National Seashore (between Watch Hill and Old Inlet) were alive at the time their collars dropped off or failed.

Interactions between Red Fox and Piping Plover

All piping plover nest, adult and brood locations west of Old Inlet in 2017 were located within the 95% kernel density home range of at least one GPS-collared fox. Based on GPS locations collected during target periods around nest exclosure and prior to hatching, collared foxes did not exhibit a clear spatial response to target events, such as nest exclosure, pipping, or hatching. Fox locations appeared to be correlated to den sites for breeding individuals, and resting sites for nonbreeding individuals, with little to no visible shift in locations before vs. after target events. The role of foxes in plover reproductive output requires continued study.

In 2016, when foxes were not present on Smith Point and Fire Island National Seashore east of Old Inlet, plover reproductive output (R_o) was very high (2.38 and 2 chicks/pair) more than twice the R_o in the other management areas, suggesting an important role for foxes. However, in 2016, nest success was very high in all management areas (probability of a nest surviving to hatch = 0.68–1.00 in the 4 management areas) suggesting the possible influence of a variable, perhaps weather, affecting nest success in a similar way in all areas. In contrast, chick survival in 2016 was quite high in the fox-free areas (0.73 and 0.84. in Fire Island Wilderness, and Smith Point respectively) compared to other areas (0.37 and 0.22, in Robert Moses State Park and Cupsogue County Park, respectively) consistent with the possibility of a strong influence of foxes on chick survival. The combination of high nest success with low chick survival in Robert Moses State Park and Cupsogue County Park resulted in a modest reproductive output in those areas (1.0, 1.1 Chicks/pair).

Fox numbers and movements in 2017 are still being analyzed but one or a few foxes were on Smith Point and the Fire Island National Seashore, east of Old Inlet. In 2017 the Fire Island management units had similar plover reproductive output, all substantially above the level needed for population stationarity (1.50–1.65). Recreational disturbance (DeRose-Wilson et al *in revision*) and habitat variability (Loefering and Fraser 1995, Elias et al. 2000, Cohen et al 2009), likely affect these results. As of this writing, the mange zoonotic seems to be decreasing the fox population west of old inlet, so next year's reproductive output in Robert Moses State Park may shed additional light on the effect of foxes on piping plover reproductive output.

The high density and widespread presence of foxes west of Old Inlet and presence of non-breeding floater foxes and long distance movements suggests that red fox eradication using normal methods would be difficult (if possible) and expensive.

Background Information and 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 USFWS North Atlantic Landscape Conservation Cooperative Beach Resiliency Project (Resiliency Project), and by 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 one cumulative annual report, to provide the best overview of conditions of the piping plover population, the habitat that supports it, the red fox predator population and its interactions with piping plovers. The Army Corps of Engineers 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.

In 2012, Hurricane Sandy struck the barrier islands of New York and New Jersey, creating multiple breaches and overwashes. Three breaches were created within our 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. Breaches and overwashes can result in creation of new foraging and nesting habitat. These alterations form intertidal flats that can serve as foraging habitat (Cohen et al. 2009). Piping plovers forage on terrestrial and marine invertebrates (Shaffer and Laporte 1994, Cuthbert et al. 1999, Majka and Shaffer 2008, Cisek 2013). The quantity of prey available can increase habitat quality and have an effect on the piping plover nesting densities (Cohen et al. 2009). Natal territories with ample high quality available foraging habitat, can contribute fitness advantages to offspring (Catlin et al. 2014). Plover chicks with access to high densities of terrestrial invertebrates, weigh more, have higher foraging rates and are more likely to survive (Cohen et al. 2009, Catlin et al. 2015). Prey abundance and location can be a contributing factor to nest site location, as plovers favor nesting in close proximity to moist substrates for availability to food sources during the nesting period. (Cohen et al. 2009). Habitat changes resulting from anthropogenic and natural forces have an impact on the piping plover population dynamics (Cohen et al. 2009, Catlin et al. 2015).

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 winter 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 piping plover (*Charadrius melodus*; threatened) and seabeach amaranth (*Amaranthus pumilus*; threatened); the BO for the FIMI included these species as well as the red knot (*Calidris canutus rufa*; threatened; USFWS 1995 and 2014). Anthropogenic alterations to habitat including dredging and 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 in 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 6 ha (15.8 ac) of bay side habitat south of Carter's Island.

A program of targeted monitoring to evaluate the efficacy of the BO conservation measures, developed by the USFWS, with the assistance of Virginia Tech, are the subject of this report. 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, the County of Suffolk, and the USACE to obtain the necessary permits to undertake this work.

In January–March and September–December we conducted data Q & A, data analyses, and project support (e.g., renting houses, equipment maintenance, technician hiring, etc.). In April–August we conducted field monitoring of piping plovers and red foxes. From our monitoring efforts of the piping plover from April through August 2013–2017, we present the results of our banding efforts, data on piping plover reproductive output in various habitats, clutch quality, chick behavior and responses to disturbance in different management units, and information on invertebrate prey abundance in different habitats.

The goal of this project is to provide a broader ecological understanding of the ways in which the breaches and USACE breach-fill 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, overwash areas, and other areas. Ultimately our results will help refine their understanding of the time frame and manner in which piping plover habitat develops and persists.

The following report of piping plover monitoring includes the following objectives:

- Objective 1: Compare changes in numbers, density, and distribution of piping plovers in various habitats on Fire Island
- Objective 2: Estimate the “effect distance” of newly created habitat
- Objective 3: Compare egg weights of birds with access to bay side intertidal flats with egg weights of birds without such access
- Objective 4: Compare piping plover nest and chick survival rates in various habitats across Fire Island
- Objective 5: Compare the relative contribution of local growth (recruitment) vs immigration in population growth associated with new habitats
- Objective 6: Compare prey densities at several habitat types
- Objective 7: Quantify the interactions of predators with shorebird nests and chicks

The following report of fox monitoring includes the following objectives:

- Objective 1: Quantify the abundance and occupancy of red foxes and relate changes in these parameters to changes in habitat and predator management activities.
- Objective 2: Assess habitat use versus availability by red foxes and any variation in habitat selection related to changing habitat composition and predator management.
- Objective 3: Quantify home range size, long-range movements, and variation in movement ecology of red foxes and relate to changing habitat composition and predator management.
- Objective 4: Estimate red fox reproductive success and adult survival, and variation in these population parameters, related to habitat composition and predator management.
- Objective 5: Quantify the interactions of red fox with shorebird nests and chicks.

- Objective 6: Utilize information collected in objectives 1-5 to make recommendation on utility of, and recommended methodologies for, management of red fox populations in the study area and in other coastal habitats.

Study Site

Our study area consisted of 27.1 km of Fire Island and Westhampton Island within the administrative boundaries 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). From 2013 to June 2015, the study area 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).

Prior to Hurricane Sandy, our 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 that was filled with a berm 9+ ft. National Geodetic Vertical Datum (NGVD) in elevation, 150 ft. wide and 2000 ft. long (presentation on Coastal Breach Closures, Bocamazo 2013). The Narrow Bay West breach was filled with a berm about 12 ft. NGVD in elevation (based on a completion drawing of Smith Point breach fill, 2013). 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 Dredge 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 as close as possible to Zeigler et al. 2017.

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

Habitat	Definition
Restoration Area	The New Made and Great Gun restoration areas, created between the 2014 and 2015 piping plover breeding seasons.
Overwash	Any area that overwashed during Hurricane Sandy. This is a broad overarching category that can be further defined into manipulated or natural overwash.
Manipulated Overwash	Areas of Fire Island that overwashed during Hurricane Sandy, but have been manipulated by breach filling, dune manipulation (including snow fencing and Christmas tree placement) or vegetation management.
Natural Overwash	Areas of Fire Island that overwashed during Hurricane Sandy, but have not been further modified.
Breach Fill	The areas of Fire Island and Cupsogue County Park that breached during Hurricane Sandy but were filled shortly after by USACE. Both breach fills fall within the boundaries of manipulated overwashes.
Manipulated Dune	Any dune or berm created by other than natural sources. This includes snow fencing, Christmas trees, planting and USACE/other engineering.
Management Area	Areas defined by USACE to manage under the FIMI.
Nourishment	Beach widening by the USACE in Smith Point County Park from 2014–2015.
Other	A broad category, where the area does not fall into the category of overwash, or restoration area. All following categories would further be defined under the 'other' category, depending on the specificity of the question.
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.
Swale	A depression or blow-out behind the primary dunes, between ridges, either vegetated or unvegetated.
Beach	The moist area that is between the high and low water line. Can be further defined as ocean, bay or inlet depending on the geologic location.
Dredge Spoil	Stockpiles and graded areas of dredged sand created by New York State.
Mud Flat	Areas where finer-grained sediments from the bay have been deposited in the low-energy shallows behind the island.
Sand Bar	Areas where tidal currents are strong enough to keep fine sediments in suspension. In this study area, sand bars occur north of the inlets.
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 is tidally exposed. They are often not static and may move seasonal, and annually.

Throughout the season, 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 around the time chicks hatched and re-opened around the time the chicks fledged (Table 2). However, in 2013 there was still vehicle presence at Smith Point County Park and in 2017 there was still vehicle presence at Robert Moses State Park-South in subsite OSS (Table 3). Official dates of closures for 2013 are uncertain. These closures did not apply to law enforcement, park staff and monitors, who continued to drive All-Terrain Vehicles (ATVs) or Utility-Terrain Vehicles (UTVs) in those areas.

Table 2. Beach closure dates for vehicle use at Smith Point County Park and Robert Moses State Park 2013–2017.

Management Unit	2013	2014	2015	2016	2017
Smith Point County Park	closed, but dates unknown	09 Jun–01 Aug	09 Jul–13 Aug	22 May–25 Aug	27 May–05 Aug
Robert Moses State Park - North	-	-	06 Jun–04 Aug	06 Jul–22 Jul	07 Jun–16 Jul
Robert Moses State Park - South	-	-	10 Jul–04 Aug	25 Jun–05 Aug	24 May–10 Aug

Each study site was divided into multiple smaller subsites intended to be of a similar geomorphological and ecological structure (e.g. areas where overwash occurred were distinct from highly vegetated tall dunes, Table 3, Figure 1 and Figure 2). These subsites allowed for general location referencing when GPS accuracy was not available. Monitoring surveys were performed by subsite.

Table 3. Subsite names, location descriptions, and habitat at Fire Island and Westhampton Island, 2017.

Name	Subsite Code	Description	Habitat Type
Demo 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 and Interior
Stockpile	STP	Sand stock pile area on bay side of island; begins at west end of golf course and extends west to the northernmost extent of the rock jetty.	Dredge Spoil, Beach, Interior
The Golf Course	GOLF	Begins at the 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 at 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 at 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 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 and Manipulated Dune
Old Inlet West Reference	OIWR	Stretch of beach between the east end of Old Inlet West and extends west to 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 duneline 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
Bridge East	BRE	Begins where the primary duneline begins at 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 duneline between Narrow Bay West and Pattersquash overwashes.	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 Cut 2 on Burma Road and extends west to the west 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 Cut 2 of Burma Road.	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 County Park. Begins at Moriches Inlet and extends east to the west edge of the breach-fill.	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 east edge of the breach-fill and extends east the west end of the parking lot.	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

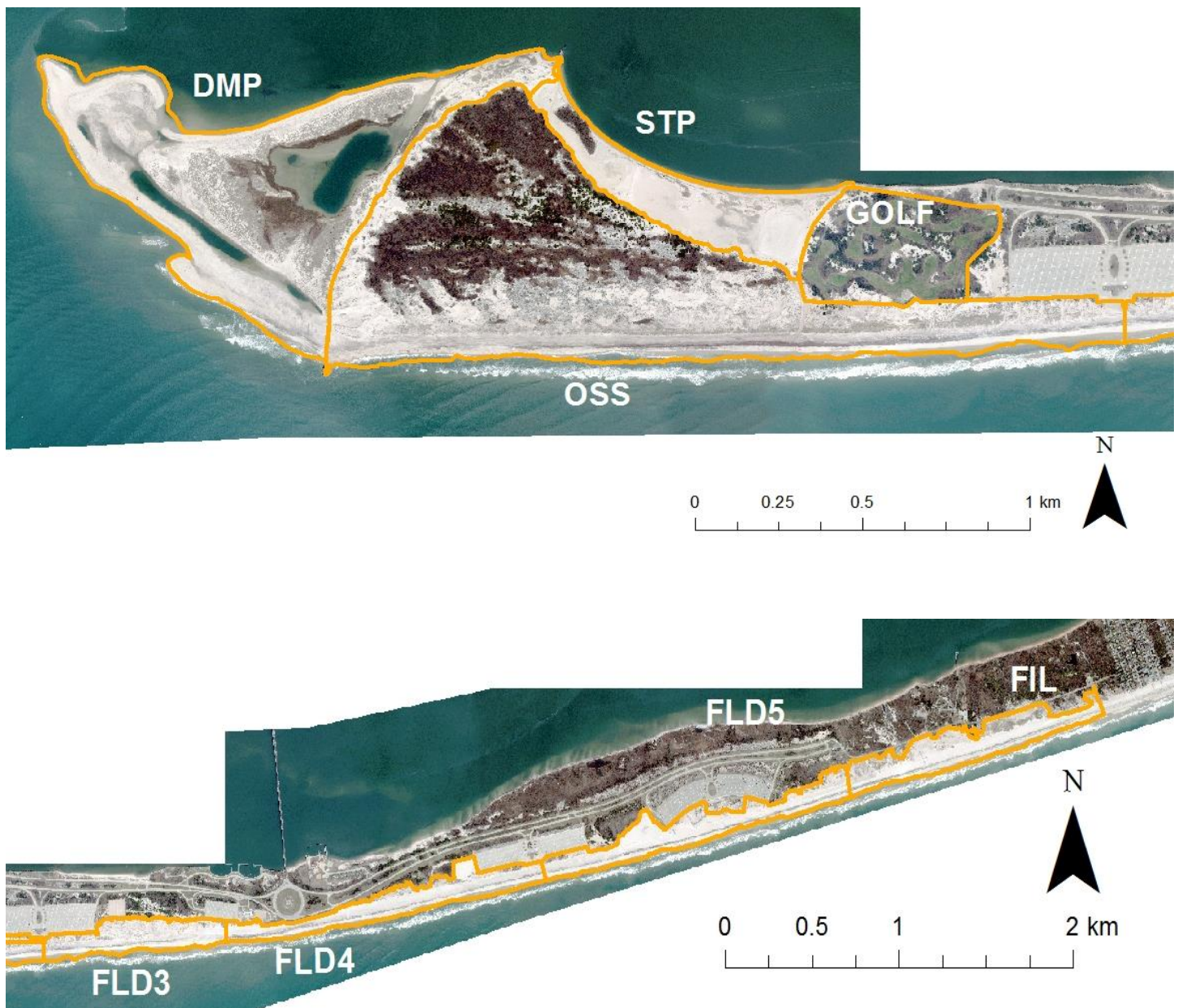


Figure 1. Study area at Robert Moses State Park including Fire Island Lighthouse Tract, shown from west to east. Subsite boundaries are highlighted in orange. See Table 3 for subsite names and descriptions. Imagery captured April, 2017.

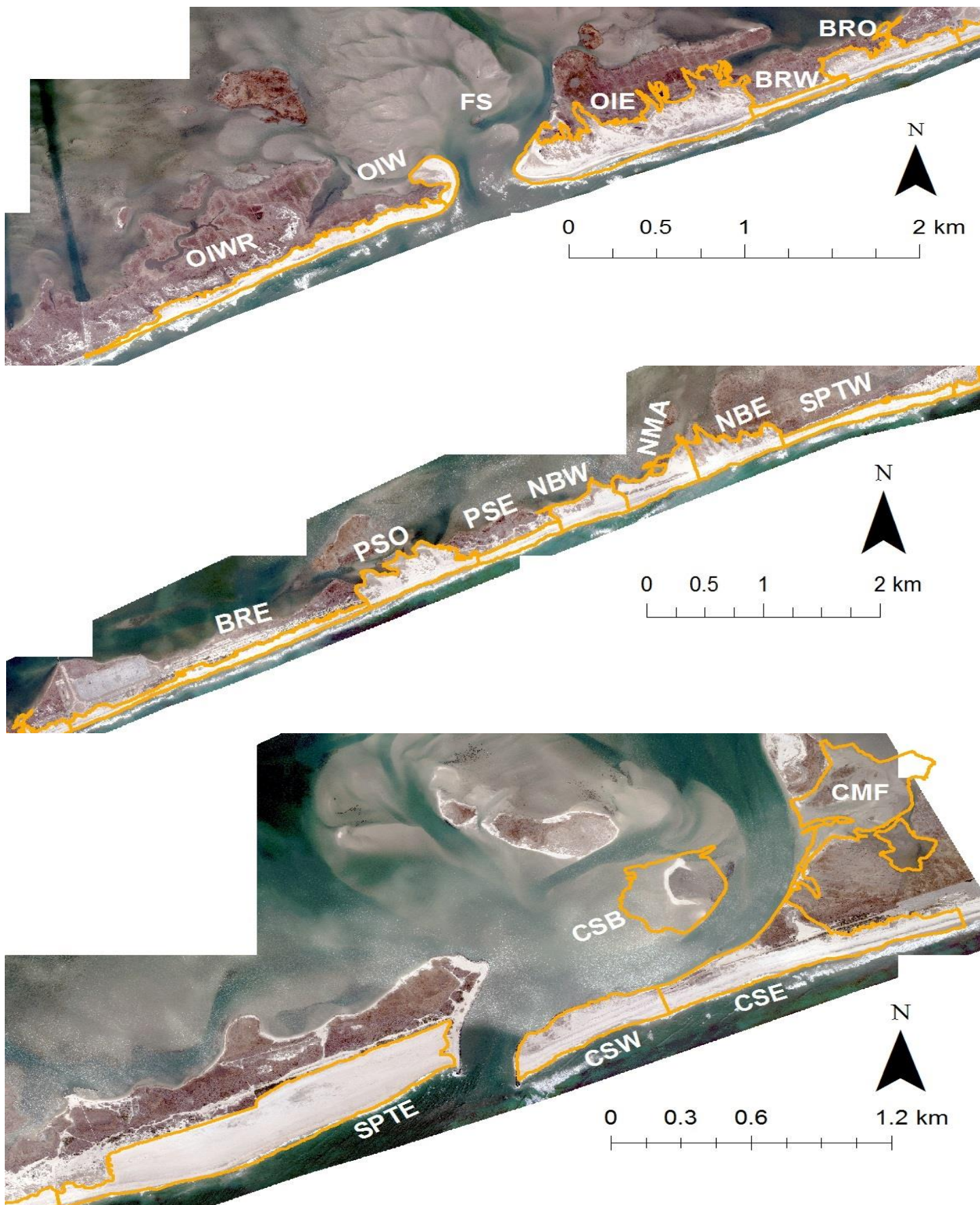


Figure 2. Study area at Eastern Barrier Islands, including Fire Island and Westhampton Island, shown from west to east. Subsite boundaries are highlighted in orange. See Table 3 for subsite names and descriptions. Imagery captured in April, 2017.

Part I: Monitoring of Piping Plover on Fire Island and Westhampton Island

Field Methods

Habitat and Tide Mapping

We used a Trimble Geo7x (Trimble Navigation, Ltd., Sunnyvale, California, USA) in 2015–2017 to map the spring high and low tides along the bay and ocean beach (Figure 3). In 2013 and 2014, we collected all mapping data with a Garmin 78 (Garmin International, Olathe, KS). In 2017, spring tides were mapped in May and the snow and symbolic fencing during late June through August. In 2016, spring tides were mapped during May till mid-September, while all other mapping (snow and symbolic fencing) were mapped during late-July and August. In 2015, the spring high tide was mapped during mid-May and early June, while all other mapping (wrack line, subsites) were mapped during late July and early August. In 2013 and 2014, mapping was carried out throughout the entire season; most was finished by August.



Figure 3. Using a Trimble Geo7X to map the spring tides.

Adult Survey

We surveyed each subsite in the study area for adult piping plovers every 1–3 days, April–August, 2017. We surveyed subsites on foot and by UTV. Some subsites that did not have bay side habitat and consisted only of narrow ocean beach were surveyed by driving a UTV slowly along the ocean beach and then back 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 surveyed by foot to search for plover tracks, scrapes, and nests. When conducting foot surveys, observers walked parallel 50-100 meters apart depending the size of the subsite, either east or west facing. 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 4). Upon sighting a plover, we recorded its band combination (if banded), behavior, and habitat. In 2016 and 2017, we used a Nikon laser rangefinder and a Trimble GPS unit to record the location of plovers. In 2013–2015, we recorded the location of individuals with a Garmin GPS unit.



Figure 4. Surveying for piping plovers using spotting scopes in the Otis Pike Wilderness area.

Nest Location

We searched all potential nesting areas in the study site, including backshore, overwashes and sparsely vegetated areas. Nests locations were recorded using a Trimble GPS (2015–2017, Figure 5) unit or a Garmin GPS unit (2013–2014). We floated all the eggs of nests found initially with 4 eggs, to estimate the initial lay and hatch date (Alberico 1995). Eggs were not floated in the western wilderness of Fire Island National Seashore (subsites OIW and OIWR) after the 2014 season.



Figure 5. Using a Trimble Geo7X to map the location of a piping plover nest.

Egg Masses

Once a nest reached four eggs, the eggs were weighed using a digital pocket scale (American Weigh, Norcross, GA, USA). We only weighed eggs in nests that were found before reaching a full clutch (typically 4 eggs). Nests in the western wilderness of Fire Island National Seashore (subsites OIW and OIWR) were not weighed after the 2014 season.

Nest Checks

We checked nests every 1–3 days throughout the incubation period. As the hatching date approached, nests were checked for signs of hatching approximately every day. The fate of nests was determined from observations from the nesting area.

Trapping and Banding

We trapped adults during the incubation period, over a full clutch, using a drop-box trap or shoo-in funnel trap. Once captured, the plover was removed from the trap, banded, weighed, and measured (Figure 6a). If the second adult had not yet been trapped in 2017, we reset the trap while banding the first adult. We attempted to capture unbanded and banded plovers to assess the condition of bands and flags and leg health. If no adults entered the trap within 20 minutes of its deployment, we removed the trap and did not return to the nest that day. We captured, banded and processed plover chicks on the initial nest visit post-hatching (Figure 6b) and recaptured them at a minimum of 20 days to weigh and measure them a second time.



Figure 6. a) Taking the tarsus measurement of an adult plover. b) Taking the weight of a plover chick.

Banding Schemes

In 2015–2017, we banded all adults and chicks with dark green flags with a three-digit alphanumeric ID etched in white on one upper leg, and an numbered orange Darvic band on the opposite upper leg (Figure 7). In previous years, some birds 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, DeRose-Wilson et al. 2014).

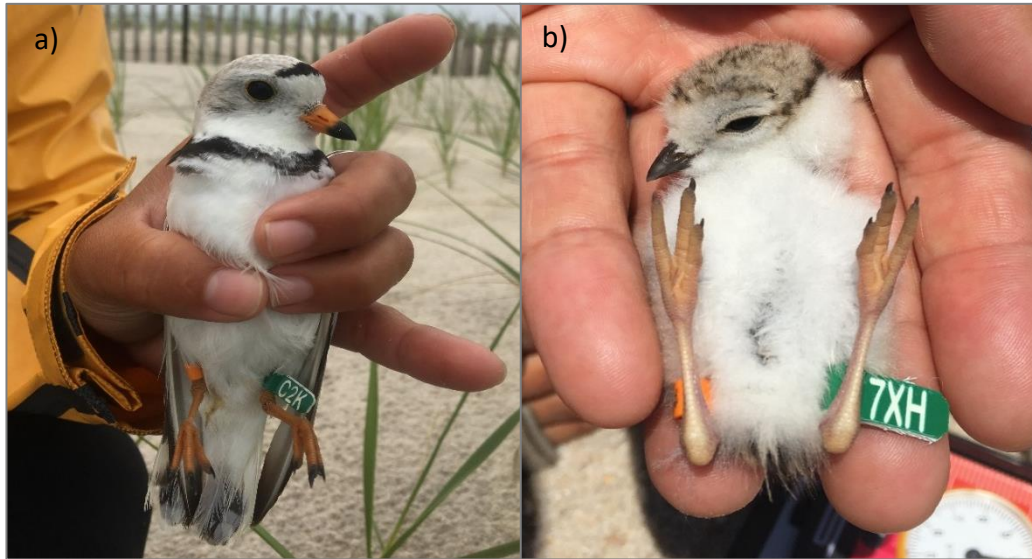


Figure 4. a) A piping plover adult and b) chick banded with an alphanumeric flag and color band.

Brood Observations

We searched for plover chicks for 30 days post-hatching, regardless of brood fate. If weather permitted, we visited broods at least every other day. We collected a behavioral sample on at least one randomly selected chick in each brood at each visit, using a random number generator to determine the start time of the behavioral sample after locating a brood. For five minutes, we recorded forage rate (peck, pull, probe) continuously, and behavioral state (forage, sit, run, walk, chase, flee, preen, encounter with another individual) and habitat (moist sand, moist vegetation, dry sand, dry vegetation, and wrack) every ten seconds (Altmann 1974). We recorded the number and type of disturbances (humans, vehicles, predators, competing piping plovers, etc.) 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 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 a chick and conducted another behavioral sample at the new location.

Invertebrate Sampling

We sampled terrestrial invertebrates from 2014–2017 in various habitat types (backshore, overwash, breach fill, restoration area, beach), from the time that the first chicks hatched until the time the last chicks fledged. We collected four samples during each invertebrate sampling set, two on the beach and two randomly selected locations. For the beach samples, we followed a transect parallel to the beach at the upper edge of the intertidal zone. Using a Garmin GPS unit, we navigated to the last beach data point collected during the previous sample in the subsite and continued a random distance, in meters, using a randomly generated number 1–50, for the first sample of the current sampling. We walked 50-m along the beach for the subsequent sample. The third and fourth samples for the set were selected from a predetermined random selection of points, selecting a random point within 100-m of the previous sample. We sampled invertebrates using two paint stirrers coated with Tanglefoot Tree Insect Barrier (Tanglefoot Company, Grand Rapids, Michigan). One stick had one side coated with Tanglefoot, and was placed horizontally in the sand with the Tanglefoot side facing up (Figure 8). The sticky area on the horizontal stick was 64.5 cm² and was 129 cm² on the vertical stick. The second stick was coated on both sides, and placed vertically in the sand. We placed a cage over the trapping station to prevent any birds from getting stuck in the Tanglefoot. We returned to each sample location after 30 minutes and counted, measured (± 1 mm), and identified arthropods to order.



Figure 5. Invertebrate trapping station.

Fecal Sampling

Piping plover fecal samples were collected opportunistically while conducting field surveys. Fresh fecal samples were collected, with as little as sand as possible, and placed in vials using tweezers (Figure 10). We collected the location of the sample in the Trimble Geo7x. After every collection, the tweezers were wiped off with ethanol wipes, before being used to collect another fecal sample. Vials were kept in pouches with coolers in the field until storing them in the field site freezer until the end of the season. Thirty-four samples were collected across the study site in 2017, 39 were collected in 2016, 7 were collected in 2013, and 37 in 2014. In 2013 and 2014, the samples were stored in alcohol at room temperature and transported to the MRI, where organisms were counted and identified. Samples collected for 2017 and 2016 were transported to contractors at Jonah Ventures in Boulder, Colorado for meta-barcoding DNA analysis. We did not collect fecal samples in 2015.



Figure 6. Collecting a fecal sample, using tweezers and marking the location with Trimble Geo7x.

Nest Cameras

In 2016 and 2017, we placed remote cameras (Reconyx Rapidfire, Holmen, WI, USA) at nests to detect the abundance of disturbances. All cameras were placed in a camouflaged box for theft protection. We programmed the cameras to have a 5-minute picture interval and take 3 pictures per trigger, with 5 seconds 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 and our second priority was to face the boxes north or south, to avoid positioning them directly in the sun's path to prevent whited-out photos. We placed the camera box against the outside of the enclosure, and aimed it towards the nest, attaching it with a bungee cord to secure it (Figure 11). We would perform any maintenance needed to cameras, such as replacing memory cards and batteries, while we conducted daily nest checks so as to not add camera caused disturbance. The cameras were removed when the brood permanently left the enclosure nest area.



Figure 7. A remote nest camera installed outside an enclosure.

Analytical Methods

Objective 1: Compare changes in numbers, density, and distribution of piping plovers in various habitats on Fire Island

We compared the number of pairs that used the various habitats 2013–2017. The various habitats compared were backshore, restoration area, natural overwash, and manipulated overwash. See Table 1 for habitat definitions. Using the GPS locations for each nest, we assigned each nest a discrete habitat type, and compared the proportion of nests in each habitat type. For the number of nests in each habitat type, we only included renests of stable pairs (same male and female) in the count if their new nest was in a different subsite and different habitat class. We assumed that if a male renested in the same subsite and habitat type, the location of the renest was not independent of the first nest.

We calculated annual pair counts using the males within pairs. We used males for pair counts and productivity estimates because when pairs separate, the males often keep their territory when they partner with a new female. Additionally, males are the primary territory holders and defenders. Using banded adult information

and nest date overlaps, we used the highest number of male territories per management unit 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 of unbanded pairs nests to try 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.

Objective 2: Estimate the “effect distance” of newly created habitat

To estimate the “effect distance” (the distance which birds traveled to use the habitat) of newly created habitats, we compiled all adult locations from 2014–2017, and divided them into two sets. The first set contained all of the points of banded individuals that nested within the study area, during the time that they had an active nest. The second set of points contained all of the points of banded individuals while they did not have a known active nest or brood, including both before and after nest initiation. Before nest initiation, we used their first nest location for distance calculations, and if a nest failed and a bird renested, we used the second nest location starting the estimated nest initiation date. Using the package “geosphere” (Hijmans 2016) in program R (R Core Team 2016) we calculated the distance of each location from their nest location. If a bird had multiple nests within a given year, we calculated the distance from each location to the nest that was active closest in time to the location. Similar to adult locations, we compiled all of the fledgling locations from 2014 to 2017 and calculated the distance from each location to the nest from which the fledgling hatched.

We joined all three of the point location datasets with a habitat GIS layer, including the flood shoals of Old Inlet and Moriches Inlet, overwashes and restoration areas. We summarized the distance that birds moved to each of the newly created habitats during the different biological periods.

In addition, to assess the importance of bayside foraging habitat to nesting piping plovers, we used a Kruskal-Wallis test to compare the distance adults moved from active nests with direct bay access and nests without bay access. Direct bay access included any subsite where a plover could walk to the bay intertidal area within the subsite of the nest location without encountering a solid line of vegetation within the habitat classification, or water. Subsites with direct bay access included Narrow Bay West, Pattersquash Overwash, Cupsogue West and the Stockpile.

Objective 3: Compare egg weights of birds with access to bay side intertidal flats with egg weights of birds without such access

We used simple linear regression to estimate the total clutch mass at the time of completion, as although we attempted to weigh nests the same day as nest completion, we were not always able to do so. We used all clutches found at less than a full clutch 2013–2017. To determine if clutch masses differed in piping plovers that nested where they had direct bay access compared to those that did not have bay access, we used the Wilcoxon Rank Sum Test. We considered a nest to have bay access if the subsite a nest was laid in had bayside intertidal habitat. Areas with nests that had bay access include Narrow Bay West, Pattersquash Overwash, Cupsogue West and the Stockpile. All other sites were considered to not have direct bay intertidal habitat access.

Objective 4: Compare piping plover nest and chick survival rates in various habitats across Fire Island

We used a logistic exposure model (Rotella et al. 2004, Shaffer 2004) to estimate the daily survival rates of exclosed and unexclosed nests. A nest was considered exclosed only during the period when the exclosure was present. In this model, hatch date was set at 34 days from initiation date (Heisey et al. 2007).

To estimate chick survival, we used the young survival from marked adults model, implemented in Program Mark (Lukacs et al. 2004, White and Burnham 1999). We modeled chick survival for each year from 2013 to 2017 ($n = 111$ broods). We estimated daily survival over a 31-day period. Our model set included year and habitat types. We ran all subsets of models including year and habitat terms for both apparent survival and detection probability. To estimate the probability of survival to fledge, we raised the daily survival estimate to 25 days, and used the delta method to obtain variance estimates (Powell 2014). We also considered a second model set, using the distinct management units, with year and management unit variation.

We estimated minimum convex polygons as brood rearing areas from 2014–2017 using the “adehabitatHR” package in Program R (Calenge 2006, R Version 3.3.1). We only calculated home range size for broods that had ≥ 5 unique points available, and compared the data in two sets. The first was using all brood home ranges, and the second was using just broods that fledged at least one chick, to control for broods that had smaller sample size due to early chick mortality. A Wilcoxon Rank Sum test was used to compare brood home ranges between broods that had direct access to bay foraging habitat within the subsite their nest was in, and those that did not have such access. We also used simple linear regression to test for the difference between broods with bay access or not, controlling for the number of points used in the estimate the MCP.

We also calculated the movement of broods between successive daily observations. To do this, we randomly selected one point per day, per brood, and then used the “geosphere” package in Program R to calculate the Euclidean distance from every point to the one preceding it (Hijmans 2016).

Objective 5: Compare the relative contribution of local growth (local recruitment) vs immigration in population growth associated with new habitats

Immigration and recruitment

Using banded and known age birds, we estimated immigration and local recruitment for 2014–2017. We estimated immigration rate using a Pradel (1996) model. The parameters estimated from the Pradel model are apparent survival, resight rate, and fecundity rate. We used the fecundity rate as an estimate of the immigration rate (# immigrants in $t+1$ per individual in year t), and ran a single model holding apparent survival and resight rate constant, to estimate annual immigration rates (Catlin et al. 2015). Starting in 2016, estimates of the immigration rate also include the Democrat Point site. We estimated annual local recruitment, as the number of individuals banded as chicks in years prior that returned to breed for the first time in the population in a given year. In addition to immigration rates, we assessed how many of the newly banded individuals (immigrants), local recruits and returning nesters settled into new habitats created by Hurricane Sandy or mitigation areas.

To determine if immigrants, natal recruits, and returning nesters are selecting for or against the habitat classes, we compared percent area and percent used of the habitat types. We used a classification layer of dry sand within each of the habitat areas, as defined by delineated ocean-to-bay overwash and USACE provided shapefiles, as available, and calculated the proportion of habitat use as the proportion of natal recruits or immigrants in each habitat type. We conducted a chi-squared test to determine if there was overall selection

occurring, and compared 95% confidence intervals on proportion of use to availability to determine selection for or against particular habitat types (Byers et al. 1984, Catlin et al. 2011). We currently only have the selection analysis completed for the eastern study area for 2015, as in 2013 we could not estimate immigrants or natal recruits without prior banded birds, and for 2014 there no imagery available. We intend to complete 2016 and 2017 soon.

Survival and site fidelity

To estimate true survival of piping plovers in the Fire Island study area, we used the Barker live-dead encounter model (Barker 1997). The Barker model uses supplemental (off-site) resightings to distinguish permanent emigration from mortality, for a more precise estimate of survival. Our primary interest for this report was in true survival (S) of hatch year (hatch to first year returning; HY) and after-hatch-year (AHY) piping plovers. Other parameters that we were able to estimate using this model are detection probability (p), the probability of being reported dead during the supplemental period (r), the probability of being resighted during and surviving the supplemental period (R), the probability of being resighted during the supplemental period and dying (R'), site fidelity (F), and the probability of returning to the study area, if you have emigrated (F'). We fixed r to 0, as we did not have any reports of dead individuals during the supplemental period. We modeled variation in all of the parameters, other than r , with year and age variation. We tested for overdispersion ($\hat{c}$) using a model with age and year variation in all models and corrected for overdispersion if confidence intervals $\hat{c}$ did not overlap 1.

Objective 6: Compare prey densities at several habitat types

To assess the difference in the mean number of terrestrial invertebrates collected using sticky sticks on bay beaches, ocean beaches, and barrier interior we performed an ANOVA using the invertebrate transects from 2017. We also used a negative binomial mixed model to predict the mean invertebrate catch over a three-hour period in each of four habitat types (bay intertidal, ephemeral pool, ocean intertidal, island interior) and each of three treatment levels (overwash, breach fill, restoration area) with the combined invertebrate data from 2014–2017. For the grouping variables within the analysis, habitat type was assigned for each individual invertebrate sample, at the time of data collection. Treatment level was assigned based on the joining of a point file of all invertebrate sample locations with shapefiles of the overwashes, breach fills and restoration areas. Results were extrapolated to three-hour periods to compare to past studies.

Objective 7: Quantify the interactions of predators with shorebird nests and chicks

We installed remote cameras at exclosed nests, to quantify the interactions predators have with plover nests and chicks. We analyzed the camera data using a 30-minute detection period as a cutoff from the start of one photo series to determine when to record the following photos as part of the current event versus a new event. This allows us to distinguish between different photo capture “events” for non-uniquely identifiable animals. Individuals detected on camera but not included in the results were piping plovers, monitors, and monitor vehicles. Also see details on use of high frequency GPS collar locations of red foxes during exclosure setup and egg hatching periods in the Red Fox report section below (Part IV: Methods, GPS tracking).

Results

Objective 1: Compare changes in numbers, density, and distribution of piping plovers in various habitats on Fire Island

From April–August 2017, we banded 36 adult piping plovers and 129 chicks (Table 4). Six chicks of the 135 that hatched were not banded: 2 chicks from separate broods from East Wilderness Area, 2 chicks from separate broods from Smith Point County Park, 1 chick from Cupsogue Beach County Park, and 1 chick from Democrat Point at Robert Moses State Park. None of these unbanded chicks were believed to survive to fledging. The remaining chicks in 2017 were banded between 0 and 6 days old. We recaptured 39 adult piping plovers that were banded during previous years. Of those recaptured, 18 of them were banded as chicks on Fire Island, and 21 were banded as adults on Fire Island.

From 2013–2017, we banded 151 adult piping plovers (82 females and 69 males) and 352 chicks (Table 5). All chicks were banded between 0 and 13 days old. In 2013 and 2014, 8 nests were found at Watch Hill. Five adults and three chicks from those nests were banded. We did not monitor nests at Watch Hill 2015 through 2017. We did not band any birds at Robert Moses State Park or the Lighthouse Tract in 2013 and 2014 as we did not begin any work at those sites until 2015. Of the 352 total chicks banded, 37% of them were banded in the 2017 season, 45.3% in 2016, 19.2% in 2015, 18.3% in 2014 and 15.2% in 2013.

Table 4. Piping plover adults and chicks banded on Fire Island and Westhampton Island during April to August 2017 per management unit. Subsites within each management unit are abbreviated in parentheses. See Table 3 for a detailed description of these subsites.

Management Unit	Adults		Chicks
	Females	Males	
Cupsogue Beach County Park (CSE, CSW, CMF, CSB)	4	2	15
Smith Point County Park (BRE, PSO, PSE, NBW, NMA, NBE, SPTW, SPTE)	8	6	61
Fire Island National Seashore - east of breach (OIE, BRW, BRO)	5	3	35
Fire Island National Seashore - west of breach (OIW, OIWR)	0	0	0
Fire Island National Seashore - Lighthouse Tract (FIL)	1	0	0
Robert Moses State Park (FLD5, FLD4, FLD3, OSS, STP, DMP, GOLF)	4	3	18
Total	22	14	129

Table 5. Piping plover adults and chicks banded on Fire Island and Westhampton Island during 2013–2017 per management unit. Subsites within each management unit are abbreviated in parentheses. See Table 3 for a detailed description of these subsites.

Management Unit	Adults		Chicks
	Females	Males	
Cupsogue Beach County Park (CSE, CSW, CMF, CSB)	15	11	80
Smith Point County Park (BRE, PSO, PSE, NBW, NMA, NBE, SPTW, SPTE)	27	24	125
Fire Island National Seashore - east of breach (OIE, BRW, BRO)	13	12	73
Fire Island National Seashore - west of breach (OIW, OIWR)	1	3	14
Fire Island National Seashore - Lighthouse Tract (FIL)	1	1	0
Robert Moses State Park (FLD5, FLD4, FLD3, OSS, STP, DMP, GOLF) ^a	23	15	57
Off Site – Bellport to Watch Hill ^b	2	3	3
Total	82	69	352

^a Birds were only banded here from 2015–2017

^b Birds were only banded here from 2013–2014

We found 62 nests in 2017, 40 of which survived to hatching and 22 failed (Table 6). Of the 22 that failed, nine were abandoned, four were depredated and two flooded (Table 6). Of the four nests depredated in 2017, three were suspected as avian predation and one was suspected as red fox predation. From 2013–2017, we located and monitored 202 nests. Of these nests, 115 (56.9%) of them hatched, and 87 (43.1%) of them failed (Table 7). The most common reason for nest failure over our whole study period was depredation (Table 8).

Table 6. Piping plover nest distributions and fates at Fire Island and Westhampton Island, in various habitats of each management unit, April–August 2017. Nest numbers include renests of some pairs.

Management Unit	Habitat	Nests			
		Found	Hatched	Failed	Reason for loss
Fire Island National Seashore - Lighthouse Tract	Barrier Interior	1	0	1	Abandoned
Robert Moses State Park	Backshore	11	6	7	Abandoned (2), Predated (1), Flooded (1), Unknown (3)
Robert Moses State Park	Barrier Interior	1	1	0	
Robert Moses State Park	Dredge Spoil	1	1	0	
Cupsogue Beach County Park	Manipulated Overwash	9	4	5	Abandoned (3), Predated (1), Unknown (1)
Cupsogue Beach County Park	Barrier Interior	2	1	1	Unknown
Fire Island National Seashore - east of breach	Natural Overwash	13	10	3	Abandoned (1), Flooded (1), Unknown (1)
Smith Point County Park	Backshore	3	2	1	Abandoned
Smith Point County Park	New Made Restoration Area	1	1	0	
Smith Point County Park	Great Gun Restoration Area	5	5	0	
Smith Point County Park	Manipulated Overwash	15	11	4	Abandoned (1), Predated (2), Unknown (1)
Total		62	40	22	

Table 7. Piping plover nest distributions and status at Fire Island and Westhampton Island, in various habitats of each management unit, 2013–2017. Nest numbers include renests of some pairs.

Management Unit	Habitat	Found	Hatched	Failed	Hatched %	Failed %
Fire Island National Seashore - Lighthouse Tract ^a	Barrier Interior	2	0	2	0	100
Robert Moses State Park ^a	Backshore	23	13	10	56.5	43.5
Robert Moses State Park ^a	Barrier Interior	7	3	4	42.9	57.1
Robert Moses State Park ^a	Dredge Spoil	5	3	2	60	40
Cupsogue Beach County Park	Manipulated Overwash	27	18	9	66.7	33.3
Cupsogue Beach County Park ^b	Barrier Interior	10	7	3	70	30
Fire Island National Seashore - east of breach	Natural Overwash	34	23	11	67.6	32.4
Fire Island National Seashore - west of breach	Natural Overwash	5	4	1	80	20
Smith Point County Park	Backshore	13	6	7	46.2	53.8
Smith Point County Park ^a	New Made Restoration Area	8	2	6	25	75
Smith Point County Park ^a	Great Gun Restoration Area	10	7	3	70	30
Smith Point County Park	Manipulated Overwash	50	25	25	50	50
Off Site – Bellport to Watch Hill ^c	Backshore	8	4	4	50	50
Total		202	115	87	56.9	43.1

^a Not monitored 2013–2014

^b Not monitored in 2013

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

Table 8. Fate of nests on Fire Island and Westhampton Island, by year, in each management unit, 2013–2017.

Year	Management	Total Nests	Successful	Failed			
				Abandoned	Depredated	Overwash/Flood	Unknown
2013	Fire Island National Seashore	8	5 (63%)	1 (13%)	-	1 (13%)	1 (13%)
	Smith Point County Park	9	3 (33%)	-	4 (44%)	1 (11%)	1 (11%)
	Cupsogue Beach County Park	4	3 (75%)	-	-	-	1 (25%)
2014	Fire Island National Seashore	10	4 (40%)	4 (40%)	2 (20%)	-	-
	Smith Point County Park	9	3 (33%)	-	5 (56%)	-	1 (11%)
	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%)	16 (70%)	-	4 (17%)
	Cupsogue Beach County Park	11	7 (64%)	-	3 (27%)	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	15	11 (73%)	1 (7%)	1 (7%)	-	2 (13%)
	Cupsogue Beach County Park	6	6 (100%)	-	-	-	-
2017	Robert Moses State Park	13	6 (46%)	2 (15%)	1 (8%)	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%)	3 (27%)	1 (9%)	-	2 (18%)
Total		197	110 (56%)	26 (13%)	39 (20%)	5 (3%)	17 (9%)

The number of pairs we monitored increased from 19 in 2013 to 45 in 2017, however the study site boundaries have changed throughout the years of monitoring (Table 9). The majority of piping plovers nested in backshore and overwash habitat. Restoration areas were not created until after the 2014 nesting season, but they were used by piping plovers thereafter (Table 9). Using information from USFWS and NPS, we estimated pairs for the entire study area (Robert Moses State Park to Cupsogue Beach County Park) for all four years of the monitoring effort. Though there was fluctuation within sites, the overall number of pairs have increased since Hurricane Sandy in 2012 (Table 10).

Table 9. Number of nesting piping plover pairs in each of the various habitats. Restoration areas were not created until after the 2014 nesting season. Numbers increased in the “backshore” habitat in 2015 due to the addition of the Robert Moses State Park study area. Some pairs had more than one nest per year. Footnotes to table indicate changes in monitoring area among years.

Habitat	Year				
	2013 ^a	2014 ^a	2015 ^b	2016 ^b	2017 ^b
Overwash	11	17	15	14	23
Breach Fill	1	1	3	3	3
Restoration	N/A	N/A	5	4	6
Backshore/Barrier Interior	7	3	12	16	13
Total	19	21	35	37	45

^a Includes Watch Hill nests, only monitored 2013–2014

^b Includes Robert Moses and the Lighthouse Tract, only monitored 2015–2017

Table 10. Number of pairs nesting at each management unit. Pair counts for areas not monitored by Virginia Tech in a given year (e.g. Robert Moses State Park in 2013–2014) were obtained from USFWS and NPS correspondence and annual reports, and are italicized. Some pairs had more than one nest per year.

Management Unit	Year				
	2013	2014	2015	2016	2017
Robert Moses State Park	<i>12</i>	<i>10</i>	10	9	8
Fire Island Lighthouse	<i>1</i>	<i>1</i>	0	1	1
Fire Island National Seashore West	6	4	1	1	0
Fire Island National Seashore East	4	4	2	6	10
Smith Point County Park	6	8	13	13	20
Cupsogue County Park	3	5	7	6	6
Total	32	32	33	36	45

Objective 2: Estimate the “effect distance” of newly created habitat

Adult Movement

The greatest distance that an adult with an active nest was observed from its nest within a breeding season was 7,412 m; in 2014, an adult with a nest at Narrow Bay East was observed at Old Inlet East. The greatest distance an adult was observed from an inactive nest was 33km; an adult that nested in Field 3 of Robert Moses State Park was observed foraging at the Old Inlet Flood Shoals after brood loss.

In 2014, the average distance that birds with active nests were observed from their nest was 236 m (SE = 39), and the average distance that birds without active nests (i.e., pre- or post-nesting) were observed from their nests was 536 m (SE = 110). In 2015, the average distance that birds with active nests were observed from their nest was 350 m (SE = 54), and the average distance that birds without active nests (i.e., pre- or post-nesting) were observed from their nests was 873 m (SE = 106). In 2016, the average distance that birds with active nests were observed from their nests was 150 m (SE = 15.1), and the average distance that birds without active nests (i.e., pre- or post-nesting) were observed from their nest was 537 m (SE = 110.4). In 2017, the average distance that birds with active nests were observed from their nest was 133 m (SE = 10.4), and the average distance that birds without active nests (i.e., pre- or post-nesting) were observed from their nest location was 716.4 m (SE = 87).

During the time that birds had active nests, we only recorded one location at the Old Inlet Flood Shoal, and it was the location of female nesting 150 m from the resight location. We recorded eight locations of non-nesting adults at the Old Inlet Flood Shoal. The greatest of these non-nesting movements was 33 km from Robert Moses State Park, though all observed movements to this area were greater than 800m (Table 12).

We observed 17 individuals with active nests and 78 individuals without active nests using the Cupsogue sandbar over the three years of the study. The greatest distance that an individual with a nest moved to the Cupsogue sandbar was 2,141 m from the Great Gun restoration area. All of the observations of nesting individuals using the Cupsogue sandbar were nesting either at Cupsogue Beach County Park or Great Gun restoration area/Smith Point East. Of the 71 instances of individuals using the sand bar that were not actively nesting, the greatest observed distance was 8.3 km and was an individual who had nested at Pattersquash Overwash earlier in the season. This bird from Pattersquash was observed foraging at the Cupsogue Sand Bar both before and after nest initiation.

At the Cupsogue mud flat, we observed 8 individuals with active nests foraging. Similar to the sandbar, all of these individuals were nesting either at Cupsogue Beach County Park or the Great Gun restoration area. The greatest distance for these observations was 2.5km from an individual nesting at Great Gun restoration area. We recorded 10 pre- or post- nesting individuals foraging at the Cupsogue Mud Flat, with the greatest distance from their nest of 6km, and was an individual that had previously nested at Narrow Bay East.

During both the time that nests were active and the time that nests were inactive, individuals used all of the overwashes within the study area (Table 11, Table 12). We observed 38 individuals in locations other than the subsite they were nesting in, 17 of which were foraging. Most (N = 14) of these locations were individuals foraging at Narrow Bay West at the bayside mudflat, and the birds moved from Narrow Bay East and New Made. Individuals also moved from Narrow Bay East and New Made to Old Inlet East, and from Old Inlet East to Bridge Overwash. While many of the observations of piping plovers at the overwashes were observations of birds that had a nest within the overwash, birds with active nests elsewhere did use the overwashes.

Specifically, the greatest distance moved while birds were actively nesting was to the Old Inlet Overwash, adjacent to the Old Inlet Breach—a new habitat created by Hurricane Sandy.

Table 11. Distance from nest for piping plovers with active nests observed in Hurricane Sandy created habitats, breach fills, Cupsogue sandbar and mudflat, and restoration areas, Fire Island and Westhampton Island, 2014–2017.

Habitat	Distance from Nest (m)				95% Confidence Interval	
	N	Max	Mean	SE	Lower	Upper
Bridge Overwash	12	1326.7	254.2	142.2	-58.8	567.2
Cupsogue Mud Flat	8	2537.2	1377.4	269.6	739.8	2015.0
Cupsogue Sand Bar	17	2141.2	907.2	104.4	685.8	1128.6
Cupsogue Overwash	196	1419.8	119.0	11.6	96.0	141.9
Cupsogue Breach	14	440.1	118.6	37.0	38.8	198.4
Old Inlet Flood Shoal	1	150.4	150.4	-	-	-
Great Gun restoration area	35	3466.1	179.2	98.1	-20.1	378.5
Narrow Bay Breach	22	276.6	57.2	12.1	31.9	82.4
NBE Overwash	86	3466.1	186.3	47.0	92.9	279.8
NBW Overwash	128	2006.2	141.3	16.9	107.9	174.7
New Made	7	2453.5	418.7	339.8	-412.8	1250.3
OIE Overwash	262	7412.7	206.6	46.2	115.6	297.7
OIW	2	938.2	933.3	4.9	870.9	995.7
OIWR Overwash	4	87.9	61.0	11.4	24.7	97.2
PSO Overwash	133	1751.9	138.3	15.5	107.6	169.1

Table 12. Distance from nest to Hurricane Sandy create habitats, restoration areas, and the Cupsogue sandbar and mudflat, for adult plovers before or after (but not during) nesting, on Fire Island and Westhampton Island, 2014–2017.

Habitat	Distance from Nest (m)				95% Confidence Interval	
	N	Max	Mean	SE	Lower	Upper
Bridge Overwash	2	1048.8	1044.8	4.1	993.2	1096.4
Cupsogue Mud Flat	10	6070.8	2102.7	481.1	1014.3	3191.1
Cupsogue Sand Bar	78	8326.3	2047.4	278.6	1492.6	2602.1
Cupsogue Overwash	258	7762.2	577.7	79.9	420.4	735.1
Cupsogue Breach	22	535.9	146.3	35.1	73.3	219.4
Old Inlet Flood Shoal	8	33079.4	10960.9	3593.9	2462.6	19459.2
Great Gun restoration area	18	2374.8	488.4	146.6	179.2	797.7
Narrow Bay Breach	5	69.7	55.7	7.8	34.0	77.5
NBE Overwash	127	12426.0	1074.9	241.1	597.7	1552.1
NBW Overwash	89	3324.8	255.0	56.2	143.2	366.7
New Made	11	2505.2	331.8	218.1	-154.1	817.8
OIE Overwash	237	12877.6	479.8	112.5	258.3	701.4
OIW	5	1447.3	1307.3	52.5	1161.5	1453.2
OIWR Overwash	19	8097.0	2309.3	381.8	1507.2	3111.3
PSO Overwash	218	5797.9	344.0	44.6	256.0	432.0

In 2014–2017, adult plovers with an active nest and direct bay access were seen closer to their nests than plovers ($\bar{x} = 117.8$, SE = 14.54) that did not have direct bay access ($\bar{x} = 196.8$, SE = 28.6; df = 1, $P = 0.002$, Figure 11).

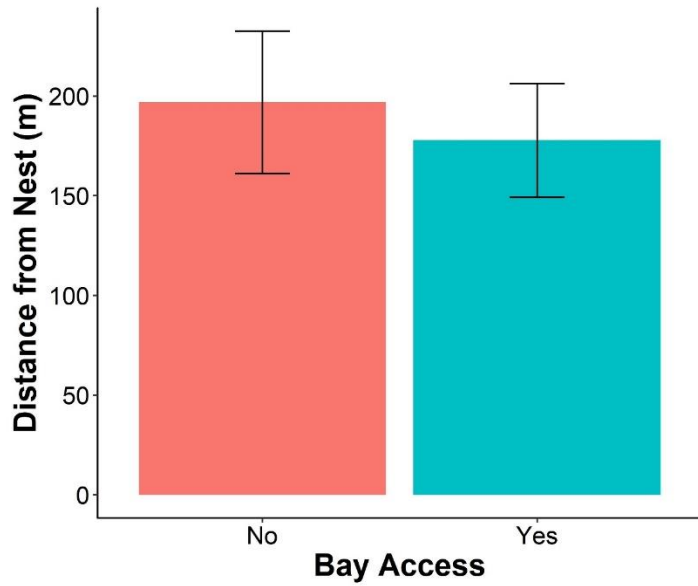


Figure 11. Average distance adult plovers were observed from their nest 2014–2017.

Fledgling Movement

We assessed the use of newly created habitats and other habitats of interest by fledgling piping plovers relative to the nest they hatched from. From 2014–2017, we recorded 664 observations of fledglings from the study area. On average, those individuals were observed 2,515m (SE = 219; Table 13) from where they hatched that season. The maximum distance observed was 49.08km; a fledged chick from Great Gun restoration area was observed foraging at the tip of Robert Moses State Park in August of 2017. On average, fledglings moved farthest to the Old Inlet Flood Shoals, Cupsogue mud flat, and Cupsogue sandbar. After creation of the restoration areas, we only observed fledglings that had hatched there. From 2014–2016, we observed five fledglings return to nest as first-year birds within a subsite where they had been observed as a fledgling in the year prior.

Table 13. Distance moved by fledgling piping plovers, from their nest location.

Habitat	Distance from Nest (m)				95% Confidence Interval	
	N	Max	Mean	SE	Lower	Upper
Bridge Overwash	17	10815.6	1505.3	606.8	316.0	2694.6
Cupsogue Breach	2	481.0	434.5	46.5	343.3	525.7
Cupsogue Mud Flat	11	13703.0	7209.3	1569.0	4133.9	10284.6
Cupsogue Overwash	72	12410.7	1344.9	349.0	661.0	2028.9
Cupsogue Sand Bar	43	12927.2	3103.6	548.8	2028.0	4179.3
Great Gun restoration area	2	572.5	490.7	81.8	330.3	651.0
Narrow Bay Breach	3	44.0	40.6	2.6	35.5	45.8
NBE Overwash	9	1108.2	379.7	93.4	196.6	562.8
NBW Overwash	42	7280.1	2073.5	379.8	1329.1	2817.9
New Made	4	387.6	387.6	-	-	-
OIE Overwash	139	12644.6	1769.8	264.7	1251.0	2288.6
OIW	9	2578.9	2032.0	192.5	1654.8	2409.3
OIWR Overwash	1	13104.0	13104.0	-	-	-
Old Inlet Flood Shoal	35	35841.9	4632.6	1072.8	2529.9	6735.3
PSO Overwash	101	7421.8	2120.5	243.9	1642.4	2598.6

Objective 3: Compare egg weights of birds with access to bay side intertidal flats with egg weights of birds without such access

In 2017, we weighed 32 nests at full clutch. There was no difference in the clutch masses of piping plovers that had bay access as opposed to those that did not have bay access ($Z=0.81$, $df=1$, $P=0.42$, $\bar{x} = 39.09$). Similarly, from 2013–2017, we weighed 115 nests at full clutch. There was no difference in the clutch masses of piping plovers that had bay access versus those that did not have bay access ($Z=0.10$, $df=1$, $P=0.92$, $\bar{x} = 39.23$).

Objective 4: Compare piping plover nest and chick survival rates in various habitats across Fire Island

Nest Survival

In 2017, nests that were exclosed during laying or after clutch completion had a 66% predicted probability of surviving to hatching. Nests that were never exclosed had a 42% predicted probability of surviving to hatching

(Table 14). The daily survival rate for exclosed and unexclosed nests was 0.99 and 0.97, respectively. Nests that were exclosed had a lower abandonment rate (9%) than nests that did not have exclosures (24%, Table 15).

When comparing across years, the probability of surviving to hatch varied (Table 14). In 2015, midway through the nesting season, exclosures were removed from all of the Smith Point County Parks nests (approximately 15 days after clutch completion) due to a high incidence of fox harassment at exclosed nests. The nests in 2015 that were “previously exclosed” had a 0% probability of surviving to hatching (Monk et al. 2015). During the years of the concurrent fox study, the probability of a nest surviving to hatching increased from 2015 to 2016, but decreased from 2016 to 2017 (Table 16). However, small samples sizes resulted in overlapping confidence intervals among all sites and years.

In 2013–2017, 27 of 197 nests were abandoned (Table 15). From the 27 abandoned nests, 5 nests belonged to a pair of unbanded or unknown adults, 9 nests belonged to a pair with only one banded adult and 12 nests belonged to a pair of banded adults. Of the 12 nests with a pair of banded adults, one individual from each nest (50%) was seen after the nest was abandoned. Of the 9 nests with one banded adult, eight banded individuals (89%) were seen after the nest was abandoned. Of 33 banded birds from the 27 abandoned nests, 20 banded individuals (61%) were seen after the nest was abandoned. The resight rate of birds following nest abandonment is lower than the annual survival rate, but not low enough to conclude that all nests were abandoned due to adult mortality.

Table 14. The probability of piping plover nests surviving to hatching with an exclosure and without an exclosure on Fire Island and Westhampton Island, 2013–2017.

Year	Caged	Number of Nests	Failed	Hatched	Apparent Survival	Probability of surviving to hatching	Standard error	Daily survival	Standard error
2013	N	6	5	1	0.17	0.23	0.12	0.96	0.02
2013	Y	15	4	11	0.73	0.49	0.12	0.98	0.01
2014	N	9	6	3	0.33	0.31	0.10	0.97	0.01
2014	Y	16	7	9	0.56	0.57	0.09	0.99	0.00
2015	N	25	20	5	0.20	0.25	0.07	0.96	0.01
2015	Y	23 ^a	12	11	0.48	0.51	0.07	0.98	0.00
2016	N	8	6	5	0.63	0.68	0.10	0.99	0.00
2016	Y	32	4	28	0.88	0.83	0.06	1.00	0.00
2017	N	16	14	2	0.13	0.42	0.09	0.97	0.01
2017	Y	45	7	38	0.84	0.66	0.06	0.99	0.00
2015	Removed ^b	5	5	0	0.00	0.00	0.00	0.47	0.25

^a Includes the 5 nests that had exclosures removed due to concerns of fox depredation

^b Probability of survival once exclosures had been removed from a previously exclosed nest

Table 15. Abandonment rate of piping plover nests with exclosures and without exclosures from Fire Island and Westhampton Island 2013–2017.

Year	Caged	# of Nests	Failed	Hatched	Abandoned	Abandonment ^a Rate
2013	N	6	5	1	0	0.000
2013	Y	15	4	11	1	0.067
2014	N	9	6	3	0	0.000
2014	Y	16	7	9	5	0.313
2015	N	25	20	5	2	0.080
2015	Y	23	12	11	5	0.217
2016	N	8	6	5	2	0.250
2016	Y	32	4	28	4	0.125
2017	N	17	15	2	4	0.235
2017	Y	45	7	38	4	0.089
2015	Removed	5	5	0	0	0.000
All	N	65	52	16	8	0.123
All	Y	131	34	37	19	0.145

^a Calculated as the # of abandoned nests / total nests

Table 16. Predicted probability of surviving to hatch (33 days from initiation) for all four management units, during the years of the concurrent fox study. 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	N Nests	Survival Estimate	SE
2013	Fire Island National Seashore	8	0.68	0.18
	Smith Point County Park	9	0.23	0.14
	Cupsogue County Park	4	0.64	0.28
2014	Fire Island National Seashore	11	0.35	0.14
	Smith Point County Park	9	0.48	0.16
	Cupsogue County Park	5	0.76	0.21
2015	Robert Moses State Park	10	0.62	0.15
	Fire Island National Seashore	4	0.52	0.24
	Smith Point County Park	23	0.11	0.05
	Cupsogue County Park	11	0.57	0.16
2016	Robert Moses State Park	11	0.68	0.15
	Fire Island National Seashore	9	0.75	0.15
	Smith Point County Park	14	0.83	0.11
	Cupsogue County Park	6	1.00	0.001
2017	Robert Moses State Park	12	0.42	0.15
	Fire Island National Seashore	14	0.70	0.13
	Smith Point County Park	24	0.77	0.09
	Cupsogue County Park	12	0.38	0.15

Chick Survival

Daily survival rates for chicks, and probability to fledging, varied among years and among management units (Table 17). For 2017, relative to 2016, survival to fledge increased at Robert Moses State Park and Cupsogue County Park, and decreased at Smith Point County Park and Fire Island National Seashore. Causes of chick survival remain challenging to identify, however in 2015 and 2016, we observed chicks depredated by gulls (Herring and Great Black-Backed) at Robert Moses State Park. In 2017, one chick starved at Fire Island National Seashore, in the Wilderness Area east of Old Inlet, based on necropsy by the USGS National Wildlife Health Center.

Table 17. Daily survival rate and probability of fledgling for chicks in all management units and years, estimated from Lukacs young survival model in Program MARK. Probability of surviving to fledging represents the probability of surviving to 25 days old.

Management Unit	Year	N Broods	DSR	S _{fledge}	SE(S _{fledge})
Robert Moses	2015	5	0.95	0.33	0.11
	2016	8	0.96	0.37	0.09
	2017	6	0.99	0.78	0.10
Fire Island National Seashore	2013	5	0.92	0.15	0.08
	2014	3	0.96	0.38	0.13
	2015	2	0.99	0.83	0.15
	2016	7	0.99	0.73	0.09
	2017	10	0.97	0.53	0.08
Smith Point	2013	3	0.96	0.39	0.18
	2014	4	0.99	0.77	0.11
	2015	2	0.98	0.55	0.19
	2016	12	0.99	0.84	0.06
	2017	19	0.97	0.49	0.06
Cupsogue	2013	3	0.88	0.05	0.05
	2014	4	0.96	0.36	0.12
	2015	7	0.90	0.07	0.04
	2016	6	0.94	0.22	0.08
	2017	5	0.96	0.34	0.12

For daily chick survival among habitat types (bay access, overwash, breach fill or restoration area), adding in the 2017 chick survival data, we did not have any significant relationships between chick survival and habitat. Year was the only significant variable included in any model (Figure 12). For future analyses, we will be incorporating continuous measures of habitats (such as vegetation density and least cost distance to bay) to try and get at the question of variation in chick survival among habitat types.

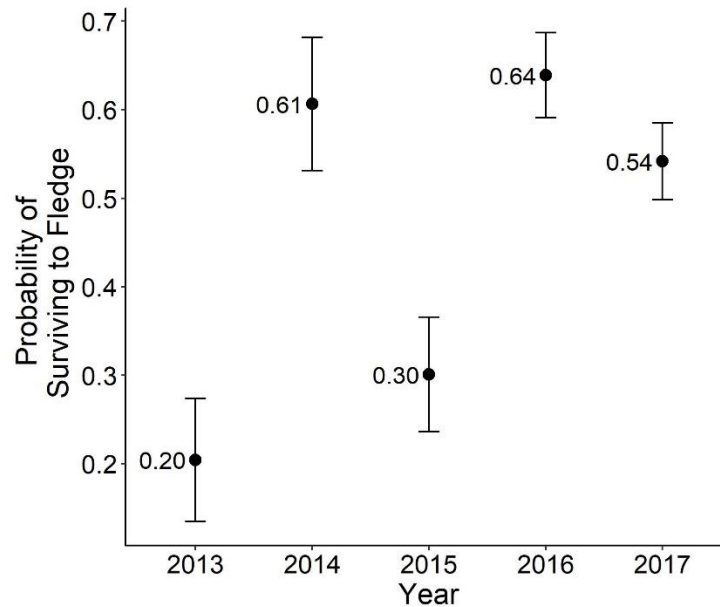


Figure 12. Probability of surviving to fledge (25 days) for chicks on Fire Island and Cupsogue.

Productivity

In 2017, 56% of chicks survived to fledging. Average productivity was 1.44 fledges/pair. The highest productivity was at Smith Point County Park, whereas the lowest productivity was at Cupsogue Beach County Park (Table 18). When comparing across the study site from 2013–2017, 2016 had the highest rate of productivity and 2015 had the lowest with 0.39 fledges/pair (Table 18).

Table 18. Piping plover pair counts and productivity (fledges/pair) in four different management units on Fire Island and Westhampton Island, May–August 2017.

Management Unit	2013		2014		2015		2016		2017	
	Pairs	Fledges /Pair	Pairs	Fledges /Pair	Pairs	Fledges /Pair	Pairs	Fledges /Pair	Pairs	Fledges /Pair
Robert Moses State Park	-	-	-	-	10	0.3	9	1.11	8	1.50
Fire Island National Seashore	10	0.6	8	0.75	3	1.67	8	2	11	1.56
Smith Point County Park	6	0.5	8	1.5	13	0.31	13	2.38	20	1.65
Cupsogue County Park	3	0	5	1.2	7	0.14	6	1	6	0.50
Total	19	0.47	21	1.14	33	0.39	36	1.75	45	1.44

Brood Home Range

Brood home range size increased in size with every day a brood aged (Figure 13), therefore to account for potential mortality of broods before fledgling, we only compared brood home range between habitat types for broods that successfully fledged at least one chick ($n = 71$). For the plover broods in 2017 that had at least one chick survive to fledging, with enough points to estimate a home range size ($n = 30$), we found that there was no difference between broods that had direct access to the bay within the subsite of their nest (STP, PSO, NBW and CSW) as compared to those that did not have direct access to the bay ($W = 120$, $P = 0.39$). However, over the four years of the study, we found that broods that have direct access to bay intertidal foraging within the subsite of their nest had smaller home range size than broods that did not have such access (Figure 14, $W = 865$, $P < 0.001$). This remained true when accounting for the number of points used to estimate the home range in a linear regression. Finally, we found that movement between observations of broods was smaller, on average, for broods that had bay access ($\bar{x} = 121.63$ m, $SE = 5.63$) than broods that did not ($\bar{x} = 174.01$ m, $SE = 7.71$; $W = 149890$, $P < 0.001$).

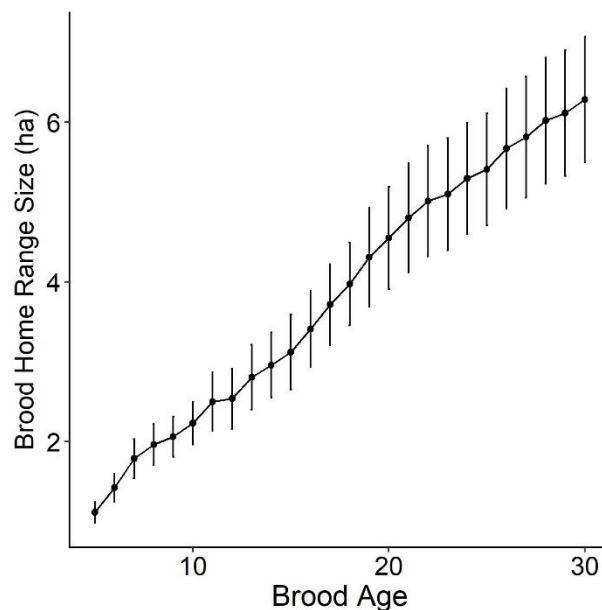


Figure 13. Cumulative home range size for piping plover broods as age increased. Home ranges were only calculated for age classes where up to that point, the brood had at least 5 locations.

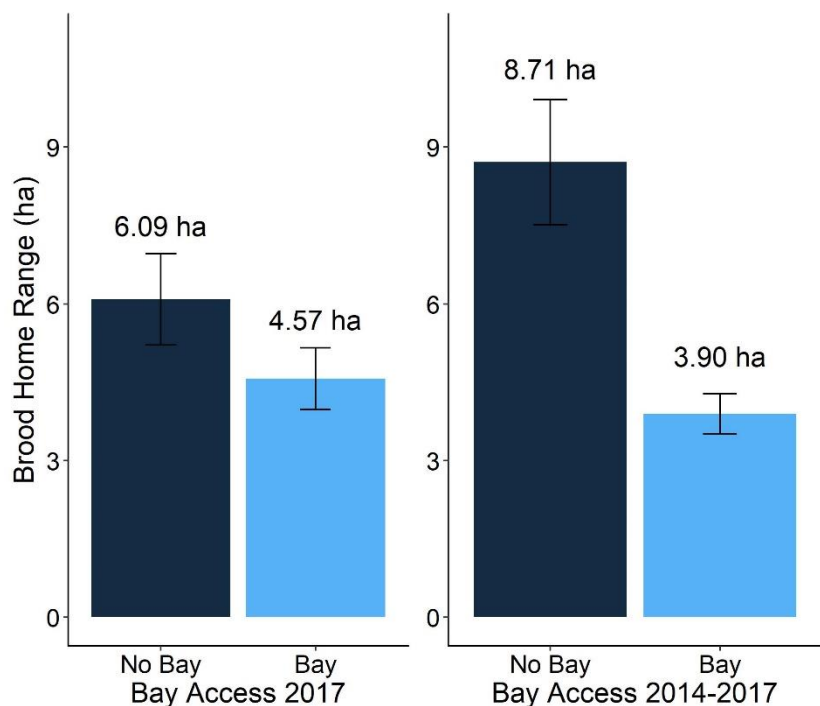


Figure 14. Brood home range size for 2017 (left) and 2014–2017 (right), estimated using minimum convex polygons for broods that fledged at least one chick.

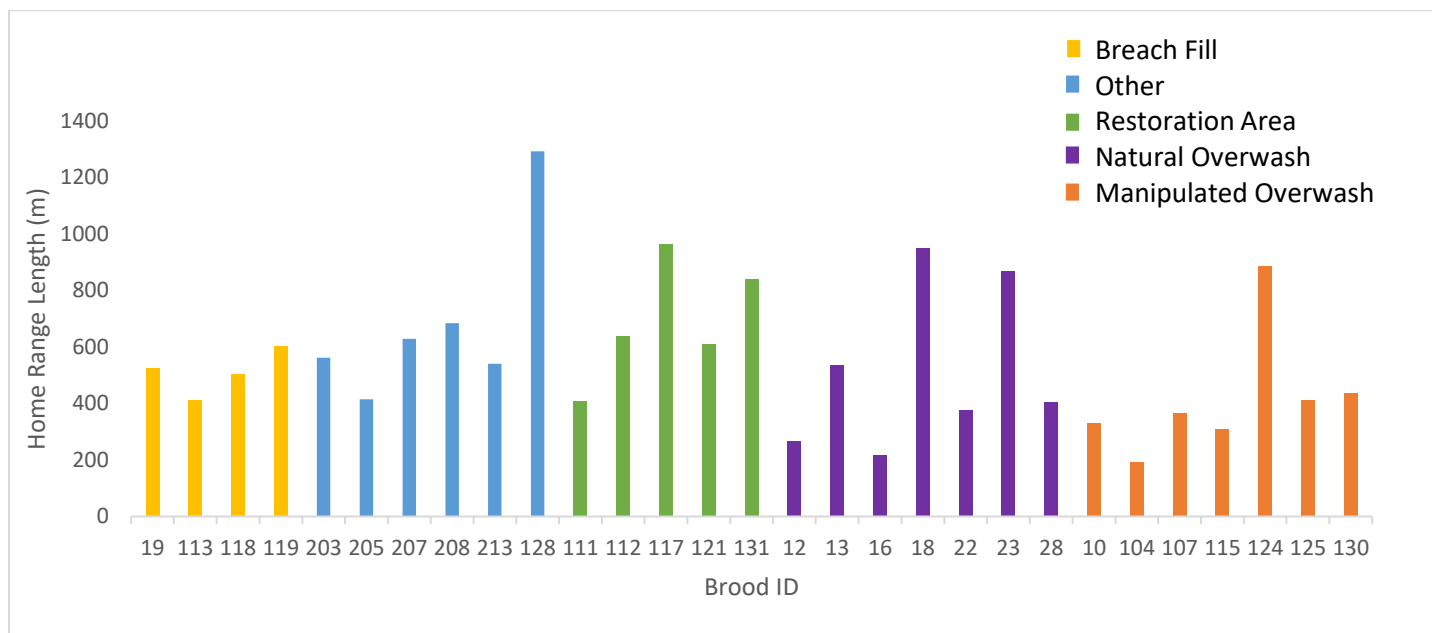


Figure 15. Brood home range lengths of 2017 piping plover broods that foraged on filled breach, other, restoration, natural overwash and manipulated. This figure only shows broods that had ≥ 10 home range location points available.



Figure 8. Brood ranges of piping plovers at Robert Moses State Park. Imagery captured in April, 2017.

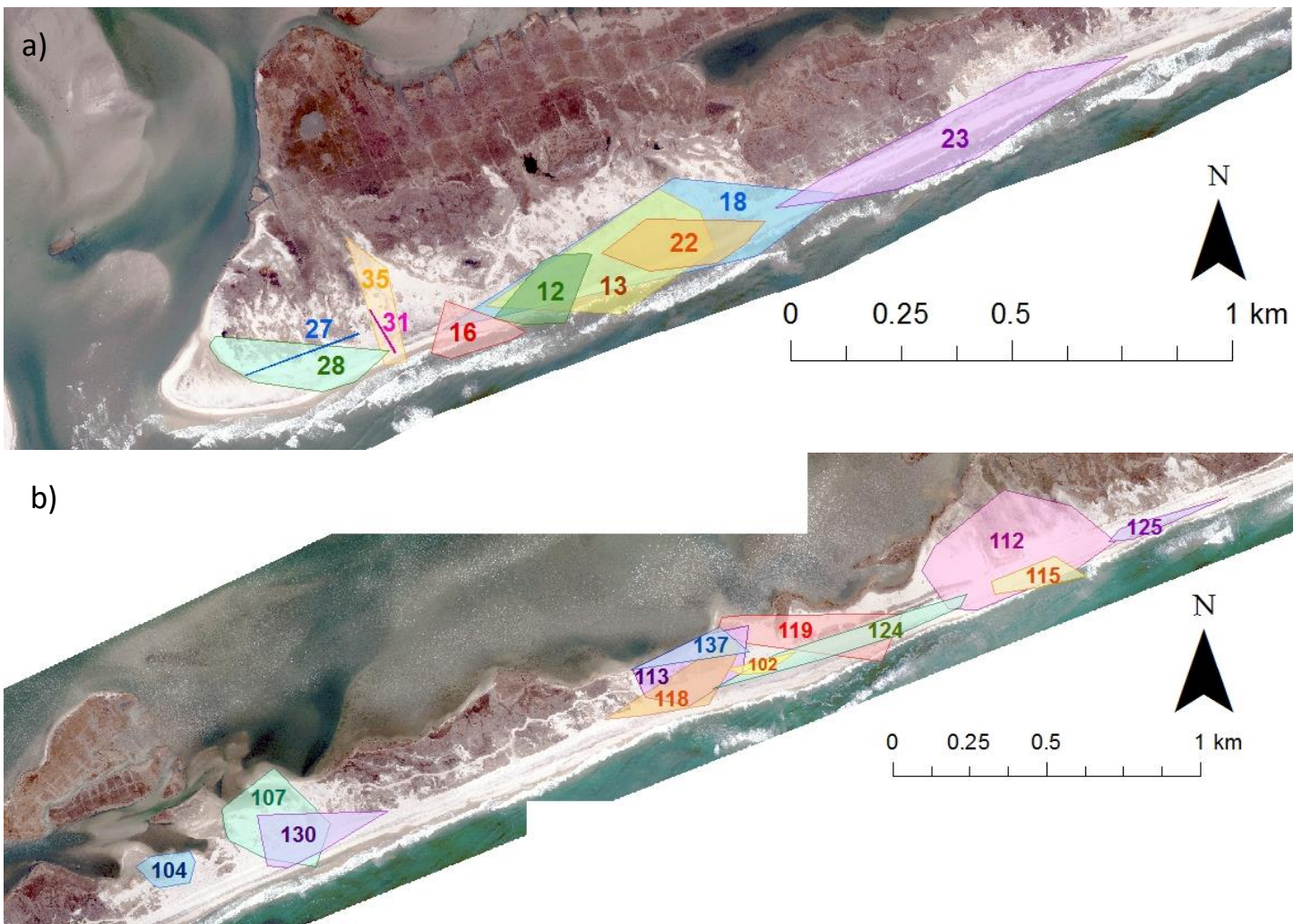


Figure 9. Brood ranges of piping plovers at a) Old Inlet East and b) Western section of Smith Point County Park (including subsites: PSO through NBE). Imagery captured in April, 2017.

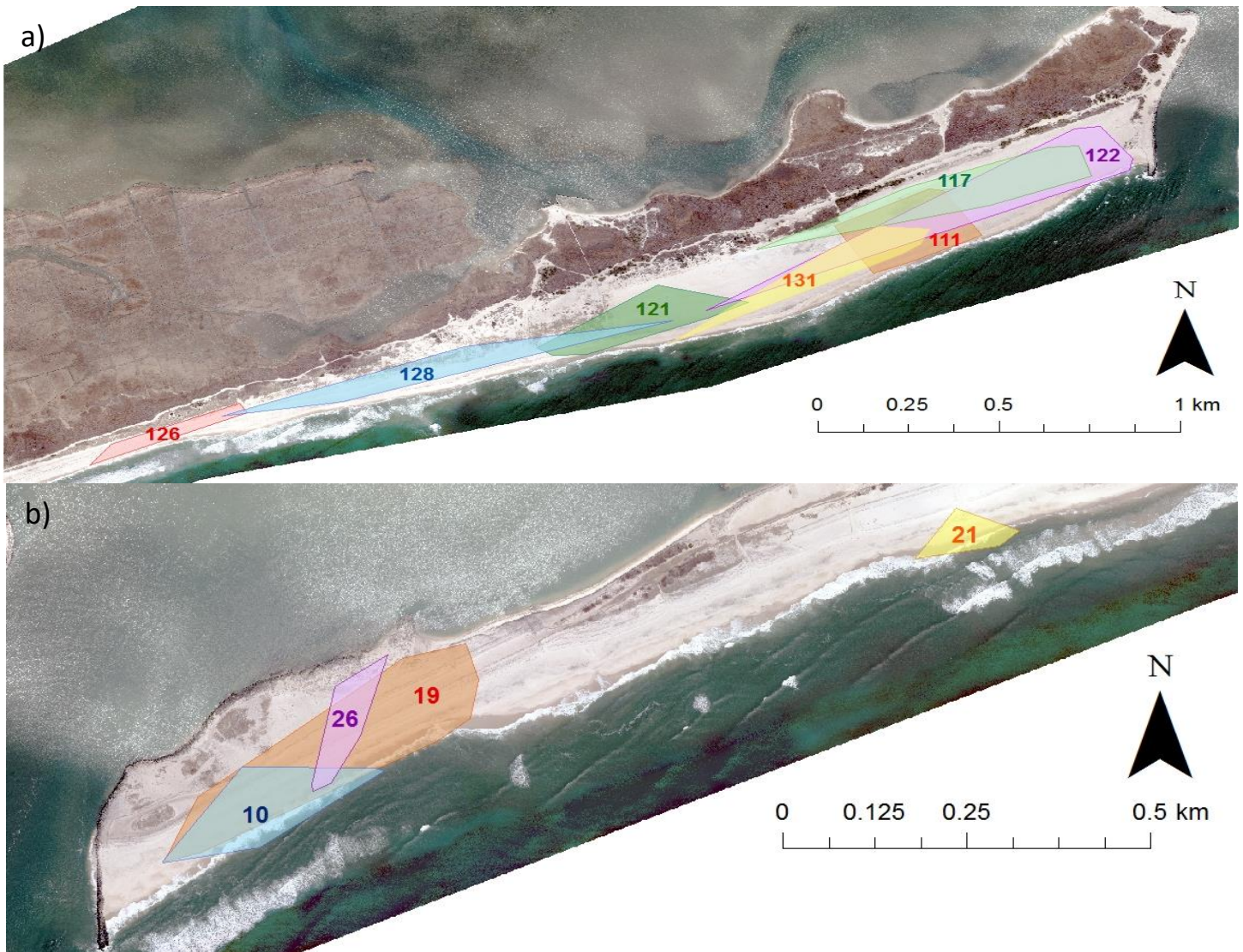


Figure 10. Brood ranges of piping plovers at a) Eastern section of Smith Point County Park (including subsites: SPTW-SPTE) and b) Cupsogue Beach County Park, May-August 2017. Imagery captured in April, 2017.

Objective 5: Compare the relative contribution of local growth (local recruitment) vs immigration in population growth associated with new habitats

Immigration and recruitment

For our study, the breeding population was made up of 31–54% returning nesters annually (Table 19). The remainder of the birds that nested in the study area hatched in the study area the year before (6–19%) or were unbanded birds captured during the year (immigrants, 35–63%). The count of immigrants may be a slight over-estimate due to a small number of birds that did not get banded in the year prior, due to early nest failure or trap aversion. Our top model did not include support for annual variation in immigration rates, as long as Democrat Point in 2015 is accounted for. Estimates from the top Pradel model with annual variation suggest the highest immigration rate in 2016, and the lowest in 2015, though confidence intervals overlap for all four years, indicating non-significant differences. Additionally, for 2016 and 2017, we were able to account

for birds at Robert Moses State Park, making it challenging to compare estimates from before 2016 to those for 2016 and later.

Following creation of the restoration areas, at least one locally recruited young nested in at least one of the restoration areas in each year. Among all four years, at least 50% of local recruits nested in new habitats created since Hurricane Sandy (Table 20). Restoration areas and manipulated overwashes supported most natal recruits with all years combined, manipulated and natural overwashes supported the most immigrants and returning nesters (Table 20, Table 21, and Table 22).

Table 19. Nesting adult piping plovers broken into categories of number of individuals that had nested in the study area previously the number of recruited young and the number of immigrants. Numbers for 2016 and 2017 include the entire study area (Robert Moses State Park to Cuposuge Beach County Park), Numbers for 2014 and 2015 are only the eastern study area population (NPS Wilderness Area to Cupsogue Beach County Park).

Year	Returning nesters	Local Recruits ^a	Immigrants ^b	Immigration rate ($\pm$ SE) ^c
2014	17 (46%)	2 (8%)	17 (46%)	0.89 (0.27)
2015	20 (54%)	4 (11%)	13 (35%)	0.54 (0.16)
2016	21 (31%)	4 (6%)	42 (63%)	0.91 (0.19)
2017	32 (38%)	16 (19%)	36 (43%)	0.71 (0.13)

^a Local recruits refer to individuals that were hatched as chicks in the study area and returned the year following to nest

^b Immigrants refers to a count of the individuals that were newly banded, and any birds that were known to have hatched outside of the study area and nested on Fire Island or Cupsogue

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

Table 20. The number of yearlings recruited into the population, which were banded as chicks in the year prior and returned the following year to breed, in new habitats on Fire Island and at Cupsogue following Hurricane Sandy and subsequent habitat alteration. The numbers are the numbers of recruits that had a nest in that habitat type, with percentage of total yearling recruits that year.

Year	Total # Recruits	Natural Overwash	Manipulated Overwash	Breach Fill	Restoration Area	Percentage of recruits in new habitats
2014	2	0	1 (50%)	0	NA	50%
2015	5	0	2 (40%)	0	3 (60%)	100%
2016	4	1 (25%)	0	1 (25%)	1 (25%)	75%
2017	15	3 (20%)	4 (27%)	2 (13%)	3 (20%)	80%
Total	26	4 (15%)	7 (27%)	3 (12%)	7 (27%)	81%

Table 21. The number of individuals that immigrated into the population (unbanded individuals that were banded), divided into the new habitat types on Fire Island following Hurricane Sandy and subsequent habitat alterations. The numbers represented are the number of immigrants that had a nest in that habitat type, with percentage of total immigrants that number represents.

Year	Total # Immigrants	Natural Overwash	Manipulated Overwash	Breach Fill	Restoration Area	Percentage of immigrants in new habitats
2014	16	3 (19%)	8 (50%)	2 (13%)	NA	81%
2015	13	1 (8%)	5 (38%)	1 (8%)	3 (23%)	77%
2016	26	7 (27%)	7 (27%)	1 (4%)	4 (15%)	73%
2017	30	9 (30%)	6 (20%)	3 (10%)	7 (23%)	83%
Total	85	20 (24%)	26 (31%)	7 (8%)	14 (16%)	79%

Table 22. The number of returning nesters, divided into the new habitat types on Fire Island following Hurricane Sandy and subsequent habitat alterations. The numbers represented are the number of returning nesters that had a nest in that habitat type, with percentage of total returning nesters that number represents.

Year	Total # Returners	Natural Overwash	Manipulated Overwash	Breach Fill	Restoration Area	Percentage of returners in new habitats
2014	16	5 (31%)	5 (31%)	2 (13%)	NA	75%
2015	20	3 (15%)	7 (35%)	5 (25%)	1 (5%)	80%
2016	21	6 (29%)	8 (38%)	4 (19%)	0	58%
2017	32	8 (25%)	11 (34%)	3 (9%)	2 (6%)	75%

In 2015, immigrant piping plovers used habitat in proportion to their availability ($\chi^2=8.33$, $P = 0.08$). Returning nesters selected breach fills and avoided pre-sandy habitat ($\chi^2=47.49$, $P < 0.001$, Figure 19). Natal recruits did not use habitat equal to availability ($\chi^2=20.662$, $P < 0.001$), as they did not use pre-Sandy habitat (“other”), natural overwashes or breach fills at all. Sample sizes were small, leading to large confidence intervals. When immigrants and natal recruits are combined into a “new birds” category, they are avoiding natural overwash, and avoiding pre-Sandy habitat ($\chi^2=21.38$, $P < 0.001$).

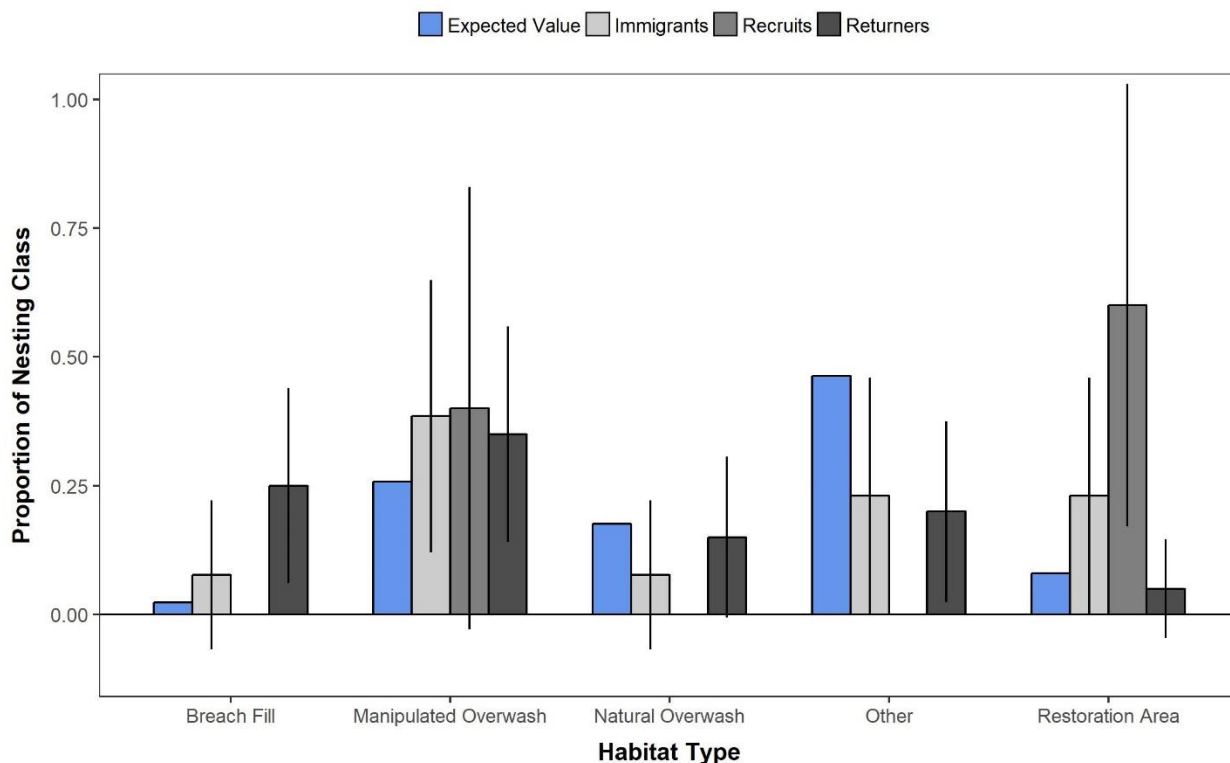


Figure 19. Used habitat by immigrant, natal recruit and returning piping plovers (and 95% confidence intervals), compared to the expected amount of use based on availability of habitat. The expected amount of use is represented in blue, and selection is evident when the expected amount is not within the confidence intervals of the used amount.

Survival and site fidelity

Our top model from the Barker model set indicated varying survival for HY birds among years with a constant adult survival rate for all years, and an age effect on site fidelity (F). Point estimates of HY survival vary each year of the study, with high years of HY survival relating to years of high chick survival to fledgling (Figure 20). AHY survival ($S = 0.72 \pm 0.03$) is within the range of past studies of piping plover adult survival. Site fidelity for AHY birds ($F = 0.89 \pm 0.04$) was higher than for HY birds ($F = 0.77 \pm 0.08$). Detection probability was constant over both ages and all years ($p = 0.91 \pm 0.04$).

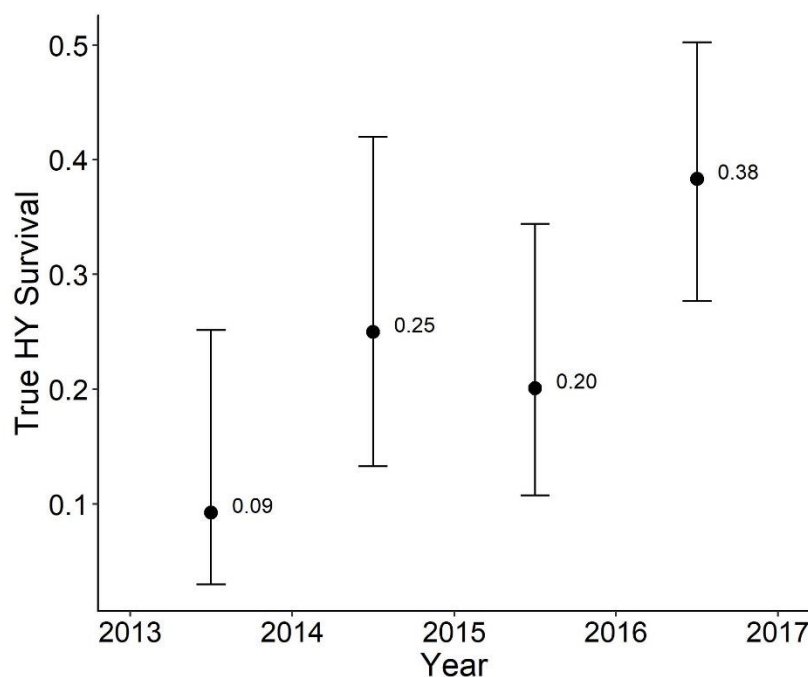


Figure 20. True survival (S) of hatch-year (HY; from hatch to year 1) piping plovers on Fire Island from 2013 to 2016. Estimates are from the top-model with an additive effect of year and age. Error bars represent 95% confidence intervals.

Objective 6: Compare prey densities at several habitat types

Terrestrial Invertebrates

In 2017, there was a significant difference in the number of invertebrates collected among the three different geologic locations of bay beach, barrier interior and ocean beach (K-W $\chi^2 = 49.12$, $df = 2$, $P < 0.001$).

Invertebrate samples in the bay beach had a greater mean arthropod catch than barrier interior or ocean beach invertebrate samples (Table 23).

Table 23. Mean number of invertebrates captured during a 30-minute period at bay beach, ocean beach, and interior habitats on Fire Island and Westhampton Island, May through August 2017.

Geologic Location	Mean Arthropods/ 30min	SE	N
Bay Beach	4.89	1.00	71
Barrier Interior	1.67	0.31	165
Ocean Beach	1.07	0.21	98

Within habitat types, ephemeral pools and the bay beach had greater projected invertebrate catches than the ocean beach or barrier interior. For the treatment types, overwashes had greater invertebrate catches than the restoration areas and the breach fills (Table 24, Figure 21). Although the invertebrate catch rate was high in ephemeral pools everywhere, ephemeral pools were rare in restoration areas and breach fill areas, as indicated by the small sample size (Figure 21).

Table 24. Mean predicted number of invertebrates caught over a three-hour period in various habitats from 2014–2017. Actual samples were collected over 30-minute periods and projected over three hours for comparison to past studies.

Habitat	Overwash	SE	Breach Fill	SE	Restoration	SE
Bay Beach	22.90	3.65	14.01	3.23	11.46	3.87
Ephemeral Pool	35.81	7.93	21.91	5.86	17.92	6.59
Ocean Beach	10.35	1.65	6.33	1.39	5.18	1.66
Barrier Interior	5.49	0.77	3.36	0.70	2.75	0.86

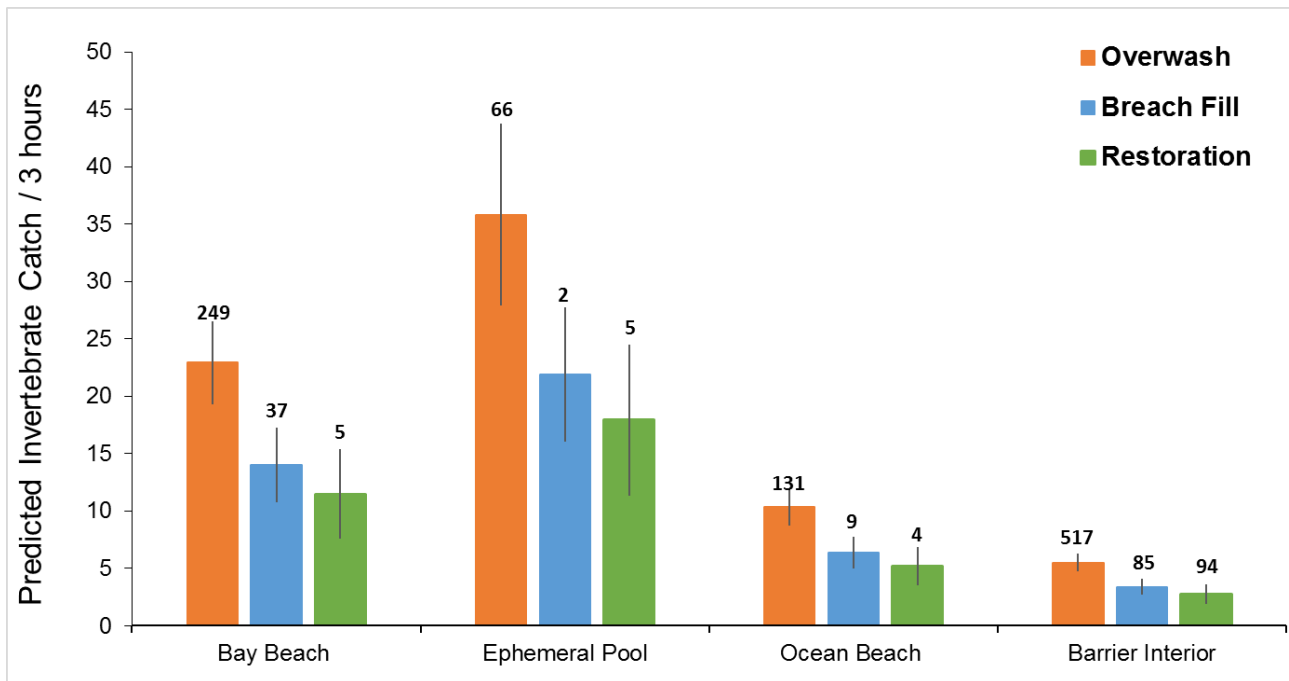


Figure 21. Invertebrate caught over a projected three-hour period in various habitats from 2014–2017, predicted using a negative binomial mixed model. Invertebrate samples were collected over 30-minutes, but projected to 3-hours to facilitate comparison to similar piping plover studies. Numbers above bars indicate sample sizes. Sample sizes differed among habitats because samples were collected in proportion to the habitat availability

Fecal Samples

A total of 34 fecal samples were collected during the 2017 field season and has been sent away for analysis.

Objective 7: Quantify the interactions of predators with shorebird nests and chicks

In 2017, we placed 23 remote nest cameras across the entire study site, with 8 cameras in Smith Point County Park, 2 in Cupsogue Beach County Park, 7 in Fire Island National Seashore (east of Old Inlet breach), and 6 in Robert Moses State Park. The length of time a camera remained present on an exclosure ranged from 2 to 26 days. The duration of time each camera was on an exclosure varied by nests. The length of time was determined by when we were able to capture both adults to when the nest hatched and the brood left the exclosure. Gulls and humans had the highest volume of detections, with 383 and 247, respectively (Table 25). The lowest type of disturbances were an owl, rabbit, and rodent with each only 1 detection (Table 25).

Table 25. Number of detections on 23 remote cameras placed on exclosures and the subsite of the detection. See Table 3 for a detailed description of these subsites.

Subsite(s) of Nest Camera	Disturbance ID	Number of Detections
BRO	Owl	1
CSE	Rabbit	1
CSW	Rodent	1
OIE	Unknown mammal	2
CSE, OIE, DMP	Unknown	3
DMP	American oystercatcher	7
STP, CSW, DMP	Crow	7
OIE, SPTE, NMA	Raccoon	7
STP, NMA, OSS, DMP, FLD4	Red fox	8
DMP, OIE, NBW	Toad	10
OIE, PSO, CSW, SPTE, FLD4	Unknown bird	10
OIE, CSW	Tern	16
OIE, SPTE, STP, DMP, OSS, FLD4	Deer	32
OSS, DMP	Crab	39
FLD3, FLD4, OSS, DMP	Human ^a	247
CSW, SPTE, DMP, FLD4, OSS	Gull	383

^a Only including non-monitors

Discussion

In a highly dynamic system such as barrier islands, changes in habitat distribution and quality can have substantial impacts on the organisms that rely on such systems. Hurricane Sandy and subsequent beach engineering caused significant habitat change, potentially impacting the nesting piping plovers. We studied the piping plovers, their habitats, and their prey on Fire Island and a portion of the nearby Westhampton Island, and aimed to assess the overall impacts on piping plovers and their invertebrate prey after Hurricane Sandy made landfall in 2012.

Prey availability in different habitat types likely drives habitat selection and chick survival. Bay intertidal and ephemeral pool habitats had greater terrestrial invertebrate availability, and more terrestrial invertebrates were caught in overwashes compared to breach fills and restoration areas. Greater invertebrate abundance and densities are likely to improve foraging efficiency of both adult plovers and chicks, leading to improved growth rates and eventually survival, and are thus one piece of the puzzle to explain piping plover habitat selection.

The available habitats on Fire Island and Westhampton Island for nesting and brood-rearing territories included overwashes (both natural and manipulated), breach fills, restoration area, backshore and barrier interior. The overwashes were created by high waters during Hurricane Sandy, and the breach fills were areas that were breached during the Hurricane and subsequently filled in. The restoration areas include New Made and Great Gun restoration area which were created prior to the 2015 nesting season. Each year, young from the previous year nested in areas of overwash and, when available, restoration areas, with overwashes and restoration areas contributing the most to the recruitment and immigration. In each year that the restoration areas were available, at least one new local recruit and immigrant nested in them. The Fire Island plover population is composed of at least 50% immigrants and natal recruits each year. Many of those individuals are using newly created habitat.

Although we did not see an irruptive population increase on Fire Island immediately following Hurricane Sandy, as was seen following past habitat creating events (Wilcox 1959, Cohen et al 2009, Hunt 2016, Robinson et al. *in prep.*). We did see a 20% pair increase from 2012 to 2016, of 30 to 36 pairs across the study area, and a 25% increase to 45 pairs from 2016–2017, resulting in a 40.6% increase since the hurricane. The slower increase relative to other irruptions was likely due to low reproduction in New York from 2009–2013 (mean = 0.84 chicks/pair, USFWS unpublished data). Given this level of reproduction, we would have expected a population decline or stability through 2014, due to lack of potential recruits, if there was no habitat effect. However, 2016 was a highly productive season for our study area, which may have produced an excess of individuals to colonize open habitats on Fire Island. With high levels of reproduction in some areas in 2017, the prospects for a further population increase in 2018 seem good.

Piping plovers traveled relatively long distances to forage along mud flats and habitats created by breaches and overwashes. Individuals also moved long distances to forage at the overwashes, sand bars, and mud flats whether they had an active nest or not. Ocean to bay overwashes exposed bayside foraging areas, especially the Pattersquash and Narrow Bay West overwashes, and the sand bar and mud flat at Cupsogue Beach County Park appear to be important to birds preparing to nest and to migrate. Long distance movements by plovers with active nests could be due to birds without access to bayside foraging searching for areas with greater prey abundances. In 2016, however, the distance moved by adults without direct bay access was not significantly different from that of the adults that did have bay access. The ability of adults with active nests to travel great distances to access the habitat could be a reason that clutch masses did not differ in piping plovers

that had direct access to bayside habitat when compared to those that did not have direct access. Fledglings also were found to fly long distances to Hurricane Sandy created habitats and the Cupsogue sandbar and mudflat, some of which are not protected during migration. A portion of these movements could be due to prospecting for future nesting sites, adding support for more protection of important plover sites during migration (Davis et al. 2017).

Nests had a higher probability of hatching in 2017 than in the three first years of the study, but lower than in 2016. In 2015, Smith Point County Park had many foxes, which depredated some of the nests by climbing into the 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. After the 2015 piping plover breeding season and prior to the 2016 breeding season, a mange outbreak left Smith Point County Park and the Wilderness east of Old Inlet without foxes, potentially contributing to the greater nest success in 2016 and 2017. At least one fox, however, was present east of Old Inlet during the breeding season of 2017 and depredated at least one plover nest.

The role of foxes in plover reproductive output requires continued study. In 2016, when foxes were not present on Smith Point and Fire Island National Seashore east of Old Inlet, plover reproductive output was very high (2.38 and 2 chicks/pair) more than twice the reproductive output in the other management areas, suggesting an important role for foxes. However, in 2016, Nest success was very high in all management areas (probability of a nest surviving to hatch = 0.75–0.85 in the 4 management areas) suggesting the possible influence of a variable, perhaps weather, affecting nest success in a similar way in all areas. In contrast, chick survival in 2016 was quite high in the fox –free areas (0.73 and 0.84. in Fire Island Wilderness, and Smith Point respectively) compared to other areas (0.37 and 0.22, in Robert Moses and Cupsogue, respectively) consistent with the possibility of a strong influence of foxes on chick survival. The combination of high nest success with low chick survival in Robert Moses and Cupsogue resulted in a modest reproductive output in those areas (1.0, 1.1 Chicks/pair). Fox numbers and movements in 2017 are still being analyzed but one or a few foxes were on Smith Point and the Fire Island National Seashore, east of Old Inlet. In 2017 the Fire Island management units had similar reproductive output, all substantially above the level needed for population stationarity (1.50–1.65). Recreational disturbance (DeRose-Wilson et al. *in revision*) and habitat variability (Loefering and Fraser 1995, Elias et al. 2000, Cohen et al 2009), likely affect these results. As of this writing, the mange zoonotic seems to be decreasing the fox population west of old inlet, so next year’s reproductive output in Robert Moses State Park may shed additional light on the effect of foxes on piping plover reproductive output.

With the combined results from adult movement, recruitment, immigration and prey availability, it is clear that the overwashes and other habitats created by Hurricane Sandy are important to piping plovers. This conclusion is supported by past studies of plovers, that have seen population increases following large habitat creating events, such as past storms in New York in the 1930s’ (Wilcox 1959) and 1990s’ (Cohen et al. 2009). Initial predator-prey observations on Fire Island suggest that piping plovers in some areas are more productive in years with lower fox densities, but that they are also impacted by their proximity to overwashes and bayside habitat (See Part III). The relationship between plovers and overwashes is likely related to the suppression of vegetation following the storm and the low-energy bayside foraging habitat created by ocean to bay overwash. Studies regarding the impact of habitats created by Hurricane Sandy, and predators on piping plovers are ongoing.

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Part II: Piping plover nest site selection before and after Hurricane Sandy: natural and human-created habitat on Fire Island and Cupsogue Beach County Park, New York

Introduction

Selection refers to the behavioral choices made by animals (Buskirk and Millspaugh 2006). Determining species selection is a common goal within the wildlife field and is often achieved through the use of resource selection functions. Resource selection functions measure non-random relationships among locations that describe an animal's position within habitats (Morris et al. 2016). Understanding what habitat an animal selects is an important component to wildlife management and is especially important in species of conservation concern (Morris et al. 2016).

Piping plover nesting habitat is susceptible to many changes due to human use and development, vegetation growth, and storm events. Hurricane Sandy altered habitats on Fire Island and Westhampton Island, and subsequent actions by the U.S. Army Corps of Engineers (USACE) and others further changed the landscape. Virginia Tech's projects have aimed at understanding how piping plovers responded to natural and anthropogenic habitat changes and how to provide a basis for adaptive management.

In various studies, piping plover nest site selection has been driven by the need to select habitat features that maximize access to reliable food sources (Loefering and Fraser 1995, Elias et al. 2000, Fraser et al. 2005, Cohen et al. 2009, and Cohen and Fraser 2010), while distancing themselves from the high tide (Cohen 2005) and from predator-prone vegetation (Cohen et al. 2009). Additionally, piping plover densities have been found to be lower on beaches close to development (Houghton 2005). Using this *a priori* knowledge about piping plover habitat selection, we set out to model piping plover nest site selection before and after Hurricane Sandy. With habitat selection models, we can evaluate the effects of natural coastal processes on this threatened shorebird and, in particular, compare the suitability of natural and engineered habitats. The objectives of this study were to: 1) identify coarse-scale habitat variables that influence piping plover nest site selection; 2) determine if piping plovers selected nest sites differently before and after Hurricane Sandy; 3) estimate how much suitable piping plover habitat was available before and after Hurricane Sandy; 4) highlight areas of high probability nesting for informing future management.

Methods

Our study site spanned Robert Moses State Park to Cupsogue Beach County Park (Figure 22). We modeled piping plover habitat in ArcGIS 10.2.2 using 15-cm resolution multispectral imagery from April, 2010 (New York State) and April, 2015. We fit models based on imagery and nest locations from 2010 and 2015. We used spring imagery to ensure that the vegetation growth stage in the imagery approximately matched the vegetation stage when piping plovers return to the breeding grounds in March and select their territories.

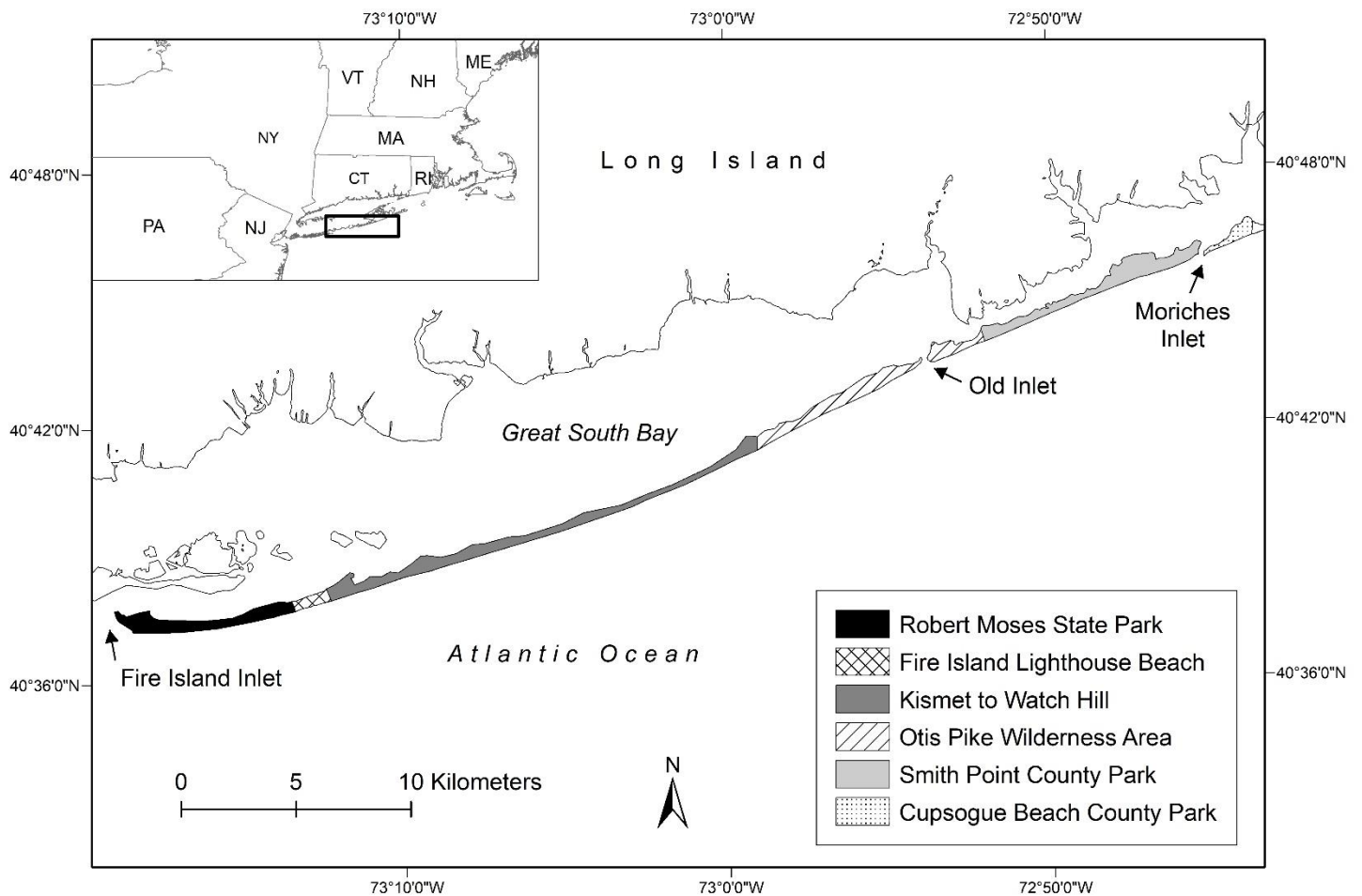


Figure 22. Fire Island and Westhampton Island study site, divided by management unit.

To set a boundary for habitat classifications, we digitized 2010 wet dry tide lines using the 2010 imagery. For 2015, we used habitat above spring high tide for habitat classifications. Spring tides are semidiurnal tides of increased range that occur with new or full moons (Pugh 1987). We used the spring high tide line to approximate the ocean- and bayside limits of useable nesting habitat. If a nest was below the spring high tide line, it would not persist through an incubation cycle (approximately 34 days from first egg to hatch) without a high probability of tidal flooding. Spring high tide lines within the study area were mapped by foot and by ATV in May and June of 2015 using handheld Trimble GPS units.

Field mapping did not cover the entire ocean beach, therefore, we used the elevation of the mapped line to complete the ocean high tide line across the study site. In ArcMap, we buffered the field-mapped ocean high tide line by 3m on each side. We extracted 1m LiDAR elevation values (collected in April, 2015) to these buffers to determine the difference in elevation below and above the field-mapped ocean high tide line. We calculated the 75th percentile values from the buffer below the high tide line (2.23 m) and the buffer above the high tide line (2.31 m). We reclassified the LiDAR to contain only values from 2.23–2.31 meters. This created a line of elevation values within this range across the entire ocean beach. For 2015 areas that hadn't been field-mapped, we digitized these pixels to complete the ocean high tide line. 2015 bay high tide lines

were mapped entirely in the field at the same time as the ocean high tide line. For our 2010 model, we digitized the high tide line based on imagery.

The United States Fish and Wildlife Service (USFWS) Long Island Field Office provided coordinates of 2010 plover nest locations. We collected the coordinates of piping plover nest locations using a Trimble GPS unit during the 2015 breeding season. We generated random points, matching them to the number of nests each year (2010, n=61; 2015, n=46). We limited our random points to dry sand because we wanted to know the characteristics of dry sand locations piping plovers selected. We used a maximum likelihood supervised classification to identify four cover type classes within the imagery: dry sand, wet sand, vegetation, and water.

Model Variables

We created ArcGIS-based datasets using 8 variables for 2010 and 2015. We used ArcGIS tools and created raster layers for each, from which we characterized nests and random points with respect to the variables.

Elevation

We extracted 1-m resolution LiDAR elevation values to every nest and random location (LiDAR collected in April, 2010 and April 2015).

Distance to development

We digitized development using April 2015 imagery. Development included all man-made structures including roads, buildings, parking lots, and boardwalks. We calculated the Euclidean distance from nests and random points to development polygons.

Distance to ocean/bay high tide

In addition to the field-mapped bay high tide line, we digitized bay access areas that were accessible to a flying bird. We used the field-mapped/digitized spring high tide lines to determine Euclidean distances from nests and random points to the ocean and bay high tide lines. Bay access areas included areas of foraging habitat (intertidal sand) on the island that were disconnected from dry sand habitat but may have been accessible to adult plovers by flight.

Least cost distance to ocean/bay high tide

We estimated the least cost path distance to the ocean spring high tide and bay spring high tide lines. Ocean spring high tide access included all ocean side intertidal access, as well as high-energy tidal access along inlets. Bay high tide consisted of all sandy, low-energy, bay tidal access as well as persisting, low-energy, eutrophic pools. We estimated the least cost distance to these features, such that the path did not pass through a pixel of vegetation or water but only through wet and dry sand.

Dry sand within a 500-m radius

To obtain an index to the amount of open sand habitat near nests, we counted the number of pixels of dry sand that were within a 500-m radius of each sand pixel.

Backshore width

We define backshore width as the distance perpendicular to the ocean tideline to the nearest back-beach barrier. The back-beach barrier (BBB) was defined as the farthest bayward barrier that would prevent plover chick movement. The BBB was digitized using each respective year's imagery and LiDAR. We digitized where the sandy backshore met dunes, vegetation, development, roads, and bay high tide lines.

We first created lines that were perpendicular to the ocean high tide. These lines were converted to polygons and given a backshore width value. Due to the differences in azimuth of the spring tide line on various parts of the ocean front, we created a number of polygons to ensure that polygons were as close to perpendicular to the shore as possible. Each polygon was 10 meters wide and covered the entire width of the island at that

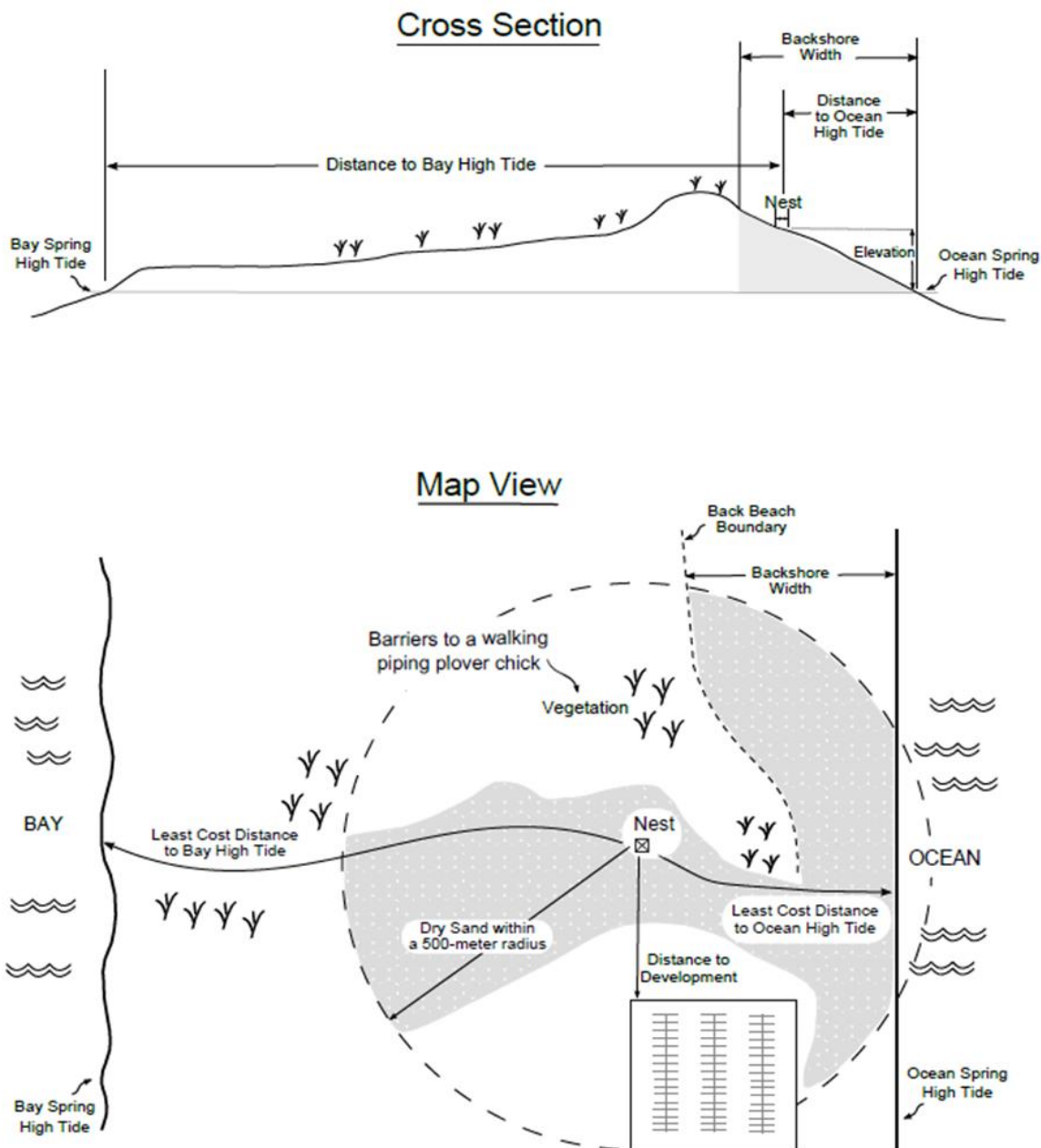


Figure 11. Cross-section and overhead diagram illustrating the 8 variables used in model selection.

location. We determined the distance to ocean from every polygon where it met the BBB line. We then attributed these distance-to-ocean values to the entire width of the polygon, from the high tide line to the BBB. Therefore, along the same perpendicular, a pixel 100 meters from the shore has the same beach width value as a pixel 1,000 meters from the shore (Figure 31). We converted the polygons and their corresponding backshore width values to a continuous raster so that every pixel had a backshore width value.

Analysis

We fit logistic regression models to the 2010 and 2015 datasets. We standardized all variables prior to analysis and checked for correlations among them. Least cost distance to ocean and distance to ocean were highly correlated ($r=0.99$). Backshore width and dry sand within 500-m were correlated in both 2010 and 2015 ($r=0.78$ and $r=0.79$). We removed distance to ocean and dry sand within a 500-m radius from our analysis; we kept least cost distance to ocean and backshore width as variables because they were retained longer than their respective collinear variable in backward selection, thus better explaining the variability of nest sites.

We considered each nest, including renests, to be independent because the random effect of female did not explain variability within the 2015 regression. We conducted backward selection logistic regressions in SAS on the six remaining covariates for both years. We used Akaike's Information Criterion corrected for small sample bias (AIC_c) to rank models. We started with the global model and removed variables until the AIC_c increased. We selected the model with the lowest AIC_c as the top model (Burnham and Anderson 2002). We considered variables significant if their 95% confidence intervals did not overlap zero.

We created a standardized raster for each variable. We input the estimates from our top model into the logistic regression equation and we generated a raster with a probability of nest occurrence (0–1) for every pixel. We created the 2010 suitable habitat map by projecting the 2010 top model onto the 2010 imagery. We created the 2015 suitable habitat map using the 2010 and 2015 top models, both projected onto the 2015 imagery, thus accounting for a piping plover's nest site selection pre- and post-hurricane. We assumed the characteristics that piping plovers found suitable in 2010 would have remained suitable, but that new habitat types might also be selected by piping plovers. We used a suitability threshold of 0.5 to facilitate comparison between years. We termed dry sand pixels with a probability of use ≥ 0.5 as "suitable piping plover habitat" and pixels with a probability < 0.5 as "unsuitable." Thus, for the 2015 map, all suitable pixels, whether calculated from the 2010 or 2015 model, were merged and weighted equally as suitable piping plover habitat (Figure 24–Figure 28).

We calculated the number of dry sand pixels and the number of suitable pixels ($p \geq 0.5$) and summed each by habitat type—natural overwash, manipulated overwash, plover restoration area, breach fill, planted dune, manipulated dune, beach nourishment, dredge spoil, and other (see Table 1). USACE provided restoration area polygons and we created polygons to delineate all other habitats. Planted dune refers to the regions of USACE-created dune that were planted with beachgrass. Manipulated dune encompassed all other dunes constructed to stabilize the island through USACE-engineering, snow fencing, or Christmas trees. The area of sand placed in beach nourishment by the USACE—pumping of sand on to the berm and beachfront from November 2014 to March 2015—was calculated by creating a polygon that encompassed the area between the 2014 and 2015 spring high tide lines. "Other" included all sand that was not redeposited inland during Hurricane Sandy nor engineered following the storm. Various regions of the island were engineered following overwash or previous engineering. For example, plover restoration was built atop previously overwashed

sand, and manipulated dunes were engineered over breach fills. For our estimates, we categorized dry and suitable sand into the most recent habitat creation and did not count any sand area twice.

We estimated the amount of 2015 suitable piping plover habitat within protected areas. We termed protected areas as regions that were enclosed with symbolic fencing. Symbolic fencing was mapped in the field in 2015 and 2016. In regions where 2015 data were absent, we used 2016 for delineating protected area boundaries as the symbolic fencing didn't vary significantly from year to year (A. McIntyre, personal communication). We estimated the amount of suitable habitat protected and unprotected across management units and across natural and engineered habitat types. Some symbolic fencing was adjusted or erected mid-season to accommodate for nesting birds, such as within the communities from Kismet to Watch Hill (L. Ries, personal communication). These areas were excluded from our analysis because we used the best estimate of symbolic fenced areas at the earliest point in the season, when birds were beginning to set up their territories. Protected areas that were not fully bordered by symbolic fencing were due to primary dunes or dense vegetation, therefore, the boundaries were digitized along those natural barriers. We hypothesized that piping plovers were selecting to nest within protected areas more often than would be expected if they were randomly distributed across available dry sand. To investigate whether piping plovers showed disproportionate use, we conducted a chi-squared (χ^2) analysis for nests from 2015–2017.

Results

In 2010, plovers selected nest sites farther from the ocean and bay and with lower elevations than would be expected at random. The top model (AIC_c , 133.13; Table 28) included least cost distance to ocean as the strongest predictor ($\beta=4.27$), followed by distance to bay ($\beta=0.79$), least cost distance to bay ($\beta=0.52$), and elevation ($\beta=-0.52$).

In 2015, piping plovers selected nest sites with short least cost distances to bay and ocean and wide backshores. The top model (AIC_c , 105.87; Table 28) contained least cost distance to bay, least cost distance to ocean, and backshore width. Least cost distance to bay was the strongest predictor of nest site selection ($\beta=-2.82$), followed by least cost distance to ocean ($\beta=-0.60$), and then backshore width ($\beta=0.41$; Table 27). The 95% confidence interval for least cost distance to bay and least cost distance to ocean did not include 0, but the confidence interval for backshore width included 0. Removal of backshore width from the model increased AIC_c from 105.87 to 106.14, therefore, backshore width was retained in the final model.

Table 26. Logistic regression models for determining the suitability of 15-cm pixels for piping plover nesting on Fire Island and Cupsogue Beach County Park, New York in 2010 and 2015. Models were created using piping plover nest locations random points confined to dry sand, and multispectral imagery from respective years. Model ranking was based on a backward selection using Akaike’s Information Criterion corrected for small sample size (AIC_c), differences between those values (ΔAIC_c), and Akaike’s weights (w_i).

	Model	AIC_c	K^a	ΔAIC_c	w_i
2010	Least cost distance to ocean + distance to bay + least cost distance to bay + elevation	133.13	5	0	0.65
	Least cost distance to ocean + distance to bay + least cost distance to bay + elevation + backshore width	134.92	6	1.79	0.26
	Least cost distance to ocean + distance to bay + least cost distance to bay + elevation + backshore width + distance to development	137.14	7	4.01	0.09
2015	Least cost distance to bay + least cost distance to ocean + backshore width	105.87	4	0	0.58
	Least cost distance to bay + least cost distance to ocean + backshore width + elevation	107.05	5	1.42	0.28
	Least cost distance to bay + least cost distance to ocean + backshore width + elevation + distance to bay	108.80	6	3.46	0.10
	Least cost distance to bay + least cost distance to ocean + backshore width + elevation + distance to bay + distance to development	110.63	7	5.64	0.04

Table 27. Standardized estimates (β), standard errors (SE), and lower (LCL) and upper (UCL) 95% confidence intervals for variables affecting nest site selection of piping plovers on Fire Island and Cupsogue Beach County Park, NY in 2010 and 2015.

	Variable	β	SE	LCL	UCL
2010	Intercept	0.70	0.31	0.10	1.30
	Least cost distance to ocean	4.27	1.04	2.23	6.30
	Distance to bay	0.79	0.31	0.18	1.41
	Least cost distance to bay	0.52	0.24	0.05	1.00
	Elevation	-0.52	0.26	-1.03	-0.01
2015	Intercept	-0.76	0.55	-1.85	0.32
	Least cost distance to bay	-2.82	1.33	-5.43	-0.20
	Least cost distance to ocean	-0.60	0.29	-1.17	-0.04
	Backshore width	0.41	0.28	-0.14	0.96

Suitability maps (Figure 24–Figure 28) illustrate suitable and unsuitable piping plover habitat based on the 2010 model projected onto the 2010 imagery (2010 images) and the 2010 and 2015 models projected onto 2015 imagery (2015 images). For the 2015 map, all suitable pixels—whether calculated from the 2010 or 2015 model—were merged and weighted equally as suitable piping plover habitat. Suitable habitat shifted from narrow beaches in 2010 to primarily occurring within Hurricane Sandy-created overwashes and areas of wide ocean-to-bay expanses of dry sand in 2015. In 2010, most unsuitable habitat was closer to the ocean if habitat was available farther bayward. In 2015, unsuitable habitat occurred closer to communities, human use areas, and along narrow backshores.

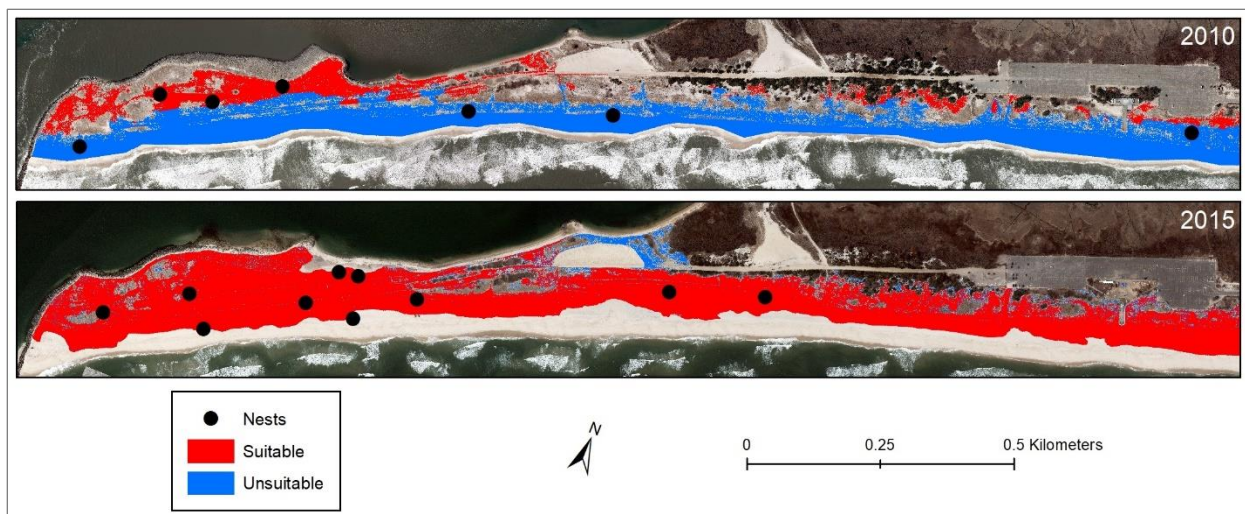


Figure 24. Cupsogue Beach County Park suitable and unsuitable habitat in 2010 and 2015.

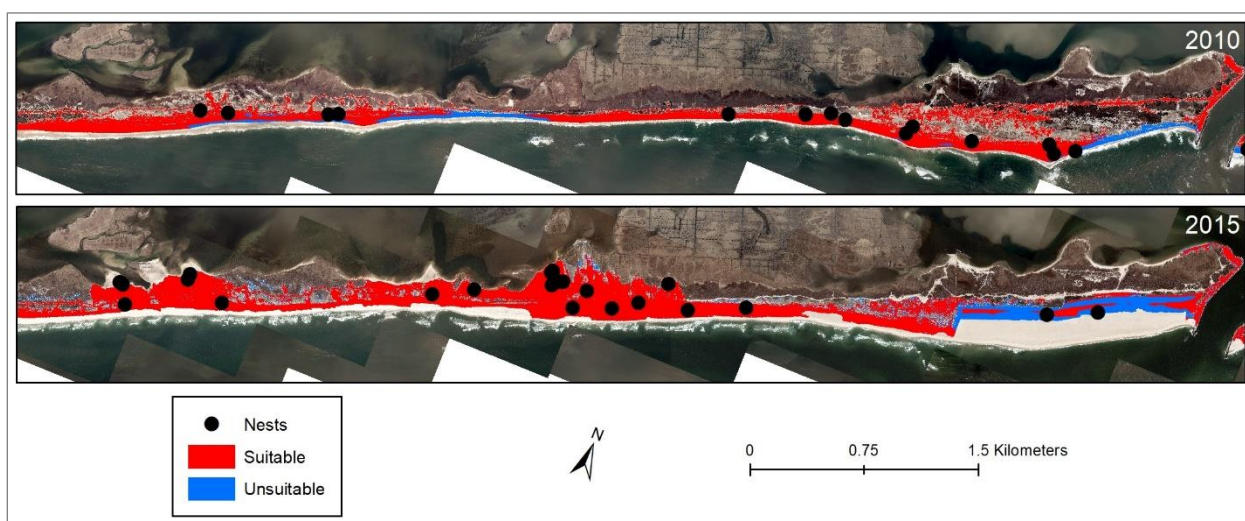


Figure 25. Smith Point County Park suitable and unsuitable habitat in 2010 and 2015.

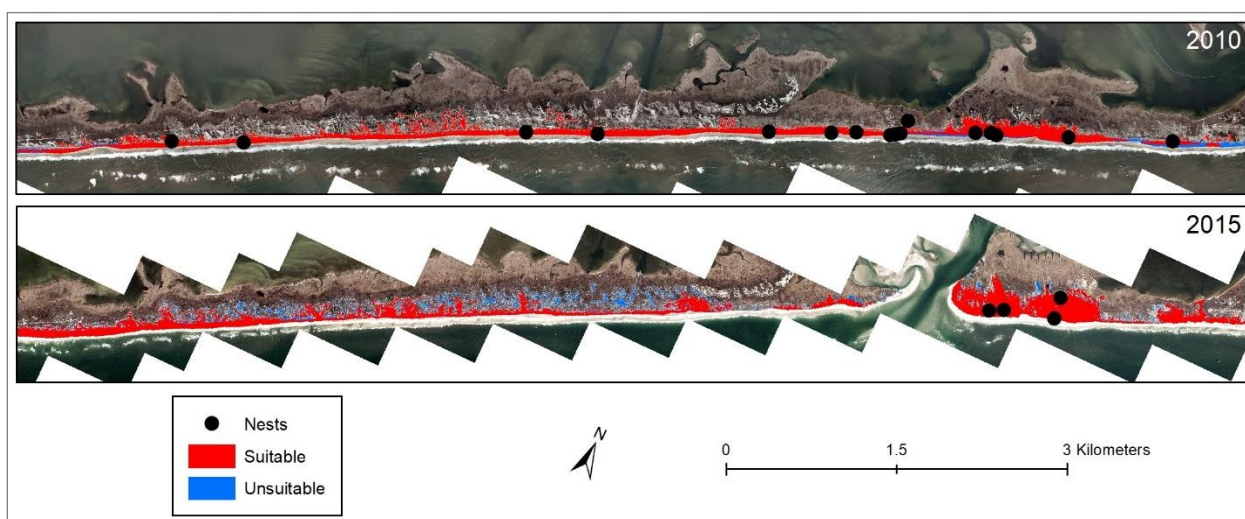


Figure 26. Otis Pike Wilderness Area suitable and unsuitable habitat in 2010 and 2015.

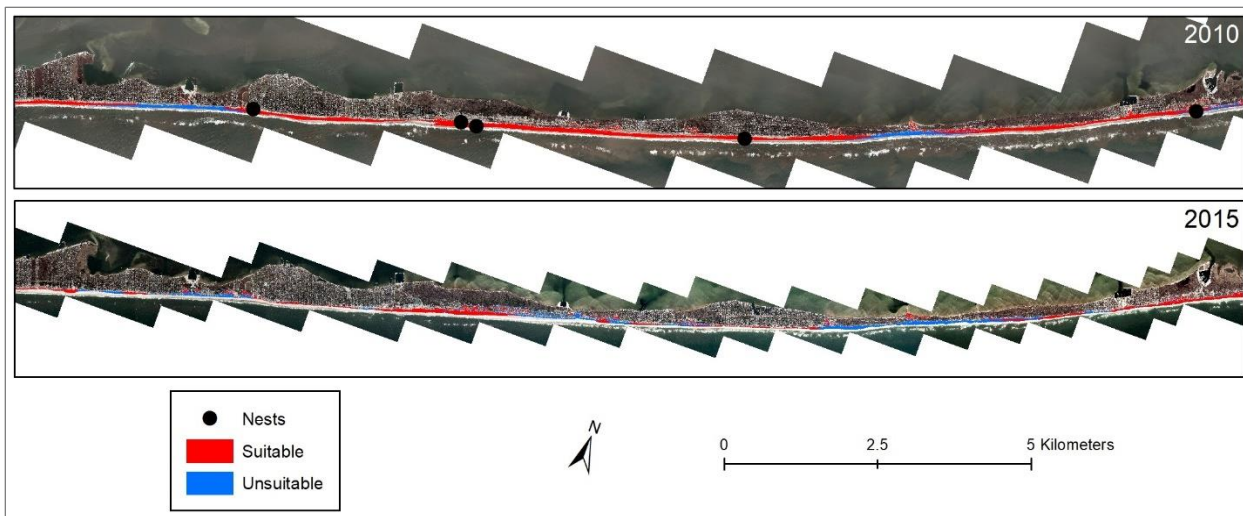


Figure 27. Kismet to Watch Hill suitable and unsuitable habitat in 2010 and 2015.

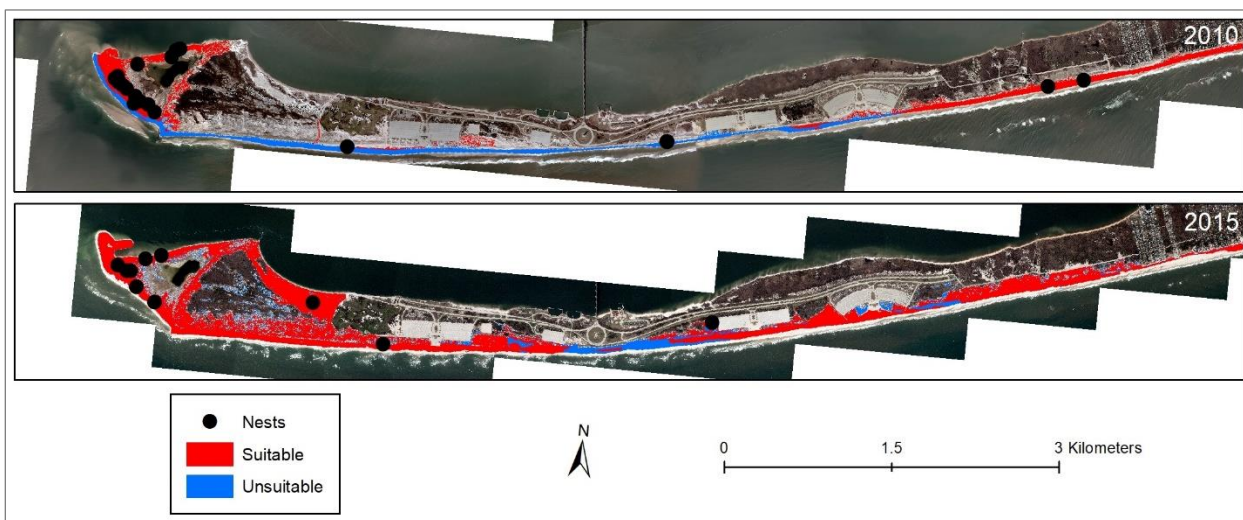


Figure 28. Robert Moses State Park and Fire Island Lighthouse suitable and unsuitable habitat in 2010 and 2015.

Figures 24–28. Piping plover nest site suitability maps for Cupsogue Beach County Park and Fire Island, New York in 2010 and 2015. Red pixels indicate suitable piping plover nesting habitat, with probability of occurrence values of ≥ 0.5 . Blue pixels indicate unsuitable habitat ($p < 0.5$). 2010 maps were created using the 2010 model applied to the 2010 imagery. 2015 maps were created using the 2015 and 2010 models applied to 2015 imagery, both years weighted equally. Only dry sand pixels above the high tide lines were modeled for suitability. Nest locations are in accordance with map year.

The amount of unvegetated dry sand on Fire Island and Cupsogue Beach County Park increased by 34% between 2010 and 2015 (Table 28). New sand in 2015 included 37 hectares created by natural overwashes, 35 hectares created by manipulated overwashes, 15 hectares placed or devegetated by the USACE in plover habitat restoration, 3 hectares placed by the USACE when filling Hurricane Sandy breaches, 7 hectares placed by USACE through creation of planted dunes, 10 hectares created by manipulated dunes, 12 hectares placed by USACE in beach nourishment, 16 hectares of dredge spoil, and 313 hectares of dry sand that was not overwashed nor engineered after the storm (Table 28). The amount of suitable piping plover habitat increased by 36% between 2010 and 2015. Man-made habitat changes created 54 hectares of suitable piping plover

habitat between 2010 and 2015, while natural storm overwashing processes created 70 hectares of suitable piping plover habitat (Table 28).

Table 28. Total dry sand and suitable habitat within each management unit in 2010 and 2015 divided by habitat type, rounded to the nearest hectare. 2010 estimates were produced by projecting the 2010 model onto 2010 imagery. 2015 estimates were produced by projecting the 2010 and 2015 models onto 2015 imagery, weighing each model equally.

Habitat & Sand Type		Robert Moses State Park and Fire Island Lighthouse		Kismet through Watch Hill		Otis Pike Wilderness		Smith Point County Park		Cupsogue Beach County Park		Total Study Area	
		2010	2015	2010	2015	2010	2015	2010	2015	2010	2015	2010	2015
Natural Overwash^a	Suitable ^j	0	0	0	0	0	35	0	0	0	0	0	35
	Dry Sand	0	0	0	0	0	37	0	0	0	0	0	37
Manipulated Overwash^b	Suitable ^k	0	0	0	0	0	0	0	31	0	4	0	35
	Dry Sand	0	0	0	0	0	0	0	31	0	4	0	35
Plover Restoration^c	Suitable	0	0	0	0	0	0	0	8	0	0	0	8
	Dry Sand	0	0	0	0	0	0	0	15	0	0	0	15
Breach Fills^d	Suitable	0	0	0	0	0	0	0	2	0	1	0	3
	Dry Sand	0	0	0	0	0	0	0	2	0	1	0	3
Planted Dunes^e	Suitable	0	0	0	0	0	0	0	6	0	0	0	6
	Dry Sand	0	0	0	0	0	0	0	7	0	0	0	7
Manipulated Dunes^f	Suitable	0	0	0	0	0	0	0	4	0	5	0	9
	Dry Sand	0	0	0	0	0	0	0	5	0	5	0	10
Beach Nourishment^g	Suitable	0	0	0	0	0	0	0	12	0	0	0	12
	Dry Sand	0	0	0	0	0	0	0	12	0	0	0	12
Dredge Spoil^h	Suitable	0	16	0	0	0	0	0	0	0	0	0	16
	Dry Sand	0	16	0	0	0	0	0	0	0	0	0	16
Otherⁱ	Suitable	36	107	90	42	62	43	73	38	4	7	265	236
	Dry Sand	61	127	110	75	68	55	80	48	15	8	334	313
Total	Suitable	36	123	90	41	62	78	73	101	4	17	265	360
	Dry Sand	61	143	110	75	68	92	80	120	15	18	334	448

^aDune sand redeposited by overwashing waves and never manipulated.

^bDune sand redeposited by overwashing waves which was then manipulated via USACE and/or other engineering.

^cUSACE-restored piping plover habitat.

^dHurricane Sandy breaches that were filled by the USACE.

^eUSACE dunes planted with beachgrass.

^fDunes created by snow fencing, Christmas trees, or USACE engineering.

^gBeach/berm widening by the USACE in Smith Point County Park.

^hDredged sand placed within Robert Moses State Park.

ⁱSand that was not overwashed during Hurricane Sandy nor engineered following the storm.

^jTotal hectares of dry sand with a probability of nesting ≥ 0.5

^kTotal hectares dry sand that was modeled for suitability.

Forty-five percent of suitable habitat within the study area was protected from recreational use in 2015 (Figure 29, Figure 30). Kismet to Watch Hill contained entirely unprotected habitat. The county parks contained the most protected area relative to the amount of suitable habitat, with Smith Point County Park and Cupsogue County Park having 72% and 68% protected from recreational use, respectively. Comparatively, at Robert Moses State Park and Fire Island Lighthouse and Otis Pike Wilderness Area, 39% and 40% of suitable piping plover habitat was protected from recreational use, respectively.

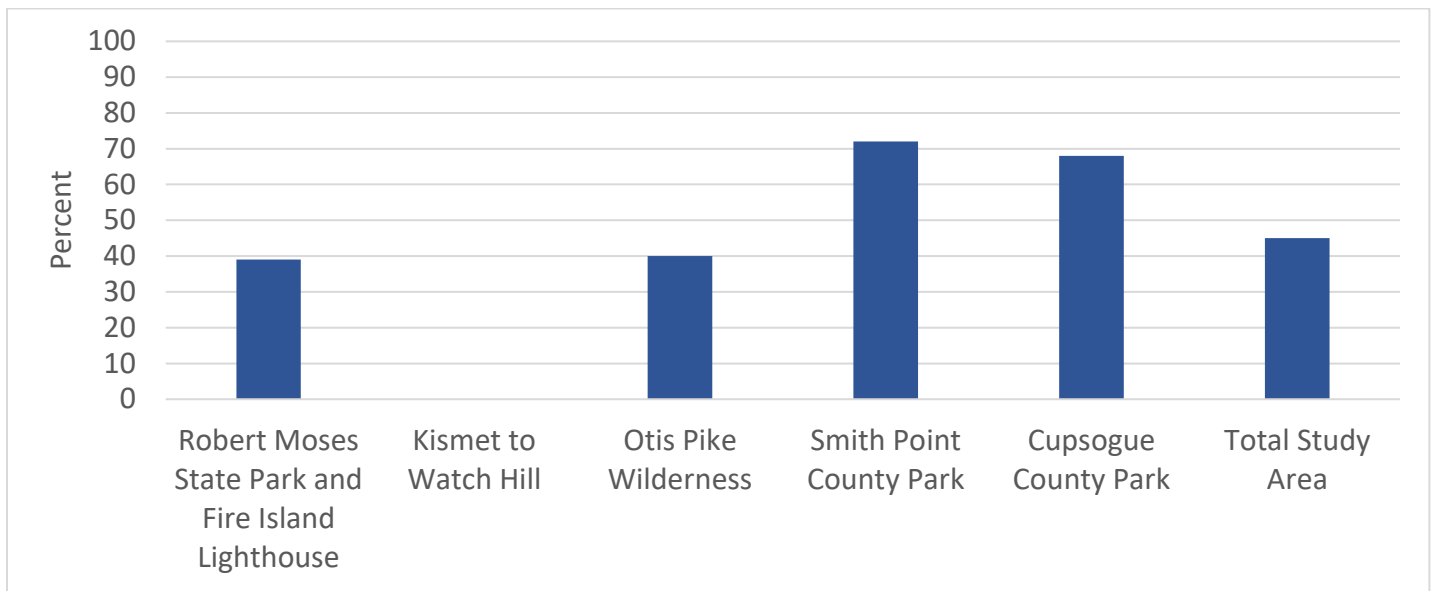


Figure 29. Percent suitable piping plover habitat protected from recreational use with symbolic fencing across management units on Fire Island and Westhampton Island, NY.

All of the suitable habitat within plover restoration areas was protected (Figure 30). Natural and manipulated overwashes were 89% and 86% protected, respectively, while breach fills were 85% protected. Planted dunes were 98% protected while manipulated dunes were 92% protected. Across all engineered habitat types, beach nourishment showed the lowest proportion protected, with only 44% closed to recreation. Twenty-eight percent of “other” habitat, that was not overwashed nor engineered following Hurricane Sandy, was within protected areas.

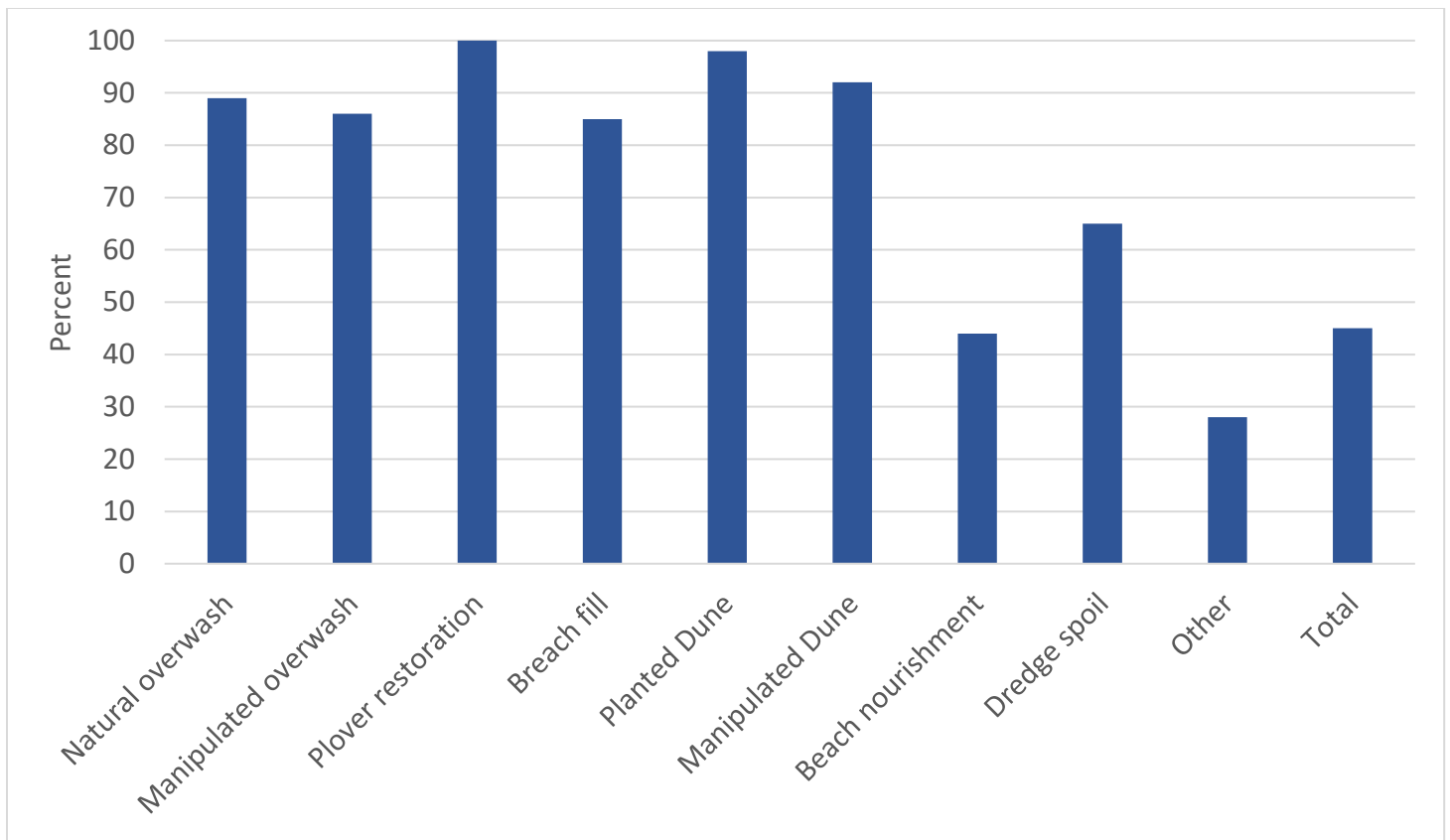


Figure 30. Percent suitable piping plover habitat protected from recreational use with symbolic fencing across habitat types on Fire Island and Westhampton Island, NY.

Nest distribution differed between protected and unprotected areas ($\chi^2=158.85$, $P<0.001$). Piping plovers showed preference for protected areas, with only 6 out of 151 nests placed outside of symbolic fencing from 2015–2017. In 2015, one nest was laid on a stockpile outside symbolic fencing in Field 4. In 2016, one nest was laid outside the symbolic fencing at OIWR. In 2017, two nests were outside the symbolic fencing at CSW, one at OSS, and one in the dunes at Field 3 behind the symbolic fencing.

Discussion

Piping plovers selected habitat differently before and after Hurricane Sandy. In 2010, when the only available nesting habitat was on narrow stretches of beach, piping plovers selected nest sites farther from the ocean, farther from the bay, and at lower elevations than would be expected at random. They selected nest sites that likely reduced the probability of tidal flooding of nests. Piping plovers selected nest sites farther than would be expected at random from the bay as the bird would fly (distance to bay) and as the bird would walk (least cost distance to bay). This pattern was a function of the birds spreading themselves out along the ocean beach, many nests being in the middle, wider portion of Fire Island, such as in Otis Pike Wilderness Area and the eastern portion of Smith Point County Park. Selection for lower elevation may be due to avoidance of dunes, where there may have been a high risk of predation in 2010.

In 2015, piping plovers selected nest sites that provided unimpeded access to bay and ocean-side foraging. The strongest predictors of site selection were least cost distance to foraging habitats. Proximity to bayside intertidal was the strongest driver of selection potentially because of higher invertebrate abundance than ocean intertidal zones (Patterson et al. 1991, Loegering and Fraser 1995, Elias et al. 2000, Fraser et al. 2005,

Cohen et al. 2009, and Cohen and Fraser 2010, Part II Objective 6: Compare prey densities at several habitat types). Chicks with access to bay habitats had higher daily survival rates than chicks without access to bay on Fire Island from 2014–2016 (Carey et al. 2016). Similarly, in Maryland, chicks foraging on bay beach had higher foraging rates, faster growth, and higher survival than chicks on ocean beach (Loegering and Fraser 1995). This could explain the selection of nest sites near bay-side foraging habitat. Plover nests in or near vegetation can have a higher rate of predation than those in open territories (Prindiville Gaines and Ryan 1988). Selection for wide backshores suggests that piping plovers maintained a large buffer from their nests to dunes and vegetation so they could readily see approaching predators.

Following Hurricane Sandy, new habitat was not occupied immediately, because of low local and regional reproductive output in 2012–2013 (USFWS unpublished data), and low local reproductive output in 2015 (Carey et al. 2016). In addition, it is possible that time was needed for development of adequate prey, for plover's acquisition of knowledge about the new habitat, and for locally hatched birds to return to the site and occupy new territories (Catlin et al. 2015). However, the increase in colonization in bay-to-ocean habitat indicates piping plover preference for these areas over narrower, oceanside beaches (Figure 31). Furthermore, pairs continue to colonize the USACE-created Great Gun restoration area. We expect that if reproductive output and survival remains high, the population will continue to grow in response to the increase in suitable habitat until it reaches the new carrying capacity or habitat degrades through vegetation succession.

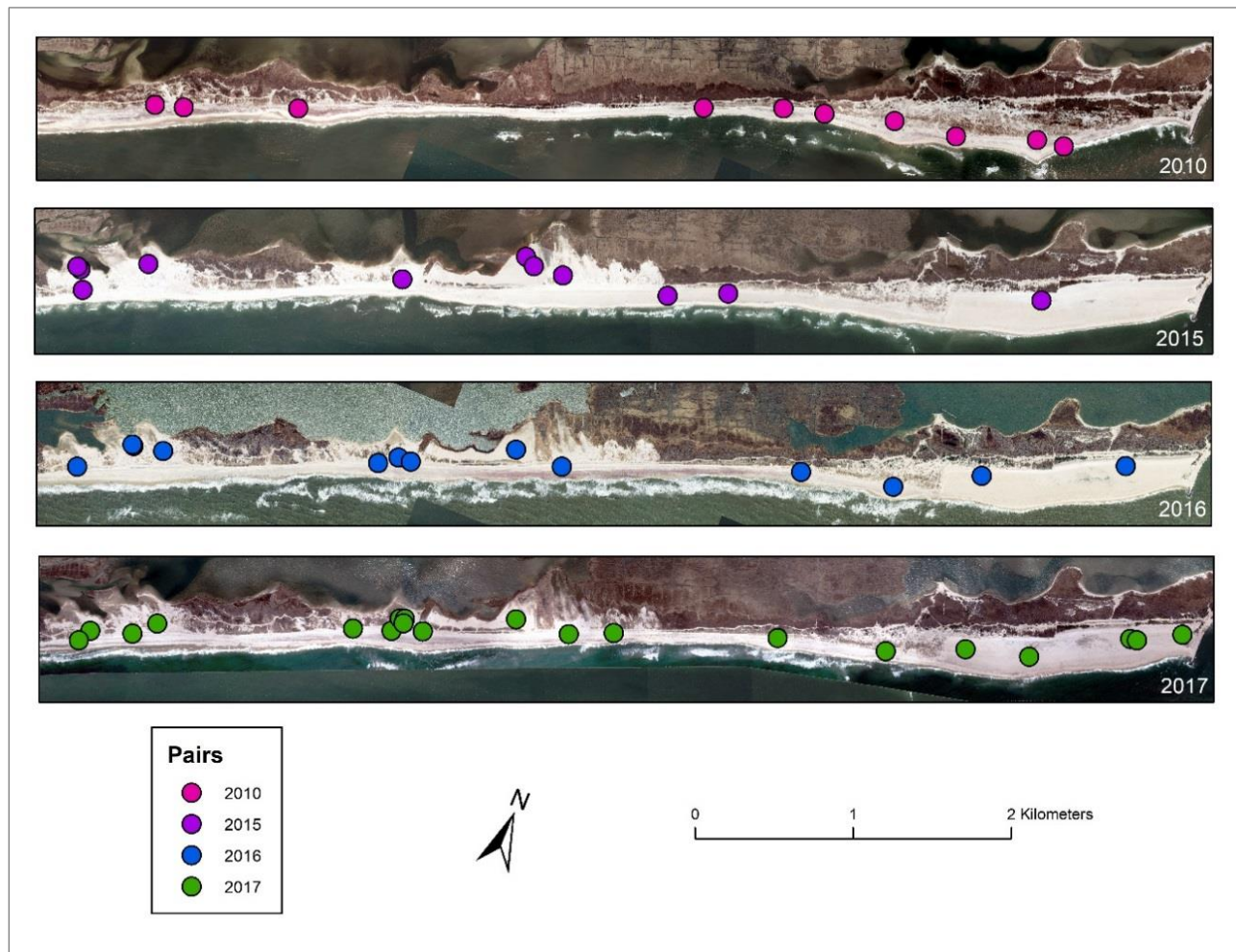


Figure 12. Smith Point County Park plover pair locations in 2010 and 2015–2017. Imagery is in accordance with year.

The amount of natural- and human-created habitat on Fire Island and Cupsogue Beach County Park demonstrates the importance of these areas as probable piping plover nesting locations in the future. Since 2012, the breeding population in this study area has increased 50%, from 30 to 45 pairs (Figure 32).

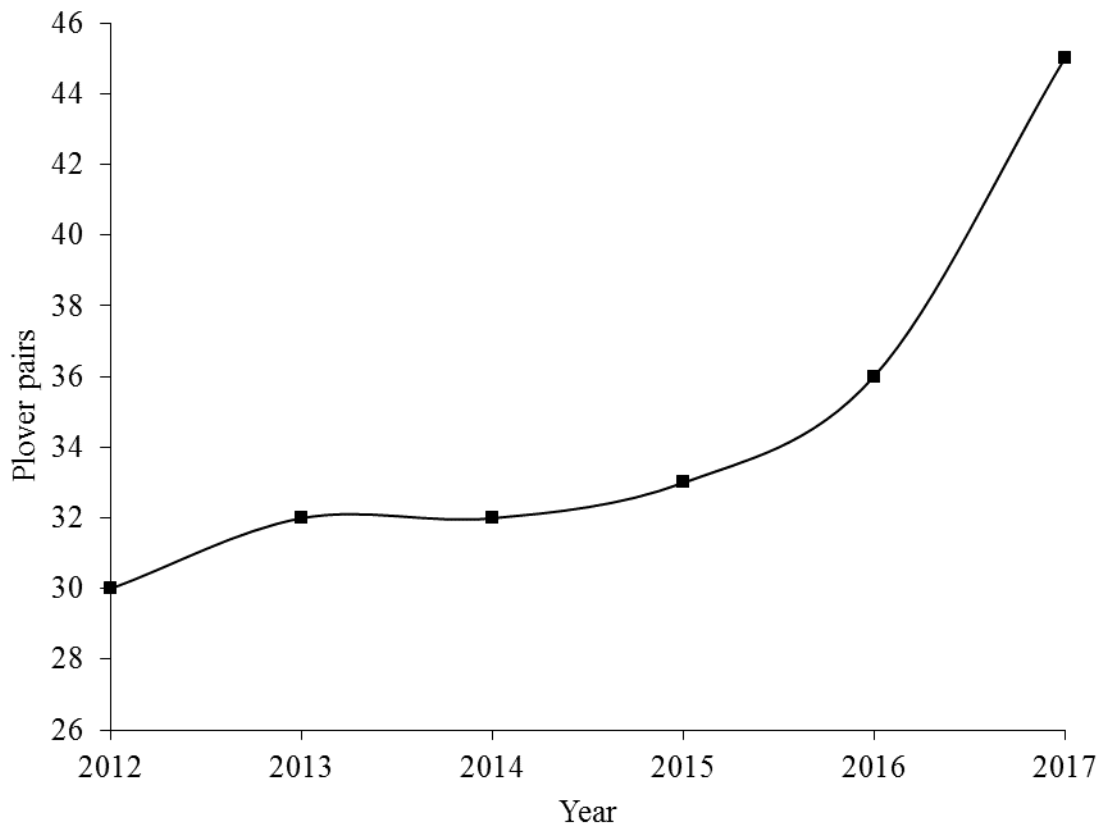


Figure 32. Breeding plover pairs on Fire Island and Cupsogue Beach County Park, New York immediately before and following Hurricane Sandy.

Hurricane Sandy and subsequent beach engineering added 124 hectares of suitable piping plover nesting habitat on Fire Island and Cupsogue Beach County Park, New York. Fifty-six percent of new suitable habitat was from natural overwashing storm processes. Despite 360 hectares of suitable piping plover habitat in 2015, only 162 hectares were protected from recreational use. The majority of storm-created and engineered habitats were well protected for nesting piping plovers. However, the majority of suitable habitat closer to the ocean high tide (beach nourishment and “other”), did not have substantial protection from driving and pedestrian use. Our analyses demonstrate that unprotected areas, which encompass 55% of suitable habitat within our study area, is avoided by piping plovers. Therefore, piping plover nesting relies on geomorphically suitable habitat that can also be protected from recreational use.

We should continue to view storm habitat creation as a critical component of piping plover conservation. Barrier island ecosystems like Fire Island are susceptible to constant changes in island morphology and therefore should be modeled often to account for temporal changes and subsequent behavioral responses. As conservation efforts continue, habitat selection should continue to be addressed, especially with regards to storm-induced habitat change.

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Part III: Piping plover use of restoration areas, managed overwashes and natural overwashes on Fire Island

Introduction

An increase in human coastal development, habitation, and recreation has co-occurred with coastal stabilization and habitat modification along the coasts of New Jersey and New York (Maslo et al. 2012, Rice et al. 2016). These habitat modifications can affect coastal ecosystems and may be a concern for species such as breeding shorebirds (Brown & McLachlan 2002, Defeo et al. 2009). Many species have life histories rooted in dynamic coastal systems, including the piping plover (*Charadrius melodus*) which has evolved to depend on such coastal processes (USFWS 2014).

Following Hurricane Sandy, the USACE proposed the Fire Island Inlet to Moriches Inlet Stabilization Project (FIMI), a coastal risk reduction project designed to reduce future impacts of large nor'easter storms and hurricanes. The FIMI project promises storm risk reduction for coastal communities on Fire Island and across the bay on the south shore of Long Island.

Under section 7 of the Endangered Species Act the FIMI was subject to a BO by the USFWS to assess the potential impact this project could have on the breeding piping plovers present in the area (USFWS 2014). Under this BO, habitat management commitments were made to mitigate the detrimental effects of disturbing critical piping plover habitat. The BO requires the USACE to create mitigation areas and manage for vegetation encroachment. In response, 45 hectares of habitat was created at Great Gun and New Made restoration areas between 2014 and 2015. Further details of the island-wide habitat alterations following Hurricane Sandy can be found in the General Introduction.

The USACE agreed to de-vegetate areas with >30% vegetation cover (USFWS 2014) by clearing vegetation of Great Gun restoration area (approximately 33.7 ha) and New Made (approximately 6 ha) to provide early successional habitat, nesting and foraging habitat through mechanical, manual and chemical treatments (USFWS 2014). Additionally, the USACE committed to bayside de-vegetation of the manipulated overwash (Pattersquash Island Overwash (approximately 13 ha), Smith Point Breach Overwash (6.1 ha) and New Made Island Overwash (10.5 ha)) when areas reached >30% vegetation cover.

Our objective was to quantify piping plover use in three habitat types resulting from Hurricane Sandy and the subsequent human modifications from 2015–2017 (post-Hurricane Sandy manipulated overwash, USACE restoration areas, and comparable Hurricane Sandy created natural overwash).

Study Area

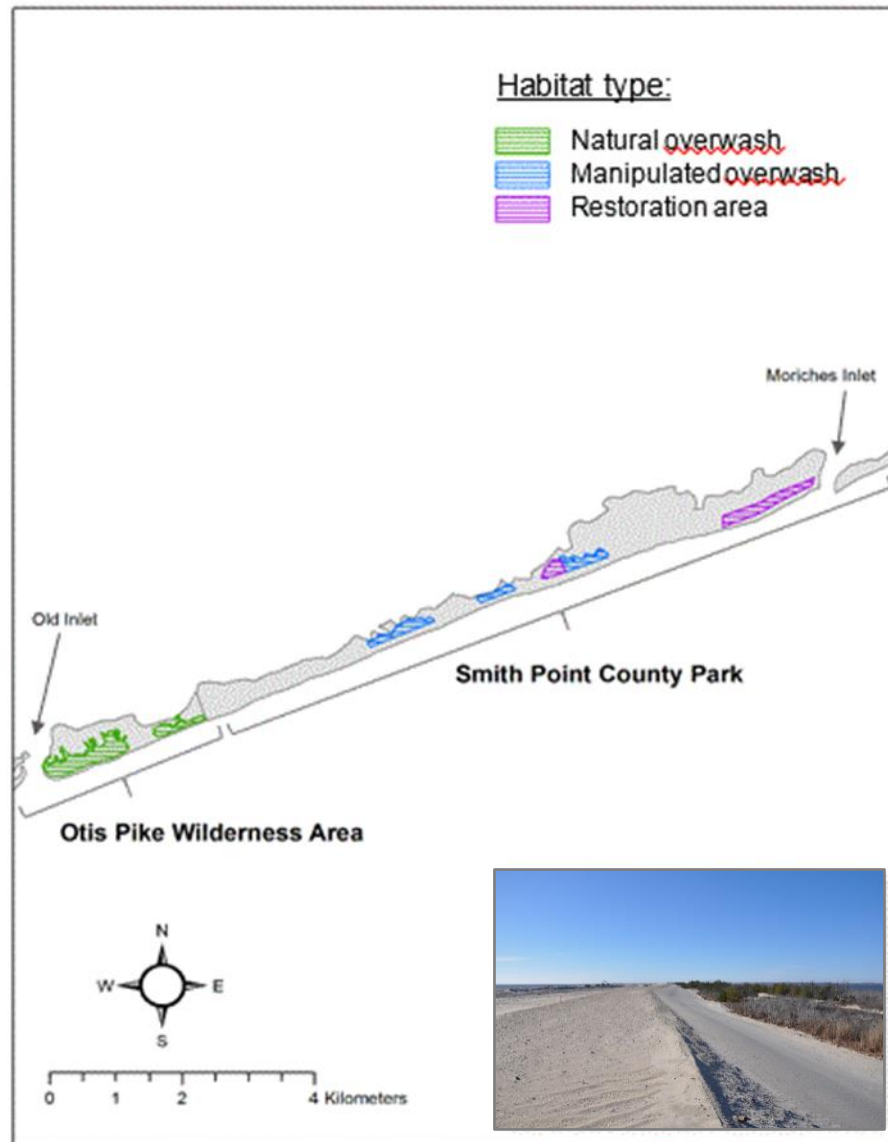


Figure 33. Study area on Fire Island, New York. Including two management units (Otis Pike Wilderness Area (National Park Service) and Smith Point County Park). The restoration areas are shown in purple (New Made is in the middle of the island and Great Gun restoration area is the eastern-most section near Moriches Inlet), the natural overwashes in green and the manipulated overwashes in blue. Inset: Picture taken from the Northwest corner of Great Gun restoration area following construction showing the expansive homogenous habitat.

Within our study area three major habitats were identified – natural overwashes, manipulated overwashes and restoration areas (Figure 41, Table 29). Natural overwash was defined as areas of Fire Island that overwashed during Hurricane Sandy, but have not been further modified (Table 1). Here we included the overwash to the east of Old Inlet (Old Inlet East (OIE)) and the smaller overwash further east (Bridge Overwash (BRO)). Manipulated overwash was defined as areas of Fire Island that overwashed during Hurricane Sandy, but have been manipulated by breach filling or stabilization, dune manipulation (including snow fencing and Christmas tree placement) or vegetation management (Table 1).

Table 29. Habitat definitions and annual area (ha) based on USACE polygons (USFWS 2014; USACE 2014), bound by annual spring tides. Further details of how areas were obtained, and vary from year to year, are described in the methods below. Management actions describe any management implemented since habitat creation.

Habitat type	Subsite(s)	Area (ha)			Management actions
		2015	2016	2017	
Manipulated overwash	Pattersquash overwash, Narrow Bay Breach Management Area, Narrow Bay East	26.99	24.84	28.13	- Annual snow fencing - Silt fencing in 2017
Natural overwash	Old Inlet East overwash, Bridge overwash	42.33	42.53	40.75	NONE
Restoration area	New Made (NMA), Great Gun (GG) restoration area	27.2	27.2	27.2	- NMA bayside mechanical vegetation removal. - GG minor physical removal of phragmites along north edge (black plastic sheeting 470m ² *)

*Calculated using aerial imagery to hand digitize the boundaries of the black plastic sheeting

Methods

Habitat

We used 4-band, high resolution (15cm) aerial imagery for each year (April 2015, February 2016 and April 2017) collected by Axis Geospatial (Axis Geospatial, Easton, MD). We hand digitized the habitat boundaries, as described by USACE (2014) and bound these sites by the appropriate annual spring high tide in ArcMap 10.4. Tide data were collected by walking the water's edge during the spring high tide (May) using a Trimble Geo7x handheld GPS unit (Trimble, Sunnyvale, CA).

The three habitat types were manipulated overwash, natural overwash and restoration area (Table 29). We calculated total area in hectares for each habitat type. In the manipulated overwash habitat, we excluded anything south of Burma Road (dune or oceanside habitat) as this fell outside of the USACE polygon specifications (USACE 2014). In the natural overwash habitat we excluded bayside marsh habitat and used the overwash boundaries collected in August 2013 following Hurricane Sandy. There was some variation in hectares from year to year because of changes in annual spring tides (bay and ocean) or erosion.

We used maximum likelihood classification in ArcMap 10.4 to identify sand only pixels from fall imagery (October 2015, September 2016, and September 2017) within the habitat boundaries for 2015 and 2016.

Bird banding and resighting surveys

We used mark, recapture and re-sight methods outlined in the Part 1-Field Methods above.

Daily surveys were conducted across the study site to identify banded and un-banded birds using 20–60x spotting scopes. Each time a bird was resighted we took the exact location of the bird using a Trimble GPS unit and Nikon laser rangefinder (Nikon, Tokyo, Japan). In addition, we recorded the habitat type and geological location for each resighting.

Nest monitoring

We monitored nests as outlined in the General Methods above. We considered a nest successful if ≥ 1 egg hatched and a brood successful if ≥ 1 chick survived to fledgling (where a chick was seen sustaining flight off the ground for 1 meter).

We recorded the distance from the nest bowl to the nearest edge of dense vegetation (where the vegetated area $\geq 5\text{m}^2$) in 2016 and 2017.

Pair densities

We calculated the pair density in each habitat (manipulated overwash, natural overwash and restoration area) by dividing the number of pairs by the given habitat area for the total area (pairs/ha), and the sand only pixels (pairs/ha sand). If a pair were present in more than one subsite, the subsite in which they were successful or spent the longest time was used.

Results

The number of piping plover pairs within the three habitat types were 14 in 2015, 14 in 2016, and 21 in 2017. These were a subset of Fire Island study area pairs mentioned in Sections I and II of this report (35 in 2015, 37 in 2016, and 45 in 2017).

The density of pairs per hectare was different among years and habitats (Table 32). In 2015, pair densities were highest in the manipulated overwash and the restoration area (0.22 pairs/ha for both; Table 32). In 2016, pair density was highest in the manipulated overwash (0.20 pairs/ha; Table 30). In 2017, pair density was highest in the natural overwash (0.27 pairs/ha; Table 30).

Since piping plovers primarily use open sand for nesting, we used sand only pixels to calculate the habitat area in 2015 and 2016. We observed an increasing trend in pair density with sand only pixels compared with total area due to reduced available habitat. Pair densities were higher in the manipulated overwash than the restoration area in 2015, 2016 and 2017 (0.38, 0.41 and 0.55 pairs/ha respectively; Table 30).

Table 30. The density of piping plover pairs in manipulated overwash, natural overwash and restoration areas for 2015, 2016 and 2017 for total habitat area and sand only habitat. Total area includes all pixels within the habitat boundaries, and sand only includes pixels classified as sand from our vegetation report (Ritter et al 2017).

Year	Habitat	Total area (ha)	Pairs/ha	Sand only (ha)	Pairs/ha sand
2015	Manipulated Overwash	26.99	0.22	15.7	0.38
	Natural Overwash	42.33	0.05	33.63	0.06
	Restoration Area	27.19	0.22	24.5	0.24
2016	Manipulated Overwash	25.63	0.20	12.1	0.41
	Natural Overwash	42.53	0.14	-	-
	Restoration Area	27.19	0.11	21.6	0.14
2017	Manipulated Overwash	28.92	0.14	7.3	0.55
	Natural Overwash	40.75	0.27	-	-
	Restoration Area	27.19	0.22	17.5	0.34

‘-’ Denotes data unavailable at this time as yet to complete classification of the appropriate imagery.

Nests

Across habitat

There was an overall increasing trend in nest numbers across the three habitat types with 17, 16, and 24 nests in 2015, 2016 and 2017, respectively (Figure 34). These were a subset of Fire Island study area nest numbers mentioned in Sections I and II of this report (48 in 2015, 42 in 2016, and 62 in 2017). For 2016 and 2017, the mean distance from the nest bowl to dense vegetation was 37.26m (SD± 32.08m). The mean distance to dense

vegetation for all nests presented in Sections I and II was 50.17m (SD± 58.14m, SE ± 5.93m, Range: 2.28m - 288.40m, 1st Qu: 19.23m, Median: 29.67m, 3rd Qu: 54.34m).

The observed increasing trend in nest numbers was particularly driven by nesting in natural overwash (Figure 34). In 2017, nest numbers in natural overwash were more than twice that of the other two habitats.

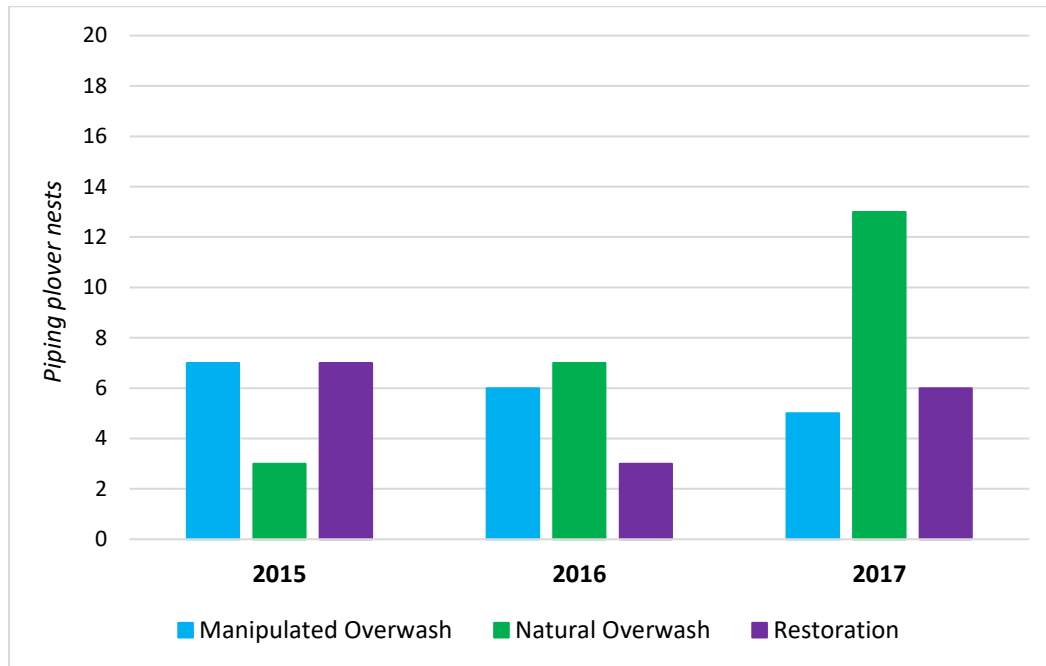


Figure 34. Piping plover nests in 2015, 2016 and 2017 in the three habitat types – manipulated overwash (blue), natural overwash (green) and restoration area (purple).

Restoration areas nests

The number of nests in the two restoration areas ranged from 1 to 5 following habitat construction (Figure 35). The mean number of nests in the restoration areas across three years was 2.7 nests (SD± 1.9). The number of nests increased to 5 at Great Gun restoration area while the converse was observed at New Made (Figure 43). For the last 2 years, only one nest per year was found at the New Made site.

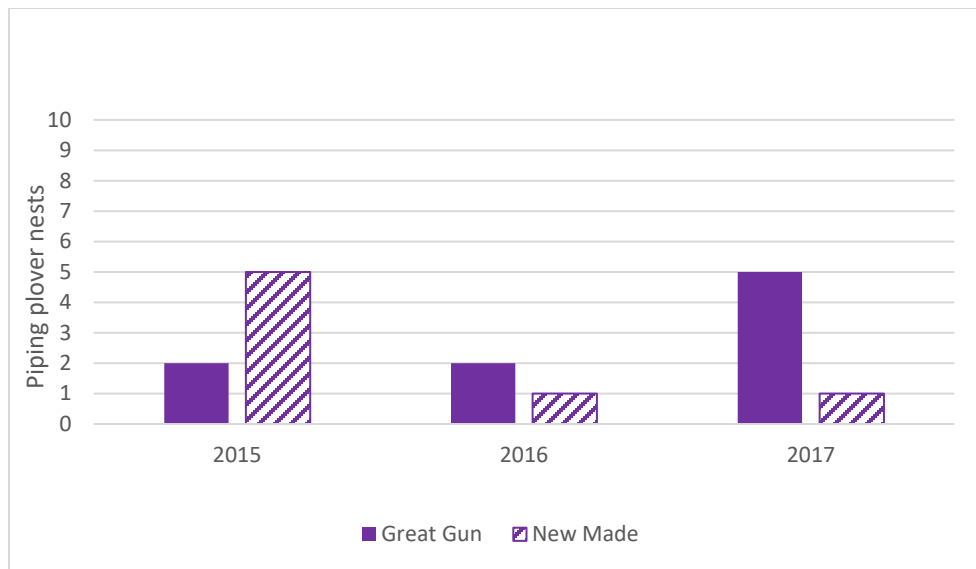


Figure 35. The number of piping plover nests found in each of the two sites that make up the restoration area habitat. All nest attempts were included (≥ 1).

Piping plover success across habitat types

Nest success varied by year and habitat type, ranging from 0% to 100% (Table 33). In 2015, 0 of the 7 nest attempts were successful in the restoration area. However, this increased to 100% nest success in 2016. The highest nest success was in natural overwash in 2015 (33%), and in the restoration area in 2016 and 2017 (100%).

The proportion of successful broods varied across habitats within the three years (Table 33). Brood success ranged from 0% to 100% (Table 33). In 2015, the highest proportion of successful broods (33%) was observed in the natural overwash habitat. In 2016 all three habitat types had >65% of successful broods (restoration area was greatest at 100%). In 2017, 100% of broods in the restoration areas were successful (Table 33).

Table 31. The number of annual piping plover nests within the habitat types, the number, and proportion, of successful piping plover nests, and the number, and proportion, of successful broods across the three habitat types in 2015, 2016 and 2017. Any nest and reneest within the habitat was included (≥ 1 egg). A nest was considered successful if at least one egg hatched, and a successful brood was considered successful if one chick per brood survived to fledge.

Year	Habitat	Nest Attempts	Successful Nests	Successful Nests (%)	Successful Broods	Successful Broods (%)
2015	Manipulated Overwash	7	1	14	1	14
	Natural Overwash	3	1	33	1	33
	Restoration Area	7	0	0	0	0
2016	Manipulated Overwash	6	4	67	4	67
	Natural Overwash	7	6	86	5	71
	Restoration Area	3	3	100	3	100
2017	Manipulated Overwash	5	3	60	2	40
	Natural Overwash	13	10	77	7	54
	Restoration Area	6	6	100	6	100

Return rates

The rate of returning male piping plovers, to the habitat they nested in the previous year, varied with habitat and year. In 2015, the manipulated and natural overwash was 50%. In 2016 and 2017 the male fidelity was highest in the manipulated overwash (60% and 50% respectively; Table 34). In females, site fidelity was lower or equal than males across all habitats, except in the natural overwash in 2017 where it was 12% greater. Female site fidelity was highest in the natural overwash in 2015 (50%), and manipulated overwash in 2016 (40%) and 2017 (50%) (Table 34).

Table 32. The number of returning individuals across habitat types for 2015, 2016 and 2017 by sex. Fidelity was defined as a returning adult who nested in the same habitat as the previous year, not necessarily the same specific subsite. Given the restoration areas were created in 2014–2015, there was no 2014 pair data to calculate 2015 site fidelity in the restoration areas for 2015.

Year	Habitat	Male Fidelity (%)	Female Fidelity (%)
2015	Manipulated Overwash	50	33
	Natural Overwash	50	50
	Restoration Areas	-	-
2016	Manipulated Overwash	60	40
	Natural Overwash	33	33
	Restoration Areas	0	0
2017	Manipulated Overwash	50	50
	Natural Overwash	18	30
	Restoration Areas	33	0

Discussion

We observed increasing pair densities and nest attempts in natural overwash, and an increase in pair densities and constant nest attempts in manipulated overwash following creation. Pair density and nest numbers in the restoration areas, however, were fairly constant despite increasing male site fidelity and nest success.

The USACE built two restoration sites on Fire Island that offered new sandy habitat. Engineered habitats may be biologically effective, helping to reach local and region recovery goals if they meet the biological requirements of the target species (Catlin et al. 2011, Maslo et al. 2012). Previous studies show piping plovers show preference for engineered habitats where they have good nest success (Catlin et al. 2011). Both restoration areas have experienced high chick survival over two of the last three years, and pair and nest numbers have increased at Great Gun restoration area. However, at New Made the number of pairs declined from 2015, only supporting one pair per year in 2016 and 2017. It is possible that vegetation encroachment degraded habitat at New Made, decreasing its suitability. New Made is bound by stands of Common reed (*Phragmites australis*) with poor shoreline foraging access, forcing piping plovers that nested there to take their chicks elsewhere. The surrounding vegetation stands may have acted as a barrier, particularly for flightless chicks, making foraging trips to other habitats challenging. For example, chicks from the New Made nests were regularly seen foraging in other locations outside of the New Made habitat boundaries.

Access to foraging habitat affects pair densities (Cohen et al. 2009). Cohen et al (2009) estimated pair densities of 0.5 pair/ha in one-sided, ocean foraging habitat but 1 pair/ha in two-sided, ocean and bay, habitats. Densities in the manipulated overwash, a two sided habitat, have shown an increase over the last three years (0.55/sand ha in 2017), suggesting these areas are not yet full. Additionally, pairs/ha sand for the restoration areas, one-sided habitat, are well below 0.5 pairs/ha (0.34/sand ha in 2017) suggesting these areas are also

not at their full carrying capacity. However, these habitats are susceptible to vegetation encroachment, reducing the available nesting and foraging areas which may impact the pair densities.

Habitat management is key to providing longevity to the piping plover nesting habitats (Catlin et al 2011). An increase in vegetation cover may lead to a less desirable nesting habitat. All three habitat types have experienced vegetation encroachment from American beachgrass (*Ammophila breviligulata*) and Common reed. These plants are well adapted to respond to sand burial with evidence showing this process can simulate growth (Disraeli 1984). The natural overwash falls on National Park Service property where habitat management is not conducted. The natural overwash has had high nest and brood success, however, without habitat management, it will degrade as re-vegetation continues. The manipulated overwash and restoration habitats are on Suffolk County Park property where some vegetation removal has occurred, but vegetation is >30% in all areas except Great Gun restoration area (Ritter et al. 2017).

Overall, male site fidelity was higher than female fidelity, except in the natural overwash in 2017. This is similar to piping plover site fidelity on the Missouri River (Friedrich et al. 2014). Male site fidelity in the manipulated overwash habitat has been highest across the three habitats for the last two years. In the restoration areas, no females ever returned to nest. While in 2017, we recorded the return of two males who had nested in this habitat the previous year. Haas (1998) suggested that higher nest reproductive success leads to a higher probability of site fidelity, so we might expect the fidelity of the restoration habitat to be higher given that the brood success has been high for the last two breeding seasons.

Continued monitoring of the restoration areas, and other manipulated habitats are crucial in understanding plover habitat use on Fire Island, particularly as these habitats re-vegetate and degrade, or are maintained. Overall, restoration efforts have been successful in attracting nesting piping plovers, and more recently have produced successful broods. However, regular mechanical or chemical habitat management will be needed to maintain these habitats. Further analysis of piping plover habitat use and selection in relation to habitat availability is ongoing.

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Part IV: Red Fox Movements, Habitat Selection, and Demography on Fire Island, New York

Introduction

Many studies have shown that predation is a key determinant of piping plover reproductive output, and predator management often is used to improve piping plover breeding success. The red fox (*Vulpes vulpes*) is an important predator of piping plovers and other beach nesting birds on the U.S. Atlantic Coast. On Long Island, Cohen et al. (2009), showed that plover reproductive output in Westhampton Dunes was improved when foxes and cats were removed. Our ongoing piping plover monitoring work at Robert Moses State Park, Fire Island National Seashore and Smith Point County Park on Long Island (see Part I of report) showed that foxes were common in all areas in some years and an important predator of piping plovers on Fire Island, with impacts varying across years. Predator management is an important part of piping plover management along the entire U.S. coast, and management of red foxes at these sites on Long Island may be important for piping plover populations (USFWS 1996, 2014). Nevertheless, there still are many gaps in our knowledge of the foxes, fox responses to management, and the effects of foxes and fox management on piping plover habitat use and reproductive output.

As part of monitoring in connection with the Fire Island Inlet to Moriches Inlet, Fire Island Stabilization Project (FIMI), we are collecting data needed to best design and implement red fox monitoring and management activities.

- Objective 1: Quantify the abundance and occupancy of red foxes and relate changes in these parameters to changes in habitat and predator management activities. (*Methods: Camera-trapping, den surveys, habitat assessment, collaboration with predator management partners*)
- Objective 2: Assess habitat use versus availability by red foxes and any variation in habitat selection related to changing habitat composition and predator management. (*Methods: GPS collars, habitat assessment, collaborations with predator management partners*)
- Objective 3: Quantify home range size, long-range movements, and variation in movement ecology of red foxes and relate to changing habitat composition and predator management. (*Methods: GPS collars, habitat assessment, collaboration with predator management partners*)
- Objective 4: Estimate red fox reproductive success and adult survival, and variation in these population parameters, related to habitat composition and predator management. (*Methods: GPS collars, den surveys, habitat assessment, collaboration with predator management partners*)
- Objective 5: Quantify the interactions of red foxes with shorebird nests and chicks. (*Methods: GPS collars, camera traps on shorebird nests, scat collection and diet analysis, collaboration with shorebird management partners*)
- Objective 6: Utilize information collected in objectives 1–5 to make recommendations on the utility of, and recommended methodologies for, management of red fox populations in the study area and in other coastal habitats.

Methods

Study site

We focused red fox monitoring efforts from Fire Island Inlet to Moriches Inlet on Fire Island, within the administrative boundaries of the National Park Service, the County of Suffolk (Smith Point County Park), and the State of New York (Robert Moses State Park). When conducting red fox monitoring activities during the piping plover nesting and brood-rearing seasons, the red fox monitoring team recorded all activities as they pertained to the sub-sites as described in the “Study Site” section of the introductory information to the overall report. However, the red fox monitoring efforts were conducted across the entirety of Fire Island as GPS-collared foxes used the entire area in 2015 and 2017, with the exception of a temporary absence east of Old Inlet between mid-April 2016 and March 2017 due to a mange outbreak (described below in more detail).

Sign, scat, and den surveys

Field methods

We used ArcMap 10.3 GIS software to create and overlay a grid onto each of the 3 study sub-sites (Robert Moses State Park, Fire Island National Seashore, and Smith Point County Park), subdividing them into 300 m by 300 m cells, and identified the center point of each cell. The goal of this subdivision was to focus survey efforts (for sign, scat, dens, and remote camera surveys) such that each potential core range of a fox was sampled with equal probability. Previous work with red foxes at Assateague Island National Seashore (Gieder 2015) showed that foxes rarely moved between camera stations spaced 300 m from each other. GPS collaring data from foxes monitored in each of the 3 study subsites during 2015–2017 (see GPS tracking results) suggests that this spacing is more than sufficient to cover all potential home ranges, as the smallest observed red fox home range in each area during each year had a radius >300m. Thus, there should be no gaps in potential detection based on the overlap of our grid size and the observed movements from Gieder (2015) and our own collars.

We walked 96 transects (24 in Robert Moses State Park, 40 in Fire Island National Seashore; and 32 in Smith Point County Park, Figure 36) from the ocean beach high tide line to the bayside high tide line approximately once per month during the red fox denning and piping plover breeding season (May, June, July) and again outside of the breeding seasons in November, December, and January (2015–2017). Transects connected the center points across the island of the 300 m grid described above. Specifically, we walked game trails, or in as close to a straight line as possible if no game trails present, between the grid cell center points from ocean beach to bayside as possible on each transect. All potential red fox scats located on transects identified as ≤ 1 month old were collected and labeled by estimated freshness (Table 33) and a unique sample number corresponding to the GPS waypoint recorded. Scats were frozen for later genetic confirmation of the species that deposited the scat and analyses of prey items. All observed red foxes were recorded and GPS locations noted. All inactive and active red fox den sites were also recorded and GPS locations noted. Dens with fresh tracks leading in and out of entrance holes and/or foxes nearby were considered “active.” In 2017, we conducted additional transect surveys in late April/early May to assist in den searching. During these surveys, multiple observers worked together to search all accessible areas for potential dens, in addition to the game trails and transect areas typically surveyed during other months.

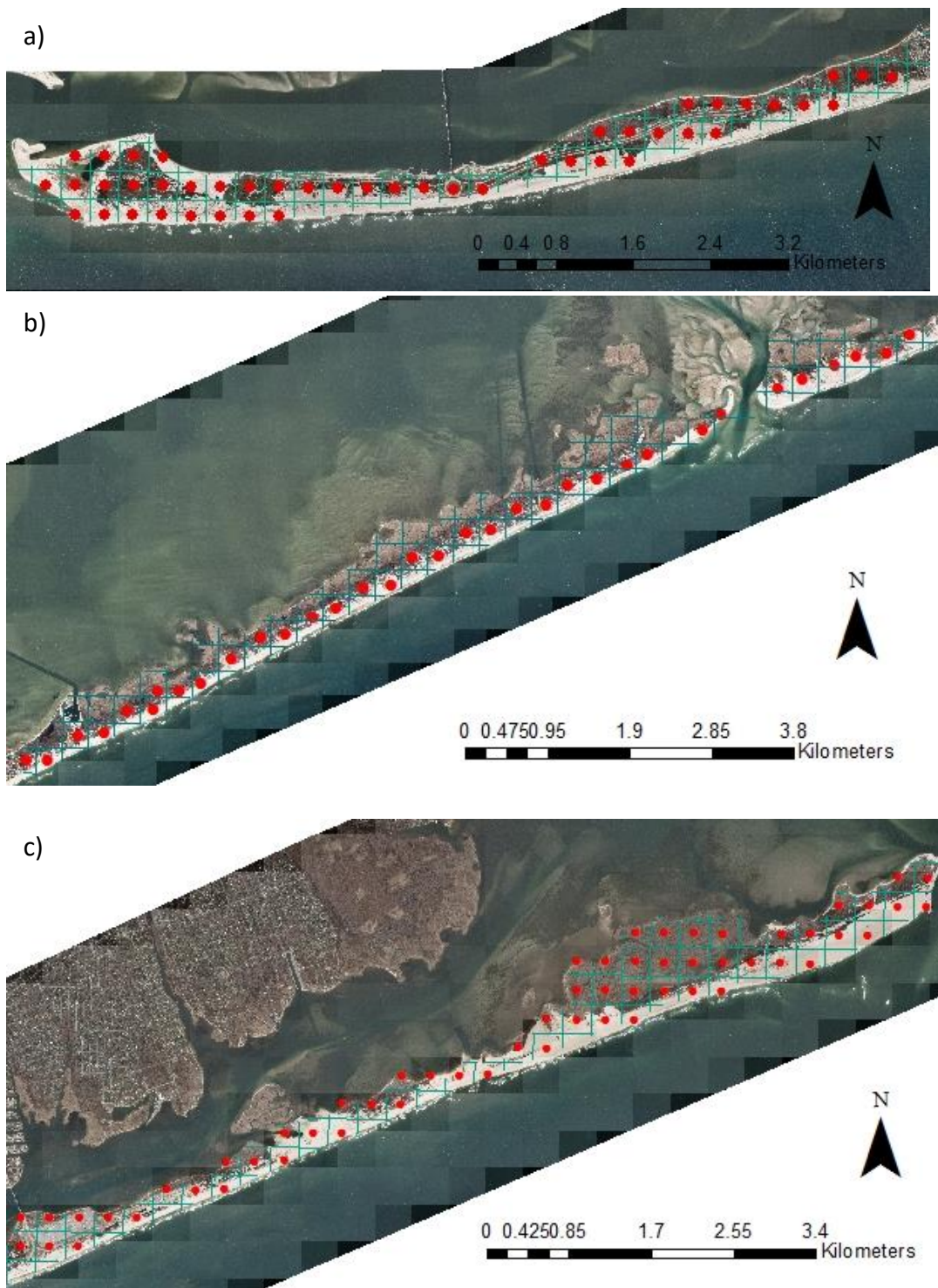


Figure 36. a) Transect locations in Robert Moses State Park and Fire Island National Seashore Lighthouse Tract. b) Transect locations in Fire Island National Seashore. c) Transect locations in Smith Point County Park. Monitors walked as close to straight-line transects as possible while following game trails and wildlife travelways between grid cell center points, shown here as red dots, from ocean beach high tide line to bayside high tide line to identify fox sign, collect fox scat, and locate fox dens in 2016–2017; April–July and October–January.

Lab methods

All scat samples were stored in a freezer, then transported on ice back to Virginia Tech's campus and stored in a -20 C walk-in freezer until analysis. For each sample, we separated a small portion (approximately 2mL in volume) of fecal material from the outside edges and tapered ends, and submerged this portion in 95% ethanol for DNA preservation. These preserved subsamples will be sent to the Mammalian Ecology and Conservation Unit (MECU) in the Veterinary Genetics Lab at UC Davis for mitochondrial DNA sequencing to verify the source species. Morin et al. (2016) has shown that misidentification of predator species in the field can significantly bias fecal dietary analysis results. To minimize potential bias, the authors recommend genetic verification, particularly in areas where multiple predator species may deposit morphologically similar scats. Feral cats, domestic dogs, and raccoons are also present on Fire Island, and even experienced field biologists can have difficulty distinguishing among scats from these species.

For a subset of these samples shown to contain high quality red fox DNA, we plan to collaborate with the MECU to conduct further genetic analyses to determine the phylogenetic origins of red foxes on Fire Island. This will be done using red fox-specific primers and assays, and genotyping protocols previously developed by the MECU. We will compare the resulting haplotypes and nuclear DNA information from samples collected on Fire Island to reference samples of known origins, and determine whether red foxes on the island are likely native to the area, the result of natural range expansions from further north, or the descendants of nonnative foxes brought over from Europe for hunting. Results from a previous study of red fox mitochondrial DNA in various parts of the United States suggest that samples from the southeastern US were most similar to reference samples from native populations in the northeastern US (Statham et al. 2012). However, a more recent study based both mitochondrial and Y-chromosome DNA found that many samples collected in New York, Pennsylvania, and along the mid-Atlantic coastal plain aligned more closely with non-native reference samples (Kasproicz et al. 2016). Neither of these studies included any samples from Long Island, leaving both nonnative translocation and native range expansion as possible explanations for the origins of red foxes on Long Island and its barrier islands.

We will use the remaining portion of each scat sample for fecal dietary analysis in our lab on Virginia Tech's campus. After drying, samples are dissected and contents sorted for identification to the lowest taxonomic level possible (Figure 45) through comparison to reference materials and collections (Reynolds and Aebischer 1991). We then calculate percent occurrence and frequency of occurrence for each diet item or group, to assess reliance on different food sources, and compare these results among study sites and among samples collected during months when piping plovers are present versus absent



Figure 13. After drying, scat samples are dissected and contents sorted for identification to the lowest taxonomic level possible. Photo: Earle Berge.

Table 33. Scat collection criteria used during sign, scat and den surveys. “Fecal material” refers to the unidentifiable digested waste material contained in a sample, as opposed to the identifiable prey item remains.

Age Category	Color	Moisture	Biodegradation	Notes	Collect?
Very fresh	Varied	Very moist	None	Observed a fox depositing the sample and/or signs indicate <24 hrs. old	Yes
Fresh	Usually dark	Somewhat moist	None		Yes
Mid	None or minimal, no sun bleaching	May be moist due to recent dew/humidity	None, outer fecal material intact		Yes
Old	May be sun bleached	Usually dry	Some, outer fecal material beginning to wear away but still present		Only if outer fecal material is mostly intact
Very Old	Sun bleached	Very dry	Significant, hardly any outer fecal material left		No

Den monitoring

Field methods

We monitored all active dens located during den transects, through GPS tracking of breeding foxes, or from opportunistic locations from the time of discovery until at least 2 weeks after the last signs of active den use. All discovered dens were revisited within 1 week of discovery and re-assessed to determine the likelihood of current use based on tracks and sign; dens that still appeared active during this secondary assessment were added to our ongoing den monitoring. We visited active dens once every 2 weeks and collected scats, recorded and photographed prey remains in the immediate area, and recorded other signs of active use, including tracks, new prey remains since the last visit, and foxes observed in the area.

Where surrounding habitat provided sufficient cover and human use was low enough to minimize the likelihood of drawing undue attentions to dens, we also set up 1 or more remote cameras (Moultrie M880i, Pradco Incorporated) aimed at the den entrance and/or small clearings directly outside of the den. We changed memory cards and batteries and adjusted camera angling during den monitoring visits.

Analyses

We used a combination of remote camera photos, direct observations, and GPS collaring data from breeding foxes to determine the number of adults and kits associated with each den and, when possible, to identify likely family groups and track den relocations during over time. For dens that had no collared adults attending, we determined likely family groups and den relocations based on the proximity and dates of use of newly active dens relative to previously active dens.

Trapping, ear-tagging, and collaring

We attempted to capture up to 6 foxes in each management unit to attach GPS collars (Telemetry Solutions, Quantum 4000E, Figure 46), and ear-tags during August–December 2015, April–May 2016, and February–mid-April 2017. During these trapping periods, we also attempted to recapture foxes from previous years whose GPS collars failed to drop off as programmed.

We used capture techniques as recommended by the Association of Fish and Wildlife Agencies Best Management Practices for Trapping Red Foxes in the United States (Updated 2014). We used Victor 1 1/2 Soft Catch padded foothold traps and Bridger #2 Offset coil spring foothold traps, set to 2-lb trigger pressure to avoid capturing lighter animals (Figure 47). We discontinued the use of Tomahawk 2-door fox-size box traps attempted in 2015, because no foxes were successfully captured in these in 2015, and there was substantial bycatch of raccoons, which were all released in good condition.



Figure 14. Red fox at Robert Moses State Park outfitted with a Quantum 4000E GPS collar and numeric ear-tag.



Figure 15. Setting a padded foothold trap.
Photo: Matt Kynoch.

Upon capture, we manually restrained foxes using a catch-pole and attached a muzzle, leg-ties and a blindfold; no sedatives were used (Figure 48). We attached the Quantum 4000E collar, placed an ear-tag in the right ear, and conducted morphometric measurements. We used serially-numbered white ear-tags at Robert Moses State Park, serially-numbered blue ear-tags at Smith Point County Park, and small (Approx. size: 1/4" x 5/8") metal ear-tags at Fire Island National Seashore (Figure 49). These metal ear-tags are not likely to be visible in our remote camera photos, but will assist in identifying any individuals recaptured in future years.



Figure 16. Captured fox being processed. We manually restrained foxes and took morphometric measurements before releasing them. Photo: Caitlin Kupferman.

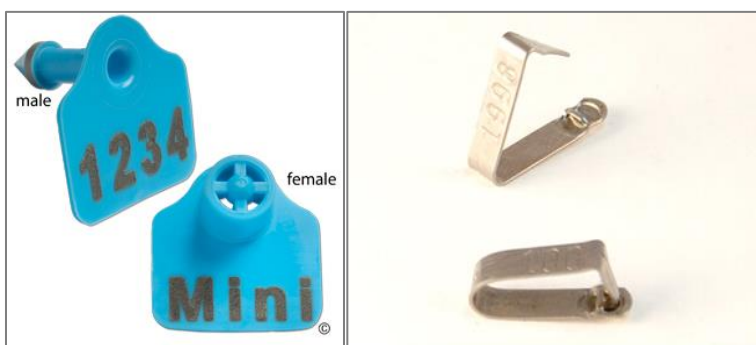


Figure 17. Numeric ear-tags used to mark captured foxes at Robert Moses State Park and Smith Point County Park (left) and small metal ear-tags used to mark captured foxes at Fire Island National Seashore (right) Photos: Premier 1, National Band and Tag Company.

GPS tracking

Field methods

In 2016 and 2017, we programmed the Quantum 4000E collars to record GPS locations once per hour, every day from April 20th through the end of May, to allow us to determine which individuals were using each plover monitoring subsite, and then once every 2 hours, on every other day until programmed collar drop-off in early August (originally programmed for August 10th). In 2015, we programmed the collars to record GPS locations once per hour, every day for 30 days from their deployment date, and then once per hour, every other day from day 31 through programmed collar drop-off (end dates varied depending on deployment date). We used Collar SW v 2.09 software (Telemetry Solutions) to remotely download data from each collar every 10–14 days after deployment and to reprogram collars as needed. To do this, we relocated the collared fox via radio telemetry tracking of the VHF signal on the collar, approached to between 10–100m, connected wirelessly to the collar using the Quantum Base Station, and downloaded the data and/or reprogrammed the collar.

In 2016 and 2017, as soon as possible after the discovery of piping plover nests in areas being used by collared foxes, we remotely reprogrammed collars to temporarily increase the frequency of location recording to collect fine scale data on fox spatial response to plover nesting activities. Due to battery life limitations, we could only sustain this high frequency scheduling for 8 days per collar. We chose to focus these days around plover nest establishment, predator exclosure setup (where applicable), and just prior to hatching. Through close collaboration with piping plover field crews and park managers, we identified expected exclosure setup dates and expected hatch dates and reprogrammed collars to record locations every 15 minutes for 2 days before and after planned exclosure setup, and 2 days before and after egg pipping was estimated to occur. In situations where multiple nests were laid within a given fox's regular use area, we based our programming on the dates corresponding to the first laid nest. Similarly, for nests placed in areas used by multiple foxes, we paired nests with foxes based on the proximity of fox locations to nest locations.

For various logistical reasons and biological reasons, the estimated dates of these events were not always the same as the actual dates on which they occurred, particularly surrounding pipping and hatch dates that had been estimated several weeks in advance. Therefore, our high frequency data collection at some nests did not line up exactly with the events that we had scheduled them around. For our analyses, we used the highest frequency data available surrounding the actual dates, using 2–2.5 days of location data before and after the target event to examine fine-scale spatial response.

Analyses

We mapped GPS locations for each collared red fox in ArcMap for exploratory analyses, including the determination of fox proximity to nesting areas. We overlaid these points onto the most recent 6-inch resolution orthoimagery.

We then used the “Minimum Bounding Geometry (MBG)” tool in ArcMap (in 2016) and R package “adehabitat” (in 2017) to create minimum convex polygons (MCP) using each individual's locations, for a very basic estimate of home range and area used, then used the “Calculate Geometry” tool in ArcMap to estimate

polygon area in hectares. These polygons include the minimum area encompassing all locations recorded for each fox that can be contained in a single convex polygon. Due to the nature of this polygon estimation, MCPs can include large areas that may not have actually been used by foxes, but are between two or more locations that were used. We caution against interpreting these polygons as regularly used home ranges; instead, we suggest that they be considered as a metric of how far an individual fox might range over the course of the tracking period.

To obtain more refined home range estimates, we used R package “adehabitatHR” to calculate 50% and 95% kernel density home ranges for each individual, then imported these contours into ArcMap to overlay them on orthophotograph imagery for comparison to recorded locations. We then used the “Calculate Geometry” tool in ArcMap to estimate the area of each contour in hectares; 50% kernel densities are commonly interpreted as an animal’s “core area,” or an area of intense use, and 95% kernel densities are commonly interpreted as an individual’s larger “home range”, which encompasses this core area but is not used as intensively (Laver and Kelly 2008).

Due to the underlying mathematics used to calculate MCPs and kernel densities, both of these descriptors of red fox space use can incorporate areas that are most likely not used (e.g., deep water) into a polygon or home range contour, if these areas are adjacent to areas of regular use. To avoid including large amounts of open ocean or bay in our maps and area estimates, we clipped all polygons and contours to the extent of the island using the “Clip” tool in ArcMap. Similarly, we clipped the kernel density contours of animals on either side of Old Inlet to the extent of the portion of the island that they were located on, as we did not observe any fox crossing Old Inlet.

We currently are exploring a dynamic Brownian Bridge approach to home range estimation using R package ‘move’. This approach models the likelihood of use at a given point based on recorded locations, similar to the kernel density approach, but accounts for variable time intervals between points and incorporates the chronological order of points and the path taken in between points, rather than assuming spatial and temporal independence among all locations. This approach also models likelihood of use as a non-normal probability distribution (as opposed to kernel density estimators, which assume a normal distribution), and is well suited to variable location sampling schedules (Kranstauber et al. 2012). We believe that this approach offers a more realistic approximation of red fox space use by excluding unused areas included with MCP and kernel density estimators. We aim to transition to using this home range estimation approach in future reports, and will re-calculate home range sizes for animals tracked during previous seasons using this approach to present alongside future results.

Remote camera surveys

Field methods

We used the same 300 m grid cells and center points created in ArcMap to guide camera placement in each of the 3 study sites, as described above for the transect surveys for sign, scat, and dens. We established an un-baited camera station within 50 m of the center point of each grid cell containing suitable sites. We excluded hard surface developed areas and inundated marsh from survey grids. Each station consisted of 2 different models of Moultrie game cameras (Pradco Incorporated, Moultrie M880c and M880 Gen2 cameras), each mounted on a plastic t-post and facing towards a game trail, clearing, or other potential wildlife trail (Figure 50). We set each camera on opposite sides of the game trail photograph both sides of passing animals (Figure 51), and angled them to be offset from each other.



Figure 18. Cameras were mounted on plastic t-posts along game trails and wildlife travelways.



Figure 43. We set one camera on each side of game trails and wildlife travelways to photograph both sides of passing animals.

We programmed the cameras to take 3 pictures per trigger and 5 seconds delay after each trigger. We returned every 10–14 days to switch memory cards, replace batteries as needed, and check each station. We set up each station in the same location as the previous year’s station, or as close to it as possible. The exact locations and numbers of camera stations in each grid varied slightly among years due to habitat changes. We established 30–35 stations in each of the 3 main study areas, starting with Smith Point County Park and Fire Island National Seashore (surveyed late September–November of each year) and then surveying Robert Moses State Park (late November to early January of each year) (Figure 44). Each grid was or will be operational for a minimum of 1200 trap nights, defined as the number of camera stations multiplied by the number of nights the cameras are in operation.

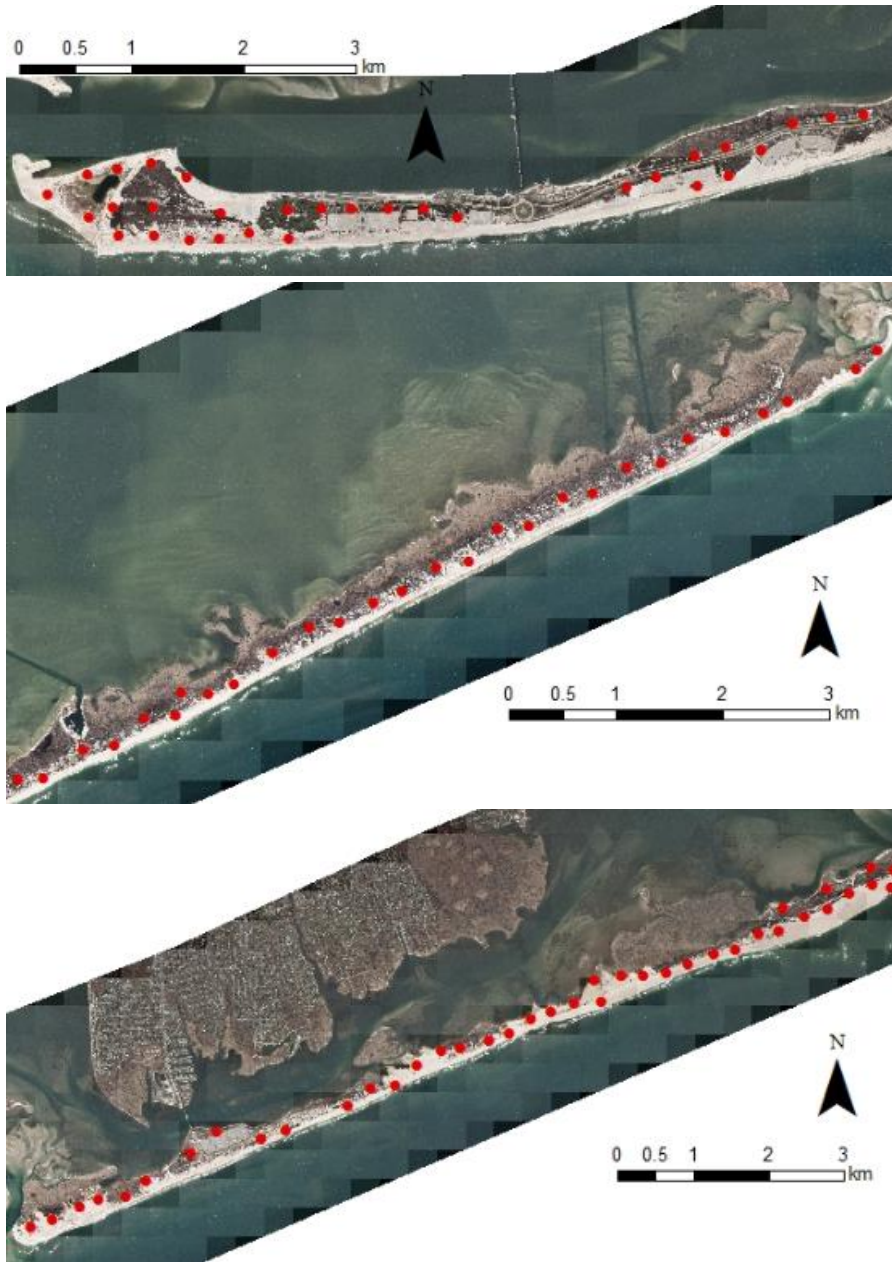


Figure 44. Approximate camera station locations at Robert Moses State Park (top), Fire Island National Seashore between Watch Hill and Old Inlet (middle), and Fire Island National Seashore east of Old Inlet and Smith Point County Park (bottom). Exact location and number of stations varied slightly among years.

Baited camera stations

In examining photos taken during our remote camera surveys in the fall 2015, we had some difficulty reading the ear-tag numbers for many of the ear-tagged foxes photographed. To determine whether using bait at stations might improve tag readability in photos, we set up approximately 25 temporary bait stations at Robert Moses State Park in January 2017, following the completion of our normal, unbaited camera grid surveys. We set one bait station in each 300m north-south section of the island in locations likely to be used regularly by foxes based on sign and previous GPS collaring data. Results from our previous home range analyses suggested that this spacing would allow for at least 2 bait stations in each potential fox home range. We baited each station with tinned sardines attached to a wooden landscaping stake, and located bait stations >50m away from any location used for our unbaited camera grid surveys, to minimize the risk of biasing detection in future unbaited surveys. We aimed 1–3 remote cameras at each baited stake and used the same t-post mounting methods described above for our remote camera grids. We refreshed the bait after approximately 1 week, and took down the bait stations and camera after approximately 2 weeks total survey time. These temporary baited camera stations were tested only at Robert Moses State Park, as no foxes were detected at Smith Point County Park during this time, and the ear-tags used at Fire Island National Seashore are quite small and not meant to be detectable or readable in photos.

Analyses

In this report, we present results and analyses from remote camera data collection in 2015–2016 and in 2016–2017. We are in the process of further examining photos from regular grids and baited camera stations to determine whether individual red foxes can be identified in photos using ear-tags, GPS collars, and variation in pelage in order to estimate abundance and density in each management area. We also plan to explore the use of abundance modeling approaches for animals that cannot be individually identified, such as mark-resight modeling. Based on our observed GPS collaring data and camera spacing, we likely had multiple camera stations within each potential home range, which will greatly assist in our efforts to model abundance and density. Because Fire Island National Seashore east of Old Inlet is geographically isolated from the rest of the national park area but directly adjacent to Smith Point County Park, we combined detection data from the Old Inlet East portion of Fire Island National Seashore with data from Smith Point County Park in our analyses.

Using this detection data, we calculated trap success for red foxes and other potential predator species. Trap success can be used as an index of species activity in a study area. For these analyses, trap success was calculated as the number of independent detection events for a given species divided by the total number of trap nights that cameras were operational for, then multiplied by 100.

We used a single species, single season framework in Program PRESENCE to estimate occupancy ψ (i.e., the probability that a site is occupied by red fox) and detection probability p (i.e., the probability of detecting a red fox during a given survey, provided it is present). Occupancy does not translate directly to abundance, as it does not yield an estimate of individuals detected, just the probability of presence.

We used the null model, where no covariates affected occupancy (ψ) or detection (p), and the survey-specific detection model, where no covariates affected occupancy (ψ) but detection (p) varied by day. We used

Akaike's Information Criterion to select the model that best fit the observed data. We did not collapse data, but rather each day was counted as a survey period, thus our estimated occupancy relates to the whole survey period, while detection rates are interpreted as detection by day. We will model habitat covariates in future analyses and annual reports.

Results and Discussion

Our results and discussion are organized by objective to better reflect progress towards completion.

Objective 1 and 2: Quantify the abundance and occupancy of red foxes, and habitat use vs. availability, and relate changes in these parameters to changes in habitat and predator management activities

Abundance estimates from den surveys and transect monitoring

We combined the remote camera data gathered from each den with GPS collar data and field observations of ear-tagged individuals to estimate minimum fox abundance at each study site (Robert Moses State Park, Fire Island National Seashore from Watch Hill to Old Inlet, Fire Island National Seashore east of Old Inlet + Smith Point County Park). We assumed that each den had 2 adults associated with it. We suspect that this assumption leads to a conservative estimate of population size, as our observations suggest that there may be helper, non-breeding foxes in the population. For example, we have observed >2 adult foxes attending the same den via remote camera photos during both the 2016 and 2017 den monitoring. In another case, we observed an adult female fox attending the den of a breeding pair during a monitoring visit. Both breeding adult foxes at this den were GPS collared and ear-tagged this year, while the other attending female was collared and ear-tagged last year. In these cases, we included confirmed additional attending adults in our overall abundance counts.



Figure 19. Two kits at the entrance of one of the dens that we monitored at Fire Island National Seashore. Photo: Matt Kynoch.

We estimate that there were at least 10 adults and 8 sub-adults (partially grown kits born in the spring of that year) inhabiting Robert Moses State Park and the Fire Island National Seashore Lighthouse Tract at the beginning of August 2017. We estimate that there were at least 13 adults and 2 sub-adults inhabiting the area between Watch Hill and Old Inlet at the beginning of August 2017 (Table 34).

Starting in June 2017, we observed signs of sarcoptic mange (caused by the *Sarcoptes scabiei* mite) in red foxes at Robert Moses State Park. From June to August 2017, 3 GPS collared foxes died from confirmed or strongly suspected mange-related causes, and one collared fox and several unmarked foxes showed signs of infection west of Old Inlet. Our observations are consistent with anecdotal reports of foxes with mange in the western villages of Fire Island in early summer and a general decrease in fox abundance. More recent observations suggest that the outbreak is still ongoing in the state park.

In December 2017, we observed early signs of mange in some foxes via remote camera photos in the Wilderness Area just west of Old Inlet and near Watch Hill. Our current fall camera grids will provide important information needed to assess the overall impact of this ongoing mange outbreak on the red fox population west of Old Inlet. The minimum abundance estimates presented here do not include collared foxes known to have died during the GPS collar tracking period or dead unmarked foxes discovered during monitoring activities (see Objective 4 results for further discussion of observed fox mortalities from mange and non-mange causes).

At the time of our last report, there had been no observations of red fox or likely red fox sign at Smith Point Count Park since April 2016, following large-scale die-off from a previous sarcoptic mange outbreak east of Old Inlet in September 2015–April 2016. During our first round of spring den/sign/scat surveys, both surveyors observed potential red fox tracks in multiple areas of the park. We confirmed the presence of at least one adult red fox in the park using temporary remote camera stations in March 2017. We continue to see tracks during sign/scat surveys since, and unsuccessfully attempted to capture the fox(es) in April 2017. We did not find any evidence of breeding dens in the park or anywhere east of Old Inlet during this breeding season.

We estimated minimum red fox densities in each of the 3 main study areas. Averaging the fox densities from surveyed areas and extrapolating to the unsurveyed areas west of Old Inlet yields an estimated red fox population density of 2.43 adults/km² and 1.05 sub-adults/km² and a total red fox population size of 39 adults and 16 sub-adults west of Old Inlet. There is at least one adult fox east of the inlet, and the minimum density is 0.21 adults/km².

Table 34. Minimum estimated red fox adult and sub-adult abundances and total densities by study area and year. Estimates for all area west of Old Inlet were derived by averaging estimated densities in surveyed areas and extrapolating to the amount of unsurveyed area available.

Study Area	2016			2017		
	Adults	Sub-adults	All foxes/km ²	Adults	Sub-adults	All foxes/km ²
Robert Moses State Park+ Lighthouse Tract	12	16	7.12	10	8	3.79
Fire Island National Seashore	8	14	4.64	13	2	3.16
<i>All area W of Old Inlet</i>	<i>39</i>	<i>57</i>	<i>3.51</i>	<i>39</i>	<i>16</i>	<i>3.48</i>
Smith Point County Park + Old Inlet East	0	0	0	1	0	0.21

Trap success

In addition to red fox, we detected raccoons, feral cats, domestic dogs, gulls, crows, and several species of bird of prey during our remote camera surveys. Red fox was the most commonly detected predator species at sites west of Old Inlet in both years. Interestingly, we did not observe a substantial increase in the trap success of other mammalian predators in the absence of foxes east of the inlet in Fall–Winter 2016, but did observe an increase in the trap success of aerial predators (although our surveys were not designed for aerial species). Species of aerial predators varied among sites and years, and included American kestrel (*Falco sparverius*), Northern harrier (*Circus cyaneus*), red-tailed hawk (*Buteo jamaicensis*), Cooper’s hawk (*Accipiter cooperii*) and short-eared owl (*Asio flammeus*). We caution that trap success should be interpreted as an index of activity rather than abundance. In addition, because we avoided setting camera stations in marsh habitats and on the open beach, our estimates may be biased low for species that primarily use those habitats.

Table 35. Trap success for potential shorebird predator species during fall-winter remote camera surveys.

Study area	Red fox	Raccoon	Feral cat	Domestic dog	Gull	Crow	Bird of prey
Fall-Winter 2015							
RMSP	89.18	13.70	8.22	2.12	0.62	0.14	0.27
FINS wilderness only	57.01	17.23	1.09	0.19	0.45	3.46	0.70
SPCP+OIE	13.99	19.64	2.45	1.07	0.31	0.13	2.20
Fall-Winter 2016							
RMSP	119.45	26.14	8.26	0.84	0.28	0.28	0.22
FINS wilderness only	103.36	24.39	0.39	1.29	0.39	10.03	1.68
SPCP+OIE	0.00	21.84	2.79	0.21	1.55	0.72	5.63

Occupancy estimates from remote camera monitoring

We report here our remote camera survey findings from Fall–Winter 2016; our 2017 camera surveys are ongoing and will be reported in the next annual report.

In each of our study sites, we found that red fox occupancy was best modeled as constant whereas red fox detection varied across sites and years in whether it was best modeled as constant or by day of the survey period (Table 36).

Red foxes were photographed at least once at every camera station in Robert Moses State Park and Fire Island National Seashore west of Old Inlet in Fall-Winter 2016, resulting in an occupancy of 1. Detection was estimated as a constant of 0.67(0.01) at Robert Moses State Park and 0.54 (SE 0.01) at Fire Island National Seashore. At Smith Point County Park and the Old Inlet East portion of Fire Island National Seashore, we did not detect any red foxes, leading to an estimated occupancy of 0 (Table 39). These findings are in agreement with our abundance estimates (above) and home range analyses (see below, Objective 3).

Table 36. Model selection tables from PRESENCE output for occupancy modeling using Fall–Winter remote camera detection data. In each area, the model that allowed detection probability to vary by day had the lowest AIC value, signifying a better fit between the model and the observed data.

Study Area	Model ^a	AIC	ΔAIC	AIC weight	Model Likelihood	# of Parameters
Fall-Winter 2015						
Robert Moses State Park	ψ(.) p(day)	1953.57	0	1	1	53
	ψ(.) p(.)	2021.14	67.57	0	0	2
Fire Island National Seashore	ψ(.) p(day)	1985.15	0	1	1	62
	ψ(.) p(.)	2000.6	15.45	0	0	2
Smith Point County Park+OIE	ψ(.) p(day)	1098.5	0	1	1	47
	ψ(.) p(.)	1124.24	25.74	0	0	2
Fall-Winter 2016						
Robert Moses State Park	ψ(.) p(.)	2257.46	0	1	1	2
	ψ(.) p(day)	2292.93	35.47	0	0	62
Fire Island National Seashore	ψ(.) p(.)	2176.80	0	0.99	1	2
	ψ(.) p(day)	1886.26	9.46	<0.01	<0.01	58
Smith Point County Park+OIE	N/A	-	-	-	-	-
	N/A	-	-	-	-	-

^a ψ=Occupancy, p= detection probability, (.)= constant/no covariates, N/A=unable to model due to lack of detections

Table 37. Occupancy (ψ) and detection probability (p) and estimates from the top model for each study area and year.

Study Area	Fall-Winter 2015			Fall-Winter 2016		
	Top Model	ψ (SE)	p ^a	Top Model	ψ (SE)	p ^a
Robert Moses State Park	ψ(.) p(day)	1 (0)	0.03-0.72	ψ(.) p(.)	1(0)	0.67(0.01)
Fire Island National Seashore	ψ(.) p(day)	1 (0)	0.11-0.61	ψ(.) p(.)	1(0)	0.54(0.01)
Smith Point County Park+OIE	ψ(.) p(day)	0.86 (0.06)	0.03-0.25	n/a	0	n/a

^aIf estimated detection probability (p) varied by day, the range of values is reported here. If constant, a single value is reported.

Baited camera stations

Preliminary examination of photos taken at temporary baited camera stations suggests that placing bait in front of the cameras did not significantly increase tag readability in photos of ear-tagged foxes as compared to unbaited grid cameras. Many of the foxes photographed did not show a clear behavioral response to the bait or approach it to investigate. Those that did tended to spend a longer time in front of the camera, resulting in more photos taken compared to foxes detected during our unbaited camera grids, but this increase in photos did not yield very many photos in which the ear tag numbers were clearly readable. Based on the lack of improved readability, we do not plan to repeat these baited camera stations in future years.

Objective 3: Quantify home range size, long-range movements, and variation in movement ecology of red foxes and relate to changing habitat composition and predator management.

We trapped 17 foxes from February through March 2017 over 3 nights at Robert Moses State Park and 5 nights at Fire Island National Seashore (Table 38). Of these 17 foxes, we outfitted 12 with GPS collars. The remainder were ear-tagged but not outfitted with GPS collars. Three of the captured foxes were recaptures, originally captured in August 2015 (1 fox) and April 2016 (2 foxes). Another of these 17 foxes was captured 2 nights in a row in March 2017 during our unsuccessful attempts to recapture another fox whose collar failed to drop off as programmed at the end of the 2016 monitoring period. We did not successfully trap any foxes at Smith Point County Park over 3 trap nights in April 2017.

We collected data on 3 males and 3 females at Robert Moses State Park, 1 male and 1 female at Fire Island National Seashore's Lighthouse Tract, and 1 males and 2 females at Fire Island National Seashore between Watch Hill and Old Inlet. An additional female was collared just west of Old Inlet, but disappeared soon after. We discovered her remains in the village of Ocean Bay and recovered the GPS collar and recorded data. From the locations recorded, we estimate that she died before the collars were programmed to begin their normal sampling schedule on April 20th, resulting in <10 reliable location points. Thus, this individual was excluded from our spatial analyses. For other foxes that died before the end of the monitoring period, we estimated the date of death using location data, and excluded any post-mortem points from our analyses (see Objective 4 for more information on observed mortalities).

Table 38. Foxes GPS collared during each tracking period. All captured foxes were ear-tagged where permits allowed. Additional foxes were ear-tagged but not fitted with GPS collars due to a limited number of collars.

Management Unit	2015		2016		2017	
	Males	Females	Males	Females	Males	Females
Smith Point County Park + OIE	2	1	0	0	0	0
Fire Island National Seashore – Wilderness Area	3	0	3	3	1	3
Robert Moses State Park + Lighthouse Tract	2	3	4	2	4	4
Total	7	4	7	5	5	7

Home range estimates: MCPs and Kernel densities

In this section of the report, we show home range maps for one male and one female fox from the 2017 tracking period (Figure 46). Home range maps for all foxes tracked, including those tracked during previous monitoring periods, are shown in Appendix B.

We found that Spring-Summer 2017 home range estimates varied widely among individual foxes (Table 39), with an average 50% core area of 166.84 ha for males and 11.53 ha for females, an average 95% home range area of 336.50 ha for males and 48.84 ha for females, and a 100% MCP of 477.24 ha for males and 97.04 ha for females across all three sites (Table 40).

Average home range sizes by sex varied among study areas, with the largest average male home range observed at either Robert Moses State Park or Fire Island National Seashore (depending on estimation method) and the largest average female home range observed at Fire Island National Seashore. Similar to the observations from previous years, we found that females had smaller home range estimates than males, due to the long distance movements of 2 males during the tracking period (See “Long distance movements” below). We hypothesize that the 2 males that exhibited these long distance movements did not breed this year, as we never tracked them to a den, but likely were transient individuals with no defined territory (Dekker et al. 2001).

One pair of collared red foxes at Fire Island National Seashore’s Lighthouse Tract exhibited a high degree of overlap in home range. The female appeared to be breeding and was associated with the both den sites monitored in the area. The male was more loosely associated with the den sites, and may have been either the breeding male or a non-breeding member of the same family group. Aside from these overlapping home ranges, resident individuals appeared to use distinct sections of the island spanning from the bay to ocean as their home ranges, as opposed to using long sections spanning only a narrow portion of the bay or ocean sides of the island.

Breeding individuals, particularly females, used very small areas during the beginning of the Spring-Summer 2017 monitoring period, when their locations were mostly near dens. They expanded their use areas throughout the spring and summer, becoming less restricted to den areas as time progressed. We observed similar patterns during the Spring-Summer 2016 monitoring period.

When the 50% and 95% core and home range areas of all GPS-collared foxes are overlaid, the resulting contours encompass all of Robert Moses State Park and the Fire Island National Seashore Lighthouse Tract, about half of the Fire Island National Seashore Wilderness Area, including all areas east of Bellport Beach where Virginia Tech monitors piping plovers and other shorebirds, and several large areas in the villages (Figure 47).

Table 39. 50% and 95% kernel density and 100% MCP area estimates (ha) for individual red foxes tracked during each monitoring period.

Area	ID	50% kernel	95% kernel	100% MCP
Fall 2015-Winter 2016				
RMSP	F-01	8.9	39.3	72.7
RMSP	M-01	383	1397	1440.2
RMSP	F-02	25.1	72.6	122.3
RMSP	M-02	231.2	833.8	1406.8
RMSP	F-03	18.8	80	170.8
FINS	M-03	210.4	1202.5	1405.1
FINS	M-05	48.9	155.9	425.2
FINS	M-06	269.8	594.3	469
SPCP	M-07	196.2	594.3	528.9
SPCP	F-04	270.5	376.5	437
Spring-Summer 2016				
RMSP	M-08	11.8	53.5	84.6
RMSP	F-05	16	53.5	158.8
RMSP	M-10	108.7	698.8	976.3
RMSP	F-06	15.5	56.8	74.2
RMSP	M-11	664.2	1617.9	1586
FINS	M-13	11.6	39.3	42
FINS	M-14	37.4	140.9	348.8
FINS	F-07	13.1	57	65.3
FINS	F-08	17.5	91	112.8
FINS	F-09	26.4	97.2	117.5
Spring-Summer 2017				
RMSP	F-03	5.9	25.99	43.98
RMSP	F-11	11.83	48.51	54.34
RMSP	F-12	23.23	73.73	205.41
RMSP	M-08	19.02	73.36	68.48
RMSP	M-14	148.22	450.35	819.57
RMSP	M-15	561.63	820.48	797.04
FINS-L	F-13	19.57	107.11	162.38
FINS-L	M-16	17.91	72.04	84.31
FINS	F-14	6.12	24.79	61.38
FINS	F-15	2.55	12.92	42.77
FINS	M-17	87.41	266.28	616.82

Table 40. Mean 50% and 95% kernel and 100% MCP area home range estimates (ha) by tracking period, study area, and sex.

Study area	Sex	50% kernel area	95% kernel area	MCP area
Fall 2015-Winter 2016				
RMSP	M	307.1	1115.4	1423.5
	F	17.6	64.0	121.9
FINS (Wilderness)	M	129.6	679.2	915.1
	F	n/a	n/a	n/a
SPCP + FINS (E of inlet)	M	233.0	594.3	499.0
	F	270.5	376.5	437.0
All areas	M	223.3	796.3	945.9
	F	80.8	142.1	200.7
Spring-Summer 2016				
RMSP	M	261.6	790.1	882.3
	F	15.7	55.1	116.5
FINS (Wilderness)	M	24.5	90.1	195.4
	F	19.0	81.7	98.5
All areas	M	166.7	510.1	607.6
	F	17.7	71.1	105.7
Spring-Summer 2017				
RMSP+ Lighthouse Tract	M	186.70	354.06	442.35
	F	15.13	63.84	116.53
FINS (Wilderness)	M	87.41	266.28	616.82
	F	4.34	18.86	52.08
All areas	M	166.84	336.50	477.24
	F	11.53	48.84	95.04

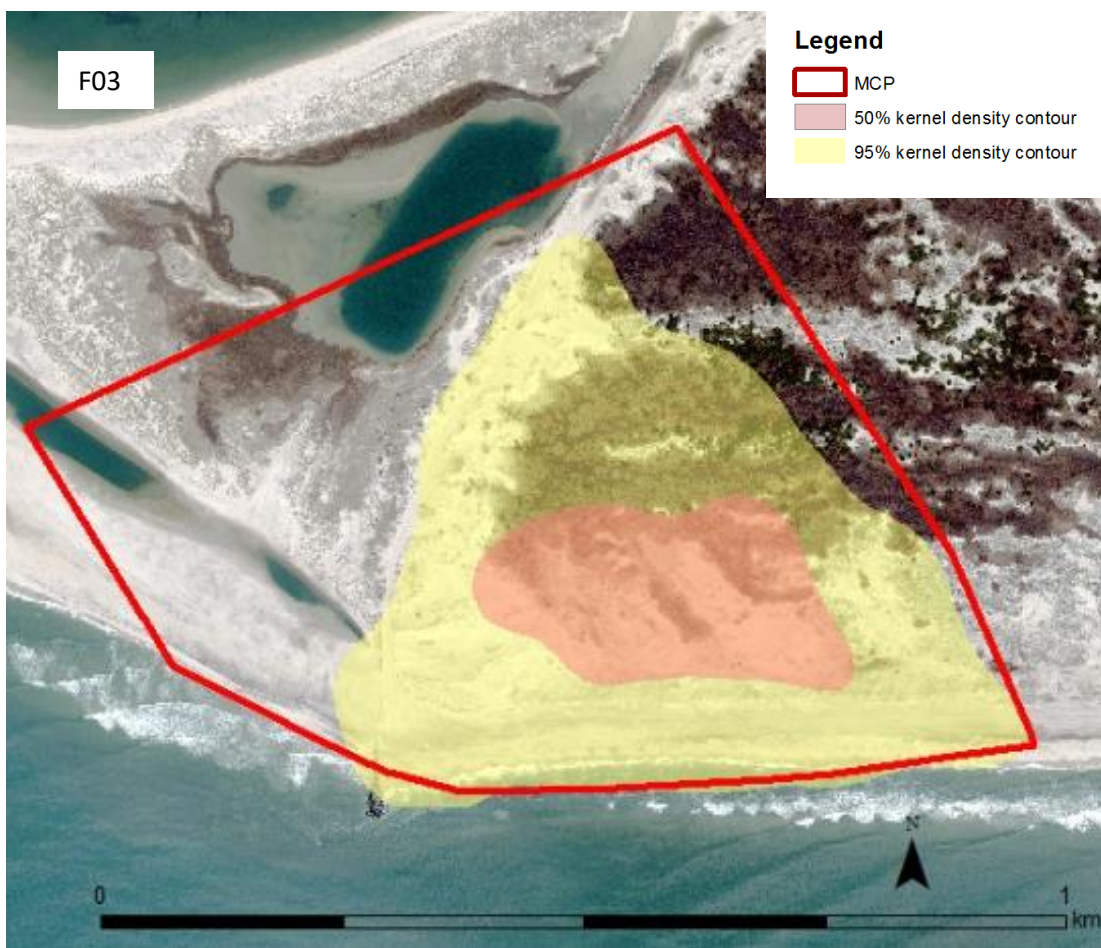
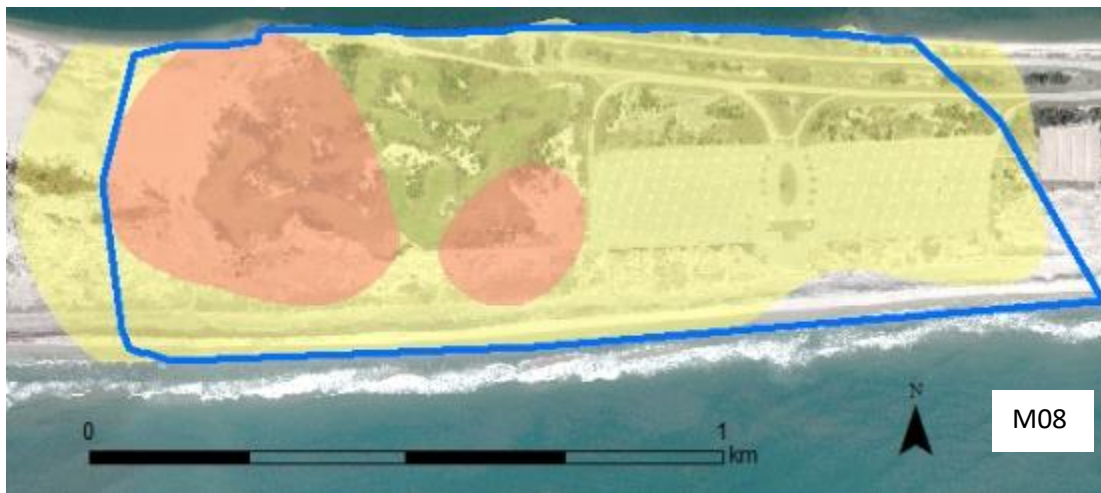


Figure 46. MCP, 50% and 95% kernel density contours for 1 male red fox (top) and 1 female red fox (bottom) at Robert Moses State Park, Spring-Summer 2017.



Figure 47. 50% and 95% kernel density contours for all GPS-collared red foxes in Spring-Summer 2017 at Robert Moses State Park and Fire Island National Seashore Lighthouse Tract (top) and Fire Island National Seashore Wilderness Area (west of Old Inlet, bottom) overlaid on top of each other.

Long distance movements

During previous monitoring periods, we observed multiple GPS-collared male foxes making long distance movements, as described in previous reports. During the 2017 tracking period we observed 2 additional GPS-collared male foxes making similar long distance movements, between the westernmost end of Fire Island and the Sailor's Haven area. Both were initially captured at Robert Moses State Park, and moved regularly between the park, the villages, and Sailor's Haven until their deaths (both died before the end of the tracking period, one due to unknown causes and one due to suspected drowning/complications of mange; see 'Adult Survival' in Objective 4 Results for further information on observed mortalities).

Objective 4: Estimate red fox reproductive success and adult survival, and variation in these population parameters, related to habitat composition and predator management.

Reproductive success

We monitored 7 breeding dens at Robert Moses State Park and, 2 breeding dens at Fire Island National Seashore's Lighthouse Tract, and 2 breeding dens at Fire Island National Seashore (between Watch Hill and Old Inlet) during April–July 2017 (Table 41). No active dens were detected at Smith Point County Park during this time period.

Red foxes at all sites used multiple den sites within a breeding season and moved between dens as the season progressed, similar to our observations during last year's breeding season. We identified dens active from late April–May as natal dens, and dens that were not active until June–July as secondary dens used by family groups that had relocated from their natal dens. Based on remote camera photos, timing of use, and amounts of activity observed, 3 of the 7 breeding dens at Robert Moses State Park appeared to be natal dens. At Fire Island National Seashore's Lighthouse Tract, one of the 2 breeding dens appeared to be a natal den. The remaining breeding dens appeared to be secondary dens, used later in the breeding season. In the Fire Island National Seashore wilderness area, both of the 2 breeding dens appeared to be natal dens. The family groups associated with each of these dens were not tracked to distinct secondary dens later in the season, but did appear to use nearby above-ground areas more heavily and shift away from regular use of natal dens as the season progressed.

Similar to last year, we located additional dens during our transect surveys in Robert Moses State Park and Fire Island National Seashore. These dens did not appear to be breeding dens, and likely served as thermal refugia from the summertime heat and/or daytime underground resting sites used by adults. We did not find any signs of kits at these dens. Most were only temporarily active, and were not monitored beyond 2 weeks past the last signs of activity.

At Robert Moses State Park and Fire Island National Seashore's Lighthouse Tract, 13 kits were observed at breeding dens among 4 likely family groups, for an average of 3.25 kits per pair. 2 of these kits were later found dead during den monitoring activities or reported dead to us by other Virginia Tech monitors (see below). Between Watch Hill and Old Inlet, we observed 2 kits among 2 likely family groups, for an average of 1 kit per pair. We observed an additional apparently lactating female during den monitoring activities between Watch Hill and Old Inlet, but did not observe any signs of kits at that den, despite remote camera monitoring and thorough searching for kit-sized scats and tracks. Compared to our 2016 estimates, the numbers of kits per pair in 2017 were similar in Robert Moses State Park and Fire Island National Seashore's Lighthouse Tract (4 kits per pair in 2016), but significantly lower between Watch Hill and Old Inlet (3.5 kits per pair in 2016). This difference in reproduction is likely due in part to the fact that one of the 2016 dens had at least 7 kits, which may have skewed that year's average kits per pair. It could also be related to a decrease in anthropogenic food availability due to the closure of the Watch Hill marina and decreased visitor numbers.

Table 41. Breeding dens located on transect surveys, through GPS tracking, and opportunistically in April–July 2016 and 2017, and the associated number of likely family groups.

Management Unit	2016		2017	
	Breeding dens	Likely family groups	Breeding dens	Likely family groups
Smith Point County Park + Old Inlet East	0	0	0	0
Fire Island National Seashore – Wilderness Area	7	4	2	2
Robert Moses State Park + Lighthouse Tract	10	4	7	4
Total	17	8	9	6

Adult Survival

Three of the 8 foxes that were GPS collared in 2017 at Robert Moses State Park and the Fire Island National Seashore Lighthouse Tract, and 3 of the 4 foxes at Fire Island National Seashore (between Watch Hill and Old Inlet) were alive at the time their collars dropped off or failed. Of the 6 deceased foxes, 2 died (of confirmed or strongly suspected) sarcoptic mange and/or related complications, and one most likely died of drowning in a water treatment pit. A second male collared fox was also found dead in the water treatment pit, but it is unclear whether he ultimately died from drowning or complications of mange, as he had been observed within the previous week showing signs of severe mange. Two foxes died of unknown causes. One of these deaths may have been due to vehicle collision, but the carcass was too decomposed upon discovery to conclude a cause of death. We collected the carcass of the other fox that died of unknown causes for future necropsy.

In addition, we observed or received reliable reports from collaborators or other Virginia Tech monitors of the deaths of 4 unmarked adult foxes and 2 unmarked kits in Robert Moses State Park between January and August 2017. 3 adults and one kit were suspected or reported to have died of vehicle collisions. The remaining adult was killed with a crossbow. The remaining kit was discovered <50m from an active den during den monitoring activities, and died of unknown causes.

We ear-tagged but did not collar 2 foxes (1 was a recapture from the previous year who had lost her ear tag) at Robert Moses State Park, and 3 foxes in the wilderness area of the Seashore; we do not have reliable information yet on whether these foxes survived through the end of the summer.

Based on known survival/mortality of GPS collared individuals, we estimate adult survival during our 2017 monitoring periods as 37.5% at Robert Moses State Park and the Fire Island National Seashore Lighthouse Tract and 75% at Fire Island National Seashore (between Watch Hill and Old Inlet). These estimates are lower than those resulting from the 2016 monitoring period (100% in both areas where foxes were collared and monitored from April–August 2016), likely due to the ongoing mange outbreak and observed mortalities. With

no marked foxes east of the inlet, we cannot estimate survival in Smith Point County Park or Old Inlet East during the 2017 monitoring season.

Recaptures, resightings, and VHF tracking of previously collared foxes whose GPS collar drop-offs failed allowed us to begin confirming the survival of some foxes from previous monitoring periods. VHF tracking of 2 adult foxes first captured and collared in April and May 2016 confirmed their survival through at least February 2017. One of these adults was also resighted in the field on April 24, 2017. We recaptured two adult foxes first captured in August 2015 and April 2016, respectively, and recorded their survival through collar drop-off in October 2017 and up until shortly before collar drop-off in August 2017, respectively. We resighted one adult fox that was originally captured in April 2016 multiple times during the 2017 spring and summer den/sign/scat transects and den monitoring. We anticipate that continued data collection and analysis will allow us to estimate survival between monitoring periods, based on remote camera detections, potential recaptures, field observations, and known-fate survival modeling, in future reports.

Objective 5: Quantify the interactions of red foxes with shorebird nests and chicks.

Diet analyses

We collected 47 scat samples during our transect surveys, den monitoring, and opportunistically between September 2016 and March 2017, and 87 samples from April–August 2017 (Table 42). To date, we have completed preliminary fecal dietary analysis on 186 samples, and are in the process of preparing samples to send for genetic verification of source species. From these samples, we have identified 823 distinct items to varying taxonomic levels. Preliminary estimates, pooled across multiple survey seasons and study sites of percent occurrence (defined as the percentage of total items identified belonging to a given source category) and frequency of occurrence (defined as the number of samples containing at least one item belonging to a given source category) suggest that red foxes on the island have highly variable diets (Table 43). Approximately 6.32% of items identified were from birds (feathers, bird bones, beaks, etc.), and approximately 21.51% of samples collected included at least one item of avian origin. The majority of bird remains found so far have not been readily identifiable to species due to a lack of unique identifying characteristics (bone and beak fragments) or degradation during the digestion process (feathers).



Figure 20. Some anthropogenic food items found at active dens during den monitoring activities, May–July 2017.

In addition, we recorded a wide variety of prey remains outside of den entrances and in the immediate vicinity during our monitoring activities from April–August 2017, including remains from little skates (*Leucoraja erinacea*), spiny dogfish (*Squalus acanthias*), assorted other fish, soft-shell clams (*Mya arenaria*), and white-tailed deer (*Odocoileus virginianus*). We also observed a variety of anthropogenic food items, including fast food bags and wrappers (Figure 56).

In future analyses, we plan to compare prey remains recorded during den monitoring to the results of our fecal dietary analysis of scats collected from those dens to determine whether certain items of interest might be underestimated using feces due to limited post-digestion identifiability.

Table 42. Scat samples collected during transect surveys, den monitoring, and opportunistically during the course of the study.

Management Unit	June–July 2015	April–August 2016	September 2016–March 2017	April–August 2017
Smith Point County Park	17	0	1	3
Fire Island National Seashore – east of breach	4	0	0	0
Fire Island National Seashore – west of breach	16	54	27	39
Fire Island National Seashore – Lighthouse Tract	5	5	2	5
Robert Moses State Park	34	42	17	40
Total	76	101	47	87

Table 43. Percent occurrence and frequency of occurrence for source categories of 823 items identified during preliminary fecal dietary analysis of 186 scat samples.

Source	% Occurrence	Frequency of Occurrence
Anthropogenic	4.13	0.15
Avian	6.32	0.22
Crustacean	8.51	0.32
Fish	2.07	0.09
Insect	19.93	0.63
Mammalian	25.52	0.74
Unidentifiable	10.21	0.38
Unknown bone	1.94	0.08
Vegetation	21.39	0.75

Spatial overlap with piping plover nests and locations

We used two sets of data from our Spring–Summer 2017 GPS collared foxes to explore spatial interactions between red foxes and piping plovers: the hourly/every two hour data set collected throughout the season and the 15-minute high frequency sampling conducted around exclosure setup and hatching periods.

Hourly/Every Two Hour Location Data

We found, that all dry land area in Robert Moses State Park and the Fire Island National Seashore Lighthouse Tract and approximately half of the Wilderness Area west of Old Inlet was within the home range of at least one collared red fox (see Objective 3, Figure 77). There were many more foxes on the island beyond those we had collared; these results only describe home ranges and overlap among collared foxes and piping plovers. All 14 piping plover nests in Robert Moses State Park and the Fire Island National Seashore Lighthouse Tract were located within the 50% and 95% kernel density home range of at least one fox (Figure 49). Of these nests, 6 hatched successfully.

All piping plover adult and brood locations west of Old Inlet were also located within the 95% kernel density home range of at least one fox, and most were also located within the 50% kernel density core area of at least one fox (Figure 58). During Spring-Summer 2017, no red foxes were collared at Smith Point County Park nor the Old Inlet East unit of Fire Island National Seashore, thus no nests, brood locations, or adult locations in those areas were overlapping with collared red fox territories.

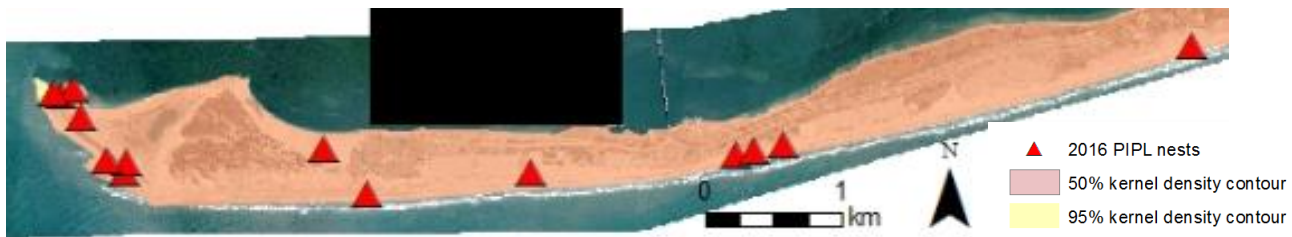


Figure 49. Plover nest locations, 50% and 95% kernel density contours for all GPS-collared red foxes in Spring-Summer 2017 at Robert Moses State Park and Fire Island National Seashore Lighthouse Tract overlaid on top of each other.

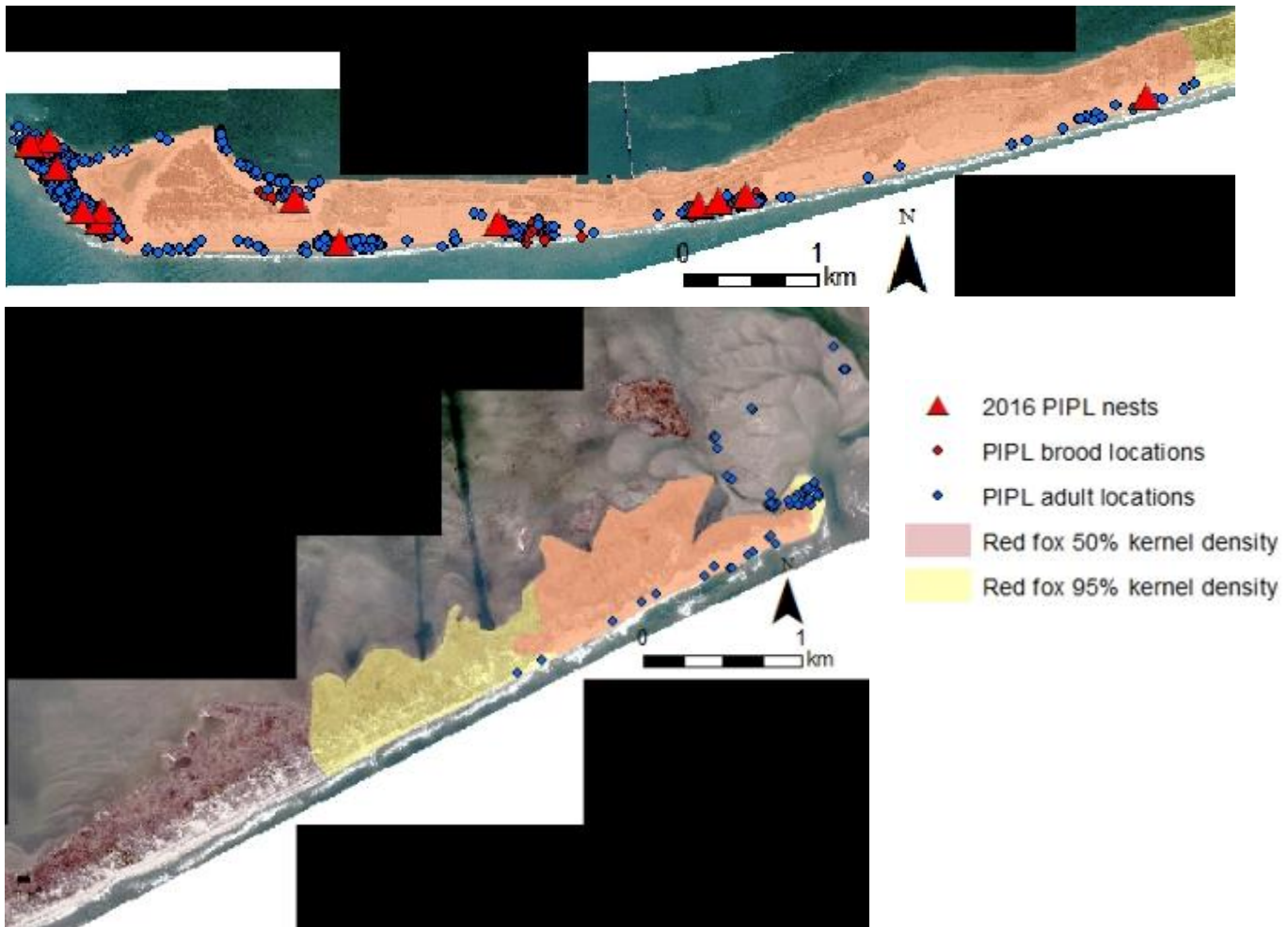


Figure 21. Piping plover nest, brood, and adult locations overlaid onto overall red fox kernel density contours at Robert Moses State Park and Fire Island National Seashore Lighthouse Tract (top) and Fire Island National Seashore Wilderness Area (bottom).

High frequency (15-min) Location Data

We were able to reprogram collars and collect data on high frequency sampling schedules for 5 GPS collared foxes surrounding nest discovery and exclosure, and for 3 of these foxes surrounding expected pipping and hatch dates.

We show one set of maps with fox locations relative to plover nests before vs. after exclosure setup and before vs. after nest pipping and hatching in the figures below (Figure 59, Figure 52); All maps for high frequency location sampling are included in Appendix C. Based on GPS locations collected during these focal sampling periods, collared foxes did not exhibit a clear spatial response (detectable at the sampling frequency) to nest exclosure, pipping, or hatching. Fox locations appeared to be correlated to den sites for breeding individuals, and resting sites for nonbreeding individuals, with little to no visible shift in locations before vs. after target events. However, this is only preliminary information, based on data from only a few individuals, and more information is needed to assess whether our observations are representative of the larger fox population's response to target events. While we did not observe (via our GPS locations or nest cameras) the circling behavior around exclosures observed in other studies, this does not mean that this behavior never occurs.

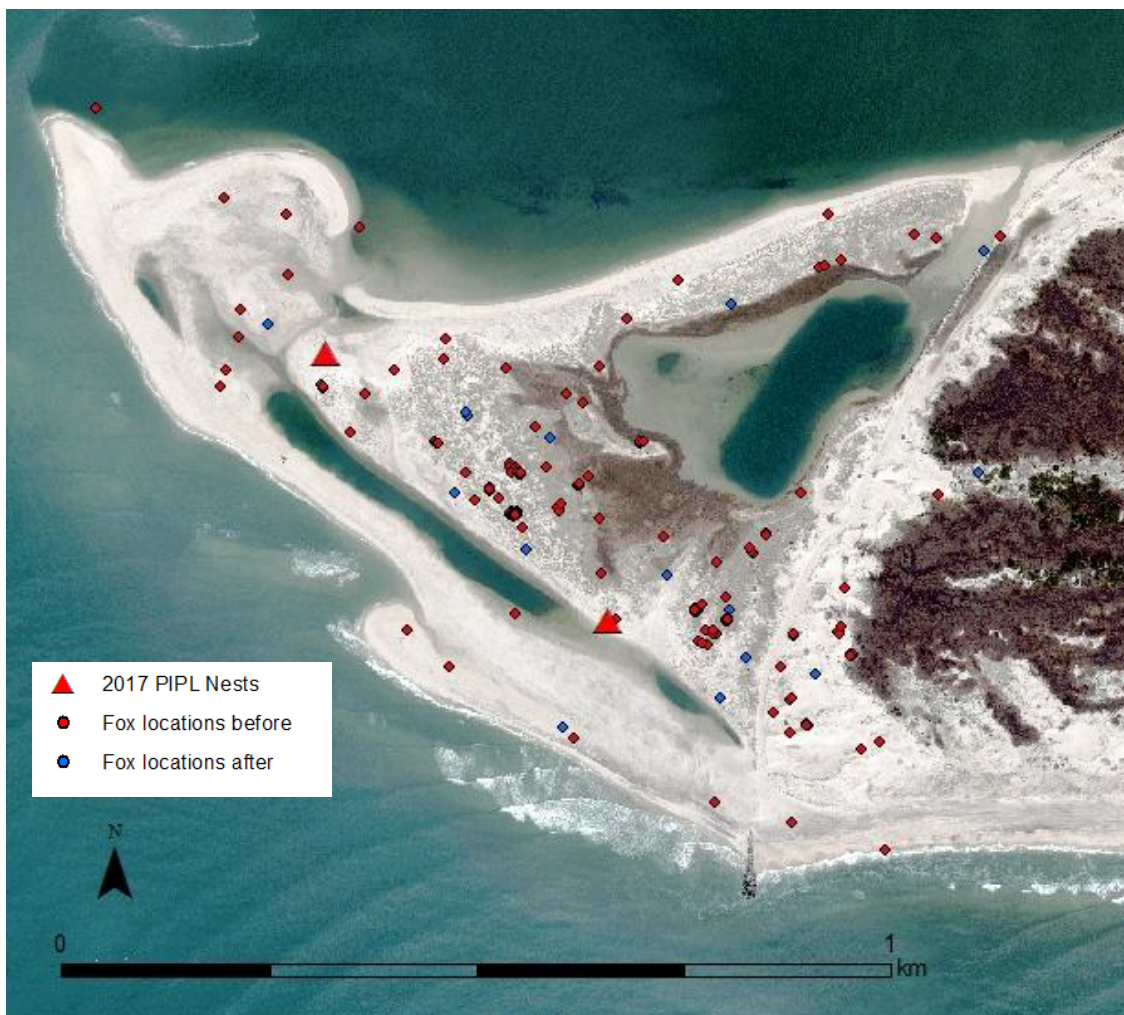


Figure 22. Locations for F11 before and after nest exclosure at Robert Moses State Park, PIPL nests 203 and 204.

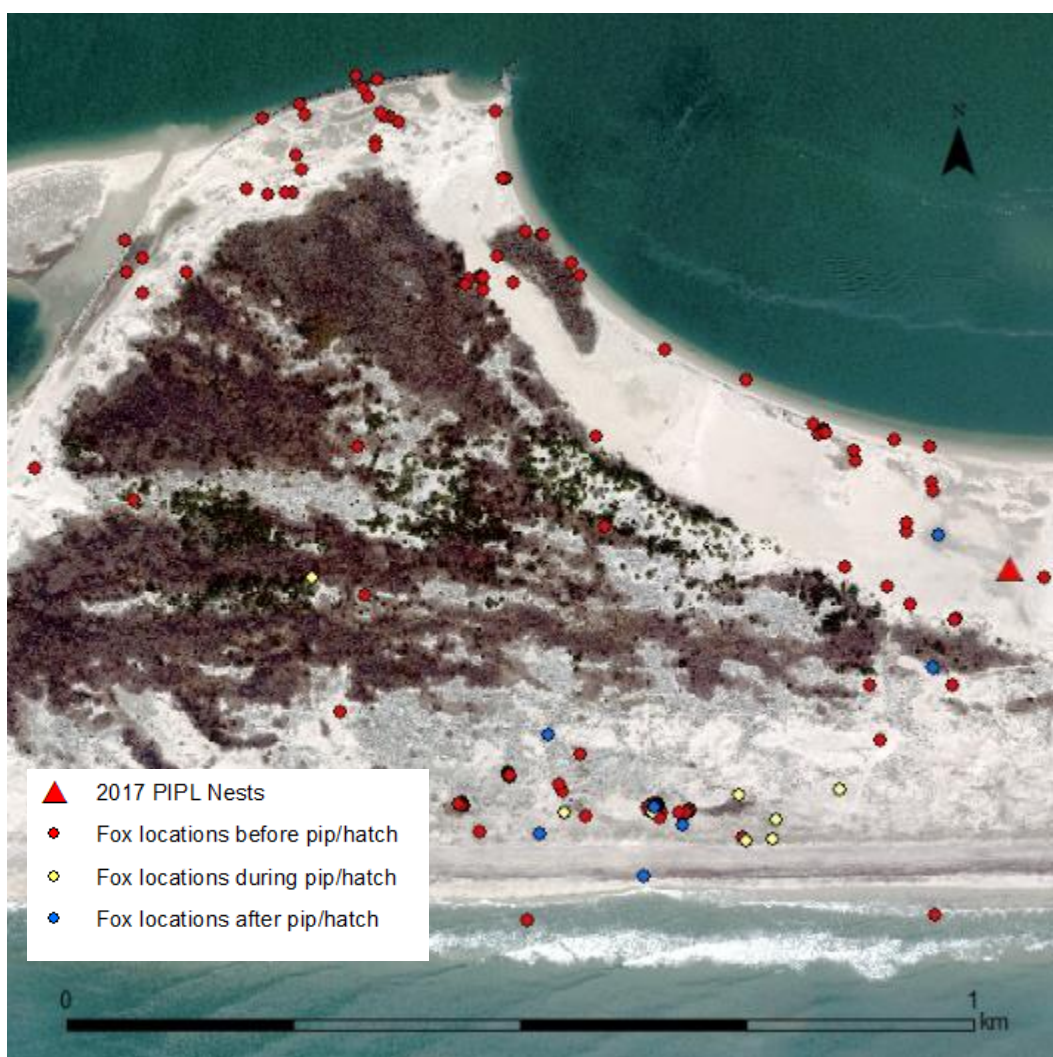


Figure 52. Locations for F12 before, during, and after nest pipping/hatching at Robert Moses State Park, PIPL nest 205.

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Appendices

Appendix A: Injured Piping Plover Observations

Table 44. Piping plovers seen with a swollen leg or limping, April–August, 2017 after banding at Fire Island and Westhampton Island.

Flag ID	Age of Banding	Date Banded	Subsite	First Seen Limping	Seen not Limping	Comments
GF(YCY)	Chick	7/22/2015	CSW	6/5/2017	N/A	Reported by Rob Longiaru, that the foot/leg was broken. From pictures, believed not band related. Seen 8/1/17 at Cupsogue, leg missing below joint. Appears to be moving well enough to carry on essential life functions, band and flag still in place and in good condition.
GF(A98)	Chick	6/8/2016	PSO	3/27/2017	N/A	Dragging flag leg. Flag sealed, not necrotic, flag not over wrist, leg swollen to flag, no clear injury, still getting blood flow, using leg from time to time. Likely lost leg, as a bird was seen with a single leg, with an orange band in the same location in the fall of 2017.
GF(V34)	Chick	6/26/2016	OSS	8/15/2016	N/A	Have a photo and video. Discoloration apparent in the leg. Assumed the flag was lost and/or the bird died based on the photographs from October. Seen again in summer 2017 with missing leg, but alive. Determined from orange band.
GF(1YA)	Chick	6/29/2017	SPT	8/1/2017	N/A	Limping, favoring right leg. Leg is swollen from the orange band down to the foot. Walking gingerly, and not putting much weight down on right leg. Could not tell if band was impacted.
GF(NLC)	Chick	6/26/2017	NBW	7/25/2017	8/11/2017- New Jersey	Seen holding flag leg up while walking. Not putting pressure on left leg unless at rest. Leg visibly swollen above joint, and leg looked discolored a darker red.
GF(1C1)	Chick	7/3/2017	OIE	7/27/2017	8/8/2017	Limping and favoring his right leg, didn't put weight on it, was hobbling around while foraging and walking, and could put weight on it when necessary.

Appendix B. Home Range Maps

Individual home range maps and for all GPS collared foxes tracked during Spring-Summer 2017 and previous monitoring periods are shown below, grouped by sex and study area. We also show 50% and 95% kernel contours for all tracked foxes in each study area and tracking period overlaid on top of each other, to show broader patterns of area use among the overall sample population.

Spring-Summer 2017

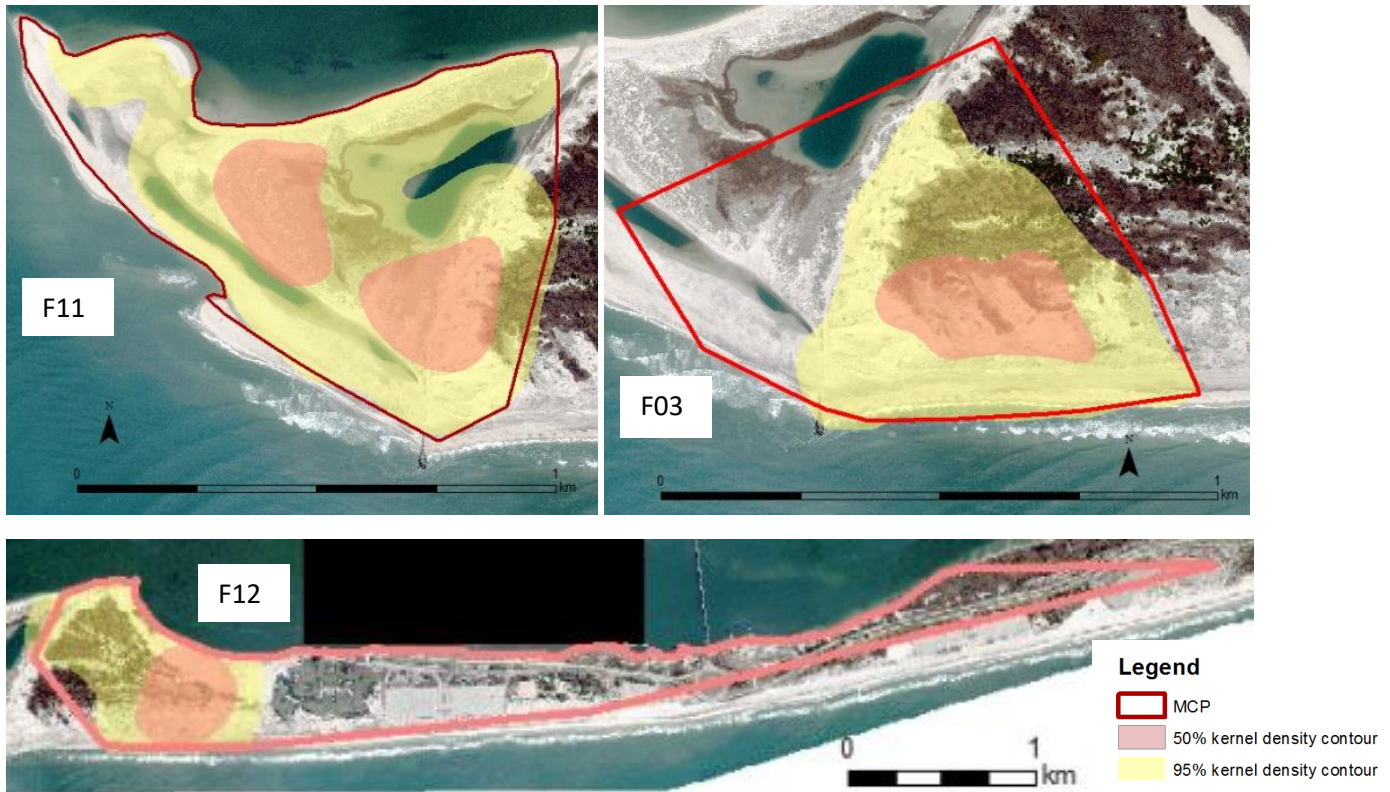


Figure 53. MCPs (solid lines), 50% and 95% kernel density contours for 3 female red foxes at Robert Moses State Park, Spring-Summer 2017.

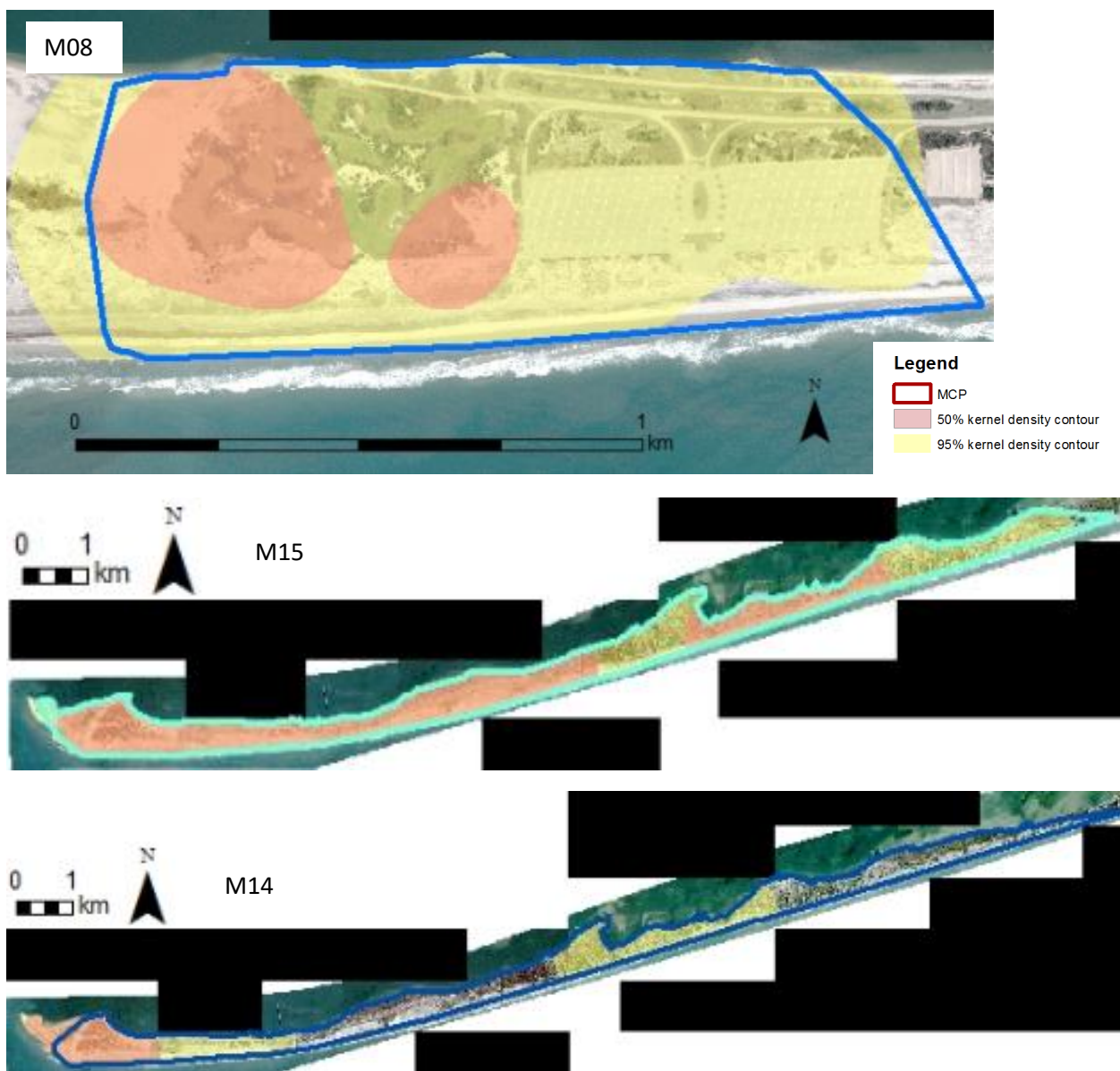


Figure 54. MCPs, 50% and 95% kernel density contours for 3 male red foxes originally captured at Robert Moses State Park, Spring-Summer 2017.

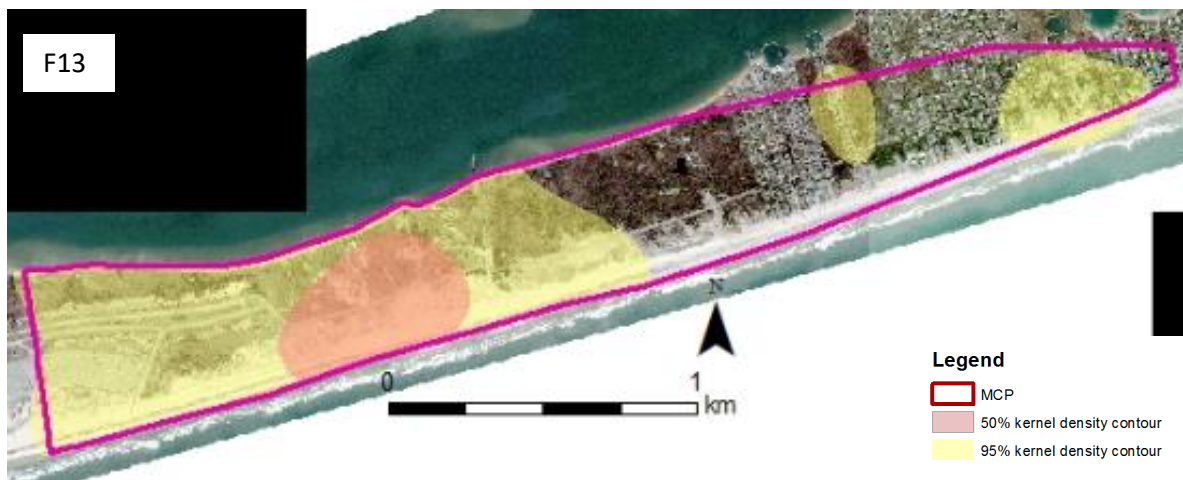


Figure 55. MCP, 50% and 95% kernel density contours for one female red fox at Robert Moses State Park/Fire Island National Seashore Lighthouse Tract, Spring-Summer 2017.

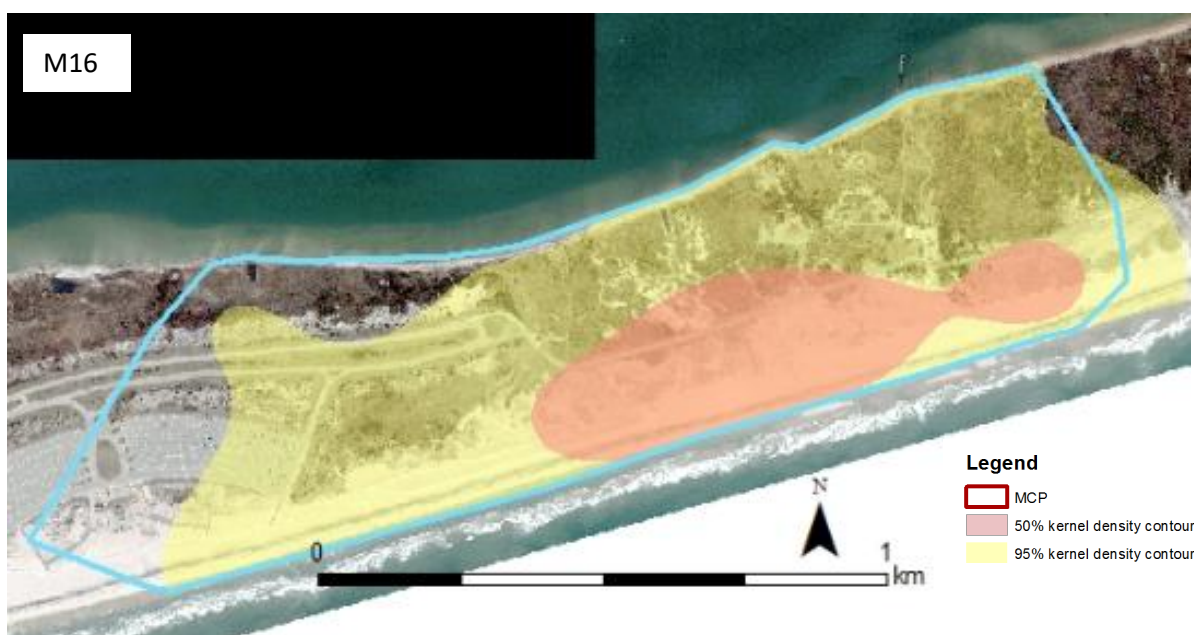


Figure 56. MCP, 50% and 95% kernel density contours for one male red fox at Robert Moses State Park/Fire Island National Seashore Lighthouse Tract, Spring-Summer 2017.

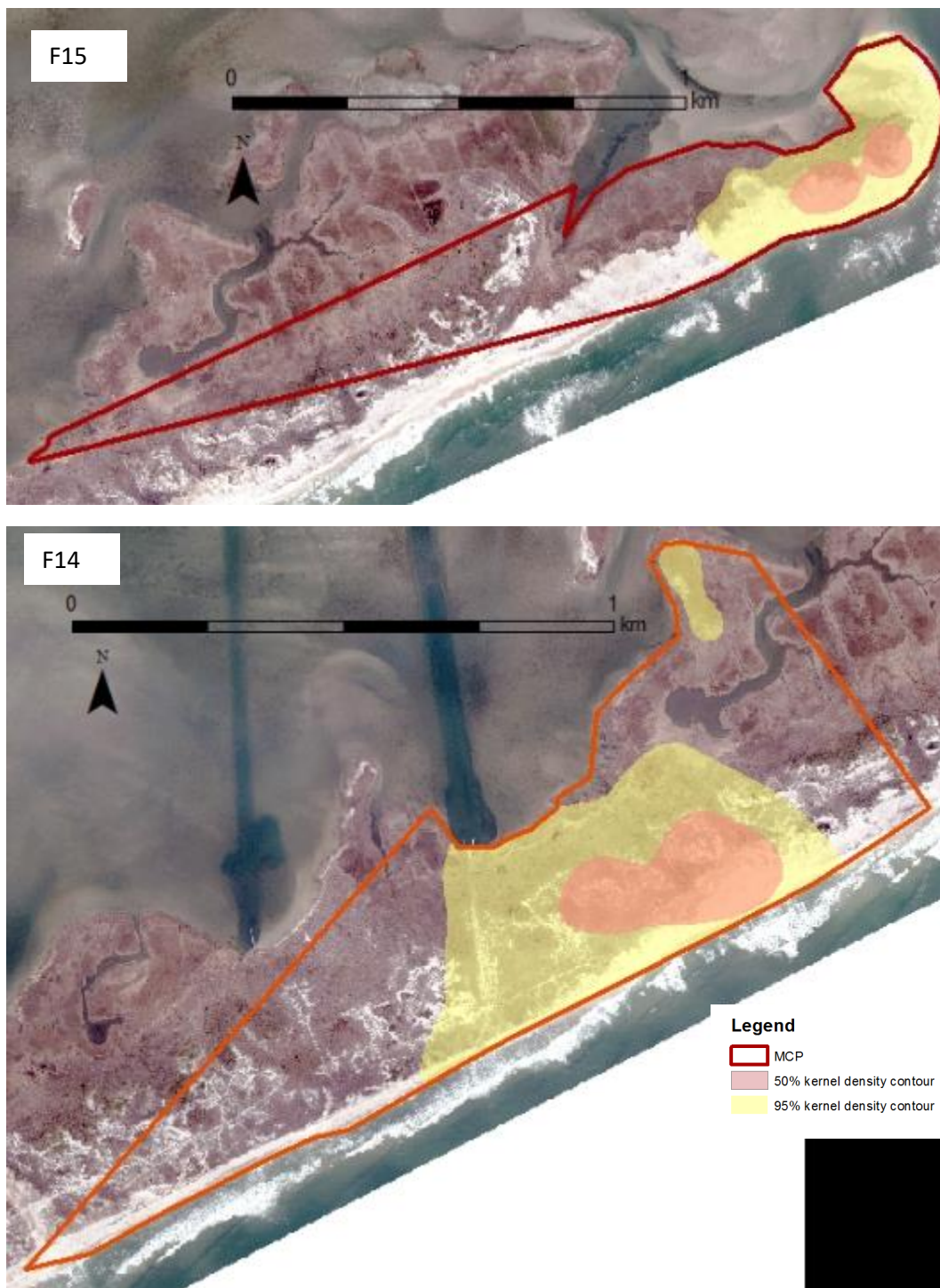


Figure 57. MCPs, 50% and 95% kernel density contours for 2 female red foxes at Fire Island National Seashore, Spring-Summer 2017.

M17

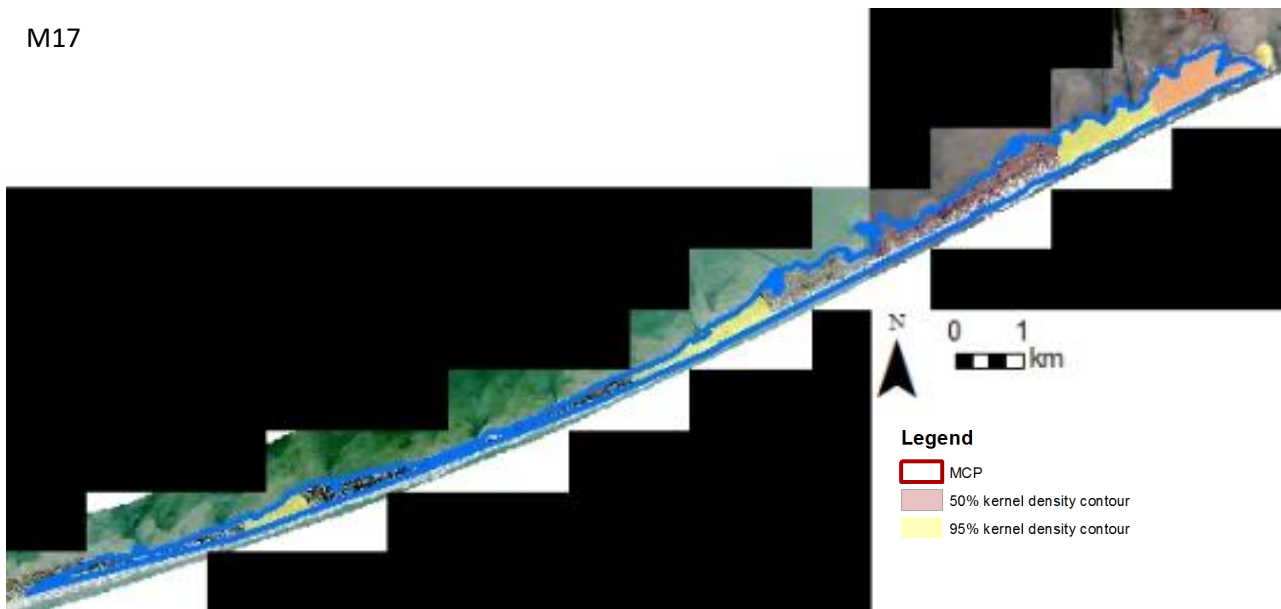


Figure 58. MCP, 50% and 95% kernel density contours for one male red fox at Fire Island National Seashore Lighthouse Tract, Spring-Summer 2017.

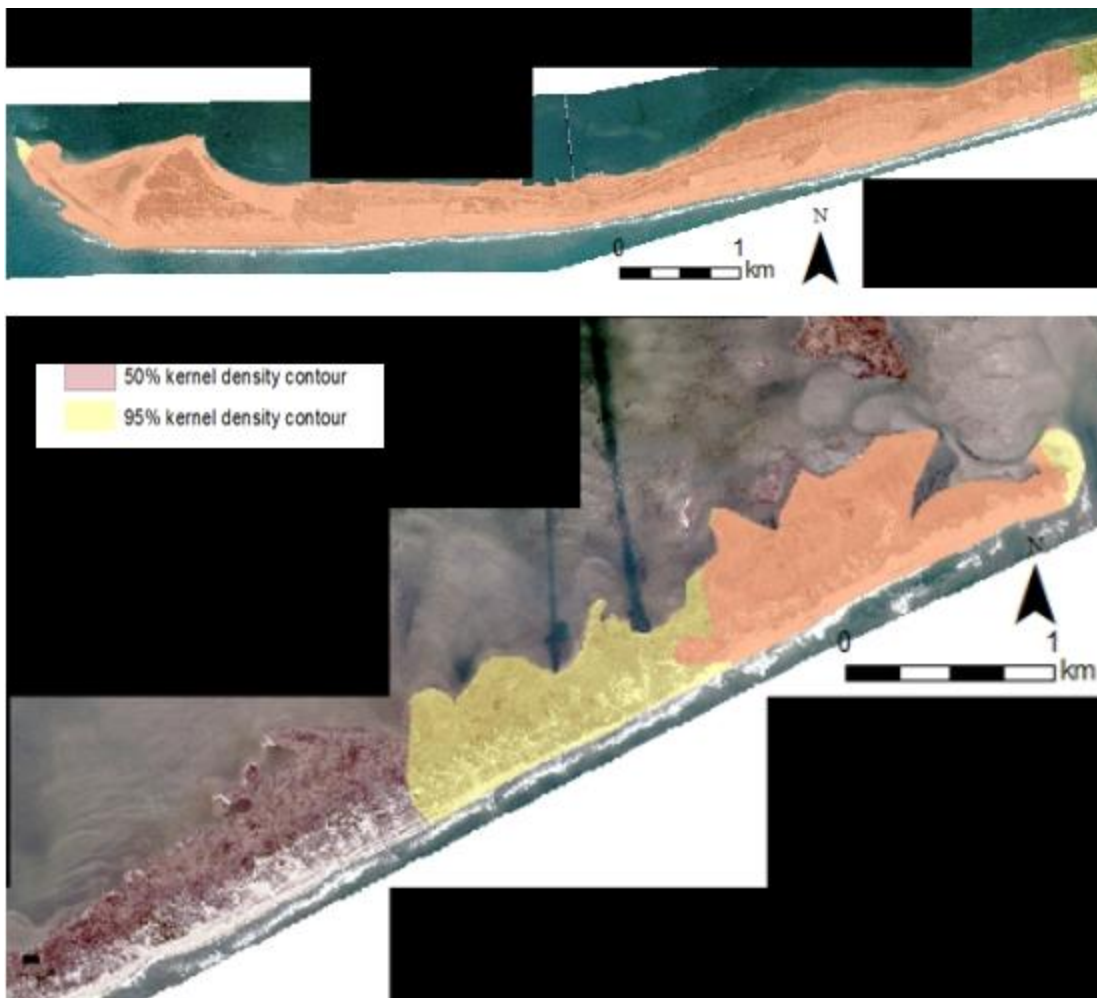


Figure 59. 50% and 95% kernel density contours for all GPS-collared red foxes in Spring-Summer 2017 at Robert Moses State Park and Fire Island National Seashore Lighthouse Tract (top) and Fire Island National Seashore Wilderness Area (west of Old Inlet, bottom) overlaid on top of each other.

Previous monitoring periods (in chronological order):

Fall 2015-Winter 2016

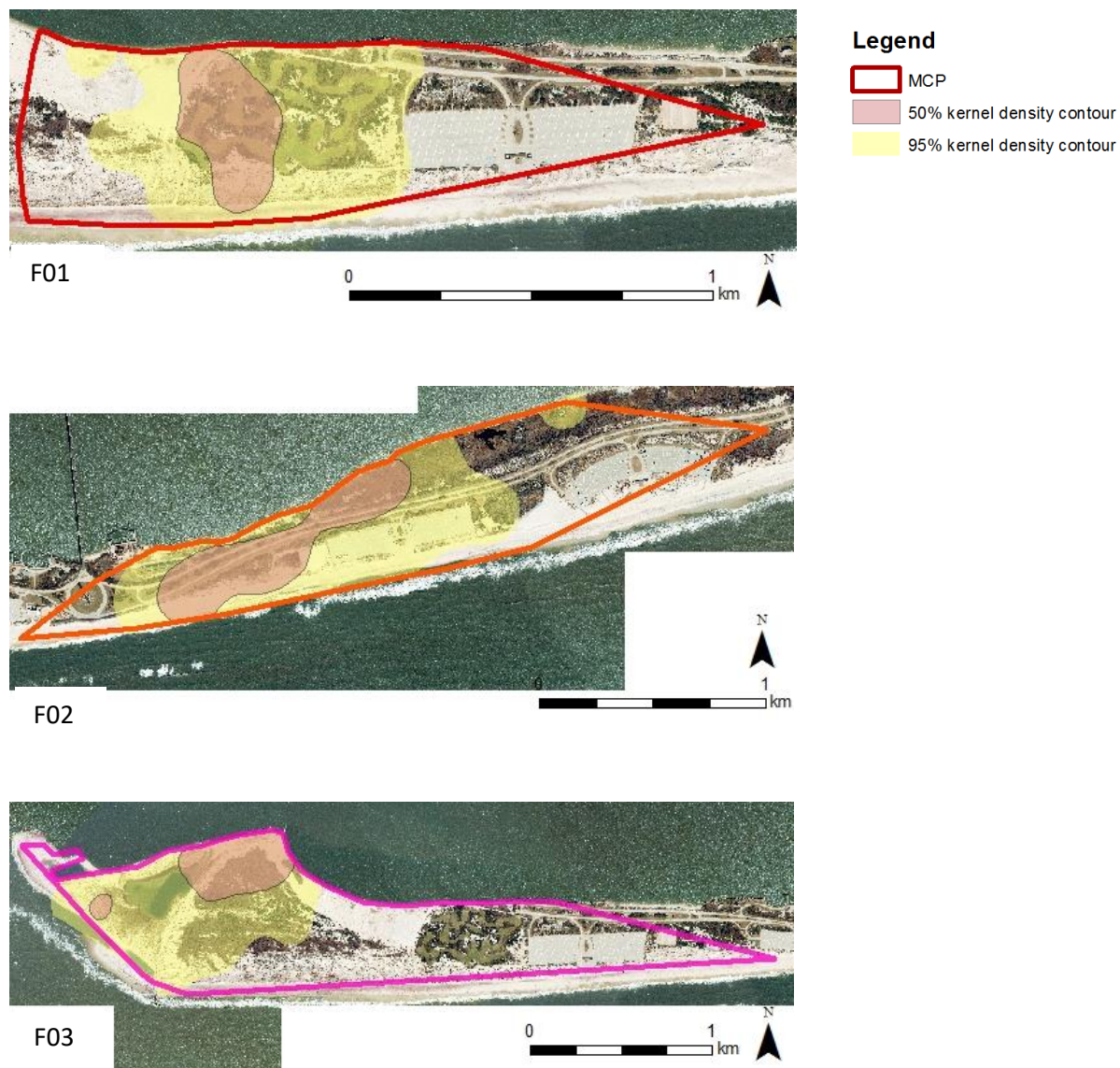


Figure 60. MCPs (solid lines), 50% and 95% kernel density contours for 3 female red foxes at Robert Moses State Park, Fall-Winter 2015.

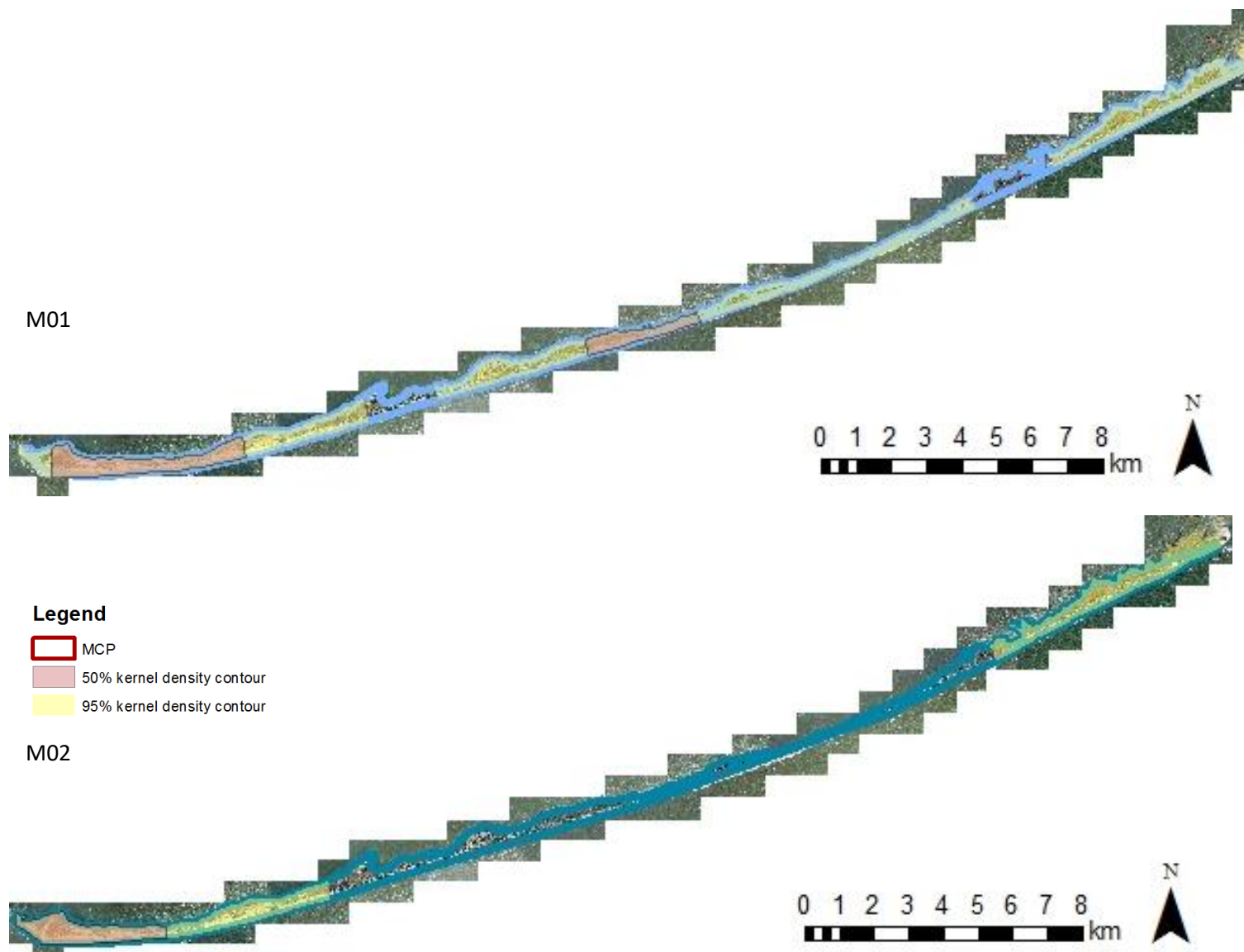


Figure 61. MCPs, 50% and 95% kernel density contours for 2 male red foxes originally captured at Robert Moses State Park during Fall-Winter 2015.

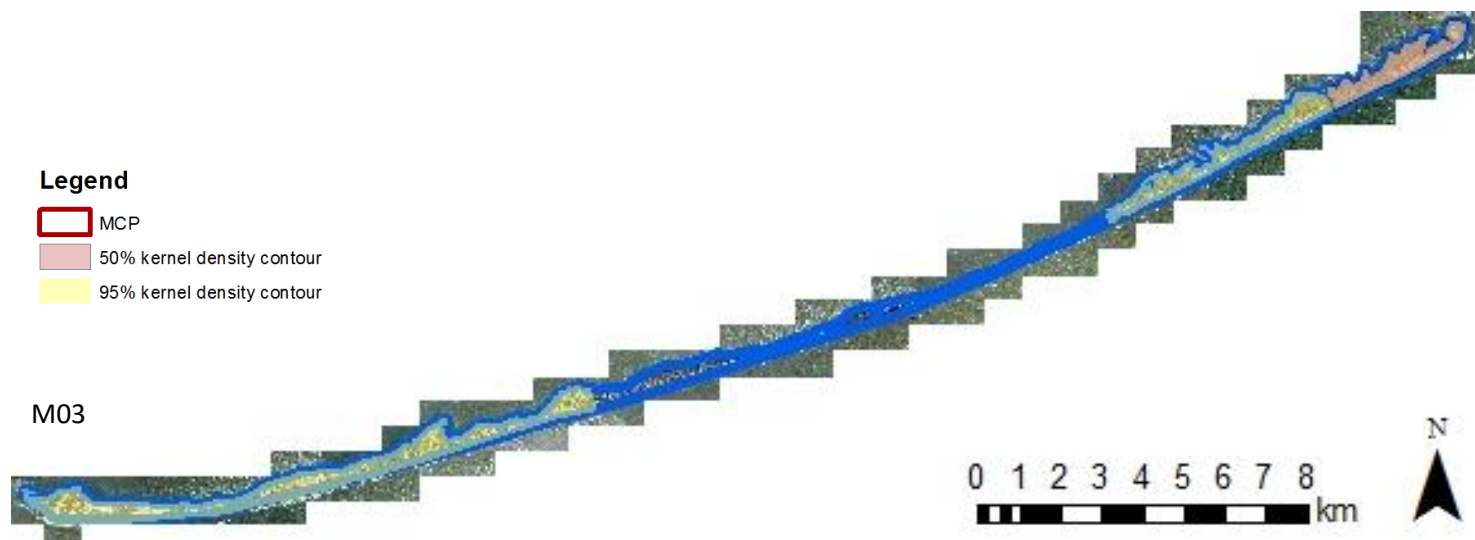


Figure 62. MCP, 50% and 95% kernel density contour for 1 male red fox originally collared at Fire Island National Seashore in Fall-Winter 2015.

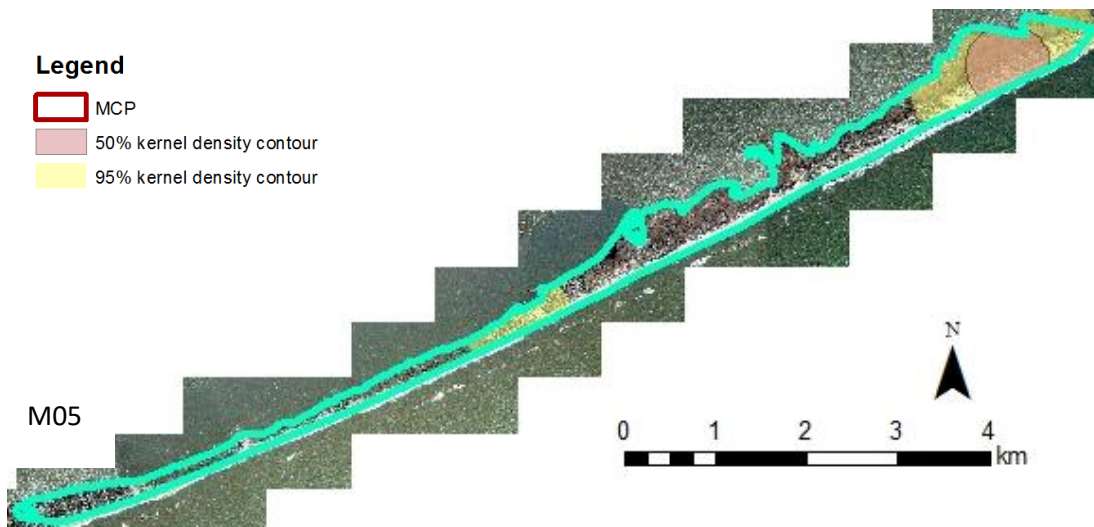


Figure 63. MCP, 50% and 95% kernel density contour for 1 male red fox at Fire Island National Seashore in Fall-Winter 2015.

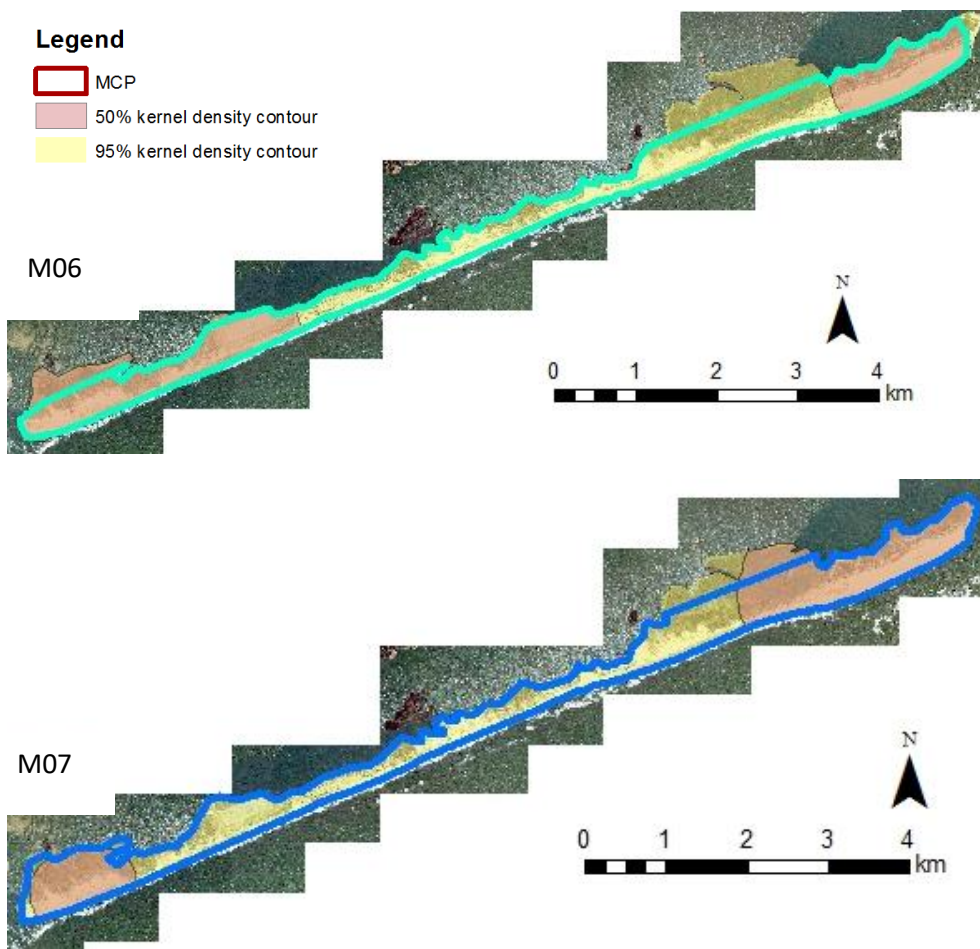


Figure 64. MCP, 50% and 95% kernel density contour for 2 male red foxes at Fire Island National Seashore (Old Inlet East) and Smith Point County Park in Fall-Winter 2015.

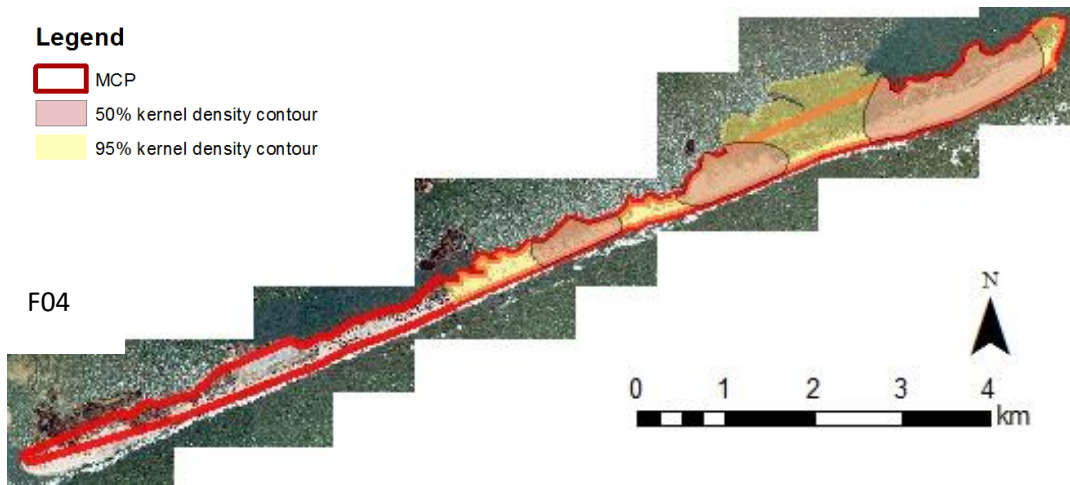


Figure 65. MCP, 50% and 95% kernel density contour for 1 female red fox at Fire Island National Seashore (Old Inlet East) and Smith Point County Park in Fall-Winter 2015.

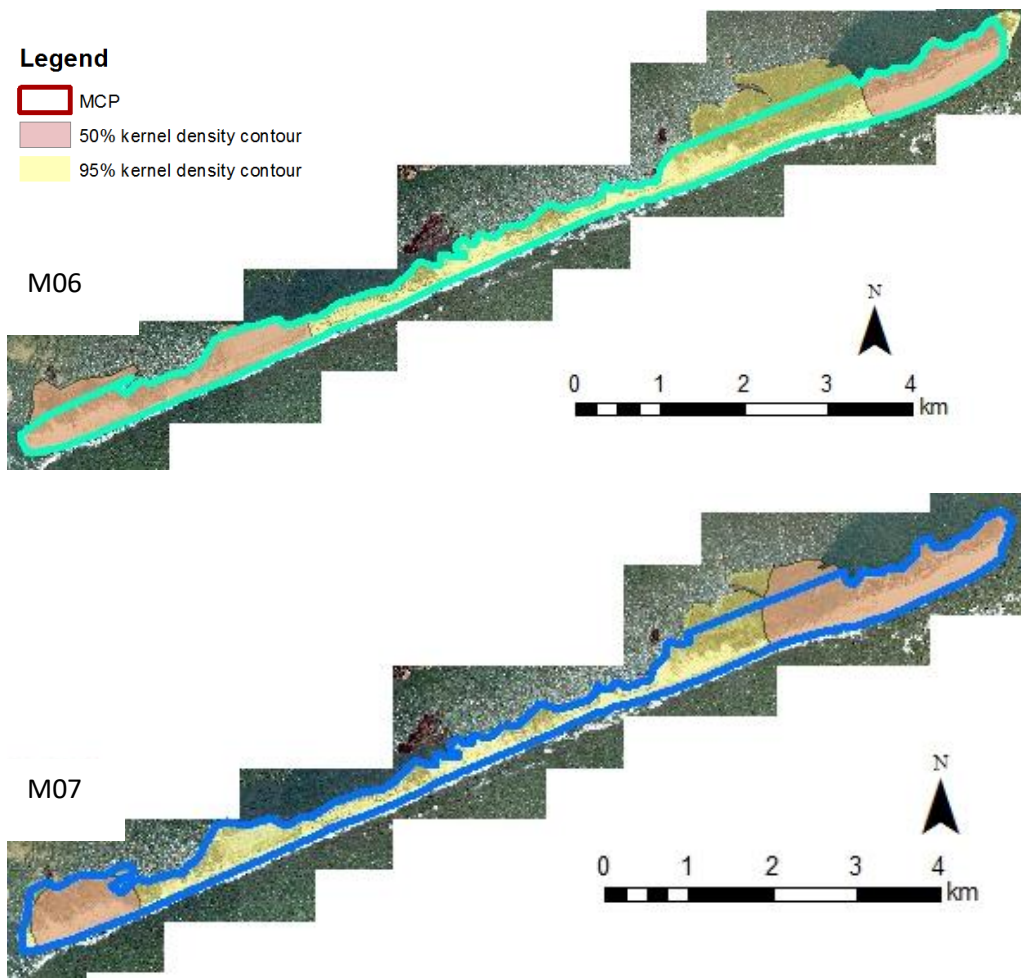


Figure 66. MCP, 50% and 95% kernel density contour for 2 male red foxes at Fire Island National Seashore (Old Inlet East) and Smith Point County Park in Fall-Winter 2015.

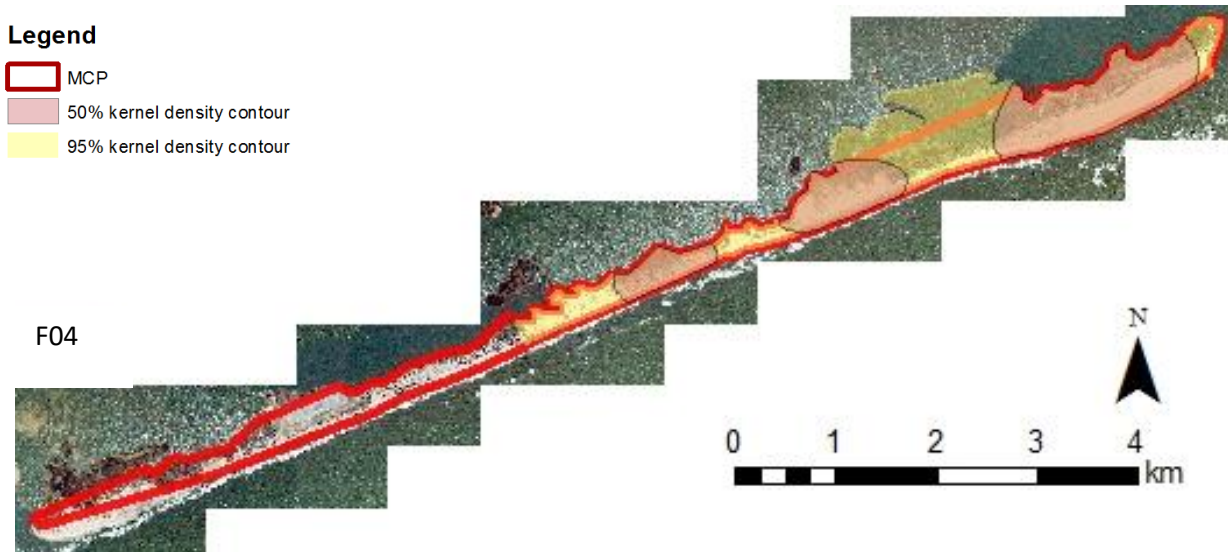


Figure 67. MCP, 50% and 95% kernel density contour for 1 female red fox at Fire Island National Seashore (Old Inlet East) and Smith Point County Park in Fall-Winter 2015.



Figure 68. 50% and 95% kernel density contours for all GPS-collared red foxes in Fall-Winter 2015 at Robert Moses State Park and Fire Island National Seashore Lighthouse Tract overlaid on top of each other.

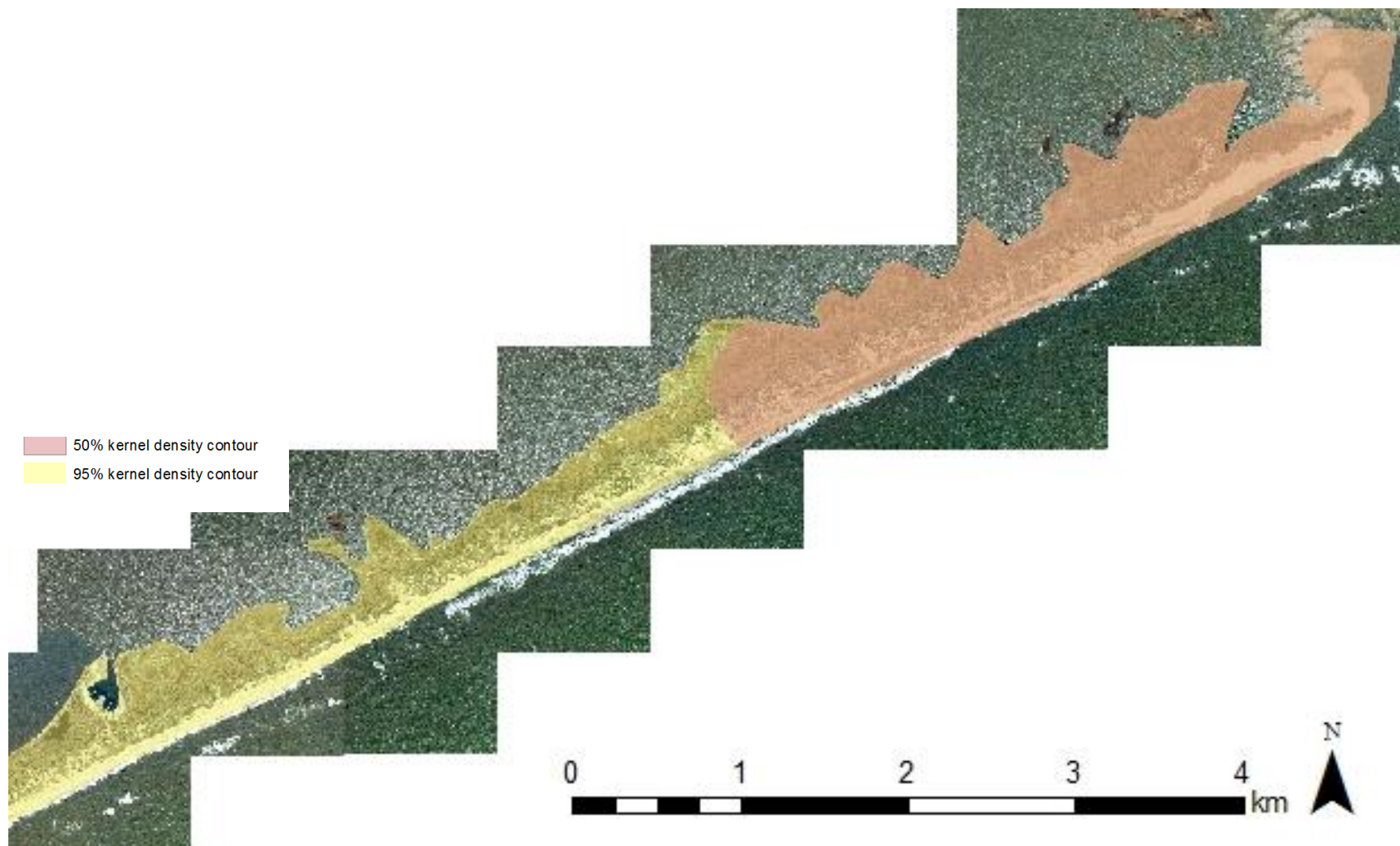


Figure 69. 50% and 95% kernel density contours for all GPS-collared red foxes in Fall-Winter 2015 at Fire Island National Seashore (Watch Hill to inlet) overlaid on top of each other.

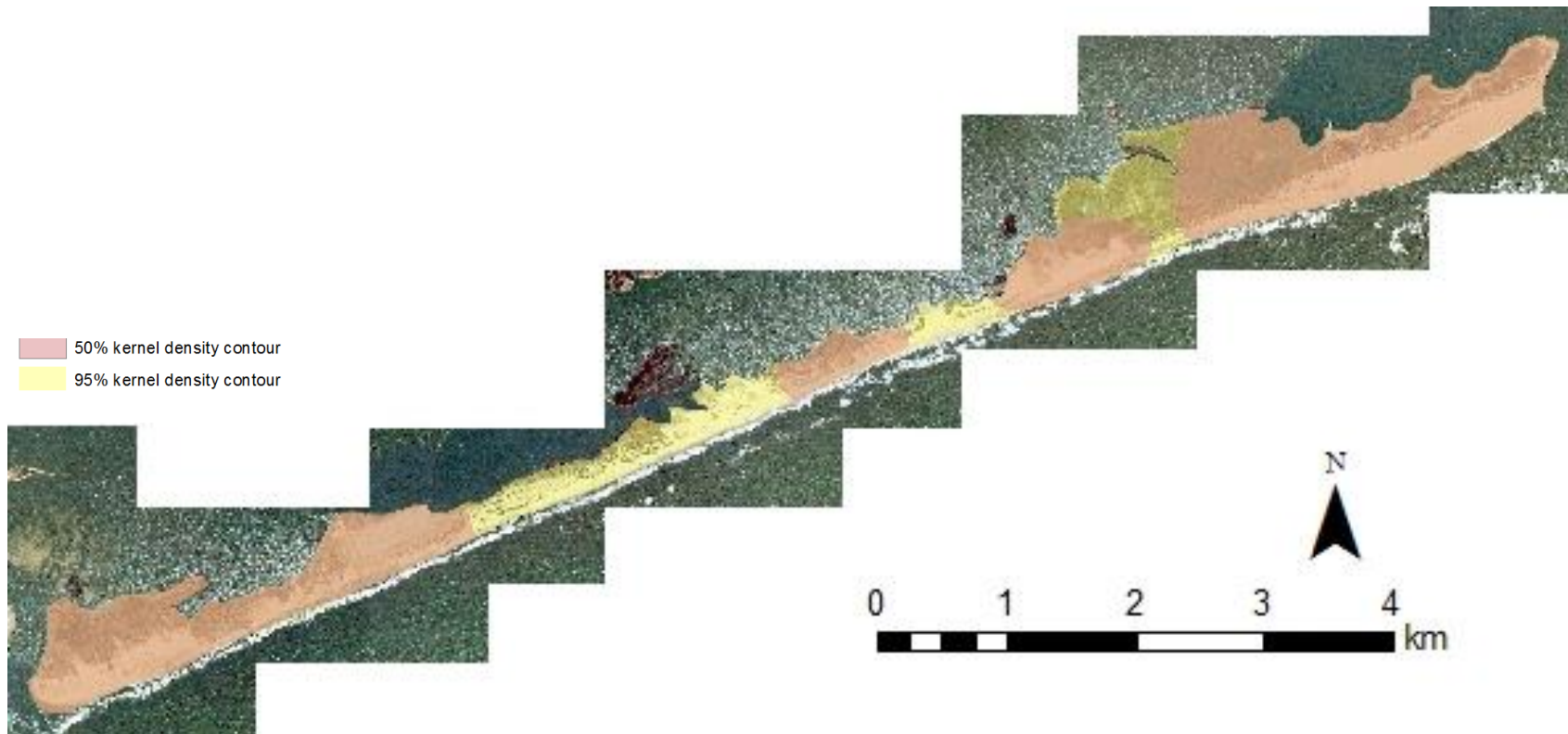


Figure 70. 50% and 95% kernel density contours for all GPS-collared red foxes in Fall-Winter 2015 at Fire Island National Seashore (Old Inlet East) and Smith Point County Park overlaid on top of each other.



Figure 71. MCP, 50% and 95% kernel density contours for 1 male red fox at Robert Moses State Park, Spring-Summer 2016.

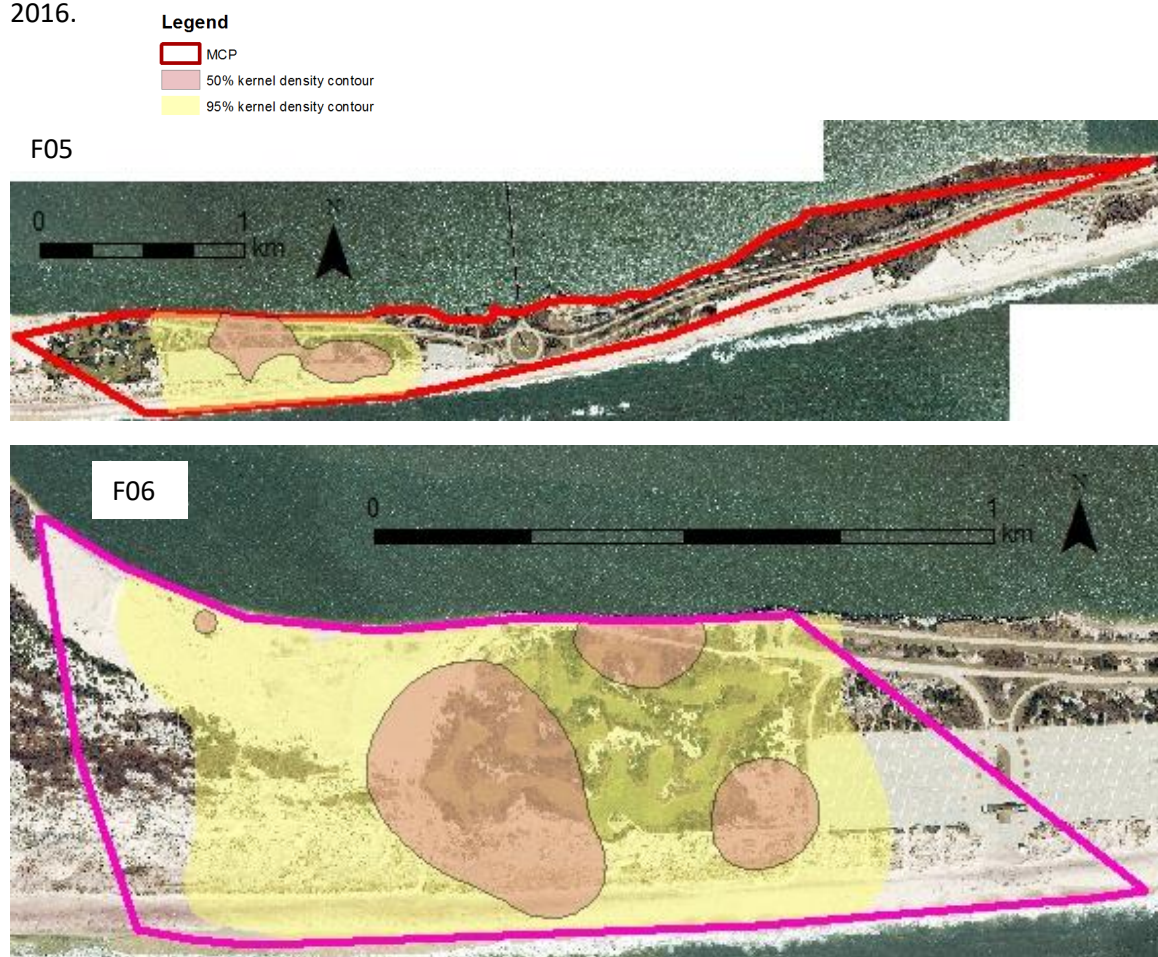


Figure 72. MCP, 50% and 95% kernel density contours for 2 female red foxes at Robert Moses State Park, Spring-Summer 2016.

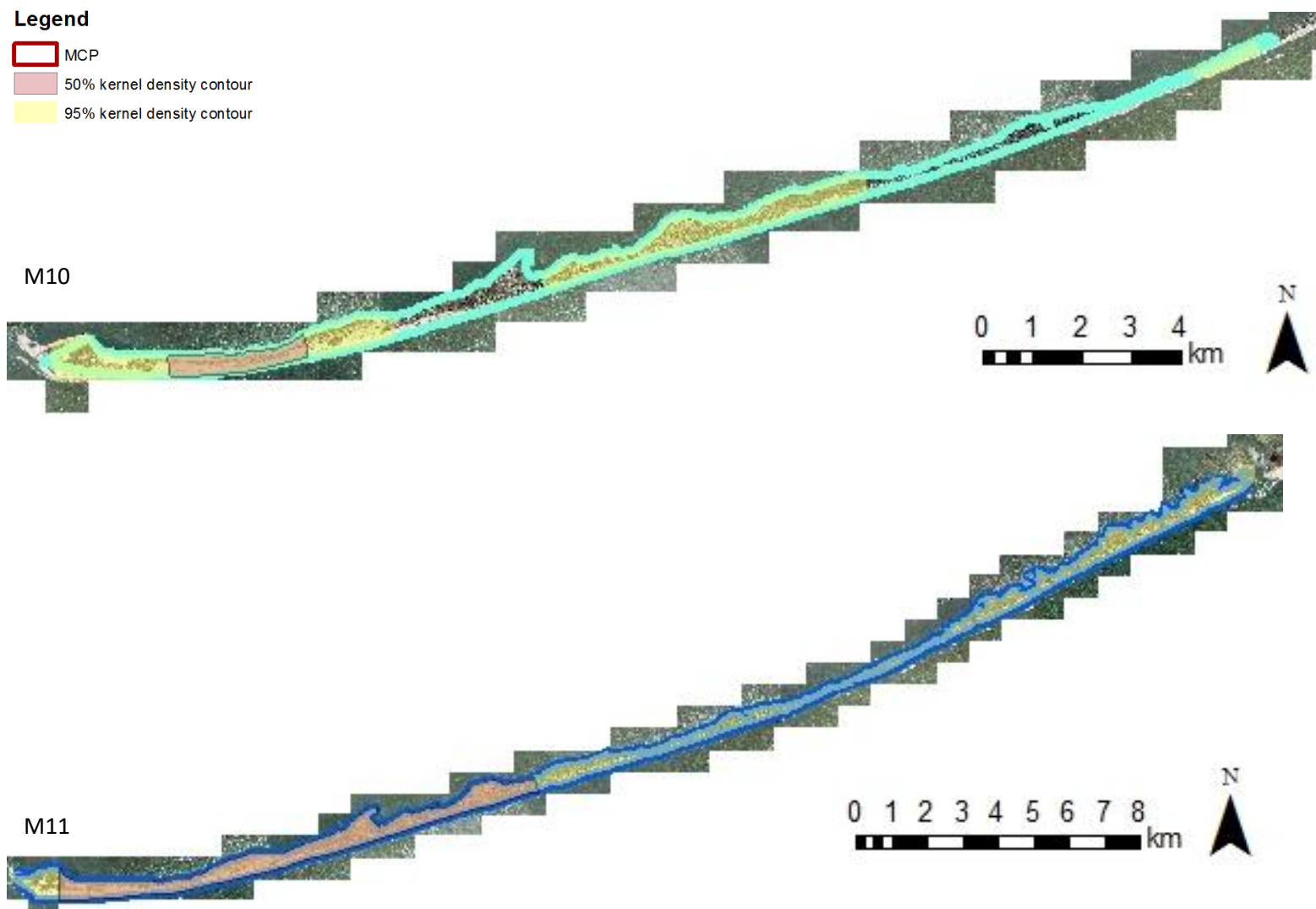


Figure 73. MCP, 50% and 95% kernel density contours for 2 male red foxes originally captured at Robert Moses State Park, Spring-Summer 2016.

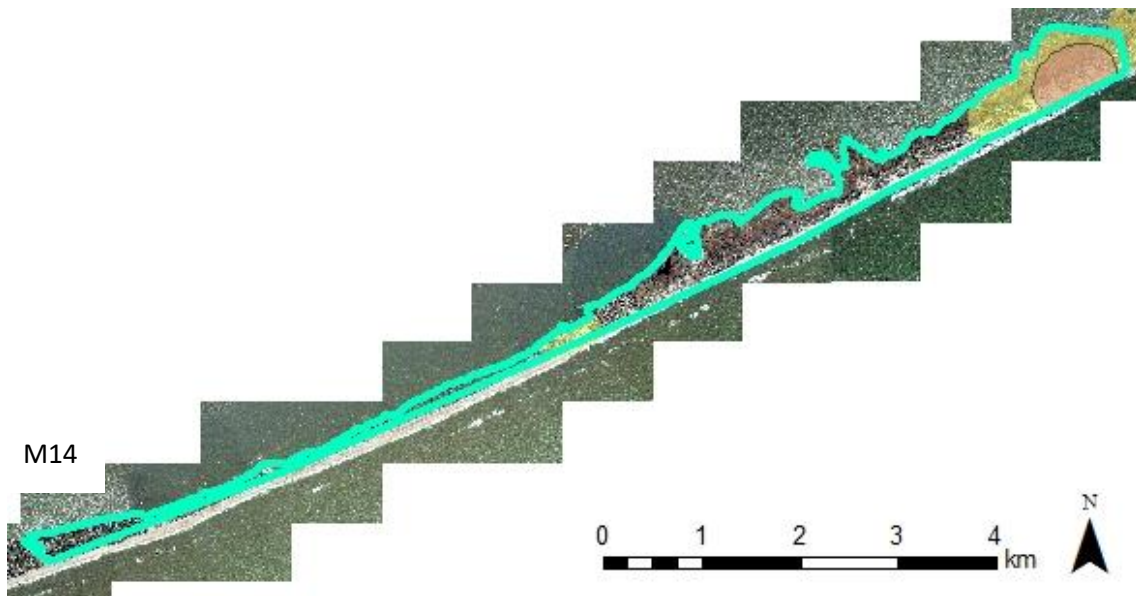
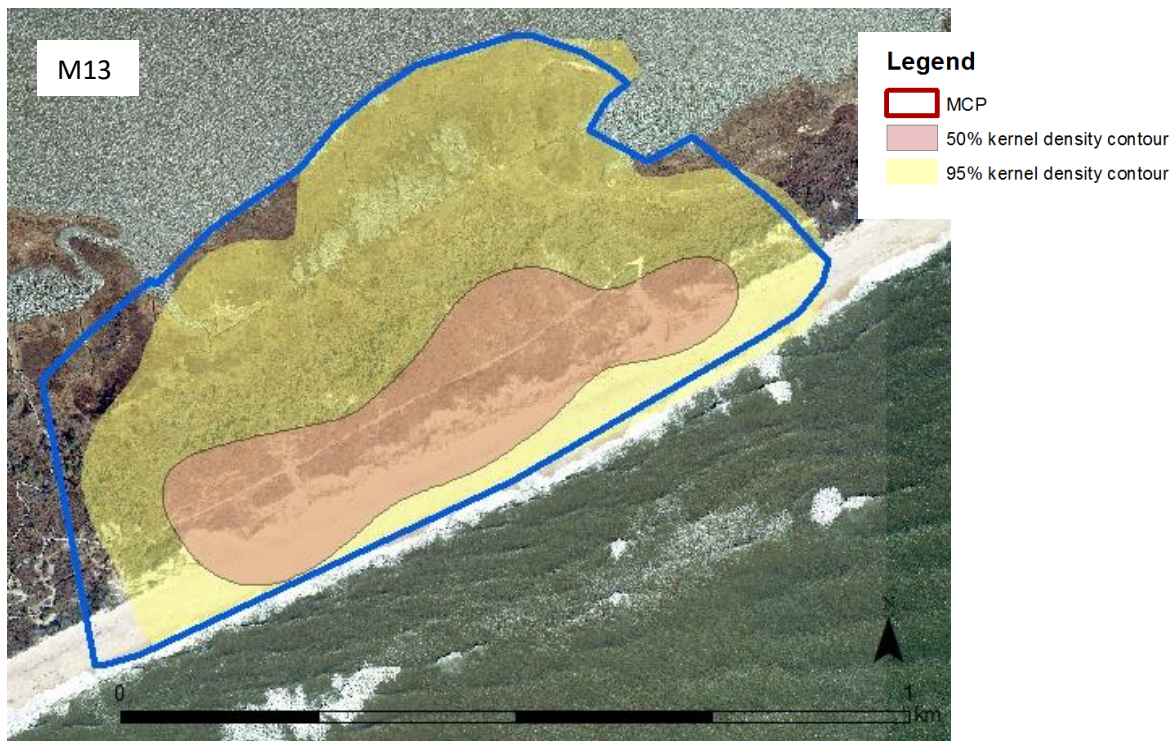
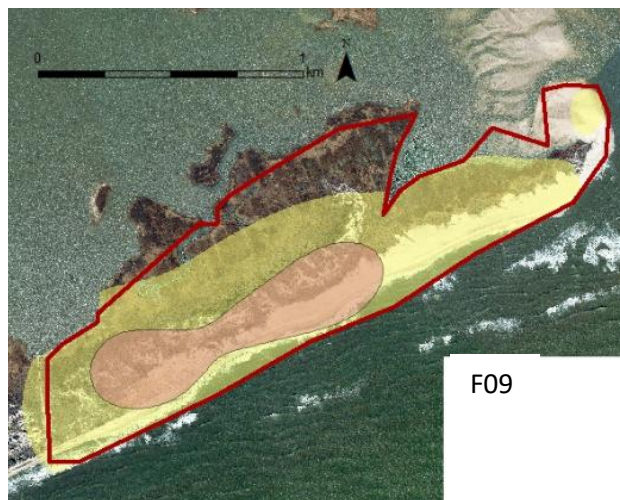
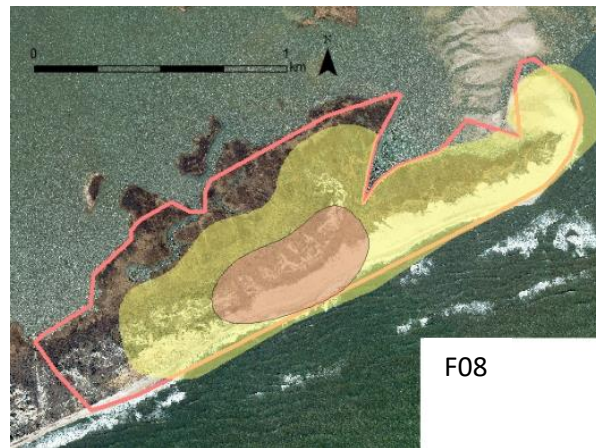


Figure 74. MCP, 50% and 95% kernel density contours for 2 male red foxes at Fire Island National Seashore, Spring-Summer 2016.



Legend

- MCP
- 50% kernel density contour
- 95% kernel density contour

Figure 75. MCP, 50% and 95% kernel density contours for 3 female red foxes at Fire Island National Seashore, Spring-Summer 2016.

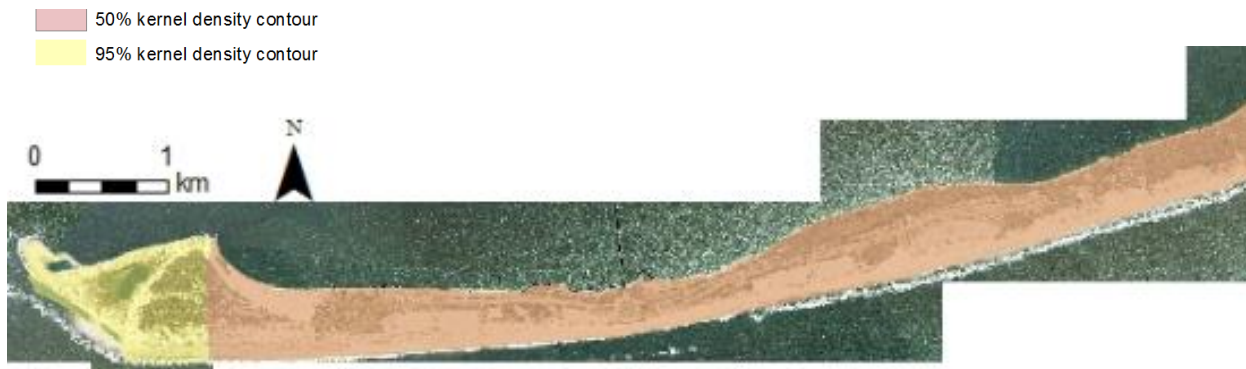


Figure 76. 50% and 95% kernel density contours for all GPS-collared red foxes in Spring-Summer 2016 at Robert Moses State Park overlaid on top of each other.

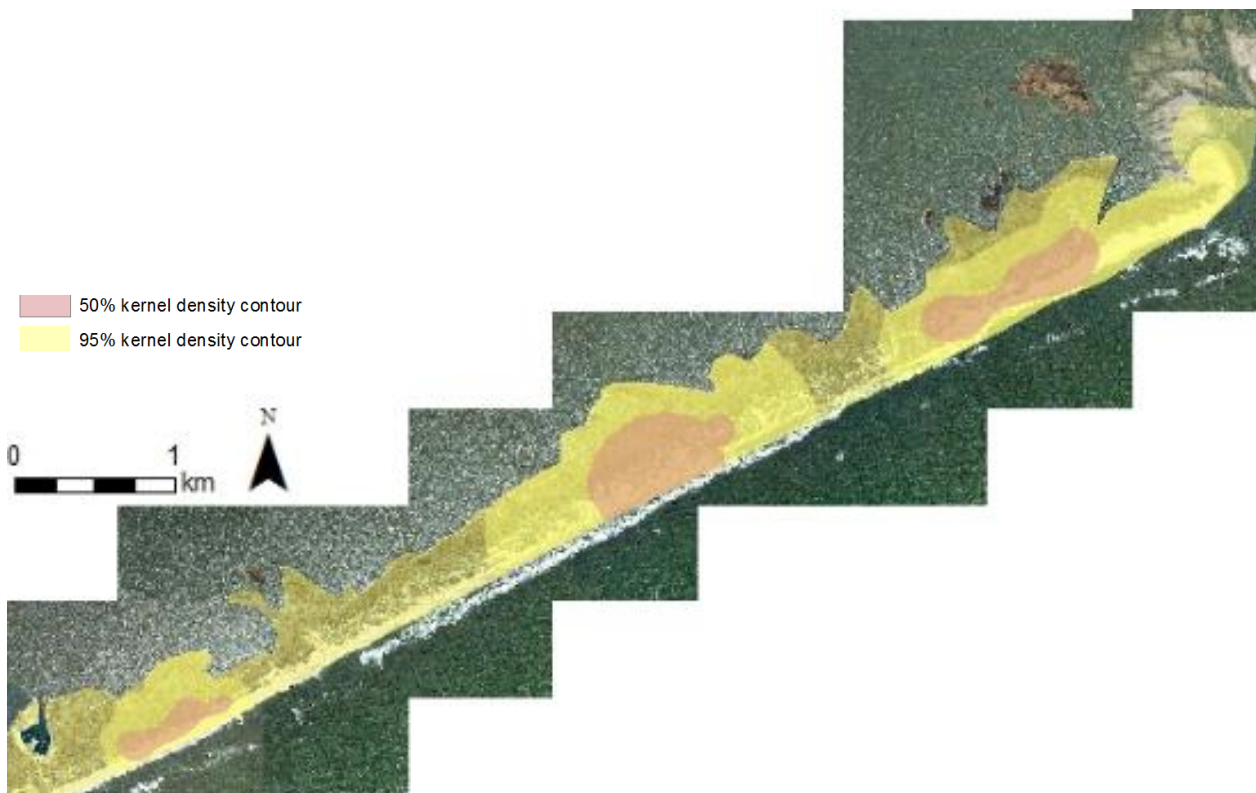


Figure 77. 50% and 95% kernel density contours for all GPS-collared red foxes in Spring-Summer 2016 at Fire Island National Seashore (west of Old Inlet) overlaid on top of each other.

Appendix C. Spatial response of red foxes to nest enclosure setup, pipping, and hatching

Maps of fox locations during high frequency location sampling surrounding nest enclosure set-up and estimated nest pipping and hatching dates are shown below. For various logistical and biological reasons described in the report, our high frequency data collection periods at some nests did not line up exactly with the events that we had scheduled them around. We used the highest location frequency collected surrounding the actual dates, using 2-2.5 days of location data before and after the target event to examine fine-scale spatial response. Thus, the location frequency and number of locations shown may differ among time periods and across maps.

Spring-Summer 2017 monitoring period:

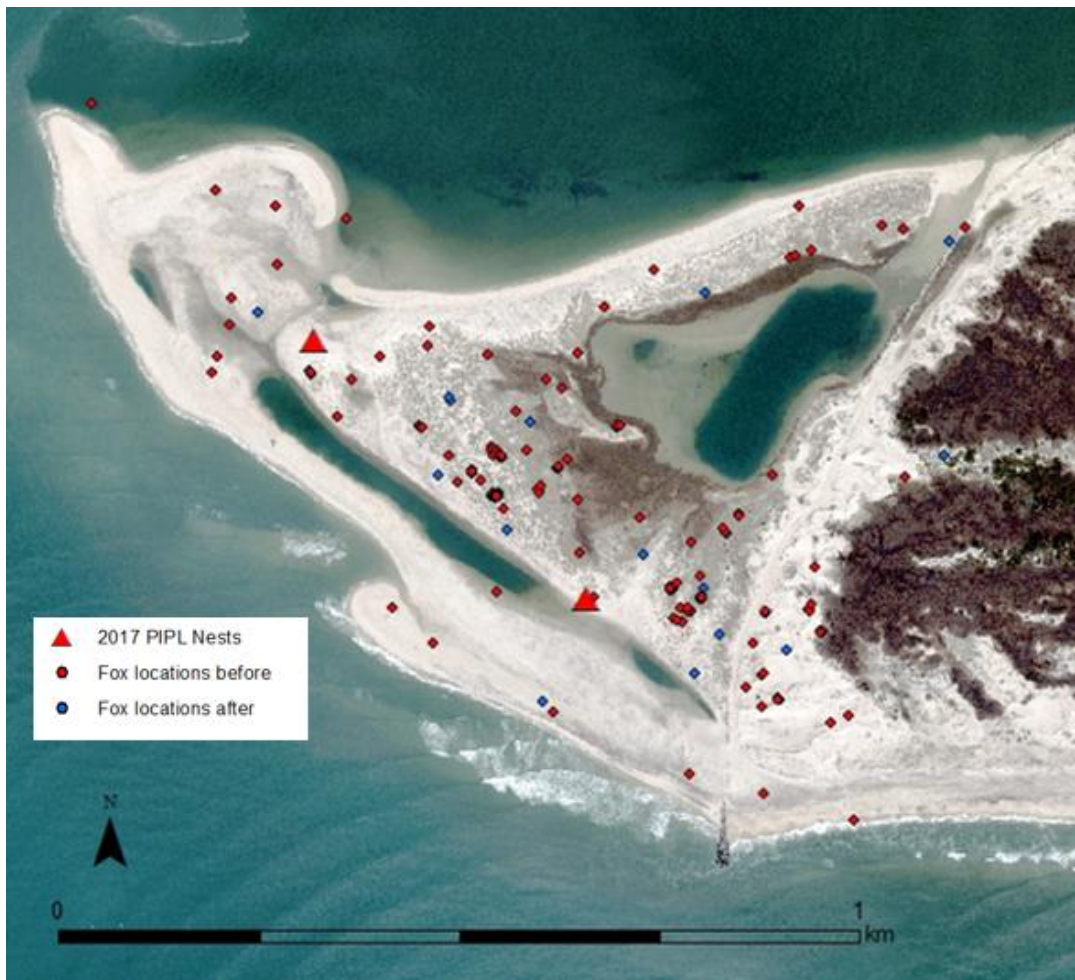


Figure 78. Locations for F11 before and after nest enclosure at Robert Moses State Park, PIPL nests 203 and 204.

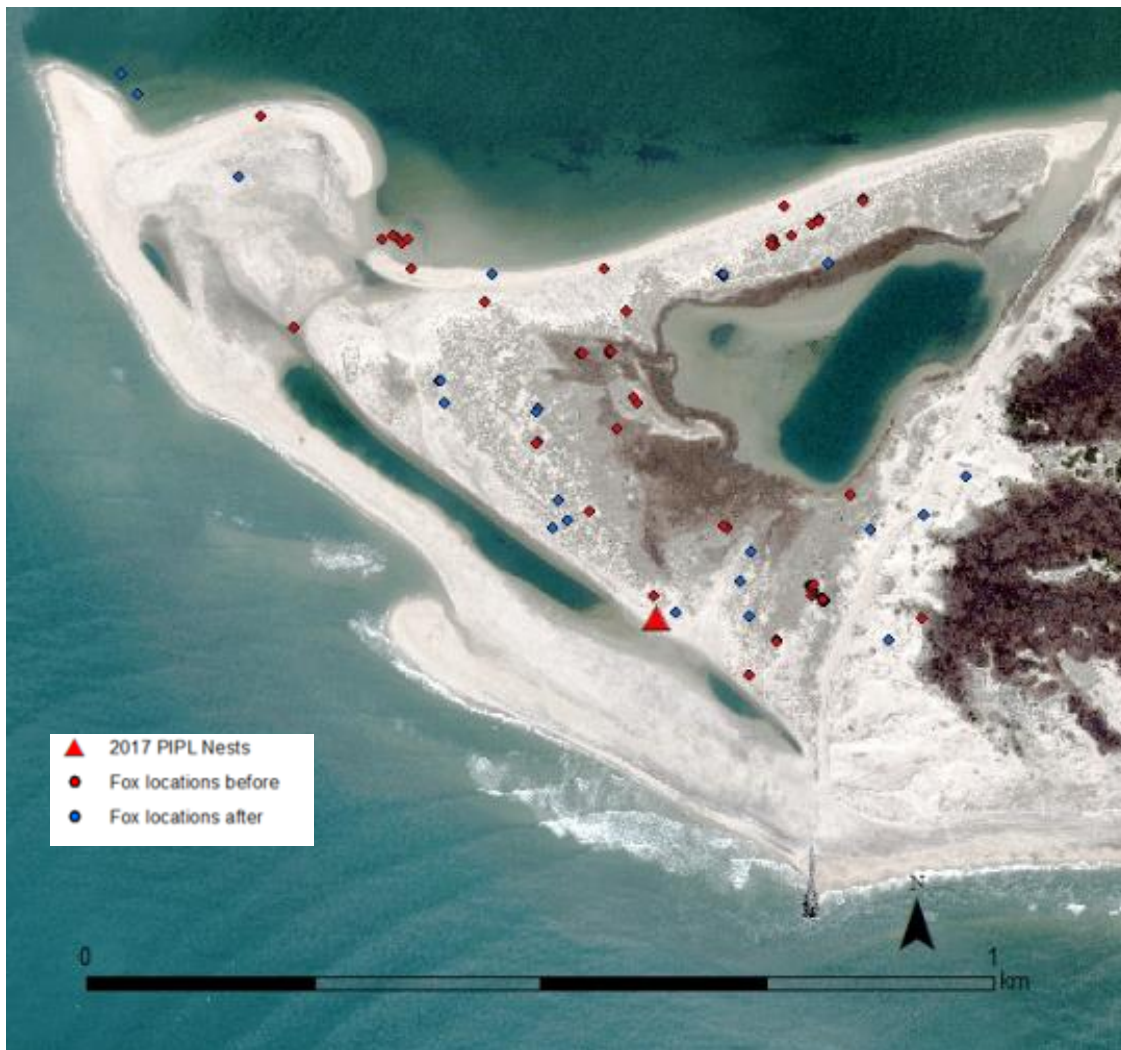


Figure 79. Locations for F11 before and after nest hatch date at Robert Moses State Park, PIPL nest 203 (nest 204 failed).

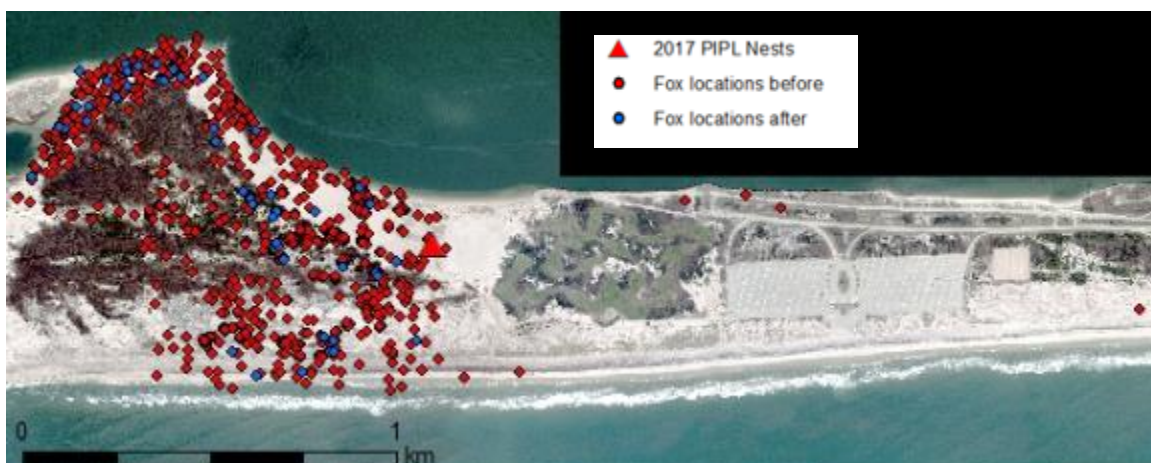


Figure 80. Locations for F12 before and after nest exclosure at Robert Moses State Park, PIPL nest 205.

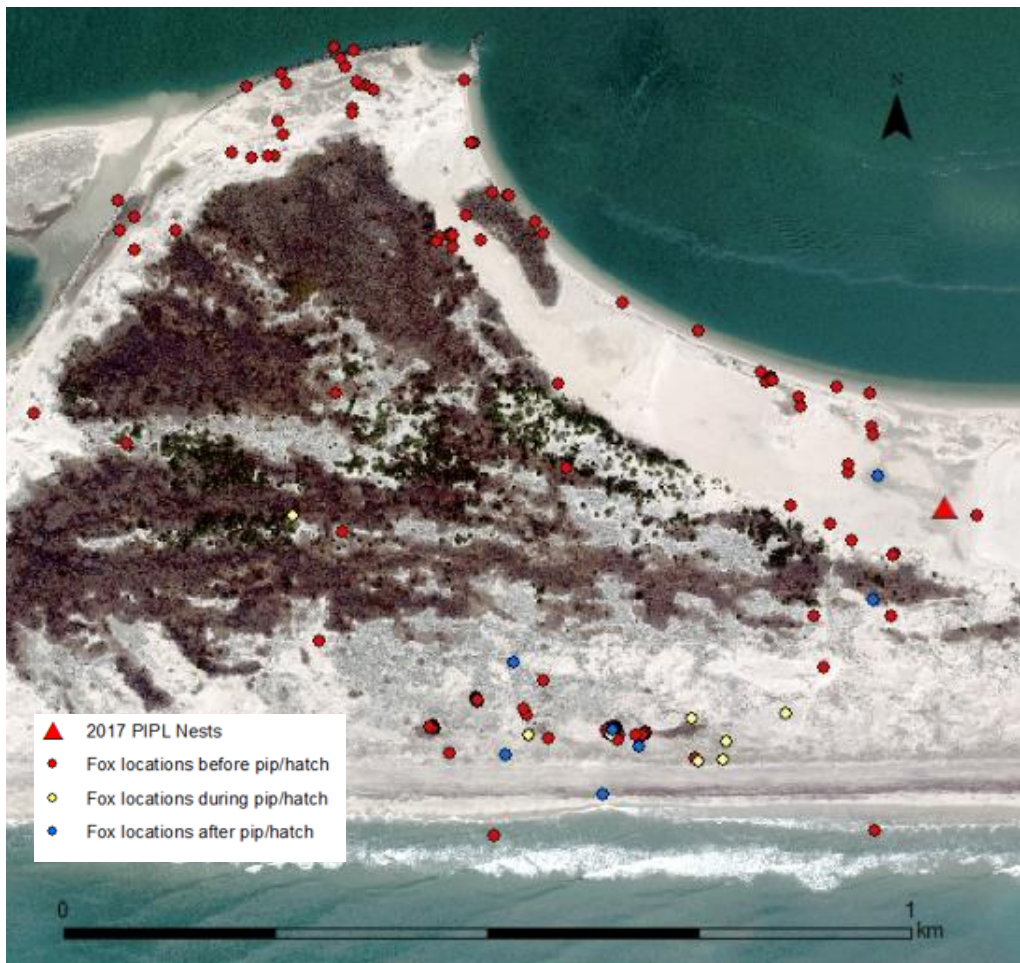


Figure 81. Locations for F12 before, during, and after nest pipping/hatching at Robert Moses State Park, PIPL nest 205.

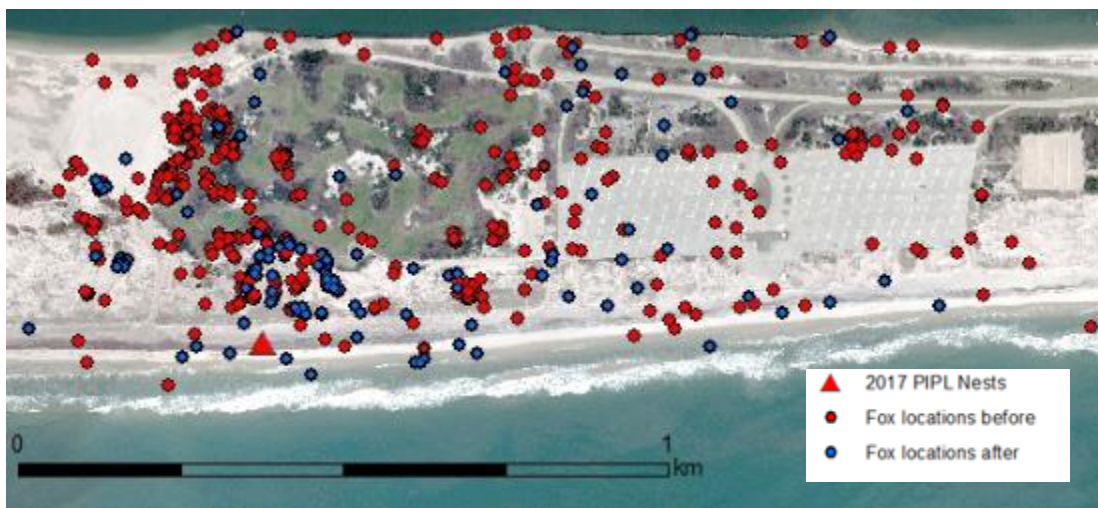


Figure 82. Locations for M08 before and after nest exclosure at Robert Moses State Park, PIPL nest 214.

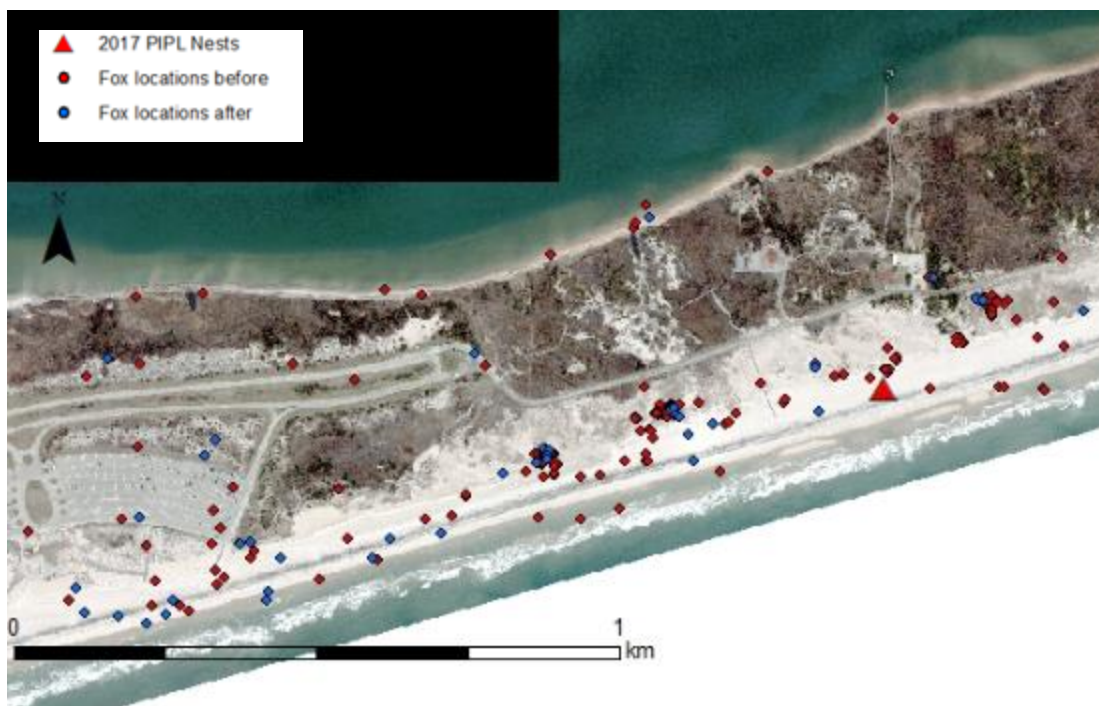


Figure 83. Locations for M16 before and after nest exclosure at Robert Moses State Park, PIPL nest 211.

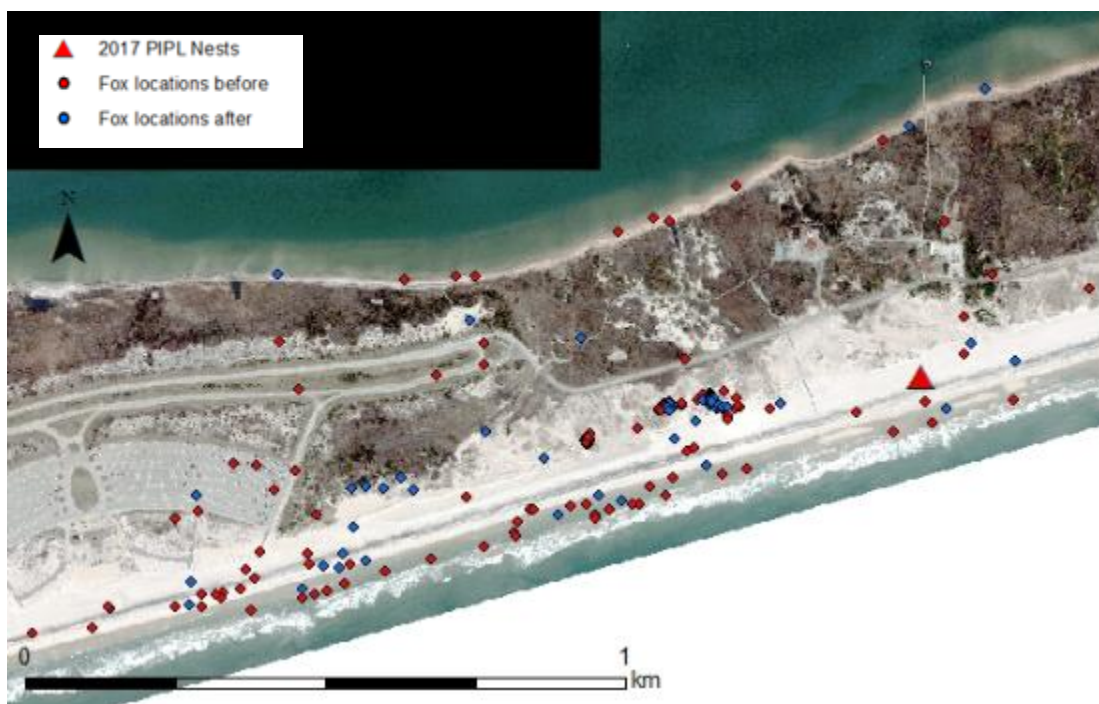


Figure 84. Locations for F13 before and after nest exclosure at Robert Moses State Park, PIPL nest 211.

Spring-Summer 2016 monitoring period:

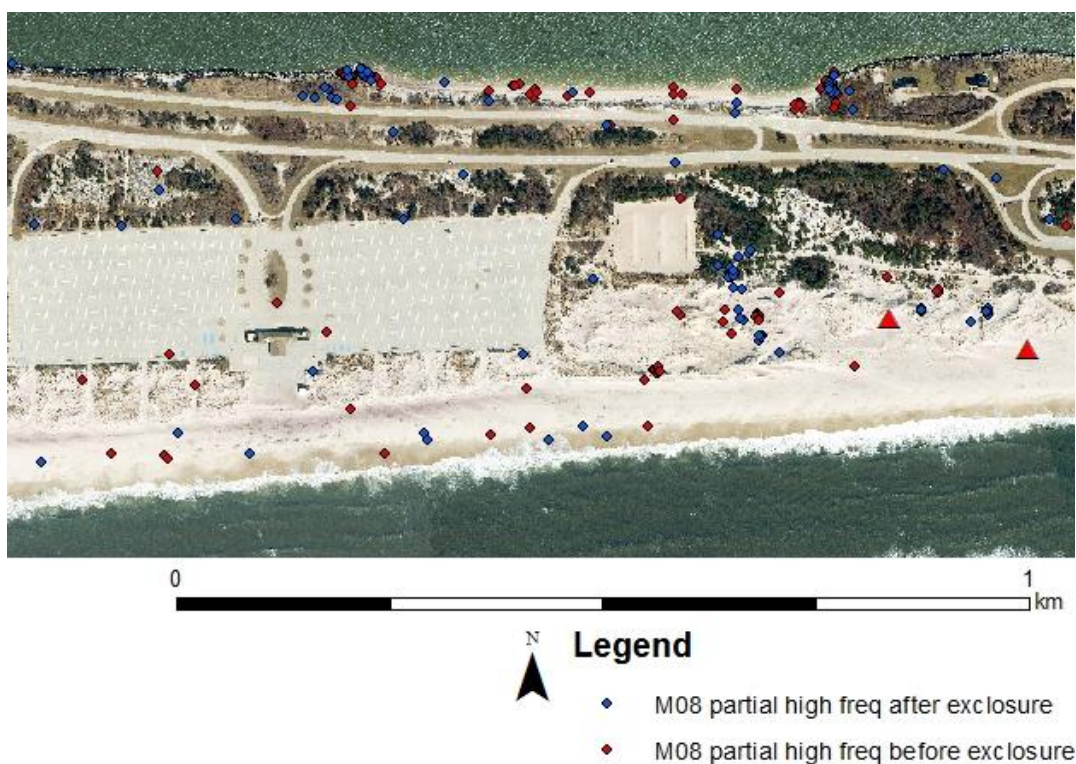


Figure 23. Locations for fox M08 before and after nest enclosure at Robert Moses State Park, PIPL nest 211.



Figure 86. Locations for fox M08 before, during, and after nest pipping/hatching at Robert Moses State Park, PIPL nest 211.

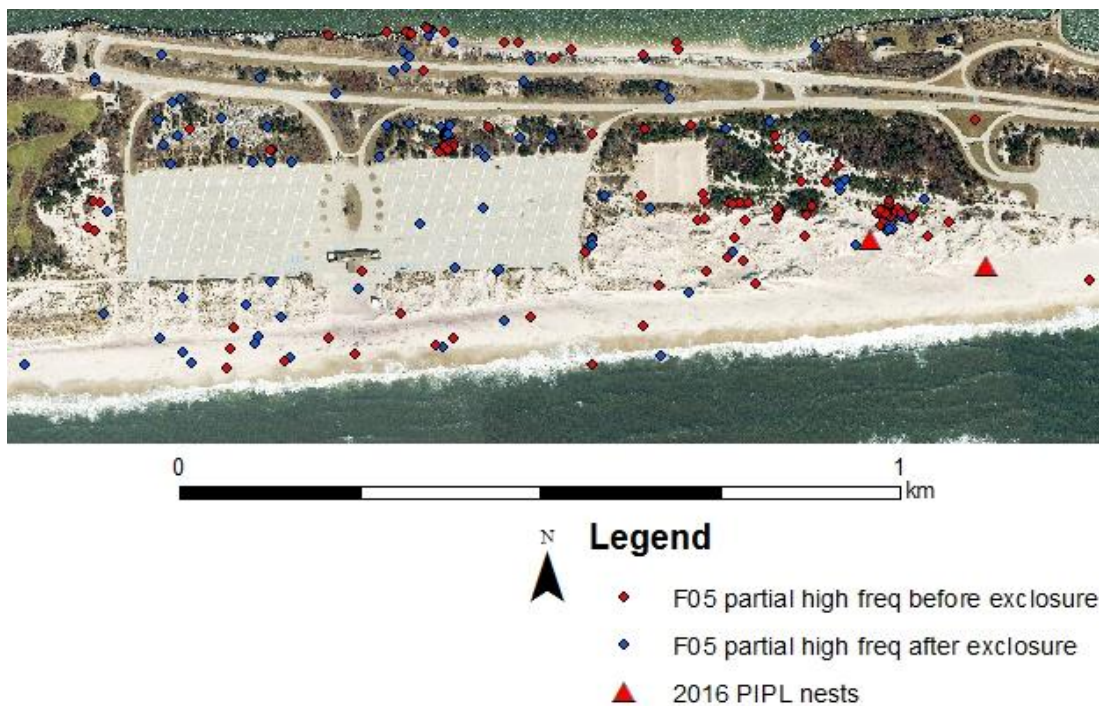


Figure 87. Locations for fox F05 before and after nest exclosure at Robert Moses State Park, PIPL nest 211.

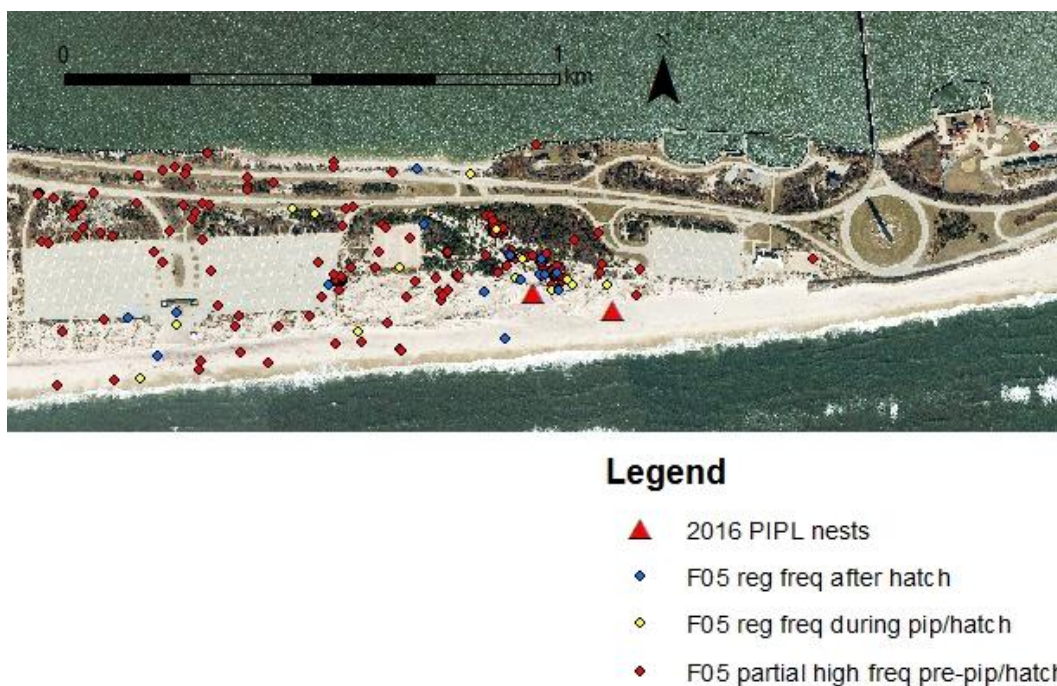


Figure 88. Locations for fox F05 before, during, and after nest pipping/hatching at Robert Moses State Park, PIPL nest 211.

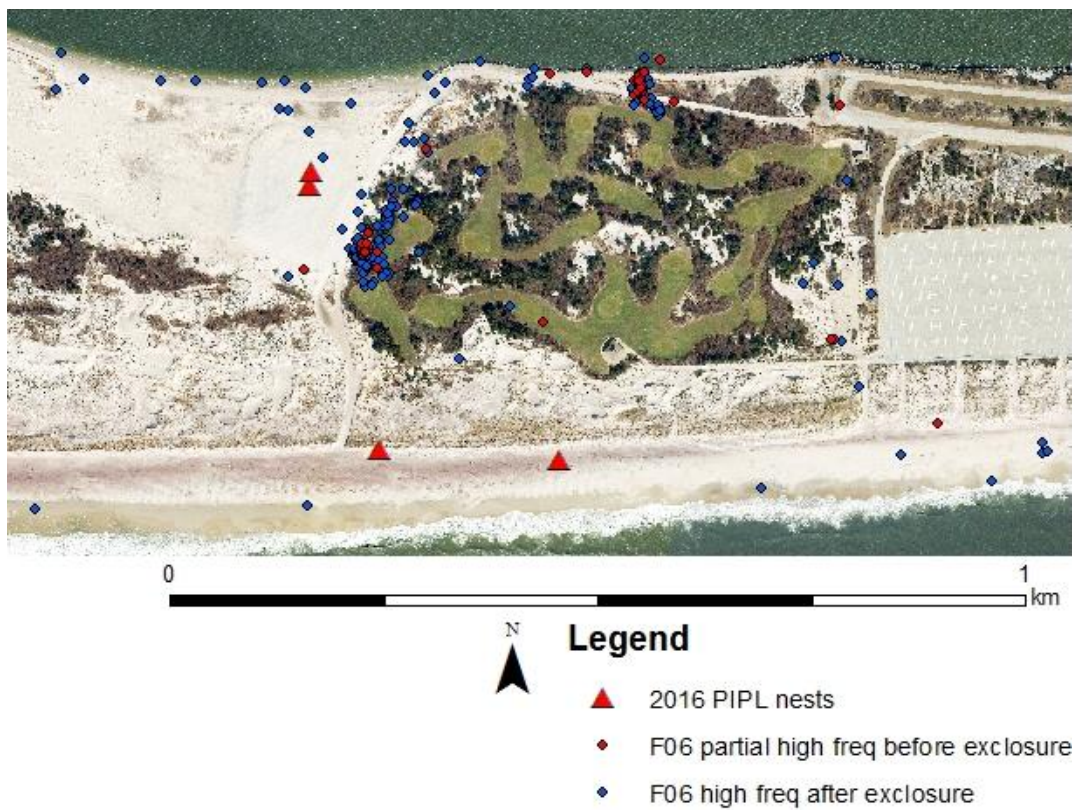


Figure 89. Locations for fox F06 before and after nest exclosure at Robert Moses State Park, PIPL nests 204 and 205 (on southern beachfront).



Figure 90. Locations for fox F06 before, during, and after nest pipping/hatching at Robert Moses State Park, PIPL nests 204 and 205 (on southern beachfront).

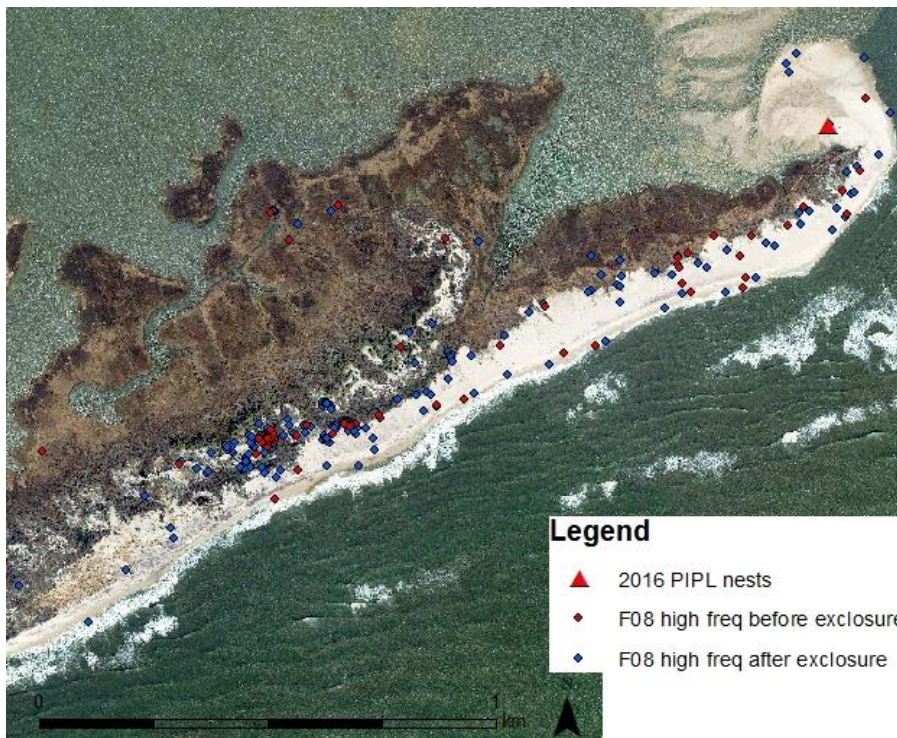


Figure 91. Locations for fox F08 before and after nest exclosure at Fire Island National Seashore, PIPL nest 32.

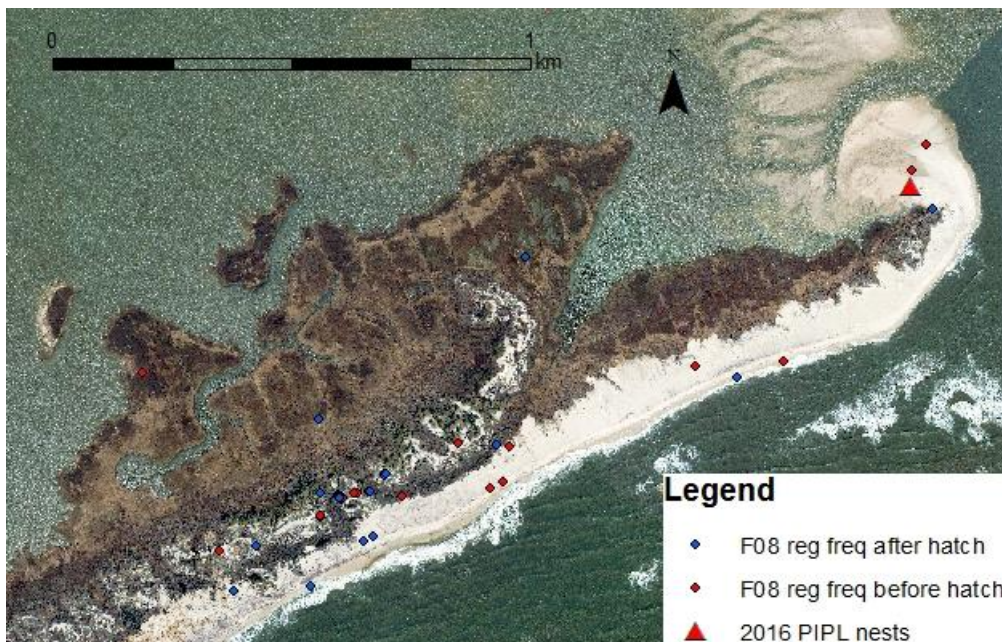


Figure 92. Locations for fox F08 before and after nest hatching at Fire Island National Seashore, PIPL nest 32.

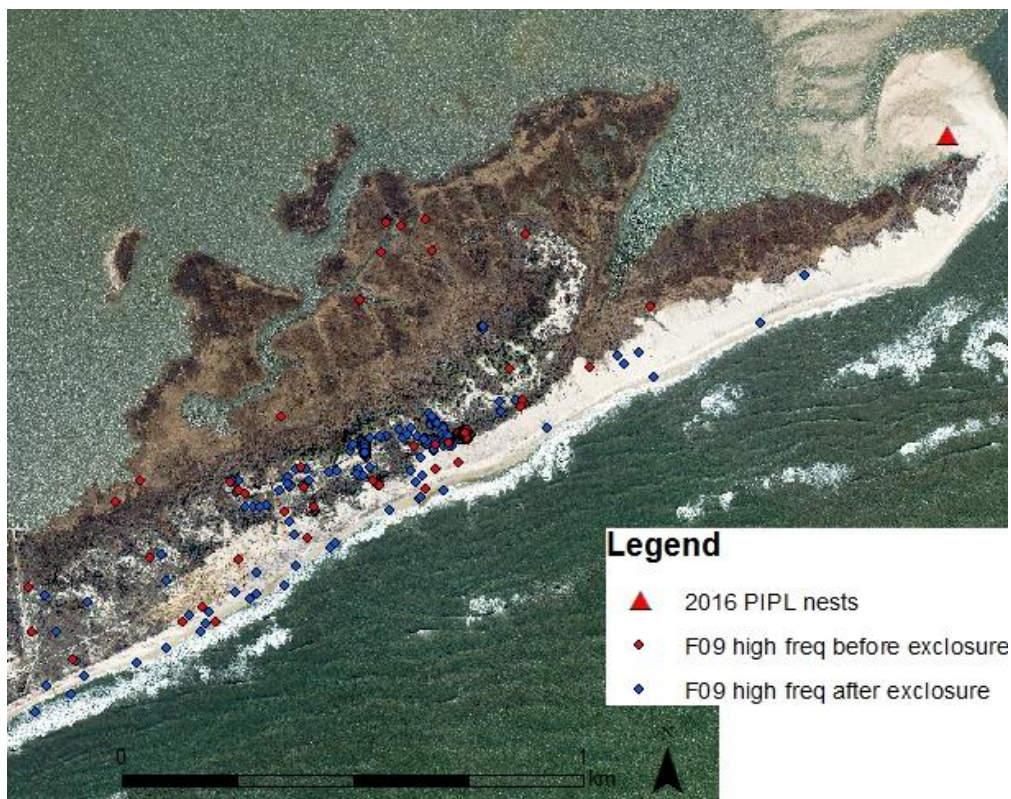


Figure 93. Locations for fox F09 before and after nest exclosure at Fire Island National Seashore, PIPL nest 32.

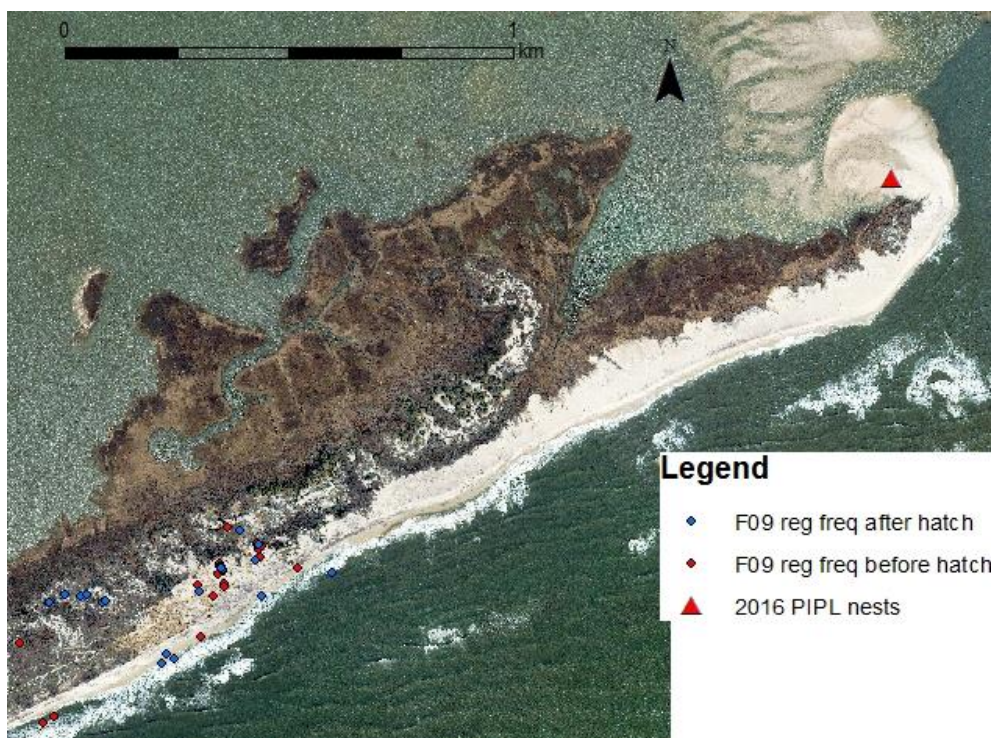


Figure 94. Locations for fox F09 before and after nest hatching at Fire Island National Seashore, PIPL nest 32.

Appendix D. FIMI Restoration Area Performance

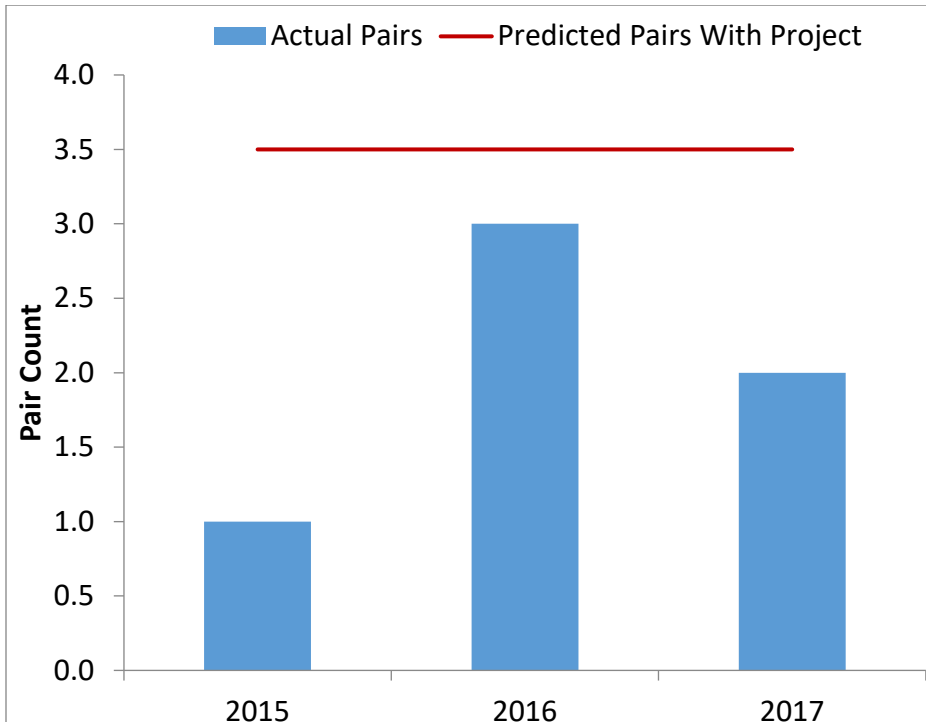


Figure 95. Robert Moses State Park Fields 2 and 3 restoration area performance based on predicted (USFWS 2014) symbolized in the red horizontal bar, and actual pairs symbolized as blue vertical bars.

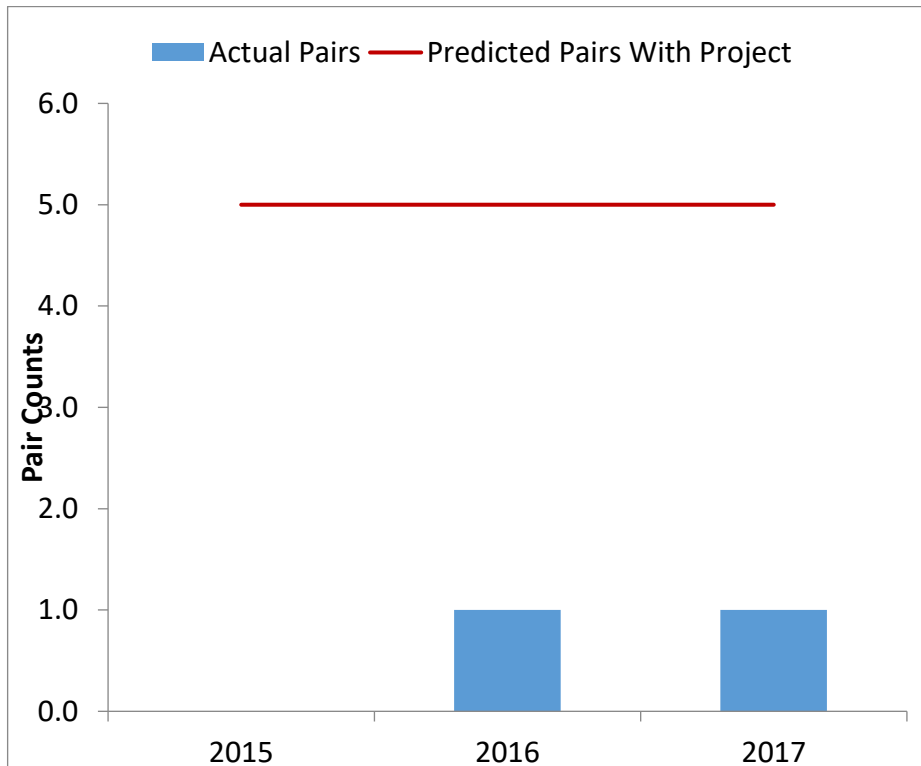


Figure 96. Lighthouse beach restoration area performance, based on predicted (USFWS 2014), symbolized by the red horizontal bar and actual pairs, symbolized by the blue vertical bars.

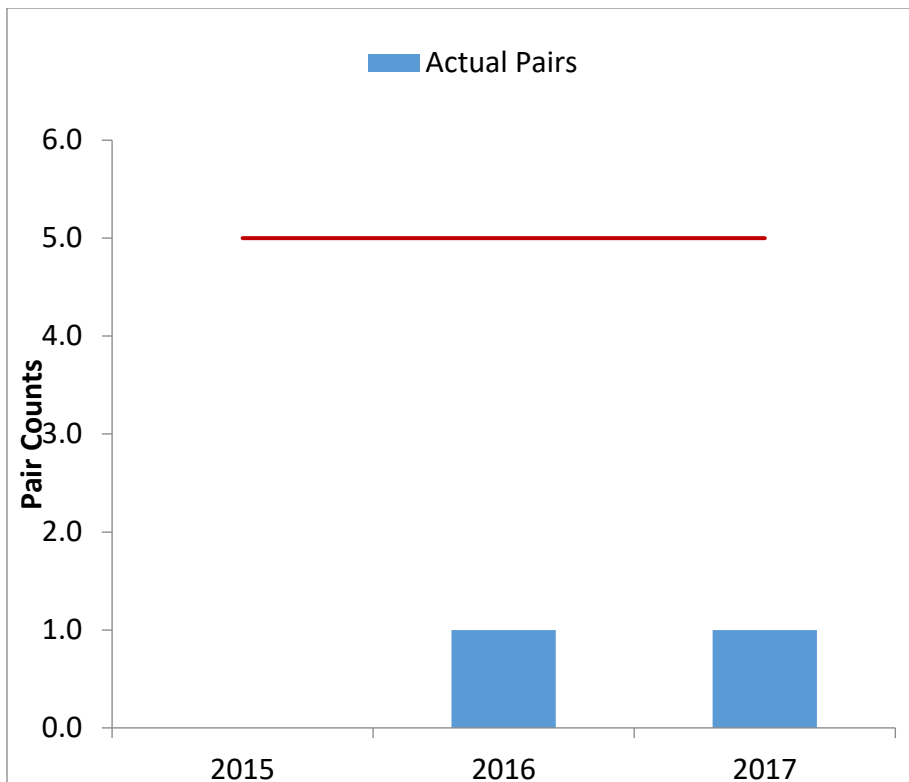


Figure 97. Lighthouse Beach Restoration Area Performance based on predicted (USFWS 2014), symbolized by the red horizontal bar and actual pairs, symbolized by the blue vertical bars.

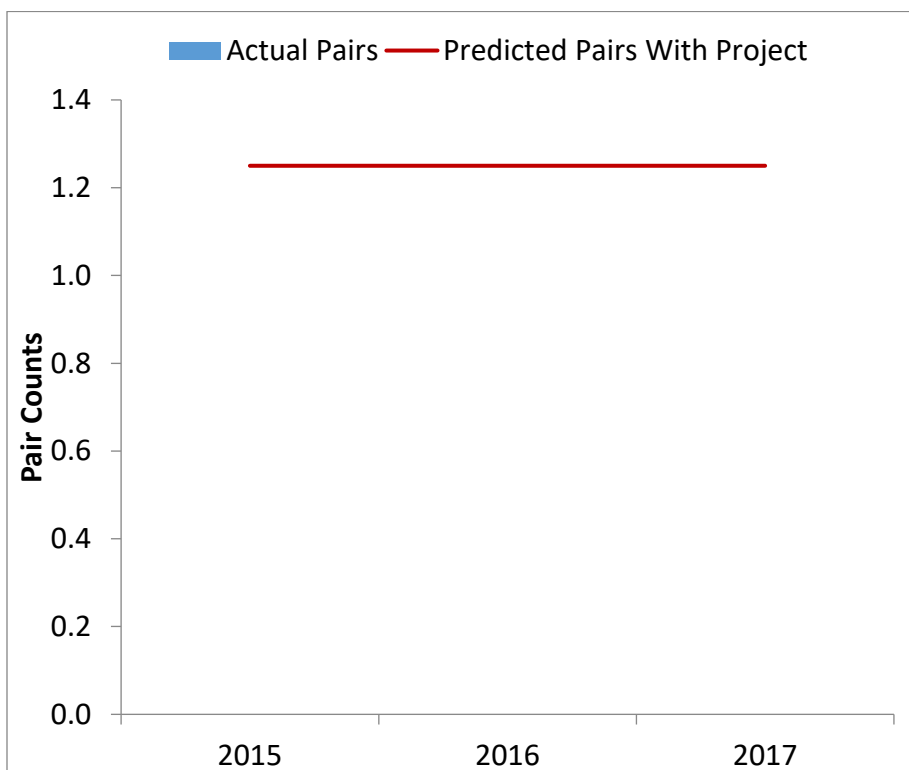


Figure 98. Robbins Rest Restoration Area performance based on predicted (USFWS 2014), symbolized by the red horizontal bar and actual pairs, symbolized by the blue vertical bars.

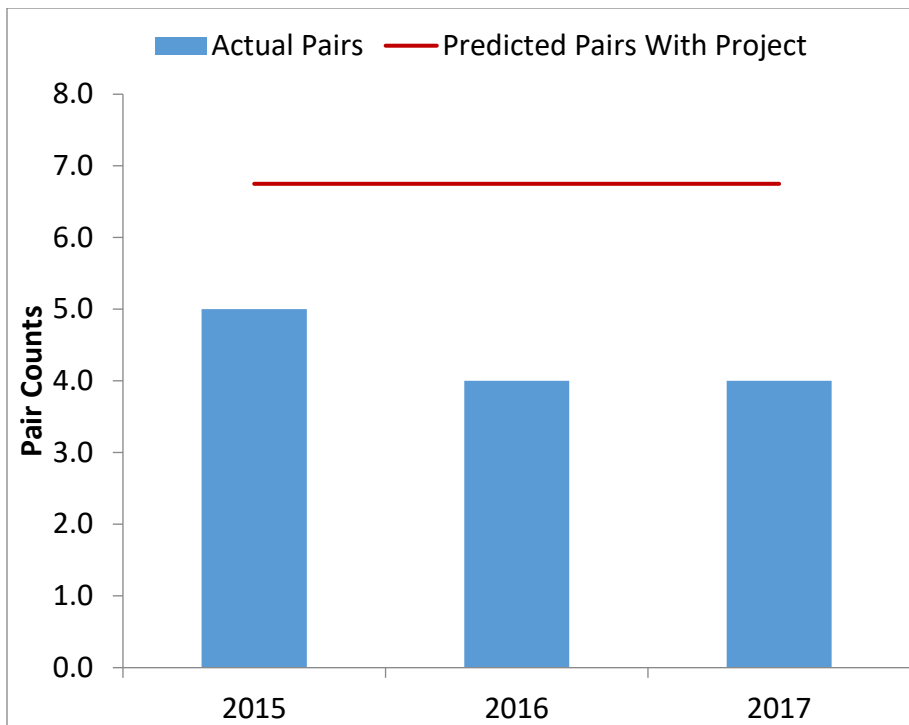


Figure 99. Pattersquash Overwash Restoration Area performance based on predicted (USFWS 2014), symbolized by the red horizontal bar and actual pairs, symbolized by the blue vertical bars.

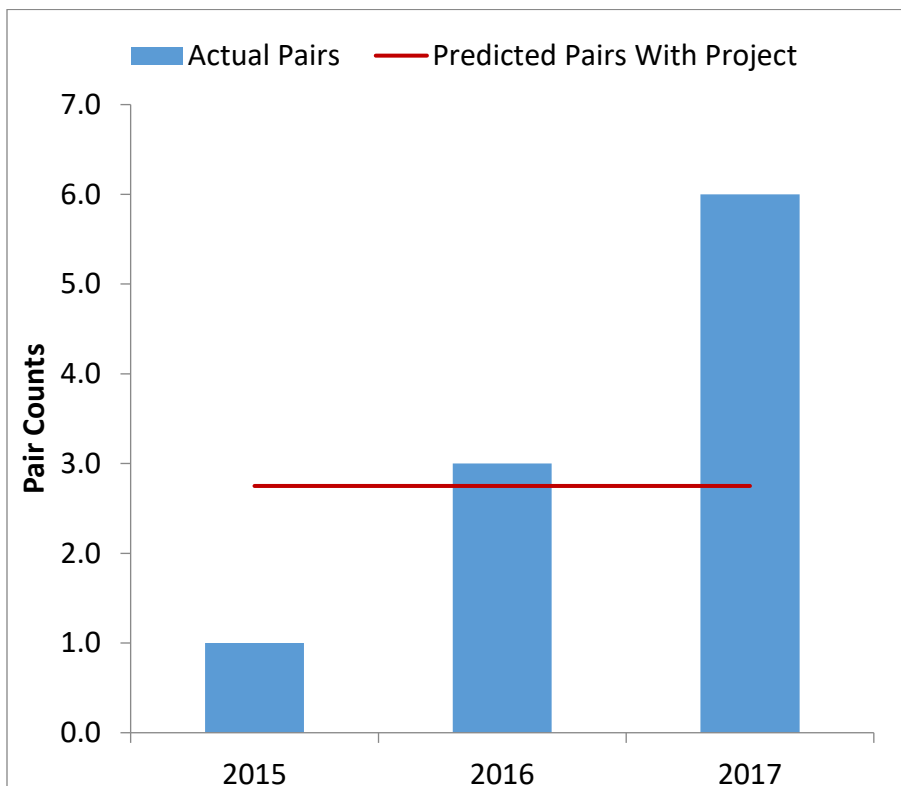


Figure 100. Narrow Bay West Restoration Area performance based on predicted (USFWS 2014), symbolized by the red horizontal bar and actual pairs, symbolized by the blue vertical bars.

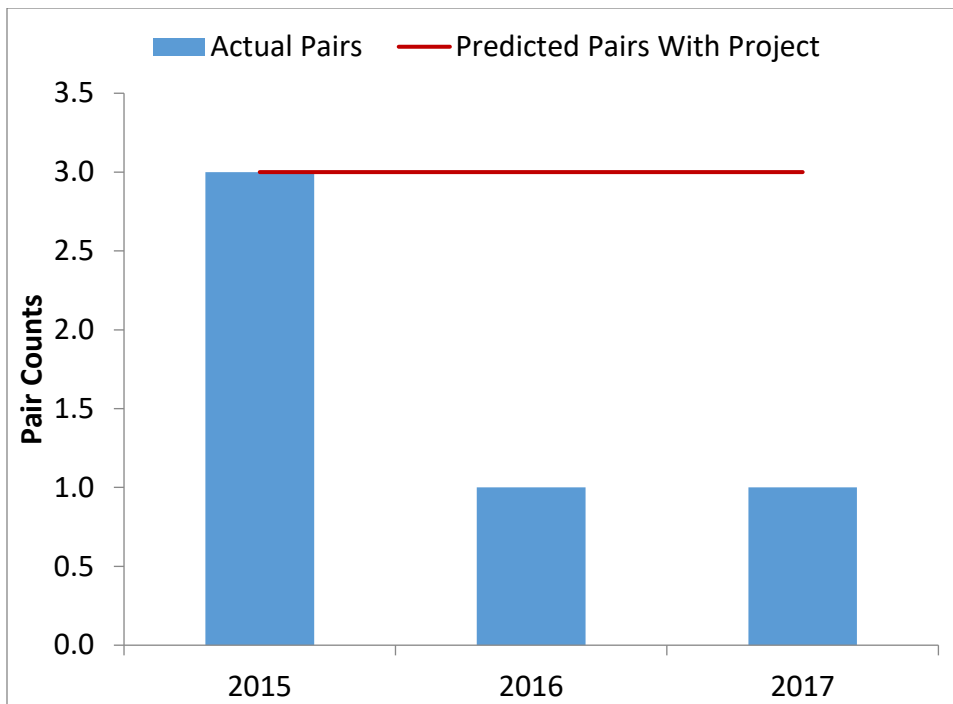


Figure 101. New Made Dredge Spoil Area Restoration Area performance based on predicted (USFWS 2014), symbolized by the red horizontal bar and actual pairs, symbolized by the blue vertical bars.

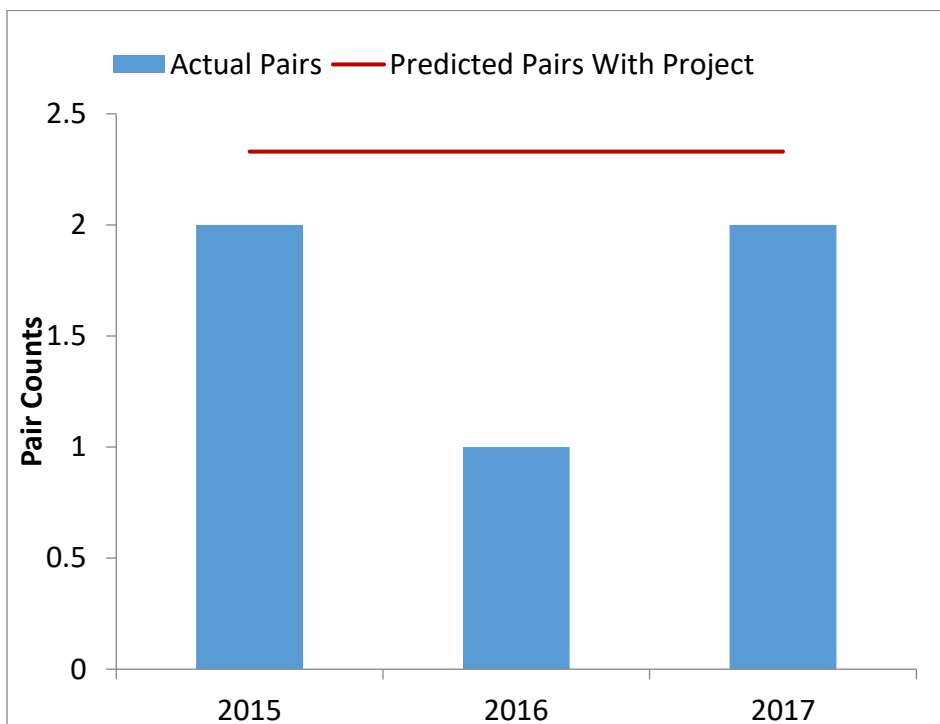


Figure 102. Narrow Bay East Restoration Area performance based on predicted (USFWS 2014), symbolized by the red horizontal bar and actual pairs, symbolized by the blue vertical bars.

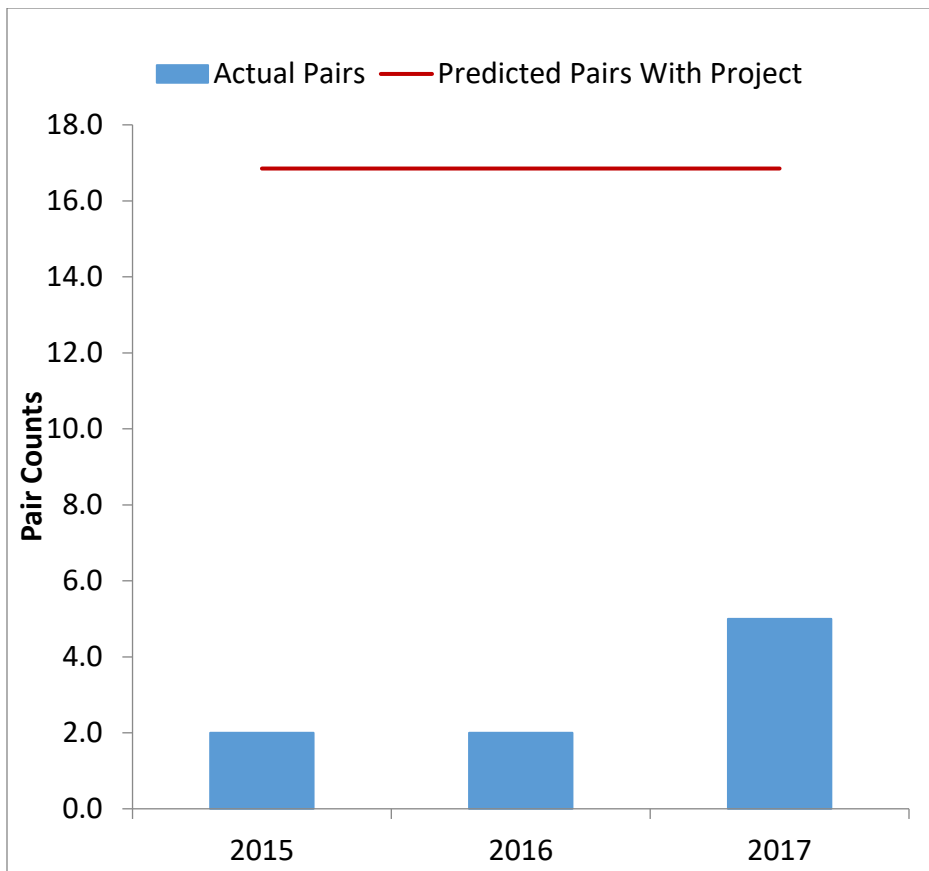


Figure 103. Great Gun Restoration Area performance based on predicted (USFWS 2014), symbolized by the red horizontal bar and actual pairs, symbolized by the blue vertical bars.