

2018 Annual Report

Effectiveness monitoring and targeted studies in support of FIMI conservation measures

Piping Plover and Red Fox Monitoring on Fire Island and Westhampton Island, New York

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

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Robert Smith, Steve Papa, Christopher Allen, Mike Bilecki, Lindsay Ries, Kaetlyn Jackson, Jordan Raphael, Nick Gibbons, Diana Sanford, Suffolk County plover stewards, Annie McIntyre, Timothy Byrne, Robert Moses State Park staff, Emily Ronis, Ruici Ong, L. Granger, C. Robertson, S. Sullivan, R. Winters, J. Deans, B. Hardt, A. Zeller, C. Helmke, E. Zawatski, M. Carroll, E. Swiecki, K. Attinger, J. Hall, C. Howell, R. Moylan, A. Carey, A. Wiese, F. Stetler, M. Spears, A. Miller, J. Gardner, M. Kynoch, C. Kupferman, E. Berge, D. Crawford, J. Ialeggio, E. Peyton, E. Brown, J. LeClaire, L. Deets, S. Bilodeau, T. Russell, and K. Sloan



Cite this report as: Hermanns, L. F., K. M. Black, S. G. Robinson, K. M. Walker, H. A. Bellman, J. D. Fraser, D. H. Catlin, S. J. Ritter, and S. M. Karpanty. 2018. Effectiveness monitoring and targeted studies in support of FIMI conservation measures: Piping plover and red fox monitoring on Fire Island and Westhampton Island, New York. Annual report prepared by the Virginia Tech Shorebird Project. Virginia Tech, Blacksburg, VA.

Cover photo was provided by Anita Regler

Green flag '884' was banded at Fire Island, NY in June 2018 and was photographed near Ponquogue Bridge, Shinnecock, Long Island, NY, in 2018.

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Executive Summary

This report summarizes Virginia Tech's Effectiveness Monitoring of certain conservation measures required under the Biological Opinion for the Fire Island Inlet to Moriches Inlet Stabilization Project in 2018. Where appropriate and possible, we provide cumulative results to facilitate the inspection of trends. In April–August of 2018, we conducted field monitoring of piping plovers and red foxes. Red fox monitoring continued year-round. In January–March and September–December, we conducted data Q&A, data analyses, report writing and project support (e.g., renting houses, equipment maintenance, technician hiring, etc.).

Piping Plover Monitoring Summary

During the 2018 piping plover breeding season, we banded 42 adults and 169 chicks. We found 70 nests; 19 within Robert Moses State Park, 13 in Fire Island National Seashore, 31 in Smith Point County Park, and 7 within Cupsogue County Park. Of the total 70 nests, 50 hatched. Nests exclosed during laying or after clutch completion had a 72% predicted probability of surviving to hatch, while nests never exclosed had a 48% predicted probability of surviving to hatch. Of the 174 chicks that hatched, 115 fledged (66%). Robert Moses State Park produced 34 fledglings, Fire Island National Seashore 29 fledglings, Smith Point County Park 25 fledglings, and Cupsogue County Park produced 7 fledglings; resulting in a reproductive output of 1.98 fledges/pair for the entire study area. By management area, productivity was 2.27 fledges/pair at Robert Moses State Park, 1.29 fledges/pair at Cupsogue County Park, 1.72 fledges/pair at Smith Point County Park and 2.64 fledges/pair in the Fire Island National Seashore. Overall, chicks had a 72% predicted probability of surviving to fledge. Robert Moses State Park, Fire Island National Seashore, and Cupsogue County Park had their highest estimated productivity since the beginning of the study. Smith Point County Park did not reach the highest reproductive output of the study, but it increased from 2017.

Piping Plover Habitat Use, Selection, and Demography

We assessed several metrics relating plovers to post-Hurricane Sandy habitat. Adult and fledgling piping plovers moved large distances (max = 36km) to access overwash, flood shoal, and mudflat habitats, some of which were not protected from disturbance during migration or otherwise. During 2014–2018, on average, access to bayside foraging habitat did not affect the distance birds were observed from their nests. There was no difference in the clutch masses of piping plovers that had bay access compared to those that did not have bay access. Using data from 2014–2018, brood home range size increased as broods aged. From 2014–2018, 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, and that mean distance between successive brood observations was smaller for broods with bay access than for broods without. In 2018, invertebrate samples in the bay beach had a greater mean arthropod catch than barrier interior or ocean beach invertebrate samples. With all years combined (2014–2018), for newly created habitats (i.e., overwashes, breach fill, or restoration areas), overwashes had greater invertebrate catches than the restoration areas and breach fills.

Restoration areas and manipulated overwashes supported the highest percentage of natal recruits from 2014 to 2018, whereas manipulated and natural overwashes supported the highest proportion of immigrants. When comparing the available habitat to the proportion that different groups of nesters (immigrants, local recruits and returning nesters) used in 2017, 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 more new birds than the rest of the study area. Settlement patterns were determined only for 2017 because it was the most recent year with habitat classification available. Immigration rate has steadily declined over the last few years, perhaps suggesting that survival and immigration decrease with increasing density. However, we have not observed a similar pattern with reproductive output, and survival has remained relatively stable.

Historically, coastal populations of piping plovers increased 42%–745% following habitat creating events (Robinson et al. *in prep*). Despite increased habitat and relatively high immigration rates, the population increase in the study area since before the hurricane (93%) has been modest compared to other habitat creating events, but within the range of past observations. Less than anticipated growth may have been related to generally low local and regional reproduction just before and immediately following the storm. An average of 31% 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.

Traditional methods (visual identification in samples) for piping plover diet seem as effective as DNA metabarcoding to identify prey to phylum, but not order, due to a proportion of samples only being identified to phylum Arthropoda using visual identification methods. We detected several invertebrate taxa using both methods to examine fecal samples collected in 2013–2014 and 2016–2017, such as a variety of flies, beetles and horseshoe crabs (*Limulus polyphemus*). We also specifically looked for fish in fecal samples due to past observations of plovers consuming fish. We detected fish in fecal samples, but only using DNA metabarcoding methods. Fish prey may only be identifiable by DNA barcoding due to partial consumption (i.e., not consuming bones and cartilage), or it may be a rare food source. There is a lack of local prey DNA examples in free DNA reference libraries, which precludes identification of some prey to species level. In addition, the fecal samples analyzed using different methods were from different years, so may represent variation in prey availability. In the future, we need to address whether DNA originated from direct consumption of plover prey, or if it was secondary consumption (i.e. consuming larvae that were consuming other organisms).

Although not described in this report, vegetation continues to rapidly encroach on plover habitat (Bellman 2018). It is not clear how much habitat will be available by 2019. Still, local and regional reproduction was high in 2018, so if unoccupied habitat remains, we may see an increase in breeding plovers in 2019. 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

We estimated a minimum red fox population size of only 7 adults as of August 2018 to the West of Old Inlet, down from a peak of 96 red foxes in the same area in 2016 due to an ongoing outbreak of sarcoptic

mange (caused by the *Sarcoptes scabiei* mite). We conservatively estimated a minimum red fox population size of 1 adult between Old Inlet and Moriches Inlet; red foxes in this site were affected by sarcoptic mange in late Summer 2015 and minimum estimates have consistently been 0-1 red foxes since that time. We did not find any evidence of breeding dens or other signs of successful reproduction in any parts of the study area during this 2018 breeding season.

The sarcoptic mange outbreak that was ongoing west of Old Inlet at the time of our last report continued in 2018, resulting in a substantial decrease in the fox population. Three of the 4 foxes captured during our 2018 trapping period showed signs of moderate to severe mange infestation. Our GPS collared foxes died prior to piping plover nest initiation in 2018, either from mange or unknown causes. We continued seeing signs of mange infestation during our field observations and in remote camera photos collected during non-breeding den monitoring through 2018. One eartagged, but uncollared, fox was last seen alive but severely affected by mange in mid-September 2018, when the last of our den cameras were taken down. During the 2018 monitoring period, we did not recapture, resight, or otherwise detect any marked or known individuals from previous monitoring periods.

More recently, we observed 2 foxes affected by mange (1 in the Fire Island villages on October 31st, 2018; 1 in the National Seashore Wilderness Area on November 8th, 2018). We have received a reliable report (supported by photos) of at least 2 healthy foxes seen in one of the Fire Island villages on November 12th, 2018. These sightings suggest that the outbreak is still ongoing, and there are at least a few foxes left on the island. Our current and final fall camera grids will continue to provide important information needed to assess the overall impact of this ongoing mange outbreak on the red fox population west of Old Inlet.

The current (2017-present) outbreak does not appear to have spread to the east side of Old Inlet, as we have not observed any signs of mange in red foxes in Smith Point County Park or in that portion of the National Seashore Wilderness Area. However, our field observations and the complete lack of fox sign during several of our monthly transect surveys suggest that the east side of Old Inlet may not have been continuously occupied by foxes from February–August 2018.

Red foxes were photographed at least once at the majority of camera stations at Robert Moses State Park and Smith Point County Park/Old Inlet East in Fall–Winter 2017, resulting in occupancy estimates of 0.97 (SE 0.03) and 0.84 (SE 0.12), respectively, in these areas. Red foxes were photographed at least once at all camera stations at Fire Island National Seashore between Watch Hill and Old Inlet in Fall–Winter 2017, resulting in an occupancy estimate of 1 in this area.

When the 50% and 95% core and home range areas of both GPS-collared foxes in spring 2018 were overlaid, the resulting contours encompassed nearly all dry land from Field 5 of Robert Moses State Park through the Fire Island National Seashore Lighthouse Tract and eastward into the villages.

Interactions between Red Fox and Piping Plover

Both GPS collared foxes at Robert Moses State Park and the National Seashore Lighthouse Tract died prior to piping plover nest initiation in 2018. Thus, we did not collect any high frequency fox location data in

2018 or compare fox home ranges to nest or brood locations. All but one of the 45 adult piping plover locations recorded in the Field 5/Lighthouse Tract area before the collared foxes died were in the 95% kernel density home range of at least one collared fox, and roughly one third of these locations also were located within the 50% kernel density core area of at least one collared fox. During Spring–Summer 2018, no red foxes were collared at Smith Point County Park, Robert Moses State Park, or Fire Island National Seashore, and no nests, brood locations, or adult locations in those areas were overlapping with known, collared red fox territories.

There was a correlation between piping plover productivity (chicks/pair) and the timing of the mange outbreaks in the red fox population in our study sites. We saw an increase in piping plover productivity east of Old Inlet following the mange outbreak in 2015–2016, and an increase in productivity west of Old Inlet during the mange outbreak in 2017–2018. Median chick survival follows the same relationship, but the data have a wider spread. In 2016, when foxes were not present on Smith Point County Park and Fire Island National Seashore east of Old Inlet, plover reproductive output (Ro) was very high (2.38 and 2 chicks/pair); more than twice the Ro in the other management areas, suggesting an important correlation with red fox declines. 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 an 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). In 2017 the Fire Island management units had similar plover reproductive output (1.50–1.65 fledge/pair), all substantially above the level needed for population stationarity and in 2018 reproductive output remained high (1.29–2.64 fledge/pair). Recreational disturbance (DeRose-Wilson et al. 2018) and habitat variability (Loefering and Fraser 1995, Elias et al. 2000, Cohen et al 2009) likely affected these results. As of now, the mange zoonotic seems to be continuing to decrease the fox population west of Old Inlet; 2019's reproductive output in Robert Moses State Park may shed additional light on the effect of red foxes on piping plover reproductive output. The mange-driven decline of the red fox in our study areas may serve as a proxy for understanding the potential impacts of future lethal fox predation management.

One scat containing a set of adult piping plover leg bands was found and collected on June 15, 2018 in Smith Point County Park. The species of origin for this scat is currently unknown, as there were no identifiable tracks or signs in the immediate area, although a plover wing was found near the exclosure of dead flagged bird. A subsample of the scat was preserved for genetic analyses, and will be sent for identification of source species' fecal DNA with the next shipment of samples to the collaborating genetics lab. Fox sign was observed in Smith Point County Park in 2018 when a fox attempted to dig under two exclosures; within the same subsite, five nests were predated when there was a pause in exclosure installations after the digging occurred.

Continued monitoring of the relationship between piping plovers and predator communities is necessary for full understanding. In the future, we will be synthesizing the two data sets to understand correlation, and to tease out the relative importance of multiple factors driving plover abundance and productivity.

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 arrived at the barrier islands of New York and New Jersey and created 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 determine habitat quality and have an effect on the piping plover nesting densities (Elias et al. 2000, Cohen et al. 2009). Natal territories with 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 (Loegering and Fraser, 1995, Cohen et al. 2009, Catlin et al. 2015). Prey abundance and location can contribute to nest site location, as plovers favor nesting close to moist substrates for availability to food sources during the nesting period. (Elias et al. 2000, 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.

This report seeks to summarize targeted monitoring programs created to evaluate the efficacy of the BO conservation measures, developed by the USFWS, with the assistance of Virginia Tech. The monitoring efforts will provide a scientific evaluation of the effectiveness of the management and habitat maintenance and restoration efforts within the FIMI project area and Fire Island, as a whole. Both Virginia Tech and the USFWS worked closely with an interagency team consisting of the National Park Service (NPS), New York State Office of Parks, Recreation, and Historic Preservation, 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, report writing, and project support (e.g., renting houses, equipment maintenance, technician hiring, etc.). In April–August we conducted field monitoring of piping plovers; red fox monitoring occurs year-round. From our piping plover monitoring efforts from April through August 2013–2018, 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 broad ecological understanding of how 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, nine overwash areas, and other areas within the study site. Ultimately our results will help refine our understanding of the timing of 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 (referred to as management units in this report) of the National Park Service (Otis Pike Wilderness Area and the Fire Island Lighthouse Tract), Suffolk County Parks (Cupsogue Beach County Park and Smith Point County Park) and the New York State Office of Parks, Recreation, and Historic Preservation (Robert Moses State Park). From 2013 to June 2015, the study area only consisted of Otis Pike Wilderness Area, Smith Point County Park, and Cupsogue Beach County Park (17.3 km of beachfront). In June 2015, when the FIMI funding became available, the study area was expanded to include Robert Moses State Park and the Fire Island Lighthouse Tract (9.8 km).

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, and a breach in the Narrow Bay West area of Smith Point County Park. The Old Inlet breach remains open. The Cupsogue breach was filled with a berm 9+ ft. National Geodetic Vertical Datum (NGVD) in elevation, 150 ft. wide and 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 using terminology in 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
Backshore	The dry sand area between the high water line and the toe of the dune.
Barrier Interior Beach	The area inland of the toe of the dune and interior of Fire Island. The area that is between the high and low water line. Can be further defined as ocean, bay or inlet depending on the geologic location. Synonymous with the intertidal zone.
Breach Fill	The areas of Fire Island and Cupsogue County Park that breached during Hurricane Sandy but were filled shortly after by USACE. Both breach fills fall adjacent to manipulated overwashes.
Dredge Spoil	Stockpiles and graded areas of dredged sand.
Ephemeral Pool	Temporary ponds that typically have ponded water only part of the year.
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.
Management Area	Areas defined by USACE to manage under the FIMI.
Manipulated Dune	Any dune or berm created by other than natural sources. This includes snow fencing, Christmas trees, planting and USACE/other engineering.
Manipulated Overwash	Areas of Fire Island that overwashed during Hurricane Sandy, but have been manipulated by breach filling, dune manipulation (including snow fencing and Christmas tree placement), or vegetation management.
Mud Flat	Mostly intertidal land forms where particles are generally < 0.05 mm in diameter and often mixed with bits of organic material. Areas where fine-grained sediments have been deposited in the low-energy shallows behind the island.
Natural Overwash	Areas of Fire Island that overwashed during Hurricane Sandy, but have not been further modified.
Nourishment	Beach widening, by the placement of sand in the backshore, by the USACE.
Other	A broad category, where the area does not fall into one of the overarching categories above. All following categories could further be defined under the 'other' category, depending on the specificity of the question.
Overwash	Any area that overwashed during Hurricane Sandy. This is a broad category that subdivided into manipulated or natural overwash.
Restoration Area	The New Made and Great Gun restoration areas, created between the 2014 and 2015 piping plover breeding seasons.
Sand Bar	Mostly intertidal landforms that are generally composed of particles > 0.05 mm. In this study area, sand bars occur north of the inlets.
Swale	A depression or blow-out behind the primary dunes, between ridges, either vegetated or unvegetated.

Throughout the plover breeding seasons, all beaches were open to pedestrians. Smith Point County Park and Robert Moses State Park also allowed recreation vehicles to drive onto the beaches. Every year, these beaches were closed to vehicles around the time chicks hatched and were re-opened around the time the chicks

fledged (Table 2). However, Smith Point County Park in 2013, and subsite OSS in Robert Moses State Park-South in 2017 and 2018, remained open to vehicle access during the hatch and fledged. 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 2013–2018.

Management Unit	2013	2014	2015	2016	2017	2018
Smith Point County Park	closed, but dates unknown	09 Jun– 01 Aug	09 Jul–13 Aug	22 May– 25 Aug	27 May– 05 Aug	11 Jun–21 Aug
Robert Moses State Park - North	-	-	06 Jun– 04 Aug	06 Jul–22 Jul	07 Jun– 16 Jul	02 Jun–03 Aug
Robert Moses State Park - South	-	-	10 Jul–04 Aug	25 Jun– 05 Aug	24 May– 10 Aug	29 Jun–03 Aug

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

Table 3. Subsite names, location descriptions, and habitat at Fire Island and Westhampton Island, 2018.

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
Watch Hill	WHL	Begins at Watch Hill and extends east to Bellport Beach.	Beach, Backshore Interior and Natural Overwash
Old Inlet West Reference	OIWR	Stretch of beach between the east end of Old Inlet West and 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 to 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

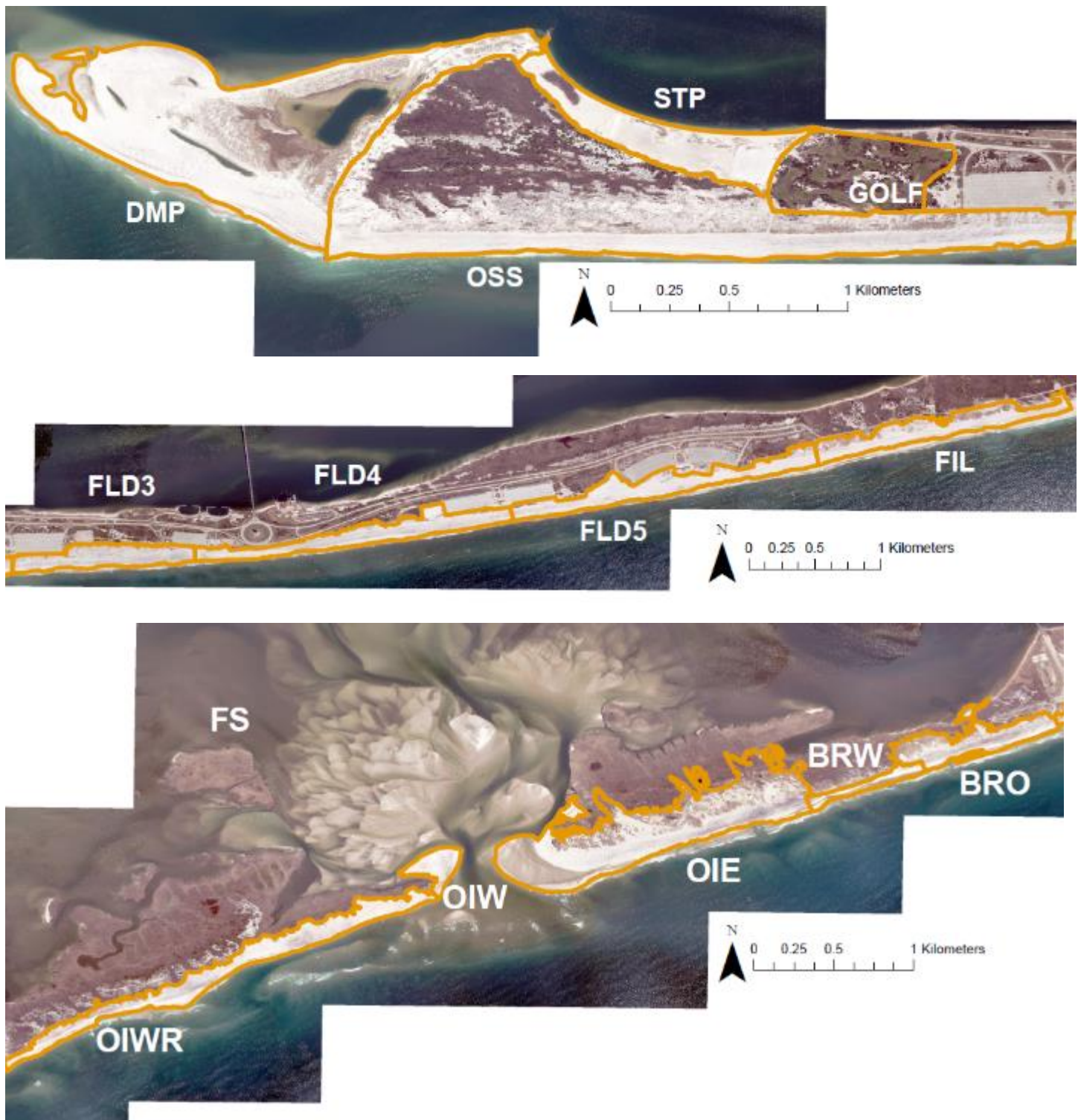


Figure 1. Study area at Robert Moses State Park (top) Fire Island Lighthouse Tract (middle) and Fire Island National Seashore (bottom). Subsite boundaries are outlined in orange. See Table 3 for subsite names and descriptions. Imagery captured April, 2018.



Figure 2. Study area at Smith Point County Park (top) and Cupsogue County Park (bottom). Subsite boundaries are outlined in orange. See Table 3 for subsite names and descriptions. Imagery captured April, 2018.

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 2015–2018 to map the shoreline during 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 2018, we mapped shorelines at spring high and low tides in May and June, and mapped snow and symbolic fencing May–August. In 2017, we mapped the shorelines during spring high and low tides in May, and the snow and symbolic fencing during late June–August. In 2016, we mapped shorelines at spring high and low tides May–September, and completed snow and symbolic fencing mapping during late-July and August. In 2015, we mapped spring high tide and low tides during mid-May and early June, and mapped various habitats and subsite boundaries during late July and early August. In 2013 and 2014, we mapped the shoreline at tides throughout the entire season, although no shoreline low tide data was collected in 2014.



Figure 1. 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, 2018. 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 tracks 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 2016–2018, 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 Democratic Point area.

Nest Location

We searched all potential nesting areas in the study site, including backshore, overwashes and sparsely vegetated areas. Nest locations were recorded using a Trimble GPS (2015–2018, Figure 5a) unit or a Garmin GPS unit (2013–2014). We floated all the eggs of nests initially found with 4 eggs to estimate the initiation and hatch date (Figure 5b) (Alberico 1995). Eggs were not floated in the western wilderness of Fire Island National Seashore (subsites OIW and OIWR) after the 2014 season.

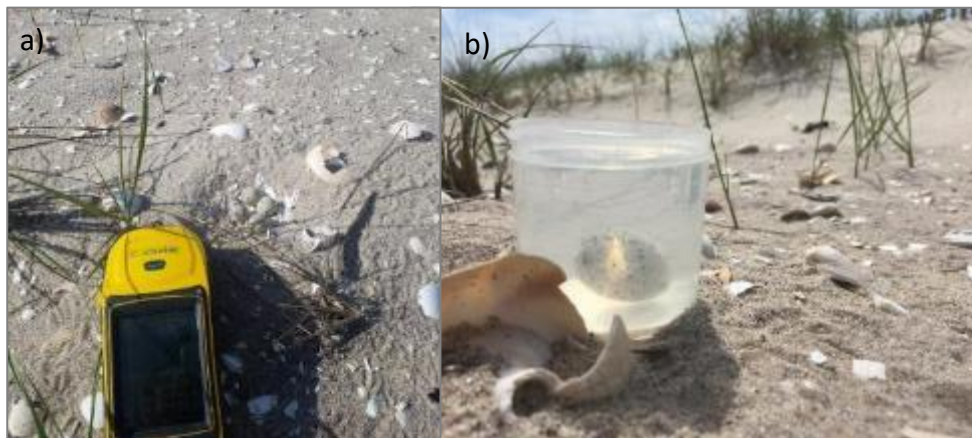


Figure 5. a) Using a Trimble Geo7X to map the location of a piping plover nest. b) Floating an egg from a four egg nest in subsite SPTW.

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 daily. 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 2018, 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 a 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 post-hatch to weigh and measure them a second time.



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

Banding Schemes

In 2014–2018, we banded all adults and chicks with dark green flags with a three-digit alphanumeric ID etched in white on one upper leg and a numbered orange Darvic band on the opposite upper leg (Figure 7). In 2013, 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 2. 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 conducted a disturbance point count, for which we recorded the presence of anything other than the brood (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 2014–2018 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 3. Invertebrate trapping station.

Fecal Sampling

In 2013–2014 and 2016–2017, we collected piping plover fecal samples opportunistically while conducting field surveys. Fresh fecal samples were collected, with as little as sand as possible, and placed in vials using tweezers (Figure 9). 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 in coolers in the field until storing them in the field site freezer until the end of the season. No fecal samples were collected in 2018. Thirty-four samples were collected across the study site in 2017, and 39 were collected in 2016. We did not collect fecal samples in 2015. Thirty-seven fecal samples were collected in 2014, and 7 in 2013. 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.



Figure 4. Collecting a fecal sample, using tweezers and marking the location with Trimble Geo7x.

Nest Cameras

In 2016–2018, we placed remote cameras (Reconyx Rapidfire, RC60 Covert IR, Holmen, WI, USA) at nests to detect anything that visited the enclosure. 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 overexposure. We placed the camera box against the outside of the enclosure and aimed the camera toward the nest. We attached the camera box to the enclosure with a bungee cord to secure it (Figure 10). We performed any maintenance needed to cameras, such as replacing memory cards and batteries, while we conducted daily nest checks to reduce disturbance. The cameras were removed when the brood permanently left the enclosure area.



Figure 10. 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 various habitats 2013–2018. 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 estimated 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 overlap, 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, resulting in a conservative pair estimate.

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 2014–2018, 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. For adults with active nests we removed locations within 30m of the nest to account for birds incubating and GPS error. Similar to adult locations, we compiled all of the fledgling locations from 2014 to 2018 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, breach fills 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 Wilcoxon Rank Sum 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, Democrat Point 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 2013–2018. 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, Old Inlet West, Democratic Point, 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 2018 ($n = 160$ broods). We estimated daily survival over a 31-day period. Our model set included year and a linear effect of time. We ran all subsets of models including year and time 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 2014–2018 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 a difference between broods with bay access or not, controlling for the number of points used in the estimate the MCP.

We 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). We then used a Wilcoxon Rank Sum test to assess whether there was a difference in daily movement rates for broods with or without direct bay access, and used a Kruskal-Wallis test whether there was difference between area of brood home range and accessible foraging habitats (filled breach, restoration, natural overwash, manipulated overwash, and other).

Finally, we compare piping plover productivity and chick survival to fledge among years in relation to the mange outbreak in the fox population. The groups are pre-mange outbreak (2013–2015 east of Old Inlet, 2013–2017 west of Old Inlet), post-mange outbreak (2016–2018 east of Old Inlet, 2018 west of Old Inlet) and unknown (Cupsogue, for which the predator community is unknown). Data input for the productivity plot is

productivity for the entirety of Fire Island east or west of Old Inlet. Data for the survival plot is survival to fledge by management unit.

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 2014–2018. 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 selected 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 have the selection analysis completed only for the eastern study area for 2015 (Granger et al. 2017) and 2017 (this report), as they are the years for which we currently have classified imagery. In 2013 we could not estimate immigrants or natal recruits without prior banded birds, and for 2014, there was no imagery available. We intend to complete 2016 and 2018 once the spring image classification is complete.

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 estimated using this model were detection probability (p), probability of being reported dead during the supplemental period (r), the probability of being resighted during and surviving the supplemental period (R), the probability of being resighted during the supplemental period and dying (R'), site fidelity (F), and the probability of returning to the study area, 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.

We used annual pre-fledge chick survival estimates from Objective 4, and our annual estimates of HY survival from the Barker model to derive post-fledging survival, the survival from fledging until following year. Post-fledge survival is HY survival divided by pre-fledge survival.

Objective 6: Compare prey densities at several habitat types

To assess the difference in the mean number of terrestrial invertebrates collected on bay beaches, ocean beaches, and barrier interior, we performed a Kruskal-Wallis chi-squared test using the invertebrate samples from 2018. 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 2014–2018. For the grouping variables within the analysis, habitat type (beach, ephemeral pool, and interior) 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.

Using fecal samples from 2013–2014 and 2016–2017, we compared two methods of prey identification. The first was visual identification in which taxonomists from the South Carolina Department of Natural Resources identified prey organisms to the lowest possible taxa (Carey et al. 2016), and this was conducted using the 2013–2014 fecal samples. The second identification method was DNA metabarcoding, in which we extracted genomic DNA from fecal samples, and amplified the DNA using taxa-specific primers. This work was contracted out to Jonah Ventures in Boulder, Colorado, and was done on the 2016–2017 fecal samples. We summarized the detection data from both methods to order and phylum.

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. The length of time a camera remained on was determined by when we were able to capture both adults to when the nest hatched and the brood left the exclosure; therefore, the duration of time each camera was on an exclosure varied by nest. 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 II: 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 2018, we banded 42 adult piping plovers and 169 chicks (Table 4). We were unable to capture and band five chicks out of the 174 that hatched: one chick from one brood from the East Wilderness Area, two chicks from the same brood from East Wilderness Area, one chick from Smith Point County Park, and one chick from Democrat Point at Robert Moses State Park. We discovered carcasses of all but one of these five unbanded chicks (see Objective 4 for further detail on chick mortality). The remaining chicks in 2018 were banded between 0 and 11 days old. We recaptured 52 adult piping plovers that were banded during previous years. Of those recaptured, 21 of them were banded as chicks on Fire Island, 29 were banded as adults on Fire Island, one was banded as an adult in New Jersey, and one was banded as a hatch year on migration at Cape Hatteras.

Between 2013–2018, we banded 193 adult piping plovers (108 females and 85 males) and 522 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 2018. 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 522 total chicks banded, 32.4% were banded in the 2018 season, 24.9% in the 2017 season, 19.3% in 2016, 9.0% in 2015, 7.8% in 2014 and 6.5% in 2013.

Table 4. Piping plover adults and chicks banded on Fire Island and Westhampton Island during April to August 2018 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	
Cuspogue Beach County Park (CSE, CSW, CMF, CSB)	5	3	21
Smith Point County Park (BRE, PSO, PSE, NBW, NMA, NBE, SPTW, SPTE)	8	5	66
Fire Island National Seashore - east of breach (OIE, BRW, BRO)	2	0	30
Fire Island National Seashore - west of breach (OIW, OIWR)	1	1	4
Fire Island National Seashore - Lighthouse Tract (FIL)	0	0	0
Robert Moses State Park (FLD5, FLD4, FLD3, OSS, STP, DMP, GOLF)	10	7	48
Total	26	16	169

Table 5. Piping plover adults and chicks banded on Fire Island and Westhampton Island during 2013–2018 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)	20	14	101
Smith Point County Park (BRE, PSO, PSE, NBW, NMA, NBE, SPTW, SPTE)	35	29	191
Fire Island National Seashore - east of breach (OIE, BRW, BRO)	15	12	104
Fire Island National Seashore - west of breach (OIW, OIWR)	2	4	18
Fire Island National Seashore - Lighthouse Tract (FIL)	1	1	0
Robert Moses State Park (FLD5, FLD4, FLD3, OSS, STP, DMP, GOLF) ^a	33	22	105
Off Site – Bellport to Watch Hill ^b	2	3	3
Total	108	85	522

^a Birds were only banded here from 2015–2018

^b Birds were only banded here from 2013–2014

We found 70 nests in 2018, 50 of which survived to hatching and 20 of which failed (Table 6). Of the 20 that failed, eight were abandoned, six were depredated, three flooded, one failed due to human interaction, in which a person entered an exclosure and destroyed the nest. Two out of the 20 failed nests were suspected predation, however, fate was undeterminable due to heavy rain and wind effects. All six nests depredated in 2018 were unexclosed. One predated nest was suspected as avian predation, with crow tracks leading up to the nest cup and yolk and eggshell found less than 5 meters away. Four nests were suspected red fox predation with fox tracks leading up to or around the nest cup. One of the six predated nests was found with a piece of eggshell in the nest cup, but no other distinguishable predator sign was observed. From 2013–2018, we located and monitored 272 nests. Of these nests, 165 (60.7%) of them hatched, and 107 (39.3%) of them failed (Table 7). The most common reason for nest failure over the entire study was depredation (14%, 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 2018. Nest numbers include renests of some pairs.

Management Unit	Habitat	Nests			
		Found	Hatched	Failed	Reason for loss
Robert Moses State Park	Backshore	14	10	4	Abandoned (1), Flooded (2), Predated (1)
Robert Moses State Park	Barrier Interior	1	1	-	-
Robert Moses State Park	Swale	2	2	-	-
Robert Moses State Park	Manipulated Dune	1	-	1	Abandoned
Robert Moses State Park	Management Area	1	1	-	-
Fire Island National Seashore	Natural Overwash	11	9	2	Abandoned (1) Predated (1)
Fire Island National Seashore - east of breach	Manipulated Dune	1	-	1	Abandoned
Fire Island National Seashore - east of breach	Backshore	1	1	-	-
Smith Point County Park	Manipulated Overwash	10	10	-	-
Smith Point County Park	Manipulated Dune	4	2	2	Predated Predated (1)
Smith Point County Park	Barrier Interior	5	3	2	Abandoned (1)
Smith Point County Park	Backshore	2	2	-	-
Smith Point County Park	Restoration Area	10	3	7	Abandoned (3) Predated (2) Unknown (2)
Cupsogue Beach County Park	Manipulated Overwash	4	3	1	Abandoned
Cupsogue Beach County Park	Manipulated Dune	1	1	-	-
Cupsogue Beach County Park	Backshore	2	2	-	-

Table 7. Piping plover nest distributions and status at Fire Island and Westhampton Island, in various habitats of each management unit, 2013–2018. 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	-	2	-	100
Robert Moses State Park ^a	Backshore	37	23	14	62.2	37.8
Robert Moses State Park ^a	Barrier Interior	8	4	4	50	50
Robert Moses State Park ^a	Dredge Spoil	5	3	2	60	40
Robert Moses State Park ^a	Swale	2	2	-	100	-
Robert Moses State Park ^a	Dune	1	-	1	-	100
Robert Moses State Park ^a	Management Area	1	1	-	100	-
Cupsogue Beach County Park	Manipulated Overwash	31	21	10	67.7	32.3
Cupsogue Beach County Park	Manipulated Dune	1	1	-	100	-
Cupsogue Beach County Park	Backshore	2	2	-	100	-
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 - east of breach	Manipulated Dune	1	-	1	-	100
Fire Island National Seashore - east of breach	Backshore	1	1	-	100	-
Fire Island National Seashore - west of breach	Natural Overwash	16	13	3	81.3	18.8
Smith Point County Park	Backshore	15	8	7	53.3	46.7
Smith Point County Park	Manipulated Dune	4	2	2	50	50
Smith Point County Park	Barrier Interior	5	3	2	60	40
Smith Point County Park ^a	New Made Restoration Area	8	2	6	25	75
Smith Point County Park ^a	Great Gun Restoration Area	20	10	10	50	50
Smith Point County Park	Manipulated Overwash	60	35	25	58.3	41.7
Off Site – Bellport to Watch Hill ^c	Backshore	8	4	4	50	50
Total		272	165	107	60.7	39.3

^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, 2013–2018.

Year	Management	Total	Successful	Abandoned	Depredated	Failed	
						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%)
2018	Robert Moses State Park	19	14 (74%)	2 (11%)	1 (5%)	2 (11%)	-
	Fire Island National Seashore	13	10 (77%)	1 (8%)	1 (8%)	1 (8%)	-
	Smith Point County Park	31	20 (65%)	4 (13%)	5 (16%)	-	2 (6%)
	Cupsogue Beach County Park	7	6 (86%)	1 (14)	-	-	-
Total		267	160 (60%)	34 (13%)	37 (14%)	8 (3%)	19 (7%)

The number of pairs we monitored increased from 19 in 2013 to 58 in 2018, 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 five years of the monitoring effort. Though there was fluctuation within sites, the overall number of pairs has 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	2018 ^b
Overwash	11	17	15	14	23	21
Breach Fill	1	1	3	3	3	3
Restoration	N/A	N/A	5	4	6	10
Backshore/Barrier Interior	7	3	12	16	13	24
Total	19	21	35	37	45	58

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

^b Includes Robert Moses and the Lighthouse Tract, only monitored 2015–2018.

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 bolded and italicized. Some pairs had more than one nest per year.

Management Unit	Year					
	2013	2014	2015	2016	2017	2018
Robert Moses State Park	<i>12</i>	<i>10</i>	10	9	8	15
Fire Island Lighthouse	<i>1</i>	<i>1</i>	0	1	1	1
Fire Island National Seashore West	6	4	1	1	0	1
Fire Island National Seashore East	4	4	2	6	10	9
Smith Point County Park	6	8	13	13	20	25
Cupsogue County Park	3	5	7	6	6	7
Total	32	32	33	36	45	58

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 10,862 m; this occurred in 2015 when an adult with a nest at Great Gun was observed at Old Inlet East. The greatest distance an adult was observed from an inactive nest was 36km; an adult that nested in Field 2 (OSS) of Robert Moses State Park was observed foraging at the Old Inlet East after nest loss in 2018.

In 2014, the average distance that birds with active nests were observed from their nest was 241 m (SE = 36), and the average distance that birds without active nests (i.e., pre- or post-nesting) were observed from their nests was 1261 m (SE = 107). In 2015, the average distance that birds with active nests were observed from their nest was 437 m (SE = 60), and the average distance that birds without active nests (i.e., pre- or post-nesting) were observed from their nests was 894 m (SE = 60). In 2016, the average distance that birds with active nests were observed from their nests was 228 m (SE = 22), and the average distance that birds without active nests (i.e., pre- or post-nesting) were observed from their nest was 540 m (SE = 93). In 2017, the average distance that birds with active nests were observed from their nest was 186 m (SE = 14), and the average distance that birds without active nests (i.e., pre- or post-nesting) were observed from their nest location was 890 m (SE = 166). In 2018, the average distance that birds with active nests were observed from their nest was 189 m (SE = 24), and the average distance that birds without active nests (i.e., pre- or post-nesting) were observed from their nest location was 1021 m (SE = 133). At this time, there does not seem to be a clear trend in the distance we have observed birds from their nests, relative to time since Hurricane Sandy.

We recorded seven 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 11). We have no observations of nesterts at the Old Inlet Flood Shoals.

We had 21 observations of birds with active nests and 114 observations of birds without active nests using the Cupsogue sandbar over the five years of the study. The greatest distance that an individual with a nest moved to the Cupsogue sandbar was 2.1 km 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 114 instances of individuals using the sand bar that were not actively nesting, the greatest observed distance was 8.4 km and was an individual who had nested at Pattersquash Overwash earlier in the season.

At the Cupsogue mud flat, there were 9 observations of individuals with an active nest. 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 29 pre- or post- nesting instances of individuals foraging at the Cupsogue Mud Flat, with the greatest distance from their nest of 6km, which 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 82 individuals during 202 observations present in locations other than the subsite they were nesting in, 107 of which were foraging observations. Most (N = 20) of these locations were individuals foraging at the Cupsogue Sand bar, followed by Narrow Bay West at the bayside mudflat, with the birds traveling most often 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 used the overwashes as well. 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, 2014–2018.

Habitat	Distance from Nest (m)				95% Confidence Interval	
	N	Max	Mean	SE	Lower	Upper
Bridge Overwash	14	1324.9	261.6	118.8	5.1	518.2
Cupsogue Mud Flat	9	2547.2	1336.2	242.1	778.0	1894.4
Cupsogue Sand Bar	20	2136.9	866.2	92.9	671.9	1060.6
Cupsogue Overwash	126	4089.6	174.3	35.0	105.1	243.5
Cupsogue Breach	93	4184.1	193.3	47.1	99.7	286.9
Great Gun Restoration Area	94	546.1	121.2	10.5	100.2	142.1
Narrow Bay Breach	88	377.2	113.0	8.6	96.0	130.0
NBE Overwash	90	8361.9	234.1	94.6	46.1	422.1
NBW Overwash	70	2005.4	200.8	32.7	135.5	266.1
New Made	21	2450.7	372.9	117.9	126.9	618.9
OIE Overwash	303	10862.5	198.9	42.7	114.8	283.1
OIW	6	1353.3	1096.5	76.1	900.9	1292.2
OIWR Overwash	6	88.0	55.7	8.0	35.3	76.2
PSO Overwash	167	5898.8	235.2	42.4	151.6	318.9

Table 12. Distance from nest to Hurricane Sandy created 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, 2014–2018.

Habitat	Distance from Nest (m)				95% Confidence Interval	
	N	Max	Mean	SE	Lower	Upper
Bridge Overwash	27	4049.0	580.3	203.8	161.4	999.3
Cupsogue Mud Flat	24	6066.4	1835.0	219.4	1381.1	2288.8
Cupsogue Sand Bar	114	8384.6	1727.1	203.9	1323.2	2130.9
Cupsogue Overwash	166	7692.1	443.5	88.6	268.6	618.4
Cupsogue Breach	164	7744.8	564.0	93.9	378.7	749.3
Old Inlet Flood Shoal	7	33049.6	8285.0	4463.8	-2637.4	19207.4
Great Gun restoration area	100	2374.8	262.2	39.4	183.9	340.4
Narrow Bay Breach	55	2072.8	159.2	44.0	71.0	247.5
NBE Overwash	142	8584.2	1529.9	248.8	1038.0	2021.9
NBW Overwash	75	7312.9	432.6	114.2	205.0	660.1
New Made	45	2502.4	286.6	84.9	115.5	457.7
OIE Overwash	465	36060.6	834.5	144.5	550.5	1118.4
OIW	59	34405.1	4093.6	745.8	2600.7	5586.5
OIWR Overwash	4	8090.4	2108.7	1993.9	-4236.9	8454.3
PSO Overwash	276	7228.4	712.6	88.2	539.0	886.3

In 2014–2018, adult plovers with an active nest and direct bay access were not seen closer to their nests ($\bar{x}$ = 199.0, SE = 9.7) than plovers that did not have direct bay access ($\bar{x}$ = 289.4, SE = 26.8; df = 1, W = 326560, P = 0.81).

Fledgling Movement

We assessed the use of newly created habitats and other habitats of interest by fledgling piping plovers relative to the nest from which they hatched. From 2014–2018, we recorded 1,179 observations of fledglings from the study area. On average, those individuals were observed 3,258 m (SE = 249 m; Table 13) from their nest. The maximum distance a fledgling was observed, within the study site, from its nest site was 50.2 km; a fledged chick from Cupsogue West was observed foraging at Fire Island Inlet at Robert Moses State Park in 2018. Some Fire Island banded fledglings were observed during off-site resighting at Jones Beach State Park, and were observed by birders elsewhere, but these locations are not considered here. On average, fledglings moved farthest to Cupsogue mud flat, Old Inlet West, Cupsogue sand bar, and the Old Inlet flood shoals. From the creation of the restoration areas to 2017, we only observed fledglings that had hatched there. In 2018, fourteen fledglings that had hatched elsewhere in Smith Point or Cupsogue County Parks were observed in Great Gun restoration area and most were observed on the oceanfront. During 2014–2018, we observed six

fledglings return to nest as first-year birds within a subsite where they had been observed as a fledgling the year prior.

Table 13. Distance moved by piping plover fledglings from their nest location on Fire Island, June–August, 2014–2018.

Habitat	Distance from Nest (m)				95% Confidence Interval	
	N	Max	Mean	SE	Lower	Upper
Bridge Overwash	22	10804.7	1337.7	472.7	354.7	2320.6
Cupsogue Breach	32	12398.8	1248.2	538.9	149.2	2347.3
Cupsogue Mud Flat	32	14650.7	7032.2	932.0	5131.3	8933.1
Cupsogue Overwash	52	12119.5	1310.1	380.8	545.7	2074.5
Cupsogue Sand Bar	52	13551.9	3854.8	594.8	2660.7	5048.9
Great Gun Restoration Area	13	4489.4	1938.1	500.2	848.2	3028.0
Narrow Bay Breach	15	5729.1	799.1	377.6	-10.8	1609.1
NBE Overwash	10	1499.3	587.6	159.1	227.7	947.6
NBW Overwash	34	7280.1	1883.0	391.6	1086.2	2679.8
New Made	5	387.0	376.7	10.3	348.0	405.4
OIE Overwash	248	13134.4	1403.7	174.1	1060.7	1746.6
OIW	55	35807.7	4114.8	737.2	2636.8	5592.9
OIWR Overwash	1	13104.0	13104.0	-	-	-
Old Inlet Flood Shoal	27	12959.6	3550.4	650.8	2212.7	4888.1
PSO Overwash	128	7414.1	1833.9	210.1	1418.2	2249.6

We assessed average movement of piping plover fledglings from their nests by year. In 2014, the average distance that piping plover fledglings were observed from their nests was 1,954 m (SE = 216; Figure 11). In 2016, the average distance that piping plover fledglings were observed from their nests was 2,893 m (SE = 567), and by 2018 the average distance increased to 4,445 m (SE = 517). Distance moved by fledglings showed a slight decrease between 2014 and 2015 and 2016 and 2017. However, there was an overall increasing trend in average distance fledglings were observed from their nest relative to time since Hurricane Sandy.

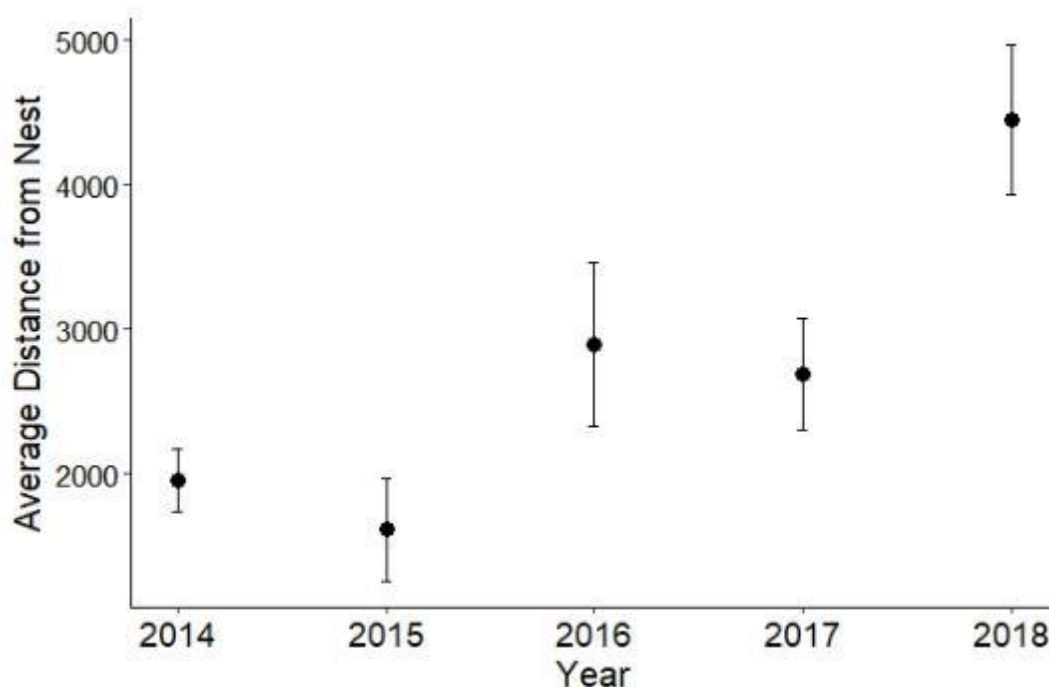


Figure 11. Average distance moved by piping plover fledglings by year from their nest location on Fire Island, June–August, 2014–2018. Error bars represent standard error.

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

In 2018, we weighed 41 nests at full clutch. There was no difference in the clutch masses of piping plovers that had bay access from those that did not have bay access ($W=173$, $df=1$, $P=0.42$, $\bar{x}=39.36$). Similarly, from 2013–2018, we weighed 163 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 ($W=3067.5$, $df=1$, $P=0.84$, $\bar{x}=39.22$).

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

Nest Survival

In 2018, nests that were exclosed during laying or after clutch completion had a 72% predicted probability of surviving to hatching. Nests that never were exclosed had a 48% predicted probability of surviving to hatching (Figure 12). The daily survival rate for exclosed and unexclosed nests was 0.99 and 0.98, respectively. Nests that were exclosed also had a lower abandonment rate (11%, Table 14) than nests that did not have exclosures (20%, Table 14).

When comparing across years, the probability of surviving to hatch varied (Table 15). 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 2013–2018, 35 of 266 nests were abandoned (Table 14). From the 35 abandoned nests, 7 nests belonged to a pair of unbanded or unknown adults, 11 nests belonged to a pair with only one banded adult and 17 nests belonged to a pair of banded adults. Of the 17 nests with a pair of banded adults, all nests had at least one individual resighted after the nest was abandoned, while 2 nests (11%) had both individuals

resighted after abandonment. Of the 11 nests with one banded adult, eight banded individuals (73%) were seen after the nest was abandoned. Of 45 banded birds from the 35 abandoned nests, 27 banded individuals (60%) were seen after the nest was abandoned. The resight rate of birds following nest abandonment is lower than the annual survival rate (see Objective 5), but not low enough to conclude that all nests were abandoned due to adult mortality.

In 2018, six banded adult mortalities were confirmed by either discovering carcasses of adults or by procuring bands or flags from predator fecal samples during field survey activities. All mortalities are believed to be predator-caused; at least one mortality was caused by a Peregrine Falcon (*Falco peregrinus*), and one mortality potentially mammalian caused. The nests belonging to three of the six dead adults were abandoned after the mortality occurred. Three of the nests survived until hatch. Of the three nests that survived to hatch, two nests fledged at least one chick.

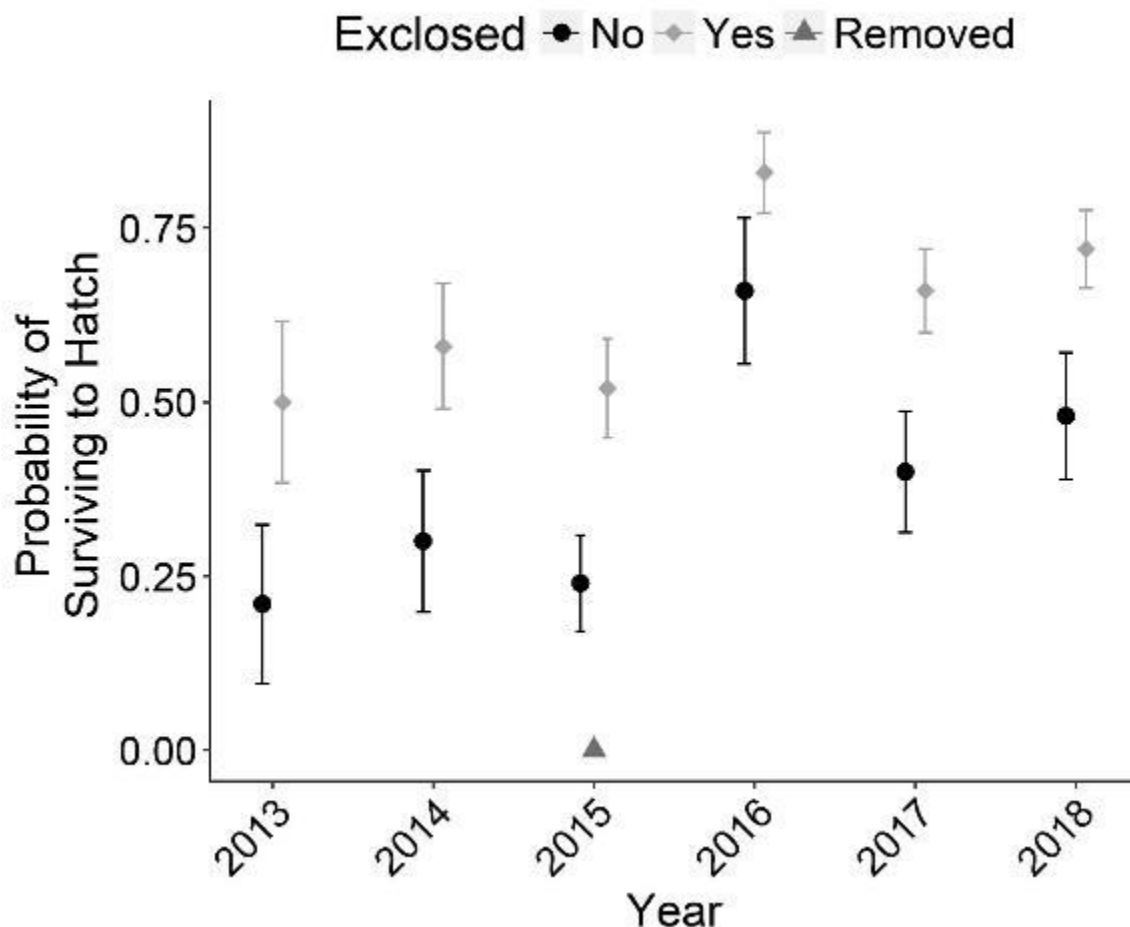


Figure 12. The probability of piping plover nests surviving to hatching with an exclosure and without an exclosure on Fire Island and Westhampton Island, 2013–2018. Error bars represent standard error.

Table 14. Abandonment rate of piping plover nests with exclosures and without exclosures from Fire Island and Westhampton Island 2013–2018.

Year	Caged	# of Nests	Failed	Hatched	Abandoned	Abandonment ^a Rate
2013	N	6	5	1	0	0
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	3	5	1	0.125
2016	Y	32	5	28	5	0.156
2017	N	17	15	2	5	0.294
2017	Y	45	7	38	4	0.089
2018	N	10	10	0	1	0.100
2018	Y	60	10	50	6	0.100
2015	Removed	5	5	0	0	0.000
All	N	75	64	16	9	0.120
All	Y	191	45	147	26	0.136

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

Table 15. Predicted probability of surviving to hatch (34 days from initiation) for all four management units in 2013-2018, 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	0.001
2017	Robert Moses State Park	12	0.42	0.15
	Fire Island National Seashore	14	0.7	0.13
	Smith Point County Park	24	0.77	0.09
	Cupsogue County Park	12	0.38	0.15
2018	Robert Moses State Park	19	0.70	0.11
	Fire Island National Seashore	13	0.74	0.13
	Smith Point County Park	31	0.65	0.09
	Cupsogue County Park	7	0.85	0.14

Chick Survival

Daily survival rates for chicks and probability to fledging varied among years (Figure 13) and among management units (Table 16). For 2018, relative to 2017, survival to fledge increased in Fire Island National Seashore, Robert Moses State Park, and Cupsogue management units, but declined in Smith Point County Park. 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 2 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. Four chicks were found dead in 2018. One chick was found dead by Suffolk County Park plover monitors in Burma Road, an off-road vehicle (ORV) sand road that extends along Smith Point County Park east to the Old Inlet. One chick was found dead in the nest bowl in subsite NBW approximately one day after hatching. The chick was observed breathing shallowly and positioned unnaturally in the bowl after hatch. The chick was not handled due to concerns of compromising chick health. Approximately 24 hours later, the chick was discovered dead in the bowl. A chick was found dead bayside NBW. Previous recapture and measurement data before the chick was found dead indicated the chick was small for its age and smaller than other chicks from that brood recaptured at the same time. In Robert Moses State Park, one chick was found dead by plover monitors with the skull detached from the body. No observations of predator tracks were present at either of these events.

For future analyses, we will be incorporating continuous measures of habitats (such as vegetation density and least cost distance to foraging habitat) to better understand variation in chick survival among habitat types.

Table 16. 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	Sfledge	SE(Sfledge)
Robert Moses State Park	2015	5	0.95	0.31	0.10
	2016	8	0.96	0.41	0.10
	2017	6	0.99	0.78	0.10
	2018	14	0.99	0.70	0.06
Fire Island National Seashore	2013	5	0.93	0.16	0.09
	2014	3	0.96	0.36	0.13
	2015	2	0.99	0.83	0.16
	2016	7	0.99	0.81	0.09
	2017	10	0.97	0.53	0.08
	2018	10	0.99	0.84	0.07
Smith Point County Park	2013	3	0.97	0.51	0.02
	2014	4	0.99	0.72	0.01
	2015	2	0.98	0.65	0.01
	2016	12	0.99	0.84	0.00
	2017	19	0.97	0.49	0.01
	2018	20	0.99	0.71	0.00
Cupsogue County Park	2013	3	0.87	0.03	0.04
	2014	4	0.97	0.43	0.12
	2015	7	0.89	0.06	0.04
	2016	6	0.94	0.22	0.08
	2017	5	0.96	0.38	0.12
	2018	6	0.97	0.49	0.10

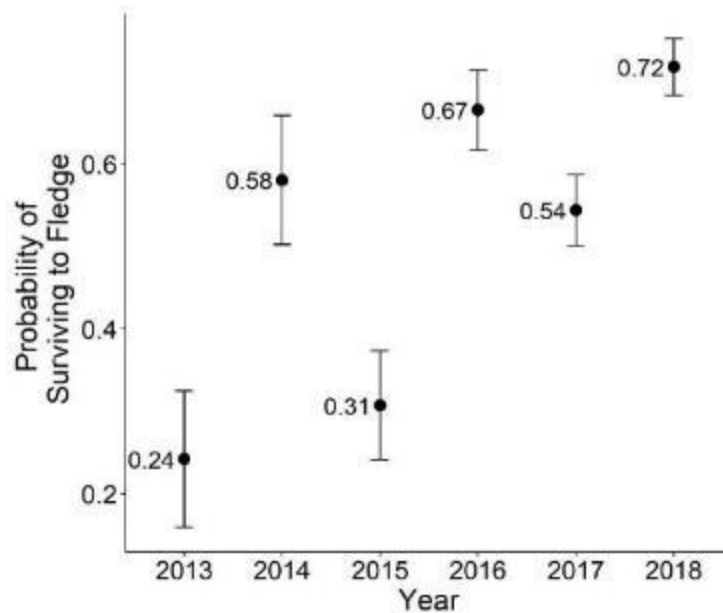


Figure 13. Probability of surviving to fledge (25 days) for chicks 2013–2018 on Fire Island and Cupsogue. Error bars represent standard error.

Productivity

In 2018, 66% of chicks survived to fledging. Average productivity was 1.98 fledges/pair. The highest productivity was at Fire Island National Seashore, whereas the lowest productivity was at Cupsogue Beach County Park (Table 17). When comparing across the study site from 2013–2018, 2018 had the highest rate of productivity and 2015 had the lowest with 0.39 fledges/pair (Table 17).

Chicks per pair seems to increase in areas that have experienced a decrease in red fox density following mange outbreaks (Figure 14), but that relationship is less obvious for chick survival. Both chick survival and productivity are consistently low at Cupsogue County Park, for which we have no information on the predator community.

Table 17. Piping plover pair counts and productivity (fledges/pair) in four different management units on Fire Island and Westhampton Island, May–August 2013–2018.

Management Unit	2013		2014		2015		2016		2017		2018	
	# Pair	Fledge /Pair	# Pair	Fledge /Pair	# Pair	Fledge /Pair	# Pair	Fledge /Pair	# Pair	Fledge /Pair	# Pair	Fledge /Pair
Robert Moses State Park	-	-	-	-	10	0.3	9	1.11	8	1.5	15	2.27
Fire Island National Seashore	10	0.6	8	0.75	3	1.67	8	2	11	1.56	11	2.64
Smith Point County Park	6	0.5	8	1.5	13	0.31	13	2.38	20	1.65	25	1.72
Cupsogue County Park	3	0	5	1.2	7	0.14	6	1	6	0.5	7	1.29
Total	19	0.47	21	1.14	33	0.39	36	1.75	45	1.44	58	1.98

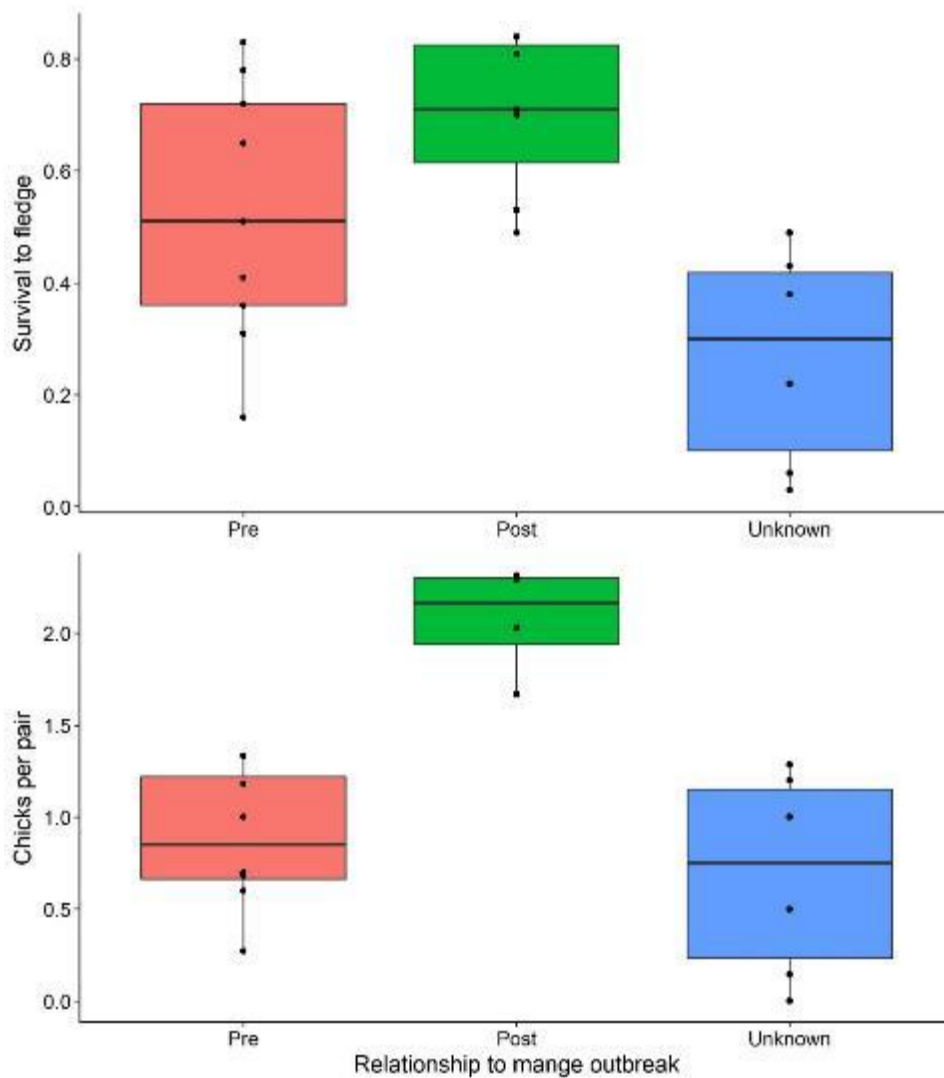


Figure 14. Survival to fledge (top) and chicks per pair (bottom) in relation to known mange outbreaks in the red fox population on Fire Island. Pre-mange outbreak includes 2013–2015 east of Old Inlet and 2013–2017 west of Old Inlet. Post-mange outbreak is all other dates. Unknown references Cupsogue County Park, as fox monitoring does not occur at Cupsogue County Park, and no foxes have been documented. Data points for survival to fledge are by management unit, and the West Wilderness area is grouped in with the East Wilderness area due to small sample size. Data points for chicks per pair are productivity for the entire portion of the island, east or west of old inlet. In 2013–2014, data for Robert Moses were primarily collected by the land management agencies.

Brood Home Range

In 2018, brood home range size increased as broods aged (Figure 15). 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 and only until a brood reached 25 days old, at which point we considered them fledged ($n = 115$). For the plover broods in 2018 that had at least one chick survive to fledging and with enough points to estimate a home range size ($n = 44$), 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, CSW and DMP) as compared to those that did not have direct access to the bay ($W = 309$, $P = 0.11$).

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 ($\bar{x} = 3.69\text{ha}$, $\text{SE} = 0.76$) than broods that did not have such access ($\bar{x} = 5.99\text{ha}$, $\text{SE} = 0.31$, Figure 14, $W = 2105$, $P = 0.005$) (Figure 16). This result remained

true when accounting for the number of points used to estimate the home range in a linear regression. We also found that movement between observations of broods was smaller, on average, for broods that had bay access ($\bar{x} = 106.4$ m, SE = 3.68) than broods that did not across all years ($\bar{x} = 160.1$ m, SE = 5.74; $W = 416090$, $P < 0.001$). Finally, we found no significant difference between brood home range area and accessible foraging habitats (K-W $\chi^2=3.15$, $P=0.53$; filled breach, restoration, natural overwash, manipulated overwash, and other; Figure 17). Individual brood home ranges throughout the study site are displayed in Figure 18, Figure 19, Figure 20, Figure 21, and Figure 22, and are symbolized by polygons stretching the extent of each home range based on Trimble GPS coordinates collected while observing broods.

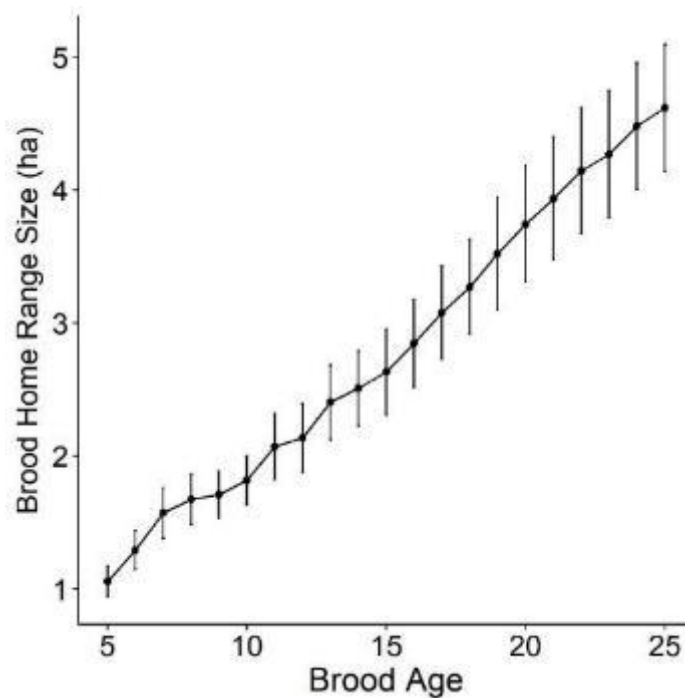


Figure 15. Cumulative home range size for piping plover broods as age increased. Home ranges were only calculated for age classes where up to that point, the brood had at least 5 locations. Error bars represent standard error.

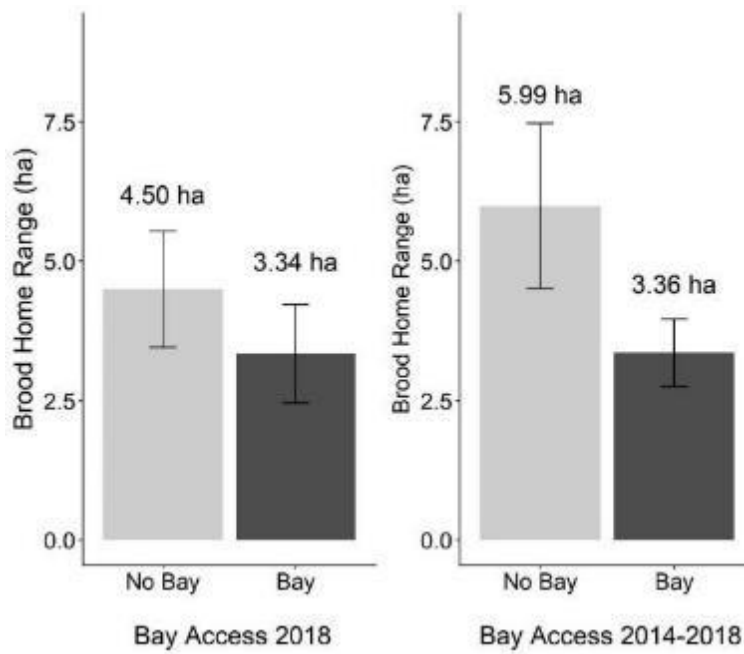


Figure 16. Brood home range size for 2018 (left) and 2014–2018 (right), estimated using minimum convex polygons for broods that fledged at least one chick. Error bars represent 95% confidence intervals.

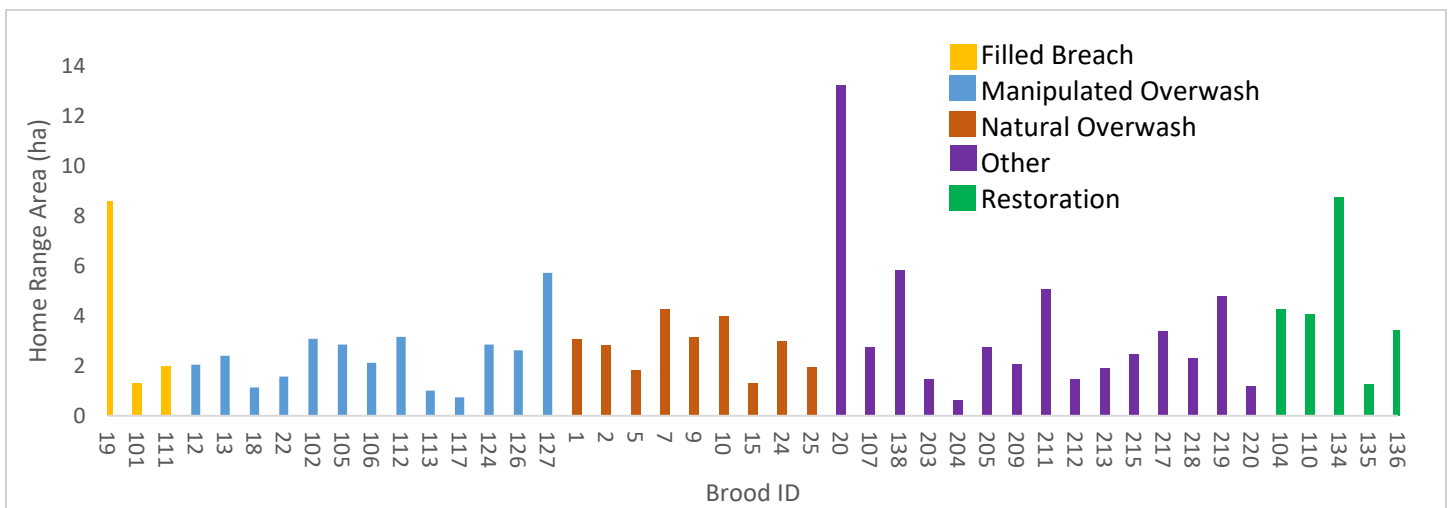


Figure 17. Brood home range area of 2018 piping plover broods that foraged on filled breach, other, restoration, natural overwash and manipulated. This figure only shows broods that had ≥ 10 home range location points available.

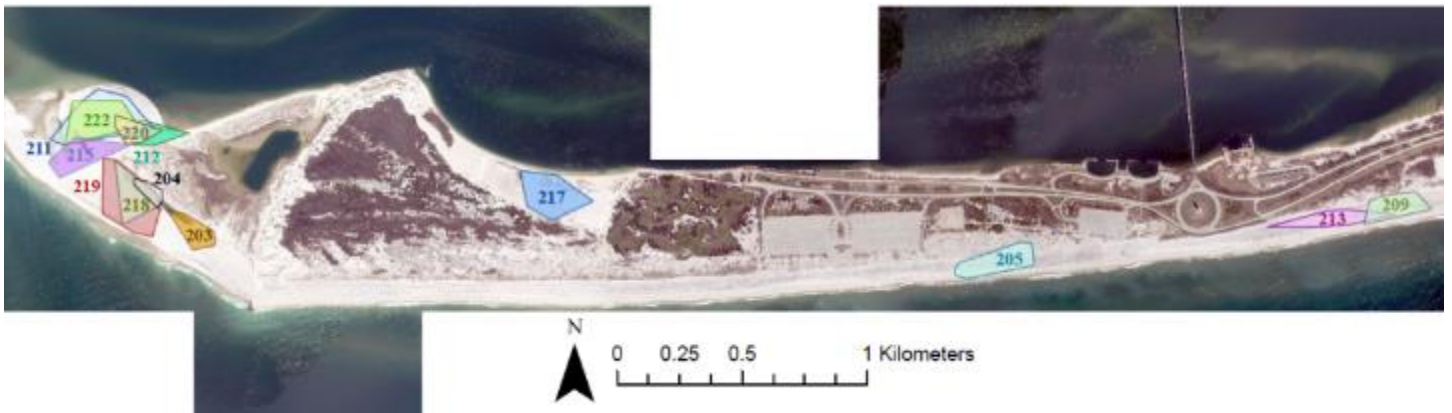


Figure 18. Brood home ranges of piping plovers at Robert Moses State Park 2018. Imagery captured April 2018.



Figure 19. Brood home ranges of piping plovers at Old Inlet East. Imagery captured in April, 2018.



Figure 20. Brood ranges of piping plovers at the western section of Smith Point County Park (including subsites: PSO through SPTW). Imagery captured April 2018.

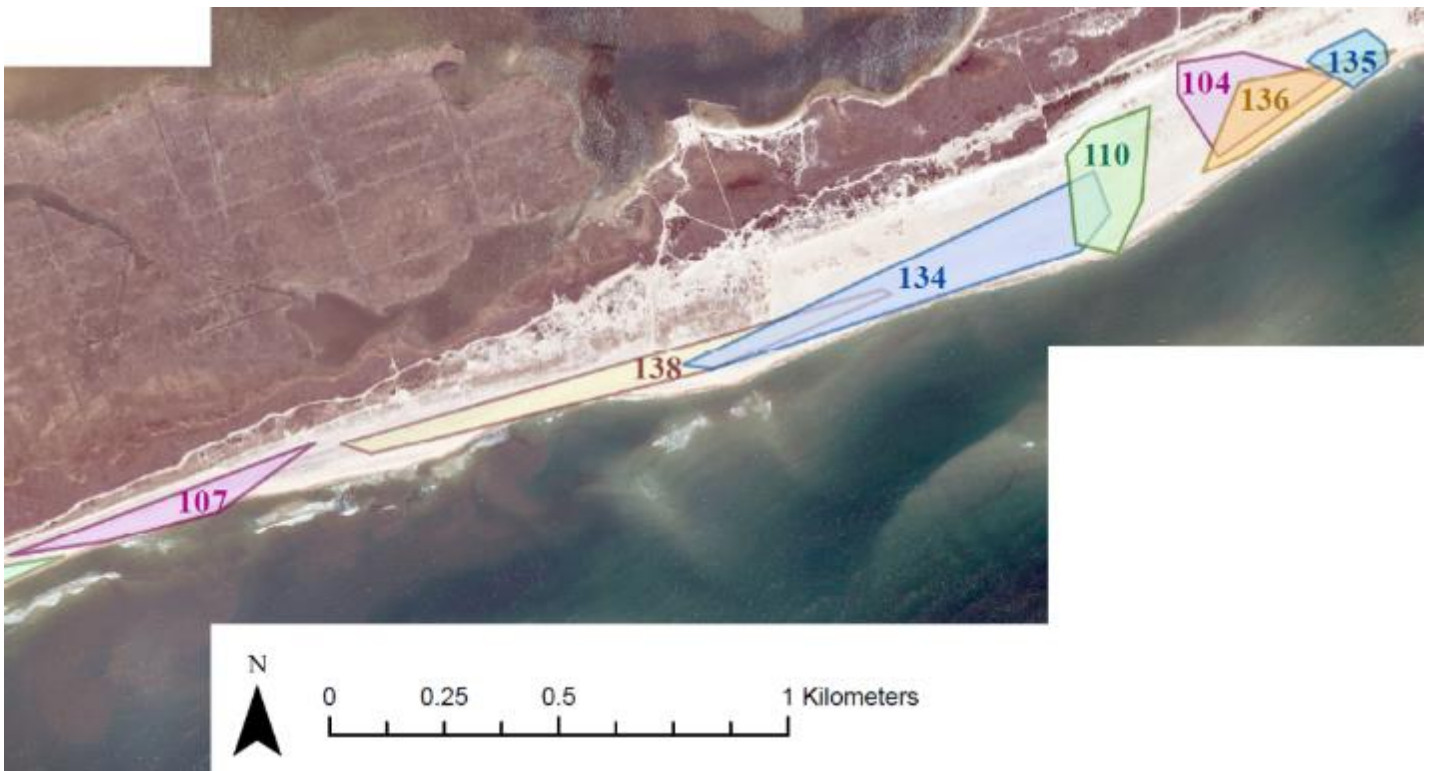


Figure 21. Brood ranges of piping plovers at the western section of Smith Point County Park (including subsites: SPTW and SPTE (Great Gun)). Imagery captured April 2018.

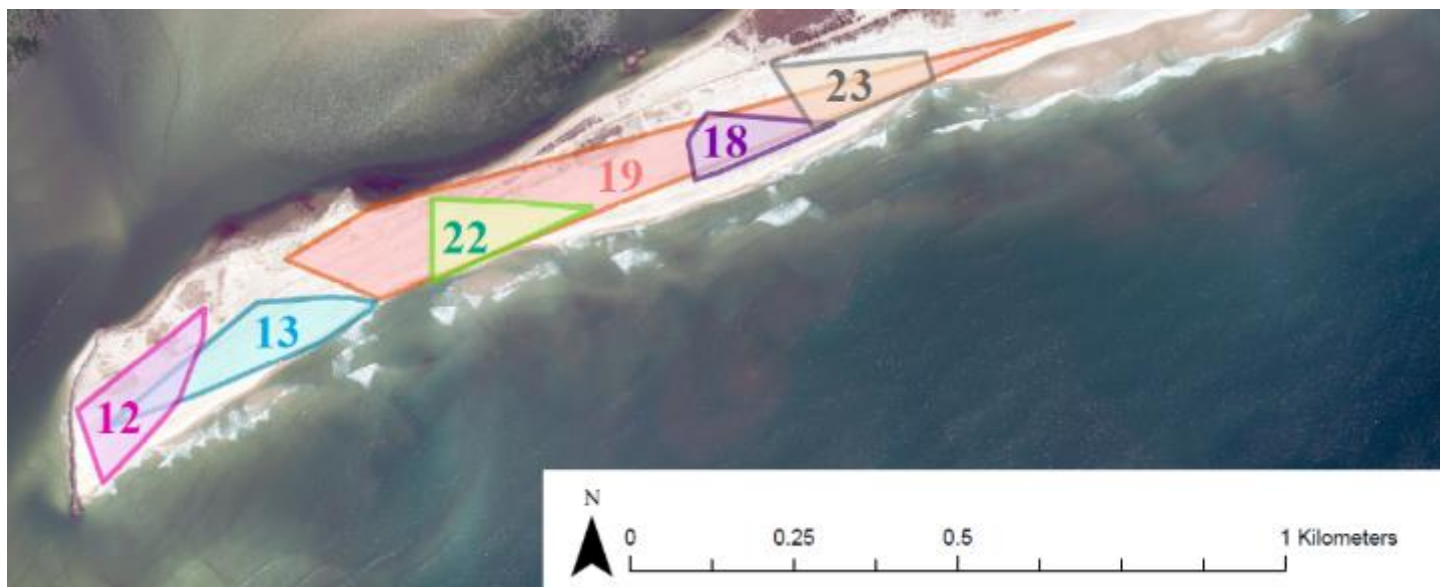


Figure 22. Brood ranges of piping plovers at Cupsogue County Park. Imagery captured April 2018.

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

Immigration and recruitment

From 2014 to 2018, the breeding population was made up of 31–53% returning nesters annually (Table 18). The remainder of the birds that nested in the study area hatched in the study area the year before (local recruits; 6–19%) or were unbanded birds captured during the year (immigrants; 32–63%). 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 Pradel model did not include support for annual variation in immigration rates, as long as Democrat Point in 2015 was accounted for (Table 19).

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 19). 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 19, Table 20, Table 21).

Table 18. 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 2016–2018 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 (53%)	5 (13%)	13 (32%)	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)
2018	49 (46%)	14 (13%)	44 (41%)	0.66 (0.11)

^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 19. 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	16	3 (20%)	4 (27%)	2 (13%)	3 (20%)	80%
2018	14	1 (7%)	2 (14%)	0	6 (43%)	64%
Total	41	5 (12%)	9 (22%)	3 (7%)	13 (32%)	73%

Table 20. 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	17	3 (19%)	8 (50%)	2 (13%)	NA	81%
2015	13	1 (8%)	5 (38%)	1 (8%)	3 (23%)	77%
2016	42	7 (27%)	7 (27%)	1 (4%)	4 (15%)	73%
2017	36	9 (25%)	6 (17%)	3 (8%)	7 (19%)	69%
2018	44	2 (5%)	7 (16%)	3 (7%)	6 (14%)	41%
Total	152	22 (15%)	33 (22%)	10 (7%)	20 (13%)	56%

Table 21. 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	17	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%
2018	49	15 (31%)	12 (25%)	3 (6%)	4 (8%)	69%
Total	139	37 (27%)	43 (31%)	17 (12%)	7 (5%)	75%

We assessed settlement patterns in relation to classified dry sand in 2017 for the eastern study area (OIWR to CSE). In 2017 the population was larger (45 pairs), thus the sample size was larger and more selection patterns were apparent relative to settlement patterns in 2015 (32 pairs; Granger et al. 2017). In 2017, immigrant piping plovers avoided pre-Sandy “other” habitat ($\chi^2_{4^2} = 20.89$, $P < 0.001$). Returning nesters selected manipulated overwash and natural overwash, avoided breach fills and restoration areas, and did not use pre-Sandy habitat ($\chi^2_{4^2} = 60.24$, $P < 0.001$, Figure 23). Natal recruits avoided pre-Sandy habitat (“other”) relative to its availability ($\chi^2_{4^2} = 17.15$, $P = 0.002$). When immigrants and natal recruits are combined into a “new birds” category, they avoided pre-Sandy habitat ($\chi^2_{4^2} = 31.76$, $P < 0.001$).

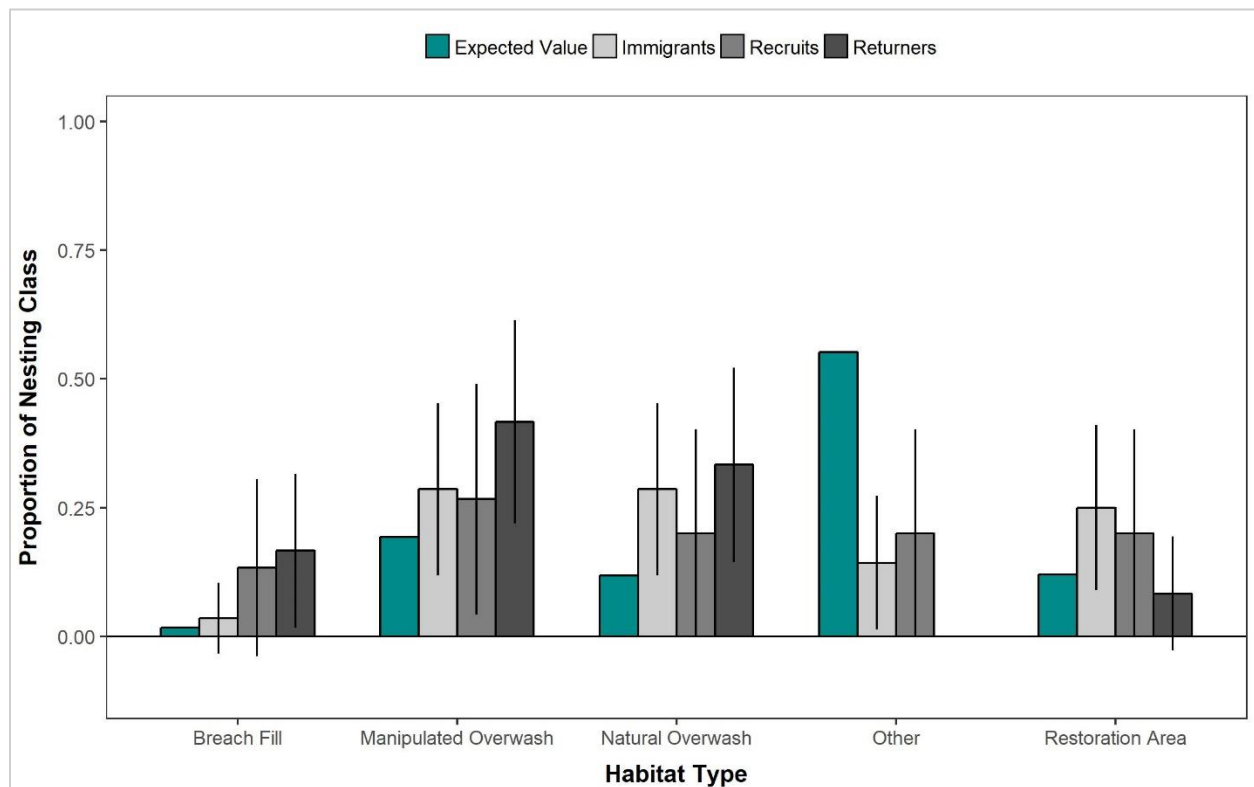


Figure 23. Used habitat by immigrant, natal recruit and returning piping plovers in 2017 (and 95% confidence intervals), compared to the expected amount of use based on availability of habitat in 2017. The expected amount of use is represented in teal, and selection is evident when the expected amount is not within the confidence intervals of the used amount.

Survival and site fidelity

The top model from the Barker model indicated varying survival for HY birds among years, a constant adult survival rate for all years, and an age effect on site fidelity (F). Estimates are reported with standard error. Point estimates of HY survival varied each year of the study, with high years of HY survival relating to years of high chick survival to fledgling (Figure 24). AHY survival ($S = 0.69 \pm 0.03$) is within the range of past studies of adult piping plover survival, and is within the confidence intervals of our estimate from 2013–2017 (Granger et al. 2017). Site fidelity for AHY birds ($F = 0.89 \pm 0.03$) was higher than for HY birds ($F = 0.69 \pm 0.06$), and HY site fidelity has declined with the addition of the 2018 data. Detection probability was constant over both ages and all years ($p = 0.94 \pm 0.03$). Post-fledgling survival, survival from fledging to returning the following year, varied substantially among years. The year of lowest post-fledge survival was 2015 and the highest post-fledge survival was 2016 (Figure 25).

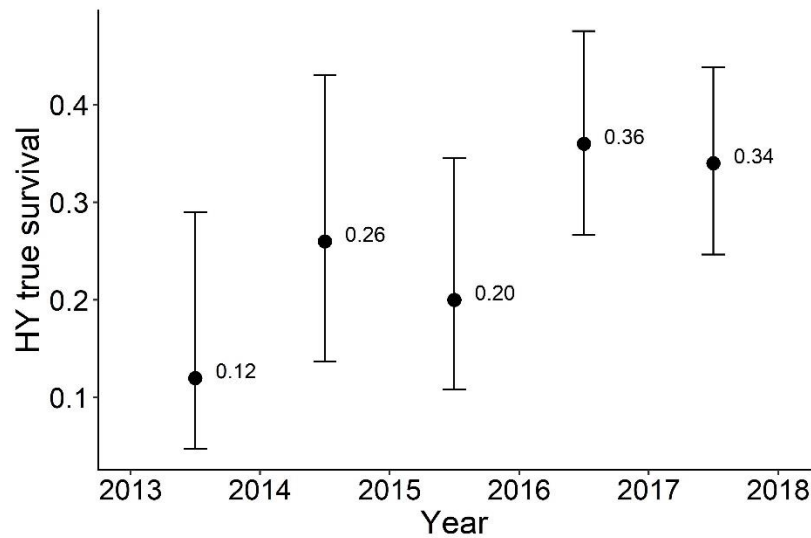


Figure 24. True survival (S) of hatch-year (HY; from hatch to year 1) piping plovers on Fire Island from 2013 to 2018. Estimates are from the top model with an additive effect of year and age (HY or AHY). Error bars represent 95% confidence intervals.

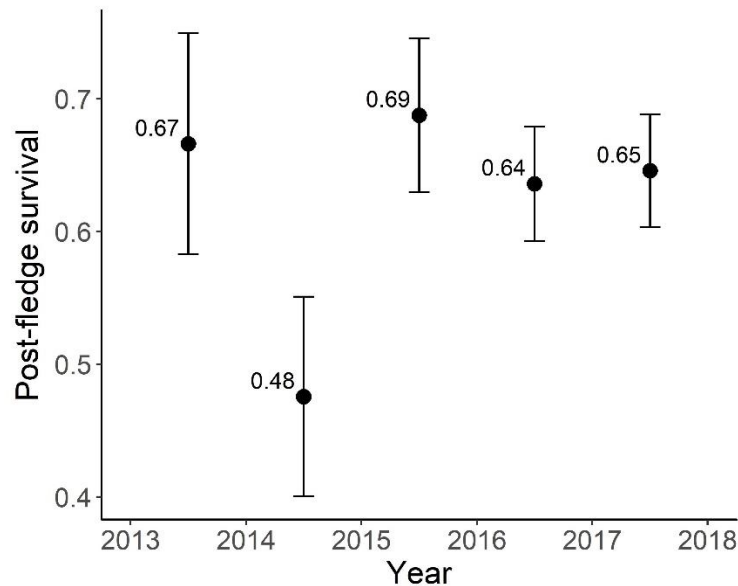


Figure 25. Post-fledge survival of piping plovers on Fire Island, New York from 2013 to 2018. Post-fledge survival is the survival from fledgling to returning the following year, and is calculated by dividing hatch year survival by pre-fledge chick survival. Error bars represent 95% confidence intervals.

Objective 6: Compare prey densities at several habitat types

Terrestrial Invertebrates

In 2018, 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= 32.71, 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 22).

Table 22. 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 2018.

Geologic Location	Mean Arthropods/ 30min	SE	N
Bay Beach	2.30	0.30	83
Barrier Interior	1.01	0.12	209
Ocean Beach	0.93	0.17	95

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 23, Figure 26). 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 26).

Table 23. Mean predicted number of invertebrates caught over a three-hour period in various habitats from 2014–2018. 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	21.60	3.11	11.84	2.47	10.23	3.05
Ephemeral Pool	37.21	7.83	20.40	5.15	17.63	5.83
Ocean Beach	10.07	1.40	5.52	1.10	4.77	1.32
Barrier Interior	5.99	0.76	3.29	0.62	2.84	0.78

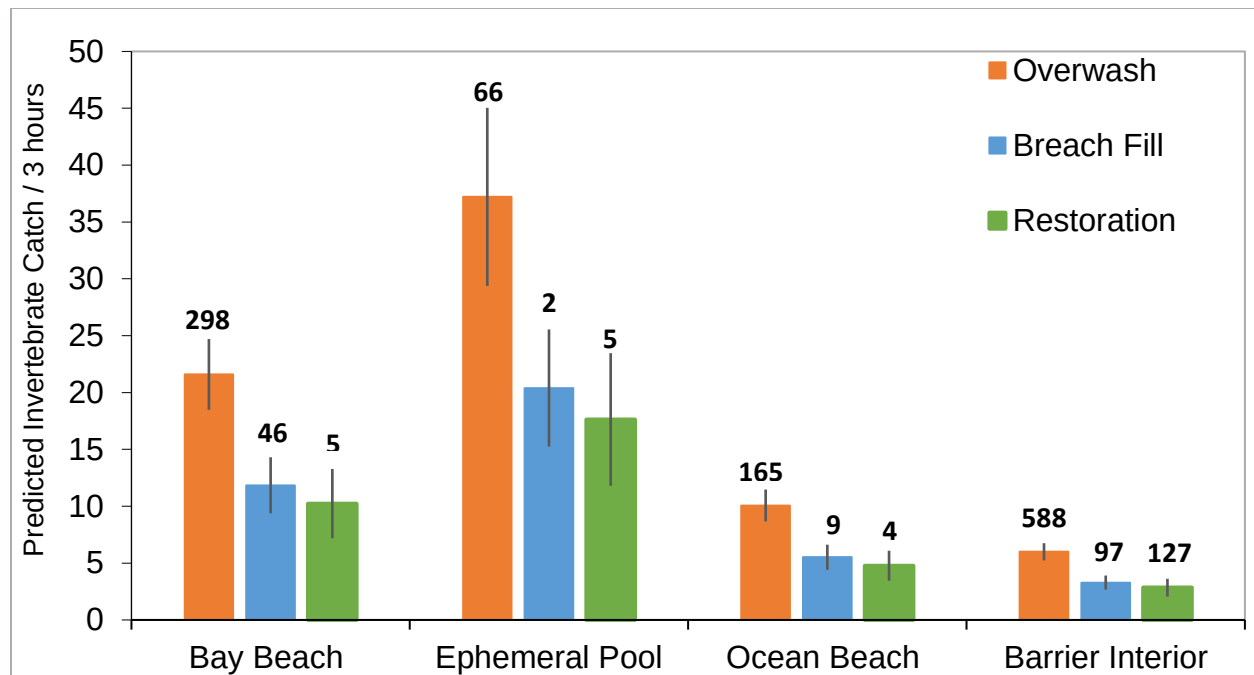


Figure 26. Invertebrates caught over a projected three-hour period in various habitats from 2014–2018, predicted using a negative binomial mixed model. Invertebrate samples were collected over 30-minutes, but projected to 3-hours to facilitate comparison to similar piping plover studies. Numbers above bars indicate sample sizes. Error bars represent standard error. Sample sizes differed among habitats because samples were collected in proportion to the habitat availability.

Fecal Samples

We collected 44 fecal samples in 2013–2014 for visual identification and 72 samples in 2016–2017 for DNA metabarcoding. No fecal samples were collected in 2018.

Polychaetes (Annelida) were detected in 8% of DNA samples and 14% of visual samples (Figure 27). Unidentifiable Arthropods were detected in 54% of visual samples (Figure 28). In visual identification samples, we detected 8 orders, 5 families, and 1 species. In DNA metabarcoded samples we detected 29 orders, 73 families, and 116 genera. There was a higher frequency of occurrence for flies (Diptera) in DNA barcoding than visual identification. Some notable findings included horeshoe crab DNA, larvae, and eggs in fecal samples. Sand lance (Uranoscopiformes) DNA was detected, a common food item for least and common terns (*Sternula antillarum*, *Sternula hirundo*). We found DNA from orders which include skate (Rajiformes), Atlantic cod (Gadiformes), and eel (Anguilliformes) DNA using DNA metabarcoding, suggesting that plovers are possibly consuming eggs or dead organisms opportunistically. We also had no conclusive evidence of bivalve prey, but we did have 1 fecal sample with gastropod DNA detected.

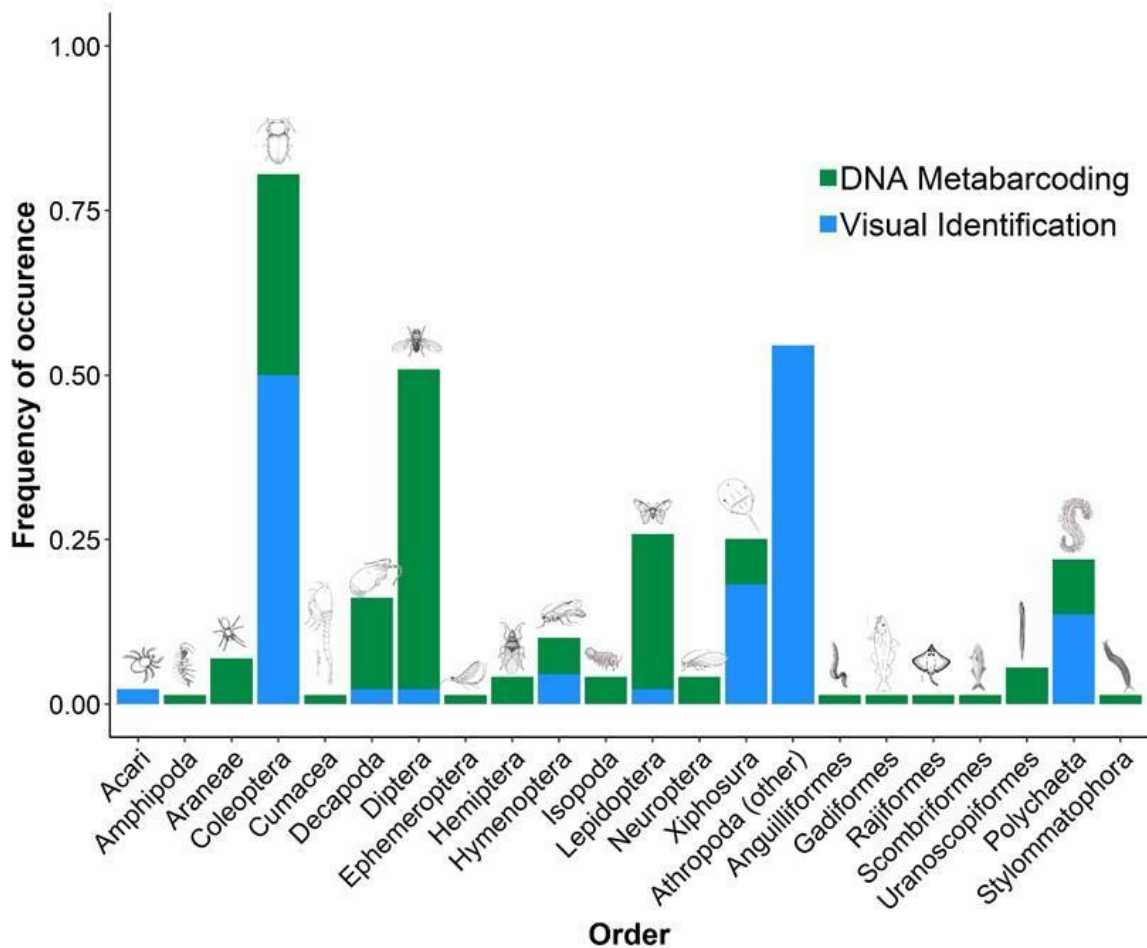


Figure 27. Frequency of occurrence for plover fecal samples with each identified taxon. Fecal samples using visual identification were collected 2013–2014, and fecal samples using DNA metabarcoding were collected 2016–2017. Frequency in fecal samples that were identified using DNA metabarcoding in green and visual identification in blue. Arthropod (other) indicates arthropods identified in fecal samples that could not be identified past phylum.

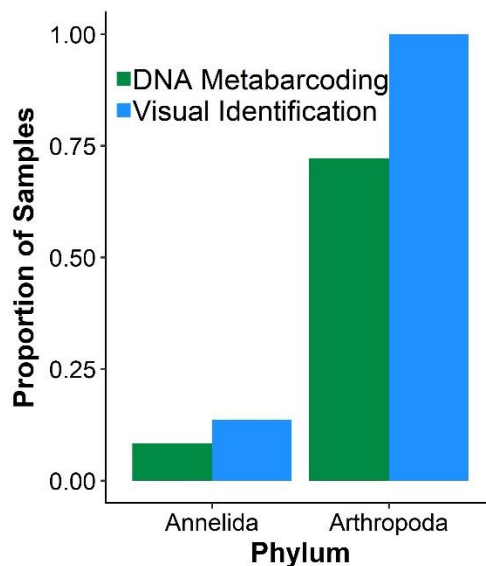


Figure 28. Proportional abundance of Arthropoda and Annelida detected in fecal samples on Fire Island and Cupsogue, using DNA metabarcoding and visual identification.

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

In 2016, we placed ten remote nest cameras across the study site, with one camera in Smith Point County Park, two in Cupsogue Beach County Park, four in Fire Island National Seashore, and three in Robert Moses State Park. The length of time a camera was present on an exclosure ranged from 3–30 days. The highest volume of detections in 2016 included red fox and crow, with 26 and 25 observed detections respectively. The lowest detections were raccoon, mourning dove, willet, and an unknown mammal, with one detection each (Table 24). In 2017, we placed 23 remote nest cameras across the entire study site, with eight cameras in Smith Point County Park, two in Cupsogue Beach County Park, seven in Fire Island National Seashore (east of Old Inlet breach), and six in Robert Moses State Park. The length of time a camera remained present on an exclosure ranged from 2–26 days. In 2017, gulls and humans had the highest volume of detections, with 383 and 247, respectively. The taxa with the lowest number of detections were owl, rabbit, and rodent with one detection event each (Table 24). We placed 24 remote nest cameras on exclosures during the 2018 field season: eight in Smith Point County Park, three in Fire Island National Seashore, four in Cupsogue Beach County Park, and nine in Robert Moses State Park. The length of time a camera remained present on an exclosure ranged from 2 to 40 days. The highest volume of detections in 2018 included unknown birds and humans, at 225 detections and 142 detections respectively. The lowest volume of detections in 2018 with one occurrence each were a black-backed gull, a rabbit, and a Semipalmated Plover. Detections from 2018 of interest include one snowy owl with 9 detections within subsite SPTE at Smith Point County Park, one peregrine falcon with 9 detections at NBW and BRO at Smith Point County Park and Fire Island National Seashore, and a red fox detected twice at FLD4 at Democratic Point.

Table 24. Number of nest camera detections per year, subsite, and what was detected in 2016–2018. See Table 3 for subsite descriptions.

Year	Subsite(s) of Nest Camera	Detection ID	Number of Detections
2016	BRO	Raccoon	1
	SPTE	Unknown mammal	1
	BRO	Willet	1
	OIE, OSS	Mockingbird	3
	OIE, BRO, OSS	Unknown	7
	CSE, BRO, DMP	Gull	8
	SPTE, BRO, CSW	Black bird	9
	BRO, DMP	Deer	12
	OIE, CSE, CSW, DMP, OSS	Human ^a	15
	OIE, DMP, OSS	Toad	15
	BRO	Unknown bird	15
	OIE, BRO	Crow	25
	DMP, OSS	Red fox	26
2017	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
2018	CSW, SPTE, DMP, FLD4, OSS	Gull	383
	SPTE	Black-backed gull	1
	CSE	Rabbit	1
	SPTE	Semi-palmated plover	1
	OIE, STP	Crow	2
	FLD4	Red fox	2
	PSO	Mourning Dove	2
	OIE, PSO	Willet	2

2018 (cnt'd)	CSE, CSW	Dog	3
	CSE, BRO, NBW	American oystercatcher	4
	BRO, PSO, NBW	Common grackle	4
	CSW, BRO	Red-winged blackbird	6
	CSW, BRO, SPTE, PSO, DMP	Unknown	7
	BRO, NBW	Peregrine falcon	9
	SPTE	Snowy owl	9
	OIE, BRO, PSO, SPTE	Toad	14
	CSE, SPTE	Least tern	30
	DMP, SPTE, CSE, OIE	Tern	32
	CSE, OIE, CSW, SPTE, DMP, FLD4, FLD3, OSS	Vehicle	47
	BRO, SPTE, PSO, NBW, DMP, FLD4	Deer	50
	OIE, CSW, CSE, BRO, SPTE, PSO, NBW, DMP, FLD4	Gull	70
	OSS, OIE, CSE, CSW, BRO, PSO, SPTE, DMP, FLD3, FLD4, STP	Human ^a	142
	OIE, CSW, CSE, BRO, SPTE, PSO, NBW, DMP, FLD4	Unknown bird	225

^a Only including non-monitors

Discussion

Barrier islands are subject to frequent changes in habitat distribution and quality, which may substantially affect organisms that rely on them. Hurricane Sandy and subsequent beach engineering caused significant habitat change, potentially affecting nesting piping plovers. We studied the piping plovers, their habitats, and their prey on Fire Island and a portion of nearby Westhampton Island, and we aimed to assess the effects on of Hurricane Sandy on piping plovers and their invertebrate prey.

The available habitats on Fire Island to Westhampton Island for nesting and brood-rearing territories included overwashes (natural and manipulated), breach fills, restoration areas, 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 were filled in. The restoration areas were New Made and Great Gun restoration areas, which were created prior to the 2015 nesting season. Prey availability in different habitat types likely drives both nest-site and foraging habitat selection, thus effecting chick survival (Loefering and Fraser 1995). 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 potentially higher survival rates (Catlin et al. 2014).

Piping plovers traveled relatively long distances to forage on 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. Specifically, fledglings 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. The sandbar is frequently accessed by boaters during the summer months, and the mudflat is frequently accessed by boaters, shell fishers and camper, so piping plover use of these habitats may benefit from protection. A portion of these movements could be prospecting for future nesting sites, adding support for more protection of important plover sites during migration (Davis et al. 2017).

The Fire Island plover population has been composed of at least 50% immigrants and natal recruits each year. Each habitat type of interest is affecting the retention and attraction of nesting plovers differently. Many of those individuals are using the newly created habitat, and most are selecting against any pre-Sandy oceanfront habitat in relation to availability. 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 nest site selection. Also, in 2017, returning nesters used overwashes more than expected compared to availability. These results contrast with the 2015 settlement patterns results, where we found that returning nesters selected breach fills, and natal recruits (of which there were few) did not use pre-Sandy habitat, natural overwashes, or breach fills (Granger et al. 2017). In each year that the restoration areas were available, at least one new local recruit and immigrant nested in them. However, in 2017, we found that returning nesters selected against the restoration areas and breach fills. That selection patterns seem to be changing as time since Hurricane Sandy increases may be a result of the vegetation succession in these habitats. Furthermore, some of the pre-Sandy “other” habitat may not have been suitable for nesting piping plovers due to proximity to vegetation, beach width and other influential variables (Walker et al. *in revision*), and thus further understanding of settlement patterns will be improved by taking into consideration habitat suitability rather than just the availability of dry sand.

The immigration rate has steadily declined over the last few years, perhaps suggesting that immigration decreased with increasing density. However, we have not observed a similar pattern with reproductive output, as reproductive output and chick survival has remained relatively high for the last several years of the study. Continued investigation into the relationship with density and demographic rates is ongoing.

More plovers nested in restoration areas in 2018 than in previous years, however, low hatch success rate in restoration areas was due to a proportionally high number of predated and abandoned nests. Four out of the ten nests initiated in the Great Gun restoration area in 2018 were predated, and three nests were abandoned. The returning piping plover population initiated more nests in natural and manipulated overwash rather than restoration areas, which natal recruits and immigrant populations used more than the returning population. Local recruits and immigrant populations are likely key in stabilizing and maintaining the plover population on Fire Island. It will be crucial to reduce or impede vegetation growth in restoration areas to prevent habitat degradation and maintain nesting habitat for the incoming birds attracted to these habitats.

Six adult mortalities were confirmed across the study site in 2018. Five of the mortalities occurred in the Smith Point County Park area and one in the Fire Island National Seashore. All mortalities are believed to be caused by predator interactions, with at least one confirmed Peregrine Falcon (*Falco peregrinus*) caused mortality, and one potential mammalian caused mortality. Further investigation on relationships between habitat use by plovers and predators with regard to productivity will benefit management decisions.

During all years of the study, exclosures were placed around nests during the laying period or after clutch completion. Typically, within the study site, exclosures are not installed around nests with fewer than four eggs to prevent nest abandonment. Two of the exclosed nests in 2018 within Smith Point County Park had signs of potential fox disturbance and digging along the edge of the exclosure. Predators have been known to cue in on exclosures as a potential food sources (Murphy et al. 2003). After the digging incidents, exclosures were not placed on several nests throughout the eastern portion of Smith Point County Park due to concern that exclosures would continue to attract predators. During this time, five unexclosed nests were predated within Smith Point County Park. After these predation events, exclosures were installed around nests with at least three eggs to prevent future predation. Nest abandonment in Smith Point County Park remained low (13%), whereas nest depredation contributed highly to nest failure (23%). In one study examining nest exclosures, exclosing nests resulted in greater population growth rates than if nests were left unexclosed. Furthermore, nest abandonment rates within the same study were more likely to be caused by an adult mortality (78%) and adult female mortality (39%, Cohen et al. 2016) than by abandonment of exclosed nests. Continuing to exclose Fire Island piping plover nests before they reach four eggs may reduce predation thus improving overall productivity in the Fire Island piping plover population.

Nests had a higher probability of hatching in 2018 than in all years except 2016. In 2015, Smith Point County Park had high fox density, and foxes 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. The mange outbreak also extended to Fire Island National Seashore and Robert Moses State Park starting in 2017 and extending into 2018, which may have contributed to higher regional nesting and productivity rates. At least one fox, however, was present east of Old Inlet during the breeding season of

2018, and it was suspected to have depredated at least one plover nest and attempted to dig under two exclosures.

We saw an increase in productivity east of Old Inlet following the mange outbreak in 2015–2016, and an increase in productivity west of Old Inlet following the mange outbreak in 2017–2018. Median chick survival follows the same relationship but the data have a wider spread, suggesting that the influences of the fox population on piping plover productivity contribute to nest and chick survival concurrently. 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 high (2.38 and 2 fledges/pair), more than twice the reproductive output in the other management areas, suggesting an important role of 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 an 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 fledge/pair). In 2017 the Fire Island management units had similar reproductive output (1.50–1.65 fledge/pair), all substantially above the level needed for population stationarity (1.25 chicks/pair). Recreational disturbance (DeRose-Wilson et al. 2018) and habitat variability (Loefering and Fraser 1995, Elias et al. 2000, Cohen et al 2009), likely affected these results. Fox numbers and movements in 2018 are still being analyzed, however, at least one fox was present in Smith Point County Park and the Fire Island National Seashore, east of Old Inlet and reproductive output in 2018 (1.29–2.64 fledge/pair) remained high even with the presence of foxes. There appears to be a relationship between piping plover productivity and the timing of mange outbreaks the fox populations.

We did not see a large population increase on Fire Island immediately following Hurricane Sandy, as was seen following some past habitat creating events (Wilcox 1959, Cohen et al 2009, Hunt 2016, Robinson et al. *in prep.*). However, we had an increase from 30 to 58 pairs (93% increase) from 2012–2018, and the population increase is beginning to look similar to other populations following habitat creation (Robinson et al. *in prep.*). We saw a 28% increase from 45 to 58 pairs between 2017 and 2018. The slower initial 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, if there was no habitat effect, we would have expected a population decline or stationarity through 2014, due to a lack of potential recruits. In 2018, reproductive output (1.98 chicks/pair) was well above the level need to replace expected losses of breeders, thus there is a good chance that the population will increase again in 2019 if there is sufficient habitat.

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 on Fire Island. This conclusion is further supported by past studies of plovers, where population increases followed 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. 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 of the effect of habitats created by Hurricane Sandy, and of the effect of predators, on piping plovers are ongoing.

Part II: 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 one of the most commonly targeted species in predator management activities for shorebird conservation along the U.S. Atlantic Coast (Hunt et al. 2018). On Long Island, Cohen et al. (2009) showed that plover reproductive output in West Hampton 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 effects varying across years. Management of foxes and other avian and mammalian predators is an important part of piping plover management along the entire U.S. coast. Management of red foxes at these sites on Long Island, whether through lethal removal of adults, destruction of active den sites, or simple nest exclosure, may be important for piping plover populations (USFWS 1996, 2014). Nonetheless, 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). However, the red fox monitoring efforts encompassed the entirety of Fire Island, as GPS-collared foxes regularly used areas outside of the parks during the monitoring periods.

Sign, scat, and den surveys

Field methods

We used ArcMap 10.3 GIS software to create and overlay a 300-m x 300-m grid onto each of the 3 study subsites: Robert Moses State Park, Fire Island National Seashore, and Smith Point County Park. 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 2015–2018 (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 was greater than 600 m at its widest point. Thus, all foxes present should be detected based on the overlap of our grid size and the observed movements from Gieder (2015) and our own collars.

We established walking transects from the ocean beach high tide line to the bayside high tide line. We walked each transect across the island 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 October/November, December, and January (2015–2018, Figure 29). Transects generally connected the center points of the grid cells described above across the island such that each transect was 300 m from the next. We walked on game trails in as close to a straight line as possible between the grid cell center points from ocean beach to bayside. All potential red fox scats located on transects identified as ≤ 1 month old were collected and labeled by estimated freshness (Table 25) and a unique sample number corresponding to the GPS waypoint was recorded. Scats were frozen for later genetic confirmation of the species that deposited the scat and for analysis of prey items. All observed red foxes were recorded and GPS locations noted. All inactive and active red fox den sites also were 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 and 2018, 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.

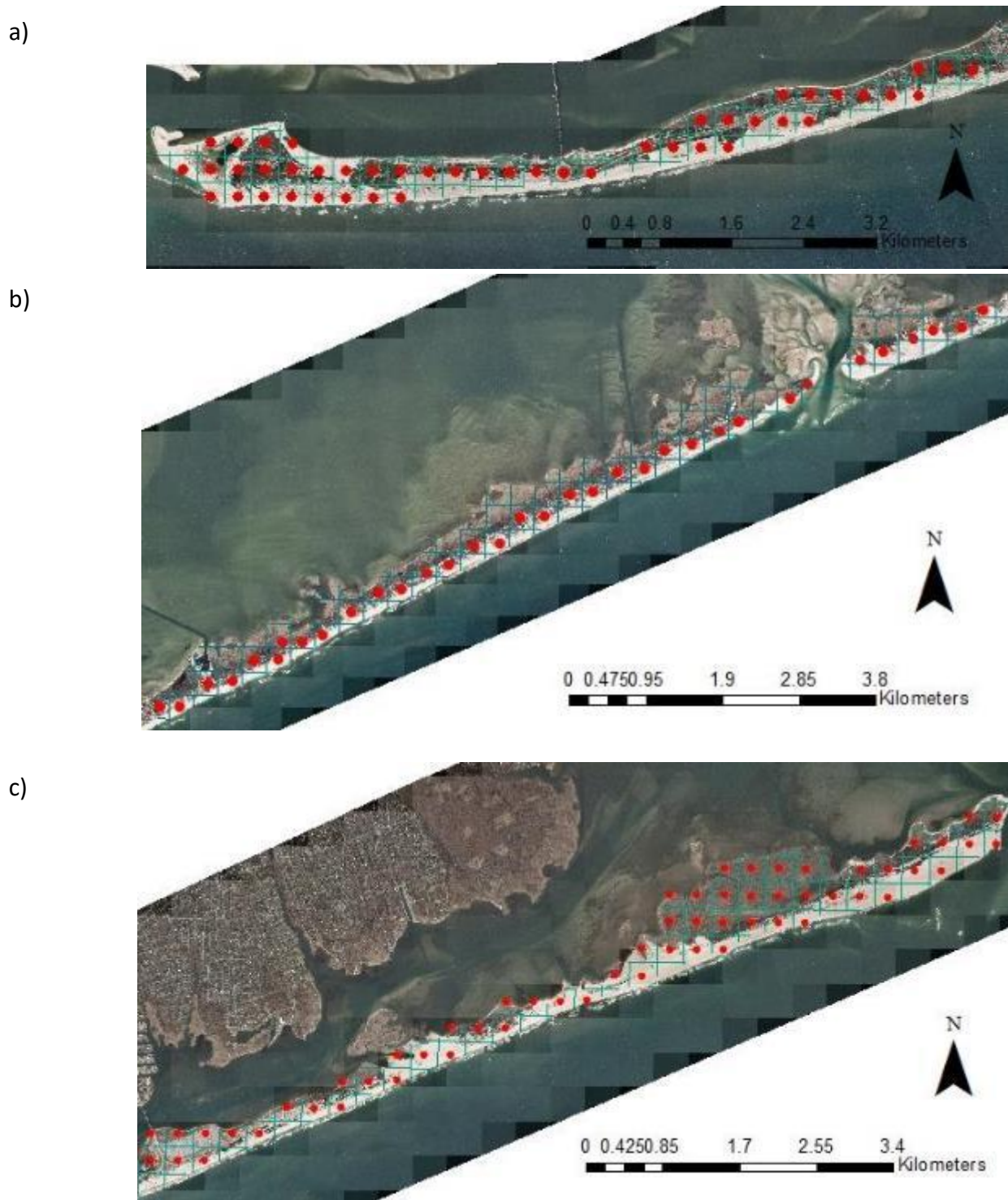


Figure 29. a) Transect locations in Robert Moses State Park and Fire Island National Seashore Lighthouse Tract. b) Transect locations in Fire Island National Seashore. c) Transect locations in Smith Point County Park. Monitors walked as close to straight-line transects as possible while following game 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 April–July and October–January of each year.

Lab methods

All scat samples were stored in a freezer, then transported on ice back to Virginia Tech and stored in a -20 C walk-in freezer until analysis. For each sample, we separated a small portion (approximately 2ml) of fecal material from the outside edges and tapered ends, and submerged this portion in 95% ethanol for DNA

preservation. These preserved subsamples have been sent (samples collected prior to October 2017) or will be sent (samples collected after October 2017) 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. Feral cats, domestic dogs, and raccoons also are present on Fire Island, and even experienced field biologists can have difficulty distinguishing among scats from these species and thus the need for genetic verification.

For a subset of these samples shown to contain high quality red fox DNA, we have collaborated with the MECU to conduct further genetic analyses to determine the phylogenetic origins of red foxes on Fire Island. This was done using red fox-specific primers and assays, and genotyping protocols previously developed by the MECU. We are comparing the resulting mitochondrial DNA 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 farther north, or the descendants of nonnative foxes imported from Europe for hunting.



Figure 30. After drying, scat samples are dissected and contents sorted for identification to the lowest taxonomic level possible. Photo: Earle Berge.

We used the remaining portion of each scat sample for fecal dietary analysis. Following standard protocols, we dried the samples and separated out contents by hand (Figure 30), then identified contents to the lowest taxonomic level possible through comparison to reference materials and collections (Reynolds and Aebischer 1991). We then calculated percent occurrence and frequency of occurrence for each diet item or group, to assess reliance on different food sources.

Table 25. 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 collared 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 were active during this second 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 a 15 m radius around the den, and recorded other signs of active use at the den entrance, 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.

We combined this den monitoring data with monitoring data from GPS collared and 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 excluded any foxes known to have died before the end of the monitoring period from these minimum abundance estimates.

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 31), and ear-tags during August–December 2015, April–May 2016, and February–April 2017 and 2018 . During these trapping periods, we also attempted to recapture foxes from previous years whose GPS collars failed to drop off as programmed.



Figure 31. Red fox at Robert Moses State Park outfitted with a Quantum 4000E GPS collar and numeric ear-tag

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 32). 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. We briefly resumed the use of box traps during our 2018 trapping season due to the low overall number of fox captures. This resulted in similar levels of raccoon and opossum bycatch, but no fox captures.



Figure 32. 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 33). 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 34). 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 33. Captured fox being processed. We manually restrained foxes and took morphometric measurements before releasing them. Photo: Caitlin Kupferman.



Figure 34. 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 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). In 2016–2018, 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). 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.

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

Home range estimation

We mapped GPS locations for each collared red fox in ArcMap for exploratory analyses, including the determination of fox proximity to nesting areas, and 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 and 2018) to create 100% 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 foxes crossing Old Inlet.

Habitat selection ratios

We used the 2-hour GPS location data for each collared fox in 2017, and habitat classification layers created from 6-inch resolution imagery to calculate individual and overall habitat selection ratios for foxes monitored in Spring–Summer 2017. Habitat classes included including dry sand, wet sand, beach grass, scrub, shrub, woodland, phragmites, marsh, water, and developed habitat.

We mapped each fox's 2-hour locations as used locations. We then randomly simulated 5 times the number of used locations per fox, within that fox's 100% MCP, to represent available habitat at the individual level. We overlaid these used and random locations onto 3 m resolution distance rasters created using the “Euclidean Distance” tool in ArcMap, and extracted the estimated distances from each used and random location to each of 10 habitat types.

We then used R package “adehabitatHS” to calculate individual and overall selection ratios comparing the mean distances to each habitat type between used and random points. Distance-based habitat selection analyses have been shown to be more robust to potential location error associated with the GPS collar data than classification-based habitat analyses (Conner et al. 2003). These selection ratios provide a preliminary comparison of habitat use relative to availability, but do not account for additional covariates that may

influence habitat selection, such as breeding status, sex, and residency. We will conduct resource selection function analyses in future reports to assess such covariates.

We are in the process of creating additional habitat classification layers using imagery collected during each monitoring period (2015-present), and will expand our analyses to fox location data collected during other monitoring periods as these classifications are completed.

Remote camera surveys

Field methods

Unbaited camera grid

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 unbaited 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 trail 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 travelway (Figure 35). We set each camera on opposite sides of the animal trail to photograph both sides of passing animals (Figure 36), and angled them to be offset from each other.



Figure 35. Cameras were mounted on plastic t-posts along animal trails and wildlife travelways.



Figure 36. Cameras were placed on opposing sides of animal trails and wildlife travelways to photograph both sides of passing animals.

We programmed the cameras to take 3 pictures per trigger with a 5 second 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. (Figure 37). 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.

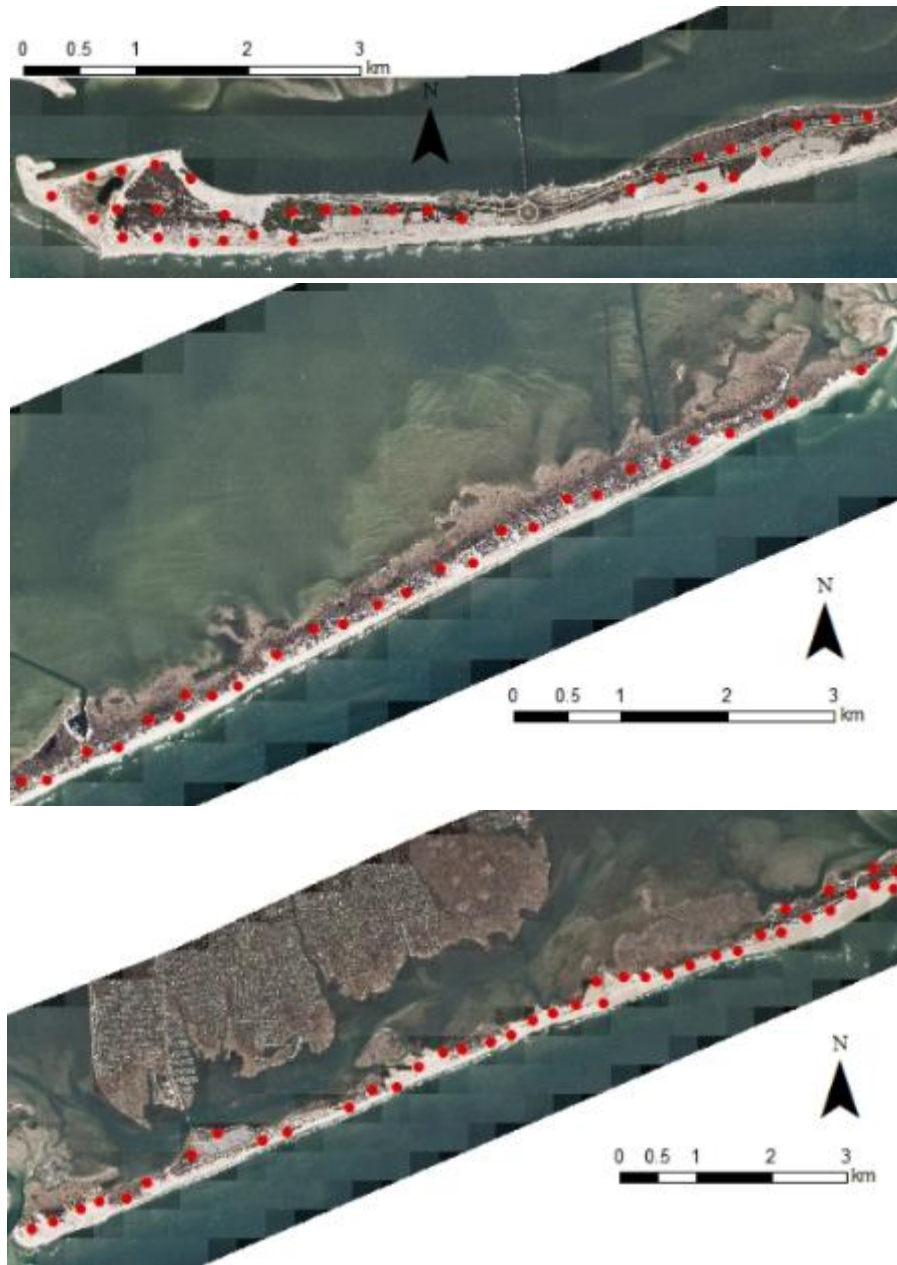


Figure 37. Approximate camera station locations at Robert Moses State Park (top), Fire Island National Seashore between Watch Hill and Old Inlet (middle), and Fire Island National Seashore east of Old Inlet and Smith Point County Park (bottom). Exact location and number of stations varied slightly among years.

Baited camera stations

In examining photos taken during our remote camera surveys in fall 2015, we had 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 were quite small and not meant to be detectable or readable in photos.

Analyses

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 is a common metric reported for remote camera studies that can be used as an index of species activity in a study area. Trap success differs from the raw number of species detections in that it is scaled by unit effort, to allow for comparisons among survey sites and periods and across studies while accounting for potential differences in numbers of detections due to differences in survey duration and/or numbers of camera stations. 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.

For a subset of the camera data collected at Smith Point County Park and Old Inlet East from Fall 2015–Winter 2017, we conducted a separate set of analyses comparing occupancy of red foxes, raccoons, and feral cats to examine potential effects of the mange die-off in those study areas on the remaining mesopredator community during that time period. For these analyses, we collapsed detection data into 4 day detection periods to allow for the inclusion of covariates in a second series of single season, single species models in Program PRESENCE. Habitat covariates included the distance to nearest road (paved or ORV), distance to ocean high tide line (as mapped during annual high tides, described in piping plover monitoring methods), management unit (county park or National Seashore), and general habitat type (ocean side non-woody

vegetation, woody vegetation, or bay side non-woody vegetation, identified by visual inspection of annual orthoimagery). We also included red fox trap success as covariates in raccoon and feral cat candidate models, to examine potential interspecific competition. We will extend this approach to include habitat covariates in occupancy modeling at all sites in future analyses and final reports.

Results

Objective 1: Quantify the abundance and occupancy of red foxes and relate changes in these parameters to changes in habitat and predator management activities

Abundance estimates from den surveys and transect monitoring

We walked 96 transects (24 in Robert Moses State Park, 40 in Fire Island National Seashore; and 32 in Smith Point County Park, 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 October/November, December, and January (2015–2018).



Figure 38. Two kits at the entrance of one of the dens monitored at Fire Island National Seashore in 2016. Photo: Matt Kynoch.

Based on our transect surveys, GPS collaring efforts, and den monitoring (Figure 38), we estimate that there were at least 3 adults inhabiting Robert Moses State Park and the Fire Island National Seashore Lighthouse Tract, at least 1 adult inhabiting the area between Watch Hill and Old Inlet, and 1 adult inhabiting Old Inlet East and Smith Point County Park at the beginning of August 2018 (Table 26).

Based on these minimum abundance estimates, we estimated minimum red fox densities in each of the 3 main study areas as of August, or the end of the piping plover nesting season. 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 0.42 adults/km² and a total red fox population size of 7 adults west of Old Inlet at the beginning of August 2018. These minimum abundance estimates are adjusted to exclude collared foxes known to have died during the GPS collar tracking period (prior to August 2018) or any other deceased foxes discovered during monitoring activities (see Objective 4 results for further discussion of observed fox mortalities from mange and non-mange causes). Also excluded from the 2018 minimum abundance estimate is one fox that was found injured in the trap during trap checks, and removed from the study area and taken to a wildlife rehabilitator per our permit requirements.

Three of the 4 foxes captured during our 2018 trapping period showed signs of moderate to severe mange infestation. We continued seeing signs of mange infestation during our field observations and in remote camera photos collected during den monitoring through mid-September 2018, when the last of our den cameras were removed.

More recently, we have observed 2 foxes affected by mange (1 in the Fire Island villages on October 31st, 2018; one in the National Seashore Wilderness Area on November 8th, 2018). We have received reliable reports and photos of one apparently healthy fox seen in the Wilderness Area on October 17th, 2018 and at least 2 healthy foxes seen in one of the villages on November 12th, 2018, along with other reliable reports (without photos) of healthy red foxes elsewhere on the island. These sightings suggest that while the outbreak is still ongoing, there are likely a few healthy foxes left on Fire Island.

We are working on results that show the change in red fox numbers from January through December 2018 to summarize the dramatic decline in numbers that we attribute primarily to sarcoptic mange; this will be added to this report in January before distribution to partners.

Table 26. Minimum estimated red fox adult and sub-adult abundances and total densities by study area and as of August of each year, or the end of the piping plover breeding season.

Study Area	2016			2017			2018		
	Adults	Sub-adults	All foxes/km ²	Adults	Sub-adults	All foxes/km ²	Adults	Sub-adults	All foxes/km ²
Robert Moses State Park+ Lighthouse Tract	12	16	5.90	10	8	3.79	3	0	0.63
Watch Hill to Old Inlet	8	14	4.64	13	2	3.16	1	0	0.21
All area W of Old Inlet ^a	39	57	5.88	39	16	3.48	7	0	0.42
Smith Point County Park + Old Inlet East	0	0	0	1	0	0.21	1	0	0.21

^aEstimates 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.

Trap success

In addition to red fox, we detected raccoons (*Procyon lotor*), feral cats (*Felis catus*), domestic dogs (*Canis familiaris*), gulls (various species, Family *Laridae*), crows (*Corvus brachyrhynchos*), and several species of bird

of prey during our remote camera surveys (Table 27). Species of aerial predators detected varied among sites and years, and included American kestrel (*Falco sparverius*), Northern harrier (*Circus hudsonius*), red-tailed hawk (*Buteo jamaicensis*), Cooper's hawk (*Accipiter cooperii*), and short-eared owl (*Asio flammeus*). Red fox was the most commonly detected predator species at sites west of Old Inlet in 2015 and 2016, but not in 2017. 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 we did observe an increase in the trap success of aerial predators (although our surveys were not designed for aerial species). However, as the mange outbreak was underway in Winter 2017, raccoon trap success increased and was higher than fox trap success at Robert Moses State Park, while feral cat trap success increased at Smith Point County Park relative to previous years.

Table 27. Trap success (# detection events/100 trap nights) for potential shorebird predator species during fall–winter remote camera surveys. Trap success is an index of relative abundance and does not account for detection probability.

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	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	119.95	26.14	8.26	0.84	0.28	0.28	0.22
FINS wilderness	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
Fall–Winter 2017							
RMSP	18.36	20.37	5.66	0.63	0	0.21	0.84
FINS wilderness	72.62	34.19	0.26	0	0	3.84	0
SPCP+OIE	3.54	15.19	4.97	0.14	0.20	1.09	0.61

Occupancy estimates from remote camera monitoring

We report here our remote camera survey findings from Fall–Winter 2015–2017; our 2018 camera surveys are ongoing and will be reported in the final report.

In each of our study sites, we found that red fox occupancy was best modelled as constant whereas red fox detection varied across sites and years in whether it was best modelled as constant or varying by day of the survey period (Table 28). For the first time during the study, red fox were not photographed at every camera station in study areas west of Old Inlet in Fall–Winter 2017, although estimated occupancy remained high at 0.97(SE 0.03) at Robert Moses State Park. Detection probability in this area varied by day, and was generally lower in Fall–Winter 2017 compared to previous years. At the National Seashore between Watch Hill and Old Inlet, occupancy remained at 1, although detection probability was somewhat lower than in the previous year. At Smith Point County Park and the Old Inlet East portion of Fire Island National Seashore, fox occupancy increased from 0 in Fall–Winter 2016 to 0.84 (SE 0.12) in Fall–Winter 2017, following fox recolonization of the

area post-mange. This occupancy estimate is comparable to the Fall–Winter 2015 occupancy estimate of 0.86 (SE 0.06) in the area, during the peak of the mange outbreak east of Old Inlet.

Table 28. 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	-	-	-	-	-
Fall–Winter 2017						
Robert Moses State Park	ψ(.) p(day)	1552.64	0	0.99	1	67
	ψ(.) p(.)	1569.44	16.80	<0.01	<0.01	2
Fire Island National Seashore	ψ(.) p(.)	2062.46	0	0.99	1	2
	ψ(.) p(day)	2072.23	9.77	<0.01	0	59
Smith Point County Park+OIE	ψ(.) p(day)	413.14	0.00	1	1	2
	ψ(.) p(.) ^b	-	-	-	-	-

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

^b Model did not converge

Table 29. 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			Fall–Winter 2017		
	Top Model	ψ (SE)	p^a	Top Model	ψ (SE)	p^a	Top Model	ψ (SE)	p^a
Robert Moses State Park	$\psi(.)$ $p(\text{day})$	1 (0)	0.03–0.72	$\psi(.)$ $p(.)$	1(0)	0.67 (0.01)	$\psi(.)$ $p(\text{day})$	0.97 (0.03)	0.03 – 0.38
Fire Island National Seashore	$\psi(.)$ $p(\text{day})$	1 (0)	0.11–0.61	$\psi(.)$ $p(.)$	1(0)	0.54 (0.01)	$\psi(.)$ $p(.)$	1(0)	0.42 (0.01)
Smith Point County Park+OIE	$\psi(.)$ $p(\text{day})$	0.86 (0.06)	0.03–0.25	n/a	0	n/a	$\psi(.)$ $p(\text{day})$	0.84 (0.12)	0.04 – 0.14

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

Pre- vs. post-mange comparison East of Old Inlet

For foxes, our top occupancy model in both years that foxes were present (2015 and 2017) held occupancy constant, while detection probability varied with distance to road. As the distance to road increased, detection probability decreased (Table 30). The inclusion of this distance to road covariate resulted in a very minor increase (~ 0.01 in both years) in occupancy estimates compared to the models without this distance to road covariate described above (Figure 39).

For raccoons, occupancy was positively correlated with red fox trap success in 2015, while detection probability varied by general habitat type. Detection probability was higher in woody vegetation compared to both non-woody vegetation types. Raccoon occupancy was high (0.80–0.97) in all three years, but did not significantly differ in years where foxes were present vs. absent from the area, or in Fall 2016 following lethal removal activities at Smith Point County Park in January 2016.

For feral cats, occupancy varied with distance to road in 2016 and fox trap success in 2017; distance to road did not show a clear directional trend in 2016, while feral cat occupancy in 2017 was negatively correlated with fox trap success. Feral cat detection probability varied by habitat in 2015 and 2017, with higher detection probability in woody vegetation compared to both non-woody vegetation types, but was best modeled as a constant in 2016. Feral cat occupancy varied (0.29–0.64) in all three years, but also showed no significant difference in years where foxes were present vs. absent.

Table 30. Top models for occupancy (ψ) and detection probabilities (p) for mesopredators at Smith Point County Park and Old Inlet East during the mange outbreak (Fall–Winter 2015), absence of foxes (Fall–Winter 2016), and following fox recolonization (Fall–Winter 2017).

Species	Fall–Winter 2015	Fall–Winter 2016	Fall–Winter 2017
Red fox	$\psi(.)$ $p(\text{distance to road})$	–	$\psi(.)$ $p(\text{distance to road})$
Raccoon	$\psi(\text{fox trap success})$ $p(\text{habitat})$	$\psi(.)$ $p(\text{habitat})$	$\psi(.)$ $p(\text{habitat})$
Feral cat	$\psi(.)$ $p(\text{habitat})$	$\psi(\text{distance to road})$ $p(.)$	$\psi(\text{fox trap success})$ $p(\text{habitat})$

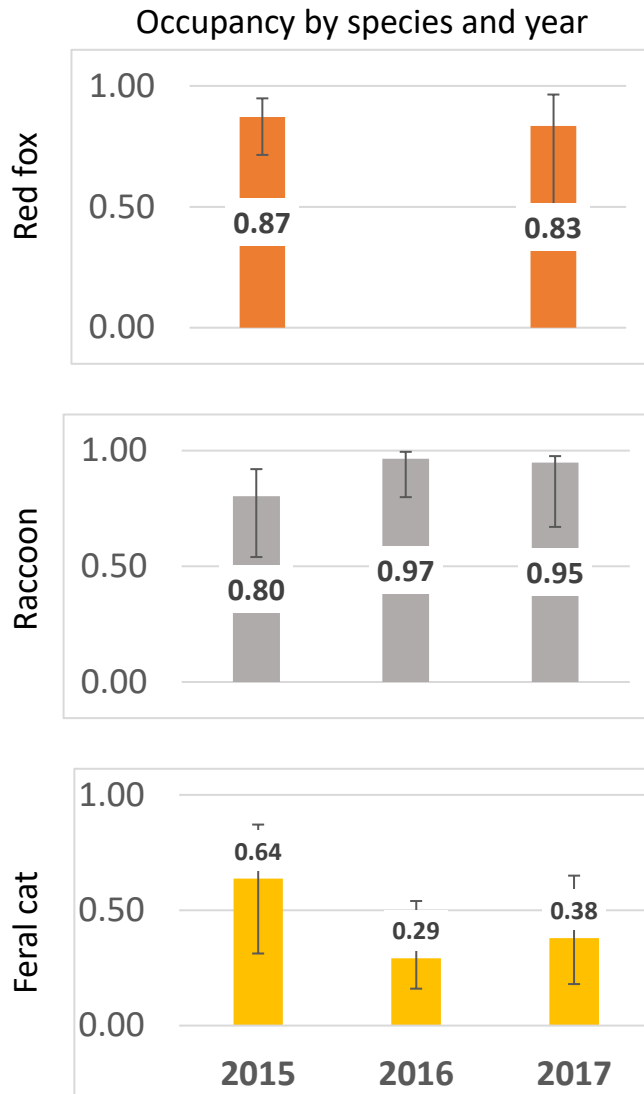


Figure 5. Occupancy (ψ) estimates for mesopredators at Smith Point County Park and Old Inlet East during the mange outbreak (Fall–Winter 2015), absence of foxes (Fall–Winter 2016), and following fox recolonization (Fall–Winter 2017).

Baited camera stations

Preliminary examination of photos taken at temporary baited camera stations in 2015–2016 suggested 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 a substantial increase in individual identifications; <1% of all fox detection events were identifiable as a specific individual. Thus, we did not repeat these baited camera stations in 2017 or 2018.

Objective 2: Habitat selection

Our preliminary analyses of red fox selection ratios indicate that foxes use habitat non-randomly (Figure 40). Red foxes used locations closer to dry sand than expected based on availability within their respective home ranges. Distances to other habitat types were not significantly different between used and available locations. Habitat selection varied among individual foxes but the selection for dry sand is clear in the individual selection ratios as well (Figure 41).

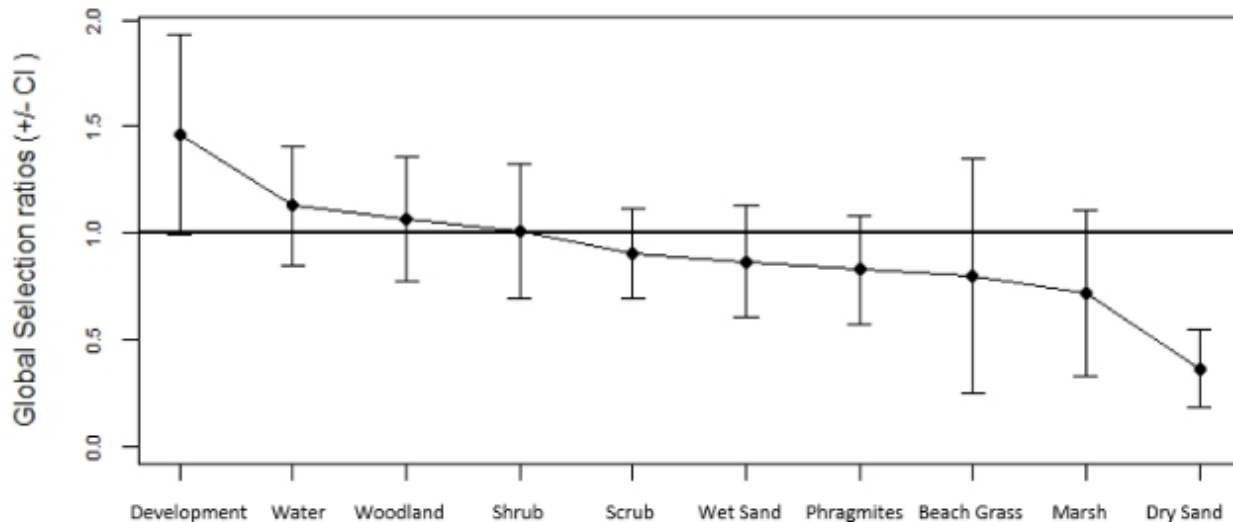


Figure 40. Overall habitat selection ratios for red foxes monitored during Spring-Summer 2017. A selection ratio <1 means that used locations were closer to a given habitat type than expected, a ratio >1 means that used locations were further from a given habitat type than expected, and a ratio of 1 represents use proportionate to availability. Lines overlapping zero show a non-significant selection ratio.

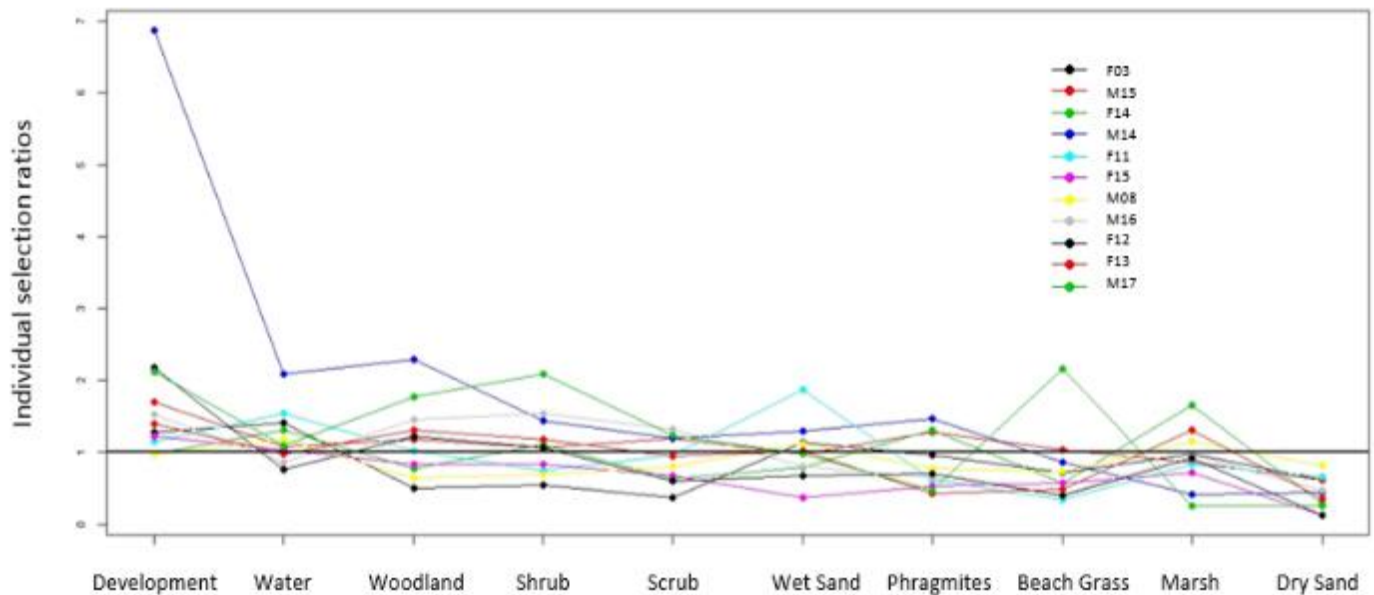


Figure 41. Individual habitat selection ratios for red foxes monitored during Spring-Summer 2017. Each line represents a single fox. A selection ratio <1 means that used locations were closer to a given habitat type than expected, a ratio >1 means that used locations were further from a given habitat type than expected, and a ratio of 1 represents use proportionate to availability. Lines overlapping zero show a non-significant selection ratio.

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 captured 4 foxes total from mid-February–late April 2018, trapping every night that weather, access/logistical restraints, and other field activities allowed. We captured 1 fox over 9 nights at Robert Moses State Park and 2 foxes over 1 night at Fire Island National Seashore’s Lighthouse Tract (Table 31). Of these 4 foxes, we outfitted 3 with GPS collars. The remaining fox was found injured in the trap during trap checks, and removed from the study area and taken to a wildlife rehabilitator per our permit requirements. We did not successfully capture any foxes over 5 nights at Fire Island National Seashore’s Wilderness area (between Watch Hill and Old Inlet) or over 5 nights at Smith Point County Park.

We collected data on 1 collared female and 1 collared male at Fire Island National Seashore’s Lighthouse Tract from April 20, 2018 until their deaths on approximately May 9th and May 20th 2018, respectively. An additional female was collared at Robert Moses State Park on April 18th, 2018, but the collar malfunctioned immediately after deployment and did not collect any usable data. Thus, this individual was excluded from our spatial analyses. We were able to recover this collar by activating the remote drop-off feature in the field after discovering the collar malfunction. Both other collared foxes died before the end of the monitoring period. We estimated the date of death using location data, and excluded post-mortem points from our analyses (see Objective 4 for more information on observed mortalities).

Table 31. 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.

Study Area	2015		2016		2017		2018	
	Males	Females	Males	Females	Males	Females	Males	Females
Smith Point County Park + OIE	2	1	0	0	0	0	0	0
Fire Island National Seashore – Wilderness Area	3	0	3	3	1	3	0	0
Robert Moses State Park + Lighthouse Tract	2	3	4	2	4	4	1	2
Total	7	4	7	5	5	7	1	2

Home range estimates: MCPs and Kernel densities

We found that Spring-Summer 2018 home range estimates varied slightly among the 2 foxes monitored (Table 32, Figure 42) with the male having a larger home range by all measures calculated compared to the female (60.4 ha, 176.4 ha, and 150.8 ha compared 34.5 ha, 137.6 ha, and 60.4 ha for 50% kernel, 95% kernel, and 100% minimum convex polygons, respectively). The kernel home range estimates for the female fox were noticeably larger than the average kernel home range estimates for female foxes monitored in the same study area during previous years, while home range estimates for the male foxes were noticeably smaller than those of male foxes monitored in the same area during previous years. The home ranges of both collared red foxes in 2018 overlapped significantly, suggesting that they may have been part of the same family group.

When the 50% and 95% core and home range areas of both GPS-collared foxes are overlaid, the resulting contours encompass nearly all dry land from Field 5 of Robert Moses State Park through the Fire Island National Seashore Lighthouse Tract, and eastward into the villages (Table 33, Figure 43). Both foxes died prior to the initiation of any piping plover nests in this area.

Home range maps for all foxes tracked, including those tracked during previous monitoring periods, are shown in Appendix B.

Table 32. 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	26.0	44.0
RMSP	F-11	11.8	48.5	54.3
RMSP	F-12	23.2	73.7	205.4
RMSP	M-08	19.0	73.4	68.5
RMSP	M-14	148.2	450.4	819.6
RMSP	M-15	561.6	820.5	797.0
FINS-L	F-13	19.6	107.1	162.4
FINS-L	M-16	17.9	72.0	84.3
FINS	F-14	6.1	24.8	61.4
FINS	F-15	2.6	12.9	42.8
FINS	M-17	87.4	266.3	616.8
Spring-Summer 2018				
FINS-L	F-16	34.5	137.6	60.4
FINS-L	M-18	60.4	176.4	150.8

Table 33. 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
Spring–Summer 2018				
RMSP + Lighthouse Tract	M ^a	60.4	176.4	150.8
	F ^a	34.5	137.6	60.4

^an=1



Figure 42. MCP, 50% and 95% kernel density contours for 1 male red fox (top) and 1 female red fox (bottom) at Robert Moses State Park and Fire Island National Seashore Lighthouse Tract, Spring-Summer 2018.

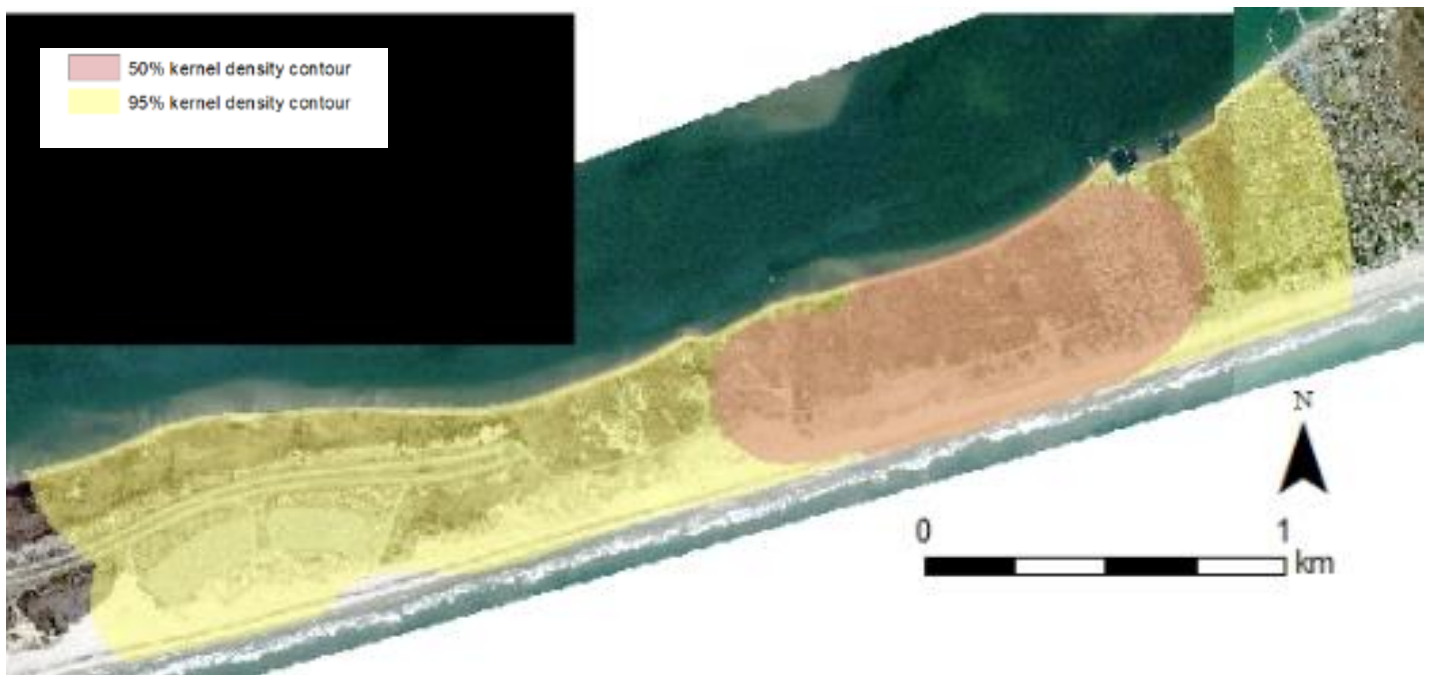


Figure 43. 50% and 95% kernel density contours for both GPS-collared red foxes in Spring-Summer 2018 at Robert Moses State Park and Fire Island National Seashore Lighthouse Tract.

Long distance movements

During previous monitoring periods, we observed multiple GPS-collared male foxes making long distance movements, as described in previous reports. We did not observe similar long distance movements during the 2018 monitoring period, possibly due to the low number and poor general condition of foxes monitored.

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 did not locate any active breeding dens or find any signs of reproduction in any of our study areas during Spring–Summer 2018, despite intensive searching. We did locate and monitor 20 non-breeding dens during this time, most of which were only temporarily active. We used photos from remote cameras placed at the 6 most active of these non-breeding dens and GPS collar data to determine that there were at least 2 likely non-breeding family groups in Spring–Summer 2018: 1 pair of foxes was photographed repeatedly visiting a den at Robert Moses State Park, and both GPS collared foxes at Fire Island National Seashore’s Lighthouse Tract appeared to be using a shared territory prior to their deaths. (Table 34).

Table 34. Breeding dens located on transect surveys, through GPS tracking, and opportunistically in April–July of 2016–2018, and the associated number of likely family groups.

Study Area	2016		2017		2018	
	Breeding dens	Family groups	Breeding dens	Family groups	Breeding dens	Family groups
Smith Point County Park + Old Inlet East	0	0	0	0	0	0
Fire Island National Seashore – Wilderness Area	7	4	2	2	0	0
Robert Moses State Park + Lighthouse Tract	10	4	7	4	0	2
Total	17	8	9	6	0	2

Adult Survival

Of the 3 foxes captured, collared and/or ear-tagged during the 2018 trapping period, only 1 survived the full duration of the spring-summer monitoring season. Of the 2 deceased foxes, 1 died of suspected sarcoptic mange. This fox, M-18, showed signs of moderate mange at the time of capture, which appeared to worsen over the following months. The last reliable GPS locations from this fox were recorded on May 20th, 2018 in a residential area of Kismet, NY. We later discovered M-18’s decomposed remains buried several feet underground in a residential area of Fair Harbor, NY nearly 2km away, while recovering the GPS collar. It is unclear whether the fox traveled there on its own or was transported there prior to burial. The other deceased fox, F-17, died of unknown causes, showing no signs of sarcoptic mange or poor health during the weeks prior to death in early May 2018. GPS locations and the low level of decomposition of the remains suggest that this fox died on approximately May 9th, 2018.

In addition, we found the remains of a previously collared and ear-tagged fox, F-09, during transect surveys in early January 2018. This was an adult female fox that we tracked during the 2015 and 2017 GPS tracking periods. F-09 was photographed showing symptoms of severe mange in October 2016. Based on the heavily decomposed condition of the remains, this fox likely died of mange-related causes soon after. We also found 1 unmarked adult fox dead in Robert Moses State Park in June 2018. This fox appears to have died sometime during the winter, based on the remaining thick winter coat. The remains did not show any signs of mange, but were fairly decomposed at the time of discovery.

Based on known survival/mortality of GPS collared individuals, we estimate adult survival during our Spring – Summer 2018 monitoring period as 100% at Robert Moses State Park (n =1) and 0% at Fire Island National Seashore’s Lighthouse Tract (n=2) (Table 35). We did not have marked foxes anywhere else on the island, and thus cannot estimate survival in other areas during the 2018 monitoring period.

In past years, 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 (see 2017 annual report for more details). During the 2018 monitoring period, we did not recapture, resight, or otherwise detect any marked or known individuals alive from previous monitoring periods.

Table 35. Proportion of GPS collared foxes known to have survived to the end of each monitoring period. Dashes indicate areas where no foxes were monitored during that time period. Asterisks indicate populations known to be affected by sarcoptic mange during the monitoring period.

Study Area	2015	2016	2017	2018
Smith Point County Park + Old Inlet East	0*	-	-	-
Fire Island National Seashore – Wilderness Area	0.66	1	0.75	-
Robert Moses State Park + Lighthouse Tract	1	1	0.38*	0.33*
Overall	0.64	1	0.50	0.33

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

Diet analyses

We collected 29 scat samples opportunistically and during transect surveys and den monitoring between September 2017 and March 2018, and 18 samples from April–August 2018 (Table 36). 180 of the 216 scat samples sent for DNA analyses so far have successfully yielded a source species identification; all but one of these samples came from a red fox.

To date, we have completed preliminary fecal dietary analysis on 190 samples. From these samples, we have identified 863 items to varying taxonomic levels. Preliminary estimates, pooled across multiple survey seasons and study sites, of percent occurrence (the percentage of total items identified belonging to a given category) and frequency of occurrence (the number of samples containing at least one item from a given category) suggest that red foxes on the island have highly variable diets (Table 37). Of items identified, 7% were from birds (feathers, bird bones, beaks, etc.), and 23% of samples collected included at least one bird part. Most bird remains have not yet been readily identifiable to species due to a lack of unique identifying characteristics (bone and beak fragments) or degradation during the digestion process (feathers).

Only one scat sample examined to date has contained definitive piping plover remains. On June 16th, 2018, Virginia Tech’s piping plover monitoring crew at Smith Point County Park discovered a scat sample containing an orange piping plover leg band and green flag (C2K). There were no identifiable tracks or signs in the immediate area indicative of the predator species. Crew members recorded the location, collected the sample, and added it to our collection of scat samples for dietary analysis. We also took a genetic subsample, which will be sent to our collaborators at the MECU with our final round of samples for genetic analyses to determine the predator species.



Figure 44. Some anthropogenic food items found at active dens during den monitoring activities, May–July 2017.

We recorded a wide variety of prey remains outside of den entrances and in the immediate vicinity during our monitoring activities from April–August 2016 and 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 44).

Table 36. Scat samples collected during transect surveys, den monitoring, and opportunistically during the course of the study.

Study Area	June– July 2015	April– August 2016	September 2016–March 2017	April– August 2017	September 2017–March 2018	April– August 2018
Smith Point County Park	17	0	1	3	4	7
Fire Island National Seashore – east of breach	4	0	0	0	0	0
Fire Island National Seashore – west of breach	16	54	27	39	19	2
Fire Island National Seashore – Lighthouse Tract	5	5	2	5	0	3
Robert Moses State Park	34	42	17	40	6	6
Total	76	101	47	87	29	18

Table 37. Percent occurrence and frequency of occurrence for source categories of 863 items identified during preliminary fecal dietary analysis of 190 scat samples collected from 2015-2018.

Source	% Occurrence	Frequency of Occurrence
Anthropogenic	4.06	0.14
Avian	6.84	0.23
Crustacean	8.34	0.33
Fish	2.32	0.11
Insect	20.39	0.63
Mammalian	24.91	0.73
Unknown bone/skin	1.04	0.05
Vegetation	20.23	0.74
Unidentifiable/Misc. non-food items	11.81	0.44

Spatial overlap with piping plover nests and locations

Both GPS collared foxes in 2018 died prior to nest initiation in the area, thus we did not collect any high frequency fox location data in 2018 or compare fox home ranges to nest locations. All maps for high frequency location sampling from previous monitoring periods are included in Appendix C.

Hourly/Every Two Hour Location Data

We found that nearly all dry land area in from Field 5 in Robert Moses State Park eastward through the Fire Island National Seashore Lighthouse Tract and the westernmost villages was within the home range of at least one collared red fox from April 20th to May 20th, 2018 (see Objective 3). There were additional foxes on the island beyond those we had collared; these results only describe home ranges and overlap among collared foxes and piping plovers.

All but one of the 45 non-breeding piping plover locations recorded before the collared foxes died were in the 95% kernel density home range of at least one collared fox, and roughly one third of these locations were also located within the 50% kernel density core area of at least one fox (Figure 45). During Spring-Summer 2018, no red foxes were collared at Smith Point County Park nor elsewhere in Robert Moses State Park or Fire Island National Seashore, thus no nests, brood locations, or adult locations in those areas were overlapping with collared red fox territories.



Figure 45. Piping plover adult locations (prior to fox mortality) overlaid onto overall red fox kernel density contours at Robert Moses State Park and Fire Island National Seashore Lighthouse Tract, spring 2018.

Discussion and Next Steps by Objective:

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

At the time of our last report, there was an ongoing sarcoptic mange outbreak in all study areas west of Old Inlet, including Robert Moses State Park and the Fire Island National Seashore's Lighthouse Tract and Wilderness Area (see 2017 Annual Report for more details). This outbreak appears to have continued throughout the winter, resulting in the substantial decreases in the fox population as documented through our monitoring. This current outbreak does not appear to have spread to the east side of Old Inlet, as we have not observed any signs of mange in foxes in Smith Point County Park or in that portion of the National Seashore Wilderness Area.

Our combined observations from transect and den surveys, GPS collaring, and remote camera surveys have documented a dramatic decline in red fox abundance and density from August 2016 to August 2018 in all management areas west of Old Inlet. For example, minimum estimated abundance at Robert Moses State Park and the Lighthouse Tract was 28 individuals (5.90 foxes/km²) in 2016, and 3 individuals (0.63 foxes/km²) in 2018. Similarly, minimum red fox abundance in the Fire Island National Seashore wilderness area (Watch Hill to Old Inlet) was 22 individuals (4.64 foxes/km²) in 2016, and 1 individual (0.21 foxes/km²) in 2018. These changes represent an 89% and 95% decrease, respectively. When these densities were averaged and extrapolated to all areas west of Old Inlet (including the villages and unsurveyed NPS areas), minimum red fox abundance to the West of Old Inlet was estimated at 96 individuals (5.88 foxes/km²) in 2016, and by August 2018 had dropped to an estimated 7 individuals (0.42 foxes/km²), a 93% decrease overall.

At Smith Point County Park and Old Inlet East, there was an earlier and even more rapid mange-driven decline and extirpation of foxes in 2015 and early 2016, such that foxes were not detected east of Old Inlet from mid-April 2016 until early March 2017. In 2017 and 2018, we conservatively estimate that 1 fox inhabited the area. However, there were some transect surveys in 2018 on which no fox sign was detected anywhere east of Old Inlet, leading to uncertainty over whether this is one fox that has persisted or whether there have been multiple colonization and extirpation events. Our field observations suggest that the east side of Old Inlet may not have been continuously occupied by foxes from February–August 2018. Given the relatively short distance and low traffic volume, it is possible that foxes and other terrestrial animals could move between Fire Island and mainland Long Island using the Smith Point Bridge. We have received anecdotal reports from contractors of foxes and other animals crossing the bridge, but have not observed any successful crossing events directly.

Despite these declines, occupancy rates, or the probability of presence accounting for imperfect detection, have remained very high in all study areas except for Smith Point County Park in 2016 when no foxes were present. These high occupancy rates are likely driven by the movement behavior of remaining foxes. Our GPS collar data from foxes at Smith Point and Old Inlet East suggest that, even when fox densities were dropping or already very low, the remaining foxes ranged much more widely than normal, and thus were photographed at nearly all camera stations, resulting in high occupancy estimates. This wide-ranging behavior among

remaining animals has been reported in other studies where a territorial species experienced a sharp population decline, including a long-term study of an urban fox population affected by mange in Bristol, UK (Baker et al. 2000, Potts et al. 2013). We hypothesize that this period of high occupancy despite low abundance may have already taken place west of Old Inlet, during Fall 2017–Winter 2018 camera surveys. Thus, we anticipate that our ongoing Fall 2018–Winter 2019 camera surveys will show a drop in occupancy west of Old Inlet.

We have finished the Fall 2018–Winter 2019 remote camera surveys at Smith Point County Park and Fire Island National Seashore, and are partway through the Fall 2018–Winter 2019 remote camera survey at Robert Moses State Park. We are now in the process of entering photo data from completed surveys in preparation for analyses. Moving forward, we plan to extend our analyses of these photo data to include estimating overall abundance and population densities, to allow for more detailed investigation of how piping plover population dynamics and reproductive success may vary with fox population sizes.

We also are conducting driving surveys in the villages and compiling reports from NPS staff and project collaborators on fox sightings on Fire Island, to determine if red foxes are persisting in these areas with human-subsidized food and shelter. Since beginning these surveys in late October 2018, we have observed one fox affected by mange in the villages on Fire Island, and received a reliable reports from bird-watchers of at least 2 healthy foxes in the villages, as described previously in our results. These data suggest that red foxes may persist in these human-modified environments and serve as a source population for recolonization of other areas of Fire Island. We propose that future monitoring focus on 1) the recolonization dynamics of red foxes post-mange and the impacts on piping plover populations, 2) the persistence of sarcoptic mange in the environment or other hosts (e.g., rabbits) to determine if mange is naturally cyclical in this system, and 3) the responses of other mammalian and avian predators to the reduced population of red fox, and their impacts on piping plovers. For example, we saw increased trap success of raccoons and feral cats in some areas following red fox declines. In 2018, we also saw novel predators impacting piping plover nests and broods in areas where red fox populations were reduced.

At this point, the outbreak of sarcoptic mange is serving as a natural experiment in red fox lethal control by greatly reducing red fox abundance and densities at Smith Point County Park in 2015 and now West of Old Inlet in 2017-present. Thus, our analyses may serve as a proxy for how red fox populations and behavior change in response to widespread lethal removal of foxes by disease in this case, or by humans in cases of predation management.

Objective 2: Assess habitat use versus availability by red foxes and any variation in habitat selection related to changing habitat composition and predator management.

We have begun to conduct analyses of habitat use versus availability, and will soon begin to analyze red fox use of different management areas (e.g., restoration areas, breach fills) before, during and after the mange outbreaks to allow for comparison with piping plover use of these areas.

Our preliminary analyses suggest that red foxes are using habitat non-randomly, and may be selecting for areas of dry sand habitat. Moving forward, we plan to expand our analyses to include additional location data,

continue exploring red fox habitat selection using more complex analytical methods, and further investigate this pattern of potential selection and its implications for predation management and piping plovers.

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.

Our tracking of GPS-collared foxes has revealed novel details about the home range, movements, and variability in movements of red foxes before, during and after mange-related die-offs across all study areas; these data will be useful as a proxy for understanding the potential effects of widespread lethal removal of foxes. In preceding years (2016–2017), we found that breeding red foxes exhibited territorial behavior during their breeding seasons and that of the piping plover, but also that some individuals, especially un-paired males, ranged widely across the monitoring time period. From 2016–2017, there were no study areas West of Old Inlet un-used by a red fox, and the same was true for Smith Point County Park and Old Inlet East following the recolonization by at least one fox in 2017. In 2018, the mange-related die-off dramatically altered red fox home range sizes and their spatial coverage of Fire Island. While signs of fox activity at Smith Point continued to be observed in all areas of the park, fox use of areas to the west of Old Inlet decreased drastically as 2018 progressed.

The home range sizes of our two GPS-collared foxes to the West of Old Inlet were dramatically smaller than the average home ranges of foxes collared from 2015-2018. For example, the MCP of the male collared fox in 2018 was 151 ha, and the female was 60 ha. In preceding years, male foxes had average MCP home ranges of 677 ha and females had average home ranges of 134 ha. The foxes that were collared in 2018 were already showing signs of mange, and we suspect their poor condition contributed to their reduced home range size. No long distance movements, from inlet to inlet, were observed this year whereas they were frequent occurrences in past year. We will be analyzing data this year to explore how the progression of mange affected the ranging behavior and habitat use of our collared foxes.

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

Prior to 2018, reproductive success was relatively high west of Old Inlet, with an average of 7 breeding groups across study areas and litter sizes ranging from 1–7 kits. No active breeding dens were located observed in any of our study areas in 2018, whereas 17 breeding dens were monitored in 2016 and 9 in 2017. We suspect that there is only one individual fox present to the east of Old Inlet, and thus no reproduction was observed. We suspect that the lack of breeding west of Old Inlet in 2018 may be due to the low number and poor condition of foxes on the west side Old Inlet. Previous work in Bristol, UK by Soulsbury et al (2007) found that foxes with advanced mange failed to breed, and that overall reproduction within the population declined during the enzootic mange phase of their long-term study.

Survival of GPS collared adults decreased over the monitoring periods. West of Old Inlet, survival ranged from 0.66 to 1 prior to 2017. Survival at Robert Moses State Park and the Lighthouse Tract dropped to 0.38 during the 2017 monitoring period, when we began observing signs of mange in this area, while survival remained at 0.75 between Watch Hill and Old Inlet during this same time, and no signs of mange had been observed in that study area. East of Old Inlet, our survival data is limited to the 2015 monitoring period, where 3 collared/ear-

tagged foxes died of mange and one died of injuries consistent with a vehicle collision, resulting in a survival of 0.

Our survival data for 2018 is limited in size and scope due to the small number of individuals collared and ear-tagged, but follows a similar downwards trend. Only one of the three foxes captured during the 2018 trapping period was alive at the end of the Spring–Summer 2018 monitoring period. This fox, F-18, was originally captured, ear-tagged, and outfitted with a GPS collar at Robert Moses State Park. Although the collar had to be removed soon after due to a hardware malfunction, the ear-tag allowed us to identify this fox in den monitoring photos taken during throughout the season. F-18 showed signs of severe mange at the time of capture. Den photos suggest that her condition may have temporarily improved during the early summer, but eventually worsened again. We have not seen F-18, via field observations or remote camera, since removing the den cameras in mid-September 2018.

In areas and monitoring periods unaffected by mange, known mortality was relatively low, and the main identifiable cause was suspected vehicle collisions, while mortality in areas and monitoring periods affected by mange has nearly always been attributable to complications of the disease.

Combined, our reproduction and survival/mortality data suggest that sarcoptic mange has been a driving force behind the recent red fox population decline on Fire Island, and will likely continue to be a primary factor influencing fox population demographics.

Moving forward, we plan to collaborate with Dr. Luis Escobar, a professor and wildlife disease expert in Virginia Tech's Department of Fish and Wildlife Conservation, to conduct necropsies on foxes whose cause of death we were unable to determine in the field. We also plan to explore statistical modeling methods, such as known-fate survival modeling, to account for the many foxes that were ear-tagged but not monitored via GPS collars, whose fates are currently unknown.

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

In all years of the monitoring, we observed high spatial overlap between red fox breeding territories and movement and the presence of piping plover nests and broods. To the west of Old Inlet, there were no areas from 2015-2017 that were used by plovers that were not also used by red foxes; this dynamic changed in 2018 along with the mange outbreak. In 2016, red foxes were absent from Smith Point County Park and Old Inlet East during the plover breeding season, however since recolonization in 2017, there have been signs of fox in all areas used by breeding plovers to the east of Old Inlet.

Our analyses of the diet of red foxes to date show that mammalian prey are the most frequently occurring prey item in fox diets, and also make up the greatest percent occurrence of prey items. Mammalian prey, vegetation and insects (primarily beetles) appear to make up the majority of fox diets, followed by crustaceans (mainly crabs), then avian prey. While avian prey do not appear to make up a large portion of overall red fox diets, there could be season or site-specific patterns, which we plan to investigate further in future analyses.

Of samples examined so far, only one has contained clearly identifiable piping plover remains, in the form of a pair of leg bands. We currently are exploring microscopic methods for identifying feathers and hairs to species level and these analyses will be presented in our final report and papers. We also will 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.

We anticipate that planned analyses will allow us to examine and quantify both direct and indirect interactions between red foxes and shorebird adults, chicks, and nests, in much more detail.

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Appendices

Appendix A. Injured Piping Plover Observations

Table 38. Piping plovers seen with a swollen leg or limping after banding at Fire Island and Westhampton Island in 2018.

Flag ID	Age of Banding	Date Banded	Site Banded	Subsite Banded	First Seen Limping/Swollen	Seen not Limping	Comments
GF(K56)	Adult	5/20/2018	FNS	NBW	6/8/2018	6/26/2018	Right leg (orange band leg) held up more often, leg seems dirty/blackened, doesn't seem to want to put weight on it. Seen in Cape Hatteris 9/6/18 limping, holding up left leg and not using it much. Last seen 9/21/18 by Chelsea Weithman limping, holding up left leg, foot somewhat swollen and dark. Flag not read but orange band read O(5?1) and assumed to be GF(K56).
GF(PPK)	Chick	5/29/2018	FNS	NBW	7/26/2018	9/6/2018	Fishing line or garbage wrapped around left foot. Still able to use foot but holding it up a lot. Looks to be causing discomfort but at present not swollen. Seen on 9/6/18 in Cape Hatteris with no comments about limping.
GF(E48)	Chick	6/2/2018	FNS	OIE	07/28/2018	N/A	Limping on flag leg, flag high on leg.
GF(E44)	Chick	06/04/2018	FNS	OIE	07/08/2018	N/A	Seen 7/8 holding flag leg up, with slight

							swelling above joint. On 7/12 the swelling increased. Already a fledgling as of 7/7, at which time leg seemed fine.
GF(92Y)	Chick	06/06/2018	FNS	OIE	7/14/2018	N/A	Flag high and leg slightly swollen, but putting full weight on it.
GF(913)	Chick	06/07/2018	FNS	NBE	07/20/2018	N/A	Putting some weight intermittently on flag leg, but not using it most of the time/limping on leg. Flag was high on leg and leg swollen.
GF(U6P)	Chick	06/08/2018	FNS	NBE	07/12/2018	N/A	Limping, using flag leg but not wanting to put weight on it. Leg slightly swollen. Flew.
GF(148)	Chick	06/12/2018	FNS	CSW	07/28/2018	08/18/2018	Limping, flag leg seems to be bothering it. No swelling. Flag not sitting high.
GF(T1H)	Chick	06/12/2018	FNS	NBE	07/18/2018	N/A	Open flag and flag high up on leg, not wanting to place weight on leg but still using it frequently. Leg was swollen.
GF(XH4)	Chick	06/12/2018	FNS	OIE	08/09/2018	N/A	Limping on flag leg, swollen around joint, flag stuck up and pointed in. Still using leg and putting pressure, only a slight limp.
GF(7EH)	Chick	07/05/2018	FNS	BRO	08/08/2018	N/A	Slight limp on flag leg. Flag sitting up slightly and swollen just above the joint. Flag pointing inward. No discoloration.

GF(E28) now GF(85X)	Chick	07/05/2018	FNS	BRO	08/06/2018	08/20/2018	Swollen flag leg, walks fine when needs to walk but holds leg up if it doesn't need to walk. Picking at flag. Recaptured 8/8/2018 with noose carpet: Removed flag and applied antibiotic to upper left leg which was discolored, swollen, and raw. Bird still able to grasp with left foot and using leg once released. No sign of lacerations or cause of swelling. Update 8/10/2018: Using left leg, not discolored. Update 8/14/2018: Putting more weight on left leg. Slight limp. Holding right leg up slightly when stopped. No swelling, flag not high.
GF(9Y8)	Chick	07/08/2018	FNS	DMP	08/15/2018	N/A	Limping on flag leg, flag high on leg and slightly swollen.

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.



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 and Fire Island National Seashore Lighthouse Tract, Spring-Summer 2018.

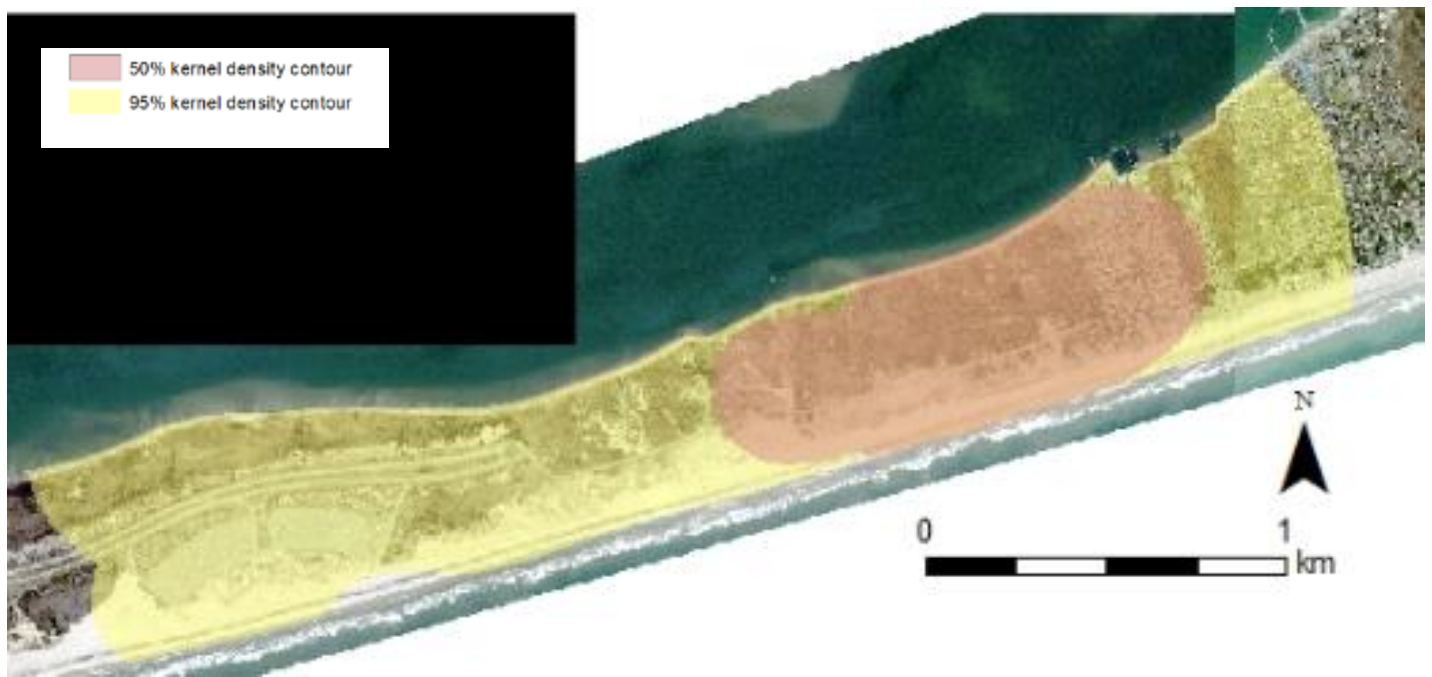


Figure 47. 50% and 95% kernel density contours for all GPS-collared red foxes in Spring-Summer 2018 at Robert Moses State Park and Fire Island National Seashore Lighthouse Tract.

Previous monitoring periods (in chronological order)

Fall 2015-Winter 2016

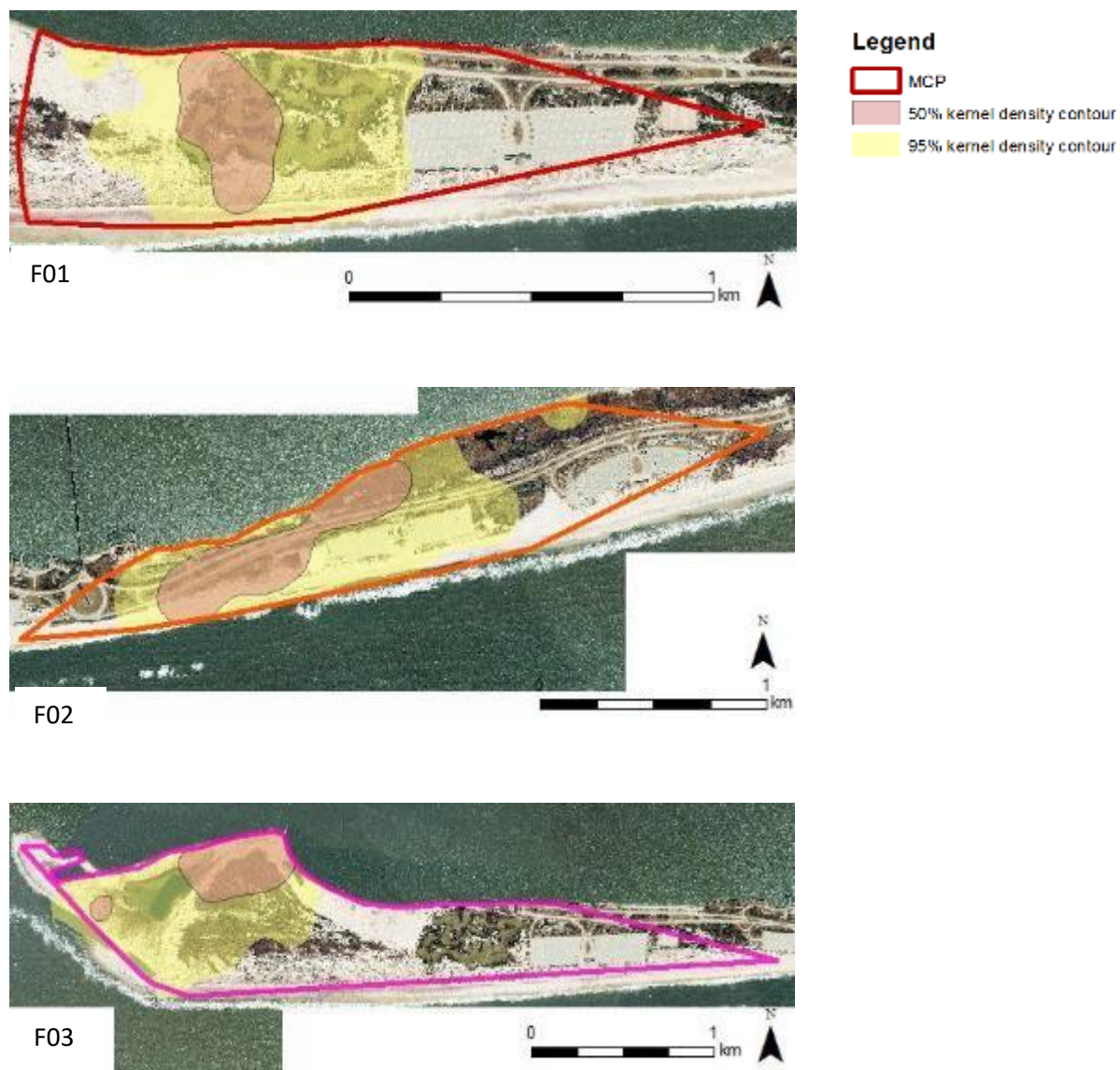


Figure 48. MCPs (solid lines), 50% and 95% kernel density contours for 3 female red foxes at Robert Moses State Park, Fall–Winter 2015.

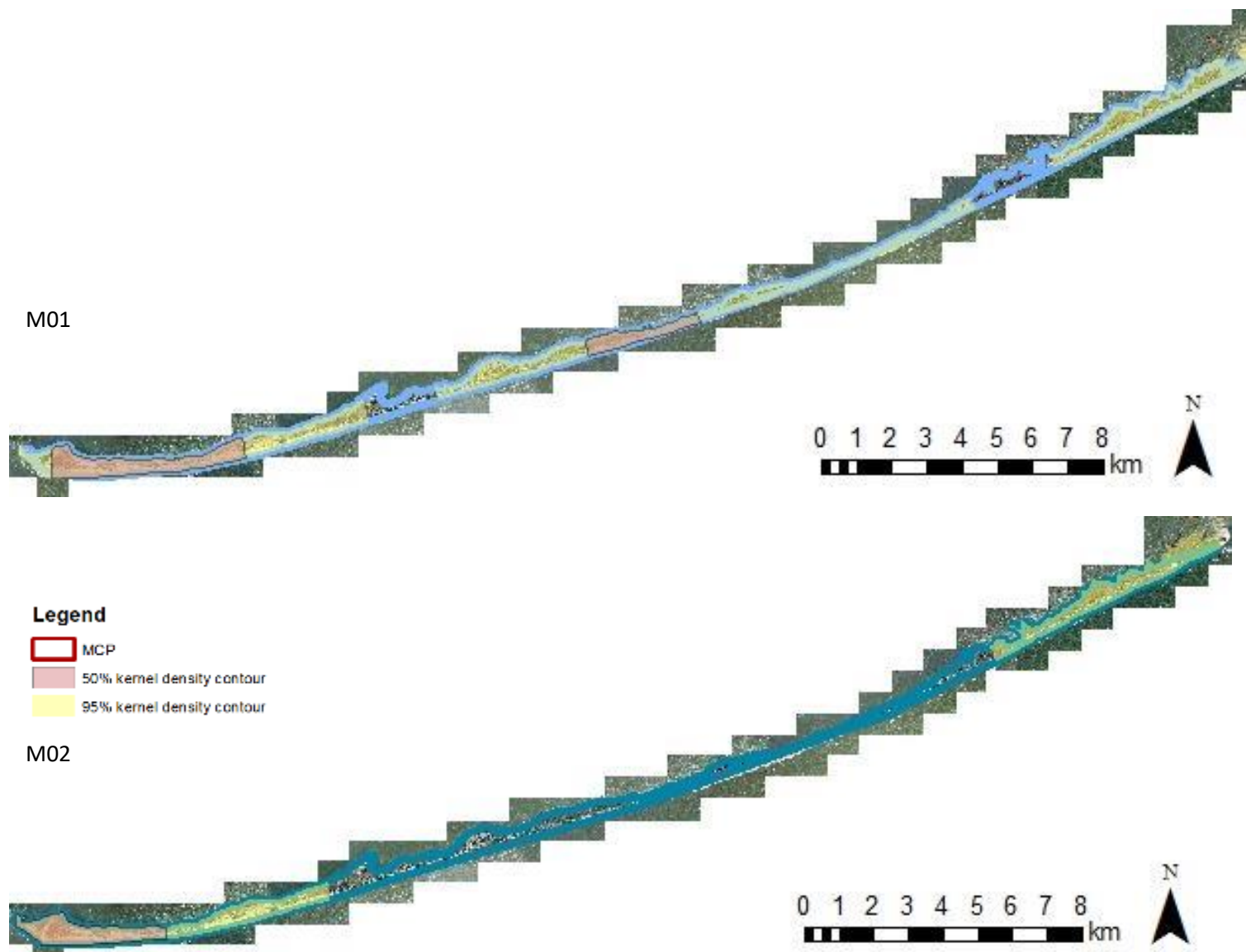


Figure 49. MCPs, 50% and 95% kernel density contours for 2 male red foxes originally captured at Robert Moses State Park during Fall–Winter 2015.

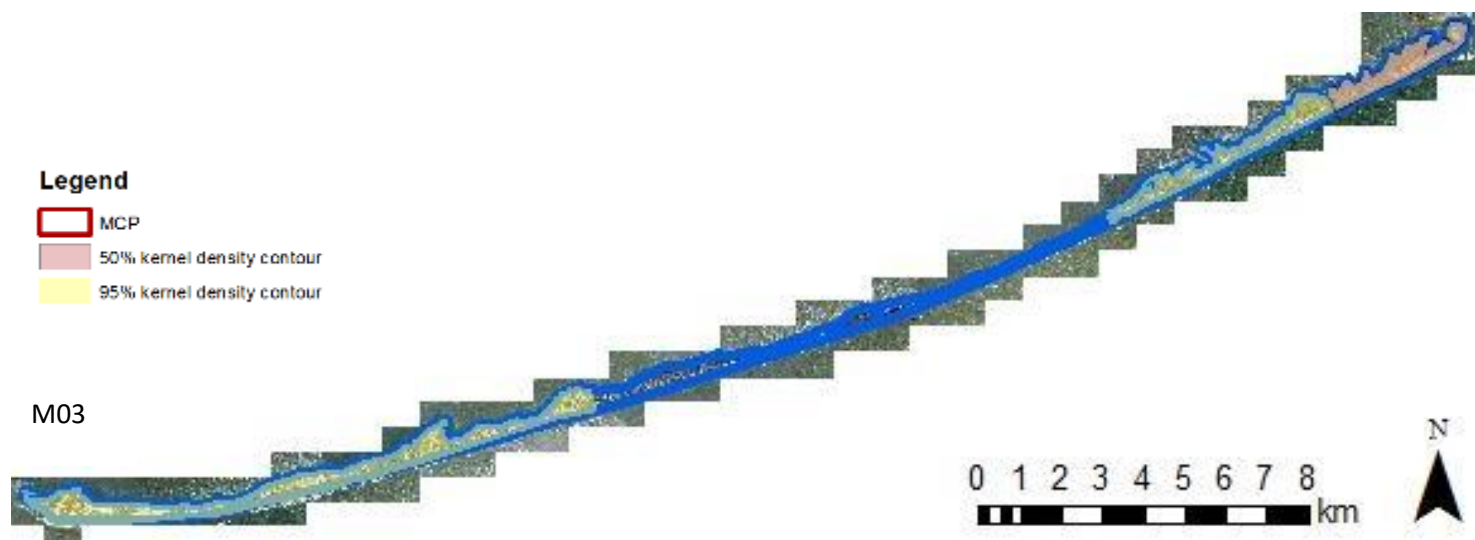


Figure 50. MCP, 50% and 95% kernel density contour for 1 male red fox originally collared at Fire Island National Seashore in Fall–Winter 2015.

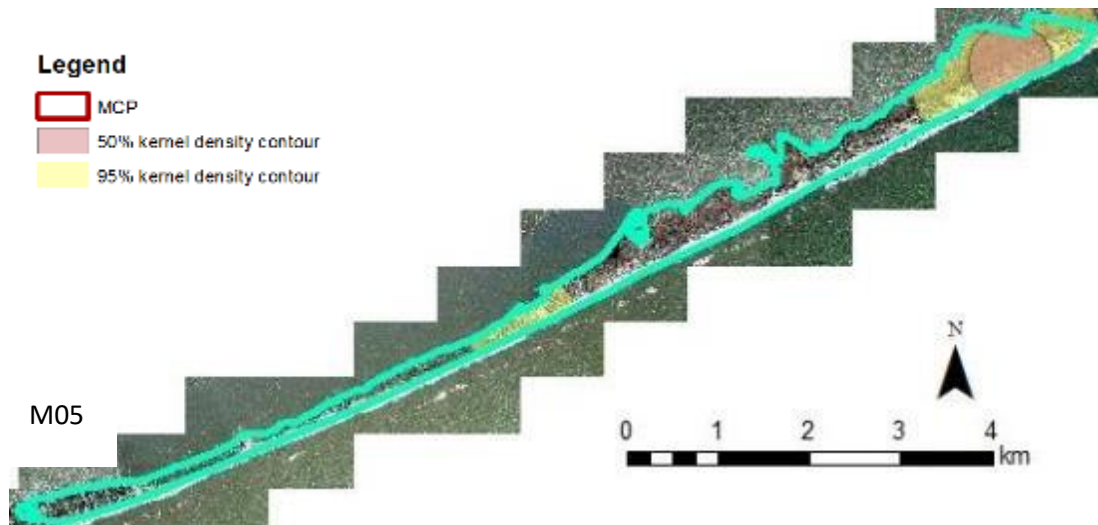


Figure 51. MCP, 50% and 95% kernel density contour for 1 male red fox at Fire Island National Seashore in Fall–Winter 2015.

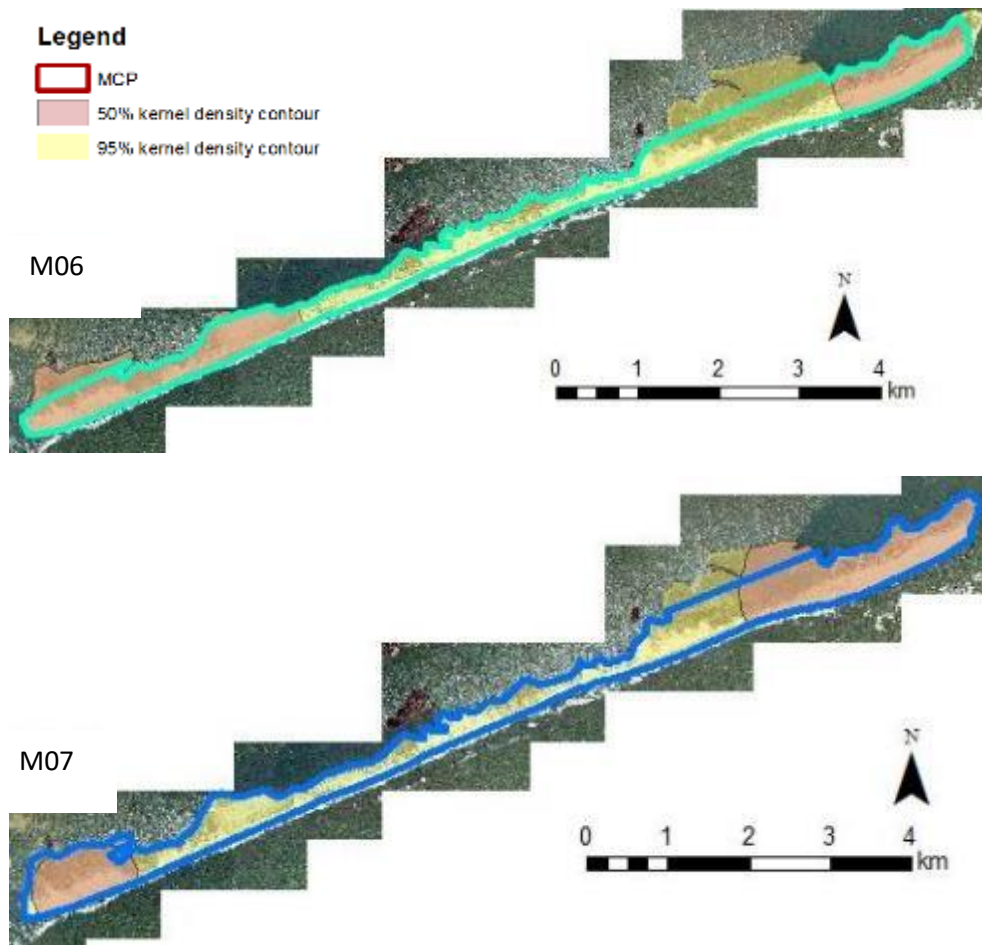


Figure 52. MCP, 50% and 95% kernel density contour for 2 male red foxes at Fire Island National Seashore (Old Inlet East) and Smith Point County Park in Fall–Winter 2015.

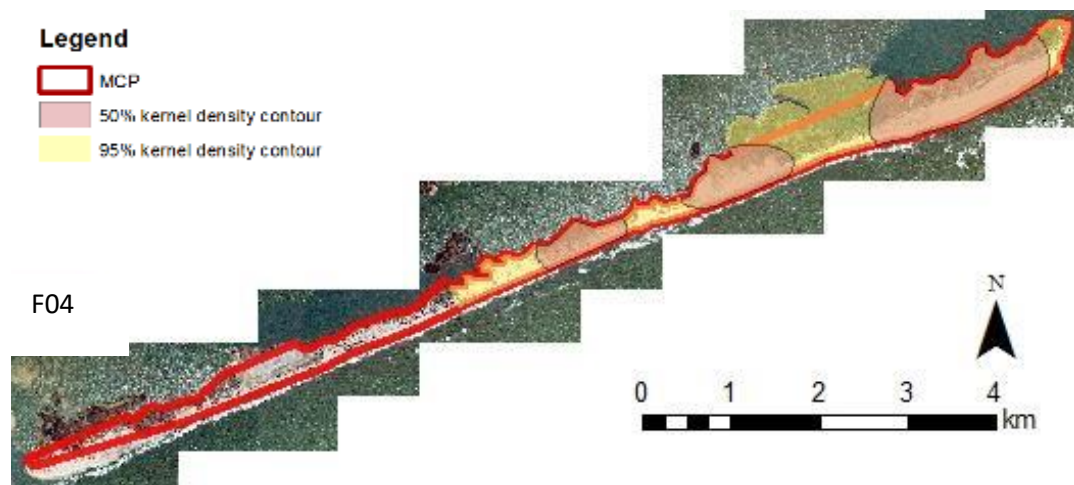
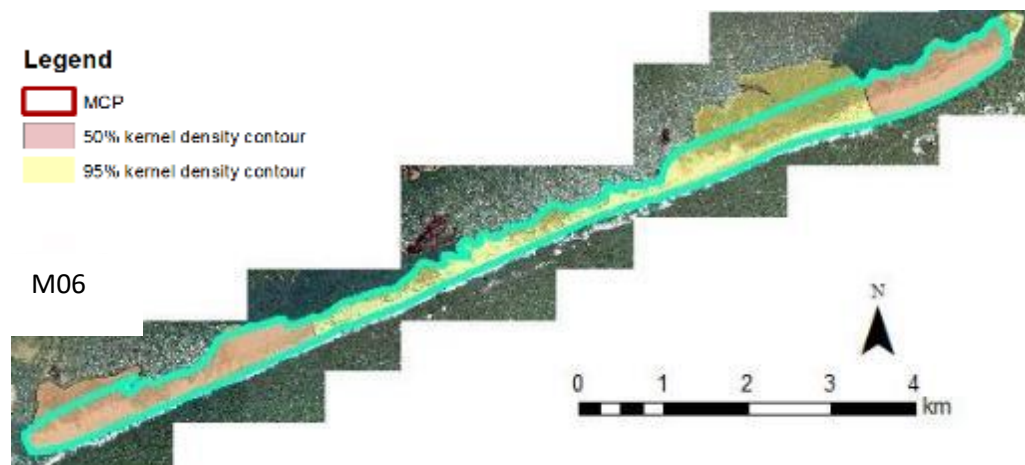


Figure 53. MCP, 50% and 95% kernel density contour for 1 female red fox at Fire Island National Seashore (Old Inlet East) and Smith Point County Park in Fall–Winter 2015.



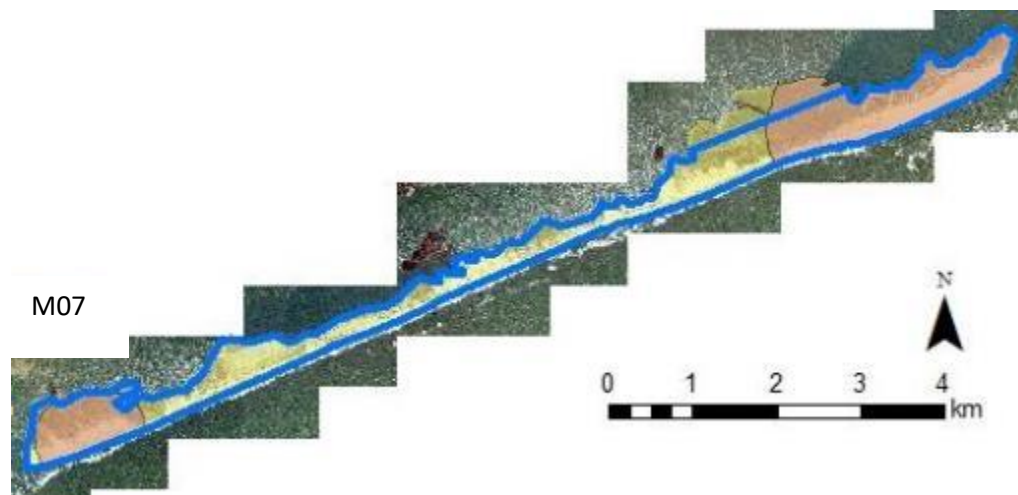


Figure 54. MCP, 50% and 95% kernel density contour for 2 male red foxes at Fire Island National Seashore (Old Inlet East) and Smith Point County Park in Fall–Winter 2015.

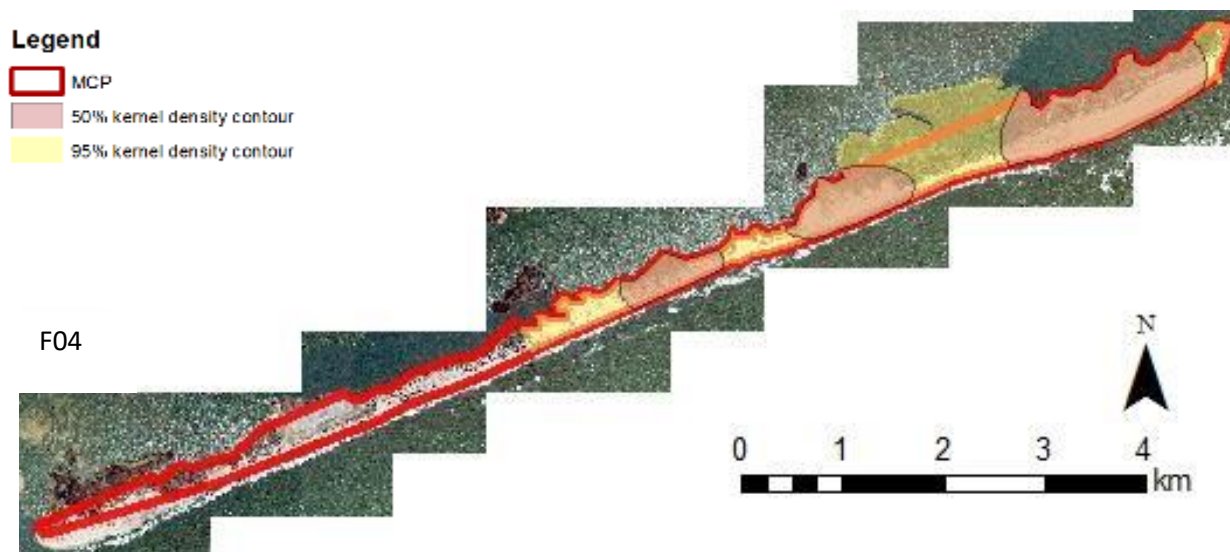


Figure 55. MCP, 50% and 95% kernel density contour for 1 female red fox at Fire Island National Seashore (Old Inlet East) and Smith Point County Park in Fall–Winter 2015.



Figure 56. 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.

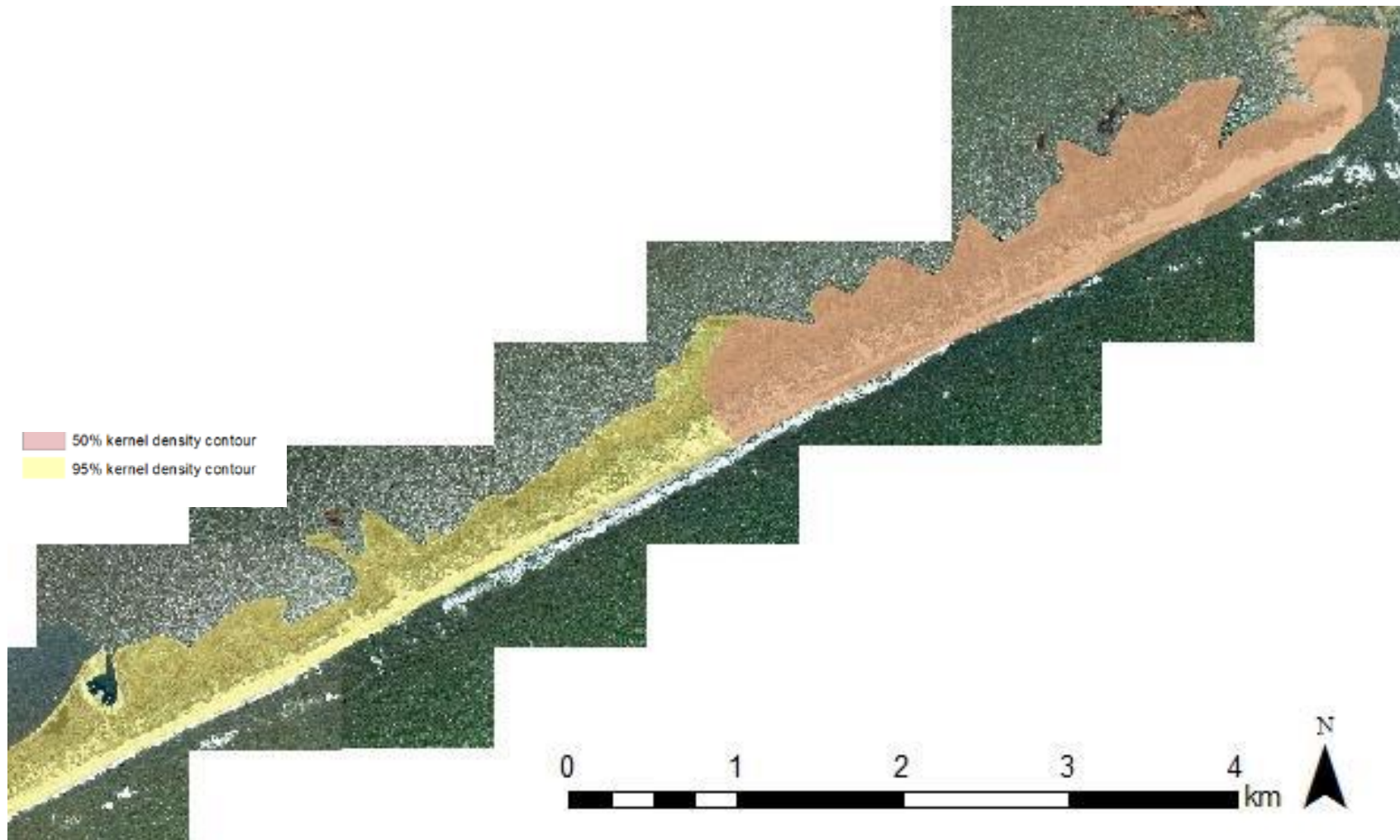


Figure 57. 50% and 95% kernel density contours for all GPS-collared red foxes in Fall–Winter 2015 at Fire Island National Seashore (Watch Hill to inlet) overlaid on top of each other.

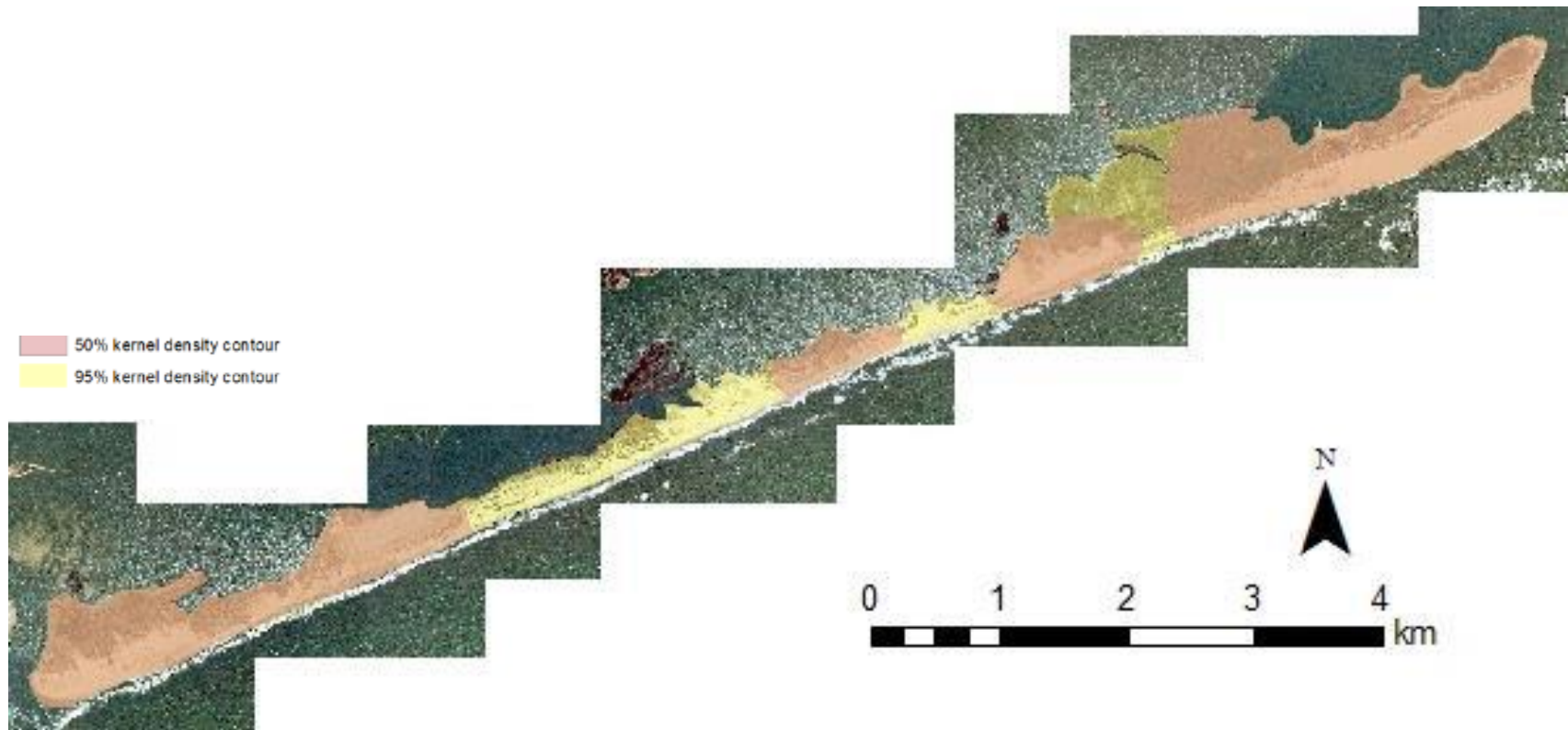


Figure 58. 50% and 95% kernel density contours for all GPS-collared red foxes in Fall–Winter 2015 at Fire Island National Seashore (Old Inlet East) and Smith Point County Park overlaid on top of each other.

Spring-Summer 2016

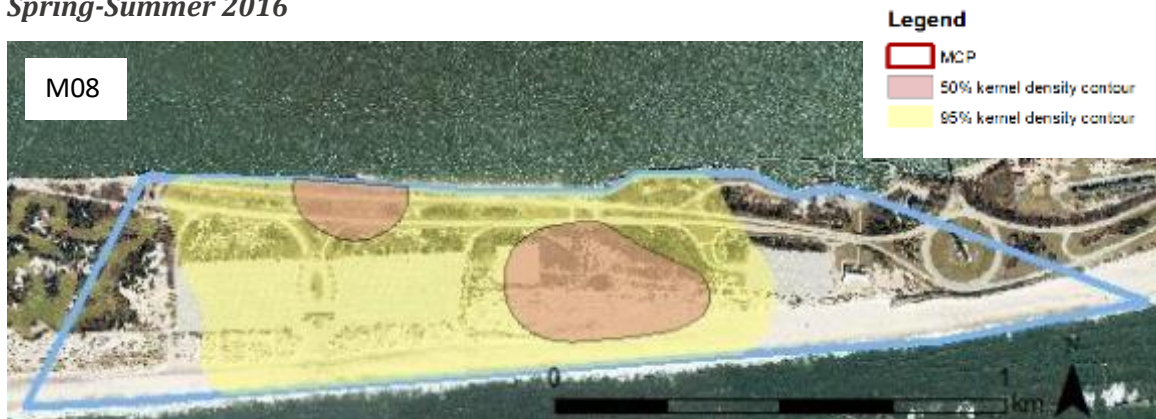


Figure 59. MCP, 50% and 95% kernel density contours for 1 male red fox at Robert Moses State Park, Spring-Summer 2016.

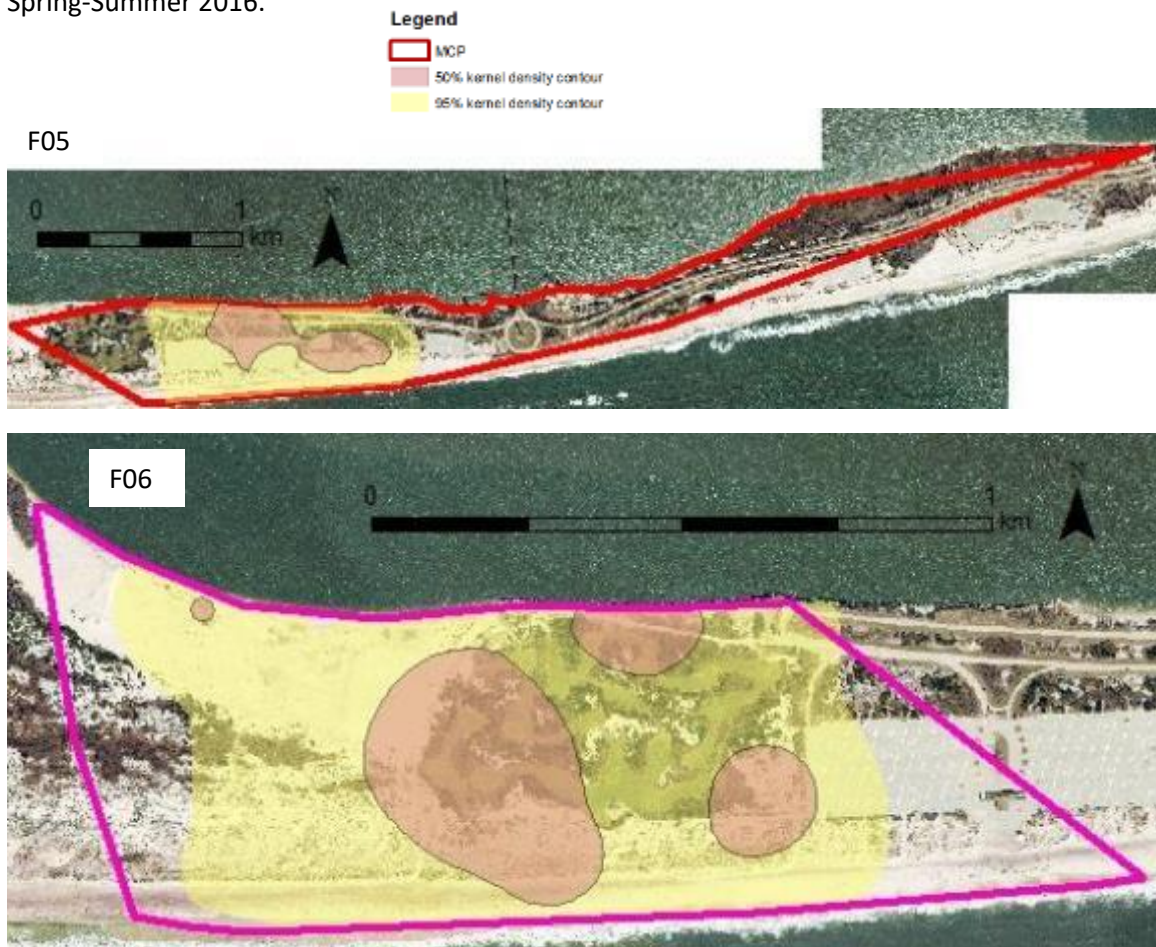


Figure 60. MCP, 50% and 95% kernel density contours for 2 female red foxes at Robert Moses State Park, Spring-Summer 2016.

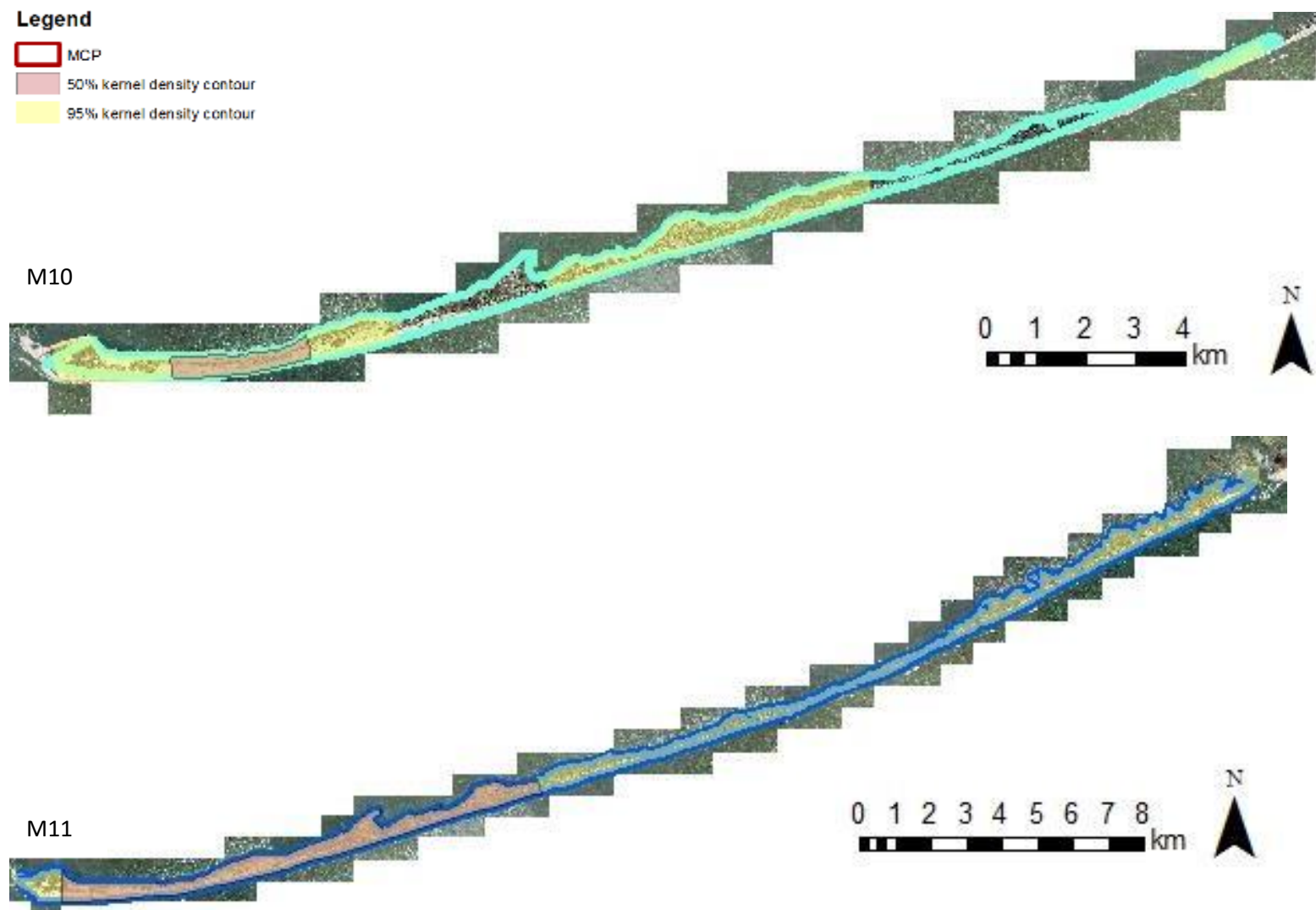


Figure 61. MCP, 50% and 95% kernel density contours for 2 male red foxes originally captured at Robert Moses State Park, Spring–Summer 2016.

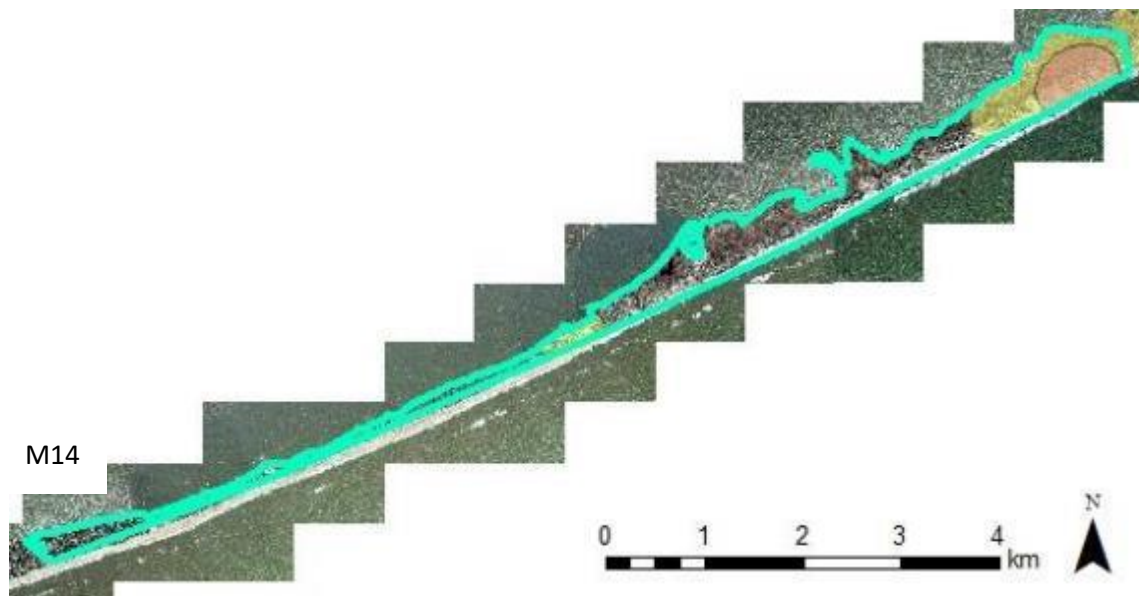
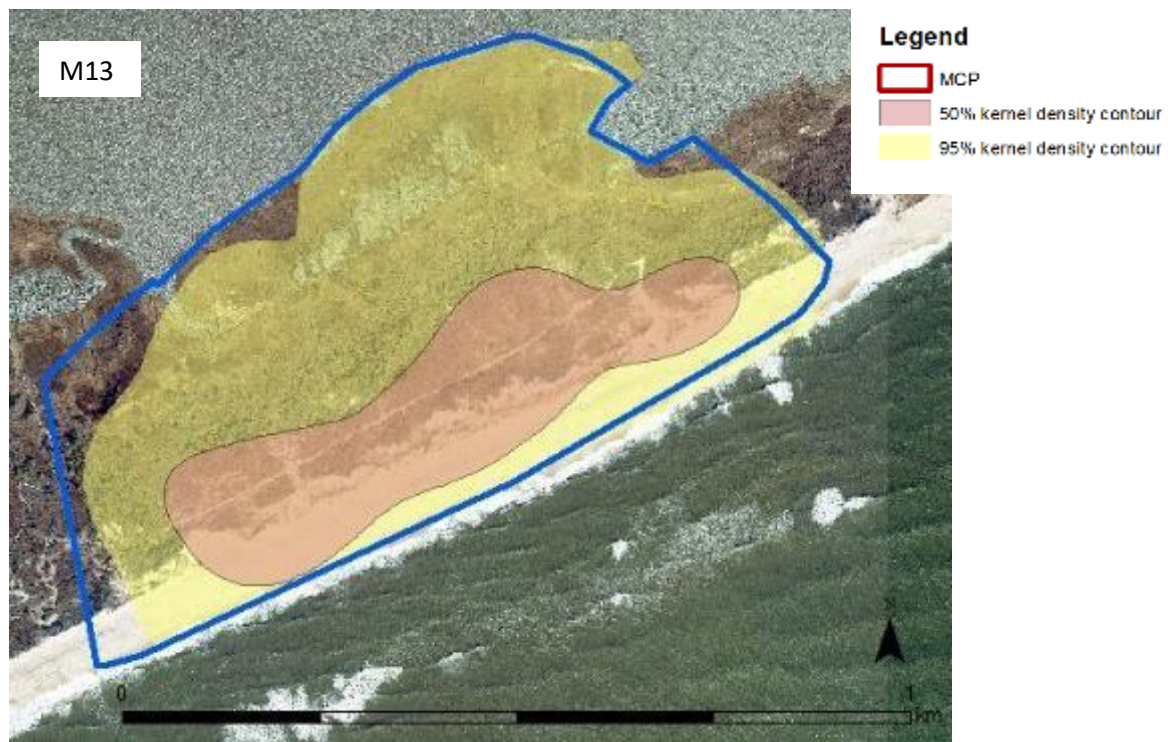
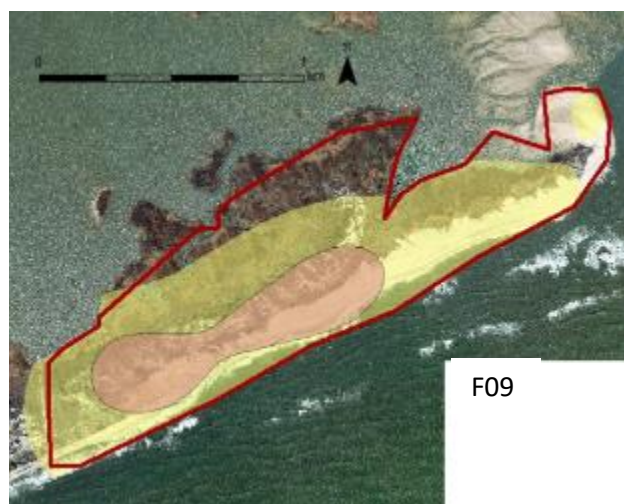
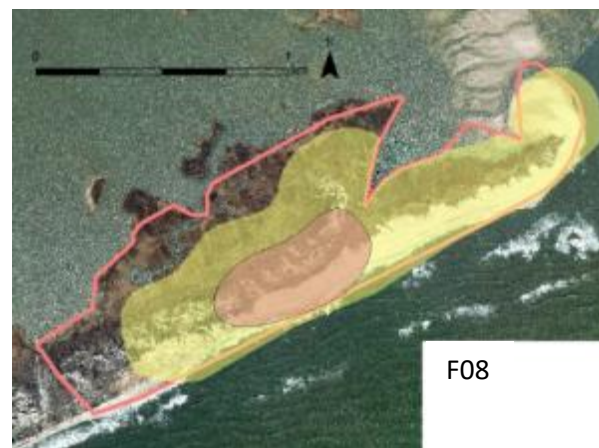


Figure 62. MCP, 50% and 95% kernel density contours for 2 male red foxes at Fire Island National Seashore, Spring–Summer 2016.



Legend

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

Figure 63. MCP, 50% and 95% kernel density contours for 3 female red foxes at Fire Island National Seashore, Spring–Summer 2016.

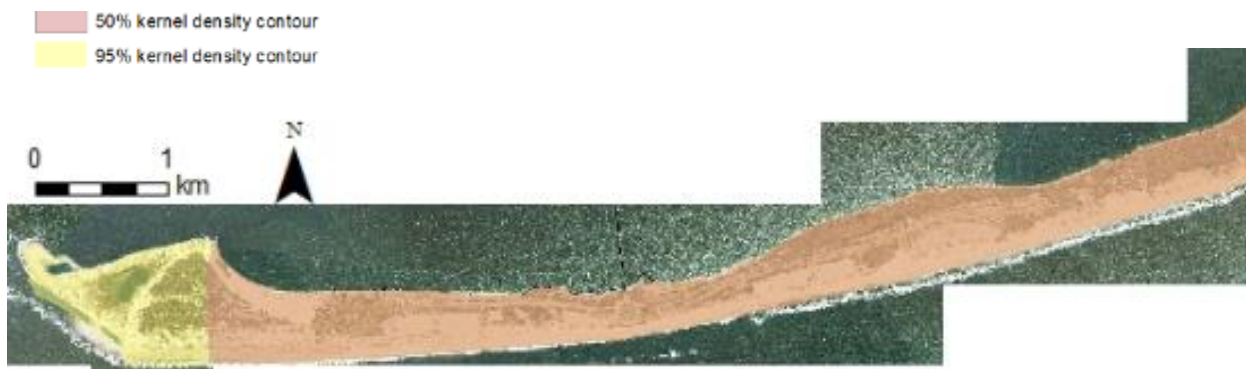


Figure 64. 50% and 95% kernel density contours for all GPS-collared red foxes in Spring-Summer 2016 at Robert Moses State Park overlaid on top of each other.

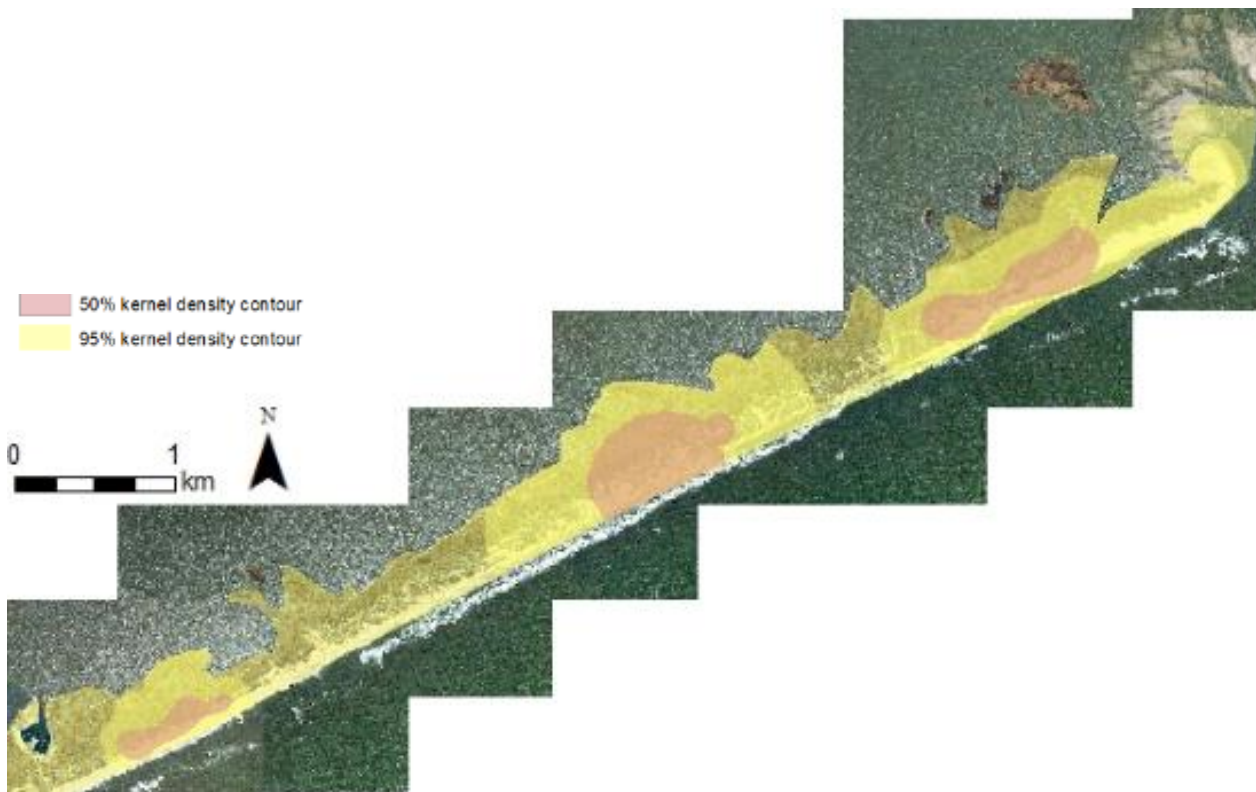


Figure 65. 50% and 95% kernel density contours for all GPS-collared red foxes in Spring-Summer 2016 at Fire Island National Seashore (west of Old Inlet) overlaid on top of each other.

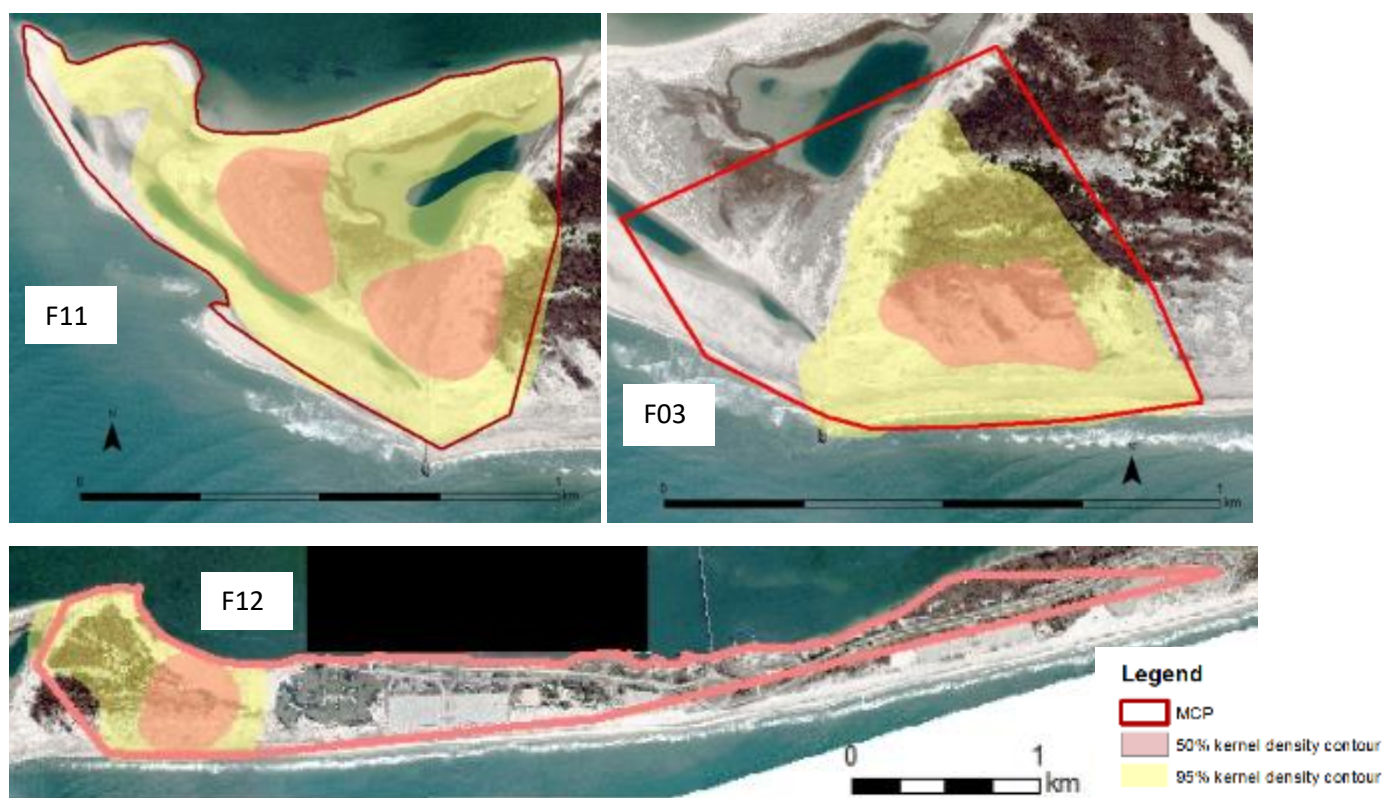


Figure 66. MCPs (solid lines), 50% and 95% kernel density contours for 3 female red foxes at Robert Moses State Park, Spring–Summer 2017.

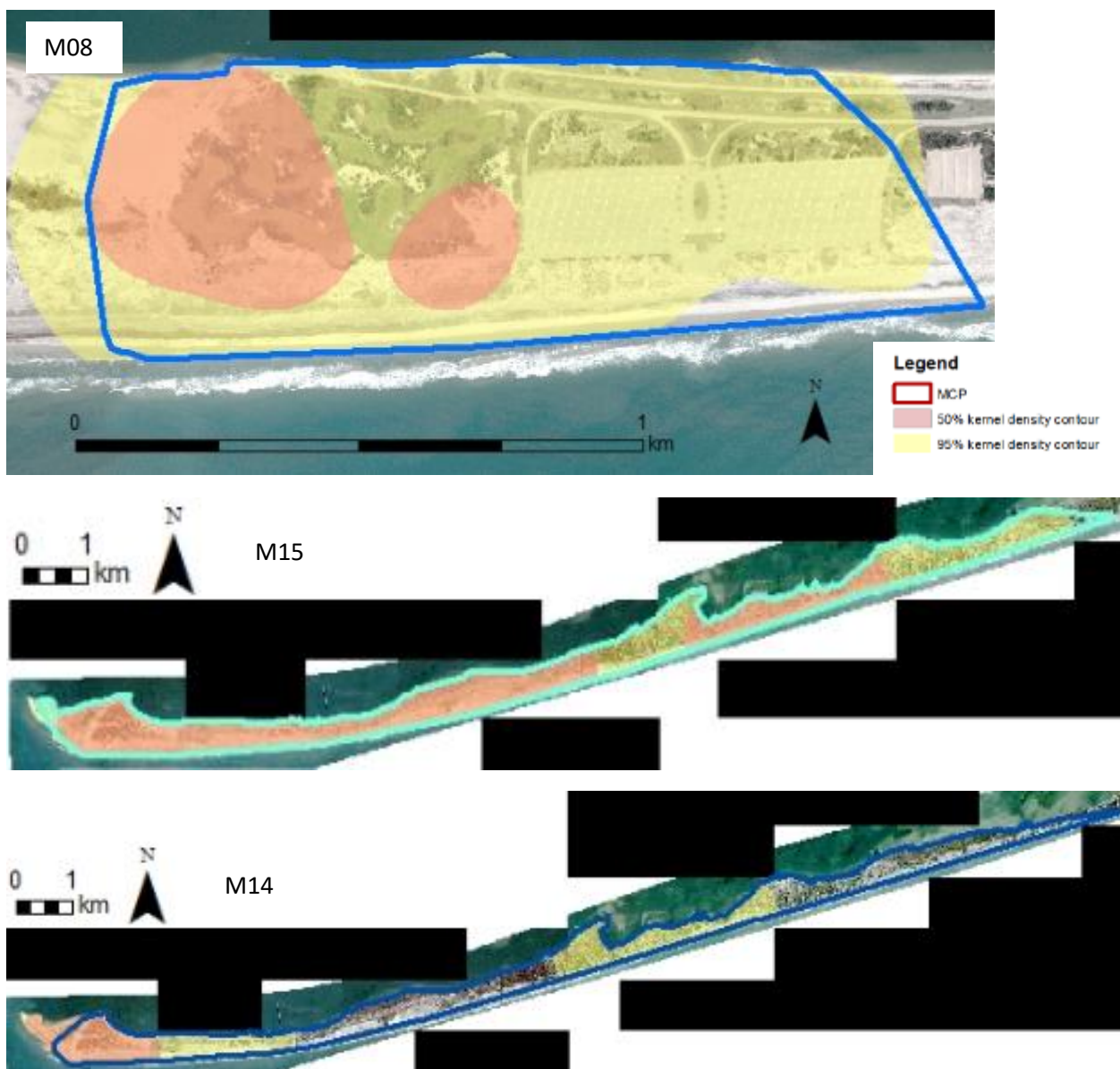


Figure 67. MCPs, 50% and 95% kernel density contours for 3 male red foxes originally captured at Robert Moses State Park, Spring–Summer 2017.

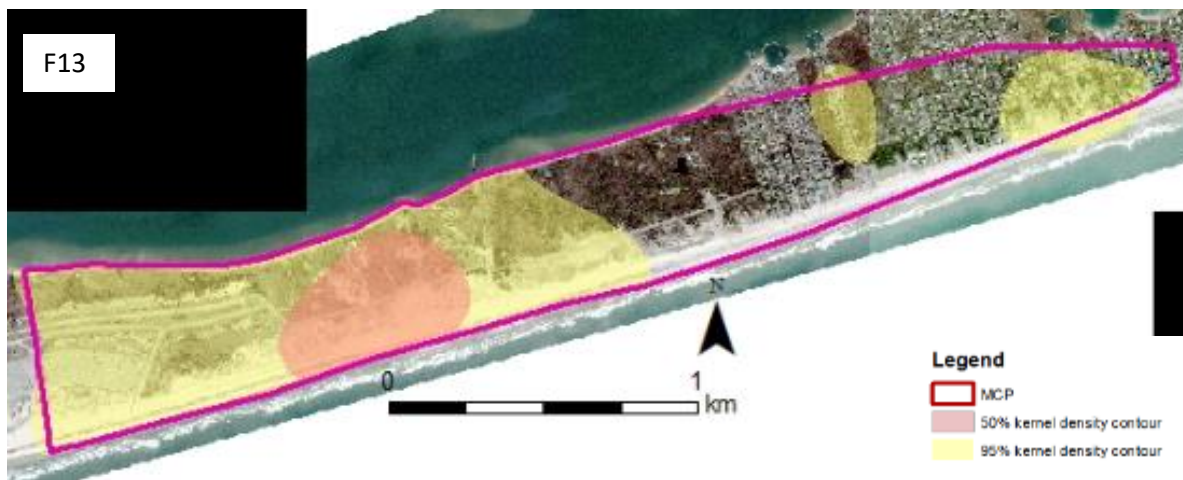


Figure 68. MCP, 50% and 95% kernel density contours for one female red fox at Robert Moses State Park/Fire Island National Seashore Lighthouse Tract, Spring–Summer 2017.

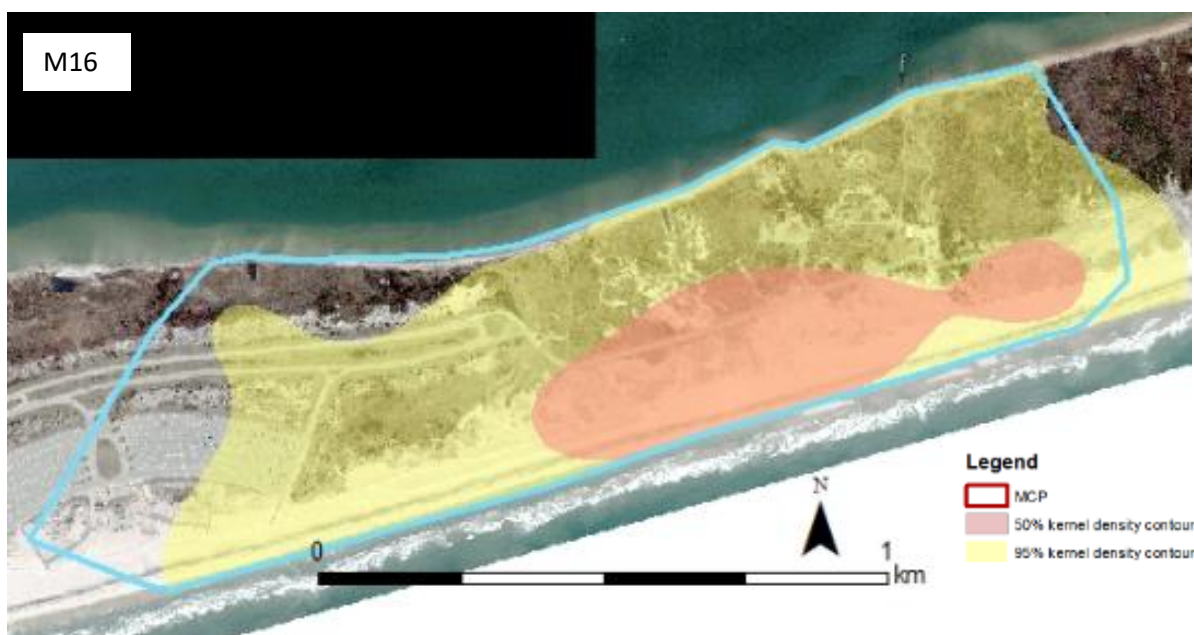


Figure 69. MCP, 50% and 95% kernel density contours for one male red fox at Robert Moses State Park/Fire Island National Seashore Lighthouse Tract, Spring–Summer 2017.

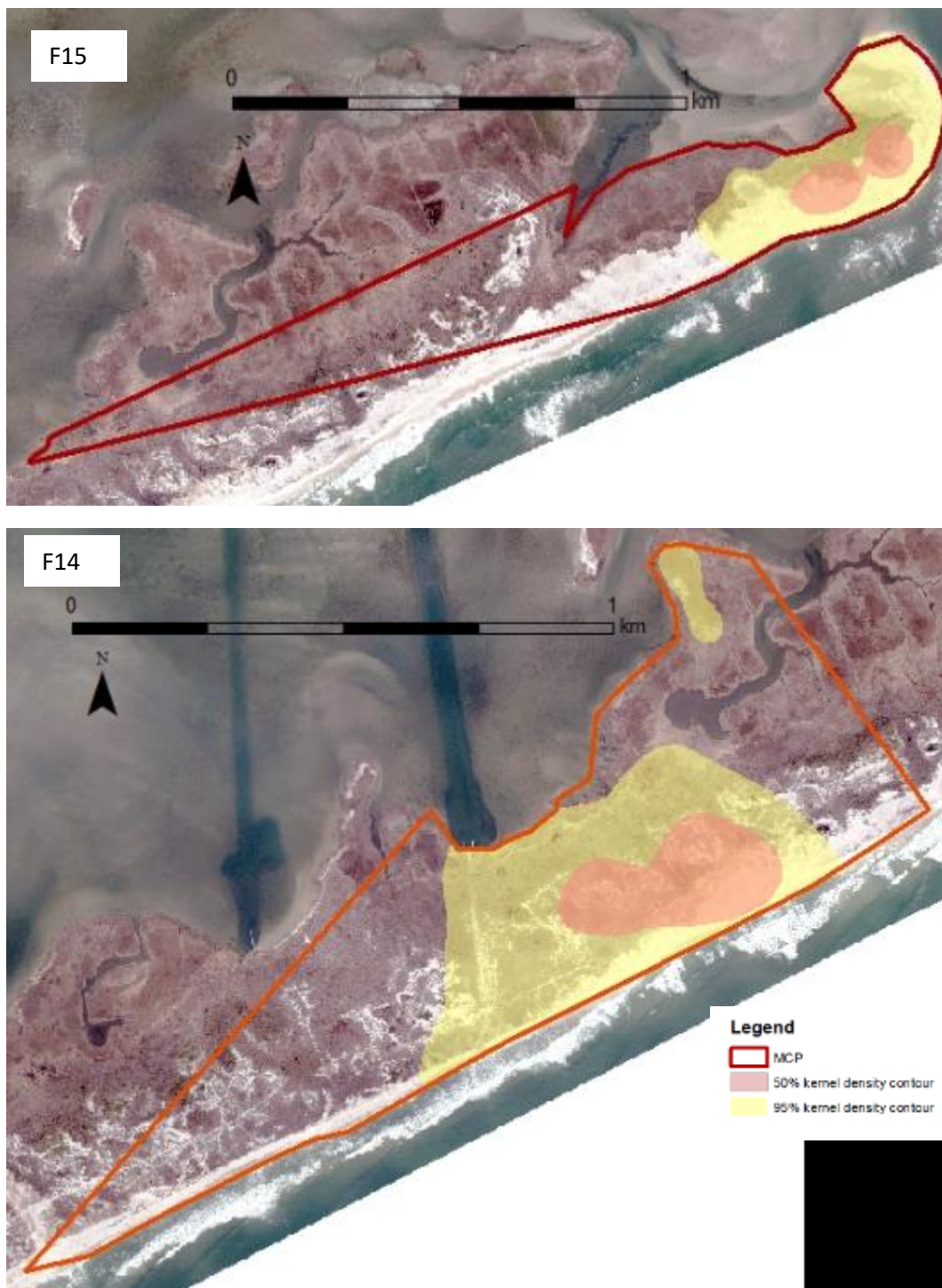


Figure 70. MCPs, 50% and 95% kernel density contours for 2 female red foxes at Fire Island National Seashore, Spring–Summer 2017.

M17

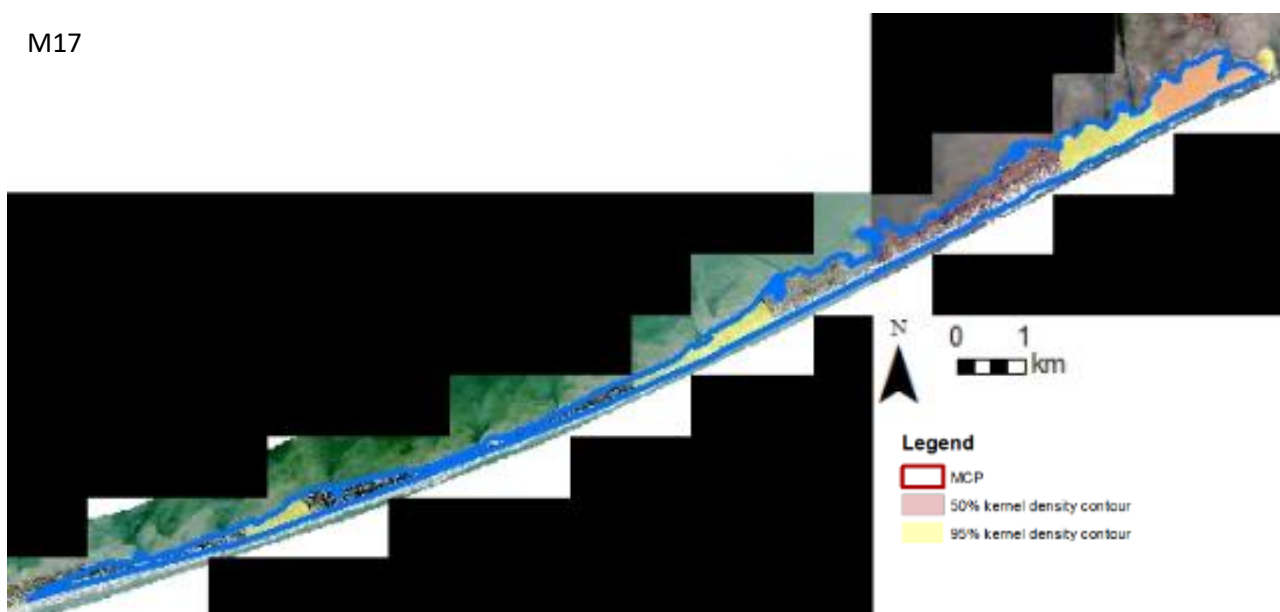


Figure 71. MCP, 50% and 95% kernel density contours for one male red fox at Fire Island National Seashore Lighthouse Tract, Spring–Summer 2017.

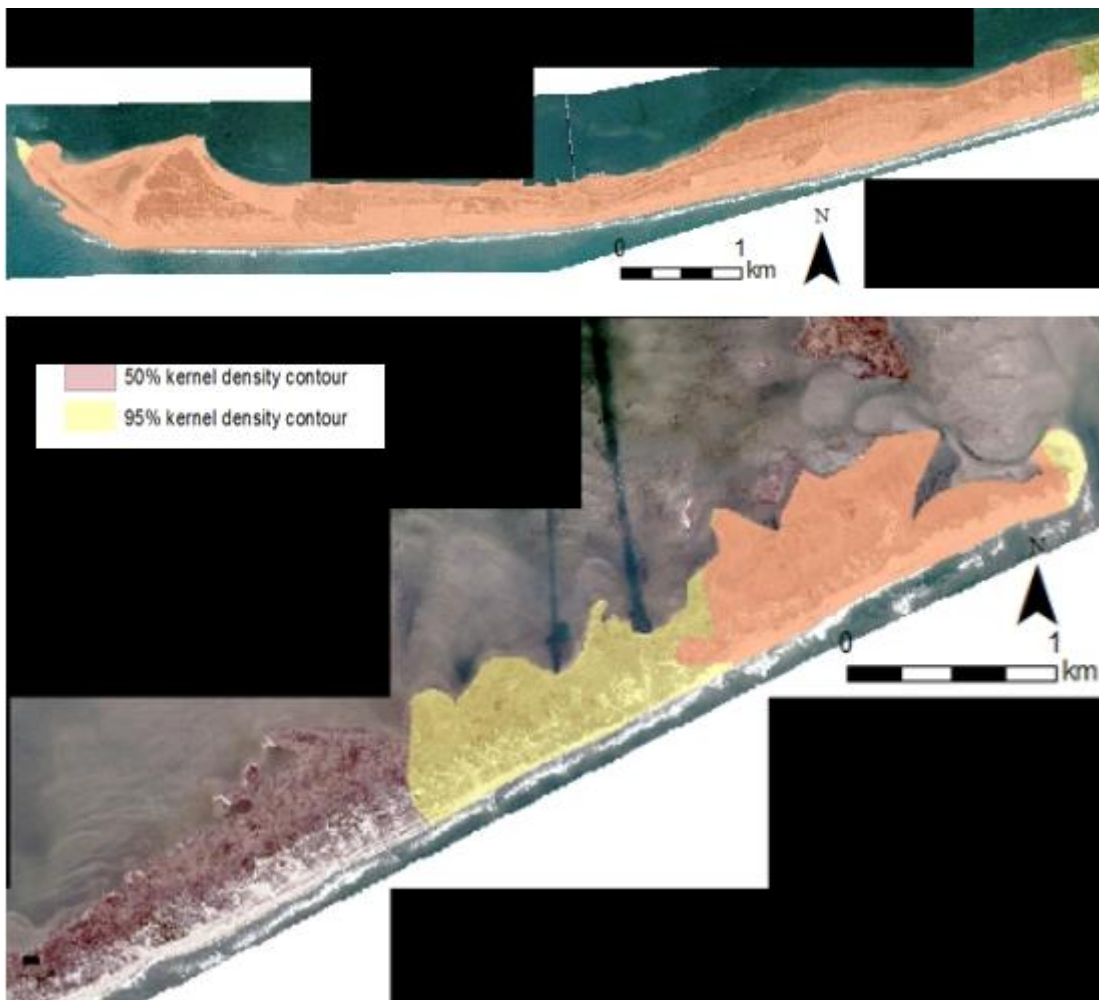


Figure 72. 50% and 95% kernel density contours for all GPS-collared red foxes in Spring–Summer 2017 at Robert Moses State Park and Fire Island National Seashore Lighthouse Tract (top) and Fire Island National Seashore Wilderness Area (west of Old Inlet, bottom) overlaid on top of each other.

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

Maps of fox locations during high frequency location sampling surrounding nest exclosure 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:

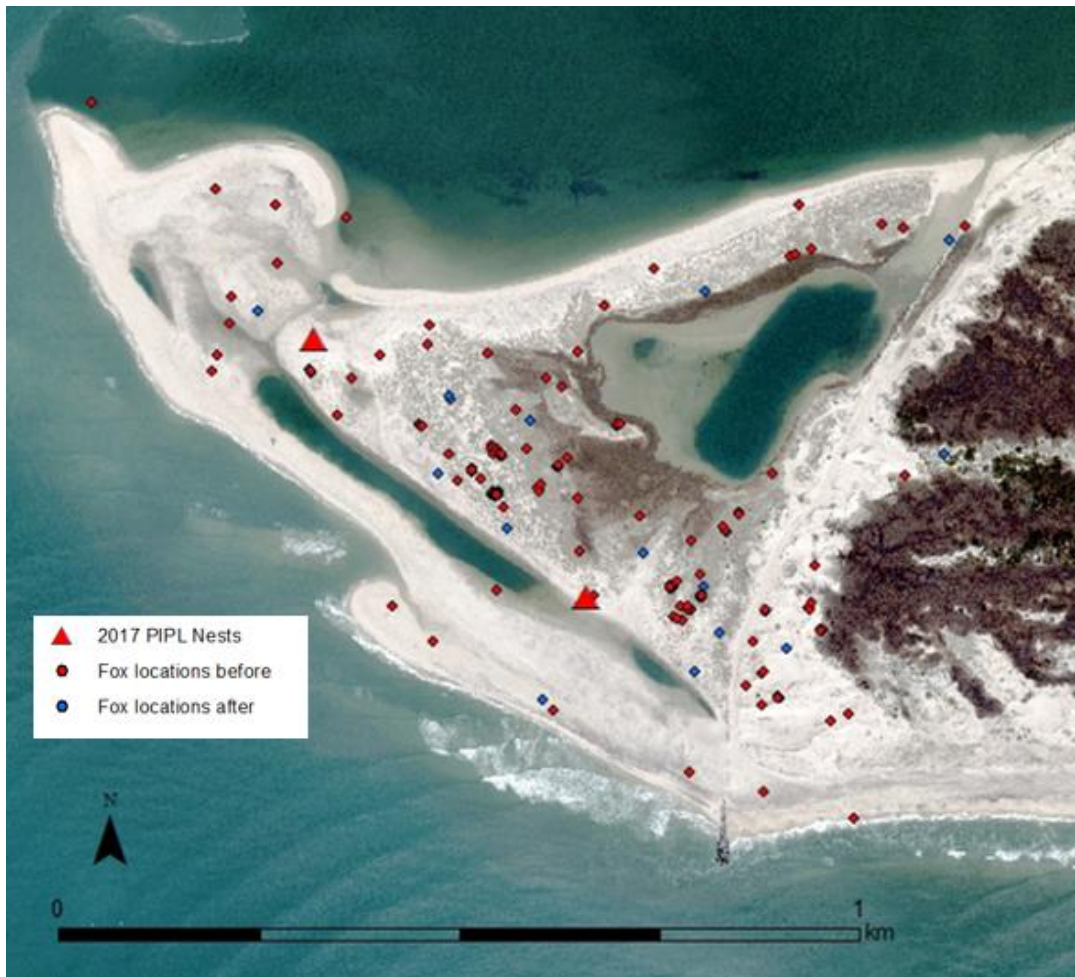


Figure 73. Locations for F11 before and after nest exclosure at Robert Moses State Park, PIPL nests 203 and 204.

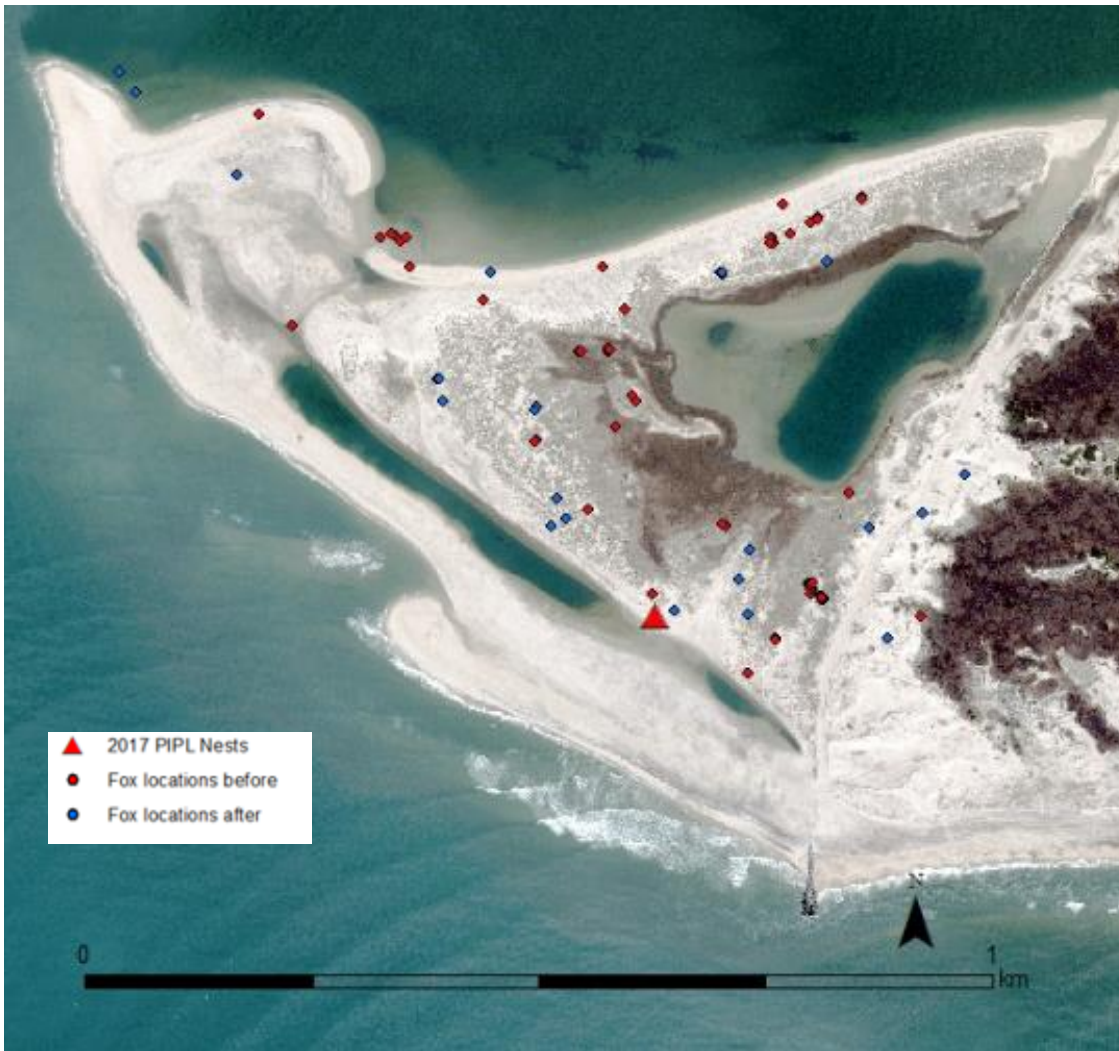


Figure 74. Locations for F11 before and after nest hatch date at Robert Moses State Park, PIPL nest 203 (nest 204 failed).

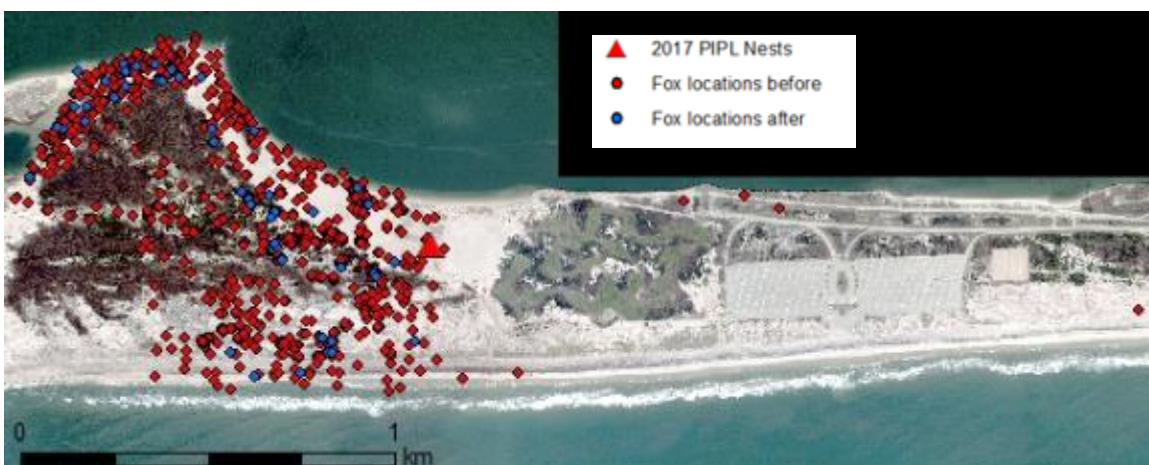


Figure 75. Locations for F12 before and after nest exclosure at Robert Moses State Park, PIPL nest 205.

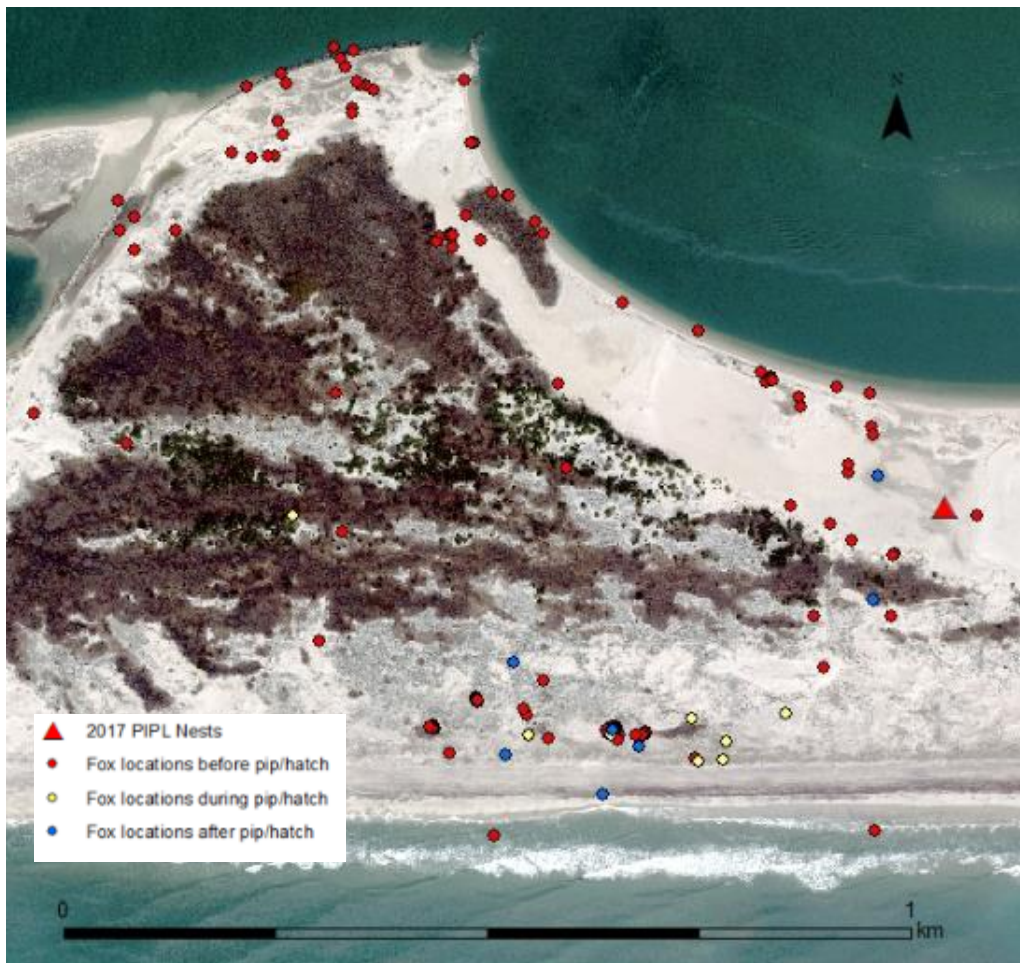


Figure 76. Locations for F12 before, during, and after nest pipping/hatching at Robert Moses State Park, PIPL nest 205.

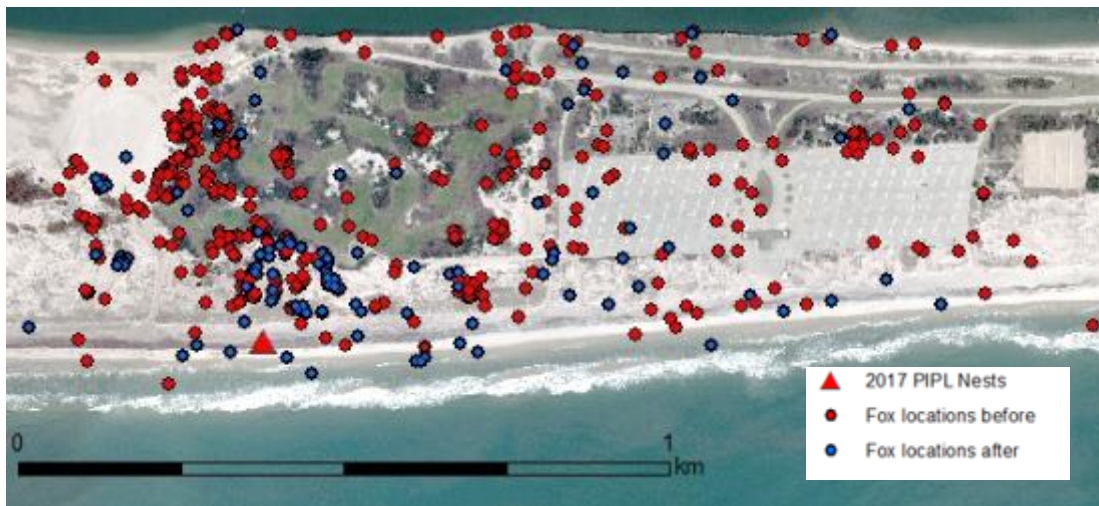


Figure 77. Locations for M08 before and after nest exclosure at Robert Moses State Park, PIPL nest 214.

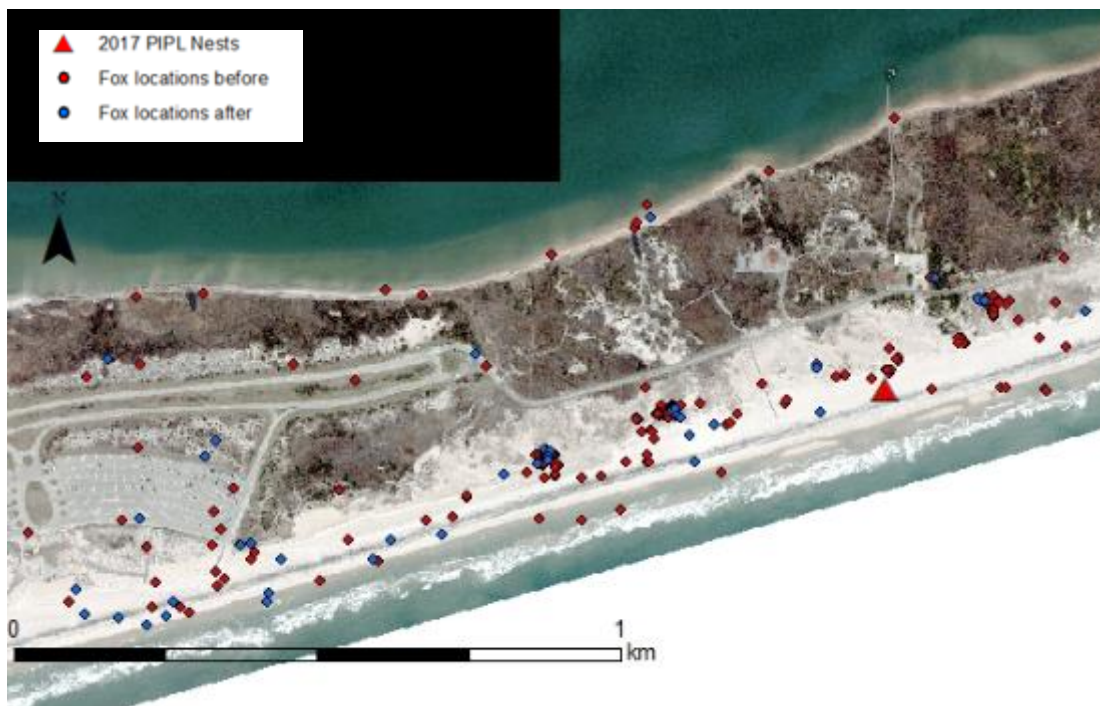


Figure 78. Locations for M16 before and after nest exclosure at Robert Moses State Park, PIPL nest 211.

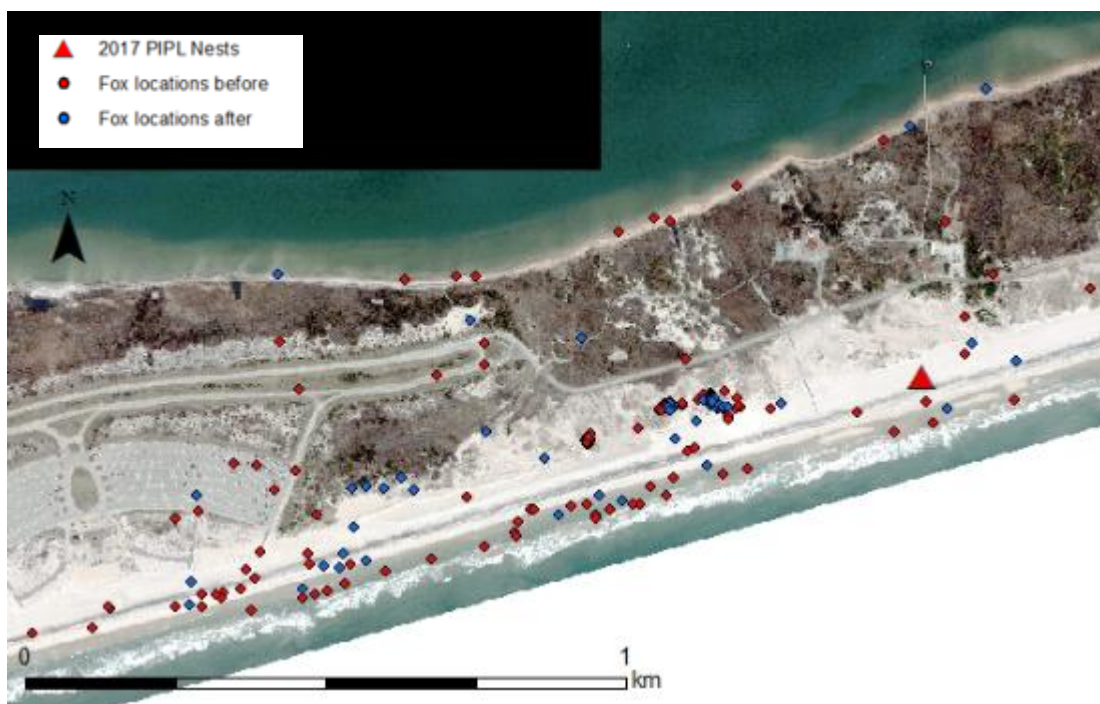


Figure 79. Locations for F13 before and after nest exclosure at Robert Moses State Park, PIPL nest 211.

Spring-Summer 2016 monitoring period:

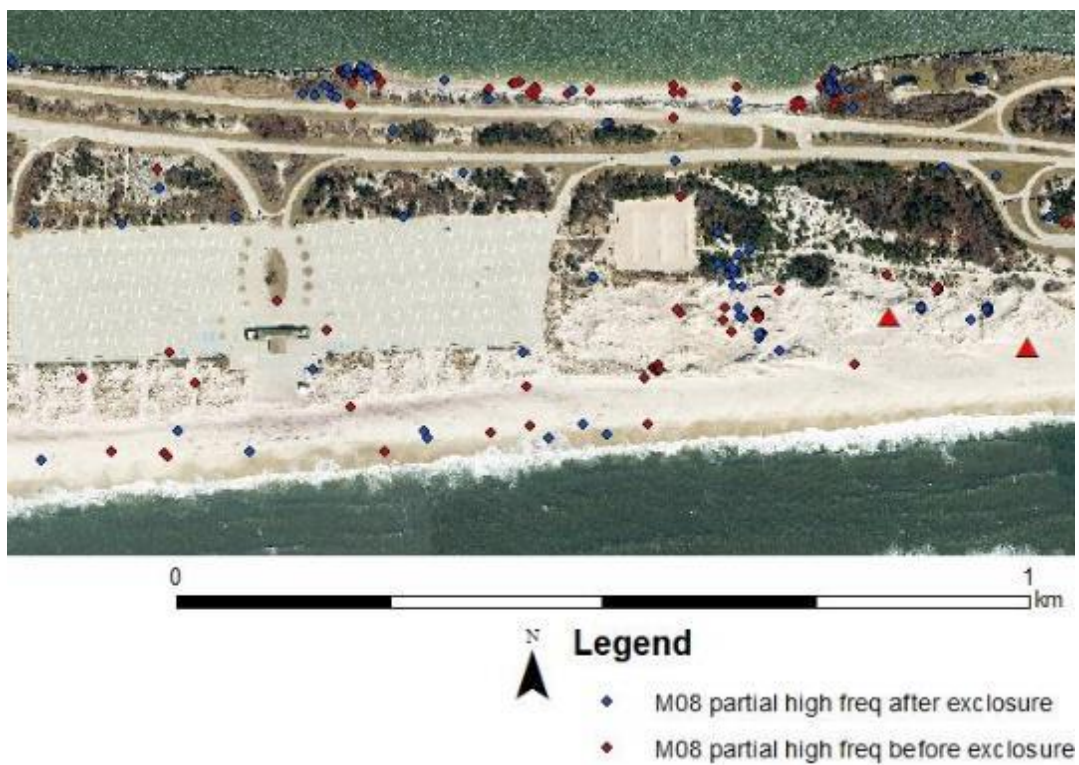
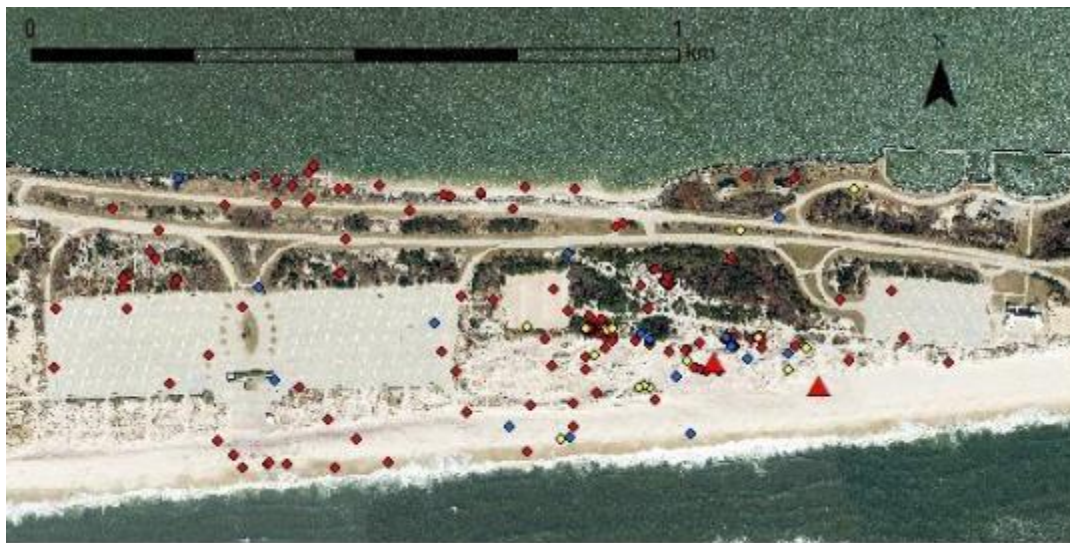


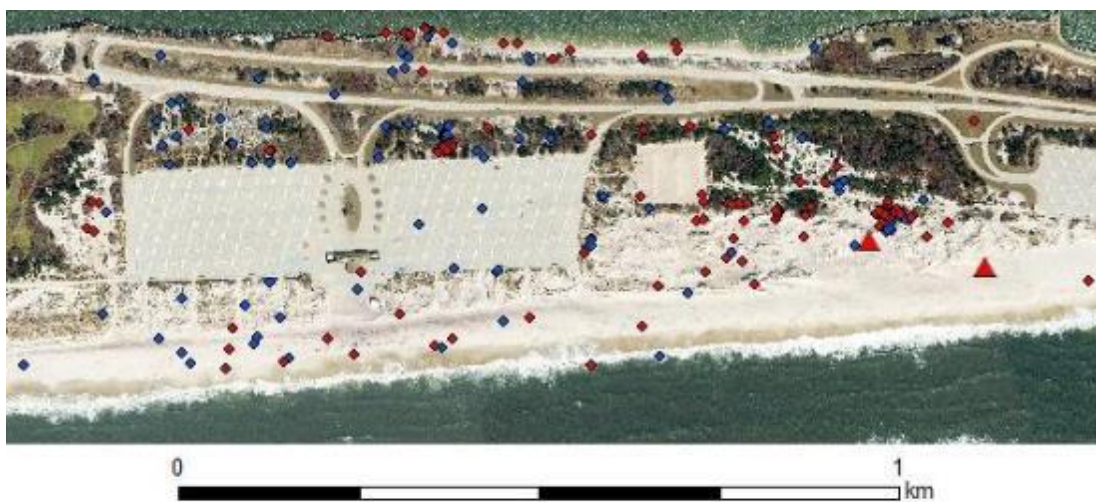
Figure 6. Locations for fox M08 before and after nest exclosure at Robert Moses State Park, PIPL nest 211.



Legend

- ▲ 2016 PIPL nests
- ◆ M08 reg freq after hatch
- ◆ M08 reg freq during pip/hatch
- ◆ M08 partial high freq pre-pip/hatch

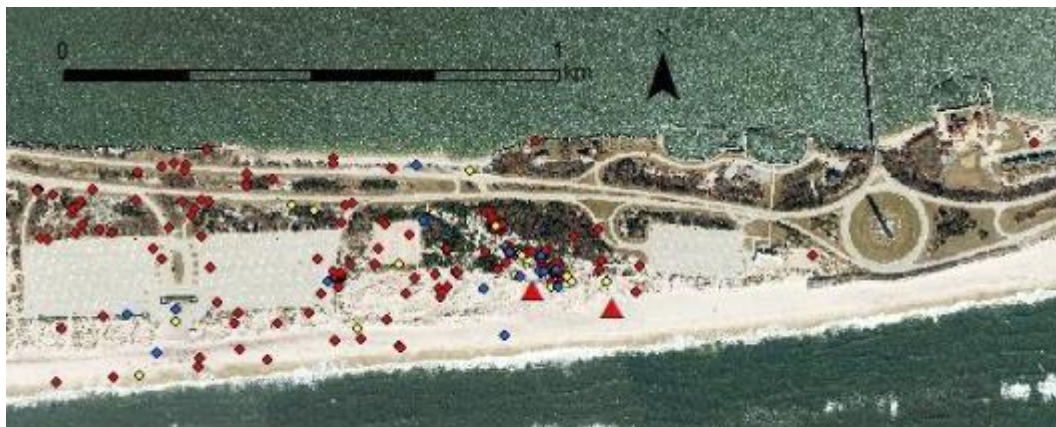
Figure 81. Locations for fox M08 before, during, and after nest pipping/hatching at Robert Moses State Park, PIPL nest 211.



Legend

- ◆ F05 partial high freq before exclosure
- ◆ F05 partial high freq after exclosure
- ▲ 2016 PIPL nests

Figure 82. Locations for fox F05 before and after nest exclosure at Robert Moses State Park, PIPL nest 211.



Legend

- ▲ 2016 PIPL nests
- ◆ F05 reg freq after hatch
- ◇ F05 reg freq during pip/hatch
- ◆ F05 partial high freq pre-pip/hatch

Figure 83. Locations for fox F05 before, during, and after nest pipping/hatching at Robert Moses State Park, PIPL nest 211.



Legend

- ▲ 2016 PIPL nests
- ◆ F06 partial high freq before exclosure
- ◆ F06 high freq after exclosure

Figure 84. Locations for fox F06 before and after nest exclosure at Robert Moses State Park, PIPL nests 204 and 205 (on southern beachfront).



Figure 85. Locations for fox F06 before, during, and after nest pipping/hatching at Robert Moses State Park, PIPL nests 204 and 205 (on southern beachfront).

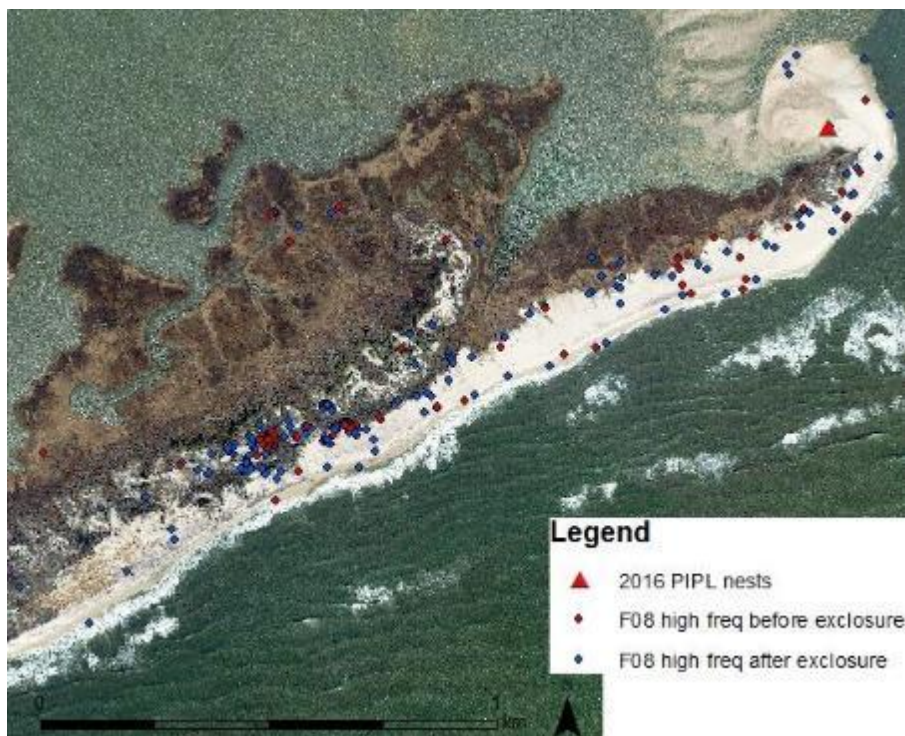


Figure 86. Locations for fox F08 before and after nest exclosure at Fire Island National Seashore, PIPL nest 32.

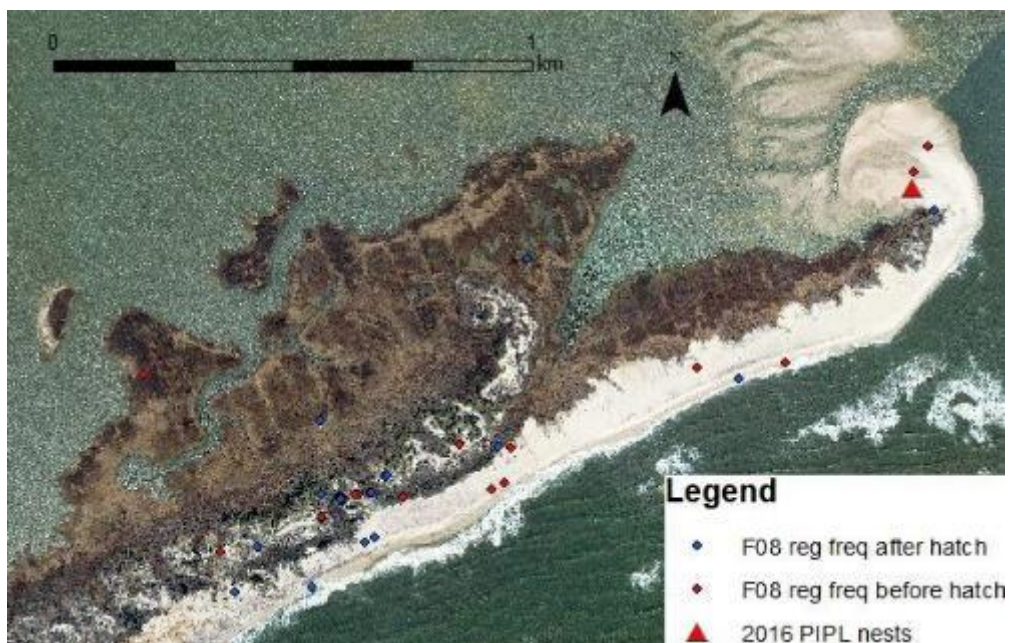


Figure 87. Locations for fox F08 before and after nest hatching at Fire Island National Seashore, PIPL nest 32.



Figure 88. Locations for fox F09 before and after nest exclosure at Fire Island National Seashore, PIPL nest 32.



Figure 89. Locations for fox F09 before and after nest hatching at Fire Island National Seashore, PIPL nest 32

