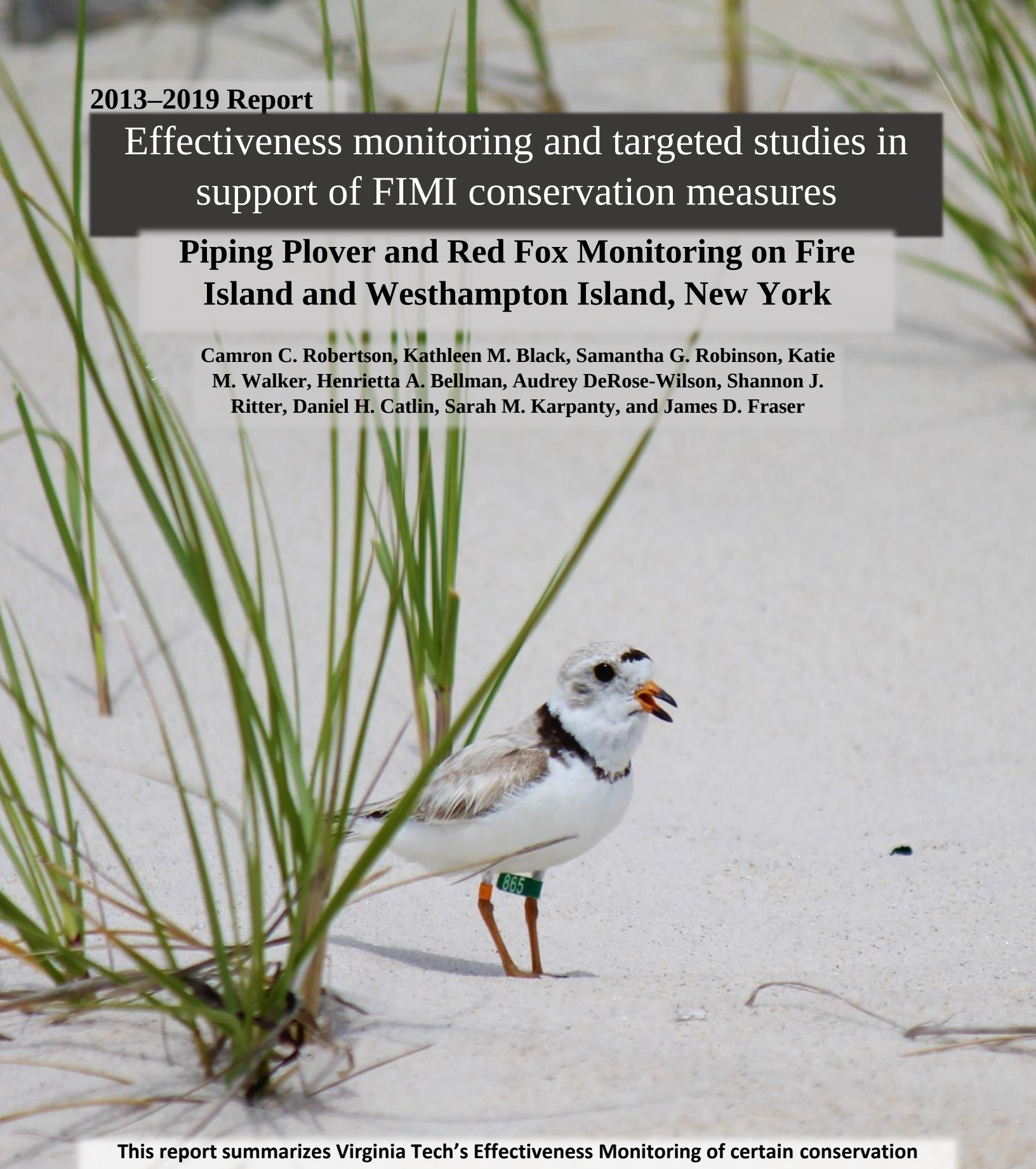
2013–2019 Report

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

Camron C. Robertson, Kathleen M. Black, Samantha G. Robinson, Katie M. Walker, Henrietta A. Bellman, Audrey DeRose-Wilson, Shannon J. Ritter, Daniel H. Catlin, Sarah M. Karpanty, and James D. Fraser



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

VIRGINIA TECH SHOREBIRD PROGRAM

With significant contributions from:

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, S. Sullivan, R. Moylan, S. Ball-Douglas, R. Bechtold, V. Schabert, N. Giordano, M. Zak, J. Domino, K. Rosamond, N. Bosco, P. Sonnega, L. Hermanns, R. Winters, J. Deans, B. Hardt, A. Zeller, C. Helmke, E. Zawatski, M.Carroll, E. Swiecki, K. Attinger, J. Hall, C. Howell, 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: Robertson, C. C., K. M. Black, S. G. Robinson, K. M. Walker, H. A. Bellman, A. DeRose-Wilson, S. J. Ritter, D. H. Catlin, S. M. Karpanty, and J. D. Fraser. 2019. 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 provided by Katie Walker

Green flag ‘865’ was banded and photographed at Cupsogue Beach County Park in Westhampton Island, NY in June, 2018.

Table of Contents

Executive Summary 9
Overall Piping Plover Monitoring Findings 9
Overall Red Fox Monitoring Findings 10
Restoration Areas 11
Vegetation Changes 11
Management Recommendations 12
Part 1: Background Information and Introduction 13
Study Area 16
Part 2: Monitoring of Piping Plover on Fire Island and Westhampton Island 24
Section 1: Field Methods 24
Habitat and Tide Mapping 24
Adult Survey 25
Nest Location 25
Egg Masses 26
Nest Checks 26
Trapping and Banding 27
Banding Schemes 27
Brood Observations 28
Invertebrate Sampling 29
Fecal Sampling 29
Substrate Invertebrate Sampling 30
Nest Cameras 31
Section 2: Classification Methods 32
Section 3: Compare changes in numbers, density, and distribution of piping plovers in various habitats on Fire Island 33
Introduction 33
Methods 34
Results 35
Density 42
Section 4: Estimate the “effect distance” of newly-created habitat 48
Introduction 48
Methods 49
Results 50
Discussion 57

Section 5: Compare egg weights of nests with access to bay side intertidal flats with egg weights of birds without such access 61
Introduction 61
Methods 61
Results 61
Discussion 62
Section 6: Compare piping plover nest and chick survival rates in various habitats across Fire Island 63
Introduction 63
Methods 64
Results 67
Discussion 80
Section 7: Compare the relative contribution of local growth (natal recruitment) vs immigration in population growth associated with new habitats 85
Introduction 85
Methods 86
Results 87
Discussion 93
Section 8: Compare prey densities at several habitat types 96
Introduction 96
Methods 97
Results 97
Discussion 99
Section 9: Quantify the interactions of predators with shorebird nests and chicks 100
Introduction 100
Methods 101
Results 101
Discussion 108
Part 3: Red Fox Movements, Habitat Selection, and Demography on Fire Island, New York 112
Section 1: Population ecology of red foxes 112
Introduction 112
Methods 113
Field methods 113
Lab methods 120
Analytical methods 121
Results 123

Discussion 140
Section 2: Spatial ecology of red foxes 143
Introduction 143
Methods 144
Results 149
Discussion 175
Section 3: Red fox response to piping plover nest exclosures and hatching 180
Introduction 180
Methods 181
Results 182
Discussion 186
Section 4: Dietary ecology of red foxes 188
Introduction 188
Methods 189
Results 192
Discussion 198
Section 5: Red Fox Summary of Findings and Management Recommendations 200
Summary of Key Findings 200
Direct impacts of red foxes and other predators on piping plovers 200
Indirect impacts via habitat use, spatial overlap and behavior of red foxes and piping plovers 202
Lethal predation management options 207
Literature Cited 208
Appendix A. Supplementary table for Part 2, Section 3: Compare changes in numbers, density, and distribution of piping plovers in various habitats on Fire Island in 'Effectiveness monitoring and targeted studies in support of FIMI conservation measures.' 223
Appendix B. Vegetation Report for 2013–2019 in Overwash and Restoration Areas at Smith Point County Park, Fire Island, NY 227
Appendix C. Supplementary tables for Part 2, Section 6: Compare piping plover nest and chick survival rates in various habitats across Fire Island in 'Effectiveness monitoring and targeted studies in support of FIMI conservation measures.' 287
Brood Home Range Figures 2014–2019 291
Appendix D. Supplementary tables for Part 2, Section 7: Compare the relative contribution of local growth (natal recruitment) vs immigration in population growth associated with new habitats 'Effectiveness monitoring and targeted studies in support of FIMI conservation measures.' 304
Appendix E. Supplementary material for Part 2, Section 8: Compare prey densities at several habitat types in 'Effectiveness monitoring and targeted studies in support of FIMI conservation measures.' 311

Appendix F. Supplementary material for Part 2, Section 8: Compare prey densities at several habitat types in 'Effectiveness monitoring and targeted studies in support of FIMI conservation measures.' 340

Appendix G. Supplementary maps for Part 3, Section 2: Red fox spatial ecology in 'Effectiveness monitoring and targeted studies in support of FIMI conservation measures.' 348

Appendix H. Supplementary material for Part 3, Section 4: Red fox dietary ecology in 'Effectiveness monitoring and targeted studies in support of FIMI conservation measures.' 352

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 2013–2019. Here, where appropriate and possible, we provide 2019-specific results in addition to cumulative results to facilitate the inspection of trends across the monitoring period. In April–August of 2013–2019, we conducted field monitoring of piping plovers (*Charadrius melodus*). Red fox (*Vulpes vulpes*) monitoring began in June 2015 and continued year-round until March 2019.

Overall Piping Plover Monitoring Findings

- During 2013–2019, the piping plover population on Fire Island and Westhampton Island increased 186%. Piping plover nesting density increased nearly 4-fold since 2013, and as of 2019 was 0.24 pairs/ha of dry sand potential nesting habitat, although bay-adjacent sites had higher nesting densities.
- Backshore and barrier interior habitats supported the most pairs, followed by overwash habitats.
- Nest failure was primarily due to predation (15%) and abandonment (13%).
- Piping plover adults and fledglings traveled relatively long distances from nest and hatch locations to forage on mud flats, sand bars, flood shoals, and Hurricane Sandy overwashes. Average adult distance moved from nest location varied by biological stage (nesting, brooding, non-breeding) but there was an increasing trend in average distances fledglings were seen from their hatch location from 2014–2019 (range 1.9–4.3 km).
- Nests in bay-adjacent subsites did not have higher clutch masses than those not bay-adjacent, suggesting that adult birds were able to find adequate nutrition, regardless of nesting juxtaposition to high-quality foraging habitat.
- Nest survival was highest for exclosed nests and was positively correlated with nesting plover density and temperature, but decreased with nest age.
- Nest survival was lower prior to the mange outbreak in the fox population and increased when mange removed most or all of the foxes.
- Chick survival was positively related to earlier hatching nests and brood age but negatively related to nesting plover density and red fox trap success.
- Reproductive output was positively correlated with study area density, suggesting that carrying capacity has not yet been reached in this population.
- The population primarily comprised returning adult nesters and immigrants in each year, with immigration rates between 0.52 and 0.75 (# immigrants in t+1 per individual in year t). Natal recruits comprised the smallest proportion of the population annually.

- Returning adults generally selected for habitat created during and following Hurricane Sandy and avoided pre-Hurricane Sandy habitats. Natal recruits selected for restoration areas in three of the five years since their creation, whereas immigrants avoided pre-Hurricane Sandy habitats.
- Hatch-year and chick survival increased throughout the monitoring period. Post-fledging survival was variable but relatively high compared to past piping plover studies.
- Predicted terrestrial invertebrate abundance was greatest in overwash habitats, ephemeral pools, and bay beaches.
- When assessed by class, avian species were the most prevalent potential nest disturbance detected around exclosures. However, at the species level, humans, gull species, unknown birds, and white-tailed deer were the most frequently detected species around nest exclosures.

Overall Red Fox Monitoring Findings

- Red foxes were important predators of piping plovers on Fire Island during our monitoring activities from 2015–2019. Their population dynamics, spatial ecology, and impacts on piping plovers were driven by high mortality caused by sarcoptic mange infections.
- Piping plover chick survival and overall reproductive output were negatively related to red fox trap success, such that plover chick survival and reproductive output were lower before versus after the mange die-offs and when fox trap success was highest. No relationship was found between feral cat or raccoon trap success and plover reproductive metrics.
- Red fox scat most frequently contained remains of mammals (75%), vegetation (73%) and insects (55%). Red fox also used anthropogenic food sources (15% of scat samples and 63% of den locations had anthropogenic food remains), even in the most remote in areas in the Fire Island National Seashore Wilderness Area. Charadriiformes feathers were found in 3% of red fox prey samples. No piping plover bands were found in 297 fox scats, contrary to the identification of piping plover bands in 1 of 3 feral cat scat samples.
- Sarcoptic mange was the largest source of mortality for all marked red foxes and drove dramatic population declines and reproductive failure of foxes across our monitoring period. Metrics of abundance and reproduction quantified the collapse of red fox populations across the monitoring area and temporary extirpation of red fox in 2016 at Smith Point County Park.
- Whereas indices of fox activity, fox abundance, fox density, and fox reproductive output all declined dramatically pre- to post-mange, red fox occupancy (i.e., probability of presence accounting for detection) remained high except during the window of fox extirpation in areas east of Old Inlet. These

results, along with the habitat selection and home range behaviors of foxes, indicate that predation risk from foxes for piping plovers persists even at very low red fox numbers.

- Male red foxes dispersed in the fall and transient, non-breeding and non-territorial males were present across the years and monitoring areas. Both dispersing and non-breeding males had very large home ranges, frequently moving between Old Inlet and both Moriches and Fire Island Inlets. However, no marked foxes were observed to cross Old Inlet.
- Foxes on Fire Island appear to be selecting areas closer to vegetation and wet sand edges, potentially including bayside marsh habitats, for daytime resting periods, while selecting more open habitats for nighttime periods of higher activity and movement.
- Red foxes did not significantly change their movement patterns following exclosure setup, piping plover egg pipping, or piping plover chick hatching.

Restoration Areas

- Piping plovers used both restoration areas immediately following construction, but their use of New Made ceased following the 2017 breeding season, likely as a result of vegetation encroachment. Use of Great Gun continued to increase.
- New Made quickly vegetated with large patches of phragmites, with vegetation reestablishing at twice the rate of Great Gun. Both areas had high fledge success (0.55 and 0.88 for Great Gun and New Made respectively) but Great Gun supported over 5x as many pairs, and fledged more than 6x as many young as New Made.
- Within areas selected for home ranges by foxes, predicted suitability was generally higher for males and for both sexes in New Made (6.43 out of 10, 10 being highest suitability) compared to Great Gun (5.80 out of 10) due to differences in vegetation and open sand between the two restoration areas.
- Five years following the creation of the restoration areas as mitigation for the FIMI, Great Gun was the more successful restoration area in terms of plover use, reproductive success, and vegetation reestablishment.

Vegetation Changes

- Vegetation cover (percent of pixels classified as vegetation using 15-cm resolution multispectral imagery) in overwashes and restoration areas ranged from 2% in 2013 to 88% in 2019.
- Vegetation cover in 2019 was lowest in the two areas where deep removal took place (Narrow Bay Breach and Great Gun)

- The rate of regrowth in areas not subject to deep vegetation removal suggests an active life of natural overwashes as plover habitat of ≤ 7 years (growing seasons).

Management Recommendations

- Balancing island stabilization with natural overwashing processes will benefit plovers in this system by increasing available nesting habitat, reducing the need for localized vegetation removal, and increasing access to abundant foraging resources.
- In the absence of natural overwash, vegetation management, as prescribed in the project description and the Biological Opinion, should be used to maintain early-successional states needed by plovers. Vegetation removal also will make areas less suitable for day-resting red foxes and reduce overall concealment ability of all mammalian predators in plover nesting areas.
- Nest exclosures should continue to be used unless predation dynamics change dramatically, and then altered only after careful consideration, as they were demonstrated to be the strongest predictor of plover nest success.
- If the red fox population remains at a low density as experienced post-mange, the local piping plover population will likely experience higher reproductive output and grow more quickly until carrying capacity is reached than if red fox densities increase rapidly to pre-mange levels, but probably will not be able to exceed the food-based carrying capacity.
- The observed dispersal and transient behavior of male red foxes suggests a high capacity for dispersal and recolonization should an occupied territory become available. These transient animals regularly moved between management areas, with most transient animals traveling the entire length of the island available to them. Thus, predation management, a conservation measure in the Biological Opinion, must consider this recolonization potential and be sustained in its deployment. This also suggests a need for continued coordination among land managers on Fire Island, as coordinated by the Corps of Engineers.
- Launching a coordinated campaign to limit feeding of wildlife and to most efficiently manage trash from recreationists probably could help keep red fox population numbers low, especially if implemented before the fox population rebounds fully from the mange-related extirpations.
- Feeding and housing feral cats should be prohibited in all management areas in order to minimize population growth of non-native plover predators.
- Future restoration efforts should use Great Gun as a starting point, with adaptive changes relative to the other findings in this report. For example, creating access to invertebrate-rich bayside habitats might increase plover density.

Part 1: 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 U.S. Fish and Wildlife Service (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 report to provide the best overview of conditions of the piping plover population, the habitat that supports it, and the red fox predator population and its interactions with piping plovers. The U.S. Army Corps of Engineers (USACE) contributions to this project until June of 2015 were made under the Breach Contingency Plan and were restricted to plover studies in the Corps Project Areas. The Resiliency project funding was intended to expand the plover project to include areas of Smith Point County Park in 2013–2016.

In 2012, Hurricane Sandy made landfall on the barrier islands of New York and New Jersey and created multiple breaches and overwashes. Three breaches were created within 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.

The breaches in the county parks were filled soon after the hurricane in accordance with the BCP. The FIMI project entailed sections of beach and dune construction along the 30.5 km of Fire Island and was initiated by USACE in the winter of 2014. The USACE formally consulted with the USFWS pursuant to the Endangered Species Act of 1973, as amended, due to the potential direct and indirect adverse impacts of these projects on the federally listed species. The USFWS's Biological Opinion (BO) for the BCP covered the piping plover (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, 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 and management, habitat maintenance and restoration, predator management, and reduction of human disturbance. Vegetation was cleared and the topography was modified by USACE in approximately 33.7 ha (83.2 ac) of ocean-side habitat at Great Gun restoration area to mitigate adverse effects of dune and beach construction. To create possible nesting and

foraging habitat for piping plovers, USACE also created New Made restoration area, 6 ha (15.8 ac) of bay-side habitat south of Carter's Island and Moriches Bay.

Breaches and overwashes can result in creation of new foraging and nesting habitat for piping plover (Cohen et al. 2009, Robinson et al. 2019), but the loss and manipulation of them by dunes built under the FIMI had the potential to disrupt natural plover breeding processes. Thus, the effect of the habitat modifications under the FIMI needed to be monitored and evaluated. Understanding quality of nesting and foraging habitats can lead to a better understanding of the plover response to Hurricane Sandy and subsequent natural and man-made habitat alterations. The response to habitat change can manifest in changes in demographic rates, growth rates, density, movement, prey abundance, and a variety of other rates (Loegering and Fraser 1995, Elias et al. 2000, Cohen et al. 2009, Catlin et al. 2014, Catlin et al. 2015).

However, plover populations often are affected by predators (Hunt et al. 2019*a*). Understanding how the local predator community responds to storm- and anthropogenic habitat alterations, and how the predators are affecting piping plovers during the same period, is necessary to gain the full picture of the effects of habitat change. Red foxes (*Vulpes vulpes*) are a predator of piping plovers on Fire Island (U.S. Fish and Wildlife Service 2014), and in other barrier island ecosystems (Rimmer and Deblinger 1990, Patterson et al. 1991, Cohen et al. 2010, Gieder 2015). Different aspects of red fox population ecology have been studied in a variety of ecosystems (see Lloyd 1980, Ables 1983 for overviews) but changes in occupancy (i.e., presence accounting for detection) have been studied only twice on the U.S. Atlantic Coast, once on Cape Cod, MA and once on Assateague Island, MD (Gieder et al. 2015, O'Connell et al. 2006). No previous monitoring has specifically examined red fox population responses to storm- and anthropogenic habitat alterations nor detailed fox population ecology in a barrier island ecosystem. Land managers frequently undertake both non-lethal (e.g. exclosures) and lethal (e.g. targeted or systematic removal) management of red foxes along the U.S. Atlantic Coast to benefit piping plovers and other imperiled shorebirds and sea turtles (Hunt et al. 2019*a*). Rarely are the efficiency and efficacy of predation management efforts assessed. There is a growing recognition by managers and the public on the need to monitor and quantitatively assess the success of predation management actions (Hunt et al. 2019*a*). Thus, the red fox monitoring in this report quantifies occupancy, home range dynamics, numbers, vital rates, habitat selection, movement ecology, and diet of red foxes in plover habitats to best guide predator management efforts.

This report seeks to summarize targeted monitoring programs created to evaluate the efficacy of the BO conservation measures. The monitoring effort seeks to provide a scientific evaluation of the effectiveness of the management and habitat maintenance and restoration efforts within the FIMI project area and Fire Island, as a whole. Both Virginia Tech and the USFWS worked closely with an interagency team consisting of the National

Park Service (NPS), New York State Office of Parks, Recreation, and Historic Preservation, Suffolk County Parks, and the USACE to obtain the necessary permits to undertake this work.

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

The 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 report of red 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 Area

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; Figure 1).

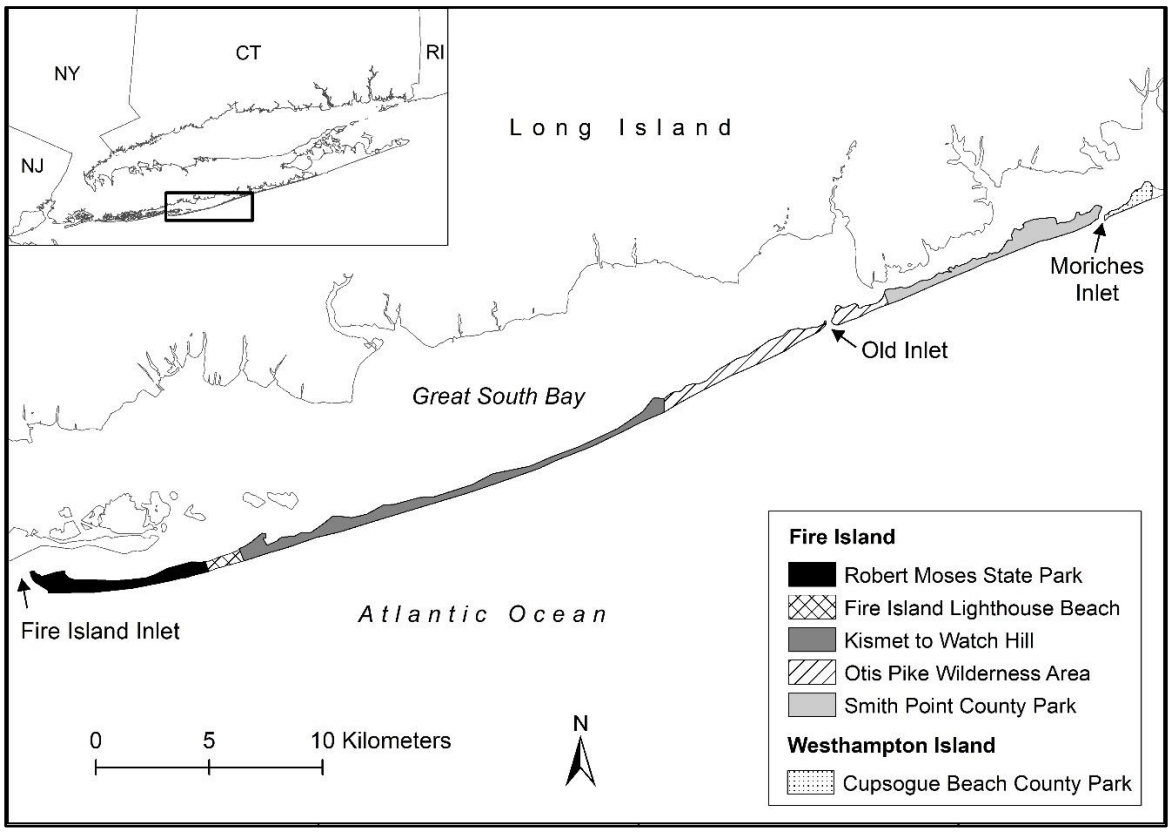


Figure 1. Map of the study area, Fire Island and Westhampton Island, NY comprising different management units.

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 restoration area (hereafter, 'New Made').

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

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

Habitat	Definition
Restoration Area	New Made and Great Gun restoration areas, created between the 2014 and 2015 piping plover breeding seasons.
Overwash	Any area that overwashed during Hurricane Sandy. This is a broad category that is subdivided into manipulated or natural overwash.
Manipulated Overwash	Areas of Fire Island that overwashed during Hurricane Sandy, but were manipulated through dune manipulation (including snow fencing and Christmas tree placement), or vegetation management.
Natural Overwash	Areas of Fire Island that overwashed during Hurricane Sandy, but have not been further modified.
Breach Fill	The areas of Fire Island and Cupsogue Beach County Park that breached during Hurricane Sandy, but were filled shortly after by USACE. Both breach fills fall adjacent to manipulated overwashes.
Manipulated Dune	Any dune or berm created by other than natural sources. This includes snow fencing, Christmas trees, planting, and USACE/other engineering.
Management Area	Areas defined by USACE to manage under the FIMI.
Nourishment	Beach widening by the USACE in Smith Point County Park from 2014–2015 and 2018-2019.
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.
Backshore	The dry sand area between the high-water line and the toe of the dune.
Barrier Interior	The area inland of the toe of the dune and interior of Fire Island.
Swale	A depression or blow-out behind the primary dunes, between ridges; either vegetated or unvegetated.
Beach	The 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.
Dredge Spoil	Stockpiles and graded areas of dredged sand.
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.
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.
Flood Shoal	A submerged ridge or bar associated with inlets consisting of or covered by sand; rising from the sea bed to near the surface so that it is tidally exposed. They are often not static and may move seasonally and annually.

We divided the site into smaller subsites intended to be relatively homogeneous in geomorphological and ecological structure (e.g., areas where overwash occurred were distinct from highly vegetated tall dunes, 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.

Throughout the plover breeding seasons, all beaches were open to pedestrians. Smith Point County Park and Robert Moses State Park also allowed recreation vehicles to drive onto the beaches. Every year, these beaches were closed to vehicles once chicks hatched and were re-opened once chicks fledged (Table 2). However, Smith Point County Park in 2013, and subsite OSS in Robert Moses State Park - South in 2017–2019, remained open to vehicle access following piping plover hatch dates. Official dates of closures for 2013 are uncertain. Despite vehicle closure to the general public, law enforcement, park staff and plover monitors were still permitted to drive through closed areas, depending on the circumstance.

Table 2. Beach closure dates for vehicle use at Smith Point County Park and Robert Moses State Park 2013–2019

Management Unit	2013	2014	2015	2016	2017	2018	2019
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	23 May– 9 Aug
Robert Moses State Park - North	-	-	06 Jun–04 Aug	06 Jul–22 Jul	07 Jun–16 Jul	02 Jun–03 Aug	06 Jun–13 Jul
Robert Moses State Park - South	-	-	10 Jul–04 Aug	25 Jun–05 Aug	24 May–10 Aug	29 Jun–03 Aug	04 Apr–end of season

Table 3**.** Subsite names, location descriptions, and habitat at Fire Island and Westhampton Island, 2019.

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

Bridge West	BRW	Stretch of backshore with primary dune line between Bridge Overwash and Old Inlet East.	Beach, Backshore, and Interior
Bridge Overwash	BRO	Overwash in the Otis Pike Wilderness, just west of the NPS Visitor Center.	Beach, Natural Overwash, and Interior
Bridge East	BRE	Begins where the primary dune line beings 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 dune line between Narrow Bay West and Pattersquash overwash.	Beach, Backshore, and Interior
Narrow Bay West	NBW	Includes the overwash and small breach-fill west of the Hazardous Area.	Beach, Manipulated Overwash, Interior, Manipulated Dunes, and Breach Fill
New Made	NMA	Begins at the east end of the filled dredge spoil and extends west to the east edge of the small breach-fill.	Beach, Backshore, Manipulated Dune, Interior, and Restoration Area
Narrow Bay East	NBE	Begins at the westernmost end of the large dune and extends to the westernmost end of the Hazardous Area.	Beach, Manipulated Overwash, Interior, and Manipulated Dunes
Smith Point West	SPTW	Begins at Cut 3 on Burma Road and extends to the end of the large dune and beginning of the Narrow Bay East manipulated overwash area.	Beach, Backshore, and Interior
Smith Point East	SPTE	Begins on west side of Moriches Inlet and extends west to Cut 3 on Burma Road.	Beach, Backshore, Manipulated Dune, Interior, and Restoration Area
Cupsogue West	CSW	Breach-fill and Moriches Inlet area of the Cupsogue Beach County Park. Begins at the west edge of the breach-fill and extends west to Moriches Inlet.	Beach, Manipulated Overwash, Manipulated Dunes, Interior, and Breach Fill
Cupsogue Sandbar	CSB	Large sandbar/shoal north of Cupsogue West.	Sand Bar
Cupsogue East	CSE	Beach and narrow strip of bayside intertidal zone at east end of the Cupsogue Beach County Park. Begins at the west end of the parking lot and extends west to the east edge of the breach-fill.	Beach, Backshore, Manipulated Dune, and Interior
Cupsogue Mud Flats	CMF	Mud flats north of the Cupsogue parking lot, and south of the high bare Island in Moriches Bay.	Mud Flat

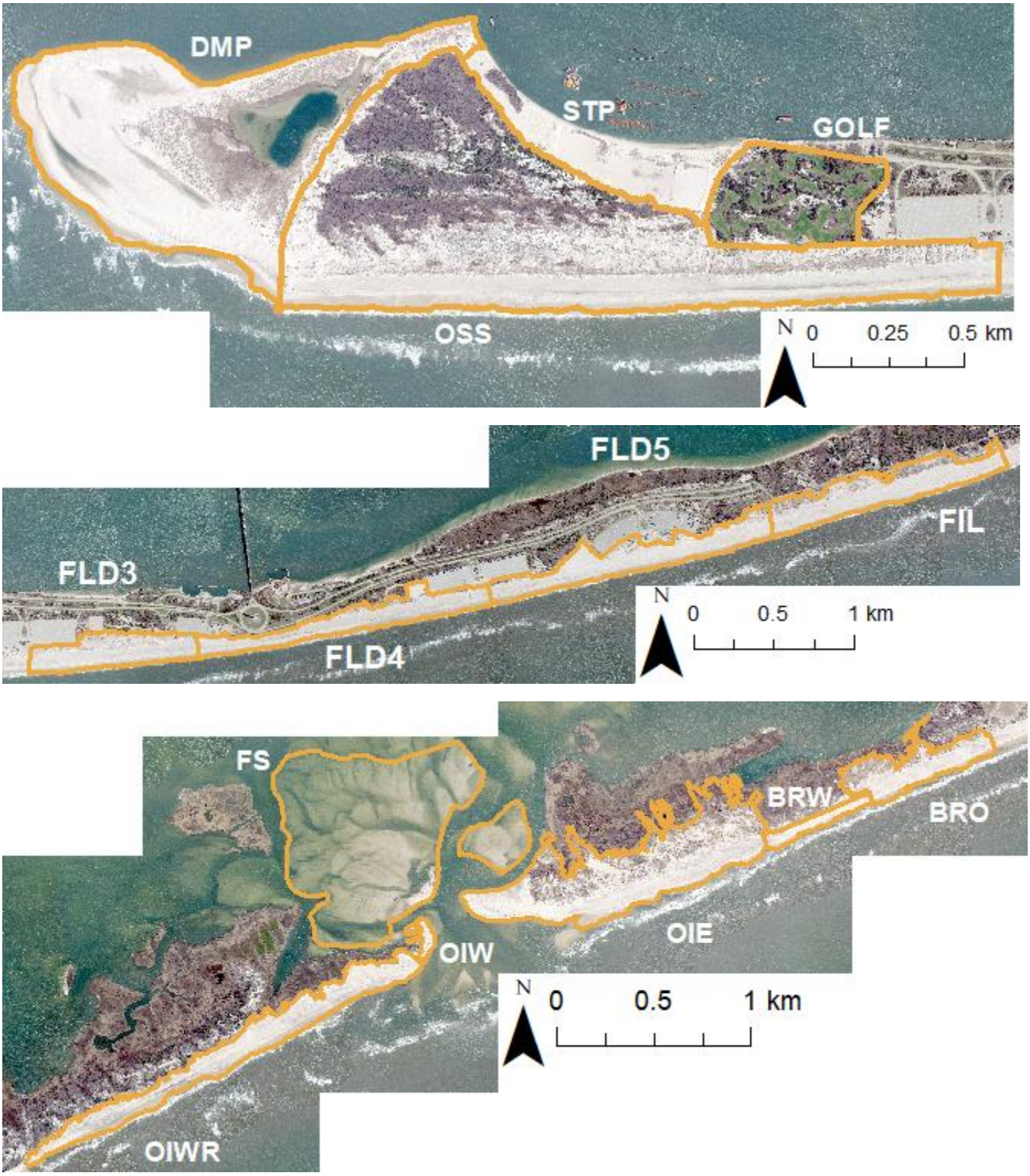


Figure 2. Study area at Robert Moses State Park (top and middle), Fire Island Lighthouse Tract (middle), and Fire Island National Seashore (bottom). Subsite boundaries are outlined in orange. See Table 3 for subsite names and descriptions. Imagery captured April, 2019.

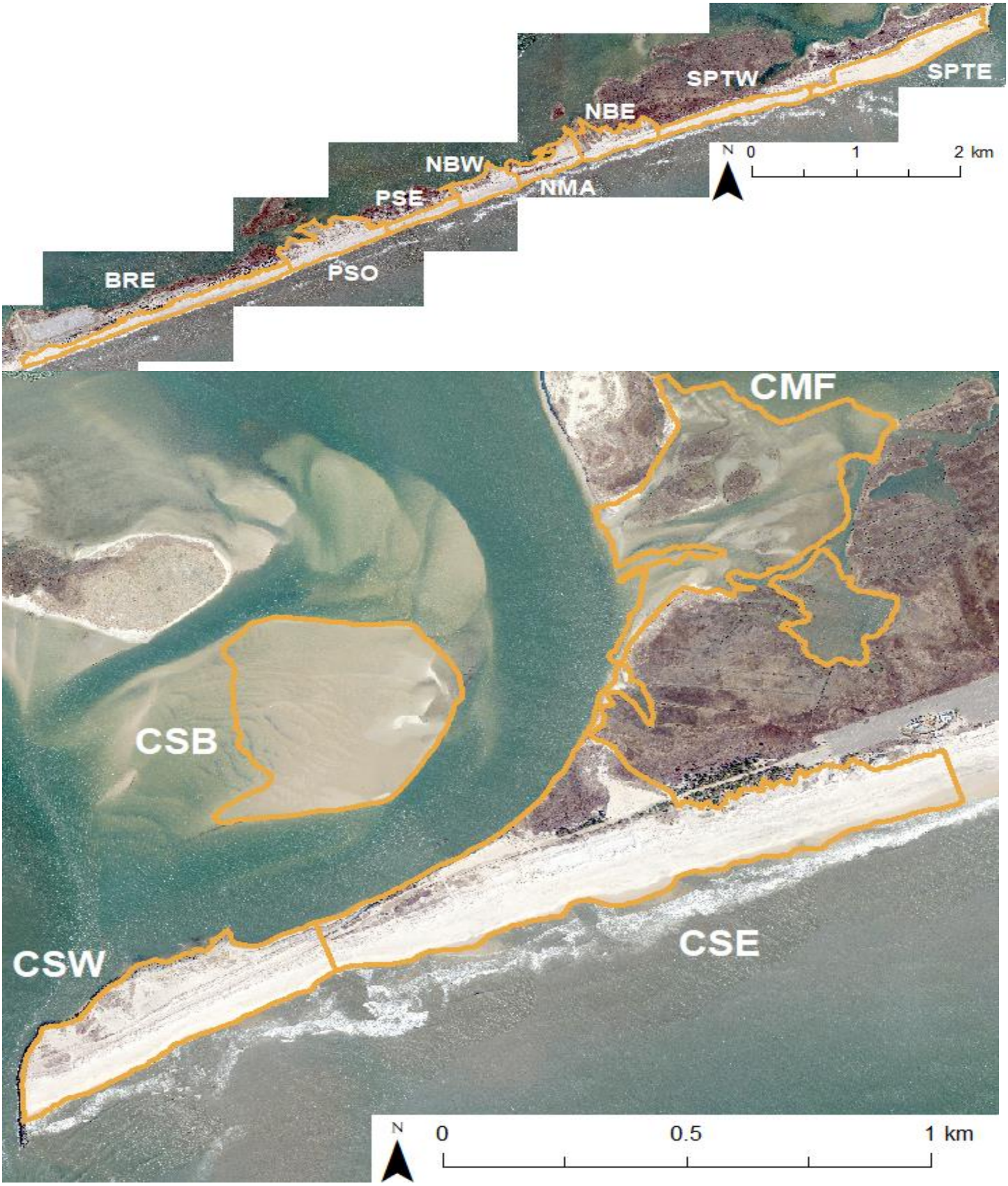


Figure 3. Study area at Smith Point County Park (top) and Cupsogue Beach County Park (bottom). Subsite boundaries are outlined in orange. See Table 3 for subsite names and descriptions. Imagery captured April, 2019.

Part 2: Monitoring of Piping Plover on Fire Island and Westhampton Island

Section 1: Field Methods

Habitat and Tide Mapping

In 2013 and 2014, we collected all mapping data with a Garmin 78 (Garmin International, Olathe, KS). We used a Trimble Geo7x (Trimble Navigation, Ltd., Sunnyvale, California, USA) in 2015–2019 to map the shoreline during spring high and low tides along the bay and ocean beach (Figure 3). Tides were mapped at the shoreline throughout the entire season, although no low tide data was collected in 2014. In 2015, we mapped spring high tide and low tides during mid-May and early June, and mapped various habitats and subsite boundaries during late July and early August. In 2016, we mapped shorelines at spring high tide and low tides May–September, and completed snow and symbolic fencing mapping during late-July and August. In 2017, we mapped the shorelines during spring high and low tides in May, and the snow and symbolic fencing during late June–August. In 2018, we mapped shorelines at spring high and low tides in May and June, and mapped snow and symbolic fencing May–August. In 2019, we mapped shorelines at spring high and low tides in May and June, and mapped snow and symbolic fencing April–July.



Figure 4. Using a Trimble Geo7x to map the spring tides by walking along the high-water line.

Adult Survey

We surveyed each subsite in the study area for adult piping plovers every 1–3 days, April–August, 2013–2019. We surveyed subsites on foot and by UTV. Some subsites that did not have bayside habitat and consisted only of narrow ocean beach were surveyed by driving a UTV slowly along the ocean beach and along the backshore. UTVs were only driven outside of areas protected by symbolic fencing established by land managers. At least twice per week during the nesting period (April–July), we 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, facing either east or west. 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 2013–2015, we recorded the location of individuals with a Garmin GPS unit. In 2016–2019, we used a Nikon laser rangefinder and a Trimble GPS unit to record the location of plovers (Robinson et al. *in press)*.



Figure 5. Surveying for piping plovers using spotting scopes inside the Great Gun restoration area at SPTE.

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 Garmin GPS unit (2013–2014) or a Trimble GPS (2015–2019, Figure 5a) unit. If a nest was found at full clutch, we floated all four eggs to estimate the initiation and hatch date (Alberico 1995; Figure 5b).

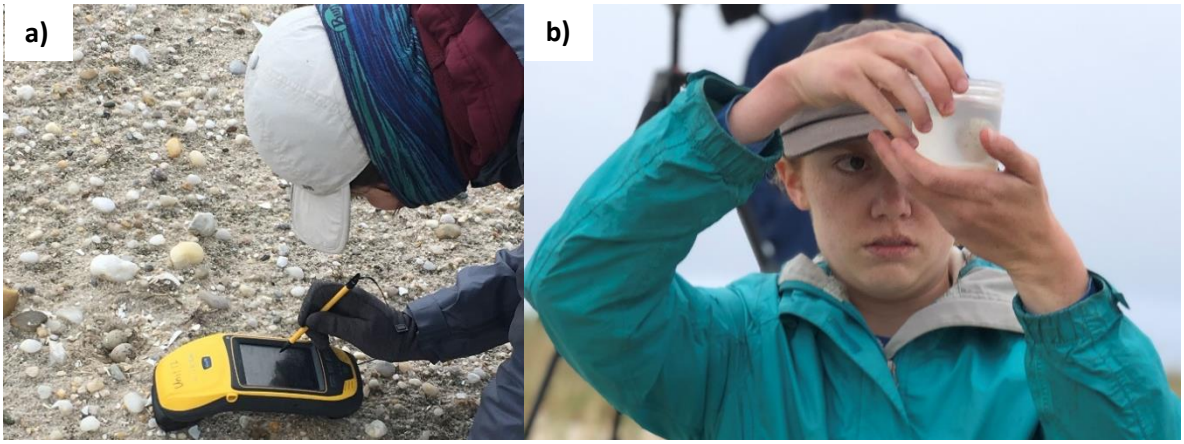


Figure 6. a) Using a Trimble Geo7x to map the location of a piping plover nest. b) Floating an egg from a four-egg nest in SPTE.

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).

Nest Checks

We checked nests every 1–3 days throughout the incubation period (Figure 6). As the hatching date approached, nests were checked for signs of hatch approximately every day. Nest fate was determined from observations of the nest and surrounding area.

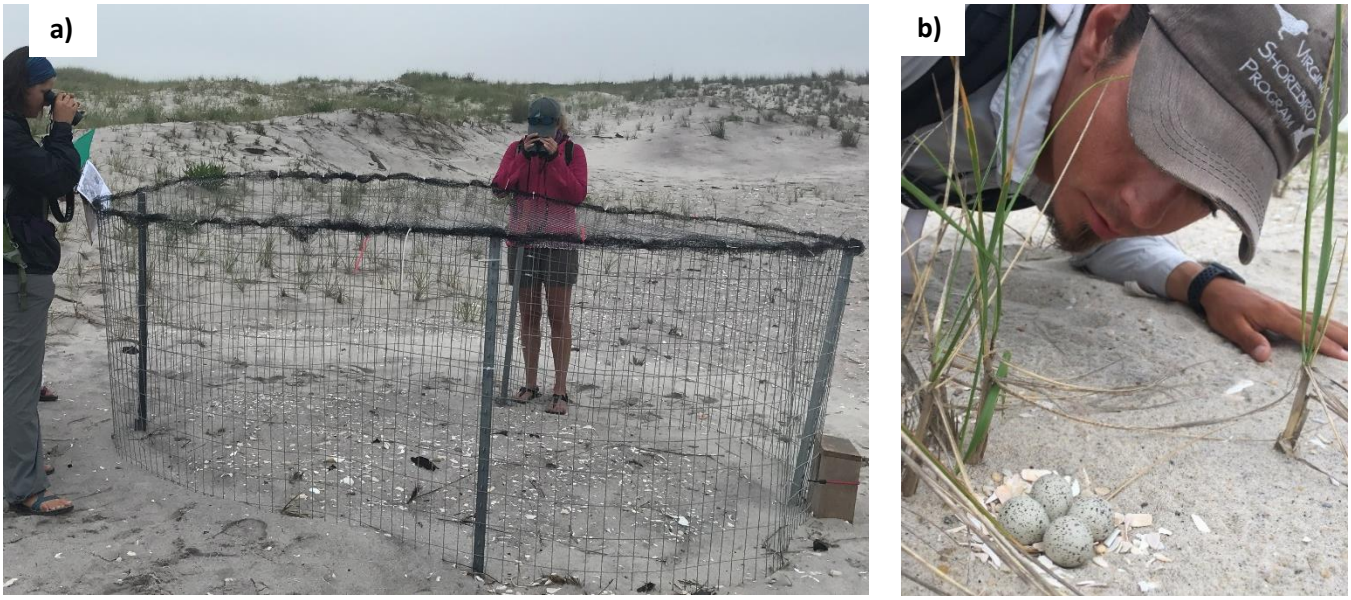


Figure 7. a) Checking an exclosed nest for hatch using binoculars. b) Checking an unexclosed nest for hatch.

Trapping and Banding

We attempted to trap banded and unbanded adults during the incubation period, over a full clutch, using a drop-box trap or shoo-in funnel trap. Once captured, plovers were removed from the trap, banded, weighed, and measured (Figure 7a). If the second adult of a particular nest had not yet been trapped that year, we reset the trap while processing the first adult. We attempted to capture banded plovers to assess the condition of bands, flags, and leg health. If no adults entered a trap within 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 7b) and recaptured them at a minimum of 20 days post-hatch to weigh and measure them a second time.

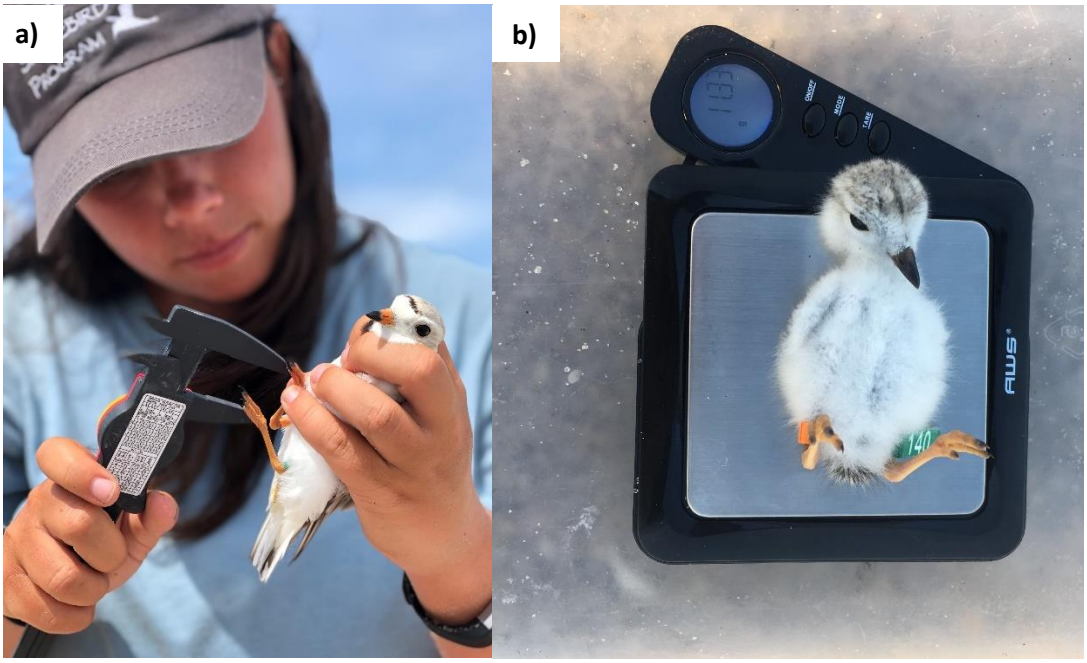


Figure 8. a) Measuring the tarsus of an adult piping plover. b) Taking the mass of a 10-day old chick.

Banding Schemes

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, 2014). In 2014–2019, 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 8).

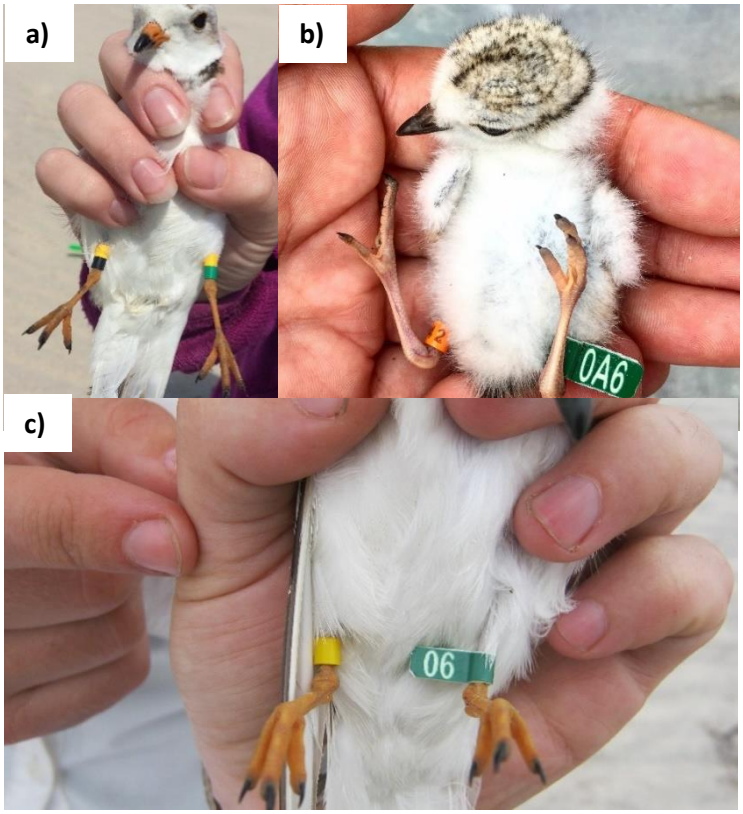


Figure 9. a) A piping plover adult with two color bands on each upper leg, b) a chick banded with a green alphanumeric flag and an orange color band, and c) an adult with a green alphanumeric flag and a yellow color band.

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 three days. We attempted to collect a behavioral sample on at least one randomly selected chick in each brood during every visit. To determine the start time of the behavioral sample after locating a brood and to randomly choose a chick, we used a random number generator. For five minutes, we recorded forage rate (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 another chick and conducted a behavioral sample at the new location.

Invertebrate Sampling

We sampled terrestrial invertebrates from 2014–2019 in various habitat types (backshore, overwash, breach fill, restoration area, beach) from the time that the first chicks hatched until the time the last chicks fledged. We collected four samples during each invertebrate sampling set. Two of these samples were taken on the beach and two were taken at randomly selected locations within the subsite. For the beach samples, we followed a transect parallel to the beach just above the intertidal zone. Using a Garmin GPS unit, we navigated to the last data point collected on the beach during the previous sampling set in the subsite. From there, we continued along the beach a randomly generated number of meters, from 1–50, to arrive at the location of the first sample in our current sampling set. We walked 50 m along the beach for the subsequent sample. The third and fourth samples for the set were chosen from a predetermined, random selection of points within 100 m of the previous sample. We sampled invertebrates using two paint stirrers coated with Tanglefoot Tree Insect Barrier (Tanglefoot Company, Grand Rapids, Michigan). One paint stirrer was coated on one side with Tanglefoot and placed flat on the sand, Tanglefoot side up. The second paint stirrer was coated on both sides with Tanglefoot and positioned vertically in the sand. The sticky area on the horizontal paint stirrer was 64.5 cm^2 and 129 cm^2 on the vertical. We placed a cage over the trapping station to prevent any birds from getting stuck in the Tanglefoot (Figure 9). We returned to each sample location after 30 minutes and counted, measured (±1 mm), and identified any arthropods stuck on the paint stirrers to order.



Figure 10**.** Invertebrate trapping station with a vertical and horizontal paint stirrer coated in Tanglefoot placed in a wire cage.

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 sand as possible, and placed in vials using tweezers (Figure 10). We collected the location of the sample in the Trimble Geo7x. After every

collection, the tweezers were wiped off with ethanol wipes before being used to collect another fecal sample. Vials were kept in pouches in coolers while in the field and then stored in a freezer until the end of the season. 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 in 2016 and 2017 were transported to contractors at Jonah Ventures in Boulder, Colorado for meta-barcoding DNA analysis. Seven fecal samples were collected in 2013, and 37 in 2014. We did not collect fecal samples in 2015. Thirty-nine samples were collected across the study site in 2016, and 34 were collected in 2017. No fecal samples were collected 2018–2019.



Figure 11**.** Collecting a fecal sample using tweezers to reduce sample contamination and marking the location with a Trimble Geo7x.

Substrate Invertebrate Sampling

To assess to availability of marine invertebrate prey and compare among habitat types, we sampled marine invertebrates on transects in all flood shoal, mud flat, and ocean-to-bay overwash intertidal habitats. In 2013, we established ten evenly spaced sampling stations in the intertidal zone at every subsite with bay or inlet intertidal habitat (Old Inlet West, both flood shoals, Old Inlet East, Bridge Overwash, Pattersquash Overwash, Narrow Bay West, Cupsogue West, and Cupsogue East). In 2015, we established 33 sampling stations in the intertidal zone at every subsite with such habitats (Flood Shoals, Cupsogue Sand Bar, Cupsogue Mud Flats, Cupsogue West, Narrow Bay West, and Pattersquash Overwash). We determined the location of the first sampling station by pacing a random number of steps from the east end of the subsite, and set each subsequent station at 10-m intervals. At each station we collected a substrate prey core (7.6 cm diameter, 10 cm deep) and sediment core (2.6 cm diameter, 10 cm deep). All samples were collected within an hour of low tide. We sieved samples with a 0.5 mm mesh sieve to remove fine grain sand, and preserved the remainder of the sample in 10%

formalin with Rose Bengal. We transported samples to a lab at the Marine Resources Institute at the Department of Natural resources in South Carolina to identify and count organisms.

Nest Cameras

In 2016–2019, we placed remote cameras (Reconyx Rapidfire, RC60 Covert IR, Holmen, WI, USA) at nests to detect anything that visited the nest exclosure. 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 a 5 second 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 exclosure, and chose an area that did not appear to be heavily used by adults entering or exiting the exclosure. When installing the cameras, our first priority was to place the boxes in low-traffic area to reduce camera-caused disturbance. Our second priority was to face the boxes north or south to avoid having them face directly towards the sun, preventing overexposure. We placed the camera box against the outside of the exclosure and aimed the camera toward the nest. We attached the camera box to the exclosure with a bungee cord to secure it in place (Figure 11). We performed any maintenance needed to cameras, such as replacing memory cards and batteries, while we conducted daily nest checks to reduce disturbance. The cameras were removed when the nest failed or the brood permanently left the exclosure area.



Figure 12. A remote nest camera installed outside of a nest exclosure and potentially capturing photos of a snowy owl.

Section 2: Classification Methods

A raster of classified habitat types was created from annual high-resolution (15-cm) aerial imagery with 4 bands (R, G, B, NIR), using ArcGIS Desktop 10.6 Interactive Supervised Classification (Maximum Likelihood Classification). Training samples were selected for dry sand, vegetation, wet sand, and water. Interactive Supervised Classification is an iterative process which allows refinement of the classification results. Even with this, there was a great deal of overlap in band reflectance of vegetation, water, wet sand, dark sand, and shadows. The human brain is adept at perceiving whether a pixel classified as water is actually the shadow of a tree but numerical models, as yet, are not capable of making those distinctions. In order to use these habitat classifications for modeling, various methods in ArcGIS were used to reclassify pixels which were misclassified in the initial classification of the whole island, for example, pixels classified as vegetation on the beachfront or pixels classified as wet sand or water in dry areas of the island interior.

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

In addition to correcting and masking misclassified pixels, a raster of water and intertidal sand was classified from DEM (digital elevation model) data on the ocean side of the island and bayside overwashes. The oceanfront intertidal range was 0–4.11 ft in elevation, based on the Astronomical High Water at NOAA Moriches Inlet tidal station. The bayside intertidal range was 0–1.46 ft in elevation, based on mean high water at NOAA Smith Point Bridge tidal station. We reclassified each DEM to select pixels within both the oceanfront and bayside intertidal and subtidal range. These pixels were reclassified with numeric values to represent intertidal wet sand and water, respectively. The refined layers from each of these methods were superimposed over the original maximum likelihood classification, masking the misclassified pixels.

Section 3: Compare changes in numbers, density, and distribution of piping plovers in various habitats on Fire Island

Introduction

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

Habitat changes resulting from anthropogenic and natural forces affect piping plover population dynamics (Cohen et al. 2009, Catlin et al. 2015). Many plover populations are regulated by density dependent forces on the breeding grounds (Cohen et al. 2009, Catlin et al. 2019*a*). There is evidence that adult survival, breeding propensity, nest survival, and chick survival are reduced when nesting densities are high (Weithman et al. 2017, Hunt et al. 2018*a*). Lower densities may reduce intra- and inter-specific competition among individuals for nesting and foraging space. Reduced competition for space and thus an increase in available nesting and foraging territories may lead to a population increase in density-dependent species, and has been observed in piping plover on several occasions (e.g., Wilcox 1959, Cohen et al. 2009, Catlin et al. 2015, Hunt et al. 2018*a*, Robinson et al. 2019). Given the increase in habitat following Hurricane Sandy, a similar process was predicted to unfold in the piping plover population on Fire Island and Westhampton Island.

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

Methods

Using color bands and field-readable, alphanumeric ID flags, we banded adults and chicks from 2013–2019. Our goal was to capture every unbanded plover each year to help us identify individuals for a variety of analyses. In particular, if we could capture the majority of unbanded adults and chicks in each year, we could understand the contribution of immigration to our study area from each origin class (e.g. natal recruits, immigrants).

We compared the number of pairs that used various habitats during 2013–2019, including backshore, restoration areas, natural overwashes, and manipulated overwashes to determine if plovers were using any of these newly created habitats more often than habitats that were available before Hurricane Sandy (See Table 1 for habitat definitions). Using the GPS locations collected at every nest, we assigned nests a discrete habitat type and compared the proportion of nests that occurred in each of those habitat types. We assumed that if a male renested in the same subsite or habitat type, the location of the renest was not independent of the first nest. We only included renests of stable pairs (same male and female) in the habitat-specific pair counts if their new nest was in a different subsite or different habitat class.

We estimated annual pair numbers using the males within pairs. We used males for pair counts and productivity estimates because when pairs separate, the males often keep their territory when they partner with a new female. Additionally, males are the primary territory holders and defenders (Haig and Oring 1988). Using banded adult information and nest date overlap, we used the highest number of male territories per management unit each breeding season as an estimate of the number of pairs in that unit. To account for pairs that included unbanded individuals, we used the initiation and failure dates to determine if the pairs were independent from individuals banded later in the season. We used this estimate to avoid double counting any male that nested more than once in a management unit, resulting in a conservative pair estimate.

We monitored nests each year for success and to determine the cause of any nest failure. Nests were assigned a fate of successful if they hatched at least one chick and failed if they were abandoned, depredated, flooded, or suffered total egg loss that was not due to hatch. Any time an egg was missing during a routine nest check, we attempted to determine the cause of egg loss. In most cases, egg loss could be attributed to hatch and chicks were present to confirm the nest fate. In cases where no chicks were present, we assigned the nest a fate based on any sign we could find near the nest including mammal or avian tracks (predation), egg yolk (predation), sand erosion from high water (flooding), etc. If a nest disappeared with no discernible sign of fate, we assigned it an unknown fate. If a nest seemed to be abandoned, we looked for any sign that an adult was incubating the eggs for three or more consecutive visits over six days. After six days, if there was no sign of an adult tending the eggs, we assigned the nest as abandoned.

Nests also were given a habitat designation depending on their habitat type and geologic location on the island. The habitats assigned included swale, natural overwash, manipulated overwash, dune, manipulated dune, barrier interior, backshore, or restoration area (Table 6, Table 8). Nests and their associated breeding pairs were then regrouped into overwash, breach fill, restoration area, or backshore/barrier interior to compare how many pairs nested in newly created habitat versus previously available habitat (Table 9). Overwashes, breach fills, and restoration areas all represent new habitats created after Hurricane Sandy. The USACE provided polygons delineating restoration areas (USACE 2014), and we generated polygons to delineate all additional post-Hurricane Sandy habitats. Backshore/barrier interior was habitat available to plovers before the overwashes and breaches occurred in 2012. "Natural overwash" was only assigned to a nest if it was within Hurricane Sandy-created overwashes. Nests found in natural overwashes created following Hurricane Sandy were given a habitat designation other than "natural overwash".

We estimated nesting plover pair density by subsite, management unit, and total study area. Using our delineated subsites, extending from the ocean to the bay, we calculated the hectares of dry sand in each subsite (Part 2, Section 2). Using the Region Group tool in ArcGIS, we calculated size of each dry sand patch in our study area using an 8-cell neighbor rule. We then removed all patches of dry sand less than 1-m^2 of contiguous dry sand. We assessed density using the total number of pairs that nested in the study area, management unit, or subsite. Pair counts were obtained from Robert Moses and Fire Island National Seashore staff for 2013 and 2014 in Robert Moses State Park and the Fire Island Lighthouse. For subsite density, we grouped OSS and STP (Table 3) to allow for comparisons of island-cross sections. For management unit density, we grouped the Fire Island Lighthouse Tract with Robert Moses State Park, as no perceivable barrier to piping plover movement was detectable between the two management units. We compared the density between subsites that had direct bay access (DMP, OSS-STP, PSO, NBW, and CSW) and subsites that did not have direct bay access. We used the estimated densities to test for a relationship between reproductive output and density in Part 2, Section 3.

Results

Between 2013 and 2019, we banded 241 adult piping plovers (128 females and 113 males) and 765 chicks (Table 4)**.** All chicks were banded between 0 and 16 days old. In 2013 and 2014, eight nests were found at Watch Hill. Five adults and three chicks from those nests were banded. We did not monitor nests at Watch Hill from 2015 through 2019 due to logistical limitations, but these areas were continually monitored by the National Park Service. We did not band any birds at Robert Moses State Park or the Lighthouse Tract in 2013 and 2014 as we did not begin work at those sites until 2015. Of the 765 total chicks banded, 4.4% were banded in the 2013 season, 5.4% in 2014, 6.1% in 2015, 13.2% in 2016, 17% in 2017, 22.1% in 2018, and 31.8% in 2019.

From April to August 2019, we banded 48 adult piping plovers and 243 chicks (Table 5). We were unable to capture and band 10 chicks out of the 253 that were confirmed to have hatched: three chicks from a brood at Cupsogue Beach County Park, two from a brood in SPTE at Smith Point County Park, one from a brood in NBW at Smith Point County Park, one chick from one brood in DMP at Robert Moses State Park, and three chicks from another brood in DMP at Robert Moses State Park. We discovered the carcasses of two of these unbanded chicks and the remainder were presumed dead as they were not seen again (Part 2, Section 6). We were unable to confirm whether three eggs hatched. We recaptured 57 adult piping plovers that were banded during previous years. Of those recaptured, 34 of them were banded as chicks on Fire Island, 20 were banded as adults on Fire Island, two were banded as adults in New Jersey, and one was banded as a hatch year during southward migration at Cape Hatteras, North Carolina.

Table 4. Piping plover adults and chicks banded on Fire Island and Westhampton Island during 2013–2019 per management unit. Subsites within each management unit are abbreviated in parentheses. See Table 3 for detailed description of subsites.

	Adults		Chicks
Management Unit	Females	Males	
Cupsogue Beach County Park (CSE, CSW)	23	16	126
Smith Point County Park (BRE, PSO, PSE, NBW, NMA, NBE, SPTW, SPTE)	38	36	286
Fire Island National Seashore - east of breach (OIE, BRO)	17	17	138
Fire Island National Seashore - west of breach (OIW, OIWR)	4	6	42
Fire Island National Seashore - Lighthouse Tract (FIL)	4	4	11
Robert Moses State Park (FLD5, FLD4, FLD3, OSS, STP, DMP)[a]	40	31	159
Off-Site - Bellport to Watch Hill[b]	2	3	3
Total	128	113	765

[a] Birds were only banded from 2015–2019

Table 5. Piping plover adults and chicks banded on Fire Island and Westhampton Island during April–August 2019 per management unit. Subsites within each management unit are abbreviated in parentheses. See Table 3 for a detailed description of subsites.

Management Unit	Adults		Chicks
	Females	Males	
Cupsogue Beach County Park (CSE, CSW, CMF, CSB)	3	2	25
Smith Point County Park (BRE, PSO, PSE, NBW, NMA, NBE, SPTW, SPTE)	3	7	95
Fire Island National Seashore - east of breach (OIE, BRW, BRO)	2	5	34
Fire Island National Seashore - west of breach (OIW, OIWR)	2	2	24
Fire Island National Seashore - Lighthouse Tract (FIL)	3	3	11
Robert Moses State Park (FLD5, FLD4, FLD3, OSS, STP, DMP, GOLF)	7	9	54
Total	20	28	243

During 2013–2019, we located and monitored 375 nests. Of these nests, 237 (63.2%) hatched and 138 (36.8%) failed (Table 6). Depredation and abandonment were the two largest causes of nest failure at 15% and 13%, respectively (Table 7). The highest abandonment rate was in 2014, at 20%, and the greatest depredation rate was in 2015 (46%). Of the 2015 nests, 18 were believed to have been depredated by red fox at Smith Point County Park based on fox sign found at the nest site. At least one fox had learned to enter exclosures at Smith Point by tearing the netting at the top to get to the eggs inside, leading to the removal of exclosures from nests by Suffolk County Park monitors, which all subsequently were depredated.

We found 103 nests in 2019; 72 survived to hatching and 31 failed (Table 8). Of the 31 nests that failed, 13 were abandoned, seven were depredated, two flooded, and nine had unknown fates primarily due to inclement weather preventing access to nests in a timely enough manner to determine nest fate. All seven nests depredated in 2019 were left unexclosed by Suffolk County Parks. Four of the depredated nests were suspected to be the result of an avian predator due to tracks and other avian sign such as cached eggs found near the nests. The remaining three depredated nests showed no sign apart from the egg shell and yolk found at or near the nests.

Table 6. Piping plover nest distributions and status at Fire Island and Westhampton Island, in various habitats of each management unit, 2013–2019. Nest numbers include renests of some pairs. Nests were only designated "Natural Overwash" if they were within the original Hurricane Sandy created overwashes.

Management Unit	Habitat	Found	Hatched	Failed	Hatch %	Fail %
Fire Island National Seashore - Lighthouse Tract[a]	Backshore	1	1	-	100	-
Fire Island National Seashore - Lighthouse Tract[a]	Barrier Interior	2	-	2	-	100
Fire Island National Seashore - Lighthouse Tract[a]	Manipulated Dune	2	2	-	100	-
Fire Island National Seashore - Lighthouse Tract[a]	Swale	1	1	-	100	-
Robert Moses State Park[a]	Backshore	57	38	19	67	33
Robert Moses State Park[a]	Barrier Interior	13	5	8	38	62
Robert Moses State Park[a]	Dredge Spoil	5	3	2	60	40
Robert Moses State Park[a]	Swale	3	3	-	100	-
Robert Moses State Park[a]	Dune	1	-	1	-	100
Robert Moses State Park[a]	Management Area	1	1	-	100	-
Cupsogue Beach County Park[b]	Backshore	6	4	2	67	33
Cupsogue Beach County Park[b]	Barrier Interior	11	7	4	64	36
Cupsogue Beach County Park[b]	Manipulated Dune	3	3	-	100	-
Cupsogue Beach County Park[b]	Manipulated Overwash	35	24	11	69	31
Fire Island National Seashore - east of breach	Manipulated Dune	1	-	1	-	100
Fire Island National Seashore - east of breach	Natural Overwash	52	34	18	65	35
Fire Island National Seashore - east of breach	Backshore	1	1	-	100	-
Fire Island National Seashore - west of breach	Natural Overwash	23	19	4	83	17
Smith Point County Park	Backshore	22	12	10	55	45
Smith Point County Park	Barrier Interior	12	9	3	75	25
Smith Point County Park	Manipulated Dune	8	4	4	50	50
Smith Point County Park	Manipulated Overwash	65	39	26	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	34	21	13	62	38
Off Site – Bellport to Watch Hill[c]	Backshore	8	4	4	50	50
Total		375	237	138	63	37

[a] Not monitored 2013–2014 [b] Not monitored in 2013 [c] Bellport Beach to Watch Hill only monitored 2013–2014

Table 7. Fate of nests on Fire Island and Westhampton Island, by year, in each management unit, 2013–2019.

Year	Management	Total	Successful	Failed			
				Abandoned	Depredated	Flooded	Unknown /Other
2013	Fire Island National Seashore	11	8 (73%)	1 (9%)	-	1 (9%)	1 (9%)
	Smith Point County Park	9	3 (33%)	-	5 (56%)	-	1 (11%)
	Cupsogue Beach County Park	4	3 (75%)	-	-	-	1 (25%)
2014	Fire Island National Seashore	11	4 (36%)	5 (45%)	2 (18%)	-	-
	Smith Point County Park	9	4 (44%)	-	5 (56%)	-	-
	Cupsogue Beach County Park	5	4 (80%)	-	1 (20%)	-	-
2015	Robert Moses State Park	10	5 (50%)	5 (50%)	-	-	-
	Fire Island National Seashore	4	2 (50%)	-	2 (50%)	-	-
	Smith Point County Park	23	2 (9%)	1 (4%)	18 (78%)	-	2 (9%)
	Cupsogue Beach County Park	11	7 (64%)	-	2 (18%)	1 (9%)	1 (9%)
2016	Robert Moses State Park	12	8 (67%)	4 (33%)	-	-	-
	Fire Island National Seashore	9	7 (78%)	1 (11%)	1 (11%)	-	-
	Smith Point County Park	16	12 (75%)	1 (6%)	1 (6%)	-	2 (13%)
	Cupsogue Beach County Park	6	6 (100%)	-	-	-	-
2017	Robert Moses State Park	13	6 (46%)	3 (23%)	-	1 (8%)	3 (23%)
	Fire Island National Seashore	14	10 (71%)	2 (14%)	-	1 (7%)	1 (7%)
	Smith Point County Park	24	19 (79%)	2 (8%)	2 (8%)	-	1 (4%)
	Cupsogue Beach County Park	11	5 (45%)	2 (18%)	2 (18%)	-	2 (18%)
2018	Robert Moses State Park	19	14 (74%)	2 (11%)	1 (5%)	2 (11%)	-
	Fire Island National Seashore	13	10 (77%)	1 (8%)	1 (8%)	1 (8%)	-
	Smith Point County Park	31	20 (65%)	4 (13%)	7 (23%)	-	-
	Cupsogue Beach County Park	7	6 (86%)	1 (14%)	-	-	-
2019	Robert Moses State Park	26	17 (65%)	4 (15%)	2 (8%)	2 (8%)	1 (4%)
	Fire Island National Seashore	29	21 (72%)	3 (10%)	1 (3%)	-	4 (14%)
	Smith Point County Park	37	27 (73%)	6 (22%)	2 (5%)	-	2 (5%)
	Cupsogue Beach County Park	11	7 (64%)	-	2 (17%)	-	2 (17%)
Total		375	237 (63%)	48 (13%)	57 (15%)	9 (2%)	24 (6%)

[a] Includes Watch Hill nests, only monitored 2013–2014

Table 8. Piping plover nest distributions and fates on Fire Island and Westhampton Island, in various habitats of each management unit, April–August 2019. Nest numbers include renests of some pairs. Nests were only designated "Natural Overwash" if they were within the original Hurricane Sandy created overwashes.

Management Unit	Habitat	Nests			Reason for loss
		Found	Hatched	Failed	
Robert Moses State Park	Backshore	20	15	5	Abandoned (1), Flooded (2), Predated (1), Unknown (1)
Robert Moses State Park	Barrier Interior	5	1	4	Abandoned (3), Predated (1)
Robert Moses State Park	Swale	1	1	-	-
Fire Island National Seashore	Natural Overwash	18	11	7	Abandoned (3), Unknown (4)
Fire Island National Seashore-west of Breach	Backshore	7	6	1	Predated (1)
Fire Island National Seashore-Lighthouse Tract	Swale	1	1	-	-
Fire Island National Seashore-Lighthouse Tract	Manipulated Dune	2	2	-	-
Fire Island National Seashore-Lighthouse Tract	Backshore	1	1	-	-
Smith Point County Park	Manipulated Overwash	5	4	1	Abandoned (1)
Smith Point County Park	Manipulated Dune	4	2	2	Abandoned (2)
Smith Point County Park	Barrier Interior	7	6	1	Abandoned (1)
Smith Point County Park	Backshore	7	4	3	Predated (1), Unknown (2)
Smith Point County Park	Restoration Area	14	11	3	Abandoned (2), Predated (1)
Cupsogue Beach County Park	Manipulated Overwash	4	3	1	Predated (1)
Cupsogue Beach County Park	Barrier Interior	3	2	1	Unknown (1)
Cupsogue Beach County Park	Backshore	4	2	2	Predated (1), Unknown (1)

The number of pairs we monitored increased from 19 in 2013 to 86 in 2019 (356.6%; Table 9). However, the study site boundaries have changed throughout the years of monitoring (Table 10). The entire study area population increased 168.8% since 2013, and 186.6% from the 30 pairs in the study area prior to Hurricane Sandy. The majority of these 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 each season 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 six years of the monitoring effort. Though there was fluctuation within management units, the overall number of pairs increased every year since Hurricane Sandy in 2012 (Table 10). Numbers increased in the "backshore/barrier interior" habitat between 2014 and 2015 due to the addition of the Robert Moses State Park study area, which only consists of backshore or barrier interior habitat.

Table 9. Number of monitored nesting piping plover pairs in each of the various habitats. Restoration areas were not created until after the 2014 nesting season. Some pairs had more than one nest per year, but renests were only counted if the subsequent nest was in a different subsite *and* habitat class. Table footnotes indicate changes in monitoring area among years.

	Year						
Habitat	2013[a]	2014[a]	2015[b]	2016[b]	2017[b]	2018[b]	2019[b]
Overwash	11	17	15	14	23	21	25
Breach Fill	1	1	3	3	3	3	3
Restoration	N/A	N/A	5	4	6	10	13
Backshore/Barrier Interior	7	3	12	16	13	24	45
Total	19	21	35	37	45	58	86

[a] Includes Watch Hill nesting pairs, only monitored 2013-2014

[b] Includes Robert Moses and the Lighthouse Tract only monitored 2015–2019.

Table 10. Number of pairs nesting within 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. These data appear bold and italicized.

	Year						
Management Unit	2013	2014	2015	2016	2017	2018	2019
Robert Moses State Park	***12***	***10***	10	9	8	15	22
Fire Island Lighthouse	***1***	***1***	0	1	1	1	4
Fire Island National Seashore West	3	1	1	1	0	1	7
Fire Island National Seashore East	4	4	2	6	10	9	13
Smith Point County Park	6	8	13	13	20	25	31
Cupsogue Beach County Park	3	5	7	6	6	7	9
Offsite- Bellport to Watch Hill	3	3	0	0	0	0	***3***
Offsite- Sailor's Haven	0	0	0	0	***1***	0	0
Total	32	32	33	36	46	58	89

Density

Density in the study area increased from 0.06 pairs/ha of sandy habitat patches > 1m^2 in 2013 to 0.24 pairs/ha in 2019. Density remained relatively stable from 2013–2017, and showed the most increase from 2017–2019 (Figure 12).

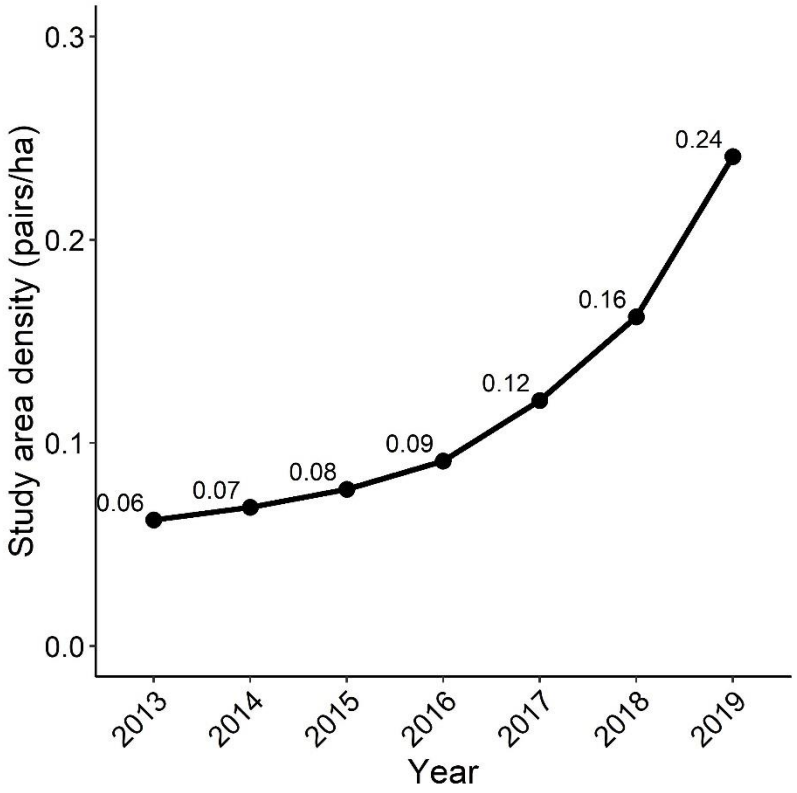


Figure 13. Estimated study area density for each year following Hurricane Sandy. Available habitat was defined as all available dry sand patches greater than 1m^2 using 15-cm classified aerial imagery. For 2013 and 2014, the number of pairs was a combination of the number of pairs monitored by Virginia Tech and the number of pairs estimated at Robert Moses State Park and the Lighthouse Tract.

From 2013 to 2019, density increased within all management units. The greatest density was observed at the East Wilderness Area (0.42 pairs/ha, Figure 13). For most management units, the lowest density was observed in 2013, directly following Hurricane Sandy. However, the West Wilderness Area did not have any pairs in 2015 or 2017, and therefore density was 0 pairs/ha. The East Wilderness Area also had a decline in density between 2014 and 2015, but increased again in 2016 (Figure 13).

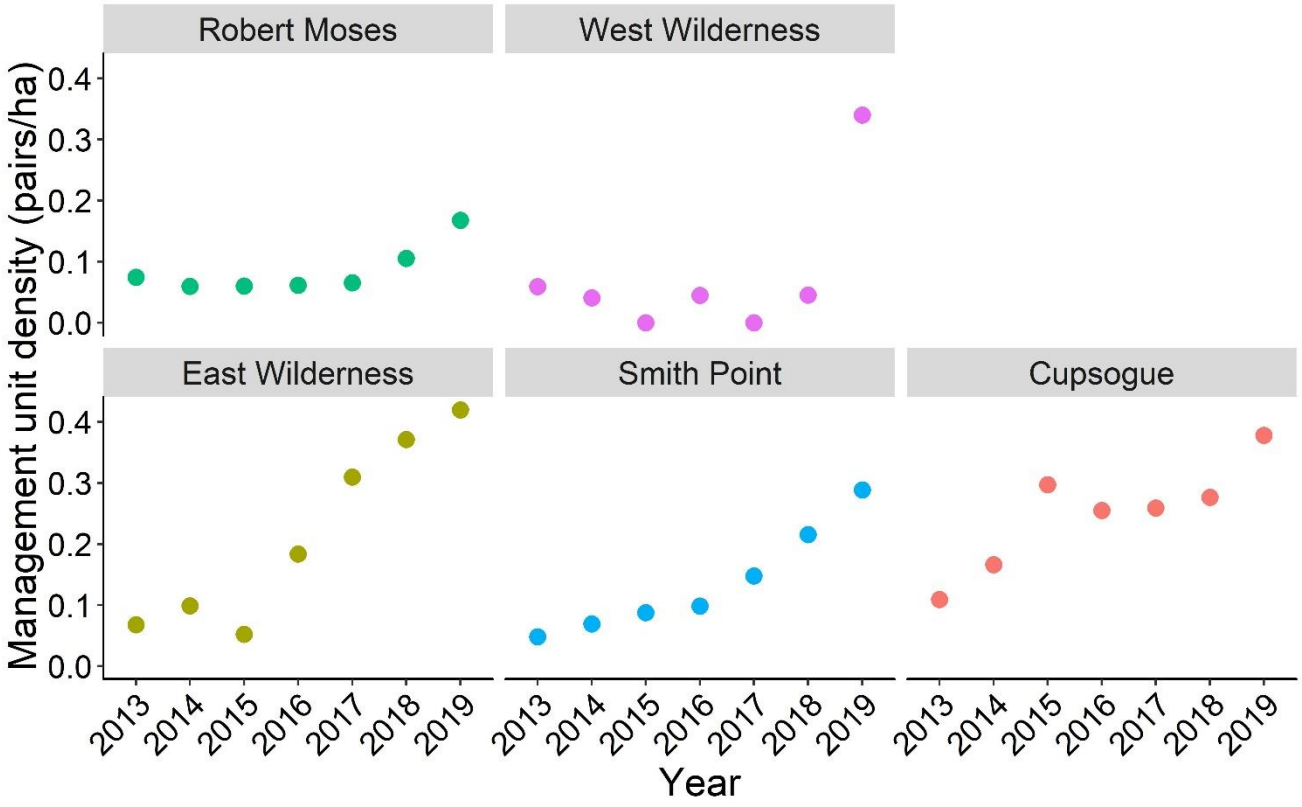


Figure 14. Estimated management unit density for each year following Hurricane Sandy. Available habitat was defined as all available dry sand patches greater than 1m^2 using 15-cm classified aerial imagery. For 2013 and 2014, the number of pairs was a combination of the number of pairs monitored by Virginia Tech and the number of pairs estimated at Robert Moses State Park and the Lighthouse Tract. The Lighthouse Tract was grouped in with Robert Moses State Park for all years.

When we estimated density by subsite, the greatest densities were observed at Old Inlet West in 2019, the small inlet subsite west of Old Inlet (1.17 pairs/ha) and Narrow Bay West at Smith Point in 2019 (0.77 pairs/ha). Annual subsite densities are available in Appendix A. Average subsite density increased over the study period (Figure 14), and when separated by subsites with and without bay access, subsites with bay access had, on average, higher densities. Both subsites with and without bay access increased in density through the study, but the average increase for bay-accessible sites increased faster as density increased and thus the difference between the two habitat types widened as density increased (Figure 15).

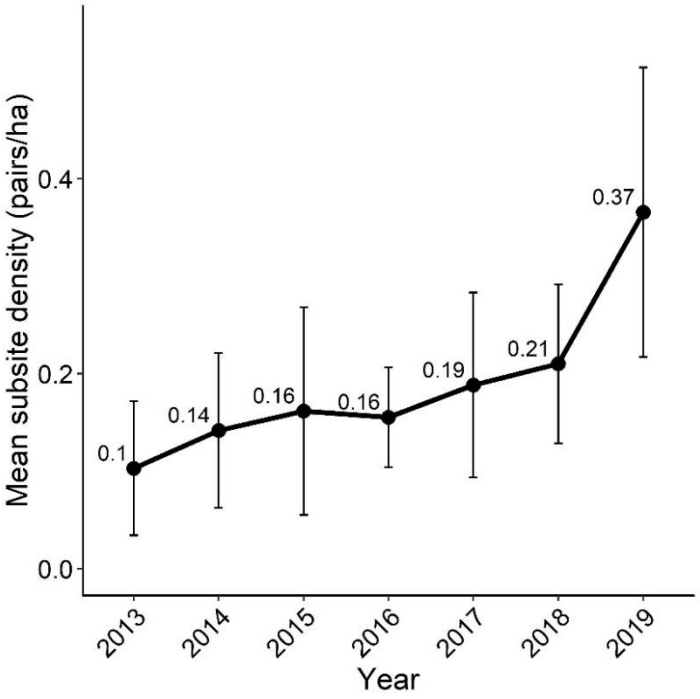


Figure 15. Average subsite density in each year. Subsite extent for density extended the entire width (north to south) of the island. Available habitat was defined as all dry sand patches greater than 1m^2 using 15-cm classified aerial imagery. Error bars represent 95% confidence intervals.

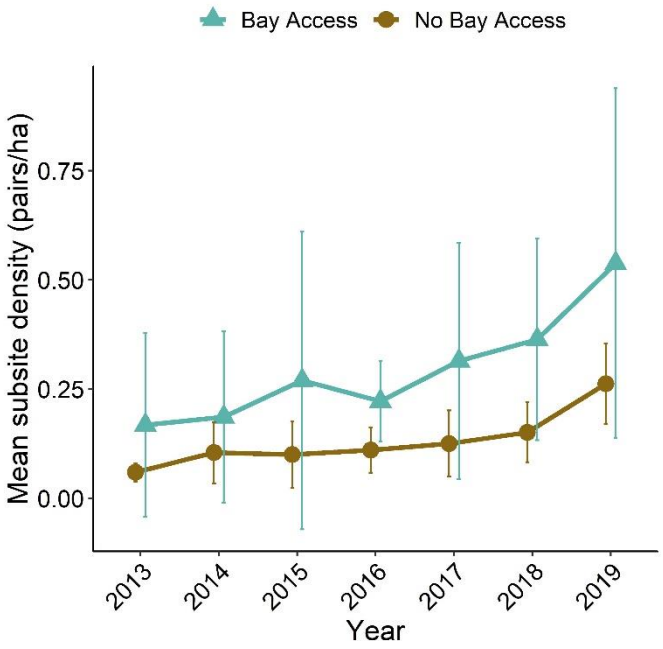


Figure 16. Average subsite density in each year for subsites with bay access (i.e., DMP, OSS/STP, PSO, NBW, CSW) compared to subsites without direct bay access. Subsite extent for density extended the entire width (north to south) of the island. Available habitat was defined as all dry sand patches greater than 1m^2 using 15-cm classified aerial imagery. Error bars represent 95% confidence intervals.

Discussion

Despite increased habitat and relatively high immigration rates, the population increase in our study area since the hurricane (186%) has been relatively modest compared to some other habitat creating events but is within the range of past observations over similar time periods (42%–745%; Robinson et al. 2019). Less than anticipated growth immediately following Hurricane Sandy may have been related to generally low local and regional reproduction just before and immediately following the storm (USFWS 2019). We estimate that an average of 31% of adult birds die annually (Part 2, Section 7), and there must be enough reproduction to replace those just to maintain a stationary population. Early in the study, the population may have decreased without immigration as it did elsewhere in the NY-NJ recovery unit in 2012–2013. The study area population may have also remained stable following 2014, as immigration was influential to the population increase (Part 2, Section 7, Robinson et al. *in prep*). Pairs at our study site increased every year throughout the study as did the number of nests, except for a decrease in nests between 2015 and 2016. There were likely more nests in 2015 than 2016 due to the frequency of nest depredation and subsequent renesting. Predators have been known to cue in on nest exclosures and associate them with a food source (Niehaus et al. 2011, Murphy et al. 2003) which, along with the removal of nest exclosures, also may explain why 2015 showed such a high percentage of nest failure compared to other years.

Red fox numbers declined across most of our study sites, and they were extirpated from some sites during 2015–2019 due to a natural outbreak of sarcoptic mange. No foxes were detected from April 2016 to March 2017 east of Old Inlet, including Smith Point County Park, and only one was believed to inhabit that area intermittently during 2017–2018 (Part 3). In the years following this fox decline, plover nests experienced lower depredation rates. Nest abandonment was the greatest reason for nest failure in every year after 2015, except 2018, when at least one red fox was present at Smith Point County Park. Population growth rate in the study area began to increase following the extirpation of foxes, and thus it is likely that red fox predation, to some degree, was limiting population growth on the Fire Island portion of the study area.

Plover pairs nested in natural and manipulated overwash habitats the most from 2013–2019. Pairs nested in backshore and barrier interior habitats nearly as often as overwash, however, when compared to what was available, generally birds have avoided backshore and barrier interior habitats (Part 2, Section 7). Another important habitat distinction of note is the difference between Hurricane Sandy-created overwashes and overwashes created by other storm events. Since 2012, there have been multiple occasions where a storm surge created overwash habitat on a smaller scale than Hurricane Sandy. For example, in 2019, we observed new overwash habitat west of Old Inlet on Fire Island National Seashore. Following this overwash, Old Inlet West pair numbers increased 7-fold, from one pair in 2018 to seven in 2019. While technically overwash habitat, many of the nests in newly-created overwashes were classified as “Backshore/Barrier Interior” since they were

not a result of Hurricane Sandy, and our objective was primarily to understand the effect of Hurricane Sandy on the plover population. Though created at different times, these overwashes are fundamentally the same as the Hurricane Sandy natural overwash and as a result, our estimates on plover usage of overwash habitats are lower than in reality.

We found that pair density nearly quadrupled from 2013, the first breeding season following Hurricane Sandy, to 2019, the last year covered in this report. Much of the trend in overall study area density was driven by Smith Point and Robert Moses, as these sites were greater than twice the area of the other management units and consistently increased in density each year. We originally hypothesized that density should begin to decline as the piping plover population on Fire Island reached carrying capacity. In a study of Westhampton Island, Cohen et al. (2009) identified carrying capacity to be 1.05 pairs/ha in two-sided habitats (ocean-side and bay-side connected by sand traversable by a walking plover) and 0.5 in one-sided habitat (no sand connection between ocean and bay), and this was the point at which the population began to decline. Remarkably, the density within this study in 2019, 7 years after Hurricane Sandy, is only about a quarter of what Cohen and coauthors found for two-sided habitats and half that found in one-sided habitat. However, density, like many population parameters, is highly sensitive to the boundaries in which it is defined. Here we classified available piping plover nesting habitat as any patch of dry sand at least 1-m^2 composed of 15-cm dry sand pixels. This fine-scale habitat delineation likely identified patches of habitat as available that would be excluded with a coarser level of habitat identification, making cross-study comparison challenging. After seven years of plover monitoring following Hurricane Sandy, the population has continued to increase, and reproductive output has not yet declined below what would be needed for stationarity (Weithman et al. 2019) nor has immigration slowed considerably (Objective 5). Therefore, it appears that our population has not reached carrying capacity. Continued monitoring of plovers and their predators following the 2019 breeding season is crucial to understand carrying capacity and what population size and density is sustainable on Fire Island and Westhampton Island in the presence of a recovering, post-mange red fox population.

Given this population's high reproductive output, the number of immigrants and vegetative succession, nesting habitat may soon limit the population. After Hurricane Sandy, vegetation increased in each of the newly created habitats from 2013–2017, especially in overwash areas (Bellman 2018), where it exceeded the 30% vegetation cover threshold established under the BO (U.S. Fish and Wildlife Service 2014; see Appendix B). From 2018–2019, we continued to see vegetation encroach on dry, sandy areas across our study site. This increase in vegetation is happening concurrently with the increase in plover density. Piping plovers generally nest in sparsely vegetated or flat, open sand due to the potential increased risk of depredation associated with dense vegetation (Espie 1996, Cohen 2008). Cohen et al. (2009) observed that the piping plover population declined at their study site (West Hampton Dunes, NY) as nesting habitat degraded due to vegetation growth.

An earlier study by Wilcox (1959) also cited increasing vegetation density as a possible reason for plovers deserting a study area east of Moriches Inlet. Therefore, we expect that suitable nesting habitats will continue to decline with vegetation succession and thereby influence population growth within our study area.

In the early spring of 2019, a large portion of vegetation at the Great Gun restoration area was removed mechanically, resulting in a 4.06 ha expanse of sand nearly devoid of vegetation. Continued vegetation management may be required to maintain open patches of preferred nesting habitat in areas of dramatic vegetation encroachment or in sites where vegetation inhibits chick access to foraging habitat.

During 2013–2019, we banded 1,006 piping plovers on Fire Island and Westhampton Island. As a result, we were able to uniquely identify individuals and accurately track population change through the years. These data are crucial for determining site fidelity and survival for plovers. Our banding effort has led to valuable insight regarding many facets of plover population dynamics that can ultimately have an influence on management decisions for the species. Without such an effort, it would have been difficult or impossible to determine the implications of Hurricane Sandy on the plover population within our study area and the mechanisms of any population change.

The implications of this study on plover population dynamics are numerous, but further investigation is required to understand the factors influencing this population's growth and eventual limitations. Although carrying capacity has not been met, with the drastic decrease in number of red foxes on Fire Island, there is one less factor potentially working against vital rates as this population as it continues to grow. Indeed, the local predator community has played an important role on reproductive output (Part 2, Section 6) and the population growth rate, but we also have observed greater conspecific disturbance and competition each year as density increases. Further, the red fox population is expected to rebound from mange-related mortalities in the coming years. The succession of plant communities at a natural rate and accelerated by anthropogenic activities like dune construction and snow fence installation also has been occurring in the overwashes and breach fill habitats for over seven years and is contributing to nesting and foraging habitat loss and fragmentation. Without the creation of new habitat, either mechanically or naturally, this population may soon begin to experience negative density dependent feedback as habitat availability decreases (Cohen et al. 2009). How the population responds to these pressures within the following few years will be crucial in determining carrying capacity in our study site and the limitations that define it.

Section 4: Estimate the "effect distance" of newly-created habitat

Introduction

Bird movement and habitat use vary across the annual and life cycle, and on finer time scales within breeding and non-breeding periods, based on reproductive requirements, energetic demands, and food availability. Movement, frequently driven by accessibility to high-quality food resources (Plissner et al. 2000), has been shown to differ in a number of seabirds based on nesting and chick-rearing periods of the breeding period (Mackley et al. 2011, Cecere et al. 2013, Walter et al. 2014) and by age (Skórka et al. 2016). As movement is associated with energetic costs (Wiltermuth et al. 2015), connectivity of a landscape may also influence how well birds can exploit high-quality resources (Farmer and Parent 1997) to maintain proper condition for various reproductive, survival and growth-related energetic needs. Although investigated in seabirds, there has been little investigation into fine-scale breeding movements in shorebirds. For imperiled species like the piping plover, it is important to address this knowledge gap in order to understand fine-scale movement and habitat use during different biological periods on the breeding grounds and inform management practices.

We may expect differences in movement and habitat-use between biological periods due to various constraints during each period. For example, adult piping plovers with nests and chicks may be limited to a smaller area surrounding their nests during the pre-fledging period. More specifically, various habitats, although functionally available by flight during brood-rearing, may become more fragmented than during the nesting phase based on the constraints of broods to ground movement and of adults to defending their young. Non-breeding birds, who are not focused on a reproductive investment, and fledglings, no longer limited to a natal territory, may travel further to use and prospect high-quality habitat.

The creation of natural and engineered habitats, particularly overwashes, breaches, and restoration areas, warrants investigation into how piping plovers are using newly available habitats and how far they will travel to get there. Understanding habitat and space use of piping plovers may help to evaluate the effects of natural and engineered habitat creation on the landscape, especially when habitats are fragmented. These data can also be useful in planning management actions such as beach closures. We evaluated how far individuals traveled from their nesting and hatching locations to other subsites in the study area. In particular, we were interested in estimating the effect distance, defined here as the distance which birds will travel to use a habitat, of Hurricane Sandy-created overwashes and restoration areas. We investigated this movement in nesting, brooding and non-breeding (including pre- and post-breeding) adult and fledgling piping plovers.

Methods

We compiled all adult locations of individuals confirmed to a nest within a given year during 2014–2019 and divided them into three 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 (hereafter; nesting). The second set contained all of the points of banded individuals when they had a known active brood (hereafter; brooding). The third set of points contained all of the points of banded individuals that were confirmed to a nest within a given year, but while they did not have a known active nest or brood, including the periods before nest initiation and after brood success or failure (hereafter; non-breeding). Similar to adult locations, we compiled all fledgling (confirmed flight) locations during 2014–2019 and calculated the distance from each location to the nest from which the bird hatched. Fire Island banded fledglings were observed during off-site surveying at Jones Beach State Park, NY and were observed by birders elsewhere, but those locations were not considered in this analysis.

Using the package "geosphere" (Hijmans 2016) in program R (R Core Team 2016), we calculated the straight-line distance of each adult location from their nest location and each fledgling location from their hatch location. For adults with active nests, we removed locations within 30 m of the nest to account for birds incubating and GPS error. 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 adult location. For non-breeding locations, we calculated the distance from the adult location to the final known nest location.

We joined all four 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 four 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 or water within the respective year's habitat classification (Part 2, Section 1). Subsites with direct bay access included Narrow Bay West, Pattersquash Overwash, Cupsogue West, Democrat Point, and the Stockpile.

Results

Adult Movement

We assessed the use of newly created habitats and other habitats of interest by adult piping plovers relative to their nest location in a given year. During 2014–2019, we recorded 7,291 observations of adult plovers from the study area. Of those, 2,089 were nesting adults, 1,883 were brooding adults, and 3,368 were non-breeding adult locations.

The greatest distance that an adult with an active nest was observed from its nest location within a breeding season was 10.8 km; this occurred in 2015 when an adult with a nest at Great Gun was observed at Old Inlet East (Table 11). The greatest distance a brooding adult was observed from its nest location was an adult nesting at Pattersquash Overwash in 2018, when she was observed 6.9 km away at Old Inlet West (Table 12). The greatest distance a nonbreeding adult was observed from its nest location was 48 km; an adult was observed at Robert Moses State Park prior to nesting in Great Gun (Table 13).

Table 11. Distance moved from nest to Hurricane Sandy-created habitats, restoration areas, breach fills, flood shoals, and Cupsogue Sand Bar and Mud Flat, for nesting adult piping plovers on Fire Island and Westhampton Island, 2014–2019.

Habitat	Distance from Nest (m)				95% Confidence Interval	
	N	Mean	Max	SE	Lower	Upper
Cupsogue Mud Flat	9	1336	2547	242	778	1894
Cupsogue Sand Bar	20	866	2137	93	672	1061
OIW	13	626	1353	158	282	970
New Made Restoration Area	21	373	2451	118	127	619
Bridge Overwash	28	229	1325	62	102	356
NBE Overwash	99	227	8362	86	57	398
Pattersquash Overwash	192	224	5899	37	151	297
Cupsogue Breach	95	195	4184	46	103	286
OIE Overwash	401	176	10862	32	113	240
NBW Overwash	100	173	2005	24	126	220
Cupsogue Overwash	147	165	4090	30	106	224
Flood Shoals	1	150	150	-	-	-
Great Gun Restoration Area	147	125	735	9	107	143
Narrow Bay Breach	102	122	1132	13	97	147
OIWR Overwash	15	75	157	10	53	98

Table 12. Distance moved from nest to Hurricane Sandy-created habitats, restoration areas, breach fills, and Cupsogue Sand Bar and Mud Flat, for brooding adult piping plovers on Fire Island and Westhampton Island, 2014–2019.

Habitat	Distance from Nest Location (m)				95% Confidence Interval	
	N	Mean	Max	SE	Lower	Upper
Cupsogue Mud Flat	4	2302	4720	807	-266	4871
OIW	32	1709	6904	291	1115	2302
Flood Shoals	1	1055	1055	-	-	-
Cupsogue Sand Bar	4	716	907	94	418	1014
NBW Overwash	83	586	4265	85	416	755
Bridge Overwash	29	483	4208	148	180	786
NBE Overwash	48	405	2898	80	244	567
Pattersquash Overwash	194	294	5302	47	201	386
OIE Overwash	341	289	5875	44	202	376
New Made Restoration Area	9	268	372	48	158	378
Great Gun Restoration Area	92	194	2018	32	130	257
Cupsogue Breach	104	189	525	13	163	215
OIWR Overwash	14	183	271	21	137	228
Cupsogue Overwash	140	128	497	7	114	142
Narrow Bay Breach	72	124	250	8	108	139

Table 13. Distance moved from nest to Hurricane Sandy-created habitats, restoration areas, breach fills, flood shoals, and Cupsogue Sand Bar and Mud Flat, for adult non-breeding piping plovers on Fire Island and Westhampton Island, 2014–2019.

Habitat	Distance from Nest Location (m)				95% Confidence Interval	
	N	Mean	Max	SE	Lower	Upper
Flood Shoals	19	9724	33050	2895	3643	15805
OIW	70	3584	34405	649	2290	4878
Cupsogue Mud Flat	39	2339	8692	287	1759	2919
Cupsogue Sand Bar	147	1860	15575	191	1482	2238
NBE Overwash	180	1753	45862	317	1126	2379
Pattersquash Overwash	333	800	8096	87	627	972
OIE Overwash	570	753	36061	121	515	991
OIWR Overwash	13	675	8090	618	-672	2022
Bridge Overwash	44	631	9880	251	126	1136
Cupsogue Breach	171	544	7745	90	365	722
Cupsogue Overwash	184	496	14895	112	274	718
NBW Overwash	109	439	7313	108	224	653
New Made Restoration Area	45	287	2502	85	115	458
Narrow Bay Breach	69	214	3607	64	87	341
Great Gun Restoration Area	176	211	2778	29	155	268

We had one observation of a nesting plover and one observation of a brooding plover at the Flood Shoals in Old Inlet. The nesting individual moved only 150 m to forage there from Old Inlet West. The individual with an active brood traveled 1054 m from Old Inlet. We recorded 19 locations of non-breeding adults at the Flood Shoals in Old Inlet. The greatest of these non-breeding movements was 33 km from Robert Moses State Park (Table 13).

We had 20 observations of birds with active nests, four observations of birds with broods, and 147 observations of birds without active nests using the Cupsogue Sand Bar over the five years of the study. The greatest distance that a nesting adult moved to the Cupsogue Sand Bar was 2.1 km from Great Gun Restoration Area. All of the observations of nesting individuals using the Cupsogue Sand Bar were nesting either at Cupsogue Beach County Park or Great Gun Restoration Area. Of the 147 instances of individuals using the

sand bar that were not actively nesting, the greatest observed distance was 15.6 km, an individual nested at Old Inlet West earlier in the season.

At the Cupsogue Mud Flat, there were nine observations of nesting adults, and four observations of brooding adults. Similar to Cupsogue Sand Bar, most of these individuals were nesting either at Cupsogue Beach County Park or Great Gun Restoration Area, although one individual with an active brood traveled from Smith Point West, which was 4.7 km from Cupsogue Mud Flat. We recorded 44 non-breeding observations of individuals foraging at the Cupsogue Mud Flat. Of these birds, the greatest distance traveled was 8.7 km from their nest, an individual that had nested that season at Pattersquash Overwash.

Across the three biological periods, individuals used all of the overwashes within the study area (Table 11, Table 12, Table 13). There were 228 observations of 96 individuals at locations other than the subsite they were nesting in, 50% of which were foraging observations. Most of these locations (17%) were of individuals foraging at the Cupsogue Sand Bar, with birds traveling from Cupsogue Beach County Park and Great Gun Restoration Area. The next most used site was Democrat Point (15%), with birds traveling most often from the Robert Moses State Park fields. Birds also were frequently observed foraging on the bayside mudflat at Narrow Bay West, with individuals traveling most often from New Made Restoration Area. Individuals also moved from Narrow Bay East and New Made Restoration Area to Old Inlet East Overwash, and from Old Inlet East Overwash to Bridge Overwash. While many of the observations of piping plovers at the overwashes were observations of adults that nested there, birds with active nests elsewhere used the overwashes as well. Specifically, the greatest distance moved by a nesting adult was to the Old Inlet East Overwash, adjacent to the Old Inlet breach, a new habitat created by Hurricane Sandy (Table 11).

On average, adults with active nests remained closer to their nest locations than birds with active broods or non-breeding adults across years (Figure 16). Non-breeding adults, that were either pre- or post-breeding, moved the farthest from their nest locations but there was no clear trend across years. Distance traveled by nesting and brooding adults stayed relatively consistent across years with brooders traveling farther from their nest locations than nesters in all years except 2015. On average, brooding adults moved more than 400 m from their nest locations from 2014–2018, however, their movement decreased in 2019 and was more similar to nesting adult distances. In 2019, nesting and brooding adults were, on average, seen closer to their nest locations than in all years prior.

During 2014–2019, nesting plovers with direct bay access were not seen significantly closer to their nest locations ($\bar{x}$ = 187.05, SE = 8.34) than nesting plovers that did not have direct bay access ($\bar{x}$ = 251.58, SE = 19.62; df = 1, W= 538490, P = 0.85).

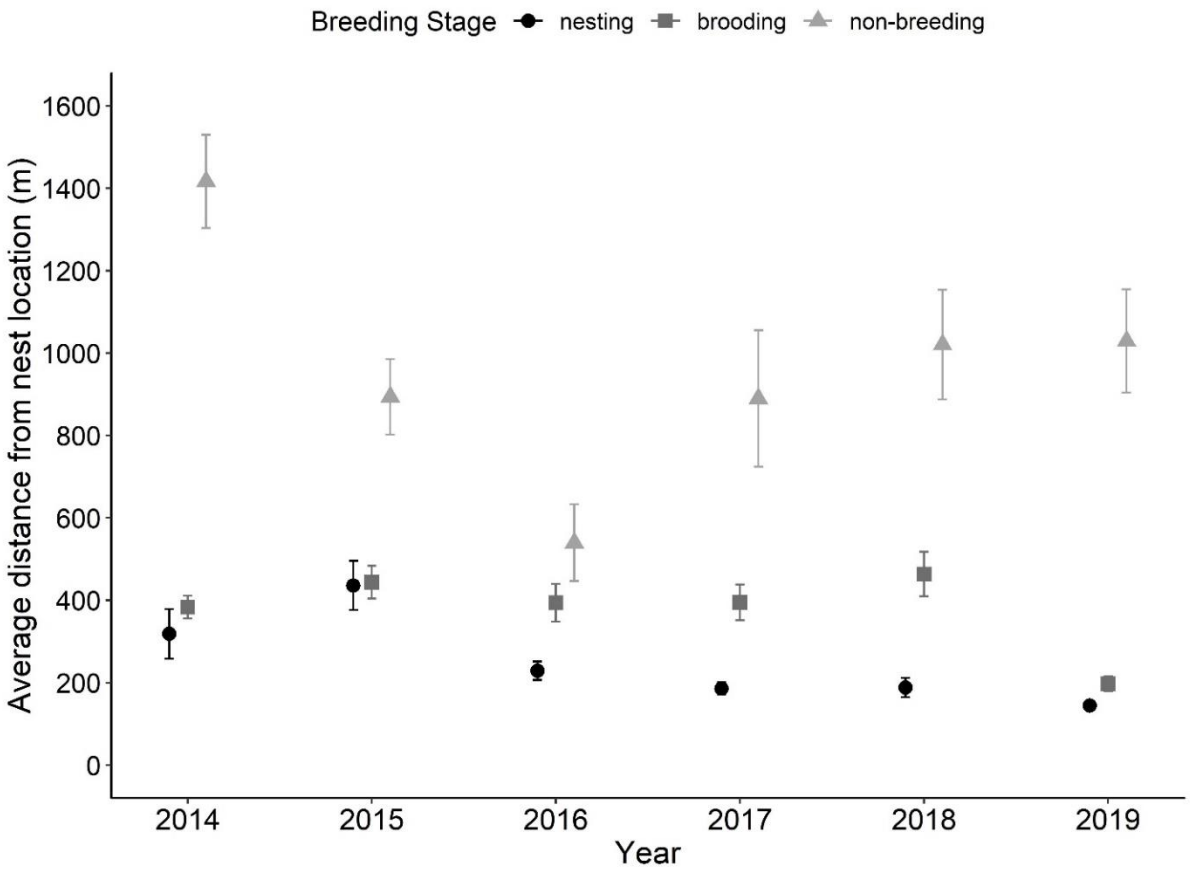


Figure 17. Average distance moved by piping plover adults from their nest location, grouped by nesting, brooding, and non-breeding individuals at the time of location collection, separated by year on Fire Island, 2014–2019. Error bars represent standard error.

Fledgling Movement

We assessed the use of newly-created habitats and other habitats of interest by fledgling piping plovers relative to the nest location from which they hatched. From 2014–2019, we recorded 1,751 observations of fledglings from the study area. On average, fledglings were observed 3,417 m (SE = 199 m; Table 14) from their hatch location. The maximum distance a fledgling was observed from its hatch location was 50.2 km; a fledged chick from Cupsogue West was observed foraging at Democrat Point, Robert Moses State Park, in 2018. On average, fledglings moved farthest to Cupsogue Mud Flat, Flood Shoals, Old Inlet West, and Cupsogue Sand Bar (Table 14). During 2015–2017, the only fledglings observed in restoration areas were individuals that had hatched there. In 2018 and 2019, 13 fledglings that had hatched elsewhere were observed in Great Gun Restoration Area and 72% of those observations were on the oceanfront. During 2014–2019, we observed 21 fledglings return to nest as first-year birds within a subsite where they had been observed as a fledgling the year prior, five of which nested within their natal subsites from the year prior.

Table 14. Distance moved from hatch location to Hurricane Sandy-created habitats, restoration areas, breach fills, and Cupsogue Sand Bar and Mud Flat, for fledgling piping plovers on Fire Island and Westhampton Island, 2014–2019.

Habitat	Distance from Hatch Location (m)				95% Confidence Interval	
	N	Max	Mean	SE	Lower	Upper
Cupsogue Mud Flat	64	6323	17212	631	5061	7585
Flood Shoals	114	5921	36258	861	4215	7626
OIW	87	4341	35808	676	2997	5685
Cupsogue Sand Bar	73	3951	16162	497	2959	4942
OIWR Overwash	5	2665	13104	2610	-4582	9911
Pattersquash Overwash	173	1716	7414	171	1378	2054
NBW Overwash	42	1715	7280	333	1043	2387
Great Gun Restoration Area	18	1447	4489	407	588	2305
Cupsogue Breach	33	1361	12399	534	273	2449
Cupsogue Overwash	61	1324	12120	341	642	2006
OIE Overwash	297	1314	13902	155	1009	1620
Bridge Overwash	30	1139	10805	353	416	1862
NBE Overwash	17	852	3294	236	352	1352
Narrow Bay Breach	18	745	5729	319	73	1418
New Made Restoration Area	5	377	387	10	348	405

We estimated average movement of piping plover fledglings from their hatch location by year. In 2014, the average distance that piping plover fledglings were observed from their hatch location was 1,954 m (SE = 216; Figure 17). By 2018, the average distance increased to 4,286 m (SE = 391). There was an overall increasing trend in average distance fledglings were observed from their hatch location relative to time since Hurricane Sandy.

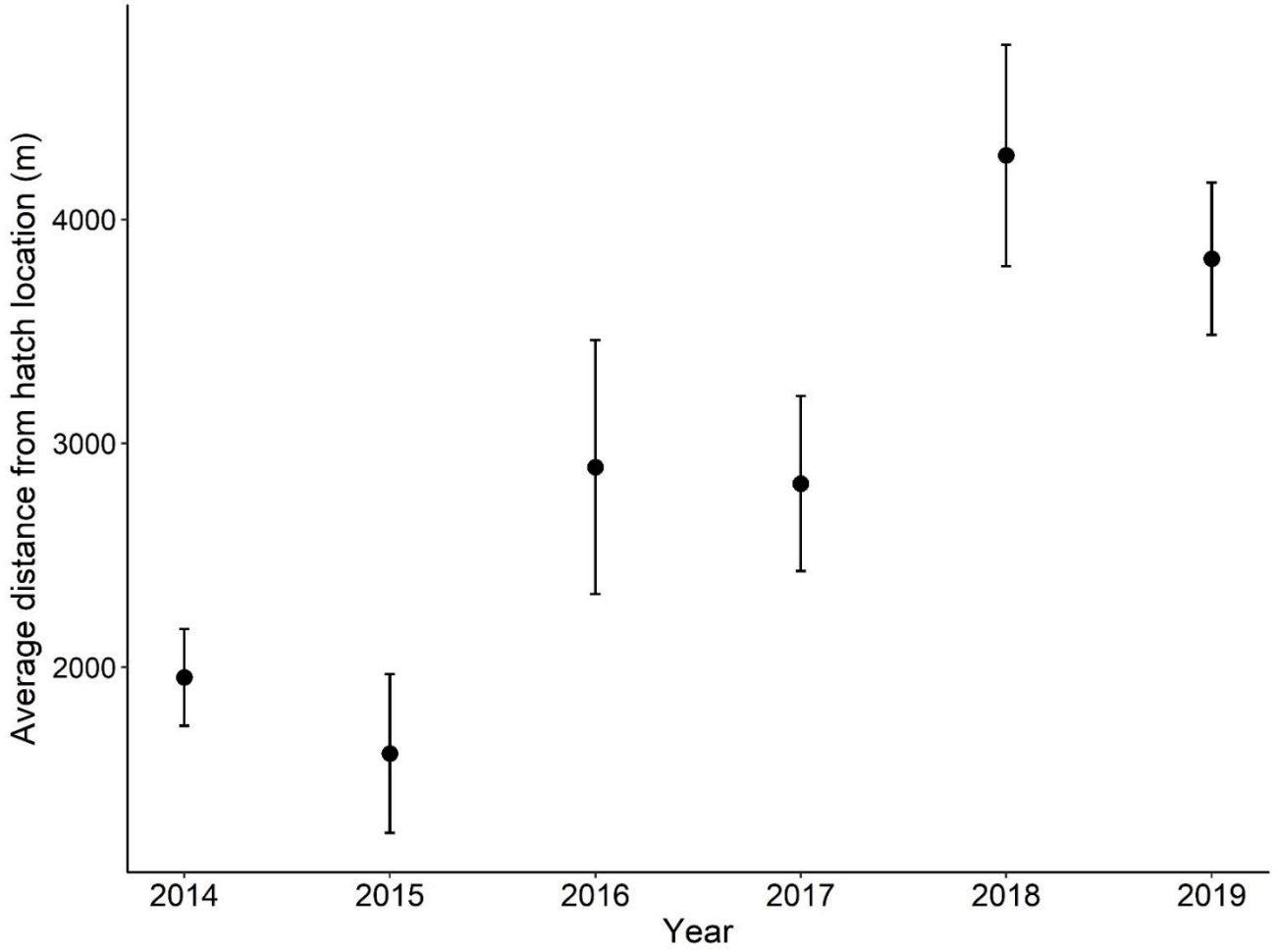


Figure 18. Average distance moved by piping plover fledglings by year from their hatch location on Fire Island, June–August, 2014–2019. Error bars represent standard error.

Discussion

Piping plover adults and fledglings traveled relatively long distances from nest and hatch locations to forage on mud flats, sand bars, flood shoals, and Hurricane Sandy overwashes. Adults on Fire Island and Westhampton Island moved similar linear distances to Great Lakes breeding plovers, whose mean linear distance traversed was 130 m to 1435 m (Haffner et al. 2009). On average, fledglings moved farthest to reach foraging habitats. Cupsogue Mud Flat, Cupsogue Sand Bar, Old Inlet Flood Shoals, and Narrow Bay East and Pattersquash overwashes all appear to be important foraging locations for adult and fledgling piping plovers. Overwashes were frequented by nesting and brooding plovers, which we expected since plovers select for them during both biological phases (Walker et al. 2019). Indeed, OIE Overwash was the site with the most observations across all categories.

Most subsites frequented by foraging piping plovers are protected by nesting closures. However, the habitats that piping plovers traveled the farthest to reach (Cupsogue Mud Flat and Cupsogue Sand Bar), are not

protected from recreational use. Because these sites are only accessible at low tides, boaters and shell fishers often occupied these foraging locations at the same time plovers were feeding. Currently, there are no limitations to human use in these areas. It is likely that plovers could benefit by protection from disturbance at these sites, which are not protected under the Biological Opinion or other management plans. We had few observations of nesting and brooding adults in foraging areas like the Flood Shoals, Cupsogue Mud Flat, and Cupsogue Sand Bar because we did not prioritize surveying these sites in the peak of breeding season during years in which shorebird surveys were not occurring. We acknowledge this survey method may have led to an observation bias, especially since these areas were frequently used by non-breeding and fledgling piping plovers. However, based on the short distances traveled by nesting and brooding adults, it is unlikely that these sites were used extensively by adults that weren't neighboring them during their nesting and brooding phases.

Compared to other habitats, adults did not travel long distances to reach restoration areas. For example, nesting and brooding adults traveled an average of less than 200 m from their nest location when they were seen in Great Gun Restoration Area, suggesting that most individuals that were seen there also nested there and that it provided adequate foraging habitat for adults. While the restoration areas were created to provide nesting habitat and ephemeral or peripheral foraging habitat, peripheral foraging grounds like ocean intertidal zones and wrack lines provided the majority of foraging at Great Gun Restoration Area. That most adults seen there were within 200 m of their nest locations is unsurprising given the extensive ocean shoreline and beach closures that minimized disturbance along the ocean and wrack foraging areas.

Average adult distance moved from nest locations varied by biological stage in the way that we expected. Non-breeders, not constrained to a territory to defend eggs or chicks, were observed farther from their nest locations than other adults, but there was no clear trend with distance moved across years. Actively nesting adults moved the shortest distances from their nest location. The distances moved by nesters decreased slightly over the years, which could be a function of increasing density in the study area. Contrary to results of Haffner et al. (2017), who found no difference in space use of piping plovers between nesting and brooding periods, our results showed a difference in movement between nesting and brooding adults in all years except 2014 and 2015. As their young hatched and became mobile, brooding adults moved greater distances, perhaps driven by the need to find adequate foraging while avoiding other territorial adults and their broods. Our results also differ from that of Mackley et al. (2011), who found that white-chinned petrels moved less during non-breeding periods compared to chick-rearing. This difference is unsurprising based on differing chick provisioning needs between seabirds and shorebirds. Adult seabirds directly feed their young and may need to travel long distances to find abundant fish resources, while shorebird chicks, who feed themselves, are not directly reliant on their parents for food. Although not evaluated in this study, condition has been found to influence movement in

varying breeding stages in Brown Pelicans (Walter et al. 2014), and it may be valuable to investigate how condition influences movement across ages and biological phases in our study area in the future.

Brooding adult movements remained relatively constant across years until 2019 when it declined. Adult movement during the brooding stage may be largely dependent on nesting pair density and may be further compounded by synchrony of hatching. Pair densities were highest in 2019 (Part 2, Section 3). Intraspecific aggression may occur in high densities and may intensify if nests hatch around the same time. Territorial aggression was observed in 2019, particularly in foraging areas, and may have constrained adults and their broods to areas closer to their nesting territories than in years past to avoid aggression.

Fledglings moved farther from their hatch location than adults did from their nest location during each year. Inconsistent with adult movements across all biological periods, there was an increasing trend in average distances fledglings were seen from their hatch location from 2014–2019. This trend may be related to the increase in pairs, densities, and fledglings produced within the study area, which may have driven fledglings farther than they may have had to go in earlier years when there were fewer pairs, fewer fledglings, and perhaps less territorial aggression. Further, increasing movements over the years may be related to vegetative increase in the study area (Bellman 2018), which may have constrained local suitable habitat use and forced fledglings to seek habitats farther from hatch locations as vegetative succession occurred. Fledglings traveled long distances from their hatching sites to Great Gun Restoration Area and other habitats that are predominantly nesting grounds rather than foraging sites, suggesting that fledglings may be prospecting for future nesting locations (Davis et al. 2017). Prospecting presumably requires wide sampling to fully evaluate habitat availability and quality to inform current and future habitat selection (Davis et al. 2017). Thus, habitat connectedness (i.e., proximity of nesting habitats to foraging habitats) may play a large role in movement of young birds (Skórka et al. 2016). This acquisition of knowledge about local habitats is not necessary for adults with high nesting site fidelity (Cohen et al. 2006, hunt e et al. 2015) that do not require exploration to evaluate resource distribution (Skórka et al. 2016), further explaining the differences we saw in movements between ages.

A small proportion of fledglings nested in subsites where they were seen as a fledgling in the year prior. Because we surveyed foraging sites (Cupsogue Mud Flat, Cupsogue Sand Bar, and the Flood Shoals) as frequently as nesting sites after peak breeding, many surveys occurred at low tides, concurrent with other surveys at neighboring nesting grounds. For this reason, we may have created another observation bias of detecting individuals foraging rather than surveying for them at higher tides when they may have been roosting or prospecting nearby. We believe there were likely more fledglings that were prospecting in our study area, but we did not detect them due to surveys frequently occurring during lower tides.

Our results indicated the differences in movement among various adult stages and fledglings on the breeding grounds and highlighted the long distances plovers traveled to reach foraging areas. Movement and

habitat use should be continually monitored to assess any potential responses to vegetative regrowth and yearly geomorphic changes that may alter foraging and nesting habitat availability and fragmentation. Continued monitoring of these movements will remain key as habitat succession continues and will be especially valuable if habitat engineering occurs in our study area in the future. We highlight the need for continued protection of overwashes and for future protection of mud flats and sand bars to ensure these important habitats are functionally available for piping plovers in future years.

Section 5: Compare egg weights of nests with access to bay side intertidal flats with egg weights of birds without such access

Introduction

Piping plovers select nest sites adjacent to low-energy, bayside intertidal flats (Elias et al. 2000), likely considering the needs of their offspring during the nest and brood-rearing phases of the breeding cycle (Fraser and Catlin 2019, Walker et al. 2019). However, proximity to foraging locations may also influence fitness of adults (Cohen et al. 2009). Female piping plovers need proper nutrition and body condition to produce high-quality eggs, therefore, it may benefit them to select nesting locations adjacent to high-quality foraging habitat (Cohen and Fraser 2010) to maximize the quality of food and minimize the energy-expenditure needed to obtain it (Farmer and Parent 1997). Piping plovers gain mass faster in low-energy habitats and their condition is positively associated with clutch mass (Catlin et al. 2014). In some shorebird species, there is a relationship between egg mass and chick mass (Ricklefs 1984), suggesting that a female's condition before nesting may influence chick condition, which is positively associated with survival (Catlin et al. 2014). On Fire Island and Westhampton Island, low wave-energy bay intertidal habitats have the highest invertebrate abundances, and thus may be higher quality habitats based on food availability (Part 2, Section 8). For this reason, it is important to investigate the influence of bay-accessibility on egg mass and clutch size. We compared the egg masses of birds in bay-adjacent subsites compared to nests without adjacent bayside habitat, and hypothesized that clutch mass of birds in bay-adjacent subsites would be greater than clutch mass of nests without adjacent bayside intertidal habitat.

Methods

We used all clutches found at less than a full clutch during 2013–2019 to determine if clutch masses at nest completion differed for piping plovers that nested bay-adjacent compared to those that were not bay-adjacent using the Wilcoxon Rank Sum Test. For the comparison, we considered a nest to be bay-adjacent if the subsite a nest was laid in had bay-adjacent intertidal habitat. Bay-adjacent subsites included Narrow Bay West, Pattersquash Overwash, Cupsogue West, Old Inlet West, Democratic Point, and Stockpile. When we were unable to weigh clutches on the day they were completed, we used simple linear regression to estimate the total clutch mass at the time of completion.

Results

In 2019, we weighed 50 nests at full clutch. There was no difference in the clutch masses of piping plovers that were in bay-adjacent subsites from those that were not bay-adjacent (W=254, df=1, P=0.50, $\bar{x}$ =38.68). Similarly, from 2013–2019, we weighed 213 nests at full clutch, and there was no difference in the

clutch masses of piping plovers that were bay-adjacent versus those that were not (W=5063, df=1, P=0.76, $\bar{x}$ = 39.08).

Discussion

Although low-energy habitats have high invertebrate abundances (Loegering and Fraser 1995, Elias et al. 2000, Cohen et al. 2009, Part 2, Section 8) and assist in higher piping plover growth rates than higher-energy habitats (Catlin et al. 2014), we found that nests in bay-adjacent subsites did not have higher clutch masses than those without adjacent bayside intertidal habitat. Pre-nesting (and post-nesting) adults moved an average of 1030 m (SE=126; Part 2, Section 4) to reach foraging habitats, suggesting that adults will travel long distances to reach optimal foraging and that those movements during this study did not affect mass of clutches. Our results suggest piping plover adults are not limited to the prey availability or quality in their nesting subsite during egg development, perhaps because they are able to fly beyond subsite boundaries to reach favorable food resources. Although the influence of bay habitat availability does seem to be essential for other portions of the plover breeding period, adults do not need to be directly adjacent to them to benefit from them, at least as quantified given the plover densities and food availability during our monitoring period.

Section 6: Compare piping plover nest and chick survival rates in various habitats across Fire Island

Introduction

Reproductive output has been shown to correlate with population change in piping plovers (Saunders et al. 2018). For Atlantic piping plovers, nest survival tends to be high due to the installation of nest exclosures (Melvin et al. 1992). Nest exclosures, however, do not prevent all causes of nest failure, such as flooding or predation by predators that can fit inside nesting exclosures (e.g., ghost crabs (*Ocypode quadrata;* Kwon et al. 2018) or American mink (*Neovision vision*; M. Stantial, personal communication)). Once precocial chicks have hatched, the nest exclosure is no longer useful at protecting the chicks, which move about and forage for themselves soon after hatch. Therefore, it is important for the next generation of plover management and conservation to better understand the best methods to improve plover chick survival and therefore reproductive output given the challenges of plover management outside of their breeding areas. Plover chick survival could be influenced by habitat suitability and could be reduced if chicks need to move around a lot to avoid predators or access suitable foraging habitat. Thus, understanding the suitability of habitat, and movement and home range size of broods could inform future management to increase survival of piping plover chicks.

Fire Island reproductive output has been assessed in terms of human disturbance (DeRose-Wilson et al. 2018) and in comparison to a more southerly site (Weithman et al. 2019). We also were interested in any relationship between reproductive output time since Hurricane Sandy, and geomorphic and habitat variables. Specifically, we were interested in effects of artificially-created nesting habitats for piping plovers, which were built intentionally (restoration areas), built for island stabilization (breach fills), and were created by the hurricane (natural and manipulated overwash; Part 1, Table 3). Home range size and movement of broods also may vary by habitat types, and may further contribute to reproductive output dynamics through effects on growth and survival in some species (Lengyel 2006, Nichols and Kaiser 1999). If it is determined that a specific habitat feature, natural or artificial, can positively influence reproductive output and thus influence the overall population in the study area, perhaps future habitat management can try to replicate those features.

Here, we focused on the components of reproductive output (nest survival, chick survival) and some potential drivers of chick survival (i.e., movement and home range) on Fire Island and Westhampton Island following Hurricane Sandy and the FIMI project. Our objectives were to understand 1) the influence of red fox (*Vulpes vulpes*), raccoon (*Procyon lotor*), and feral cat (*Felis catus*) abundance on piping plover chick survival, 2) how movement and home range size change in relation to habitat and time since Hurricane Sandy, and 3) which of these factors had the greatest influence on piping plover nest and chick survival.

Methods

Nest Survival

We used a logistic exposure model (Rotella et al. 2004, Shaffer 2004) to estimate the daily survival rates of piping plover nests. We built models that compared survival between exclosed and unexclosed nests (i.e., large, wire cages placed around nests to reduce predation; Melvin et al 1992), management units, and habitat types [breach fill, manipulated overwash, natural overwash, restoration, other]. All categorical habitat variables were analyzed in reference to the 'other' habitat type, to understand how nest survival in newly created habitats compares to pre-Sandy habitats. A nest was considered exclosed only during the period when the exclosure was present. When predicting from this model, hatch date was set at 34 days from initiation date (Heisey et al. 2007).

To distinguish whether nest survival was affected by predator abundance, habitat, density, or weather, we developed a full model, which estimated the effect of continuous variables on nest survival. We used nest initiation date, nest age, nesting plover density, maximum temperature, average wind speed, total precipitation, distance to vegetation and elevation, and least cost distance to ocean and bay. We also included a random effect of subsite and year. Using this same model, we subset our nests by the areas in which predators were monitored (i.e., not Cupsogue), and included a categorical effect of pre- and during- the mange outbreak in comparison to post-mange outbreak (Part 2, Section 1; Pre-mange was 2013–2015 East of Old Inlet and 2013–2017 West of Old Inlet; post-mange was 2016–2019 east of Old Inlet, 2018–2019 west of Old Inlet). Cupsogue was removed because we did not have knowledge of the predator community nor mange at Cupsogue. In order to focus more carefully on predator numbers, we built a second model, only including the years of the concurrent fox study (Part 2, Section 1), 2015–2018, and only including management units where foxes were monitored. This model included all of the previously mentioned variables, but instead of a categorical fox variable, we included trap success (# detections/100 trap nights; Part 2, Section 1) of red fox, feral cat (*Felis catus*) and raccoon (*Procyon lotor*).

Chick Survival

To estimate chick survival, we used a modified Dail-Madsen model, a Bayesian implementation of the young survival from marked adults' model, applied in jags (Lukacs et al. 2004, Kellner 2018). We modeled chick survival for each year from 2013 to 2019. We estimated daily survival over a 25-day period using 5-day chick detection intervals. We considered several models to derive estimates of chick survival for the whole study area. These included an annual survival model, one which estimated chick survival among management units, and another that estimated chick survival among habitat types [breach fill, manipulated overwash, natural overwash, restoration, other]. All categorical habitat variables were in reference to the 'other' habitat type,

which was any habitat that did not fall into the other four but was primarily backshore habitats that existed prior to Hurricane Sandy, to understand how newly created habitats compare to pre-Sandy habitats. To estimate the study, habitat, and management-unit-specific probability of survival to fledge, we calculated the product of the five 5-day interval survival rates from the year-specific model.

As we did for nest survival, we developed two full models to understand whether chick survival was affected by habitat, weather, density, or time. In comparison to the nest survival model, we were limited by the number of covariates we could include due to small sample size and model complexity. We included a linear trend of time, hatch day, maximum temperature within the 5-day interval, average wind speed over the 5-day interval, average least cost distance to ocean, least cost distance to bay, distance to development and distance to vegetation over each 5-day interval, and management-unit nesting plover density. Distance variables were calculated as average distance variables within 5-meters of chick use points, and were averaged over the 5-day interval prior to detection. We developed a separate model without Cupsogue, because no predator information was known from Cupsogue. We included the categorical variable of pre- and during mange as compared to post mange, as described above for nest survival. We also developed a model of chick survival using trap success of red fox, raccoon, and feral cats and nesting density as dependent variables using data from 2015–2018, the years the plover and fox projects were concurrent. Plover density and predator trap success were measured at a management-unit level.

Because we used models in a Bayesian framework, we evaluated the influence of explanatory variables using both the overlap of the 95% credible interval with 0 and the proportion of the posterior distribution that has the same sign as the mean effect (Catlin et al. 2019*b*).

Reproductive Output

We estimated reproductive output using annual monitored pair counts and the number of fledglings observed for each year. Chicks were considered fledged if they reached 25 days of age or were observed in sustained flight. Using nesting plover density estimates from Objective 1, we tested for the correlation between reproductive output and subsite nesting density (see Appendix A), management unit nesting density, and study area nesting density (Fire Island to Cupsogue), using Pearson's correlation coefficient, to determine if we could detect a relationship between reproductive output and density.

Finally, we compared piping plover productivity before and after the mange outbreak. For one management unit (Cupsogue), the predator community was unknown, and was considered separately. Fledglings produced per pair were calculated for specific management units, with the Lighthouse Tract grouped into Robert Moses State Park and the Fire Island Wilderness split into areas East and West of Old Inlet.

Brood Home Range

We created minimum convex polygons (MCP) as brood home ranges from 2014–2019 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 location points available. We first estimated cumulative home range size as the age of the brood increased to understand how age affected home range size. We then subset the home ranges from broods that fledged at least one chick, to control for broods that had fewer points and smaller home ranges due to early chick mortality. We used a Kruskal-Wallis test to test whether there was difference between area of brood home range and habitat type [breach fill, manipulated overwash, natural overwash, restoration, other], and we used linear regression to test for a difference between broods with bay access or not, controlling for the number of points used in the estimate of the MCP.

We developed a full model, which tested whether home range size varied by year, habitat type, distance to vegetation, development, least cost distance to bay and ocean, and management unit nesting density. Distance variables were measured from the brood home range centroid. Finally, to understand whether the red fox mange outbreak had an effect on home range size, we developed a model with only Fire Island broods (i.e., excluding Cupsogue) and included a post-mange variable. In all models, we also included the number of points used to estimate the home range, and log transformed brood home range size to normalize the response variable.

Brood Movement

We calculated the movement of broods between successive daily observations (time between observations = 1 day). If we collected more than one brood location in a day, we selected the last location collected in a day (Weithman 2019). If we first observed the brood outside of the nest bowl, we added in a location at the nest site on the hatch day. We then used the "geosphere" package in Program R to calculate the Euclidean distance from every point to the one preceding it (Hijmans 2016), and scaled locations by the time between observations, removing observations that the time between observations was greater than 1 day. We 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 to test whether there was difference between area of brood home range and accessible foraging habitats (breach fill, restoration, natural overwash, manipulated overwash, and other).

Results

Nest Survival

The probability of surviving to hatch varied across years (Table 15). Nest survival was highest in 2016 and lowest in 2015. Overall, nests that were exclosed during or after clutch completion had a higher predicted probability of surviving to hatch than unexclosed nests (Figure 1). Survival to hatch for exclosed nests ranged from 0.51–0.84, and survival to hatch for unexclosed nests ranged from 0.22–0.67 (Figure 19). However, in 2015, midway through the nesting season, exclosures were removed from all of the Smith Point County Parks nests by Suffolk County Parks monitors (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).

When comparing nest survival among the five habitat types [breach fill, manipulated overwash, natural overwash, restoration area, and other], nest survival was not significantly affected by habitat. The mean effect indicated lower nest survival for restoration areas, breach fills and manipulated overwash compared to 'other' habitats, but 95% credible intervals overlapped zero for all habitat types (Table C-2).

Of all variables examined "exclosed" had the greatest positive relationship with nest survival (Figure 20). Nesting plover density, temperature, and the number of eggs in a nest also correlated positively with nest survival, whereas nest age and wind had negative relationships with nest survival (Figure 20). When we added the categorical fox effect into the model (pre/during vs. post mange), nest survival was higher in the post-mange years than in the years prior to and during the mange outbreak (Figure 20). The relationship between nest survival and nesting plover density switched signs, resulting in a negative relationship, although not significant. Finally, from the model assessing fox, feral cat, and raccoon trap success during 2015–2018, trap success of red fox, feral cat and raccoon from the winter camera grid did not explain variation in nest survival, but plover nesting density was positively related to nest survival (Figure 3).

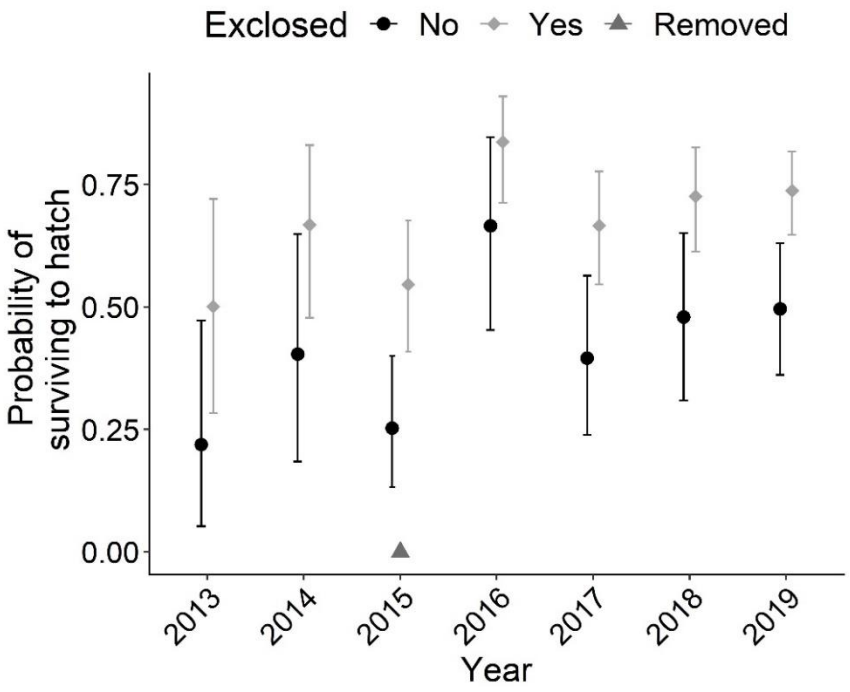


Figure 19. The probability of piping plover nests surviving to hatching with an exclosure and without an exclosure on Fire Island and Westhampton Island, 2013–2019. Error bars represent 95% credible intervals.

Table 15. Predicted probability of plover nests surviving to hatch (34 days from initiation) for all four management units in 2013–2019. 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.

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

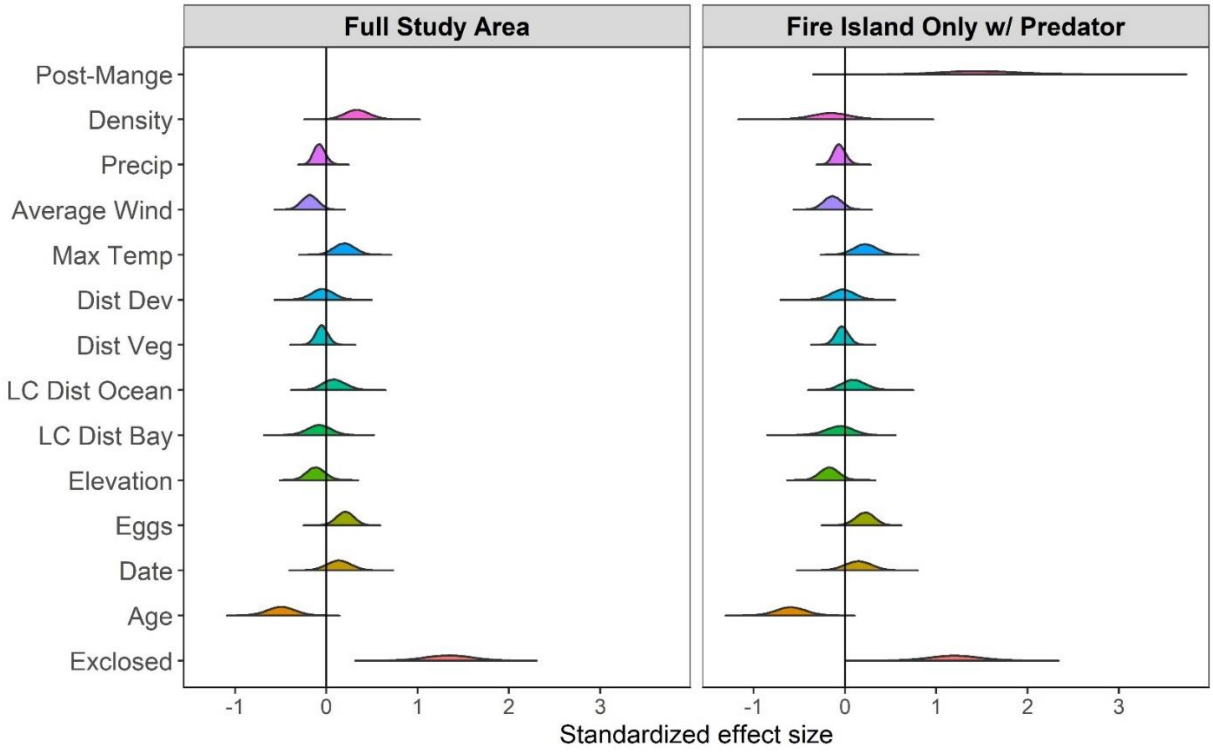


Figure 20. Posterior distribution of nest survival coefficients from a logistic exposure model testing the effects of management-unit and interval specific variables on nest survival. Model was developed using all data, 2013–2019 (left) and only Fire Island management units for the same years but including the effect on nest survival of pre- and during the sarcoptic mange outbreak on the red fox population, as compared to post-mange (right).

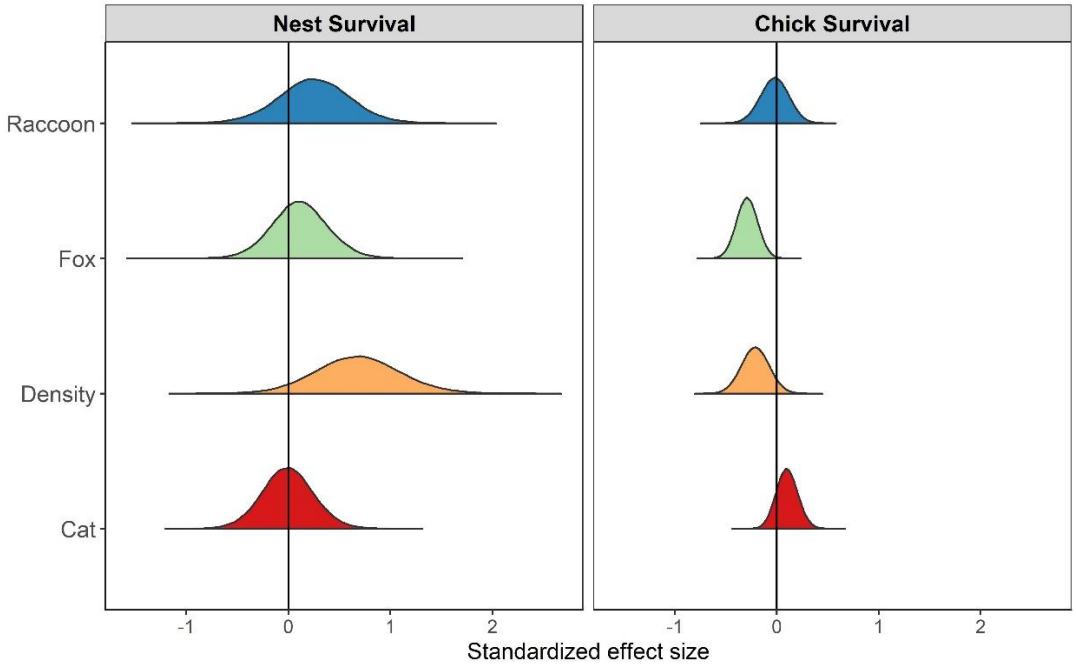


Figure 21. Posterior distribution of standardized nest and chick survival coefficients illustrating the effect of raccoon trap success, red fox trap success, nesting plover density and feral cat trap success on chick survival, on Fire Island during 2015–2018. Nest survival coefficients resulted from a logistic exposure model, and chick survival coefficients resulted from a modified Dail-Madsen chick survival model. Trap success was estimated at the management unit level as # species detections/100 trap nights from concurrent predator monitoring at game camera grids occurring from Robert Moses State Park to Smith Point County Park.

From 2013–2019, 40 of 356 nests were abandoned (11%, Table 16). From the 40 abandoned nests, nine nests belonged to a pair of unbanded or unknown adults, 11 nests belonged to a pair with one banded adult, and 20 nests belonged to a pair of banded adults. Of the 20 nests with a pair of banded adults, all but one nest (95%) had at least one individual resighted after the nest was abandoned. There were two nests (10%) where both banded individuals were seen after abandonment. Of the 11 nests with one banded adult, seven banded individuals (64%) were seen after nest abandonment. Of 51 banded birds from 40 abandoned nests, 28 banded individuals (55%) were seen after their nest was abandoned. Only nine of those 28 banded individuals renested within the same season. Throughout the duration of this study, we recorded four occasions in which a nest was abandoned around the same time as the mortality of one of the incubating adults. We discovered the remains of three of these banded adults within 90 meters of their nest exclosures.

Table 16. Abandonment rate of piping plover nests with exclosures and without exclosures from Fire Island and Cupsogue County Park, Westhampton Island 2013–2019.

Year	Exclosed	# of Nests	Failed	Hatched	Abandoned	Abandonment[a] Rate
2013	N	6	5	1	0	0.00
2013	Y	14	4	10	1	0.07
2014	N	8	5	3	0	0.00
2014	Y	12	4	8	2	0.17
2015	N	25	20	5	2	0.08
2015	Y	23	12	11	5	0.22
2016	N	8	3	5	1	0.13
2016	Y	32	4	28	4	0.13
2017	N	16	14	2	4	0.25
2017	Y	45	7	38	4	0.09
2018	N	9	9	0	1	0.11
2018	Y	58	8	50	6	0.10
2019	N	22	15	7	1	0.05
2019	Y	78	13	65	9	0.12
2015	Removed	5	5	0	0	0.00
All	N	94	71	23	9	0.10
All	Y	262	52	210	31	0.11

[a] Calculated as the # of abandoned nests / total nests

Chick Survival

From 2013 to 2019, we monitored 233 broods. Chick survival to fledge was highly variable among years, with the lowest survival rate in the first year of the study, 2013, and the highest rate in 2018 (Figure 22). Relative to other habitats, chicks hatching in a manipulated or a natural overwash relative to other habitats had a positive effect on chick survival. Hatching in a breach fill had a negative effect on chick survival, and hatching in a restoration area had no detectable effect on chick survival (Table C-2).

Based on the full chick survival model (i.e. including Cupsogue), broods hatched earlier in the summer had a higher probability of surviving to fledge than chicks hatching later, and the probability of a brood surviving to fledge increased as chicks aged. Chicks that were farther from the ocean and the bay during a 5-day

detection window had higher survival. Nesting plover density also was negatively associated with chick survival (Figure 23). When we removed Cupsogue from the model (resulting N = 195), chick survival was greater during the post-mange period, although only 90% of the posterior distribution agreed with the sign of the mean effect (Figure 23). Without Cupsogue broods, least cost distance to ocean and bay variables were no longer correlated with chick survival, however distance to development became important, suggesting that broods that were farther from development (i.e. paved roads, parking lots, buildings) had lower survival (Figure 23). When we included the categorical effect of post-mange on chick survival, chick survival was greater during the post-mange period,

Red fox trap success, followed by piping plover nesting density, best described chick survival in the 2015–2018 subset model (Figure 3). Higher plover nesting density and higher red fox trap success were correlated with lower chick survival, and that raccoon and feral cat trap success did not have an effect on chick survival (Figure 3).

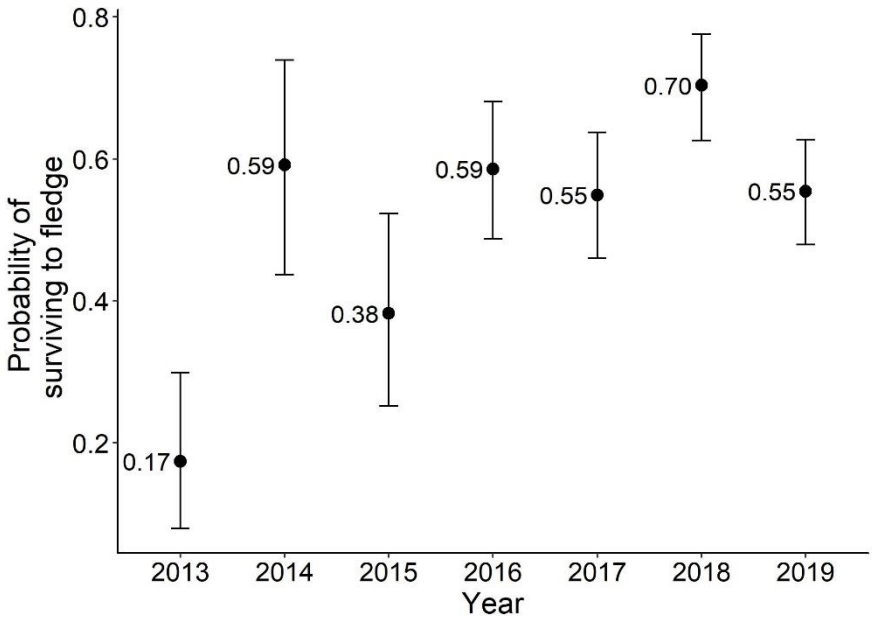


Figure 22. Probability of surviving to fledge (25 days) for chicks 2013–2019 on Fire Island and Cupsogue. Error bars represent 95% credible intervals.

Table 17. Probability of surviving to fledge for chicks in all management units and years, estimated from a young survival model. Probability of surviving to fledge represents the probability of surviving to 25 days old.

Management Unit	Year	N Broods	Survival to fledge	Lower 95% CI	Upper 95% CI
Robert Moses	2015	5	0.54	0.29	0.78
	2016	8	0.41	0.23	0.60
	2017	6	0.84	0.65	0.97
	2018	14	0.73	0.59	0.85
	2019	17	0.62	0.49	0.74
Fire Island National Seashore	2013	5	0.16	0.05	0.34
	2014	3	0.55	0.26	0.84
	2015	2	0.99	0.87	1.00
	2016	7	0.75	0.55	0.91
	2017	10	0.48	0.32	0.64
	2018	10	0.73	0.57	0.86
	2019	21	0.74	0.61	0.85
Smith Point County Park	2013	3	0.36	0.10	0.67
	2014	4	0.80	0.58	0.96
	2015	2	0.56	0.21	0.88
	2016	12	0.79	0.64	0.90
	2017	19	0.54	0.41	0.67
	2018	20	0.70	0.57	0.81
	2019	27	0.50	0.39	0.61
Cupsogue County Park	2013	3	0.10	0.01	0.30
	2014	4	0.44	0.22	0.68
	2015	7	0.12	0.03	0.28
	2016	6	0.29	0.12	0.50
	2017	5	0.45	0.22	0.70
	2018	6	0.68	0.45	0.89
	2019	8	0.30	0.16	0.47

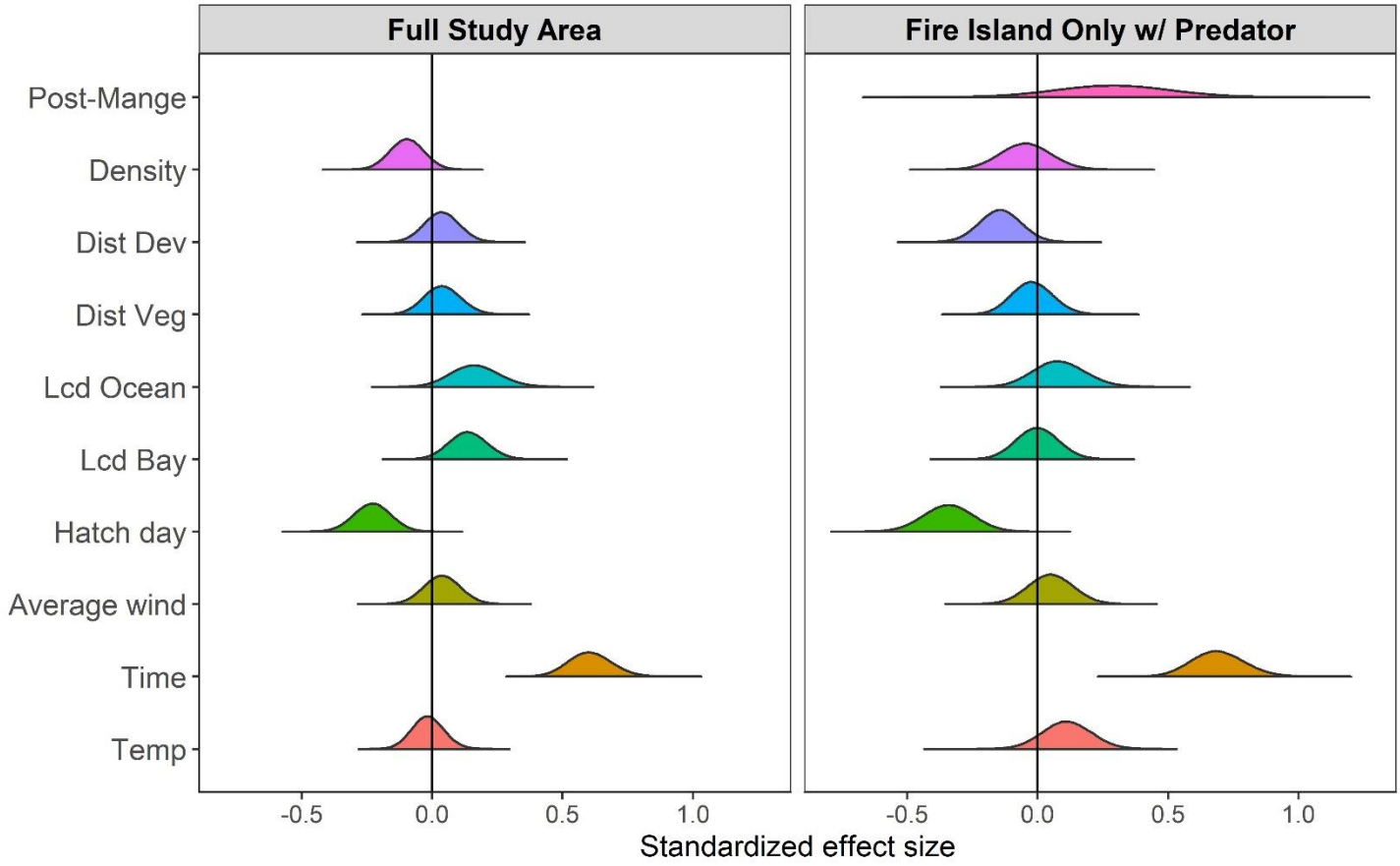


Figure 23. Standardized model coefficients from a chick survival model with the full study area (i.e., Robert Moses to Cupsogue, left) and Fire Island only (i.e., no Cupsogue) but including the effect on chick survival of pre- and during the sarcoptic mange outbreak on the red fox population, as compared to post-mange (right). Model estimates are from a modified Dail-Madsen model implemented in a Bayesian framework.

Reproductive Output

Piping plover reproductive output across the study area was variable among years, with the highest reproductive output (2 chicks/pair) in 2018 and the lowest observed reproductive output (0.39 chicks/pair) in 2015 (Table 18). Reproductive output was not correlated with subsite density (r = -0.027, *P* = 0.80), management unit density (r = 0.20, *P* = 0.26), or annual study area density (r = 0.50, *P* = 0.26). There was some evidence that reproductive output was related to the mange outbreak that occurred in 2015 on the eastern side of Old Inlet (OIE and BRO subsites) and in 2018 on the western side of Old Inlet (OIW and OIWR subsites). However, not all variation in reproductive output can be attributed to a change in predator dynamics, as some high years of reproductive output occurred prior to the mange outbreak (Figure 24).

Table 18. Piping plover pair counts and productivity (fledges/pair; Ro) in four different management units on Fire Island and Westhampton Island, May–August 2013–2019. Pairs is the number of pairs in each subsite, and Ro is the reproductive output (chicks fledged/pair).

Year		Robert Moses State Park	Fire Island National Seashore	Smith Point County Park	Cupsogue Beach County Park	Total
2013	Pairs	-	10	6	3	19
	Ro	-	0.6	0.5	0	0.47
2014	Pairs	-	8	8	5	21
	Ro	-	0.75	1.5	1.2	1.14
2015	Pairs	10	3	13	7	33
	Ro	0.3	1.67	0.31	0.14	0.39
2016	Pairs	9	8	13	6	36
	Ro	1.11	2	2.38	1	1.75
2017	Pairs	8	11	20	6	45
	Ro	1.75	1.64	1.7	0.67	1.56
2018	Pairs	15	11	25	7	58
	Ro	2.27	2.64	1.8	1.29	2.02
2019	Pairs	22	24	31	9	86
	Ro	1.36	1.83	1.32	1	1.44

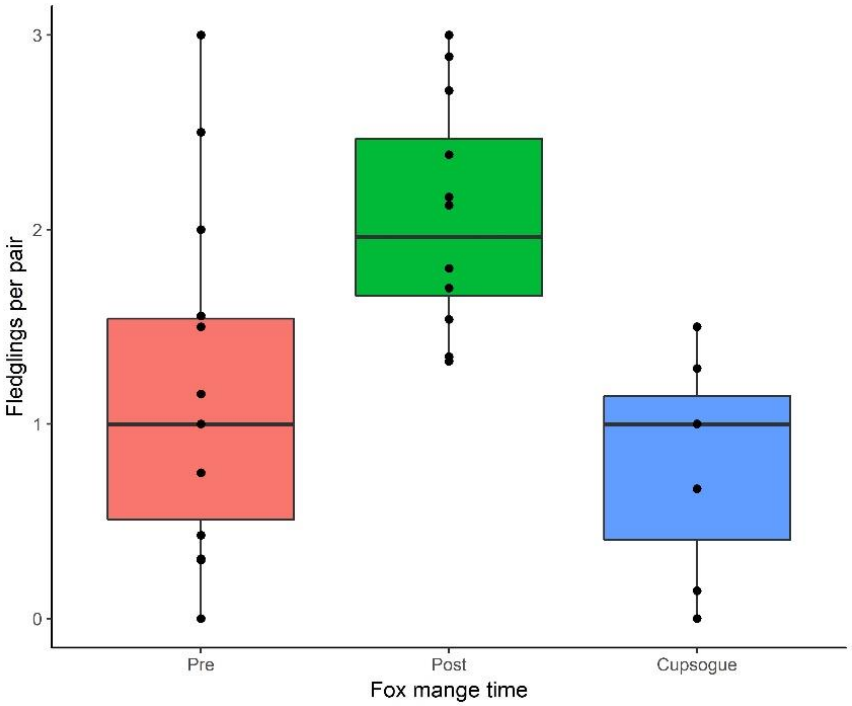


Figure 24. Fledglings per pair in relation to known mange outbreaks in the red fox population on Fire Island. Pre-mange outbreak includes 2013–2015 east of Old Inlet (OIE and BRO subsites) and 2013–2017 west of Old Inlet (OIW and OIWR subsites). Post-mange outbreak is all other dates. Cupsogue is separate, as fox monitoring does not occur at Cupsogue County Park, and few foxes or tracks have been observed. Data points for fledglings per pair are productivity for specific management units, with the Lighthouse Tract grouped into Robert Moses State Park, and the Fire Island Wilderness split into areas east and west of Old Inlet. Boxplots in this figure illustrate the median as the central horizontal line within the box and the first and third quantile as the extents of the colored box. The maximum and minimum are illustrated by lines extending out from the interquartile range. Points overlaying the plots illustrate observed reproductive output data points.

Brood Home Range

Cumulative brood home range size increased as broods aged (Figure 25); therefore, to account for potential mortality of broods, and thus smaller home ranges, we continued analyses with only broods that fledged at least one chick. Broods that fledged at least one chick (N = 169) with direct access to bayside foraging habitat in their nesting subsite (STP, PSO, NBW, CSW and DMP) had smaller home ranges ($\bar{x}$= 2.92 ha, SE =0.26) than broods that did not have such access ($\bar{x}$ = 4.74 ha, SE =0.55; Figure 26). While the relative difference was similar in 2019, the differences were not significant (K-W χ^2=4.89, *P*=0.30, Figure 8, Table 19).

Home ranges of broods that fledged at least one chick did not differ among years, but that restoration area broods had larger home ranges. When we removed Cupsogue broods and added in the pre- and during as compared to post-mange outbreak variable, home ranges were smaller in 2019, and manipulated overwash

broods had smaller home ranges (Table C-4). Home range size of plover broods did not correlate with the timing of the fox mange outbreak. Individual brood home ranges throughout the study site from 2014–2019 are displayed in Appendix C (Figures C-1 through C-6), and are symbolized by polygons stretching the extent of each home range based on Trimble GPS coordinates collected while observing broods.

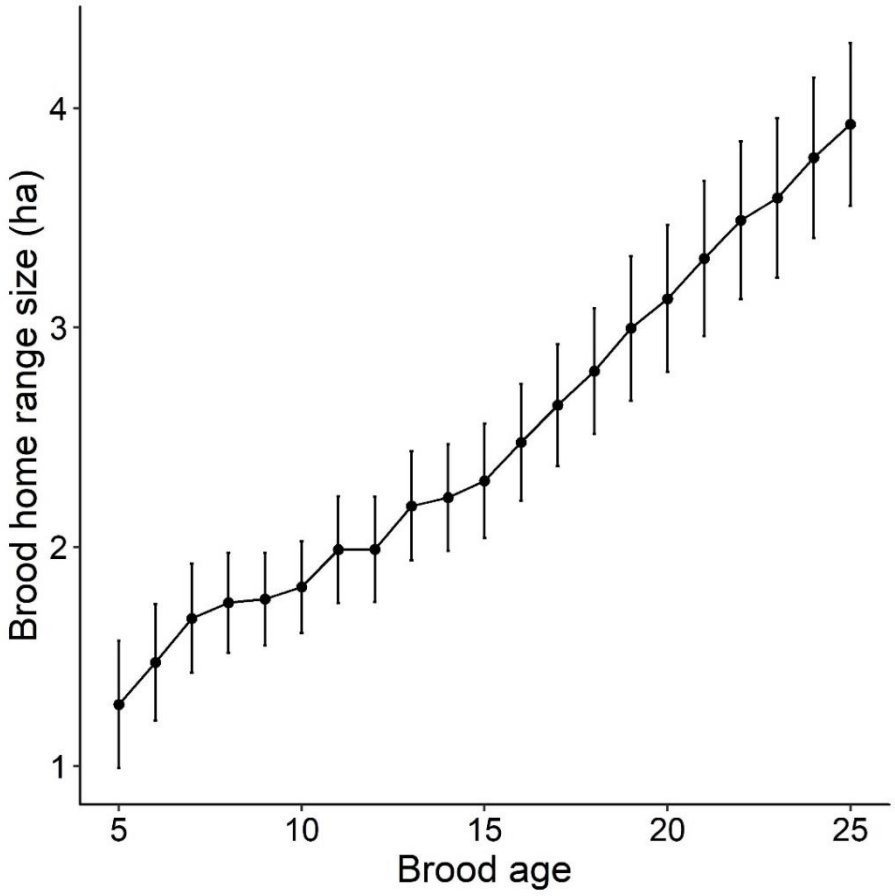


Figure 25. Cumulative home range size for piping plover broods as age increased. Home ranges were only calculated for age classes where up to that point, the brood had at least five locations. Error bars represent standard error.

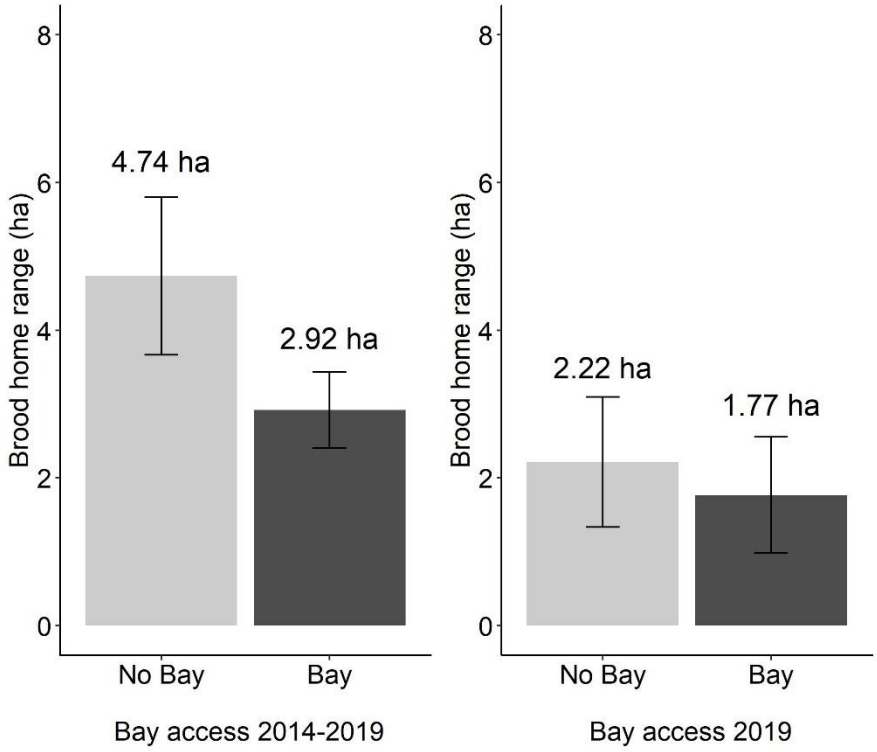


Figure 26. Average home range size among years (2014–2019) and in 2019, estimated using minimum convex polygons for broods that fledged at least one chick, of broods that did not have direct bayside access in the subsite of their nest (light gray) and the broods that did have direct bayside access in the subsite of their nests (dark gray). Error bars represent 95% confidence intervals.

Table 19. Brood home range size of broods that fledged at least one chick, by habitat type. Home ranges were created as minimum convex polygons for broods that had ≥ 5 locations.

Habitat Type	N	Mean home range area (ha)	SE	LCI	UCI
Breach fill	14	3.82	0.61	2.49	5.14
Manipulated overwash	28	3.42	0.84	1.70	5.15
Natural overwash	34	3.40	0.49	2.39	4.40
Restoration area	19	5.51	0.95	3.51	7.51
Other	73	4.22	0.65	2.92	5.53

Brood Movement

From 2014 to 2019, we recorded 1047 locations of broods on successive days. Average age of broods at observations was 10.9 days (SD = 7.79). Mean interval between observations was 1.72 days (SD = 1.43, median = 1.17). Mean daily movement was 131.68 m (SE = 4.83, median = 87.68). Movements between daily observations of broods were smaller, on average, for broods that had bay access ($\bar{x}$ = 105.16 m, SE = 5.47) than broods that did not across all years ($\bar{x}$ = 152.55 m, SE = 7.38; W = 156970, $P < 0.001$). Movement between daily observations was different among the five habitat groups (K-W $\chi^2 = 18.84$, $P < 0.001$). Broods in breach fills moved the least between daily observations and broods in natural overwashes moved the most between daily observations (Table 20).

Table 20. Average standardized distance (m/24 hr) moved by piping plover broods between successive daily observations. Breach fills included portions of Narrow Bay West and Cupsogue West. Manipulated overwashes included portions of Narrow Bay West, Narrow Bay East, Pattersquash Overwash, and Cupsogue West. Natural overwashes included Old Inlet West, Old Inlet West Reference, Old Inlet East, and Bridge Overwash. Restoration areas included Great Gun and New Made.

Habitat	N	Distance (m/24hr)	SE	95% CI
Breach Fill	107	83.04	8.37	66.45–99.62
Manipulated Overwash	152	119.33	10.26	99.06–139.61
Natural Overwash	230	149.06	10.13	129.10–169.02
Other	482	141.04	8.32	124.68–157.40
Restoration Area	76	112.96	9.43	94.17–131.74

Discussion

Exclosures

Nest exclosures had the greatest effect on nest survival of the variables studied. During all years, exclosures were placed around nests during the laying period or after clutch completion. Typically, within the study site, exclosures were not installed around nests with fewer than four eggs to prevent nest abandonment. The placement of nesting exclosures is likely to reduce any detectable effects of predators on daily nest survival unless a smart predator individual is able to reduce the effectiveness of the majority of exclosures. Predators have been known to cue in on exclosures (Murphy et al. 2003), and the managers of the plover breeding sites in the study area reacted to digging and entry into exclosures in varying manners. In 2015, Smith Point County

Park had high fox density, and foxes, or potentially a single "smart fox," took some nests by climbing into the exclosures. Following these predation events, the exclosures were removed from all other nests at the county park by Suffolk County Parks monitors. Once the exclosures were removed, the remaining nests, which were then unexclosed, were depredated. In 2018, a fox (which we believed to be the single fox in the area at this post-mange time) observed at Smith Point was digging at exclosures. After the digging incidents, Suffolk County Parks decided to not place exclosures on several nests throughout the eastern portion of Smith Point County Park in 2018 due to concern that exclosures would continue to attract predators and could lead to adult mortality. During this time, five unexclosed nests were depredated within Smith Point County Park. After these predation events, exclosures were installed around nests with at least three eggs to prevent future depredation, many of which hatched, although three were abandoned.

In one study examining plover nest exclosures, exclosing nests resulted in greater population growth rates than if nests were left unexclosed, although there is concern that nest exclosures may increase adult mortality (Cohen et al. 2016). A positive relationship between adult mortality and nest abandonment has been demonstrated (Neuman et al. 2004, Barber et al. 2010, Roche et al. 2010), and our resight rate of banded birds (55%) following nest abandonment from 2013–2019 is lower than the predicted annual survival rate of 0.69 ± 0.02 (see Objective 5). In their study of Great Lakes piping plovers, Roche et al. (2010) discovered that up to 70% of the nests monitored were abandoned due to the loss of at least one of the incubating adults. With a detection probability of 0.96 ± 0.02 (Objective 5) for adults across all years of this study, it is likely that we would have seen banded adults following nest abandonment if they were still present in our study area. Therefore, adult mortality or emigration from the study area are the two most likely causes for the relatively low percentage of banded adults resighted after nest abandonment (Roche et al. 2010). Managers who decide to continue to exclose Fire Island piping plover nests before they reach four eggs likely will reduce nest predation, improving overall productivity in the Fire Island piping plover population, but should be used with caution if nests are frequently being abandoned (Cohen et al. 2016).

Finally, we also found nest survival was negatively correlated with wind. Throughout the study, we noticed that storm events with high winds often were correlated with nest abandonment, with eggs being buried under sand and only the tops of eggs sticking out. However, because we rarely observed eggs actually blown out of nests, we suspect that the negative effect of wind on nest survival likely a result of wind improving our ability to detect that a nest had already been abandoned by covering the untended eggs with sand.

Predators

Nest survival during 2015–2018 was not related to red fox, raccoon or feral cat abundance. However, when considering all years of Fire Island data, nest survival was higher after the mange outbreak than it was

before and during the outbreak, and in the models the effect size of the mange outbreak on nest survival was greater than the effect of nest exclosures. Higher fox trap success was related to lower chick survival, and chick survival was higher in the post-mange years. Red foxes are well-known nest predators of plover nests (Patterson et al. 2011, Cohen et al. 2009, Doherty and Heath 2011), but chick mortality is challenging to document. Although we did not document foxes killing plover chicks, the relationship between red fox abundance and chick survival indicates that they may be responsible for some mortality.

Our results also suggested that the red fox population crash due to mange resulted in higher chick survival and higher reproductive output in post-mange years compared to pre-mange years despite the presence of other avian and mammalian predators in the study area (See Part 1, Section 9 and Part 2, Section 1 for information on other predators). However, the decline in the red fox population did not explain all variation in reproductive output, as plover nesting density, temperature and habitat were also influential to chick and nest survival. The change in the effect of nesting density on both chick and nest survival once the mange variables were added in the models, suggests that there may be some interaction occurring between predation by foxes and nesting density as would be expected predation was density dependent. The effect of predator removal and exclusion has been documented in plovers and other similar taxa. For example, on the Missouri River, owl trapping was followed by increased chick survival and the data suggested predation on chicks was density dependent (Catlin et al. 2011). In an experimental landscape for northern lapwings (*Vanellus vanellus*), Rickenbach et al. (2011) found that chicks protected from ground predators had higher survival rates than unprotected chicks. Future monitoring of plovers on Fire Island may benefit from aerial predator monitoring in addition to continued terrestrial predator monitoring to be able to link reproductive output to other predator taxa. These results illustrate the importance of a long-term data set to understand temporal trends in vital rates, and the importance of comparing several potential drivers of vital rates into population models.

Density

Nest survival was positively correlated with nesting plover density, and this is likely because we had a positive trend in nest survival following Hurricane Sandy, and density increased in nearly all management areas over the same period. Chick survival, however, was negatively correlated with plover-nesting density. Plovers are territorial and compete for space during both the nesting and the brooding period (Elliot-Smith and Haig 2004). Higher plover nesting density can lead to intra-specific aggression, but can also aid in the ability of predators to develop a search image for abundant prey (Whitfield 2003, Hunt et al. 2019*a*). Although the decline in red foxes did not seem to increase depredation of nests and chicks by raccoons or feral cats, there are other possible predators on the island that may have developed a search image for shorebirds as population density increased. Moreover, an increase in density likely leads to an increase in plover interspecific strife

which is obvious to human observers and may lead to increase in detection probabilities by predators and reduced anti-predator vigilance by plovers. We observed an increase adult mortality by aerial predators (Part 1, Section 7) and this may have extended to chicks. However, chicks are smaller than adults making their carcasses difficult to detect and easier for a predator to swallow whole.

Habitat

Although restoration areas had lower nest survival than other areas, when all other variables were considered, being in a restoration area did not influence chick survival. In contrast, manipulated and natural overwashes had no relationship to nest survival, but chicks within manipulated and natural overwashes had higher survival. The differences observed between habitat types were likely influenced by local density, vegetation characteristics, and access to quality foraging habitat. Lower nest survival in the restoration areas may be due to specific management decisions regarding exclosure use. Specifically, the decision by Suffolk County Parks to remove exclosures in 2015 when an individual fox was entering exclosures, and to not exclose nests in 2018 following observed canine digging around exclosures. Both instances led to high predation rates of Smith Point nests, which included the restoration areas. Overwashes had greater invertebrate abundance than breach fills and restoration areas, which could have promoted higher chick survival. Further, manipulated overwashes in particular had lower nesting densities than breach fills and natural overwash, although they were higher than restoration areas (Bellman 2018).

Bay access for chicks seemed to decrease the home range size of broods, particularly in early years of the study. There could be several explanations for this finding as the distance to bay was not related to chick home range size. Broods may not need to move as much to access adequate forage on the bayside, because the invertebrate abundance is richer (Part 1, Section 8). Nesting density may also be greater in bayside areas, constraining the available space for broods to access (See Objective 1). That the difference in home range size between the habitat types was smallest in 2019 when density was greatest suggests that local density may be playing a role in such habitat effects. However, there was little difference in home range size by categorical habitat type. Chicks in restoration areas had larger home ranges and broods moved less in breach fills and more in natural overwashes. Assessing both descriptors (movement and home range size) of space use is thus important to understanding how plover broods use the landscape.

Cupsogue

The Cupsogue study site had to be removed from models in order to explicitly compare chick and nest survival between pre- and during the mange outbreak to after the mange outbreak, because we did not have information about mange in Cupsogue. Reproductive output was consistently lower at Cupsogue relative to the rest of the study area. Although we do not have an understanding of the predator community at Cupsogue, we have found that human disturbance is influential to chick survival in this system (DeRose-Wilson et al. 2018).

Cupsogue broods are likely contributing to lower chick survival in the breach fills as Cupsogue chicks make up about half of the breach-fill brood sample size. We would expect that the access to bayside foraging habitat at both breach fill sites would improve chick survival (Patterson et al. 1991, Elias et al. 2000, Cohen et al. 2009). However, in Cupsogue, high human use of the bayside in some years prevented chick from foraging in the preferred habitat. Moreover, higher plover nesting density in the breach fills may also contribute to lower reproductive output. Two-sided habitats (where dry sand spans from the ocean- to bay-side of an island), like the breach fills in this study, have been shown to have higher nesting plover density (Cohen et al. 2009), and the breach fills had some of the highest densities in our study area, particularly the Narrow Bay Breach Fill (Bellman 2018). In addition, least cost distance to ocean and bay both related positively to chick survival when all of the data were included, however removing Cupsogue dampened this effect. This suggests that at least part of the trend of higher survival farther from foraging habitat is driven by Cupsogue, where birds may survive better when they are in the symbolic fencing and away from beachgoers.

Overall, of the variables we looked at, the primary drivers of reproductive output seem to be nesting plover density and predators, specifically red foxes. However, human disturbance, which we did not include in our models, also has an effect on reproductive output (DeRose-Wilson et al. 2018). The direct estimation of population parameters, such as reproductive output and its constituent elements, nest and chick survival, are essential to understanding change in breeding populations of threatened species (Purvis et al. 2000). Reproductive output in our population contributes to population growth (Robinson et al. *in prep*), therefore it is essential that we try to maximize the reproductive output on Fire Island and Westhampton Island to maximize the rate at which the population grows toward the carrying capacity of the study area. Understanding brood home range size and movement also can lend to the understanding of reproductive dynamics in this population, as home range size and movement can be linked to survival or growth in some species (Lengyel 2006, Nichols and Kaiser 2009). In future analyses, we hope to assess chick habitat selection and behavior to obtain a more complete picture of the mechanistic drivers of reproductive output on Fire Island.

Section 7: Compare the relative contribution of local growth (natal recruitment) vs immigration in population growth associated with new habitats

Introduction

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

When habitats change following disturbance events, increasing the quantity and/or quality of that habitat and lessening density-dependent constraints, the composition of the breeding cohort may change, altering immigration and local recruitment rates. Broadly, a nesting population can be composed of 1) birds that returned to the area in which they previously nested (adult returners), 2) birds that immigrated from other areas (immigrants), and 3), yearlings that hatched in the population and returned to breed (natal recruits; Lampila et al. 2006, Million et al. 2019). Newly created habitat may change the contribution from each cohort, such as by an increase in immigrants or natal recruits, which can settle due to newly opened territories.

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

driving population change, and 3) assessed adult and hatch-year survival to determine if increased survival could be leading to observed population change.

Methods

Immigration and recruitment

In our study population, we were able to band most chicks and adults in each year; therefore, we estimated immigrants using the number of newly banded adults in a year. We were not able to confirm each bird as a nester in each year due to trap aversion or nests in denser vegetation, particularly in 2019. Here, we assume here that our detection was not biased towards any specific nesting class. If individuals that hatched in the study population returned to breed 2+ years after hatching, we counted them as natal recruits in the year of first breeding. Although a population is likely to have floater adults that could not obtain either a mate or a breeding territory, they do not contribute to the reproductive output in a population and were not included in annual pair counts, and thus they were not included here.

Using banded and known-age birds, we estimated immigration and natal recruitment for 2014–2019. We estimated immigration rate using a Pradel (1996) model in Program Mark (White and Burnham 1999). The parameters estimated from the Pradel model are apparent survival, resight rate, and fecundity rate. We interpreted fecundity as an estimate of immigration rate (# immigrants in $t+1$ per individual in year t), which included natal recruits and immigrants. To estimate annual immigration rates, we fit a single model holding apparent survival and resight rate constant, as prior studies indicated that adult survival and detection probability were relatively constant, (Catlin et al. 2015, Weithman et al. 2019). For immigration rate in 2014 and 2015, the estimate included only the eastern portion of the study area (Old Inlet West to Cupsogue). Starting in 2016, estimates of immigration rate also included the Democrat Point site, as birds were first banded there in 2015 and thus we could first estimate the breeding cohort composition in 2016. 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 (breach fill, manipulated overwash, natural overwash, restoration area, and other, See Part 1, Table 1 for details on habitat definitions), created following Hurricane Sandy.

To determine if immigrants, natal recruits, and returning nesters selected for or against the habitat types, we compared percent area of each habitat type available and the percent each habitat type used by the classes of nesting plovers. To estimate amount of available habitat, we included patches of dry sand in the eastern study area (Old Inlet West to Cupsogue) of at least $1m^2$ following the same methods in Part 2, Section 3. We only conducted this analysis for the eastern study area because we had data for each year of the study, and habitat

types were only located in this part of the site. We then determined the amount of dry sand used the within each of the habitat areas, as defined by delineated ocean-to-bay overwash and USACE-provided shapefiles. We calculated the proportion of habitat use as the proportion of returning adult nesters, natal recruits, or immigrants in each habitat type. We conducted a chi-squared test to determine if there was overall selection occurring within nesting classes and compared 95% confidence intervals on proportion of use to availability to determine selection for or against particular habitat types (Byers et al. 1984, Catlin et al. 2011). If a breeding class did not nest in a habitat type within a year, we interpreted it as avoidance. We completed this analysis for each year, 2014–2019, to evaluate if selection changed as plover dynamics and island characteristics changed.

Survival and site fidelity

To estimate true survival of piping plovers in the Fire Island study area, we used the Barker live-dead encounter model (Barker 1997). The Barker model uses supplemental (off-site) resightings to distinguish permanent emigration from mortality for a more accurate estimate of survival than is available when just considering return rate. Our primary interest for this report was in true survival (*S*) of first year (hatch to first year b) and adult (after first breeding season) piping plovers. Other parameters that we estimated using this model were detection probability (p), probability of being reported dead during the supplemental period (*r*), the probability of being resighted during and surviving the supplemental period (*R*), the probability of being resighted during the supplemental period and dying (*R'*), site fidelity (*F*), and the probability of returning to the study area after emigration (*F'*). 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. 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 Part 2, Section 6 and our annual estimates of first-year survival from the Barker model to derive post-fledging survival, survival from fledging until the following year. Post-fledge survival is first-year survival divided by pre-fledge survival (Catlin et al. 2015, Weithman et al. 2019).

Results

Immigration and recruitment

From 2014 to 2019, the breeding population included 31–53% returning nesters annually (Table 21). The remainder of the birds that nested in the study area were unbanded birds that were captured during the year (immigrants; 34–63%) or that hatched in the study area the year before (natal recruits; 6–25. The Pradel model resulted in estimates of combined natal recruitment and immigration between 0.53 and 0.87 immigrants in year *t+1*, per individual in year *t*.

Breach fills made up the smallest proportion of the eastern study area, ranging from 2.6% to 3.2% of the study area. Restoration areas made up 10.3% to 11.0% of the dry sand in the eastern study area, natural overwashes made up 10.3 to 15.4% of the dry sand, and manipulated overwashes made up 15.1 to 21.4% of the available dry sand. Other habitats made up the greatest proportion of the available dry sand in the eastern study area, ranging from 51.4% to 61%.

In all years combined, 72% of returners nested in newly created habitats (Table 22). They nested in manipulated overwashes (27%) and natural overwashes (25%) the most, followed by breach fills (12%) and restoration areas (9%). The remaining 28% of returners nested in "other" habitat (Table 22). Returners exhibited selection for or against specific habitat types most frequently out of the three nesting classes (Figure 1). Returners selected breach fills in 2015 and 2017, manipulated overwashes in 2016 and 2018, natural overwashes from 2017–2019, and restoration areas in 2019. Returners avoided "other" habitat in all years (Figure 27).

Natal recruits made up the smallest of these three cohorts (6-25% over the years, Table 1). In all years combined, 58% of recruits nested in newly-created habitats (Table 21). They nested in restoration areas (29%) and manipulated overwashes (13%) the most, followed by breach fills (9%) and natural overwashes (8%). The remaining 42% of recruits nested in "other" habitat (Table 23). Overall, recruits nested in restoration areas in every year they were available and more than expected based on availability in 2015, 2018, and 2019. Recruits avoided nesting in breach fills in 2017 and 2018. No natal recruits nested in breach fills in 2014, 2015, and 2018, manipulated overwashes in 2016, natural overwashes in 2014, 2015, and 2019, and "other" habitat in 2015 (Figure 27).

In all years combined, 47% of immigrants nested in newly created habitats (Table 21). Immigrants nested in manipulated overwashes (18%) and natural overwashes (13%) the most, followed by restoration areas (12%) and breach fills (5%). The remaining 52% of immigrants nested in "other" habitat (Table 24). Based on availability, we would have expected more immigrants to nest in "other" habitat every year except 2019. In 2019, immigrants nested in "other" habitat proportional to its availability. Except for "other" habitat, which was avoided, immigrants nested with a frequency we would expect based on availability in every habitat type across all years except for breach fills in 2019, which immigrants avoided (Figure 27). See Figures D-1–D-6 for year-specific selection plots and statistical tests.

Table 21. Nesting adult piping plovers divided into the categories of returning nesters, local recruits, and immigrants. Numbers for 2014 and 2015 only include the eastern study area (Fire Island National Seashore Wilderness Area to Cupsogue Beach County Park). Numbers for 2016–2019 include the entire study area (Robert Moses State Park to Cupsogue Beach County Park).

Year	Returning Nesters	Natal Recruits [a]	Immigrants [b]	Immigration Rate (± SE) [c]
2014	17 (47%)	2 (6%)	17 (47%)	0.62 (0.20)
2015	20 (53%)	5 (13%)	13 (34%)	0.52 (0.16)
2016	21 (31%)	4 (6%)	42 (63%)	0.87 (0.18)
2017	32 (38%)	16 (19%)	36 (43%)	0.67 (0.12)
2018	46 (43%)	17 (16%)	44 (41%)	0.65 (0.11)
2019	60 (41%)	36 (25%)	50 (34%)	0.75 (0.11)

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

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

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

Table 22. The number of returners that nested in new habitats on Fire Island and at Cupsogue Beach County Park following Hurricane Sandy and subsequent habitat alteration.

Year	Total # Returners	Natural Overwash	Manipulated Overwash	Breach Fill	Restoration Area	% Returners in other, pre-Sandy Habitats	% Returners in New Habitats
2014	17	4 (24%)	7 (41%)	0	NA	6 (35%)	65%
2015	20	3 (15%)	7 (35%)	5 (25%)	1 (5%)	4 (20%)	80%
2016	21	6 (29%)	8 (38%)	4 (19%)	0	3 (14%)	86%
2017	32	8 (25%)	7 (22%)	7 (22%)	2 (6%)	8 (25%)	75%
2018	46	15 (33%)	12 (26%)	3 (7%)	3 (7%)	13 (28%)	72%
2019	60	13 (22%)	12 (20%)	4 (7%)	11 (18%)	20 (33%)	67%
Total	196	49 (25%)	53 (27%)	23 (12%)	17 (9%)	54 (28%)	72%

Table 23. The number of natal recruits in the population that nested in new habitats on Fire Island and at Cupsogue Beach County Park following Hurricane Sandy and subsequent habitat alteration. Recruits were banded as chicks in the year prior and returned to breed in the study area.

Year	Total # Recruits	Natural Overwash	Manipulated Overwash	Breach Fill	Restoration Area	% Recruits in other, pre-Sandy Habitats	% Recruits in New Habitats
2014	2	0	1 (50%)	0	NA	1 (50%)	50%
2015	5	0	2 (40%)	0	3 (60%)	0	100%
2016	4	1 (25%)	0	1 (25%)	1 (25%)	1 (25%)	75%
2017	16	3 (19%)	3 (19%)	3 (19%)	3 (19%)	4 (25%)	75%
2018	17	1 (6%)	2 (12%)	0	7 (41%)	7 (41%)	59%
2019	36	1 (3%)	2 (6%)	3 (8%)	9 (25%)	21 (58%)	42%
Total	80	6 (8%)	10 (13%)	7 (9%)	23 (29%)	34 (42%)	58%

Table 24. The number of individuals that immigrated into the population and nested in new habitats on Fire Island and at Cupsogue Beach County Park following Hurricane Sandy and subsequent habitat alteration.

Year	Total # Immigrants	Natural Overwash	Manipulated Overwash	Breach Fill	Restoration Area	% Immigrants in other, pre-Sandy Habitats	% Immigrants in New Habitats
2014	17	3 (18%)	6 (35%)	2 (12%)	NA	6 (35%)	65%
2015	13	1 (8%)	5 (38%)	1 (8%)	3 (23%)	3 (23%)	77%
2016	42	5 (12%)	7 (17%)	1 (2%)	4 (10%)	25 (60%)	40%
2017	36	8 (22%)	6 (17%)	3 (8%)	7 (19%)	12 (33%)	67%
2018	44	2 (5%)	7 (16%)	3 (7%)	6 (14%)	26 (59%)	41%
2019	50	7 (14%)	5 (10%)	0	2 (4%)	36 (72%)	28%
Total	202	26 (13%)	36 (18%)	10 (5%)	22 (11%)	108 (53%)	47%

Returners

	2014	2015	2016	2017	2018	2019
Breach Fill	0	+	=	+	=	=
Manipulated Overwash	=	=	+	=	+	=
Natural Overwash	=	=	=	+	+	+
Restoration Area	NA	=	0	=	=	+
Other	–	–	0	0	–	–

Natal Recruits

	2014	2015	2016	2017	2018	2019
Breach Fill	0	0	=	=	0	=
Manipulated Overwash	=	=	0	=	=	=
Natural Overwash	0	0	=	=	=	–
Restoration Area	NA	+	=	=	+	+
Other	=	0	=	–	–	=

Immigrants

	2014	2015	2016	2017	2018	2019
Breach Fill	=	=	=	=	=	0
Manipulated Overwash	=	=	=	=	=	=
Natural Overwash	=	=	=	=	=	=
Restoration Area	NA	=	=	=	=	=
Other	–	–	–	–	–	=

Figure 27. Selection by returners, natal recruits, and immigrants in each habitat type. Plus signs represent more nesters than expected in a habitat based on availability. Minus signs represent fewer nesters than expected in a habitat based on availability. If a habitat had no nesters in it that year, they were given zeroes. Restoration areas were not created until 2015, so were denoted NA in 2014. Blue indicates avoidance, with the lightest blue indicating no use whereas slightly darker blue indicates statistical avoidance, yellow indicates selection for a habitat type, and grey indicates use in proportion to what is available. See Figures D-1–D-6 for year-specific selection plots and statistical tests.

Survival and site fidelity

The top model from the survival and fidelity analysis indicated temporal variation in survival for first-year birds, a constant adult survival rate for all years, and an age effect on site fidelity (F) (Table D-1). Point estimates of first-year survival varied each year of the study, with high years of first-year survival relating to years of high chick survival (Figure 28, see also Part 3, Section 6). Adult survival (S = 0.69 ± 0.02) was within the range of past studies of adult piping plover survival and is within the confidence intervals of our estimate from 2013–2018 (Hermanns et al. 2018). Site fidelity for adult birds (F = 0.92 ± 0.02) was higher than for first-year birds (F = 0.76 ± 0.04), and first-year site fidelity increased with the addition of the 2019 data. Detection probability was constant over both ages and across all years (p = 0.96 ± 0.02). Post-fledgling survival (hatch to

fledging), survival from fledging to returning the following year, varied among years (Figure 29). We observed the highest rate of post-fledgling survival in 2013 fledglings.

In 2018, six banded adult mortalities were confirmed either by discovering carcasses of adults or by procuring bands or flags from a predator scat sample collected during field surveys. All mortalities were identified as predation. At least one mortality was caused by a peregrine falcon (*Falco peregrinus*) and one mortality was later confirmed to be caused by a cat (*Felis catus)* via mitochondrial DNA sequencing of the scat sample we collected (Part 3, Section 5). In 2019, we confirmed two adult mortalities in our study area. The flag leg of one adult was found outside of an exclosure and was likely depredated by a peregrine falcon based on field sign. We discovered evidence that at least one peregrine falcon was frequenting the area and using exclosures as perches, as several exclosures had bones, feathers, and entrails in the vicinity. The carcass of another banded adult was found in vehicle tire tracks on the driving portion of open beach in NBW. The cause of mortality was not determined, but following its discovery, a nest within 250 m was abandoned. Unfortunately, the adult was not confirmed to the nest so we cannot be certain that the two incidents are related.

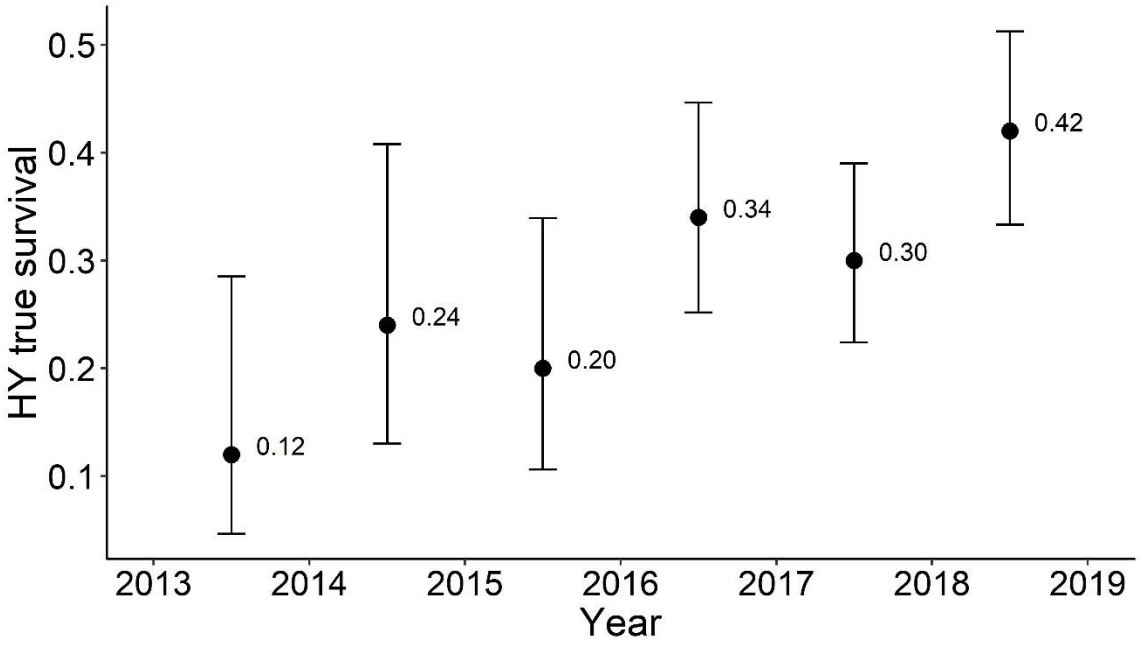


Figure 28. True survival (S) of first year (from hatch to year 1) piping plovers on Fire Island from 2013 to 2019. Estimates are from the top model with an additive effect of year and age. Error bars represent 95% confidence intervals.

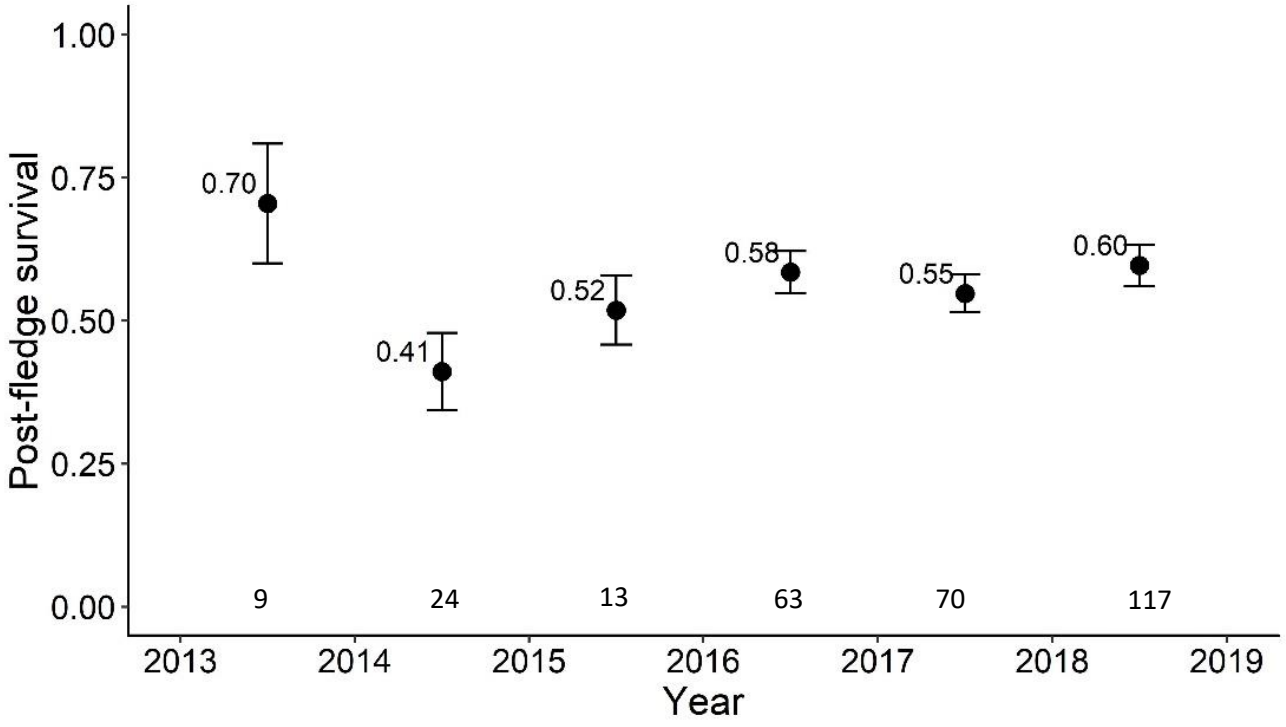


Figure 29. Post-fledge survival of piping plovers on Fire Island, New York from 2013 to 2018. Post-fledge survival is the survival from fledgling to returning the following year, and is calculated by dividing first year survival by pre-fledge chick survival. Error bars represent 95% confidence intervals. Numbers above x-axis represent number of fledglings.

Discussion

Here, we detected patterns in plover nest locations that illustrated selection and avoidance of certain habitat types by comparing the choices that adult plovers made when nesting in the study area compared to the amount of each habitat that was available on the landscape. Habitats such as backshore and barrier interior that were classified as "other" for these analyses, and were present before Hurricane Sandy, made up the greatest proportion of available plover nesting habitat based on our dry sand classification. However, during this study, birds nested in pre-Sandy areas at lower proportions when compared to the newly created habitat in each year. When combined, natal recruits, immigrants, and returners nested in habitats created since Hurricane Sandy nearly 60% of the time and only nested in "other" habitat 40% of the time despite the fact that "other" comprised 51–61% of the available nesting habitat suggesting that plovers preferred these newly- created habitats.

All three classes of nesting piping plovers avoided "other" habitats or selected for the habitats created following Hurricane Sandy, relative to what was available. Many of these areas were overwashed, or were created to appear like overwash and, thus, had greater backshore widths and wider, open sand nesting habitat. Open sandy habitat, like that created by Hurricane Sandy, can assist nesting plovers in detecting predators and

can play a role in the camouflage of their eggs and chicks (Fraser and Catlin 2019). Alternatively, habitat fragmentation and being closer to dense vegetation may make the pre-Sandy habitats less attractive to all classes of nesting birds. These pre-Sandy areas also have less distance between the high tide line and the toe of the dune, which makes nests more susceptible to flooding compared to all of the post-Hurricane Sandy habitats. Although for this analysis, we considered every patch of dry sand greater than 1-m^2 available, some of this habitat may be unsuitable to nesting birds, however much of the unsuitable habitat is likely to be the pre-Sandy "other" habitat (Walker et al. 2019).

In 2016, we observed the highest immigration rate of all the three years, although the count of immigrants may be a slight over-estimate due to a small number of birds that did not get banded in the year prior due to early nest failure or trap aversion, although overall there were few to no unbanded adults in each year. This relatively high immigration rate may have been due to low nest and chick survival in 2015, causing adults to not return to nest on Fire Island and opening up habitat for new nesting birds, especially as 2016 had the lowest proportion of returning nesters and natal recruits in the population. However, the proportion of the population that was composed of immigrants, and the proportion of immigrants in new habitats has declined in recent years. This may suggest that the Fire Island and Westhampton Island piping plover population is nearing the carrying capacity of its habitat, resulting in less available space for new birds. The new habitats may also be reaching their carrying capacity.

Overall, immigration greatly contributed to the 183% population increase on Fire Island and Westhampton Island since prior to Hurricane Sandy (Part 2, Section 3). Those immigrants generally selected storm-created and human-created habitats (breach fill and restoration areas) in proportion to their availability and generally avoided "other" habitats (largely ocean backshore). Increasing nesting density likely decreases the habitat available habitat to immigrants and natal recruits, and therefore it is possible that breach fills had become too dense for new birds to obtain territories while returning adults were remaining faithful to their nest sites, which could be why we did not observe any immigrants nesting in the breach fills in 2019. Likely, the primary way to increase and maintain immigration into the population is to maintain open, wide, sparsely vegetated sand, as immigrants seem to use it in proportion to what is available.

Similarly, on the Missouri River, immigration declined as nesting density increased (Catlin et al. 2015). On the Outer Banks of North Carolina, the small population in the southern extent of the Atlantic Coast range, population size is likely sustained by immigration (Weithman et al. 2019). In the Great Lakes plover population, immigration correlated with population growth, although less so than survival and fecundity (Saunders et al. 2018). This difference is likely due to the spatial scale differences between the Great Lakes population, which is an entire sub-population, as opposed to Fire Island and Westhampton Island, which is only a small portion of the Atlantic Coast population.

Natal recruits, or birds that hatched in our study area that returned the following year, nested in each habitat, every year, almost as much as we would expect based on area of habitat. Natal recruitment tended to be variable and higher following years of higher chick survival. Fledglings prospect for potential nesting sites prior to departing for the wintering grounds (Davis et al. 2017), thus natal recruits habitat selection could have been driven by observations of high reproductive success and conspecific adults selecting for post-Sandy habitats during their hatch year. Similar to what we have seen on Fire Island, across avian taxa, the contribution of immigration to population growth generally is higher than the contribution of natal recruitment (Million et al. 2019).

That we found adult survival to be constant among years is generally consistent with many avian populations (Saether et al. 2002, Lampila et al. 2006). Sufficient adult survival, which is usually found to have high sensitivity relative to population change (Calvert et al. 2006), is unlikely to be allowing the near doubling of the population in the Fire Island and Westhampton Island study area. Hatch-year survival generally tracks the trends for pre-fledging chick survival (Part 2, Section 6), suggesting that pre-fledging chick survival is primarily driving the annual variation in this parameter. For this report, we have used both a different model and a differing model structure to estimate chick survival, and thus derived estimates of post-fledgling survival vary from previous reports, in particular the 2013 estimate. Years with low sample size, such as 2013, are vulnerable to being altered by the overall trend in the data, which is overall higher chick survival. We also changed this model such that we use a 5-day encounter window, and use the number of eggs that hatched rather than the number of chicks ever observed as the first encounter. Only using the chicks observed certainly biased chick survival high in the past.

A population experiencing exponential growth, such as Fire Island, is likely to have very different dynamics than stationary populations (Schaub et al. 2013, Million et al. 2019). We currently are working to develop an integrated population model, which jointly estimates all of the values within this section. However, limited data preclude us from estimating annual variation in some rates, particularly adult survival. This model suggests that on Fire Island following Hurricane Sandy, immigration is the primary correlate with population growth, although reproductive output also is positively associated with population change (Robinson et al. *in prep*). Future monitoring of this population is needed to understand variation in the contribution to population change of each demographic parameter as the population approaches carrying capacity, and as vegetation increases and suitable habitat declines.

Section 8: Compare prey densities at several habitat types

Introduction

It is important to understand the effect of hurricanes and other environmental perturbations on threatened and endangered species. Disturbance on the landscape can manifest in changes to both the biotic and abiotic environment and can occur at fine and broad scales. Barrier islands, as narrow, dry sand islands that lay parallel and close to the mainland are particularly susceptible to environmental changes. Following an environmental event to a barrier island, such as a hurricane, modifications such as dune building often are conducted to protect the mainland from flooding in future events or to improve the beach for recreational purposes after hurricane-related changes. These natural and engineered changes can affect wildlife that are native to these ecosystems in both direct (e.g., loss or creation of nesting habitat space) and indirect (e.g., changes in food availability and distribution) ways.

Breaches and overwashes, such as those resulting from Hurricane Sandy, can create intertidal flats that serve as foraging habitat for shorebirds (Cohen et al. 2009). Breaches and overwashes can alter the invertebrate prey base that was present in an area or open up new areas for colonization by invertebrates. However, piping plovers' invertebrate prey base also can be altered when sand is placed in areas to stabilize barrier islands and when island breaches are filled (Speybroeck et al. 2006, Leewis et al. 2012), as was done on Fire Island following Hurricane Sandy with the Fire Island to Moriches Inlet stabilization project. Due to the federally threatened status of piping plovers, the effect of anthropogenic changes to their prey base needs to be assessed and, where appropriate, mitigated.

Piping plovers, which breed on barrier islands, forage on terrestrial and marine invertebrates by pecking for prey on the surface of sediment, probing shallowly into the substrate, and occasionally by gleaning invertebrates on vegetation (Shaffer and Laporte 1994, Cuthbert et al. 1999, Majka and Shaffer 2008, Cisek 2013). The variety of foraging methods which plovers exhibit means that they likely prey on a diverse range of taxa. The quantity of prey available in habitats can determine habitat quality and influence piping plover nesting densities (Elias et al. 2000, Cohen et al. 2009). 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). Natal territories with available high-quality foraging habitat can contribute fitness advantages to offspring (Catlin et al. 2014). Specifically, plover chicks with access to high densities of terrestrial invertebrates weigh more, have higher foraging rates, and are more likely to survive (Loegering and Fraser, 1995, Cohen et al. 2009, Catlin et al. 2015). Habitat changes resulting from anthropogenic and natural forces have an effect on plover population dynamics, and changes in prey abundance may contribute to the effect (Cohen et al. 2009, Catlin et al. 2015). Plover prey sampling in the pre-laying season has indicated that benthic prey are more available than terrestrial prey species (Cohen and Fraser 2010). Therefore, determining

whether any of the post-Hurricane Sandy habitat alterations support better piping plover prey communities may improve the overall population by suggesting how to create or modify habitat in the future.

Methods

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 2014–2019 and 2019 alone (see Part 2, Section 1 for field methods). We 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 2014–2019. 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 shapefile 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 (Houghton 2005, Cohen et al. 2009, Anteau and Sherfy 2010).

Results

During 2014–2019, we collected 2591 invertebrate samples, of which 1305 (50.4%) contained at least one invertebrate. During 2014–2019, 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 = 233.95, P < 0.001; Table 1). Across the whole study, bay beach samples had nearly 4x as many invertebrates caught over the 30-minute sample period. Similarly, in 2019, there was a significant difference in the number of invertebrates collected among the three different geologic locations (K-W= 15.04, df = 2, $P < 0.001$). In 2019, invertebrate samples in the bay and ocean beaches in 2019 had a greater mean arthropod catch than barrier interior invertebrate samples, but they did not differ from one another as in the overall dataset (Table 25).

Table 25. Mean number of invertebrates captured during a 30-minute period at bay beach, ocean beach, and barrier interior habitats on Fire Island and Westhampton Island, May through August 2014–2019.

Geologic Location	Mean Arthropods/ 30min	SE	N
Bay Beach	4.10	0.36	537
Barrier Interior	1.06	0.077	1363
Ocean Beach	1.64	0.36	594

Table 26. 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 2019.

Geologic Location	Mean Arthropods/ 30min	SE	N
Bay Beach	3.46	0.87	59
Barrier Interior	1.39	0.31	214
Ocean Beach	3.48	1.89	111

Within habitat types, ephemeral pools and the bay beach had greater projected invertebrate catches than the ocean beach or barrier interior (Table 26). For the treatment types, overwashes had greater invertebrate catches than the restoration areas and the breach fills (Table 26, Figure 30). Although the invertebrate catch rate was high in ephemeral pools, ephemeral pools were rare, as indicated by the small sample size (Figure 30).

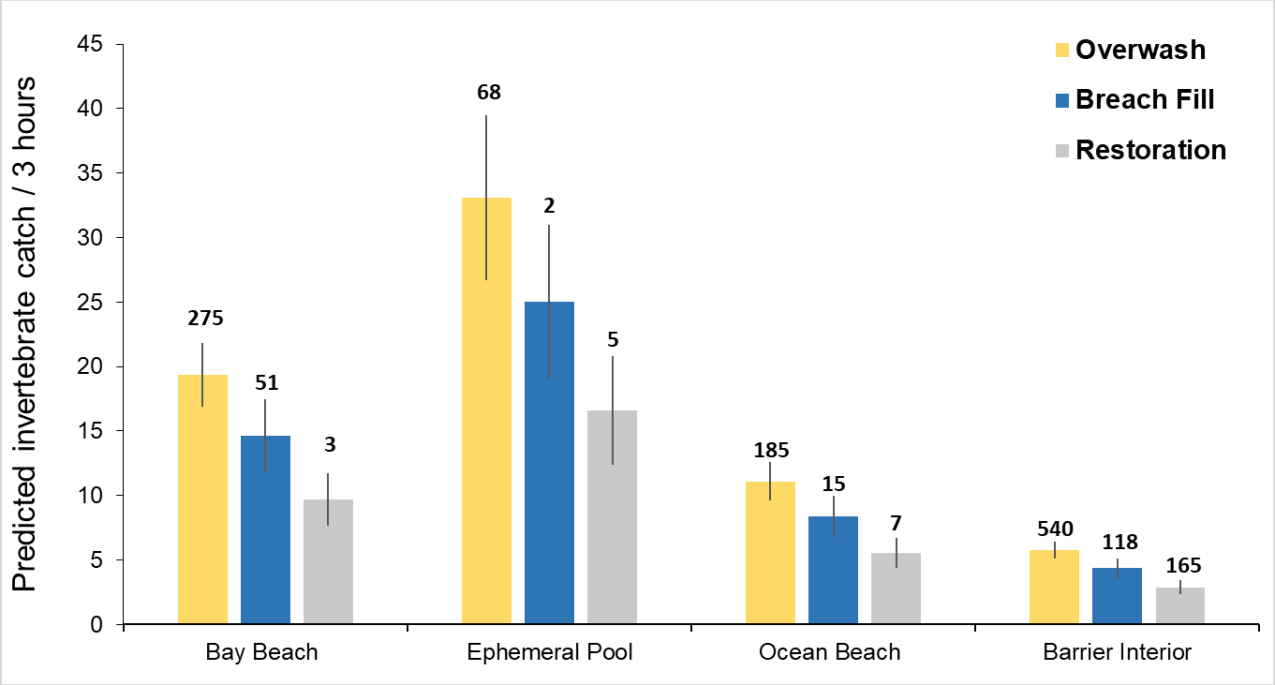


Figure 30. Predicted invertebrates catch over a projected three-hour period in various habitats from 2014–2019. We predicted invertebrate catch using a negative binomial mixed model. Invertebrate samples were collected over 30-minute periods and projected to 3-hours to facilitate comparison to similar 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 habitat availability.

In 2013 and 2015, substrate invertebrates also were collected in the Fire Island and Westhampton Island study area. The results of this study can be found in Appendix E.

Discussion

Understanding the availability of prey resources in relation to bird use is an important piece of assessing habitat quality. From terrestrial invertebrate samples on Fire Island and Cupsogue during 2014–2019, our results indicate what past studies also have indicated, that low-energy moist habitats, such as bay beach and ephemeral pools have higher invertebrate abundance than interior and ocean intertidal habitats (Loegering and Fraser 1995, Elias et al. 2000). Contrary to some studies, we found that the ocean intertidal habitat was generally more productive than interior habitats (Loegering and Fraser 1995), but invertebrate availability in the intertidal likely varies with slope, elevation, and wrack availability, which we did not evaluate here. This study also indicated that more terrestrial invertebrates were caught in overwashes compared to breach fills and restoration areas.

More invertebrates in the overwashes is particularly interesting, as the largest overwash, Old Inlet East, has overwashed multiple times since Hurricane Sandy, and it might be expected that this would continually disturb the invertebrate community (McCabe and Gotelli 2000). However, substrate invertebrates may be more disturbed by frequent overwashing than terrestrial invertebrates, such as Dipterans and Coleopterans, which can fly and recolonize an area quickly. The restoration areas did not have intertidal areas included in their design polygons, and we only included areas within the design footprint in our evaluation, which likely drove the lower predicted invertebrate abundance there. Future evaluations of the restoration areas should also consider sampling adjacent, accessible inter-tidal zone even if not explicitly in the design criteria.

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

Section 9: Quantify the interactions of predators with shorebird nests and chicks

Introduction

Predator disturbance has been shown to increase the probability of nest abandonment (Doherty 2011), thus, even in areas implementing nest exclosures, predators can cause nest failure through harassment and predation of incubating adults (Roche et al. 2010). Disturbance also can result in a change in behavior due to the perceived risk of predation (Frid and Dill 2002). In the case of plovers and other shorebirds, disturbance can influence the incubating or brooding behavior of adults and foraging behavior in chicks, both of which can have a negative influence on chick survival (Weston and Elgar 2005, Hillman et al. 2015, DeRose-Wilson 2018). Furthermore, soon after chicks hatch and are mobile, they no longer benefit from the protection of nest exclosures.

For plovers on the Atlantic Coast, many nests are protected by exclosures to prevent depredation (Melvin et al. 1992) and therefore nest survival is typically high (Catlin et al. 2015, Cohen et al. 2016) as large predators are limited in their ability to access nests. Despite this, it is still vital to understand the factors that influence potential nest and adult disturbance to gain a better perspective on nest protection and disturbance reduction, especially for aerial predators or predators that are small enough to fit through the fencing of nest exclosures (Murphy et al. 2003, Neuman et al. 2004).

Remote cameras are valuable tools for collecting data on wildlife interactions that often are difficult to observe directly (Trolliet 2014). Cameras are relatively non-invasive and can collect photo data throughout the day and night, recording interactions that occur when direct observation is difficult or impossible (Murphy et al. 2017, Hunt et al. 2019*b*). The effect of disturbance of humans and predators on wildlife species is well-understood (Gill et al. 1996, Ciuti et al. 2012, Suraci et al. 2019), however, using cameras to capture unobservable wildlife interactions can provide valuable insight into potential opportunities for management to reduce disturbance to threatened species. In this objective, we used remote cameras to quantify the interactions between predators and exclosed plover nests and to identify potential sources of disturbance to nests and chicks. Although we did not directly assess disturbance in this objective, with the information gathered from our remote cameras, we hoped to gain perspective of the local predator community and other sources of potential disturbance that may be affecting the fate of plover nests and chicks in our study area. Moving forward, this information can help focus management decisions on strategies better designed to protect nests and chicks from disturbance detected by our cameras.

Methods

We installed remote cameras (Reconyx Rapidfire, RC60 Covert IR, Holmen, WI, USA) on nest exclosures in 2016–2019, as early as May 13 and as late as July 7. Cameras were placed on nests of varying age, as we did not place cameras on nests until both incubating adults were captured. Cameras were set to capture one photo every five minutes in addition to capturing three photos in a burst whenever the motion sensor was triggered with a one second cooldown period between bursts. After installing a camera on the outside of an exclosure, we ensured the camera was accepted by the incubating adults using similar methods that managers use to confirm exclosure acceptance. If an adult did not return to incubate within 30 minutes, we removed the camera. We checked camera batteries and SD memory cards opportunistically, often during nest checks to reduce the time spent at a nest, thus the interval between each camera check varied. We removed cameras from exclosures once the nest hatched and the brood left the nest location. Therefore, the time each camera was on an exclosure varied by nest.

We used 30-minute detection periods to analyze the camera data. This allowed us to distinguish between different capture events for non-uniquely identifiable animals. For instance, if a white-tailed deer lacking identifiable features was captured in a photo and then left the frame, any white-tailed deer seen within 30 minutes of the first was assumed to be the same animal unless the two were distinguishable from one another and, thus, would only be counted once. Additionally, if a white-tailed deer remained in the camera frame for 31 minutes, it would count as two separate white-tailed deer capture events. Individuals detected on camera, but not included in the results, were plovers, plover monitors, and plover monitor vehicles. For more information regarding the interaction between plovers and red fox, 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 3, Section 3).

Results

In 2016, we placed 14 remote nest cameras across the study area. Cameras were present on exclosures between three and 34 days ($\bar{x}$ = 16.80, SE = 2.12). Red fox (*Vulpes vulpes)*, crow sp. (*Corvus sp.*), and unknown bird sp. had the highest number of capture events (Table 27). The species with the lowest number of capture events included raccoon (*Procyon lotor*), willet (*Tringa semipalmata*), unknown mammal, and grackle (*Quiscalus sp.*) with one capture event each (Table 28).

In 2017, we placed 27 remote nest cameras across the study site. Cameras were present on exclosures between two and 26 days ($\bar{x}$ = 12.07, SE = 1.02). Gulls (*Larus sp.*), humans, white-tailed deer (*Odocoileus virginianus*), and crab sp. had the highest number of capture events (Table 27). The species with the lowest number of capture events were great-horned owl (*Bubo virginianus*), eastern cottontail (*Sylvilagus floridanus*), and unknown rodent sp. with one capture event each (Table 28).

In 2018, we placed 27 remote nest cameras across the study site. Cameras were present on exclosures between one and 40 days ($\bar{x}$ =14.67, SE = 1.69). Unknown bird sp., humans, and gulls had the highest number of capture events (Table 27). The species with the lowest number of capture events were eastern cottontail, semi-palmated plover (*Charadrius semipalmatus*), and peregrine falcon (*Falco peregrinus*) with one capture event each (Table 28).

In 2019, we placed 19 remote nest cameras across the entire study site (Table 27). Cameras were present on exclosures between five and 29 days ($\bar{x}$ = 12.42, SE = 1.39). Humans, white-tailed deer, and toad (*Anaxyrus sp.*) had the greatest number of capture events at 58 and 30, respectively. The species with the lowest number of capture events were mourning dove (*Zenaida macroura*) and barn swallow (*Hirundo rustica*) with one capture event each (Table 28).

During 2016–2019, we placed 86 remote nest cameras across the study site. Cameras were present on exclosures between one and 40 days ($\bar{x}$ = 13.71, SE = 0.79). When combined, humans (non-monitors) had the greatest number of capture events, followed by gulls and unknown birds. When grouped by class, avian taxa had the greatest number of capture events followed by humans, non-human mammals, toads, crabs, and unknown capture events (Table 29). In 1,193 total days of camera trapping, footage confirmed no predation events and only one instance of a potential mammalian predator entering an exclosure. In 2017, a rat, likely a Norway rat (*Rattus norvegicus*) based on a small mammal inventory done on Fire Island in 2002 (USACE 2002), was pictured inside of the exclosure but not depredating the eggs. There were 27 capture events at two separate nests of ghost crabs (*Ocypode quadrata*), and in most of the pictures, they were seen inside of the exclosures. Toads, likely Fowler's toads (*Anaxyrus fowleri*), were captured on cameras 60 times throughout the study, and were often inside of exclosures. Examples of photo captures from nest cameras can be found in Figure 31.

Table 27. Number of nest cameras placed in each management unit per year and the three species with the greatest number of capture events each year.

Year	Management Unit	Number of Cameras Placed	Trap Nights	Species with Highest Number of Capture Events
2016	Smith Point County Park Cupsogue Beach County Park Fire Island National Seashore Robert Moses State Park	2 2 6 4	235	Crow sp. and red fox (19) Unknown bird sp. (15) Toad sp. (14)
2017	Smith Point County Park Cupsogue Beach County Park Fire Island National Seashore Robert Moses State Park	10 3 8 6	326	Gull sp. (383) Human[a] (277) Crab sp. and white-tailed deer (27 each)
2018	Smith Point County Park Cupsogue Beach County Park Fire Island National Seashore Robert Moses State Park	11 4 3 9	396	Unknown bird sp. (407) Human[a] (181) Gull sp. (85)
2019	Smith Point County Park Cupsogue Beach County Park Fire Island National Seashore Robert Moses State Park	9 1 6 3	236	Human[a] (58) White-tailed Deer (30) Toad sp. (22)

[a]Only non-plover monitor humans were counted

Table 28. Number of nest camera capture events per year and subsite, Fire Island and Westhampton Island, 2016–2019. See Table 3 for subsite descriptions.

Year	Subsite(s) of Nest Cameras with Capture Events	Capture Event ID	Number of Capture Events
2016	DMP, OSS	Red fox	19
	OIE, BRO	Crow sp.	19
	BRO	Unknown Bird sp.	15
	DMP, OSS, OIE	Toad sp.	14
	DMP, BRO	White-tailed Deer	13
	DMP, OSS, CSE	Human[a]	12
	DMP, BRO, CSE	Gull sp.	8
	OSS, OIE, BRO	Unknown	7
	NBE, SPTE, CSW	Red-winged Blackbird	6
	OSS, OIE	Northern Mockingbird	3
	BRO	Raccoon	1
	BRO	Willet	1
	BRO	Grackle sp.	1
	SPTE	Unknown mammal sp.	1
2017	CSW, SPTE, DMP, FLD4, OSS	Gull sp.	383
	FLD3, FLD4, OSS, DMP	Human[a]	277
	OIE, SPTE, STP, DMP, OSS, FLD4	White-tailed Deer	27
	OSS, DMP	Crab sp.	27
	DMP, OIE, NBW	Toad sp.	10
	OIE	Least Tern	10
	OIE, PSO, CSW, SPTE, FLD4	Unknown Bird sp.	9
	STP, NMA, OSS, DMP, FLD4	Red fox	8
	DMP, STP, CSW	Crow sp.	7
	DMP	American oystercatcher	5
	OIE, SPTE, NMA	Raccoon	4
	DMP, OIE, CSE	Unknown	3
	CSW	Common Tern	3
	OIE	Unknown mammal sp.	2
	BRO	Great-horned Owl	1
	CSE	Eastern Cottontail	1
	CSW	Rodent sp.	1
	DMP	Black Skimmer	1
2018	OIE, CSW, CSE, BRO, SPTE, PSO, NBW, DMP, FLD4	Unknown Bird sp.	407
	OSS, OIE, CSE, CSW, BRO, PSO, SPTE, DMP, FLD3, FLD4, STP	Human[a]	181

	OIE, CSW, CSE, BRO, SPTE, PSO, NBW, DMP, FLD4	Gull sp.	85
	BRO, SPTE, PSO, NBW, DMP, FLD4	White-tailed Deer	47
	OIE, SPTE, CSE	Least tern	30
	OIE, BRO, PSO, SPTE	Toad sp.	14
	SPTE	Snowy owl	9
	DMP, SPTE	Tern sp.	9
	DMP	Common Tern	9
	CSW, BRO, SPTE, PSO, DMP	Unknown	7
	CSW, BRO	Red-winged Blackbird	6
	CSE, BRO, NBW	American Oystercatcher	4
	CSE, CSW	Dog	3
	OIE, STP	Crow sp.	2
	FLD4	Red fox	2
	PSO	Mourning Dove	2
	OIE, PSO	Willet	2
	BRO, NBW	Common grackle	2
	CSE	Eastern Cottontail	1
	SPTE	Semi-palmated plover	1
	BRO, NBW	Peregrine Falcon	1
	CSE, OIE, OIWR, OIW, OSS, STP	Human[a]	58
	CSE, OIW, OIWR, SPTE, NBW, STP	White-tailed Deer	30
	OIW, SPTE, SPTW, NBW, OSS	Toad sp.	22
	OIWR, NBW, FLD3	Gull sp.	13
	OIW, OIWR	Unknown Bird sp.	12
	CSE, OIE, OIWR, SPTE	Unknown	11
	OIWR, SPTE	Red-winged Blackbird	4
2019	NBW	Least tern	3
	OIWR, NBW	American Oystercatcher	3
	OIW, OIWR	Crow sp.	3
	SPTE	Mourning Dove	1
	OIWR	Barn Swallow	1
	NBW	Boat-tailed Grackle	1
	NBW	Common grackle	1
	OIW	Grackle sp.	1

Table 29. Cumulative capture events from 2016–2019 grouped by class.

Class	Capture Event ID	Number of Capture Events	Total
	Human[a]	528	528
Mammalia	White-tailed Deer	117	160
	Red Fox	29	
	Raccoon	5	
	Dog	3	
	Unknown Mammal sp.	3	
	Eastern Cottontail	2	
	Rodent sp.	1	
Aves	Gull sp.	489	1084
	Unknown Bird sp.	443	
	Least Tern	43	
	Crow sp.	31	
	Red-winged Blackbird	16	
	Common Tern	12	
	American Oystercatcher	12	
	Snowy Owl	9	
	Tern sp.	9	
	Common Grackle	3	
	Mourning Dove	3	
	Willet	3	
	Northern Mockingbird	3	
	Grackle sp.	2	
	Boat-tailed Grackle	1	
	Semi-palmated Plover	1	
	Barn Swallow	1	
	Peregrine Falcon	1	
	Great-horned Owl	1	
	Black Skimmer	1	
Malacostraca	Crab sp.	27	27
Amphibia	Toad sp.	9	60
	Unknown	28	28

[a]Only non-plover monitor humans were counted


2016-06-02 6:44:41 AM M 3/5 57°F
2017-06-09 5:01:12 PM M 1/3 71°F
2018-05-26 08:29:00 M 3/3 64°F
10:24:48 M 1/3 70°
17-06-03 00:52:10 M 3/5 55°F



Figure 31. Examples of captures from nest cameras. Species pictured include crow sp., red fox, snowy owl, peregrine falcon, raccoon, and great-horned owl, rat sp.

Discussion

Remote cameras we placed on plover nests detected both predator and non-predator species at nest exclosures. Humans, gulls, unknown or unidentifiable birds, and white-tailed deer were the most frequently pictured species at exclosures across all years.

When capture events were considered by class, birds were the most commonly detected, suggesting that their prevalence in our study system may have an effect on nests and chicks if the number of capture events is correlated with depredation or changes in plover behavior (Table 29). There were seven times as many bird capture events as non-human mammal capture events, but 76 (12%) of the identifiable birds recorded by our

cameras (semi-palmated plover, barn swallow, black skimmer, mourning dove, willet, northern mockingbird, tern sp.) likely pose no threat to plover nests or chicks. However, gulls, crows, grackles, owls, peregrine falcons, and red-winged blackbirds (*Agelaius phoeniceus*) are all known predators of piping plover nests or chicks (Lauro and Tanacredi 2002, Ivan and Murphy 2005). American oystercatchers also pose a threat to nests as they have been observed removing eggs (Lauro and Tanacredi 2002). Many 'unknown' birds detected were unidentifiable due to poor photo quality or lighting, and sometimes the birds were at unknown distances from the camera, resulting in a lack of perspective. These unknown bird capture events likely included some potential predators, such as crows and grackles and some species that are unlikely predators, such as barn swallows and mourning doves. However, even if we assumed that all unknown bird species were non-plover predators, 52% of all avian capture events would still come from the known plover predators mentioned above. In fact, gulls were the greatest non-human potential source of disturbance captured on our nest cameras. The amount of avian presence we detected at nest exclosures suggests that birds may play an important role in nest disturbance. If presence at nest exclosures during incubation is also an indicator of future interactions with plover chicks as they leave the exclosure, avian predators with many capture events, such as gulls, may also be substantial sources of future chick predation or disturbance.

During the 2018 and 2019 field seasons, we observed peregrine falcons during plover surveys, sometimes perched on nest exclosures and hunting shorebirds in the area. On one occasion, plover monitors observed a peregrine falcon dive in the direction of an adult plover, but instead, capture a nearby least tern (*Sternula antillarum*) fledgling. In 2019, a peregrine falcon was seen eating a plover chick while on its associated nest's exclosure and the flag leg of an adult from a different nest was found at the base of the same exclosure less than a week later. In 2018, we captured photos of a peregrine falcon on a nest camera less than an hour before the carcass of one of the incubating adults was found inside the same exclosure. Given how often we observed peregrine falcons on or near exclosures while surveying, we expected more capture events than what our cameras recorded. Owls, such as the great-horned owl, also are known predators of plovers and their range encompasses our study area year-round (Catlin et al. 2011). Only one great-horned owl was captured on our cameras in all years of this study. However, in 2017, we confirmed a great-horned owl as the source of mortality for a fledgling at Old Inlet East. Further, in 2019, we discovered owl feathers on at least two exclosures and owl tracks in the sand outside of exclosures that were surrounded by unidentified shorebird feathers, suggesting that owls may also have been hunting from exclosures like peregrine falcons. Interestingly, a snowy owl was observed near exclosures at Smith Point in late May 2018. Most snowy owls have departed for the breeding grounds by the time plovers in our study area begin to initiate nests in mid to late April (Holt et al. 2015), so it is unlikely that snowy owls present much of a disturbance or source of predation to plovers in our study area. Being at ground level, our cameras were not in the ideal position to capture all interactions between

raptors and plover nests. Since larger aerial predators likely perch higher than our cameras' field of vision on the top of exclosures, more often than they walk along the ground, their presence in this system likely is being underrepresented by our camera data. In contrast, gulls may be more likely to be detected by our cameras due to their propensity for walking or roosting on the ground.

Although detected at a lower rate, mammals also contributed to the potential disturbance of plover nests in our study area. Most mammal capture events came from white-tailed deer, and while deer have been known to trample nests and consume eggs or even live birds (Pietz and Granfors 2000, Ivan and Murphy 2005), they likely do not present much of a threat to plovers in our study area given that we never observed nest failure in association with deer sign. However, red foxes were the second most detected mammal and were recorded on cameras in every year except 2019. Capture events of red foxes on our cameras decreased each year following 2016, which tracks with the timeline of the mange outbreak (See Part 3, Section 1). If these nest capture events were correlated with actual numbers, potential disturbance from red foxes on plover nests decreased substantially along with the red fox population decline. In 2018, our cameras recorded a red fox at exclosures twice, suggesting that even at very low numbers, red foxes can contribute to nest disturbance and, as known predators of plovers, play a role in the fate of nests and chicks (See Part 1 – Objective 4). Although a rat was only present during one capture event, they are known plover nest predators (Lauro and Tanacredi 2002), therefore, their presence at nests, and their ability to enter an exclosure, poses a threat of nest disturbance and egg loss.

Toads and ghost crabs were most commonly seen inside of nest exclosures. Although toads do not pose a threat to nesting plovers, ghost crabs depredate eggs (Kwon et al. 2017). In 2019, an adult plover was found decapitated in a ghost crab burrow at Watch Hill and though we do not believe the adult was killed by a ghost crab, they have been observed taking chicks in Maryland and at Cape Hatteras National Seashore in North Carolina (Loegering and Fraser 1995, Kwon et al. 2017). In this study, both nests where ghost crabs were captured showed them entering burrows within or very close to the nest exclosure. Being small enough to enter exclosures, ghost crabs are one of the potential predators in our study area that could affect the survival rate of exclosed nests if they were present in greater numbers (Kwon et al. 2017).

Human disturbance has been observed to have a negative effect on plover chick survival, foraging rate, and body condition (DeRose-Wilson 2018). Predator and human disturbance also have been linked to an increased risk of nest abandonment in piping plovers (Doherty et al. 2011). Furthermore, evidence from Beale and Monaghan (2004) suggests that birds may respond similarly to humans as they would to other predators. Given the deleterious effects of disturbance on plover nests and chicks, these findings are important to consider moving forward (Hunt et al. 2018*b*). Not only must plovers be wary of myriad potential predators within our system, but they also must contend with a near constant presence of human activity in some areas at certain

times of the breeding season. Despite the use of symbolic fencing, non-plover monitor humans were in the top two contributors to capture events in all years except 2016. Although we were unable to assess human disturbance on plovers based purely on the number of capture events, we can make some inferences based on the photo data. Some capture events showed humans recreating between the nests and important foraging habitat such as wrack and low-energy intertidal habitat (Loegering and Fraser 1995), effectively blocking any chicks from reaching them. During June–August, when beach recreation is at its highest, chicks are likely forced to avoid these preferred foraging habitats and exert more energy foraging in less optimal habitat, such as interior, to avoid people recreating on the shore (See Part 1 – Objective 6; DeRose-Wilson 2018). Other capture events showed humans disregarding symbolic fencing and directly approaching nest exclosures. Humans may also be directly affecting incubating adult behavior if nests are close enough to these heavily trafficked areas or if humans disregard symbolic fencing protections. In the future, assessing human disturbance and its behavioral effects on the plovers in our study area would provide valuable insight into the management of recreational activities in the areas of nesting plovers and any potential limitations of traditional symbolic fencing buffers (Weithman 2019).

The detected predator community varied annually within our study area and understanding how to adapt management strategies to account for such a dynamic system is vital. There was a significant avian predator presence in the study area, and managers should consider the implications of these findings when deciding to exclose nests and for how long to leave exclosures up following hatch. Raptors have been known to adapt their hunting strategies with the introduction of artificial perches or by cuing in on nests (Reinert 1984, Niehaus et al. 2004, Murphy et al. 2003), and if exclosures are being used as eating or hunting perches in our study area, leaving them up after hatch may be detrimental to plovers in the area. Additionally, there is a potential for the red fox population to recover in our study area and recommence exerting predatory pressures on the plover population dynamics (See Part 1 – Objective 4). It is crucial to continue monitoring predators via nest cameras as the community changes in the following years and, ultimately, how these changes may influence the plover population in the future. Finally, using our cameras to assess disturbance to plovers by linking behavioral changes of the incubating adult to capture events could lead to a better understanding of the indirect effects that predators, non-predators, and humans have on the plovers in our study area (Hillman et al. 2015, DeRose-Wilson et al. 2015).

Part 3: Red Fox Movements, Habitat Selection, and Demography on Fire Island, New York

Section 1: Population ecology of red foxes

This section describes data collection, analytical methods, and results related to the following objectives for red fox monitoring associated with the Fire Island Inlet to Moriches Inlet Fire Island Stabilization Project:

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

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

Introduction

Information on population size and the factors influencing animal abundance is needed for wildlife management and conservation. For threatened and endangered species conservation, information on the population ecology of both the target species and its important predators can help guide recovery efforts. Managers may seek to quantify changes in predation pressure on an imperiled species by comparing predator abundance across space and time. Managers also may seek to decrease predator abundance by manipulating the factors that influence the predator species' population size. As a predator of the federally threatened piping plover (*Charadrius melodus*) and other shorebird species of concern, red fox (*Vulpes vulpes*) populations are commonly the target of predator management efforts along the U.S. Atlantic Flyway (Hunt et al. 2019*a*)

Different aspects of red fox population ecology have been studied in a variety of ecosystems (see Lloyd 1980, Ables 1983 for overviews). To our knowledge, no previous research has specifically examined red fox population reproduction, survival, or mortality rates in a barrier island ecosystem. In the context of piping plover management, information on red fox abundance and vital rates in plover habitats is needed to guide predator management efforts.

The nocturnal nature and general trap-wariness of red foxes and other common mammalian predators can make it difficult to conduct a complete population census. In such cases, occupancy, defined as the proportion of an area occupied by a species, can be used instead of abundance to describe populations (MacKenzie et al. 2002). Occupancy estimation also is useful in cases where the inability to reliably distinguish among individuals limits the application of capture-mark-recapture methods to estimate abundance, as is often the case for red foxes (Pollock et al. 2002, Bailey et al. 2004, Güthlin et al. 2014).

So far, no published study, but one dissertation, has examined red fox occupancy and habitat relationships on barrier islands. Gieder (2015) reported a landscape level occupancy estimate of 0.43 based on surveys conducted on Assateague Island, MD. There, red fox occupancy was positively related to eastern cottontail trap success, and did not decrease following lethal control efforts. In the only other published study to examine red fox occupancy in an Atlantic coastal setting, conducted at Cape Cod, MA, O'Connell et al. (2006) reported model-averaged occupancy estimates of 0.35 and 0.41, varying with survey methods. However, this study was designed to compare multiple survey methods (camera trapping vs. field sign) for 10 species with very different spatial habits, rather than specifically investigating red fox occupancy in the area (Gieder 2015).

We aimed to fill these gaps in scientific knowledge on red fox population ecology in in coastal ecosystems while collecting information that could be incorporated into an island-wide predator management plan on Fire Island, NY. Our objectives were to estimate red fox occupancy and trap success, relative abundance, annual reproduction, and survival/mortality rates on the island and describe how these population parameters have varied in response to management actions and environmental conditions.

Methods

Field methods

GPS collaring and ear tagging

During August–December 2015, April–May 2016, and February–April 2017 and 2018, we attempted to capture up to 6 foxes per year in each management unit (Smith Point County Park, Fire Island National Seashore, and Robert Moses State Park) to attach combination GPS/VHF collars (Telemetry Solutions, Quantum 4000E) and ear-tags. We captured foxes using Victor 1 1/2 Soft Catch padded foothold traps and Bridger #2 Offset coil spring foothold traps. Tomahawk 2-door fox-size box traps were also deployed in 2015, but we discontinued the use of these traps in 2016 and 2017 due to a lack of fox captures and substantial bycatch of raccoons. We briefly resumed the use of box traps during our 2018 trapping season due to the low overall number of fox captures, but this resulted in similar levels of non-target species bycatch and no fox captures. 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 (AFWA 2014).

Upon capture, we manually restrained foxes using a catch-pole and attached a muzzle, leg-ties, and a blindfold; no sedatives were used. We attached the GPS collar, collected morphometric measurements, and placed an ear-tag in the right ear when allowed by area managers. We used serially-numbered white ear-tags at Robert Moses State Park, serially-numbered blue ear-tags at Smith Point County Park, and small (~1/4" x 5/8") metal ear-tags at Fire Island National Seashore. The colored plastic ear-tags were intended to aid in identification of individual foxes in remote camera photos and those encountered during transect surveys (see

below). In contrast, the small metal ear-tags were not intended to be visible in remote camera photos or during transect surveys, but they were meant to assist in identifying any individuals recaptured in future years.

GPS tracking

We used VHF radio telemetry to relocate collared foxes every 10–14 days until the programmed drop off date 5–6 months later (2015 monitoring period), or in early August at the end of the piping plover breeding season (2016–2018). During these relocations we remotely downloaded GPS location data stored on the collars (Part 3, Section 2). We also checked for collar mortality signals during these relocation events and followed the VHF signal to locate the remains of any foxes that died during the GPS monitoring period. After recovering GPS collars, we attempted to determine the cause of death through close inspection of remains. Depending on the condition of the remains, we either collected carcasses or disposed of them according to land managers' directions. When carcasses were not collected, we attempted to collect a tissue sample to send to the Mammalian Ecology and Conservation Unit (MECU) in the Veterinary Genetics Lab at University of California at Davis for potential use in phylogeographic analyses (described below).

Transect surveys for red fox individuals, scat, and dens

We used ArcMap 10.3 GIS software (ESRI, Redlands, CA) to create and overlay a 300-m x 300-m grid on each of the 3 focal study areas for the red fox monitoring activities of this contract: 1) Robert Moses State Park and the Fire Island National Seashore Lighthouse Tract, 2) Fire Island National Seashore wilderness area (Watch Hill to Old Inlet), and 3) Fire Island National Seashore wilderness area (Old Inlet East) and Smith Point County Park. The goal of this grid overlay was to focus survey efforts 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 suggested that this spacing was 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 (Part 3, Section 2). Thus, all foxes present should have been detected based on grid cell size relative to observed fox home range sizes.

Based on this grid overlay, we established 96 walking transects (24 in Robert Moses State Park, 40 in Fire Island National Seashore, and 32 in Smith Point County Park) connecting the center points of each grid cell from the ocean beach high tide line to the bayside high tide line, or the point where bayside transects became impassible due to deep marsh or dense shrub thickets (Figure 32). We walked each transect across the island approximately once per month during the red fox denning and piping plover breeding seasons (May, June, July) and again outside of the breeding seasons in October/November, December, and January (2015–2018). 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.

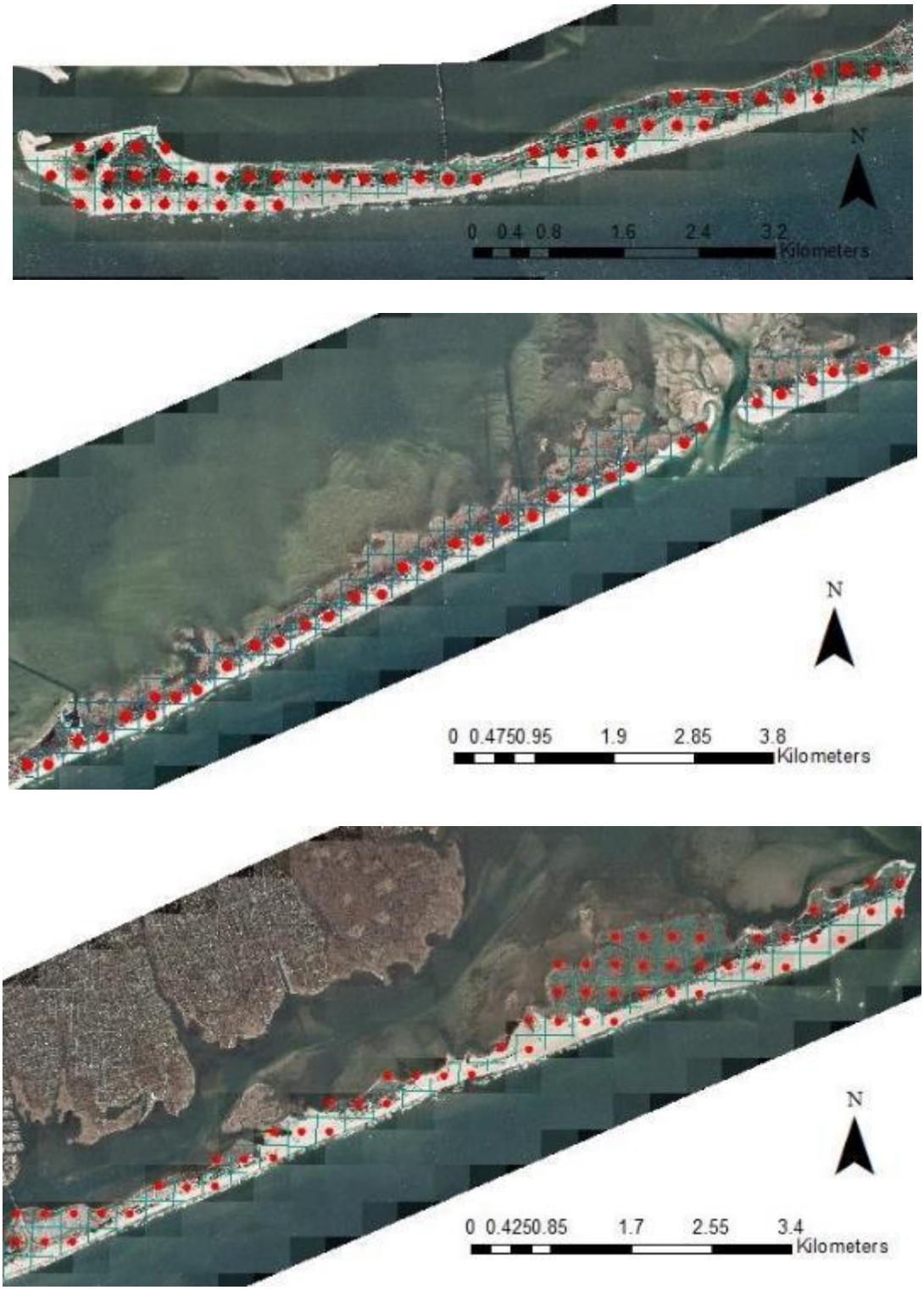


Figure 32. Transect locations in Robert Moses State Park and Fire Island National Seashore Lighthouse Tract (top), Fire Island National Seashore wilderness area (middle), and Smith Point County Park (bottom). We walked as close to straight-line transects as possible while following game trails and wildlife travel ways 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. 300-m grid cell borders are shown as green lines.

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 in a paper bag and labeled by estimated freshness and a unique sample number corresponding to the GPS waypoint that was recorded (Table 30). All observed red foxes were recorded along with the corresponding GPS location. All active and inactive red fox den sites also were recorded. 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 locate dens. 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.

In response to a mange outbreak West of Old Inlet beginning in Spring/Summer 2017 and peaking in 2018 (see results below), we conducted monthly driving surveys from October 2018–February 2019 along both the Burma Road and outer beach between Kismet and Watch Hill to determine whether foxes might have persisted in the villages following the population declines west of Old Inlet. During these surveys, we recorded the GPS locations of all fox tracks and live foxes encountered along the survey route. We also solicited information on red fox sightings on the island, particularly in the villages, from NPS staff and project collaborators.

Table 30. 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

We monitored all active dens located during transect surveys, GPS tracking of collared foxes, or opportunistically from the time of discovery until at least 2 weeks after the last signs of active den use. Any newly 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 15m 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 one or more remote cameras (Moultrie M880i, EBSCO Industries, Calera, AL) aimed at the den entrance and/or small clearings directly outside of the den to quantify the number of kits emerging from the den. We changed memory cards and batteries and adjusted camera orientation during den monitoring visits.

Remote camera grids

In late September–February of each year, we used remote camera grids to survey for foxes and other potential mammalian predator species in each of the 3 focal study areas. We used the same 300-m grid cells and center points that we used to guide our transect surveys to guide remote camera placement, and we established an unbaited camera station within 50 m of the center point of each grid cell containing suitable sites. To minimize the risk of equipment damage and theft, we avoided setting cameras out on the open beach, in low-lying areas prone to flooding, and in areas of heavy human use. For similar reasons, we also excluded grid cells that were entirely composed of large parking lots and inundated marsh. Each station consisted of 2 different models of trail cameras (Moultrie M880c and M880 Gen2, EBSCO Industries, Calera, AL), each mounted on a plastic t-post and facing towards a game trail, clearing, or other potential wildlife travel way. We set each camera on opposite sides of the animal trail to photograph both sides of passing animals and angled them to be offset from each other.

Cameras were triggered by movement in front of them. 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 33). Each grid was 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.

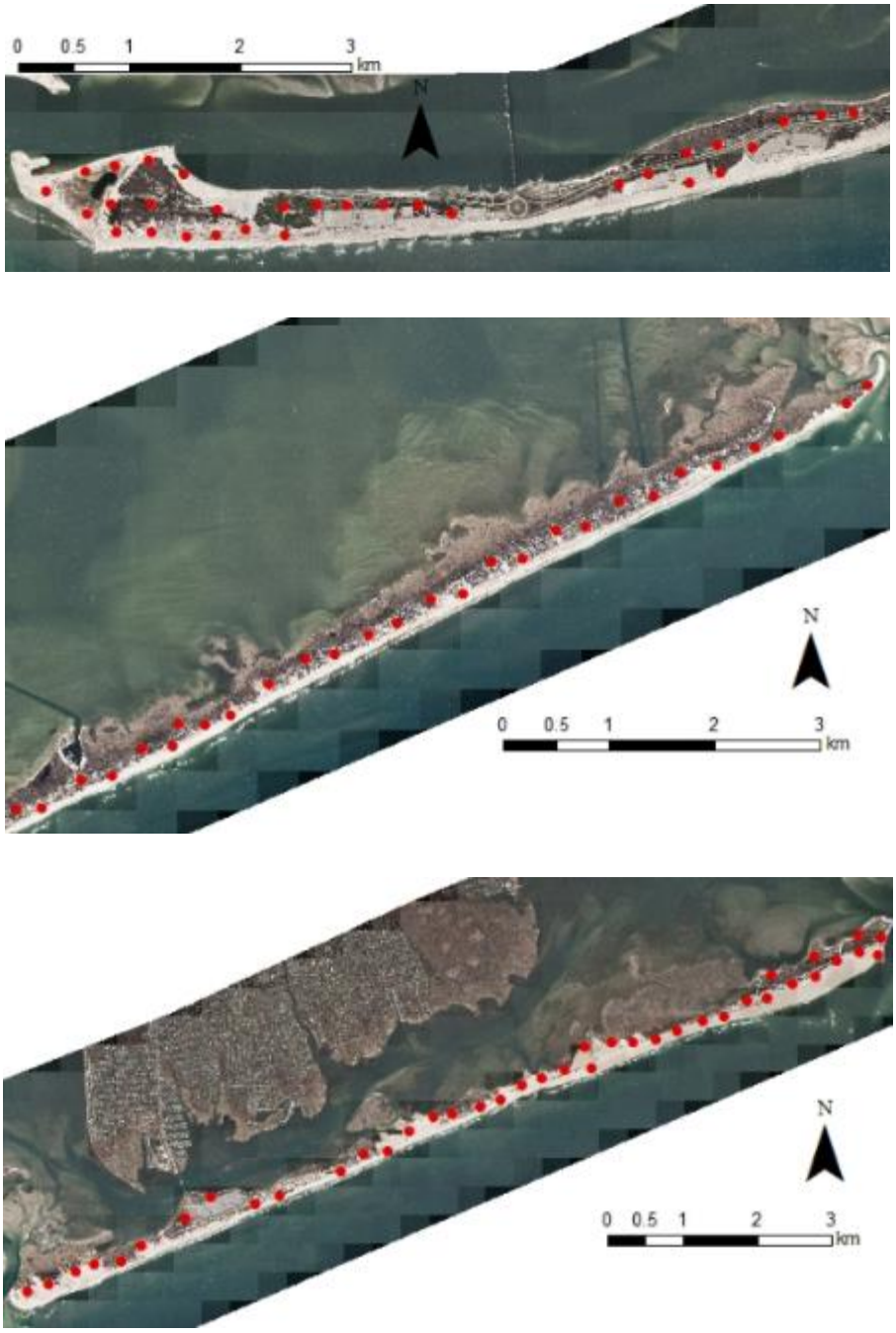


Figure 33. Approximate camera station locations at Robert Moses State Park (top), Fire Island National Seashore between Watch Hill and Old Inlet (middle), and Fire Island National Seashore east of Old Inlet and Smith Point County Park (bottom). Exact location and number of stations varied slightly among years.

In examining photos taken during our first season of remote camera surveys, 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 (Part 3, Section 2). 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 small and not meant to be detectable or readable in photos.

Lab methods

Scat samples collected during transect surveys and opportunistically during den monitoring and trapping were stored in a freezer until they could be transported on ice back to Virginia Tech, then 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 were sent to the MECU for mitochondrial DNA sequencing to verify the source species (Aubry et al. 2009, Miles et al. 2015). Feral cats (*Felis catus*), domestic dogs (*Canis lupus familiaris*), and raccoons (*Procyon lotor*) are present on Fire Island, and even experienced field biologists can have difficulty distinguishing among scats from these species, hence the need for genetic verification. Following genetic subsampling, the remaining portion of each scat sample was used for fecal dietary analysis (Part 3, Section 4).

For a subset of these scat samples shown to contain high quality red fox DNA, we collaborated with the MECU to conduct further analyses to determine the phylogenetic origins of red foxes on Fire Island. Red fox-specific primers and assays and genotyping protocols previously developed by the MECU (Sacks et al. 2010, 2011, Quinn et al. 2019), were used. Mitochondrial DNA haplotypes from samples collected on Fire Island were compared to those of samples from the MECU's reference collection and other samples of known geographic origins, to determine whether red foxes on the island are most 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. Because mitochondrial DNA is maternally inherited, haplotypes alone are not always sufficient to determine phylogenetic origins of a population. Thus, 22 nuclear DNA microsatellite loci also were examined and compared to similar data from reference samples. For samples with sufficient DNA quality, these microsatellite analyses allowed for the assignment of unique genotypes and sex markers, allowing us to distinguish among scats from different individuals.

Analytical methods

Trap success

We used photographic detection data from our fall camera grids to calculate trap success for red foxes and other potential piping plover predators. Trap success is a common metric reported for remote camera studies that can be used as an index of species activity in a given area. Trap success differs from the raw number of species detections in that it is scaled by unit effort, allowing for comparisons among survey sites and periods and across studies and 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 during which cameras were operational, then multiplied by 100 (Kelly and Holub 2008). We used 30-minute detection periods to separate consecutive photographs of animals that were not uniquely identifiable into detection events, so that any animal of that species photographed within 30 minutes of a previous photograph was counted as a single detection event rather than a separate detection event.

Occupancy

We used photographic detection data 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, MacKenzie et al. 2002). We used a single species, single season framework in Program PRESENCE (Version 2.12.21, Hines 2006). Occupancy does not translate directly to abundance, as it does not yield an estimate of individuals detected, just the probability of presence accounting for imperfect detection.

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 and a cut-off of 4 ΔAIC to identify top models (Burnham and Anderson 2007, Arnold 2010). 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.

Minimum red fox abundance estimates

Due to low rates of individual identification in remote camera photos, we were unable to estimate abundance using spatial mark-recapture analyses as we had originally planned. Instead, we combined information from our various data collection methods to estimate the minimum known fox abundance in each focal study area in August of 2016 – 2018. The August cut-off aligned with the end of the piping plover nesting period and the point when the majority of chicks would have fledged each season. Our minimum known red fox abundance was estimated by adding together 1) the number of GPS-collared and ear-tagged foxes known to be

alive in August of each year, 2) the number of additional uniquely identifiable adults associated with each breeding den 3) the maximum number of kits observed at a given time at each den. When adults associated with a breeding den were not uniquely identifiable, we assumed a minimum of 2 attending adults in our estimation of minimum population size. We adjusted these minimum estimated abundances to exclude any deceased foxes discovered during monitoring activities. Each individually identifiable fox was counted only once each season such that a GPS-collared individual observed at a den was only counted as one individual. We then divided our combined minimum abundance estimates for adults and kits by the area of each management unit to estimate minimum densities for each management unit and year.

We also were able to use the microsatellite genotyping results from genetic analysis of scat and tissue samples to independently estimate the number of unique foxes detected in a given year from the transect scat sampling. These estimates offer an additional index of minimum abundance but are not necessarily comparable to the minimum abundance estimates described above, as they were based on a different and unique data source. Due to the asynchronous nature of these scat collections which occurred both in the breeding season and winter dispersal season, we were unable to exclude animals that died during the year.

Reproduction

We used a combination of remote camera photos, direct observations during den monitoring visits, 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 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. Similarly, we used the maximum number of kits observed at any given point at a family group's dens, minus any kits know to have died during the monitoring period, to estimate the number of kits per breeding group. Because these estimates are based on remote camera detections and field-based observations after kit emergence from the den, they do not include any kits that failed to emerge from the den.

Survival/mortality

We used known survival/mortality of GPS-collared foxes to estimate adult survival rates during each Spring–Summer monitoring period. We also used microsatellite genotyping results to estimate the probability that an individual fox detected in one year would be detected during the following year. As with our minimum abundance estimates, these genetic results are not completely comparable to estimates based on known survival/mortality of GPS-collared foxes but serve as a useful additional metric of survival across years.

Results

Environmental and management drivers of red fox population dynamics: Mange outbreaks and human removals

In between the 2015 and 2016 piping plover breeding seasons, one unmarked red fox was removed from Smith Point County Park by USDA-APHIS as part of predator management efforts. No additional lethal management of foxes was conducted during the study. A sarcoptic mange outbreak, caused by the *Sarcoptes scabiei* mite, began east of Old Inlet in late summer/early fall 2015, leading to a substantial population decline (as described below) and a complete lack of fox sign from April 2016–March 2017. From March 2017 until the end of our monitoring, we observed only sporadic fox sign east of Old Inlet, and we were unsuccessful in capturing any foxes in this section of the island. A second sarcoptic mange outbreak began west of Old Inlet during the Spring–Summer 2017 monitoring period and continued through the end of data collection in early 2019, leading to another significant population decline (as described below), this time on the west side of Old Inlet.

We observed one fox with signs of moderate mange and several sets of fox tracks during our October 2018 driving surveys through the Fire Island villages but did not encounter any reliable fox signs during subsequent surveys in November 2018-February 2019. During this driving survey period, we received multiple reports of fox sightings in the villages from NPS staff (Table 31), many of which included photographs, but no reliable sightings after December 2018.

Table 31. Summary of fox sighting reports from NPS staff on Fire Island from August 2018–February 2019, after the mortality of all GPS-collared foxes from sarcoptic mange.

Month/Date	Location	# Foxes	Signs of mange?	Photos?
August 2018	Sailor's Haven	1	No	No
September 2018[a]	Watch Hill	1	No	No
10/17/2018	Wilderness Area	1	No	Yes
12/4/2018	Robin's Rest	1	No	No
12/11/2018	Kismet	1	Yes, moderate	Yes
12/12/2018	Sailor's Haven	1	Yes, low-moderate	Yes

[a] Multiple sightings reported at Watch Hill in September 2018, believed to be the same individual.

Wildlife rehabilitators removed one fox from Robert Moses County Park for treatment of mange in January 2018, and we removed two foxes from Robert Moses State Park for treatment of trapping-related injuries per

our permit requirements (one in August 2015 and one in March 2018). To our knowledge, none of these foxes were returned to the island following treatment. We do not consider these removals as mortality events in our results as described below under survival.

GPS monitoring and ear-tagging

We outfitted a total of 36 unique foxes with GPS collars between 2015 and 2018 (Table 32). Of these foxes, 16 also were given large plastic ear-tags and 14 were given small metal ear-tags. Eight additional foxes were captured and given ear-tags but not GPS collars due to the limited supply of collars. Of these foxes, 5 were given large plastic ear-tags, and 3 were given small metal ear-tags. Two foxes were captured and collared in multiple trapping seasons and were thus monitored during multiple years' tracking periods.

One non-ear-tagged fox disappeared after GPS collaring in 2015 and was never seen again or located via radio telemetry, despite extensive searching. Collar malfunctions resulted in premature collar drop-off for another non-ear-tagged fox, less than 24 hours after deployment in 2016. These two foxes were excluded from our minimum abundance and survival estimates because we were unable to verify whether or not they survived through the end of the monitoring period. We remotely activated the drop-off of another fox's collar shortly after deployment in 2018 due to a collar malfunction. We were able to identify this fox via ear-tag and confirm survival via remote camera detections throughout the monitoring period.

Table 32. Red foxes GPS collared during each tracking period from 2015-2018 on Fire Island, NY. 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.

	2015		2016		2017		2018	
Study Area	Males	Females	Males	Females	Males	Females	Males	Females
Robert Moses State Park + Lighthouse Tract	2	3	4	2	4	4	1	2
Fire Island National Seashore (Watch Hill to Old Inlet)	3	0	3	3	1	3	0	0
Smith Point County Park + Old Inlet East	2	1	0	0	0	0	0	0
Total	7	4	7	5	5	7	1	2

Trap success

In addition to red foxes, we detected raccoons, feral cats, domestic dogs, gulls (various species, Family Laridae), crows (*Corvus brachyrhynchos*), and several raptor species during our remote camera surveys (Table 33). Species of raptors 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 was much less common in 2017 and in 2018. East of Old Inlet, red fox trap success was highest in 2015 (13.99 capture events/100 trap nights) but still relatively low compared to other monitored management areas that year (Robert Moses State Park, trap success = 89.18, Fire Island National Seashore Watch Hill to Old Inlet, trap success = 57.01). Red foxes apparently were absent east of Old Inlet in 2016, followed by very low trap success in 2017 (3.54 capture events/100 trap nights) and 2018 (5.08 capture events/100 trap nights). These changes in red fox trap success align with the mange outbreaks observed on the island during our monitoring, described above. As fox trap success decreased, raccoons became the most commonly detected predator, but they were still detected far less frequently than red foxes were in the earlier years of the study. Feral cat trap success varied among sites and years, remaining very low in the wilderness area between Watch Hill and Old Inlet and increasing at Robert Moses State Park and areas east of Old Inlet in the final year of the study (Figure 34).

Table 33. Trap success (# detection events/100 trap nights) for potential shorebird predator species during fall–winter remote camera surveys from 2015-2019, Fire Island, NY. 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 spp.	American Crow	Raptor
Fall 2015–Winter 2016							
Robert Moses State Park	89.18	13.7	8.22	2.12	0.62	0.14	0.27
FINS (Watch Hill to Old Inlet)	57.01	17.23	1.09	0.19	0.45	3.46	0.7
Smith Point County Park + Old Inlet East	13.99	19.64	2.45	1.07	0.31	0.13	2.2
Fall 2016–Winter 2017							
Robert Moses State Park	119.95	26.14	8.26	0.84	0.28	0.28	0.22
FINS (Watch Hill to Old Inlet)	103.36	24.39	0.39	1.29	0.39	10.03	1.68
Smith Point County Park + Old Inlet East	0	21.84	2.79	0.21	1.55	0.72	5.63
Fall 2017–Winter 2018							
Robert Moses State Park	18.36	20.37	5.66	0.63	0	0.21	0.84
FINS (Watch Hill to Old Inlet)	72.62	34.19	0.26	0	0	3.84	0
Smith Point County Park + Old Inlet East	3.54	15.19	4.97	0.14	0.2	1.09	0.61
Fall 2018–Winter 2019							
Robert Moses State Park	4.99	23.22	17.70	1.07	0.06	0.18	0.12
FINS (Watch Hill to Old Inlet)	8.30	12.63	1.27	0.10	0.05	6.52	2.60
Smith Point County Park + Old Inlet East	5.08	8.38	10.78	0.11	0.06	1.84	1.68

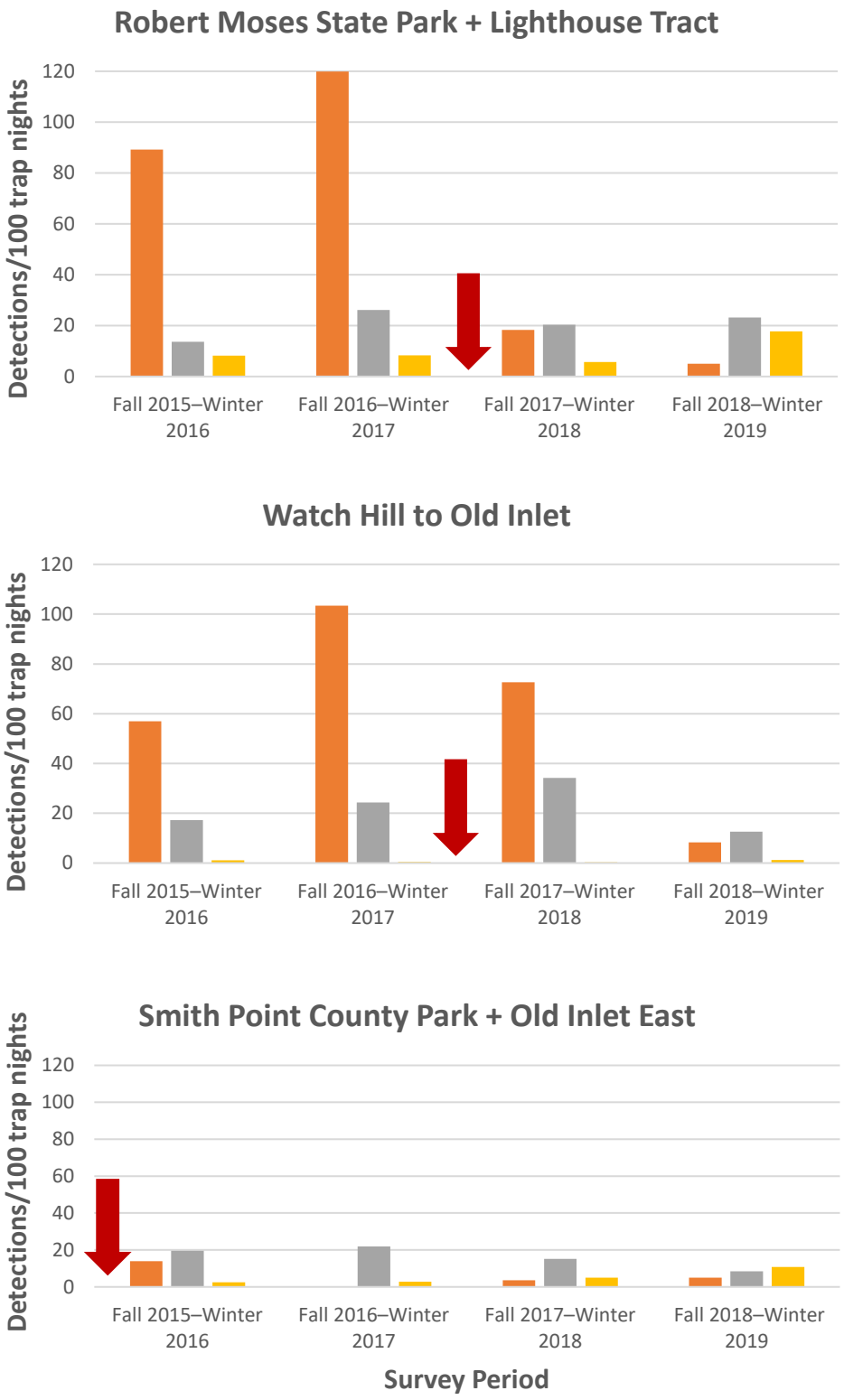


Figure 34. Trap success estimates for red fox (orange), raccoon (gray), and feral cat (yellow) at Robert Moses State Park (top), Fire Island National Seashore between Watch Hill and Old Inlet (middle), and Fire Island National Seashore east of Old Inlet and Smith Point County Park (bottom) for each remote camera survey period on Fire Island, NY from 2015–2019. Red arrows indicate mange outbreaks.

In each of our study sites, we found that red fox occupancy was best modelled as constant, but red fox detection varied across sites and years in whether it was best modelled as constant or varying by day (Table 34). Red foxes were detected at all stations west of Old Inlet until the second mange outbreak began West of Old Inlet in Spring 2017. For example, at Robert Moses State Park, occupancy (ψ) was 1.0 in both 2015 and 2016 and declined to 0.97 and 0.82 in 2017 and 2018, respectively. Thus, even after the mange outbreak resulted in high mortality rates of red foxes in these areas, the remaining red foxes were observed at a high proportion of camera stations. This same pattern was observed in the wilderness area from Watch Hill to Old Inlet (Table 34, Figure 35). Detection probabilities in these areas West of Old Inlet varied among sites and years and was generally lower following fox population declines due to mange (Table 34). At Smith Point County Park and the Old Inlet East portion of Fire Island National Seashore, fox occupancy remained similar across all years that foxes were present, while detection probability generally decreased over time (Table 34). Specifically, red fox occupancy at Smith Point County Park in 2015 before the mange outbreak was 0.86, 0 in 2016 during the outbreak, 0.84 in 2017 and 0.82 in 2018, despite evidence from other field-data that there was likely only one fox present in this system in the latter two years.

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 approach 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.

Table 34. Model selection tables for occupancy modeling using Fall–Winter remote camera detection data on Fire Island, NY from 2015-2018. 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 2015–Winter 2016						
Robert Moses State Park	ψ(.) p(day)	1953.57	0.00	1.0	1.0	53
	ψ(.) p(.)	2021.14	67.57	0.0	0.0	2
Fire Island National Seashore (Watch Hill to Old Inlet)	ψ(.) p(day)	1985.15	0.00	1.0	1.0	62
	ψ(.) p(.)	2000.60	15.45	0.0	0.0	2
Smith Point County Park + Old Inlet East	ψ(.) p(day)	1098.50	0.00	1.0	1.0	47
	ψ(.) p(.)	1124.24	25.74	0.0	0.0	2
Fall 2016–Winter 2017						
Robert Moses State Park	ψ(.) p(.)	2257.46	0.0	1.0	1.0	2
	ψ(.) p(day)	2292.93	35.47	0.0	0.0	62
Fire Island National Seashore (Watch Hill to Old Inlet)	ψ(.) p(.)	2176.80	0.00	0.99	1.0	2
	ψ(.) p(day)	1886.26	9.46	<0.01	<0.01	58
Smith Point County Park+ Old Inlet East	N/A	-	-	-	-	-
	N/A	-	-	-	-	-
Fall 2017–Winter 2018						
Robert Moses State Park	ψ(.) p(day)	1552.64	0.00	0.99	1.0	67
	ψ(.) p(.)	1569.44	16.80	<0.01	<0.01	2
Fire Island National Seashore (Watch Hill to Old Inlet)	ψ(.) p(.)	2062.46	0.00	0.99	1.0	2
	ψ(.) p(day)	2072.23	9.77	<0.01	0.0	59
Smith Point County Park+ Old Inlet East	ψ(.) p(day)	413.14	0.00	1	1	2
	ψ(.) p(.)[b]	-	-	-	-	-
Fall 2018–Winter 2019						
Robert Moses State Park	ψ(.) p(day)	663.16	0.00	1.0	1.0	58
	ψ(.) p(.)[b]	-	-	-	-	-

Fire Island National Seashore (Watch Hill to Old Inlet)	ψ(.) p(day)[b]	966.77	0.00	0.72	1.0	80
	ψ(.) p(.)	968.63	1.86	0.28	0.4	2
Smith Point County Park+ Old Inlet East	ψ(.) p(.)	684.08	0.00	1.0	1.0	2
	ψ(.) p(day)	722.67	38.59	0.0	0.0	54

[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 35 Occupancy (ψ) and detection probability (p) and estimates from the top model for each monitoring area and year on Fire Island, NY from 2015–2019.

	Study Area	Robert Moses State Park	Fire Island National Seashore (Watch Hill to Old Inlet)	Smith Point County Park + Old Inlet East
Fall 2015–Winter 2016	Top Model	ψ(.) p(day)	ψ(.) p(day)	ψ(.) p(day)
	ψ (SE)	1.00 (0.00)	1.00 (0.00)	0.86 (0.06)
	p[a]	0.03-0.72	0.11-0.61	0.03-0.25
Fall 2016–Winter 2017	Top Model	ψ(.) p(.)	ψ(.) p(.)	n/a
	ψ (SE)	1.00 (0.00)	1.00 (0.00)	0.00
	p[a]	0.67 (0.01)	0.54 (0.01)	n/a
Fall 2017–Winter 2018	Top Model	ψ(.) p(day)	ψ(.) p(.)	ψ(.) p(.)
	ψ (SE)	0.97 (0.03)	1.00 (0.00)	0.84 (0.12)
	p[a]	0.03-0.38	0.42 (0.01)	0.04-0.14
Fall 2018–Winter 2019	Top Model	ψ(.) p(day)	ψ(.) p(.)	ψ(.) p(.)
	ψ (SE)	0.82 (0.07)	0.86 (0.07)	0.80 (0.08)
	p[a]	0.00-0.18	0.08 (0.01)	0.06 (0.01)

[a] If detection probability was best modeled as a constant, a single estimate and SE are shown. If it was best modeled as varying by day, the range of estimated values is reported.

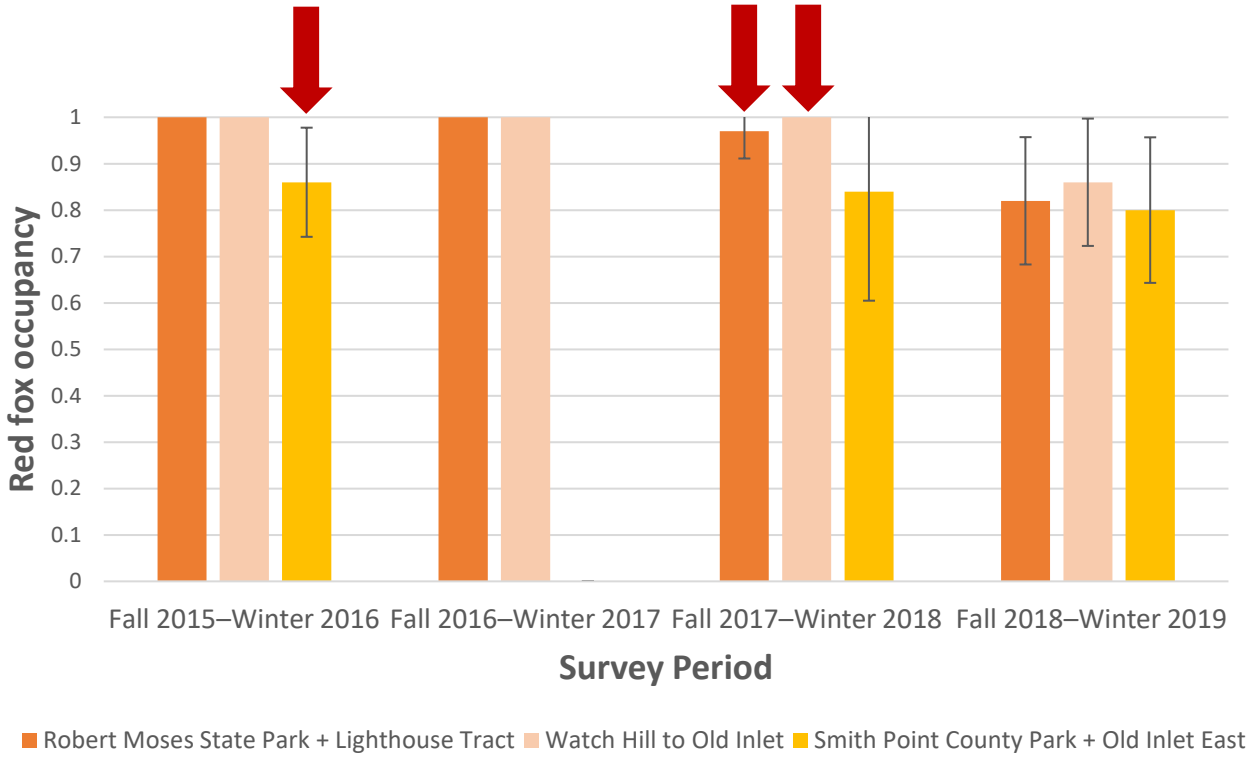


Figure 35. Red fox occupancy estimates and 95% confidence intervals for each monitoring area and survey period on Fire Island, NY from 2015–2019. Red arrows indicate survey periods following mange outbreaks in a given monitoring area.

Minimum abundance estimates

Our minimum abundance and density estimates varied by site and year (Table 36, Figure 36, Figure 37). In areas west of Old Inlet, densities were highest in August 2016 and decreased substantially through August 2018. In areas east of Old Inlet, densities were lowest in August 2016, increased slightly in 2017, and remained stationary but low in 2018. These estimates and changes over time align closely with the 2 mange outbreaks we observed.

Table 36. Minimum estimated red fox adult and sub-adult abundances and total densities by monitoring area as of August 2016-2018 of each year on Fire Island, NY.

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
Fire Island National Seashore (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

[a]Estimates for all area west of Old Inlet were derived by averaging estimated densities in surveyed areas and extrapolating to the amount of unsurveyed area available.

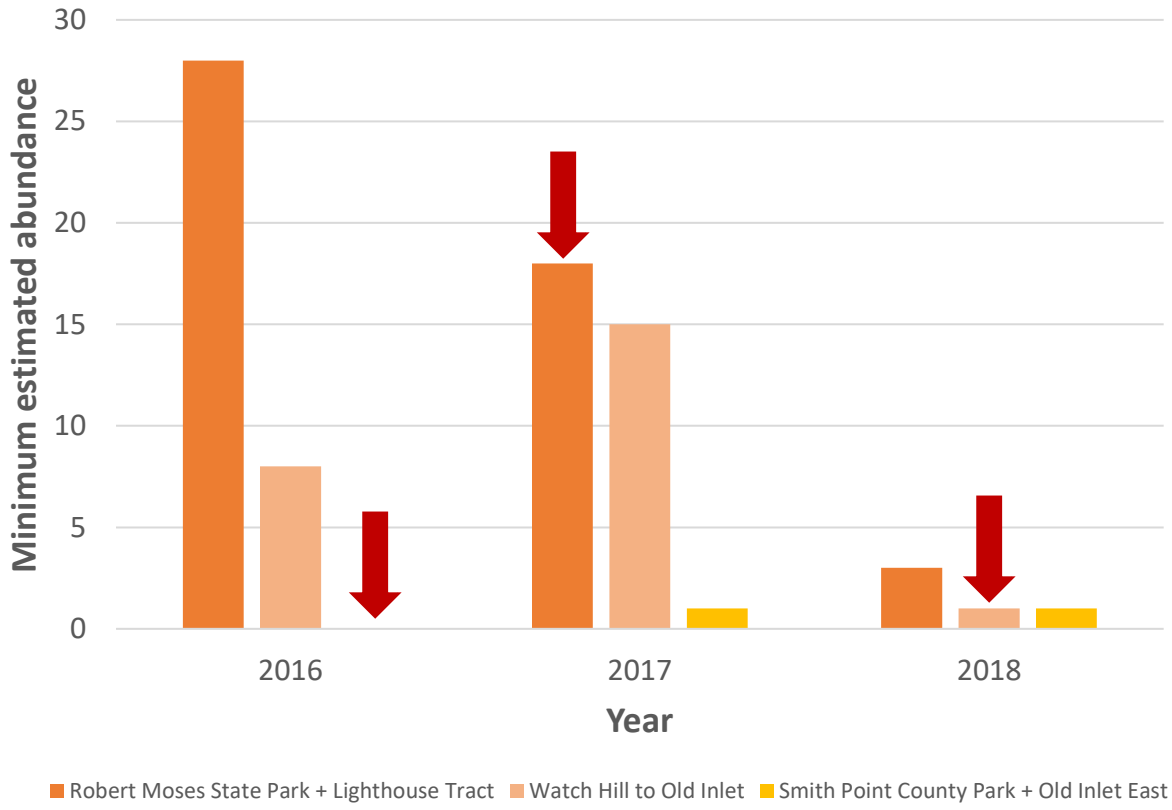


Figure 36. Minimum estimated red fox abundances (adult and sub-adults combined) in each monitoring area as of August 2016-2018, on Fire Island, NY. Red arrows indicate the beginnings of mange outbreaks or years immediately following outbreaks in a given monitoring area.

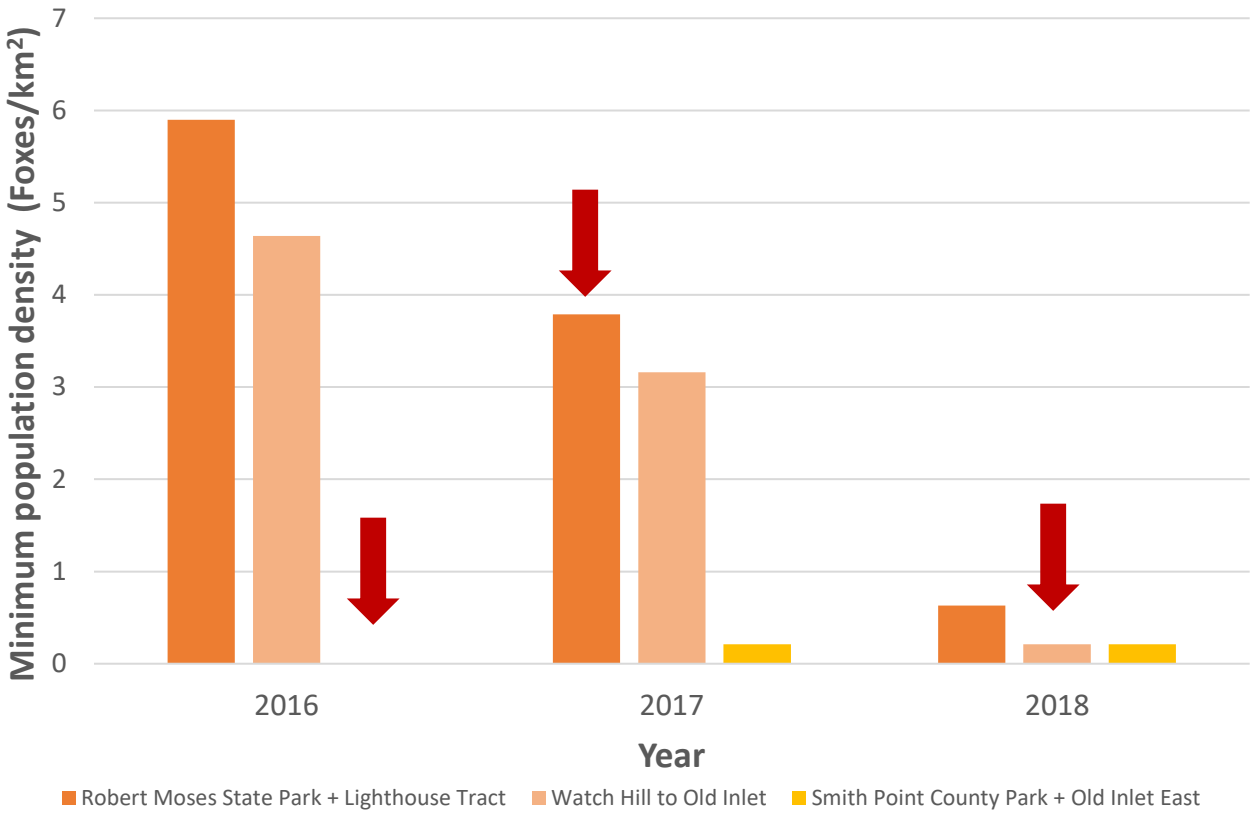


Figure 37. Minimum estimated red fox densities (adult and sub-adults combined) in each monitoring area as of August 2016-2018, on Fire Island, NY. Red arrows indicate mange outbreaks or years immediately following mange outbreaks in a given monitoring area.

Seventy-one of our scat samples contained sufficient DNA to identify the individual fox that deposited them via microsatellite genotyping. This resulted in 48 unique genotypes across samples collected from April 2016–February 2019. Four of our tissue samples also contained sufficient DNA for individual identification via microsatellite genotyping. These tissue samples yielded 4 unique genotypes, including one that matched a genotype from a scat sample collected in 2016, for a total of 51 unique genotypes across samples collected from April 2016–February 2019 (Table 37). While these genetic results are not completely comparable to the minimum abundance estimates described above, they support the idea that fox populations on the island were significantly lower after the mange outbreaks in 2017 (12 unique genotypes sampled) and 2018(14 unique genotypes sampled) compared to 2016 (32 unique genotypes sampled) which was before the second mange outbreak to the West of Old Inlet was detected (Table 37).

Table 37. Unique genotypes sample from scat samples collected during transect surveys, den monitoring, and opportunistic collection, and tissue samples sampled from carcasses collected, from 2016–February 2019 at Fire Island, NY.

Time period	Unique genotypes sampled
April–December 2016	32
January–December 2017	12
January 2018–February 2019	14

Reproduction

Breeding red foxes used multiple dens within a breeding season, with nearly all breeding family groups moving from a natal den to a secondary den as the season progressed (Table 38). Estimated dens used per family group per breeding season averaged 1.69 and ranged from 1–3, although this may be an underestimate as not all breeding dens could be assigned to a known family group. As kits developed and became more independent, family groups shifted away from using dens in favor of sheltered above-ground areas such as dune bowls and interdune clearings. In all years and across all sites, we located additional non-breeding dens that likely served as thermal refugia from the summertime heat and/or daytime underground resting sites used by adults.

The number of breeding family groups and dens varied by site and generally decreased over the course of the monitoring in each area (Figure 38). Similarly, the mean number of kits per breeding group also decreased over the course of the study, with no evidence of successful reproduction in 2018 in any of the focal study areas (Figure 39). This decline in reproduction was correlated with the population decline following the mange outbreaks in each monitoring area and the poor general condition of the few foxes surviving in 2018.

Table 38. Breeding dens located on transect surveys, through GPS tracking, and opportunistically in April–July of 2016–2018, and the associated number of likely breeding family groups.

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

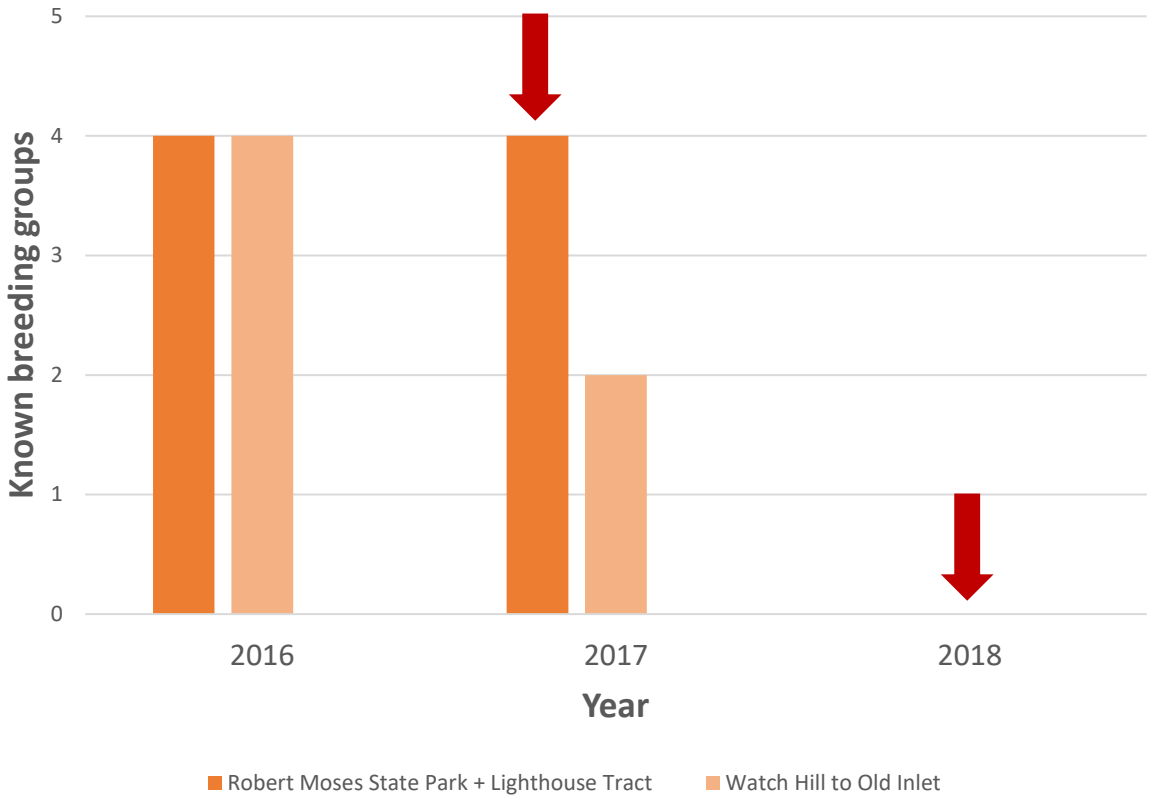


Figure 38. Known red fox breeding family groups in each monitoring area during 2016–2018 breeding seasons, on Fire Island, NY. Red arrows indicate mange outbreaks or years immediately following mange outbreaks in a given monitoring area.

Table 39 Mean number of kits observed per breeding group in 2016–2018.

Study Area	2016	2017	2018
Robert Moses State Park + Lighthouse Tract	4	3.25	0
Fire Island National Seashore (Watch Hill to Old Inlet)	3.5	1	0
Smith Point County Park + Old Inlet East	0	0	0

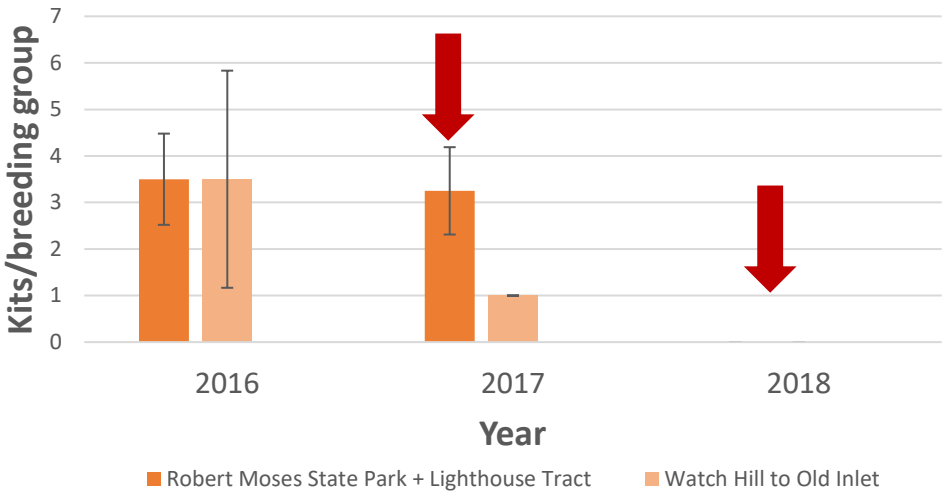


Figure 39. Mean kits per breeding group, and 95% confidence intervals, in each monitoring area during 2016-2018 breeding seasons, on Fire Island, NY. Red arrows indicate mange outbreaks.

Survival/mortality

The proportion of GPS-collared foxes surviving to the end of the monitoring period varied widely among sites and years, ranging from 0-1 among years and monitoring areas (Table 40). Of the GPS-collared foxes that died during the monitoring period, mange was the most common observed cause of death (n=9), followed by vehicle collisions (n=2), unknown causes (n=2), and drowning (n=1). An additional ear-tagged, previously GPS-collared fox, died during a poaching incident in January 2017 at Robert Moses State Park.

Table 40. Proportion of GPS collared foxes known to have survived to the end of each monitoring period on Fire Island, NY. The total number of GPS collared foxes by site and year are shown in parentheses. Dashes indicate areas where no foxes were monitored during that time period.

Study Area	2015	2016	2017	2018
Robert Moses State Park + Lighthouse Tract	1.00 (5)	1.00 (6)	0.38[a] (8)	0.33[a,b] (3)
Fire Island National Seashore (Watch Hill to Old Inlet)	0.66 (3)	1.00 (6)	0.75 (4)	-
Smith Point County Park + Old Inlet East	0.00[a] (3)	-	-	-
Overall	0.64	1	0.50	0.33

[a]Populations that were affected by sarcoptic mange during the monitoring period

[b]GPS collar was removed early in the monitoring period, but den photos allowed us to confirm survival through the monitoring period

Mitochondrial genotyping results allowed us to estimate additional indices of survival by comparing individual detection histories across monitoring periods. Of the 32 unique genotypes sampled in 2016, 5 were detected again in 2017, yielding a 15.6% estimated probability of re-detection between those years. Of the 12 unique genotypes samples in 2017, 2 were sampled in 2018, yielding a 16.7% estimated probability of redetection between those years. Only one genotype that was sampled in 2016 was sampled again in 2018. However, this genotype was not sampled in 2017, suggesting an imperfect detection probability of individuals that survived across years. Overall, only 13.7% of unique genotypes were sampled in multiple years.

Phylogenetic origins of red foxes on Fire Island

Of the 277 samples that we sent to the MECU for genetic analyses, 221 were identified as originating from a red fox. Five samples were identified as originating from another species, and the remaining samples did not contain sufficient DNA amounts and/or quality to yield a source species identification. All 221 of these red fox samples were identified as 'F' haplotypes. This is a previously characterized North American haplotype indicative of foxes originating from southeastern Canada and the northeastern U.S. (Statham et al. 2012). From these results, we can conclude that foxes on Fire Island are not descended from purely European foxes brought over to North America for fur farming. However, this haplotype also is found in the descendants of foxes that were translocated from southeastern Canada and the northeastern U.S. for fur farming elsewhere. Thus, we

cannot yet determine whether foxes arrived on Fire Island as part of natural range expansion from farther north, or were the result of human-assisted translocations from farther north in North America. Because mitochondrial DNA is maternally inherited, there is still a possibility of European admixture within the population, even in the absence of European mitochondrial DNA haplotypes. Four of the samples that we sent had sufficient quality DNA for further analyses to explore this possibility, and our MECU collaborators are currently working on these analyses.

Discussion

To our knowledge, this is the first study to investigate red fox relative abundance, reproduction, survival, and mortality on a barrier island, and it is one of very few studies to investigate red fox occupancy in a barrier island ecosystem. The mange outbreaks in 2015 and 2017 provided a unique natural experiment in lieu of planned predator removal and appear to have driven all aspects of population ecology examined.

Trap success results indicated a substantial decrease in red fox detections following the mange outbreaks. Prior to the 2017 outbreak west of Old Inlet, foxes were the most commonly detected mesopredator species. Following the outbreak, our observed red fox trap success decreased by over 90% compared to peak detections. We observed a similar but lower decline during the first outbreak east of Old Inlet. This difference is likely because the 2015 outbreak east of Old Inlet had already begun prior to our 2015 camera grids, leading to lower starting estimates.

Our combined observations from GPS collaring, transect surveys, den monitoring, remote camera surveys, and genetic analyses of scat provide further evidence of a dramatic decline in red fox abundance and density over the course of the study, particularly west of Old Inlet. For example, minimum estimated abundance at Robert Moses State Park and the Lighthouse Tract decreased from 28 individuals (5.90 foxes/km^2) in 2016, to 3 individuals (0.63 foxes/km^2) in 2018. Similarly, minimum red fox abundance in the Fire Island National Seashore wilderness area (Watch Hill to Old Inlet) decreased from 22 individuals (4.64 foxes/km^2) in 2016 to 1 individual (0.21 foxes/km^2) 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/km^2) in 2016, and by August 2018, it had dropped to an estimated 7 individuals (0.42 foxes/km^2), 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 from that monitoring area, such that foxes were not detected east of Old Inlet from mid-April 2016 until early March 2017. In 2017 and 2018, we conservatively estimated 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 winter and overnight 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 of foxes and other animals crossing the bridge, as well as a photo of a red fox crossing Smith Point Bridge that was posted on the Long Island Wildlife Photography Facebook page on 17 January 2019.

Despite the observed overall decreases in trap success and minimum estimated abundance on the island, we observed very high occupancy in all years and sites where foxes were present. With the exception of the ~1-year period where red foxes apparently were absent east of Old Inlet, our occupancy estimates were all ≥0.8 at that site even though we suspect there was only 1 fox present. For comparison, Gieder (2015) reported occupancy estimates ≤0.5 across all years and sites in their study on Assateague Island, MD and an average occupancy estimate of 0.43(±0.04 SE). O'Connell et al. (2006) also reported lower red fox occupancy estimates on Cape Cod, MA (0.35±0.17 SE). These results suggest that the remaining or recolonizing foxes during and after mange outbreaks likely expanded their areas of use to absorb territories vacated by deceased foxes, leading to a lower overall population size but no significant decrease in occupancy. 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).

Combined with the long-distance movements observed in GPS-collared foxes (Part 3, Section 2), these results suggest that lethal removal may not be effective in reducing the occupancy of foxes on Fire Island, unless these efforts result in complete removal of foxes from the island. Such a complete removal is unlikely to be achieved by human activities when the fox population is not degressed, given the general trap-wariness of foxes, relatively high cost and effort associated with lethal removal, and public controversy surrounding lethal removal of foxes and other wildlife on Fire Island. In the event of an incomplete removal-related decrease in the fox abundance, the remaining foxes are likely to either disperse into any vacated territories or simply expand the borders of their own pre-existing territories to encompass newly available areas. Gieder (2015) found a similar lack of decrease in occupancy despite intensive red fox removal efforts on Assateague Island. Other studies have found that lethal removal could have the opposite effect, leading to increased occupancy via colonization of previously unoccupied sites and/or compensatory reproduction. For example, Towerton et al. (2011) found that foxes colonized previously unused sites following lethal removal. Similarly, previous studies of red fox population responses to lethal removal call into question whether removal efforts would succeed in reducing the overall population size, regardless of occupancy levels, due to compensatory immigration (Greentree et al.

2000, Baker and Harris 2006, Harding et al, 2001). There also is the possibility of increased reproduction among remaining individuals. For example, Marlow et al. (2016) observed significantly larger litter sizes following artificial decreases of~26% in fox density compared to litter sizes at a control site.

In addition to providing further support for the decrease in fox abundance through the study, our microsatellite genotyping results offer additional insights into survival and age structure of the fox population. In previous years, low resighting rates of ear-tagged foxes following GPS collar drop-off has limited our ability to estimate survival beyond the GPS monitoring period and across years. Our genetics results suggest that part of this may be due to relatively low survival rates across years, with more than 80% of sampled genotypes in a given year representing previously undetected individuals. Conversely, these low rates of repeated detection across years could be indicative of low detection rates, warranting further investigation. Our success in the use of microsatellite genotyping to identify individuals from scat samples highlights possibilities for further examination of red fox population dynamics on the island, including potential genetic mark-recapture abundance estimation and more robust estimates of survival. We also may be able to use these results to examine red fox family group composition in more detail by comparing individual genotypes and sex ratios from genotyped scats collected at dens.

Section 2: Spatial ecology of red foxes

This section describes data collection, analytical methods, and results related to the following objectives for red fox monitoring associated with the Fire Island Inlet to Moriches Inlet Fire Island Stabilization Project:

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.

Introduction

The red fox (*Vulpes vulpes*) has the largest geographic distribution of all terrestrial carnivores, inhabiting a wide variety of global ecosystems. Like many canids, red foxes are highly adaptable in their ecology, and can shift various aspects of their behavior in response to differing conditions (Goszczyński 2002). As a result, this species has been the focus of numerous ecological studies, leading to a large body of scientific knowledge surrounding the species and its ecosystem interactions. Many of these studies are focused on red fox spatial ecology and habitat use, spanning a wide variety of habitats, geographic locations, and ecological communities (*see* Lloyd 1980, Ables 1983, Cavallini 1996, Goszczyński 2002, Baker and Harris 2004 *and others for reviews*). However, there are relatively few published studies of red fox spatial ecology in coastal systems, particularly in North America.

Barrier islands are a dynamic type of coastal ecosystem, characterized by chains of long, narrow islands that run parallel to the mainland. These islands are frequently affected by storms, resulting in erosion and overwash, and may also be modified by shoreline stabilization efforts to protect human lives and property. Barrier islands provide important wildlife habitat, particularly for nesting and migrating shorebirds such as the federally threatened piping plover (*Charadrius melodus*). Red foxes have been shown to be a key predator of piping plovers on Fire Island, New York and in other barrier island ecosystems (Rimmer and Deblinger 1990, Patterson et al. 1991, Cohen et al. 2009, U.S. Fish and Wildlife Service 2014, Gieder 2015).

As the probability of spatiotemporal overlap between a predator species and its prey increases, so does the relative predation risk. Thus, it is important to understand the factors influencing the probability of spatiotemporal overlap between red foxes and piping plovers. In the context of predation management, information on red fox home range size, activity patterns, and habitat selection can be used to predict relative predation risk over space and time and identify high risk areas of likely overlap with piping plovers. This information can then be used to guide predator management activities as well as habitat management to increase suitability for piping plovers while decreasing suitability for red foxes.

Previous studies of red fox spatial ecology on barrier islands examined occupancy patterns and movements using remote cameras (Gieder 2015) and den site selection relative to available habitat (Krim et al. 1990), but do not address key components of spatial ecology such as home range size, activity patterns, or habitat selection. To our knowledge, there are no published, in-depth studies addressing these aspects of red fox spatial ecology in a barrier island ecosystem. Thus, it is unknown how the dynamic nature, small geographic extent, and surrounding aquatic movement barriers may impact red fox spatial ecology and potential spatial responses to predator management efforts on barrier islands.

We sought to fill these gaps in ecological knowledge on red fox ecology in barrier island ecosystems and collect information that could be used to create an island-wide predator management plan to aid piping plover recovery efforts. Our objectives were to estimate home range and core use area sizes, describe daily activity patterns, and investigate habitat selection among red foxes on Fire Island, New York, and identify factors influencing each of these aspects of red fox spatial ecology on the island.

Methods

Fox trapping, GPS collaring, and tracking

We used Victor 1 ½ Soft Catch and Bridger #2 padded foothold traps to capture foxes in each monitoring area in August–November 2015, April–May 2016, and February–April 2017 and 2018, following the Association of Fish and Wildlife Agencies' Best Management Practices for Trapping Red Foxes in the United States (AFWA 2014). Traps were set to require at least 2 lbs. of pressure to avoid capturing lighter animals and minimize the chance of capturing non-target species. Upon capture, we manually restrained foxes, determined their sex and breeding status, and collected morphometric measurements. When allowed by area managers, we also placed a colored plastic numeric ear-tag (Premier 1 Supplies, Washington, IA) or small metal ear-tag (National Band and Tag Company, Newport, KY) in the right ear, to aid in identifying individuals that were observed in the field and/or recaptured during subsequent trapping periods.

We outfitted up to 6 foxes per year in each monitoring area with GPS collars (Quantum 4000E, medium size; Telemetry Solutions, Concord, CA) to collect location data over the following months. In 2015, we programmed collars to record GPS locations once per hour, every day for 30 days from their deployment date, and then once every two hours, on every other day, from day 31 through programmed collar drop-off (end dates varied depending on deployment date). In 2016–2018, we programmed collars to record GPS locations once every 1–2 hours, every day from April 20th through the end of May, then once every 2 hours, on every other day until programmed collar drop-off in early August. This variation in sampling schedule allowed us to extend collar battery life and data collection through the entire piping plover breeding season in 2016–2018. Every 10–14 days after deployment, we located collared foxes via radio telemetry tracking of the collar's VHF signal and

approached to within 10–200m to connect wirelessly to the collar and download data using a UHF antenna, Quantum Base Station, and Collar SW v 2.09 software (Telemetry Solutions).

Home range estimation

We proofed location data to include only locations that were based on enough satellites to record a "3D" fix, as designated in the collar output data. We excluded locations >10m into the ocean surrounding the island, or with recorded elevation values well above the highest known point on the island, due to satellite errors. We then used the 'tlocoh' package in R (Version 3.5.2, www.r-project.org, accessed 15 May 2019) to estimate 50% and 95% utilization distribution isopleths using a Time Local Convex Hull (T-LoCoH) approach with the k-LoCoH parameterization (Lyons et al. 2013). Like the original local convex hull (LoCoH) method, T-LoCoH is a nonparametric approach that is well-suited to situations in which hard geographic boundaries constrain animal movements, as is often the case in barrier island ecosystems where the surrounding water is the barrier (Getz et al. 2007, Lyons et al. 2013). T-LoCoH improves upon previous LoCoH methods by incorporating time into the nearest neighbor selection and hull construction processes (Lyons et al. 2013). We followed Lyons' (2014) guidelines for interactive selection of scaling parameter *s* and number of nearest neighbors *k* at the individual level. We used the resulting 50% utilization distribution isopleths to represent each animal's core use area, and the resulting 95% utilization distribution isopleths to represent each animal's home range area. We also used these 95% distribution isopleths to classify animals as residents or transients for our later analyses, with residents maintaining relatively small, well-defined home ranges throughout the monitoring period and transients maintaining relatively large, shifting home ranges throughout the monitoring period. The 50% and 95% utilization distribution isopleths of these transient individuals are more appropriately described as biding areas and transient ranges, respectively, than core use and home range areas (Hinton et al. 2015, Morin and Kelly 2017).

Home range modeling

We used a generalized linear mixed modeling (GLMM) framework to compare the impacts of several covariates on core use area and home range size, and to identify factors influencing the amount of area used by an individual. To account for individual variation and multiple years of data from some foxes, we included a unique identifier for each animal (animal ID) as a random effect in all models. Categorical covariates examined included site and year. Binary covariates examined included season (fall/winter for collared foxes monitored in 2015, spring/summer for collared foxes monitored in all other years), whether or not that animal was associated with an active breeding den (based on concurrent den monitoring data and investigation of any location hotspots), and whether an individual was affected by sarcoptic mange during the monitoring period (based on regular observations of each fox's condition during trapping, data downloads, opportunistic sightings, and collar and carcass recovery for animals that died prior to the end of the planned monitoring period). Prior to modeling,

we tested for independence among covariates using Fisher's exact test (Fisher 1925, McDonald 2014). We tested all possible combinations of the covariates of interest, excluding models that included covariates found to be strongly correlated. We then used Akaike's Information Criterion corrected for small sample size (AIC_c) to evaluate and compare models (Burnham and Anderson 2007).

Activity patterns

We estimated hourly movement rates to serve as an index of red fox activity patterns, dividing the straight-line distance between successive locations by the time difference between those locations to calculate movement rates in meters/hour. We excluded movements between locations with a time difference <0.8 hours or >4 hours from these analyses, to mitigate potential bias due to duplicated locations or missed GPS fixes. The resulting movement rates were used to calculate an average movement rate for each individual for each hour of the day, corrected for seasonal differences in sunrise time.

Activity pattern modeling

We used a generalized linear mixed modeling (GLMM) framework to compare the impacts of several covariates on individual average hourly movement rates. We included animal ID as a random effect in all models, whether or not an animal was associated with an active breeding den, and whether or not an animal was affected by mange as categorical covariates, as described above. To account for varying diel activity levels, we examined the percent change in average movement rates across all individuals over sequential hours to separate activity observations into several distinct diel activity categories, consisting of multiple sequential hours with similar activity rates. These diel activity categories were included as a categorical covariate. Following the same modeling approach described above, we examined the effects of these covariates on hourly movement rates and identifies key factors influencing these rates.

Within-home range habitat selection

To investigate within home range (Johnson's third order) habitat selection, we compared used vs. available locations using a series of distanced-based resource selection function analyses (Johnson 1980, Conner et al. 2001, Manly et al. 2002). Distance-based resource selection approaches are more robust to location uncertainty than classification-based approaches, and allow for investigation of selection patterns that are more difficult to address using classification-based approaches, such as edge effects and selection for linear and point features (Conner et al. 2003). Because red foxes are territorial, not all areas are equally available to all individuals. Thus, we randomly simulated 5 locations per used location within a given animal's 100% minimum convex polygon (MCP), clipped to the extent of the island, to approximate availability at the individual level (Cooper and Millspaugh 1999, 2001).

We used high-resolution landcover classification layers created using annual 6-inch resolution aerial imagery to characterize habitat across the island (Part 2, Section 2). To minimize classification errors due to overlapping spectral signatures of anthropogenic and natural features, areas of human development (i.e. paved roads, parking lots, buildings, and residential areas) were delineated and separated out from the imagery prior to classification. The 'Maximum Likelihood Classification' tool in ArcGIS was then used to classify pixels into one of 5 distinct undeveloped habitat types: Water, wet sand, dry sand, evergreen vegetation (primarily pitch pine *Pinus rigida* woodland and maritime forest), and all other vegetation (primarily beachgrass, herbaceous vegetation, and deciduous shrubs). We created additional shapefiles in ArcGIS delineating off-road vehicle (ORV) roads on each side of Old Inlet. These routes are unpaved, primarily on dry sand, and are regularly used for access to recreation and residential areas. We hypothesized that they could serve as a useful travel corridor for foxes and other wildlife. Conversely, they could also present areas of increased risk of human encounters and vehicle-fox collisions.

We resampled the original 6-inch resolution classification layers to create 1m resolution classification layers for our analyses, assigning each 1m cell to the majority habitat type of the 6-inch cells that it contained. We used these coarser classification layers and the "Euclidean Distance" tool in ArcMap to create 1m resolution distance-to rasters for each habitat type and ORV roads. We then overlaid all used and random locations onto these distance rasters and extracted the estimated distances, and appended attribute data for individual foxes to create our input data for resource selection function analyses.

Using this input data, we used several sets of resource selection function analyses, with location data separated by the top supported factors in the results from our previous GLMM home range and activity pattern modeling. We did this to account for potential differences in habitat selection among groups with differing space use and movement rates, and to determine whether the same factors that most strongly influenced space use and movement rates also influenced habitat use. For all sets of analyses, we included animal ID as a random effect, tested all possible combinations of distance-to-habitat covariates, and used k-fold cross-validation to assess the predictive power of the top supported model (Boyce et al. 2002).

Landscape-scale habitat selection of red foxes

We also used distance-based resource selection function analysis to investigate landscape-scale (Johnson's second order) habitat selection of red foxes (Johnson 1980, Conner et al. 2001, Manly et al. 2002). While the habitat selection analyses described above were focused on how foxes select habitat within their home range (Johnson's third order), these landscape-scale analyses aimed to investigate how foxes select from the available areas, establishing that home range (Johnson 1980). By explicitly examining the factors influencing home range selection, we can use the resulting coefficients to infer the relative likelihood that a

given part of the study area will be included in one or more fox's home range, which can then be compared to known nesting areas and similar habitat suitability information for piping plovers. Because we were specifically interested in landscape-scale red fox habitat suitability at times when piping plovers are nesting and raising chicks, we limited this set of analyses to locations collected during the piping plover breeding season.

We compared used to available locations to identify factors influencing probability of use. We randomly simulated 5 locations within the relevant monitoring area per used location to approximate availability. When predicting habitat selection across the landscape, it is important to approximate availability in a way that appropriately reflects both the spatial extent of used locations and the sampling effort used to collect these used locations. To account for the fact that our trapping efforts were not evenly distributed across the island, we limited both the used locations in our analyses and the areas from which random points were drawn to the same areas that our trapping that efforts covered in corresponding years. These areas included the portion of Robert Moses State Park running from the volleyball courts west to Democrat Point, and the portion of Fire Island National Seashore's wilderness area running from the Bellport Beach pavilion to Old Inlet for both 2016 and 2017 analyses. The portion of Fire Island National Seashore's wilderness area from Watch Hill to the Bellport Beach Pavilion was included in 2016 analyses, and the western half of Robert Moses State Park's Field 5 and the entirety of Fire Island National Seashore's Lighthouse Tract was included in 2017 analyses.
We overlaid all used and random locations onto the same 1-meter resolution rasters described above, showing the distance from each pixel to the nearest pixel of a given habitat type or feature, for the 7 habitat types and features of interest (development, water, wet sand, dry sand, evergreen vegetation, all other vegetation, and off-road vehicle (ORV) roads). We used the "extract" function in the 'raster' R package (Version 2.9-23, https://rspatial.org/raster/, Accessed 14 November 2019) to extract and append distances from locations to each habitat type. We appended the same attribute data for individual foxes described above (sex, residency status, breeding status, mange status, year, and monitoring area) to create our input data for 2nd order resource selection function analyses.

We randomly sorted 75% of our created resource selection function input data into a training dataset for creating our models, leaving 25% as a testing dataset that we used to evaluate the predictive accuracy of our models. Prior to modeling, we tested for independence among covariates using Fisher's exact test and z-standardized continuous covariates (Fisher 1925, McDonald 2014). We used the "glmer" function in R package 'lme4' (Version 1.1-21, https://github.com/lme4/lme4, Accessed 14 November 2019) to run several mixed effects logistic regression models, with whether a location was used or random as the response. We compared a model that included all 7 habitat types and features of interest to a set of additional models, which included these 7 habitat types plus each possible combination of one of the above fox attribute covariates as either an additive or interactive effect. Interactive effects were included in models as interacting with all 7 habitat types.

We included animal ID as a random effect in all models, to account for individual variation in habitat selection. We used Akaike's Information Criterion corrected for small sample size (AIC_c) to evaluate and compare models (Burnham and Anderson 2007).

Island-wide habitat suitability predictions for red foxes

We used the "Raster Calculator" tool in ArcMap 10.6 (ESRI 2018, Redlands, CA)to scale each of our distance rasters using the same mean and standard deviation values that were used to scale the corresponding habitat covariates, then projected the coefficients and equation from our top supported model onto the study area, clipped to the extent of the island. As with the results from other resource selection function models, the values in this layer do not reflect the true probability of use, rather they are a measure of relative suitability (Keating and Cherry 2004, Morris et al. 2016). We divided the resulting layer into 10 equal area bins to allow for easier interpretation (Boyce et al, 2002, Morris et al. 2016). We used the 25% of the data that was not used to create the model to test the accuracy of our relative suitability prediction layers, comparing the predicted suitability in each binned area to the average relative suitability value of points falling in that bin. We repeated these steps using each year's landcover data and created separate maps for groups of foxes (i.e., males vs. females, residents vs. transients) that our modeling results suggested differed significantly in their habitat selection patterns.

Results

Fox trapping, GPS collaring, and tracking

We captured 52 unique foxes over 55 capture events during the study. Thirty-six of these foxes were outfitted with GPS collars and monitored until programmed collar drop off, or until the fox died prior to collar drop off (Table 41). Two foxes were recaptured, collared, and monitored over a second trapping period, resulting in multiple datasets from those individuals. Collar malfunctions resulted in no usable data from 2 foxes, and insufficient data for analysis from 2 foxes. Another fox disappeared after collaring, and was never located again despite intensive searching across the island, and thus yielded no usable data. We did not observe any animals successfully crossing Old Inlet during the monitoring period, suggesting that this inlet remains as effective barrier to fox movement between the eastern and western portions of the island.

Table 41. Habits, reproductive status, mange infection status, and fate of GPS collared foxes (n=38) on Fire Island, NY 2015-2018. F denotes females, N denotes males. Season abbreviations: F/W=Fall–Winter, SP/SU=Spring–Summer. Site abbreviations: RMFIL=Robert Moses State Park/Fire Island Lighthouse Tract, FINS=Fire Island National Seashore Wilderness Area (west of Old Inlet), SPOIE=Smith Point County Park/Fire Island National Seashore Wilderness Area (east of Old Inlet). Denning and mange abbreviations: Y=Yes, N=No.

Fox	Year	Season	Site	Residency	Denning	Mange	Fate
F-01	2015	F/W	RMFIL	Resident	N/A	N	Survived to end of monitoring period
M-01	2015	F/W	RMFIL	Transient	N/A	N	Survived to end of monitoring period
F-02	2015	F/W	RMFIL	Resident	N/A	N	Survived to end of monitoring period
M-02	2015	F/W	RMFIL	Transient	N/A	N	Survived to end of monitoring period
F-03[a]	2015	F/W	RMFIL	Resident	N/A	N	Survived to end of monitoring period
M-03	2015	F/W	SPOIE	Transient	N/A	Y	Died December 2015, mange
M-04	2015	F/W	FINS	Transient	N/A	N	Survived to end of monitoring period
M-05	2015	F/W	FINS	-	-	-	Disappeared after GPS collaring, no usable data
M-06	2015	F/W	FINS	Transient	N/A	N	Survived to end of monitoring period
M-07	2015	F/W	SPOIE	Transient	N/A	N	Died January 2016, likely vehicle collision
F-04	2015	F/W	SPOIE	Transient	N/A	Y	Died March 2016, mange
M-08	2016	SP/SU	RMFIL	Resident	Y	N	Survived to end of monitoring period
F-05	2016	SP/SU	RMFIL	Resident	Y	N	Survived to end of monitoring period
M-09	2016	SP/SU	RMFIL	Resident	Y	N	Collar malfunction, insufficient data for analyses
M-10	2016	SP/SU	RMFIL	Transient	N	N	Survived to end of monitoring period
F-07	2016	SP/SU	RMFIL	Resident	Y	N	Survived to end of monitoring period
M-11	2016	SP/SU	RMFIL	Transient	N	N	Survived to end of monitoring period
M-12	2016	SP/SU	FINS	-	-	-	Collar malfunction, no usable data
M-13	2016	SP/SU	FINS	Resident	N	N	Survived to end of monitoring period
M-14	2016	SP/SU	FINS	Resident	N	N	Survived to end of monitoring period
F-08	2016	SP/SU	FINS	Resident	N	N	Survived to end of monitoring period
F-09	2016	SP/SU	FINS	Resident	N	N	Survived to end of monitoring period

F-10	2016	SP/SU	FINS	Resident	N	N	Survived to end of monitoring period
M-15	2017	SP/SU	RMFIL	Transient	N	N	Died August 2017, likely vehicle collision
F-11	2017	SP/SU	RMFIL	Resident	N	N	Survived to end of monitoring period
F-03	2017	SP/SU	RMFIL	Resident	N	N	Survived to end of monitoring period
F-12	2017	SP/SU	RMFIL	Resident	N	N	Survived to end of monitoring period
M-08	2017	SP/SU	RMFIL	Resident	Y	N	Died August 2017, likely drowned
M-16	2017	SP/SU	RMFIL	Transient	N	Y	Died June 2017, unclear whether from mange or drowning
M-17	2017	SP/SU	RMFIL	Resident	Y	Y	Died May 2017, mange
F-13	2017	SP/SU	RMFIL	Resident	Y	Y	Died August 2017, mange
F-14	2017	SP/SU	FINS	Resident	N	N	Survived to end of monitoring period
M-18	2017	SP/SU	FINS	Transient	N	N	Survived to end of monitoring period
F-15	2017	SP/SU	FINS	Resident	Y	N	Survived to end of monitoring period
F-16	2017	SP/SU	FINS	-	-	N	Died April 2017 of unknown causes, insufficient data for analyses
M-19[b]	2018	SP/SU	RMFIL	Resident	N	Y	Died May 2018, likely mange
F-17[b]	2018	SP/SU	RMFIL	Resident	N	N	Died May 2018, unknown cause
F-18	2018	SP/SU	RMFIL	-	-	Y	Collar malfunction, no usable data

[a] <300 locations recorded due to low GPS fix success rate

[b]<300 locations recorded due to death early in monitoring period

A sarcoptic mange outbreak began east of Old Inlet shortly after the start of our Fall–Winter 2015 trapping efforts, leading to a substantial die-off. We reduced the location sampling frequency to one point per day for the last remaining GPS collared fox in this area starting on 20 January 2016 to extend the collar battery life and duration of data collection until her death on 24 March 2016. After this fox's death, foxes were apparently absent in that part of the study area, based on concurrent sign surveys and remote camera monitoring (K. Black and S. Karpanty, unpublished data), from April 2016–March 2017. From March 2017 until the end of monitoring activities, we observed sporadic sign in the eastern section of the island, suggesting a very low population density and likely movement to and from the mainland. We were unsuccessful in capturing any foxes east of Old Inlet following the mange outbreak. A second sarcoptic mange outbreak began west of Old Inlet during the Spring–Summer 2017 monitoring period, leading to a sharp decline in population density over

the next year in that part of the study area. Very few foxes remained during the 2018 trapping efforts, and we estimate that ≤5 foxes remained across all monitoring areas at the end of the 2018 monitoring period.

Home range estimation

Collared fox home range sizes (95% isopleths) varied widely, averaging 164.44 ha (33.31 ha SE) and ranging from 10.09 ha to 658.55 ha (Figure 40). Core use (50% isopleths) area sizes also varied, averaging 25.68 ha (8.51 ha SE) and ranging from 0.38 ha to 267.79 ha (Figure 41). Nineteen of 33 foxes, including both foxes tracked over multiple periods, maintained distinct territories throughout the monitoring period and were classified as residents, while 12 foxes, primarily nonbreeding males, ranged widely across the island on their respective sides of Old Inlet and moved back and forth between the inlets and/or villages regularly. The proportion of resident vs. transient males varied seasonally, with all males monitored in our fall/winter monitoring period (n=6) exhibiting transient behavior, while this proportion dropped to 45% of males (n=11) monitored in our spring/summer monitoring periods.

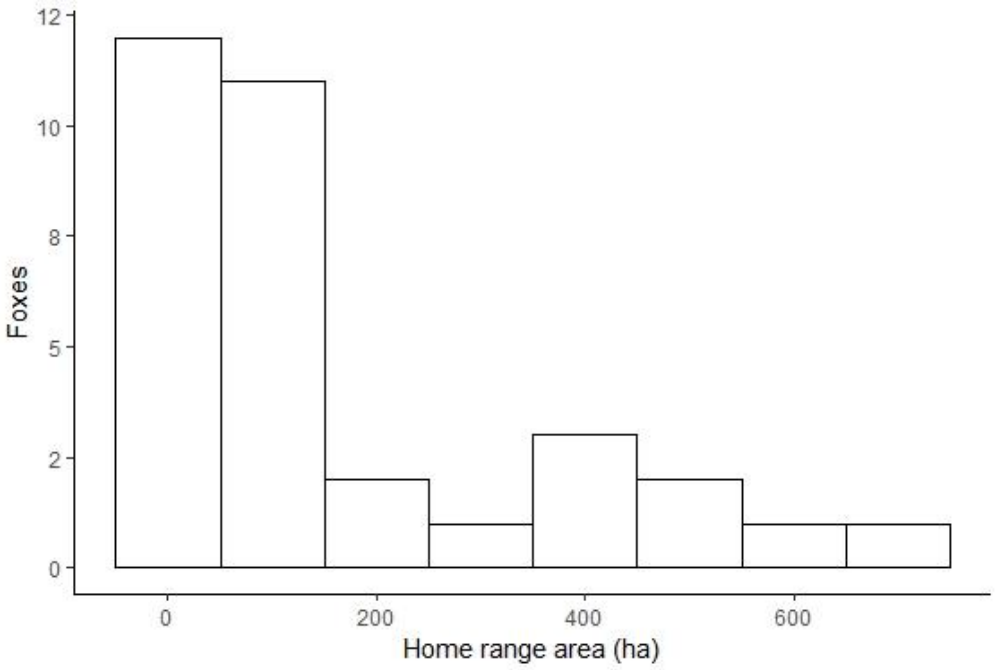


Figure 40. Home/transient range areas estimated for all GPS collared foxes (n=33) monitored on Fire Island, New York in 2015–2018.

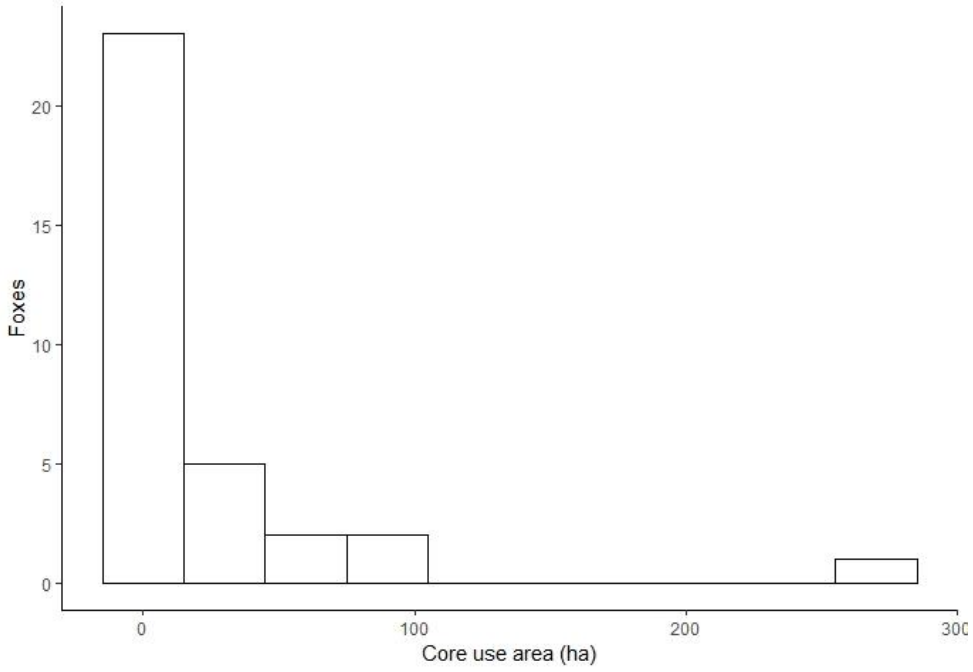


Figure 41. Core use/biding areas estimated for all GPS collared foxes (n=33) monitored on Fire Island, New York in 2015–2018.

Home range modeling

GLMM analyses identified season and sex as primary factors influencing home range size (Table 42, Table 43), and season and mange as primary factors influencing core use area size (Table 44, Table 45). Home ranges and core use areas were both larger in the fall/winter monitoring period than in spring/summer monitoring periods (Figure 42, Figure 43). Male foxes used larger home ranges than females, but core use area did not differ significantly between the sexes. Foxes affected by mange used similarly sized home ranges to healthy foxes, but used larger core areas. Mange and breeding appeared in competing models for home range size, but did not show significant directional trends. Similarly, breeding and sex appeared in competing models for core use area size, but did not show significant directional trends.

Table 42. Model selection table for GLMM analysis of collared fox home/transient range size in relation to covariates of interest. All models also included animal identity as a random effect. Only models within 4 ΔAIC_c of the top model are shown.

Model	AIC_c	ΔAIC_c	Weight
Season + Sex	59.8	0	0.596
Season + Sex + Mange	61.5	1.69	0.256
Season + Sex + Denning	62.6	2.79	0.148

Table 43. Parameter beta coefficient estimates, standard errors, and 95% confidence intervals from top model of collared fox home/transient range size in relation to covariates of interest.

Covariate	Estimate	SE	Lower CI	Upper CI
Season (spring/summer)	-0.501	0.008	-0.518	-0.485
Sex (M)	0.619	0.196	0.235	1.001

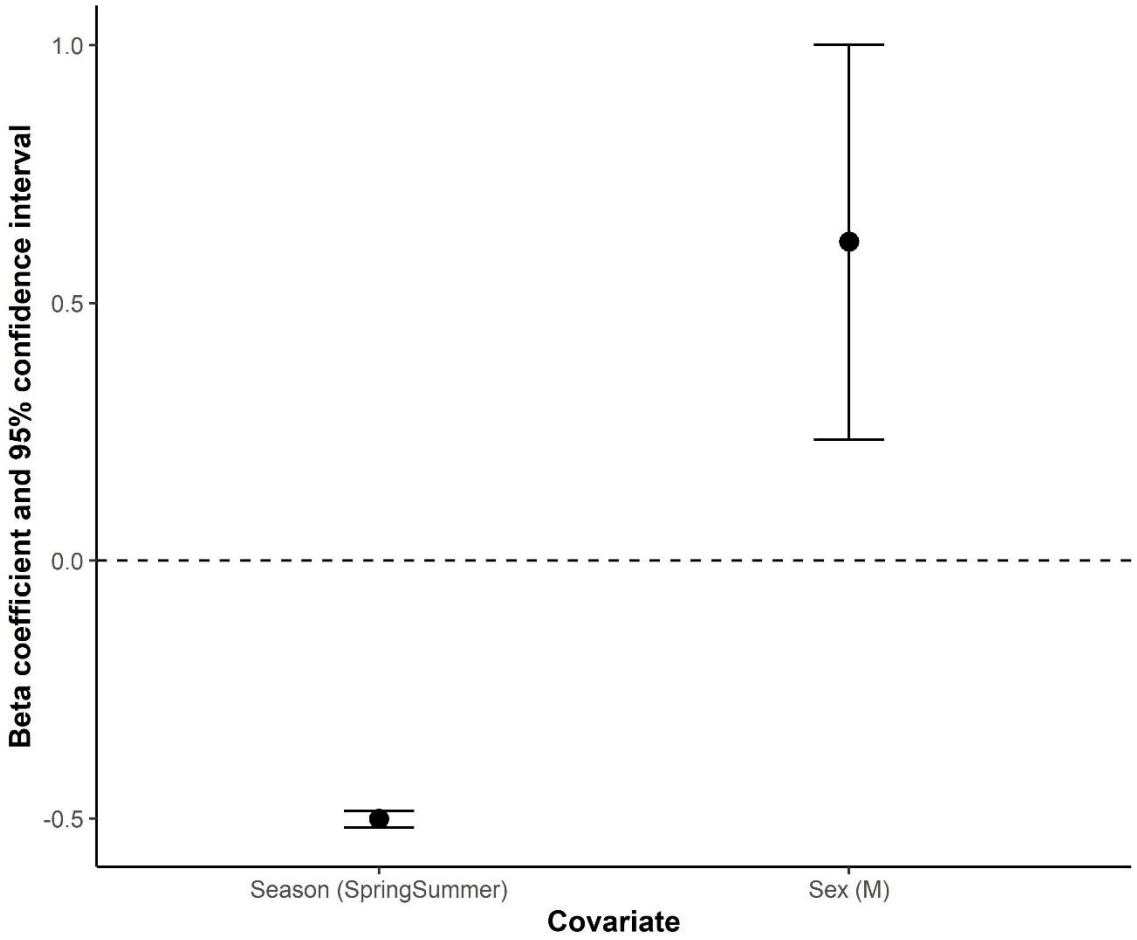


Figure 42. Parameter beta coefficient estimates and 95% confidence intervals from top model of collared fox home/transient range size in relation to covariates of interest.

Table 44. Model selection table for GLMM analysis of core use/biding area size in relation to covariates of interest. All models also included animal identity as a random effect. Only models within 4 ΔAIC_c of the top model are shown.

Model	AIC_c	ΔAIC_c	Weight
Mange + Season	62.8	0.00	0.503
Mange + Season + Denning	64.8	2.02	0.183
Mange + Season + Sex	65.0	2.22	0.166
Season	65.2	2.44	0.148

Table 45. Parameter estimates, standard errors, and 95% confidence intervals from top model of collared fox core use/biding area size in relation to covariates of interest.

Covariate	Estimate	SE	Lower CI	Upper CI
Mange (Y)	0.839	0.335	0.183	1.495
Season (spring/summer)	-0.325	0.051	-0.424	-0.224

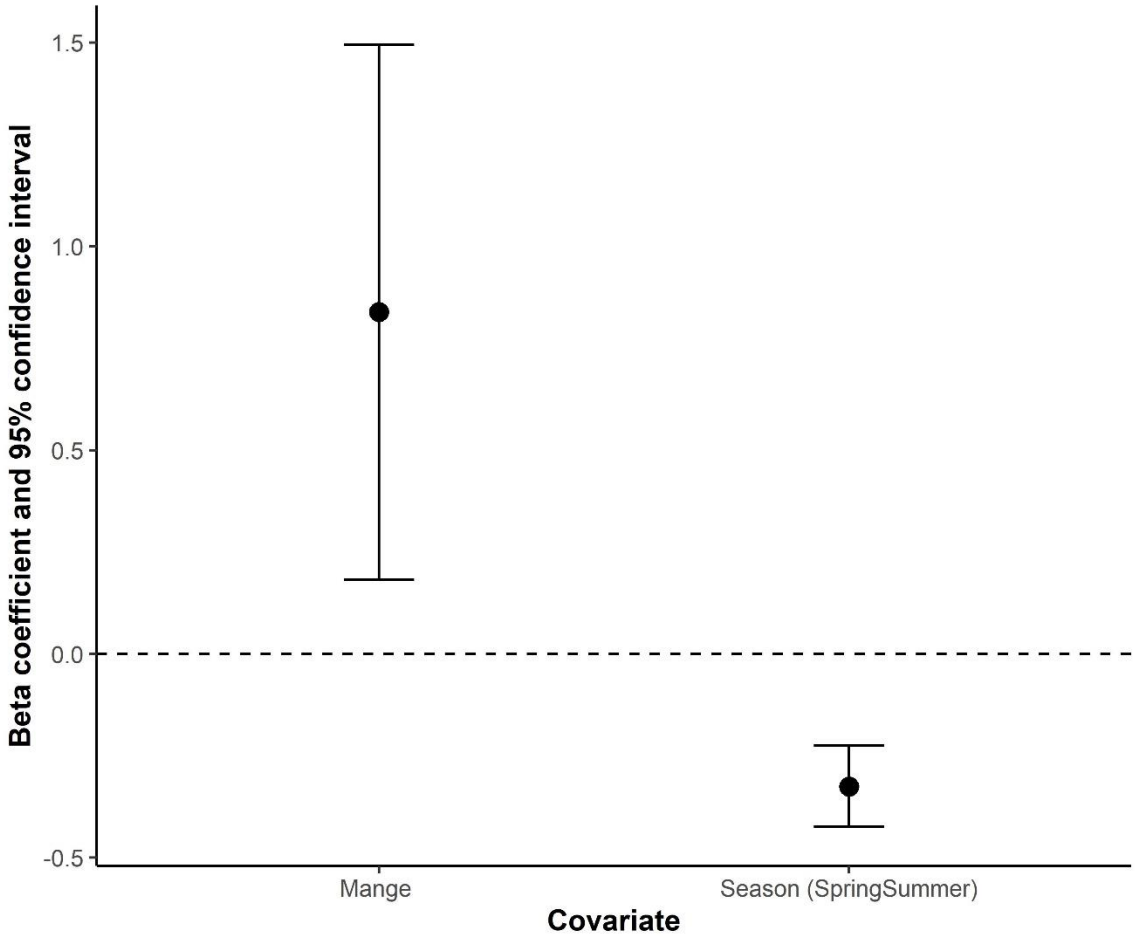


Figure 43. Parameter estimates and 95% confidence intervals from top model of collared fox core use/biding area size in relation to covariates of interest.

Activity patterns

We included residency (whether that animal was a resident or a transient) as a categorical covariate, based on our home range estimation results as an explanatory variable in activity analyses. Residency was strongly related to both sex (Fisher's exact test $p<0.001$) and season ($p<0.05$) but better explained variation in home range sizes in post hoc comparisons, and variation in activity and habitat selection in preliminary analyses. Average hourly movement rates varied across hours and individuals, ranging from 1.14 to 1766.61 m/hr and averaging 216.33 (SE 8.71) m/hr across all hours and individuals. Our comparison of the percent change in average movement rates across all individuals over sequential hours identified 3 distinct diel activity categories: a daytime period of relatively low activity (1–9 hours after sunrise), a nighttime period of relatively high activity (13–22 hours after sunrise), and a transition category for hours between these two periods characterized by moderate but rapidly changing activity levels (all other hours) (Figure 44).

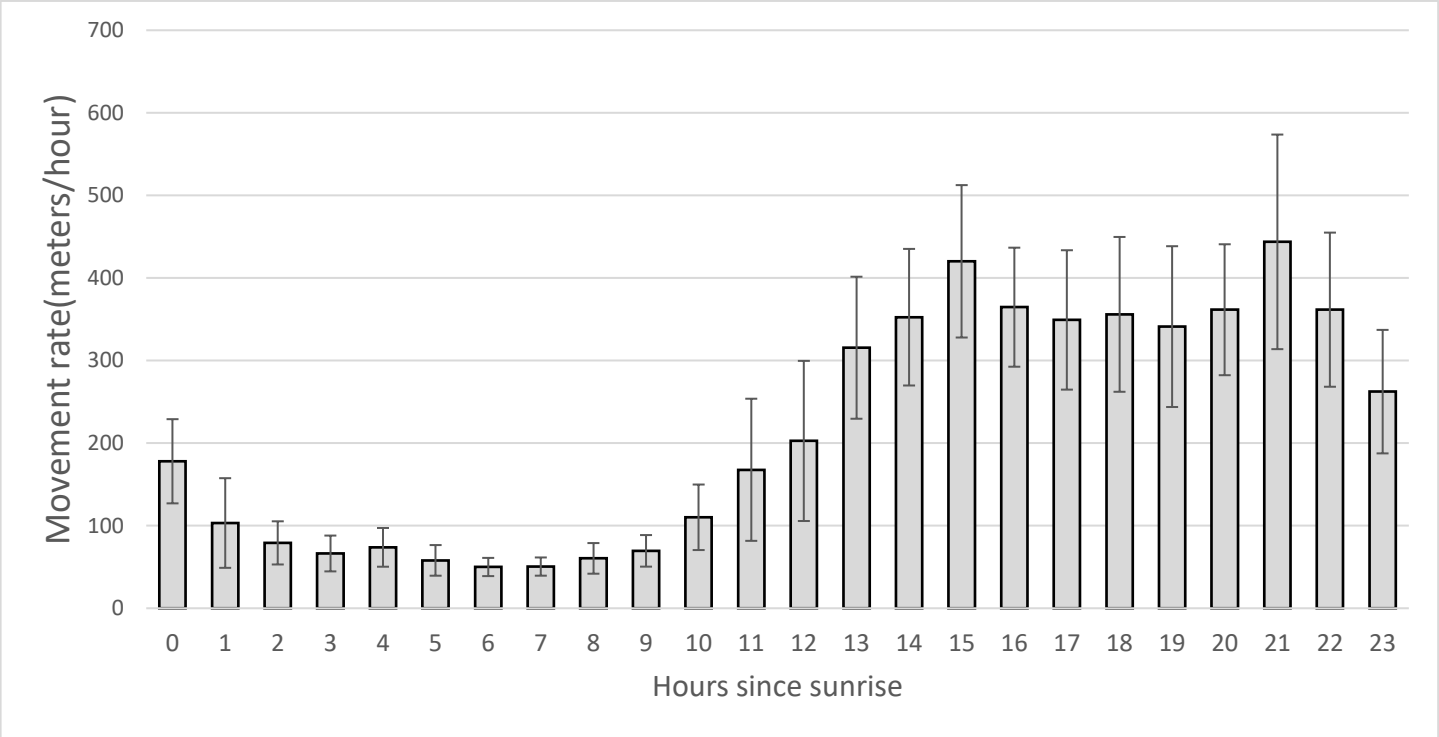


Figure 44. Mean hourly movement rates and 95% confidence intervals across all GPS collared foxes (n=33) monitored on Fire Island, New York in 2015–2018.

Activity pattern modeling

We found that the diel period, residency, and mange all were related to movement rates (Table 46, Table 47, Figure 45). Mean observed hourly fox movement rates were highest during the nighttime period (mean=366.5 m/hour, SE=14.8m/hour), lower during the transition period (mean=184.2 m/hour, SE=16.8 m/hour), and lowest during the daytime period (mean=67.8 m/hour, SE=4.41m/hour). For residency, transient foxes had higher mean observed hourly movement rates across all hours (mean=331.8 m/hour, SE=19.54 m/hour) compared to residents (mean=149.7 m/hour, SE=6.11 m/hour). For mange, infected foxes (mean=317.4 m/hour, SE=27.3 m/hour) had higher mean observed hourly movement rates across all hours compared to non-infected foxes (mean=194.3 m/hour, SE=8.5 m/hour).

Table 46. Model selection table for GLMM analysis of hourly movement rates in relation to covariates of interest. All models also included animal identity as a random effect. Only models within 4 ΔAIC_c of the top model are shown.

Model	AIC_c	ΔAIC_c	Weight
Residency + Period + Mange	62.80	0.00	0.503
Residency + Period	64.80	2.02	0.183

Table 47. Parameter estimates, standard errors, and 95% confidence intervals from top model of hourly movement rates in relation to covariates of interest.

Covariate	Estimate	SE	Lower CI	Upper CI
Residency (Transient)	0.288	0.069	0.152	0.424
Mange (Yes)	0.192	0.085	0.025	0.359
Period (Night)	0.863	0.031	0.802	0.923
Period (Transition)	0.436	0.037	0.268	0.603

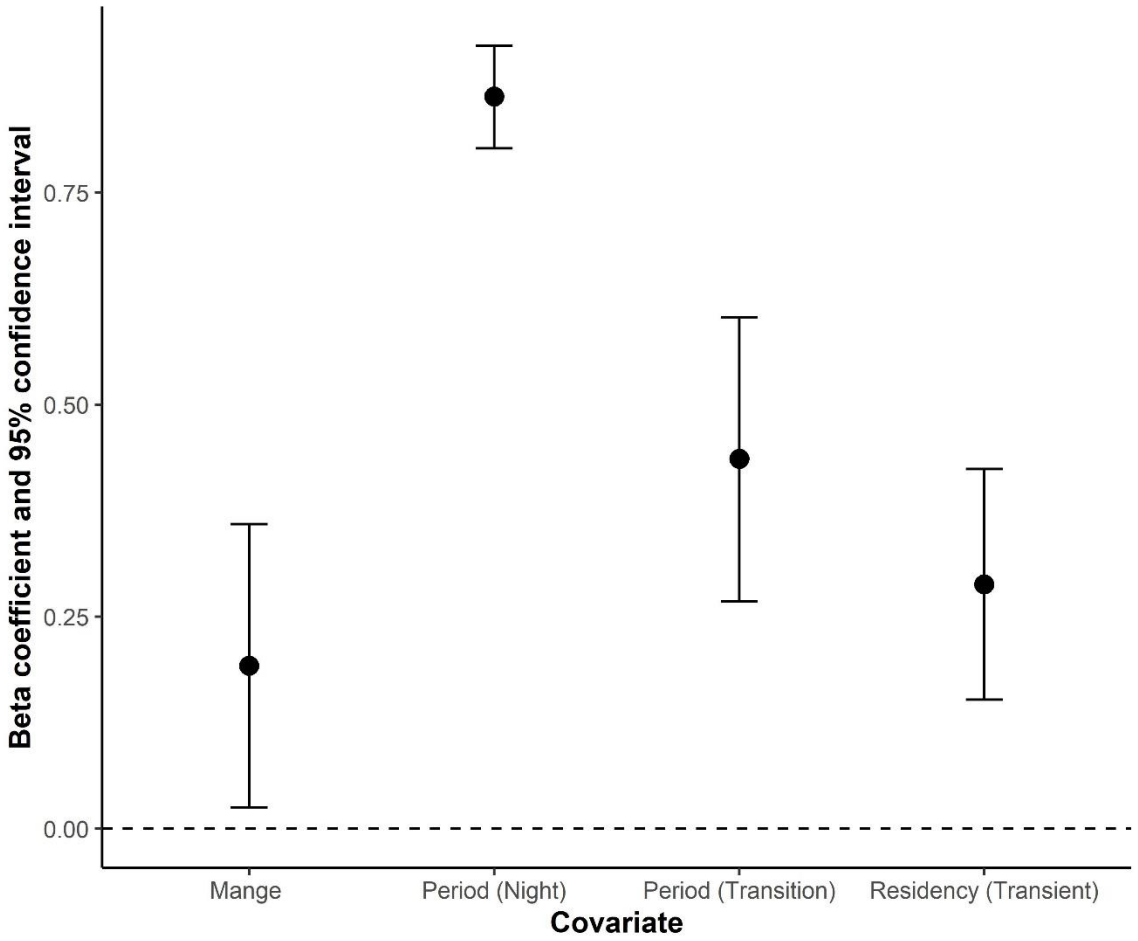


Figure 45. Parameter estimates and 95% confidence intervals from top model of hourly movement rates in relation to covariates of interest.

Within-home range habitat selection

Based on the results of our home range and activity pattern modeling, we split out within-home range habitat selection analyses by diel activity period, residency, and mange status. For our first set of analyses, we separated locations into daytime, nighttime, and transition times. For our second set of analyses, we separated out locations from residents and transients. For our 3rd set of analyses, we separated out locations from animals with and without mange.

Top models varied among our analyses, but generally included the global model or the global model minus one habitat covariate (Table 48, Table 49, Table 50). Overall, foxes selected for areas closer to wet sand, dry sand, and evergreen vegetation and further from ORV roads and water, with varying selection for or against development and vegetation (Table 51, Table 52, Table 53, Figure 46, Figure 47, Figure 48). Of these factors, proximity to dry sand and avoidance of ORV roads consistently had the strongest effects on probability of use.

There was variation in habitat selection based on diel activity period, residency, and mange status. During the day and transition times, probability of use increased as distance to vegetation decreased, while at night, probability of use increased as distance to vegetation increased. Similarly, during the day and transition times, there was no selection for or against developed habitat, but at night, probability of use increased slightly as distance to developed habitat decreased (Table 51, Figure 46). Selection patterns also differed somewhat among resident and transient animals, with residents selecting areas closer to vegetation and transients selecting areas further than vegetation. Transients also showed less avoidance of ORV roads than residents (Table 52, Figure 47). Similarly, animals affected by mange showed much less avoidance of ORV roads than animals without mange (Table 53, Figure 48).

Table 48. Model selection table for resource selection function analyses of locations recorded during each diel activity period. All models also included animal identity as a random effect. Only models within 4 ΔAIC_c of the top model are shown.

Period	Model	AIC_c	ΔAIC_c	Weight
Day	ORV road + water + wet sand + dry sand+ vegetation	55822.1	0.00	0.726
	ORV road + water + wet sand + dry sand+ vegetation + developed	55824.1	1.95	0.274
Transition	ORV road + vegetation + water + wet sand	35924.2	0.00	0.731
	ORV road + vegetation + water + wet sand + developed	35926.2	2.00	0.269
Night	ORV road + vegetation + water + wet sand	67792.3	0.00	0.674
	ORV road + vegetation + water + wet sand + developed	67793.8	1.45	0.326

Table 49. Model selection table for resource selection function analyses of locations recorded from resident vs. transient foxes. All models also included animal identity as a random effect. Only models within 4 ΔAIC_c of the top model are shown.

Residency	Model	AIC_c	ΔAIC_c	Weight
Resident	ORV road + water + wet sand + dry sand + evergreen+ vegetation	94978.2	0.00	0.646
	ORV road + water + wet sand + dry sand + evergreen+ vegetation + developed	94979.4	1.20	0.354
Transient	ORV road + water + wet sand + dry sand + evergreen+ vegetation + developed	66200.9	0.00	0.789
	ORV road + water + wet sand + dry sand + vegetation + developed	66203.5	2.64	0.211

Table 50. Model selection table for resource selection function analyses of locations recorded from foxes with mange vs. without mange. All models also included animal identity as a random effect. Only models within 4 ΔAIC_c of the top model are shown.

Mange	Model	AIC_c	ΔAIC_c	Weight
Yes	ORV road + water + wet sand + dry sand + evergreen+ vegetation	23480.8	0.000	0.362
	ORV road + water + wet sand + dry sand + vegetation	23481.1	0.30	0.311
	ORV road + water + wet sand + dry sand + evergreen+ vegetation + developed	23482.3	1.55	0.167
	ORV road + water + wet sand + dry sand + vegetation + developed	23487.9	2.21	0.120
No	ORV road + water + wet sand + dry sand + evergreen+ vegetation + developed	138486.9	0.00	0.819
	ORV road + water + wet sand + dry sand + developed	138489.9	3.01	0.181

Table 51. Parameter estimates, standard errors, and 95% confidence intervals from top resource selection function models for locations recorded during each diel activity period.

Period	Covariate	Estimate	SE	Lower CI	Upper CI
Day	ORV road	0.767	0.022	0.725	0.809
	Water	0.717	0.023	0.672	0.763
	Wet sand	-0.569	0.027	-0.621	-0.517
	Dry sand	-0.284	0.021	-0.326	-0.242
	Evergreen	-0.200	0.017	-0.234	-0.167
	Vegetation	-0.643	0.028	-0.698	-0.589
Transition	ORV road	0.827	0.029	0.771	0.883
	Water	0.420	0.025	0.371	0.470
	Wet sand	-0.259	0.029	-0.315	-0.203
	Dry sand	-0.591	0.029	-0.648	-0.534
	Evergreen	-0.146	0.021	-0.187	-0.105
	Vegetation	-0.225	0.027	-0.278	-0.172
Night	ORV road	0.814	0.022	0.772	0.857
	Wet sand	-0.296	0.013	-0.322	-0.270
	Dry sand	-0.751	0.021	-0.792	-0.709
	Evergreen	-0.057	0.014	-0.084	-0.030
	Vegetation	0.144	0.014	0.117	0.170
	Developed	-0.066	0.015	-0.096	-0.036

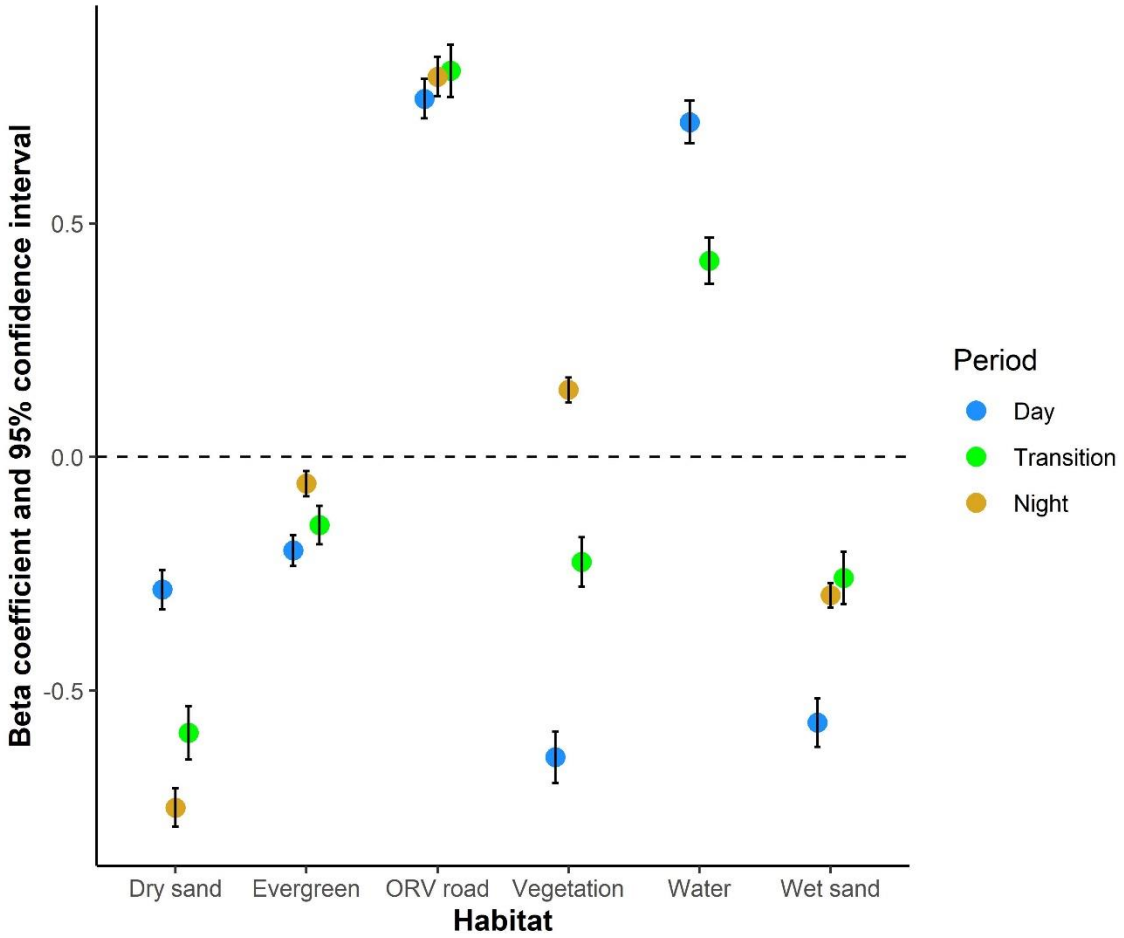


Figure 46. Parameter estimates and 95% confidence intervals from top resource selection function models for locations recorded during each diel activity period.

Table 52. Parameter estimates, standard errors, and 95% confidence intervals from top resource selection function models for locations from resident vs. transient foxes.

Residency	Covariate	Estimate	SE	Lower CI	Upper CI
Resident	ORV road	1.485	0.052	1.383	1.587
	Water	0.199	0.016	0.168	0.229
	Wet sand	-0.305	0.019	-0.341	-0.269
	Dry sand	-0.626	0.018	-0.661	-0.590
	Evergreen	-0.284	0.014	-0.312	-0.256
	Vegetation	-0.159	0.015	-0.187	-0.130
Transient	ORV road	0.627	0.011	0.605	0.649
	Water	0.405	0.022	0.363	0.447
	Wet sand	-0.351	0.024	-0.397	-0.305
	Dry sand	-0.663	0.021	-0.703	-0.623
	Evergreen	-0.029	0.014	-0.056	-0.003
	Vegetation	0.116	0.017	0.082	0.149
	Developed	-0.051	0.013	-0.077	-0.026

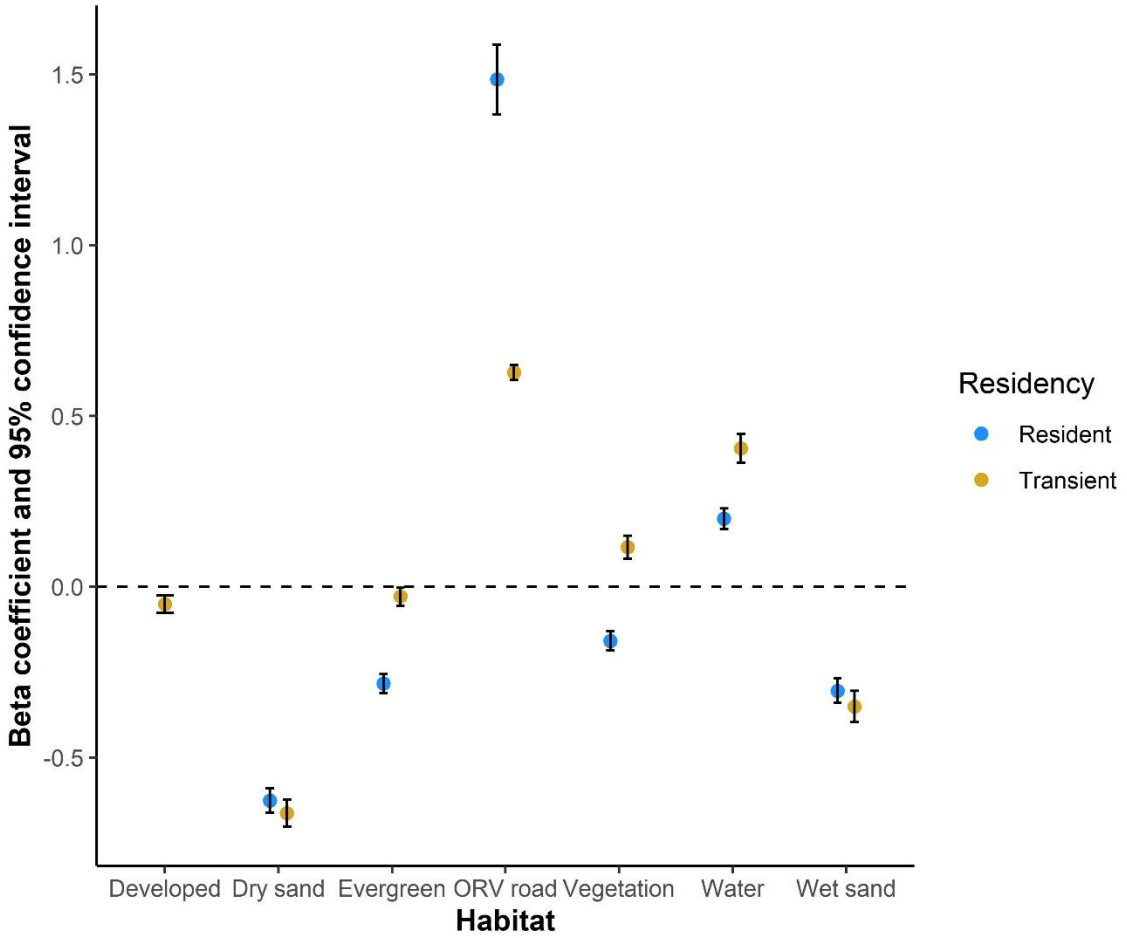


Figure 47. Parameter estimates and 95% confidence intervals from top resource selection function models for locations from resident vs. transient foxes.

Table 53. Parameter estimates, standard errors, and 95% confidence intervals from top resource selection function models for locations from foxes with vs. without mange.

Mange	Covariate	Estimate	SE	Lower CI	Upper CI
Yes	ORV road	0.052	0.016	0.021	0.084
	Water	0.143	0.045	0.055	0.231
	Wet sand	-0.150	0.046	-0.240	-0.059
	Dry sand	-0.549	0.033	-0.614	-0.484
	Evergreen	0.030	0.019	-0.008	0.068
	Vegetation	-0.185	0.025	-0.233	-0.136
No	ORV road	0.945	0.016	0.914	0.977
	Water	0.323	0.013	0.298	0.349
	Wet sand	-0.349	0.015	-0.378	-0.319
	Dry sand	-0.601	0.015	-0.629	-0.572
	Evergreen	-0.159	0.011	-0.181	-0.137
	Vegetation	-0.026	0.012	-0.049	-0.003
	Developed	-0.065	0.011	-0.088	-0.043

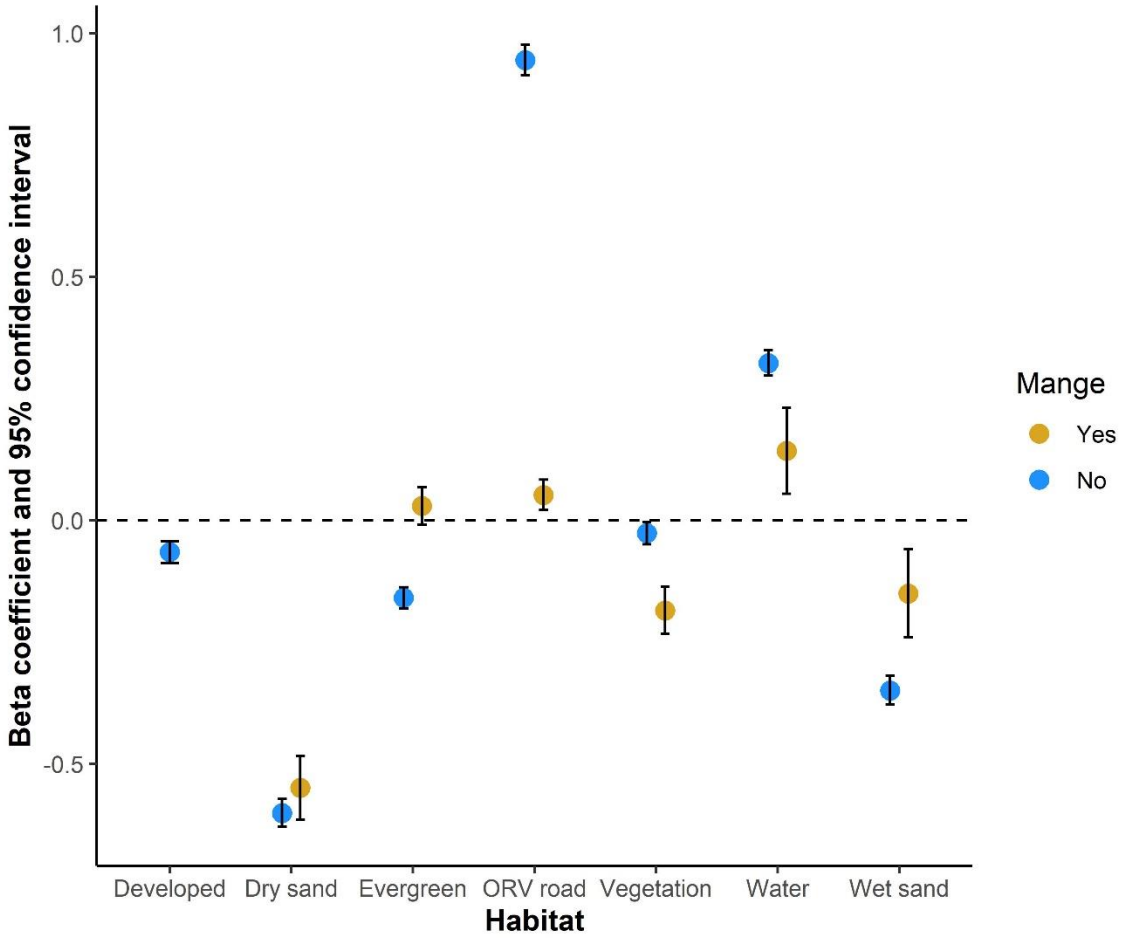


Figure 48. Parameter estimates and 95% confidence intervals from top resource selection function models for locations from foxes with vs. without mange.

Landscape-scale habitat selection of red foxes

Landscape-scale Selection patterns differed for some, but not all, habitat types by sex (Table 54, Table 55, Figure 10). Male foxes used areas closer to dry sand followed by areas closer to ORV roads at the landscape level. Female foxes similarly used areas closer to dry sand but used areas farther from ORV roads (Table 55, Figure 49. Males selected areas farther from wet sand and slightly closer to water and development, while females showed similar or somewhat stronger selection in the opposing direction for these habitats (Table 55, Figure 49). Males showed statistically insignificant selection for areas closer to evergreen habitat, while females showed stronger selection in the same direction (Table 55, Figure 49). Both sexes showed very similar selection for areas closer to vegetation (Table 55, Figure 49).

Table 54. Model selection table for landscape-level resource selection function analyses for red foxes on Fire Island, NY in 2016-2017. “Predictors” refers to the distances to the 6 habitat types and one habitat feature of interest (development, dry sand, wet sand, evergreen, other vegetation, water, and ORV roads), which were included in all but the null model. All models also included animal ID as a random effect.

Model	K	AIC_c	ΔAIC_c	Model weight
Predictors*Sex	16	57664.30	0.00	1.0
Predictors*Site	23	58729.80	1065.53	0.0
Predictors*Year	16	58960.80	1296.50	0.0
Predictors*Mange	16	59649.50	1985.26	0.0
Predictors*Breeding	16	59683.90	2019.62	0.0
Predictors*Residency	16	59900.90	2236.64	0.0
Predictors + Sex	10	60146.50	2482.25	0.0
Predictors	9	60147.50	2483.24	0.0
Predictors + Mange	10	60147.70	2483.40	0.0
Predictors + Year	10	60148.10	2483.82	0.0
Predictors + Site	11	60148.70	2484.41	0.0
Predictors + Breeding	10	60149.10	2484.85	0.0
Predictors + Residency	10	60149.40	2485.14	0.0
Null model	2	65313.30	7649.00	0.0

Table 55. Standardized beta coefficient estimates and 95% confidence intervals from top supported model for red fox landscape-level (second order) habitat selection on Fire Island, NY 2016-2017. Negative values indicate selection for areas closer to a given habitat type than expected, while positive values indicate selection for areas further from a given habitat type than expected. Confidence intervals overlapping zero indicate no clear directional selection pattern.

Habitat	Sex	Estimate	SE	Lower CI	Upper CI
ORV	F	1.454	0.040	1.377	1.532
	M	-0.868	0.042	-0.950	-0.785
Developed	F	0.034	0.021	-0.008	0.076
	M	-0.049	0.025	-0.097	-0.001
Dry sand	F	-1.071	0.040	-1.150	-0.992
	M	-1.195	0.047	-1.287	-1.103
Wet sand	F	-0.609	0.038	-0.683	-0.535
	M	0.074	0.025	0.025	0.123
Evergreen	F	-0.442	0.025	-0.492	-0.392
	M	-0.049	0.029	-0.105	0.008
Veg	F	-0.422	0.031	-0.481	-0.362
	M	-0.381	0.041	-0.461	-0.301
Water	F	0.447	0.031	0.386	0.508
	M	-0.058	0.025	-0.108	-0.008

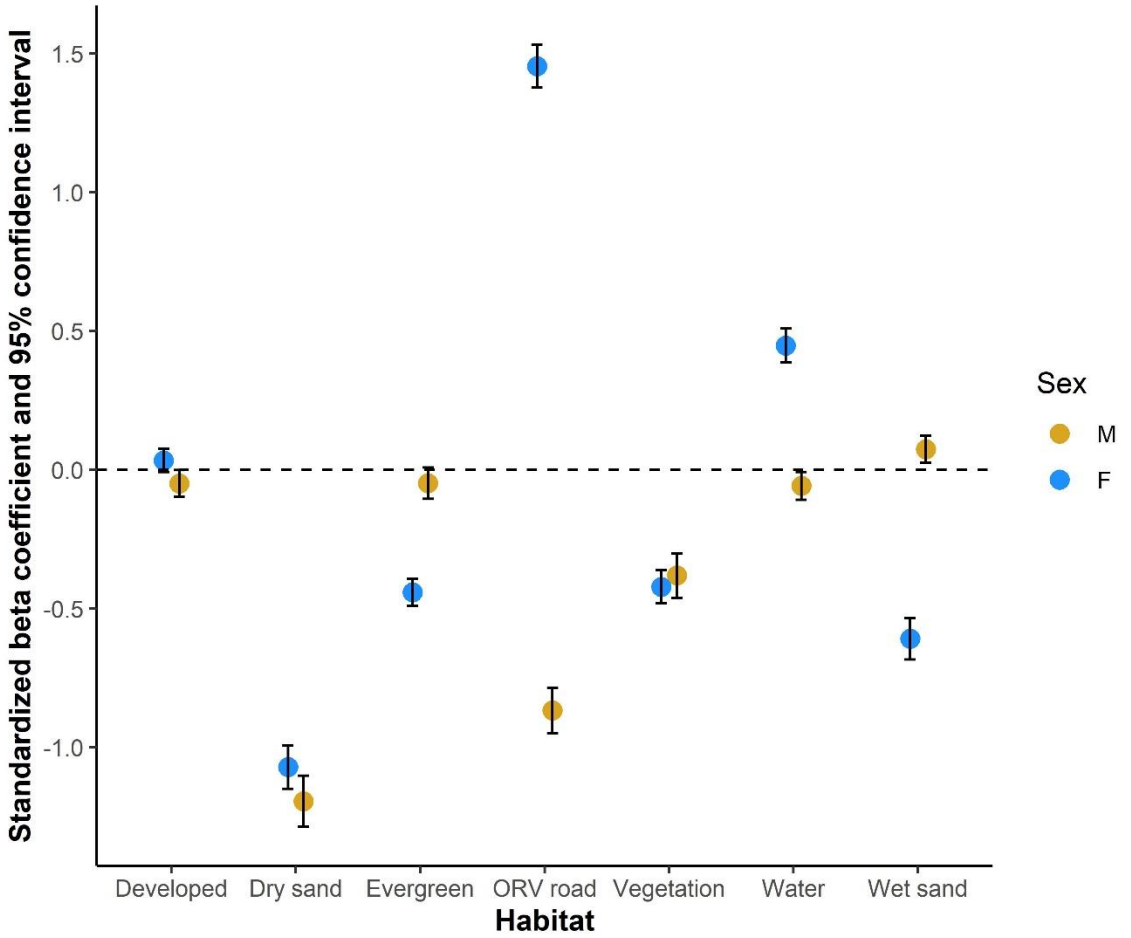


Figure 49. Standardized beta coefficient estimates and 95% confidence intervals from top supported model for red fox landscape-level (second order) habitat selection on Fire Island, NY 2016-2017. Negative values indicate selection for areas closer to a given habitat type than expected, while positive values indicate selection for areas further from a given habitat type than expected. Confidence intervals overlapping zero indicate no clear directional selection pattern.

Island-wide habitat suitability predictions for red foxes

Based on the results of our second-order, landscape-level resource selection function analyses, we created separate suitability maps for males and females, using both 2016 and 2017 landcover data. Maps from 2017 for each monitoring area are shown in Figure 50, Figure 51, and Figure 52 as examples of the those produced that will be linked with plover suitability maps in future output related to this project. Maps from 2016 are included in Appendix G. Predicted suitability maps were very similar between years, but varied noticeably between sexes, particularly within the Fire Island National Seashore wilderness area, which was predicted to be much more suitable for females than males, due to female avoidance of ORV roads in comparison to male selection for ORV roads. Predictive accuracy varied among each of these maps, ranging from 0.65 to 0.91 and averaging 0.75 across all maps (Table 56).

Table 56. Model predictive accuracy for red foxes on Fire Island, NY in 2016-2017 for each predictive habitat map created.

Year	Sex	Predictive accuracy
2016	Male	0.73
2016	Female	0.72
2017	Male	0.65
2017	Female	0.91

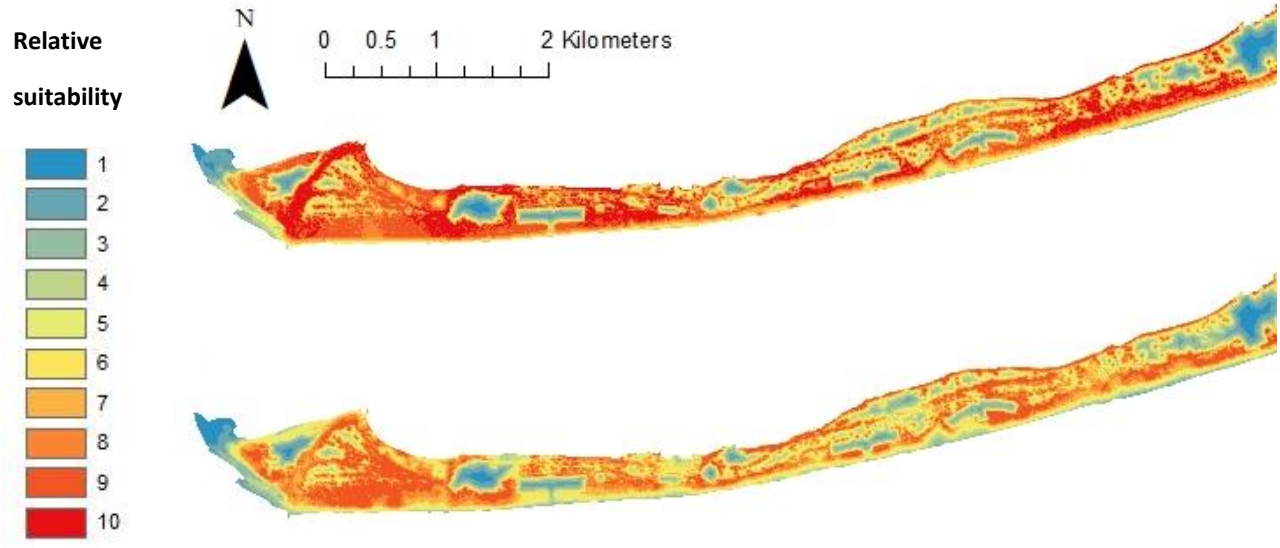


Figure 50. Predicted landscape-scale habitat suitability (10 = highest suitability to 1 = lowest suitability) maps for male (top) and female (bottom) red foxes at Robert Moses State Park and Fire Island National Seashore's Lighthouse Tract, using 2017 landcover classification data.

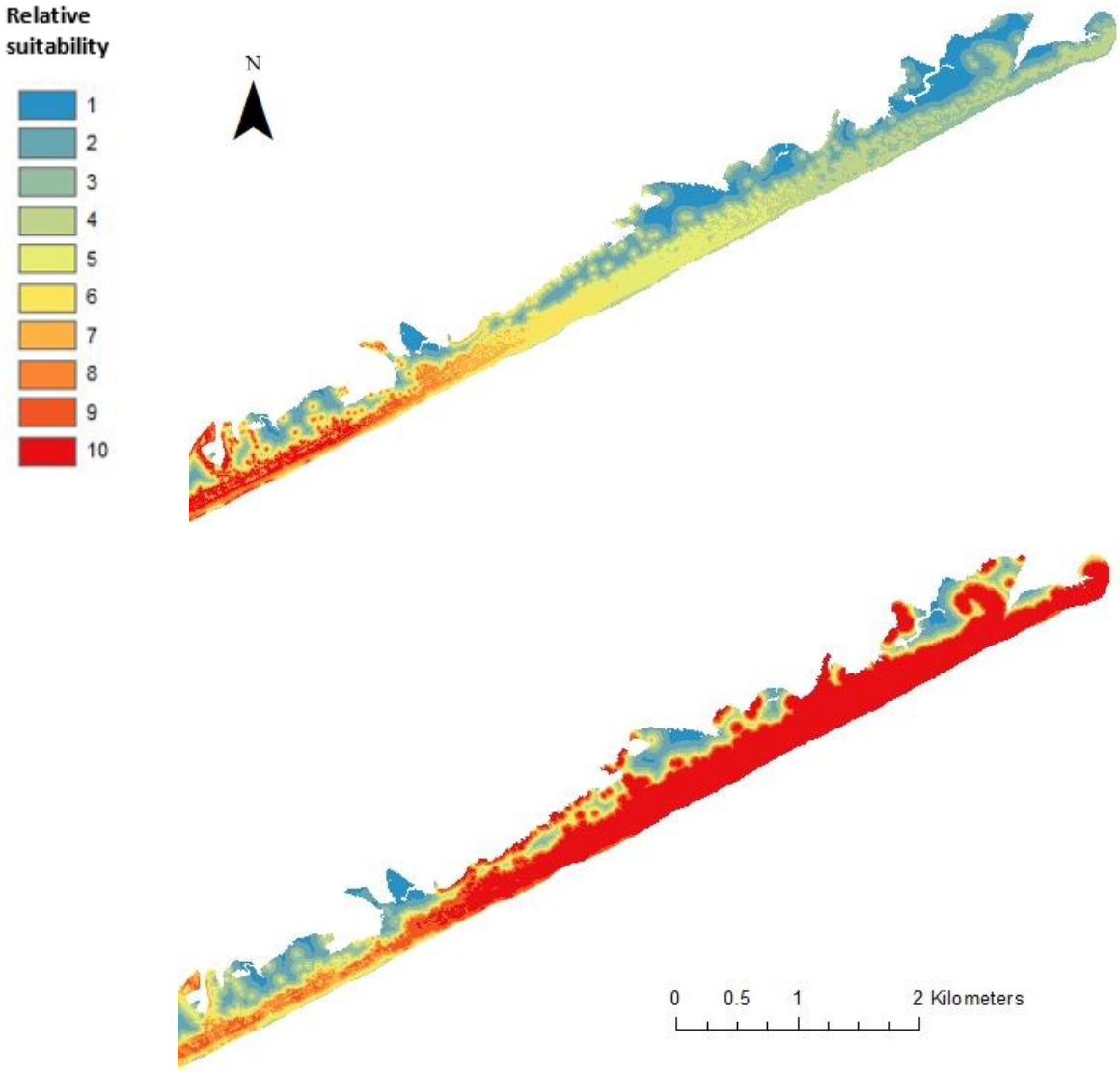


Figure 51. Predicted landscape-scale habitat suitability (10 = highest suitability to 1 = lowest suitability) maps for male (top) and female (bottom) red foxes at Fire Island National Seashore (Watch Hill to Old Inlet), using 2017 landcover classification data.

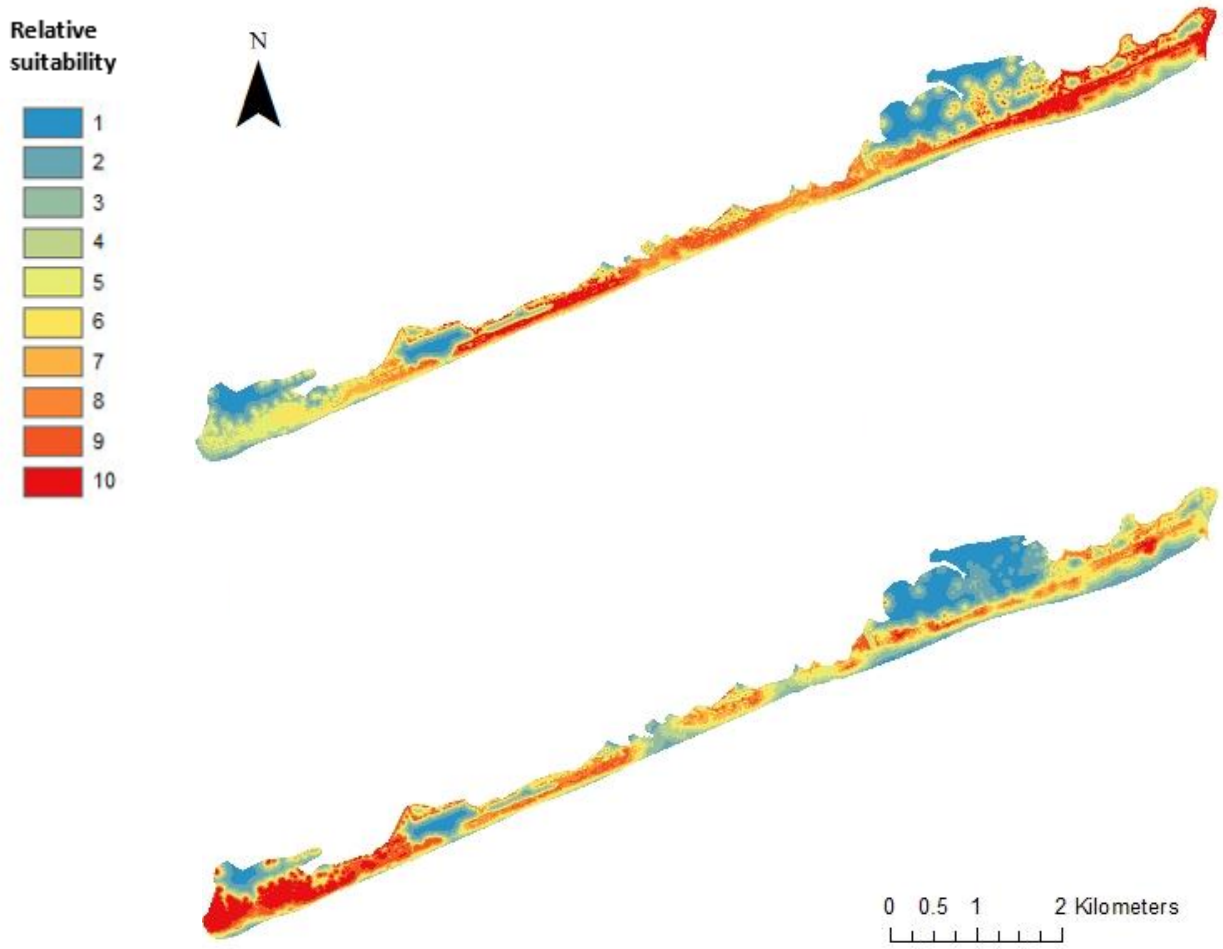


Figure 52. Predicted landscape-scale habitat suitability (10 = highest suitability to 1 = lowest suitability) maps for male (top) and female (bottom) red foxes at Fire Island National Seashore (east of Old Inlet) and Smith Point County Park, using 2017 landcover classification data.

A closer examination of the suitability maps and mean suitability values in USACE restoration areas at Smith Point County Park, calculated by averaging the suitability values for all pixels within each restoration area boundary, shows variable suitability between the sexes and between the 2 areas (Figure 53, Table 57). Within these areas, predicted suitability was generally higher for males and for both sexes in New Made compared to Great Gun. At Great Gun, suitability was highest along the main dune and in areas closest to vegetation. Similarly, suitability at New Made was highest in more vegetated areas.

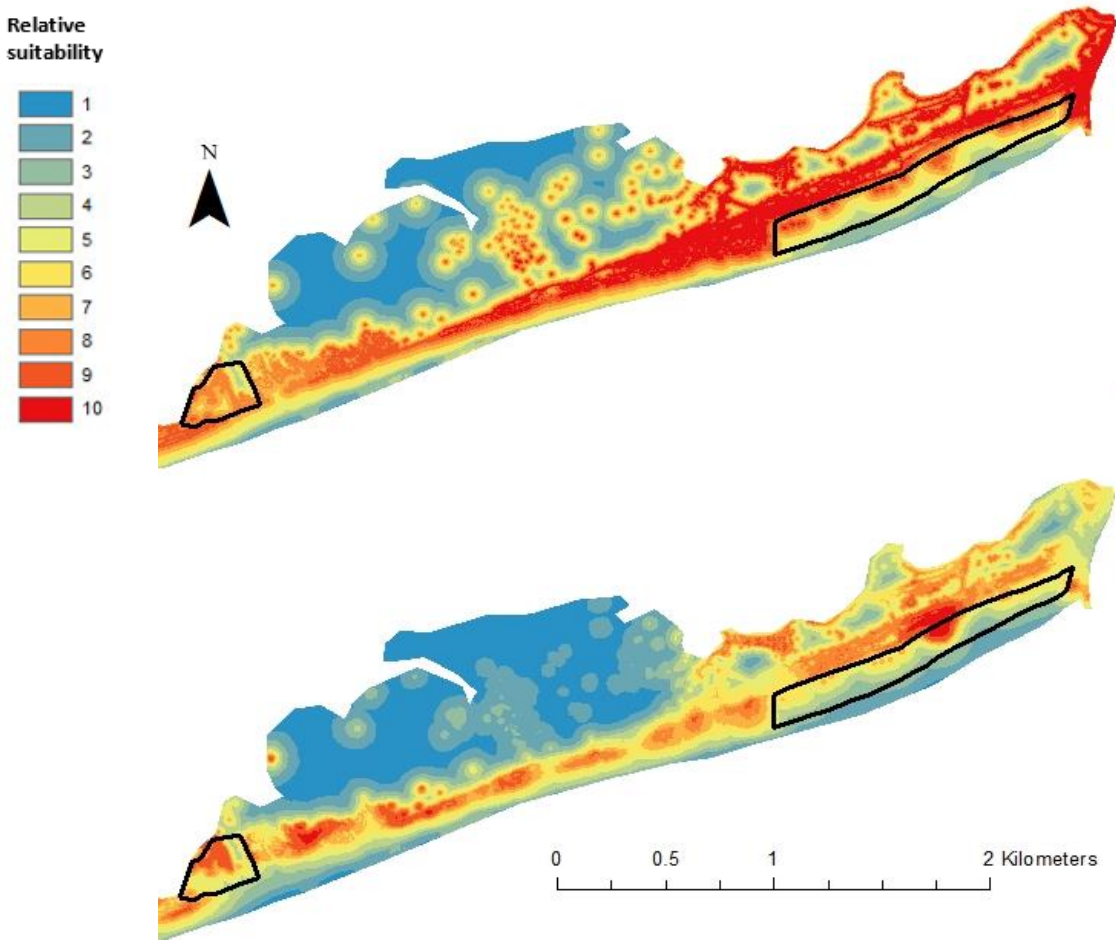


Figure 53. Predicted habitat suitability (10 = highest suitability to 1 = lowest suitability) maps for male (top) and female (bottom) red foxes within the USACE restoration areas (outlined in black) at Smith Point County Park, using 2017 landcover classification data.

Table 57. Mean predicted red fox suitability score (10 = highest suitability to 1 = lowest suitability) for habitat within the USACE restoration areas at Smith Point County Park, calculated by averaging the suitability values for all pixels within each restoration area boundary. The average suitability for each site for both years and sexes is also shown.

Sex	Year	Great Gun	New Made
Male	2016	6.58	6.77
Female	2016	4.96	5.19
Male	2017	6.59	7.13
Female	2017	5.05	6.62
Average Suitability		5.80	6.43

Discussion

To our knowledge, this is the first detailed study of red fox home range, activity patterns, and habitat selection in a barrier island ecosystem, and one of very few studies of fox spatial ecology in a North American coastal system.

Our estimated home range and core use area sizes are relatively small compared to what has been previously reported for red foxes overall, but they are similar to published estimates from non-barrier island coastal areas. Estimates of red fox home range sizes from other studies vary widely, from as few as 10 ha to nearly 5,000 ha (Lucherini and Lovari 1996, MacDonald 1987, Voight 1987). Habitat type, habitat richness, and relative food availability are thought to influence home range size (Lucherini and Lovari 1996, Macdonald et al. 2004). For example, in Baker and Harris' (2004) summary of red fox home ranges reported in 14 studies, estimates ranged from 18 ha in urban England (Baker et al. 1998, Baker et al. 2000) to over 1600 ha in the Canadian tundra (Jones and Theberge 1982). Previous studies in coastal habitats also report varying home range sizes, but within a narrower range. For example, Dekker et al (2001) observed home ranges between <50 and 650 ha in Dutch coastal dune habitat, with roughly half of the observed home ranges falling under 100 ha. Another study in Dutch wet coastal grasslands reported mean home ranges from 253-314 ha, depending on the home range estimation method used (Meisner et al. 2014). In their study of red fox home range and movement in coastal Australia, Meek and Saunders (2000) observed home ranges from 60-270 ha, with an average of 135 ha. Another study in coastal Australia, conducted by Phillips and Catling (1991), reported larger home ranges, between 120 and 520 ha, possibly due to differences in estimation methods. In general, the results from these studies suggest that home range sizes in coastal habitats appear to fall on the lower end of reported home range sizes for red fox. This also appears to be the case in coastal habitats on islands. For example, Silva et al. (2009)

reported red fox home ranges between 77 and 168 ha, depending on estimation method used, on Prince Edward Island.

Our findings that season and sex influenced home range size are consistent with the scientific literature on red fox natural history. Our fall/winter monitoring period lined up with red fox dispersal and breeding season in our study area, when we would expect foxes to be using larger areas in search of new territories and mates. Our spring/summer monitoring periods lined up with denning and kit-rearing season in our study area, when we would expect breeding fox space use to be centered around den locations. This seasonal difference in home range size is also likely connected to seasonal differences in the proportion of resident vs. transient animals. We suspect that the influence of sex on home range size in our study also is related to the disproportionate number of male vs. female transients (11 males vs. 1 female). Male foxes generally move further than female foxes (Lloyd 1980), and may be more likely to exhibit transient behaviors, as was observed in our study.

The observed transient behavior of several individuals suggests a high capacity for dispersal and recolonization should an occupied territory become available. These transient animals regularly moved between management areas, with most transient animals traveling the entire length of the island available to them, without crossing Old Inlet or the 2 motor vehicle bridges connecting Fire Island to mainland Long Island.
In our analysis of activity patterns, we observed distinct differences in movement rates during different times of day, in line with previous studies of red fox activity patterns. Our results suggest that foxes are relatively inactive during most daylight hours, with activity picking up ~10 hours after sunrise and remaining high overnight. While red foxes are generally nocturnal-crepuscular, they may strategically shift activity patterns in response to factors including prey activity, human disturbance, and seasons (Doncaster and MacDonald 1997, Díaz-Ruiz et al. 2016). For example, Díaz-Ruiz et al. (2016) found that red fox activity patterns in areas of low human disturbance in central Spain were correlated with prey activity patterns, while red foxes in more urban areas were less active during the day despite their prey being active at those times. The observed nocturnal-crepuscular activity patterns in our study area could be related to the high amount of human disturbance, but could be explained by other factors as well. We did not examine prey activity patterns in our study or compare movement rates in low human use vs. high human use areas. Thus, while we can describe red fox activity patterns on the island, we have limited ability to infer what may be driving these patterns without further investigation.

Foxes generally selected for areas closer to dry sand, wet sand, and evergreen vegetation and further from ORV roads and water, with proximity to dry sand and avoidance of ORV roads having the strongest effect on probability of habitat use. These results suggest that foxes on the island are likely selecting for dune and interdune areas of dry sand intermixed with vegetation. They also appear to be selecting for wet sand edges,

such as intertidal areas, ephemeral pools and the zone where bayside marsh meets dry land, while avoiding the water itself.

Previous studies suggest that red fox habitat preferences vary widely with the available habitat and geographic location (*see* Ables 1983). In general, studies suggest that red foxes tend to use locations with a variety of vegetation types, but avoid large continuous areas of only one habitat type. These patchwork areas likely offer an ideal combination of food and cover (Ables 1983). We did not find any previous studies examining either second or third order habitat selection in a barrier island ecosystem, although there is previous work conducted on one barrier island looking at other aspects of habitat selection. Krim et al. (1990) examined red fox den site selection on Assateague Island, MD, reporting that dens were most frequently located in shrub/early successional and dune habitats (Krim et al. 1990). Gieder (2015) examined red fox occupancy patterns on Assateague Island, but did not find any significant habitat relationships. Previous research in non-barrier island coastal habitats suggests that red foxes may select for roads and human developed areas and dry land habitat, and against forest and wetlands (Silva et al. 2009, Meisner et al. 2014).

In addition to the general within-home range habitat selection patterns described above, we found differing selection for vegetation based on the time of day, with daytime and transition time locations closer to non-evergreen vegetation than expected, while night-time locations were further from non-evergreen vegetation than expected. Considered alongside our activity pattern results, this difference in selection for or against vegetation based on time of day suggests differential habitat use for different activities. Foxes on the island appear to be selecting areas closer to vegetation and wet sand edges, potentially including bayside marsh habitats, for daytime resting periods, while selecting more open habitats for nighttime periods of higher activity and movement. The fact that residents used areas closer to vegetation, on average, while transients used areas further from vegetation, supports this idea of differential habitat use for movement. Hinton et al. (2015) found similar differences in selection for vs. against roads in transient coyotes compared to residents, and hypothesized that either more efficient movement paths or increased ability to detect residential animals and their territory boundaries could explain this difference in selection.

We also found slight differences in selection for or against development by time of day, with low levels of selection for developed areas at night but neither selection for nor against during the day. This could suggest that foxes are waiting to exploit these potentially high-resource environments until times of lower human activity. However, we did not see any significant difference in avoidance of ORV roads based on time of day. This is likely because, unlike most of the buildings and parking lots in developed parts of our study area, which close overnight, ORV roads are frequently open overnight to allow access to fishing and recreation areas. Seasonal ORV road closures to protect nesting shorebirds offer a potential avenue for exploring the factors

driving fox avoidance of ORV roads in more detail, by comparing habitat selection when ORV roads are closed vs. open.

Mange influenced almost all aspects of red fox spatial ecology on the island, although other factors appeared to influence spatial ecology more strongly. Foxes affected by mange during the monitoring period had larger core use area sizes, higher movement rates, and showed much less avoidance of ORV roads within their home range compared to unaffected foxes. We hypothesize that this is related to decreased hunting efficiency with declining health, requiring more time spent foraging to meet metabolic requirements. Carricondo-Sanchez et al. (2017) found that foxes suffering from advanced stages of mange in southeastern Norway used areas closer to developed areas compared to unaffected foxes, potentially due to increased reliance on anthropogenic food subsidies as their ability to hunt decreases with declining health. Preliminary analyses of mange detections during annual remote camera grids in our study area show similar patterns, with mange detections clustered near parking lots and restroom facilities for visitors (*C. Helmke, unpublished data*). Unaffected animals were found slightly closer to developed areas, while affected animals did not show any clear selection for or against. This difference in findings could be due to the different methods used (remote cameras vs. GPS location data), or differences in sample size between affected and unaffected animals in our study. Interestingly, we did not find evidence that mange influences landscape-level habitat selection patterns. This suggests that, while mange may influence the spatial ecology at the individual level, these changes do not translate to changes in relative predation risk across the landscape for piping plovers.

At the landscape scale, red foxes showed similar patterns of habitat selection for some, but not all, habitat types to the observed within-home range selection patterns. At both scales, foxes selected for areas closer to dry sand and evergreen vegetation and showed little or no selection for or against development. However, the strong avoidance of ORV roads observed within home ranges was not consistent at the landscape scale, with males selecting for areas closer to ORV roads while females avoided those areas. Vegetation had relatively little influence on selection within home ranges, with directional trends varying based on activity periods and residency. At the landscape scale, however, foxes consistently selected areas closer to vegetation. Much of the literature on red fox habitat selection is focused on within-home range habitat selection; we did not find any previous studies of landscape-scale habitat selection in a comparable environment.

Sex was identified as a key factor influencing habitat selection at the landscape scale, but it was not explicitly examined in our analyses within home ranges, which compared habitat selection among activity periods, between residents and transients, and by mange status. We did not compare habitat selection between sexes in those analyses because we specifically wanted to test whether residency, which was strongly correlated with both sex and season but better explained variation in home range size and movement rates, also influenced habitat selection within home ranges. In these landscape scale analyses, selection between the sexes differed

most strongly surrounding ORV roads, with females selecting for areas farther from ORV roads, while males selected for areas closer to ORV roads. This difference may reflect differing tolerance for the risks associated with ORV roads, or could be a result of differing factors driving home range selection. For example, females may select for areas of optimal habitat quality for denning and kit-rearing, while males may select for areas that maximize available food resources or travel efficiency for maintaining territorial markings.

In addition to providing insight into the factors influencing home range selection, these results also yielded relative suitability maps that can be used to predict relative predation risk and identify areas where predator management efforts should be focused. In the future, these maps could also be compared to similar habitat suitability maps for piping plovers to identify areas of high predation risk, where fox and plover suitability overlap.

Visual examination of these maps highlights the influence of ORV roads on the resulting suitability predictions for each sex. Because ORV roads spanned much of the island, areas of high suitability for females were limited. Across the sexes, these maps also highlight the importance of vegetation management, particularly in areas known to be used by plovers. Inspection of the suitability maps shows that areas of higher relative suitability for foxes align closely with the distribution of vegetation and evergreen habitat on the island. While ORV roads are unlikely to be closed or relocated, vegetation can be managed to decrease relative suitability for foxes while improving relative suitability for piping plover nesting. For example, vegetation removal at New Made restoration area would likely decrease suitability for foxes while increasing potential piping plover nesting habitat.

Section 3: Red fox response to piping plover nest exclosures and hatching

This section describes data collection, analytical methods, and results related to the following objective for red fox monitoring associated with the Fire Island Inlet to Moriches Inlet Fire Island Stabilization Project:

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

Based on the 2 very different types of data and analyses used to address Objective 5, we have split our summary of monitoring activities and results for this objective into 2 sections. This section describes the use of spatial data to examine spatial interactions between red foxes and piping plover nests. The next section, Part 3, Section 4, describes the use of fecal dietary analyses and prey items surveys at dens to examine direct predatory interactions between red foxes and piping plover chicks and adults.

Introduction

Red foxes are a predator of piping plovers on Fire Island, NY and in other barrier island ecosystems (Rimmer and Deblinger 1990, Patterson et al. 1991, Cohen et al. 2009, U.S. Fish and Wildlife Service 2014, Gieder 2015). While red foxes may prey on nests, chicks, and adults, information on the effects of red fox predation on chicks and adults is difficult to obtain reliably. Direct observations of adult and chick predation by red foxes or any other predators are rare, partially due to the high mobility and cryptic nature of plovers and many of their predators.

In contrast, nest predation can be studied and quantified more reliably via regular observation and remote camera monitoring (Part 2, Section 9). Information from other plover breeding areas along the Atlantic coast suggests that nest predation by foxes can be a significant concern. Patterson et al. (1991) identified red foxes as the most common cause of piping plover nest predation, accounting for 47.6% of predated nests on Assateague Island. Similarly, red foxes were the most common predator of unexclosed piping plover nests in studies conducted at Crane Beach, MA (Rimmer and Deblinger 1990) and Westhampton Dunes, NY (Cohen et al. 2009), comprising 44.4% and 50%, respectively, of predated nests for which the predator species could be identified.

While nest exclosures may decrease nest predation risk, some predators, particularly foxes, may learn how to penetrate these barriers, rendering them ineffective. Cohen et al. (2009) observed red fox predation at 22.9% of exclosed nests during their study on Westhampton Island. On Fire Island, it appeared as though foxes climbed into and depredated multiple exclosed nests during the 2015 plover breeding season (*A. Derose-Wilson, unpublished data*). In addition to concerns related to penetration of exclosures, there also are concerns over whether they might attract predators to nest sites, leading to increased adult predation or predation of chicks immediately after hatching (Cohen et al. 2016).

These studies suggest that further information on red fox responses to nest exclosures and hatching is needed to guide management activities and help inform predator management. Our objectives were to use fine-scale GPS data to examine spatial response of red foxes to nest exclosures and hatching.

Methods

Fox trapping, GPS collaring, and tracking

We used Victor 1 ½ Soft Catch and Bridger #2 padded foothold traps to capture foxes in the study area during April–May 2016 and February–April 2017. We focused our trapping efforts in and around piping plover nesting and chick-rearing areas from previous years. We also trapped and monitored foxes in 2015 and 2018, but the resulting fox location data did not overlap temporally with any piping plover nests or broods, and was thus excluded from this section's analyses. These trapping efforts were conducted from late August – early December in 2015 and late February–late April in 2018 (Part 3, Section 2).

We outfitted up to 6 foxes per year in each management unit with GPS collars (Quantum 4000E, medium size; Telemetry Solutions, Concord, CA) programmed collars to record locations once every 1-2 hours, every day from April 20th through the end of May, then once every 2 hours, on every other day until programmed collar drop-off in early August. Every 10–14 days after deployment, we located collared foxes via radio telemetry tracking of the collar's VHF signal and approached within <200m to connect wirelessly to the collar and downloaded data using a UHF antenna, Quantum Base Station, and Collar SW software (Version 2.09, Telemetry Solutions).

As soon as possible after the discovery of piping plover nests in areas being used by collared foxes, we reprogrammed collars to temporarily increase the frequency of location recording to collect fine scale data on fox spatial response to plover activities. Due to battery life limitations, we only could sustain this high frequency scheduling for 8 days per collar. We chose to focus these days around predator exclosure setup and hatching. Prior to hatching, chicks use an egg tooth to begin fracturing the egg shell. This process may take several days. We hypothesized that this pipping and/or emersion of chicks from the egg during hatch could draw the attention of predators to nests, and therefore looked at red fox spatial responses to both of these components of the hatching process in our analyses. With piping plover field crews and park managers, we identified expected exclosure setup dates and expected hatch dates, and we 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.

Spatial response of red foxes to piping plover nest exclosures, pipping, and hatching

We used red fox location data collected during the 48-hour periods before and after nest exclosure set-up, first sign of pipping, and hatching to examine the spatial response of foxes to these events. We separated out the location data from each 48-hour time period of interest and calculated the mean latitude and longitude from these sets of coordinates to approximate short-term home range centroids for each period. We used the "spDist" function in R package 'sp' (Version 1.3-1, cran.r-project.org/web/packages/sp, accessed 30 November 2019) to calculate the distance from each of these centroids to the nest of interest and compared the before vs. after distances for each type of event using Wilcoxon's signed rank test, a non-parametric equivalent to the t-test for paired samples (Wilcoxon 1945). If foxes keyed in on exclosure setup, pipping, or hatching, we would expect to see a decrease in the mean distance from centroids to nests after these events. Conversely, if foxes did not key in on exclosure setup, pipping, or hatching, we would expect to see no significant change in the mean distance from centroids to nests after these events.

Results

Fox trapping, GPS collaring, and tracking

We captured 26 unique foxes over 29 capture events in 2016 and 2017. Twenty-three of these foxes were outfitted with GPS collars and monitored until programmed collar drop off or until their deaths prior to collar drop off (Table 58). One fox was captured, collared, and monitored during both the 2016 and 2017 monitoring period, resulting in 2 location datasets from that fox. Malfunctioning collars resulted in no usable data from 1 fox and insufficient data for analyses from another fox. A third fox died after trapping but prior to the beginning of the planned monitoring period, resulting in insufficient data for these analyses. We have not yet been able to determine the cause of death for this fox, but we do not believe that it was trapping related, based on the time between trapping and estimated death (approx. 2 months), long distance between the capture site and where we recovered the remains (>12 km), and lack of signs of lower leg injuries during inspection of the remains.

Table 58. Year, site, residency, denning status, mange infection status, and fate of GPS collared foxes (n=24) on Fire Island, NY in 2016-2017. F denotes females, M. denotes males. Site abbreviations: RMFIL=Robert Moses State Park/Fire Island Lighthouse Tract, FINS=Fire Island National Seashore Wilderness Area (west of Old Inlet), SPOIE=Smith Point County Park/Fire Island National Seashore Wilderness Area (east of Old Inlet). Denning and mange abbreviations: Y=Yes, N=No.

Fox	Year	Site	Residency	Denning	Mange	Fate
M-08	2016	RMFIL	Resident	Y	N	Survived to end of monitoring period
F-05	2016	RMFIL	Resident	Y	N	Survived to end of monitoring period
M-09	2016	RMFIL	Resident	Y	N	Collar malfunction, insufficient data for analyses
M-10	2016	RMFIL	Transient	N	N	Survived to end of monitoring period
F-07	2016	RMFIL	Resident	Y	N	Survived to end of monitoring period
M-11	2016	RMFIL	Transient	N	N	Survived to end of monitoring period
M-12	2016	FINS	-	-	-	Collar malfunction, no usable data
M-13	2016	FINS	Resident	N	N	Survived to end of monitoring period
M-14	2016	FINS	Resident	N	N	Survived to end of monitoring period
F-08	2016	FINS	Resident	N	N	Survived to end of monitoring period
F-09	2016	FINS	Resident	N	N	Survived to end of monitoring period
F-10	2016	FINS	Resident	N	N	Survived to end of monitoring period
M-15	2017	RMFIL	Transient	N	N	Died August 2017, likely vehicle collision
F-11	2017	RMFIL	Resident	N	N	Survived to end of monitoring period
F-03	2017	RMFIL	Resident	N	N	Survived to end of monitoring period
F-12	2017	RMFIL	Resident	N	N	Survived to end of monitoring period
M-08	2017	RMFIL	Resident	Y	N	Died August 2017, likely drowned
M-16	2017	RMFIL	Transient	N	Y	Died June 2017, unclear whether from mange or drowning
M-17	2017	RMFIL	Resident	Y	Y	Died May 2017, mange
F-13	2017	RMFIL	Resident	Y	Y	Died August 2017, mange
F-14	2017	FINS	Resident	N	N	Survived to end of monitoring period
M-18	2017	FINS	Transient	N	N	Survived to end of monitoring period
F-15	2017	FINS	Resident	Y	N	Survived to end of monitoring period
F-16	2017	FINS	-	-	-	Died April 2017 of unknown causes, insufficient data for analyses

A sarcoptic mange outbreak began east of Old Inlet during late summer/early fall 2015, leading to a substantial die-off and a complete lack of fox sign from April 2016–March 2017. From March 2017 until the end of the study, we observed only sporadic fox sign east of Old Inlet and were unsuccessful in capturing any foxes in this section of the island. A second sarcoptic mange outbreak began west of Old Inlet during the spring/summer 2017 monitoring period, leading to the death of 2 GPS collared foxes prior to the end of the planned data collection. In addition, one fox died of a suspected vehicle collision, one fox died of apparent drowning, and a third fox died of either mange or drowning during the final weeks of data collection.

Spatial response of red foxes to piping plover nest exclosures, pipping, and hatching

We successfully reprogrammed GPS collars to the high frequency sampling schedule for 5 foxes in 2016 and 5 foxes in 2017 whose home ranges and core use areas overlapped with plover nesting areas. For various logistical and biological reasons, the estimated dates for nest exclosure setup, pre-hatch pipping and hatch did not always line up with the actual dates. Therefore, our high frequency data collection at some nests did not line up as planned with the events that we had scheduled them around. For our analyses, we used the highest frequency location data available for the 48-hour periods before and after the target event, meaning that analyses were based on a mix of high frequency and regular frequency location data. Exclosure setup dates were recorded, but not times, so we used 1200 (noon) on the recorded setup date as our cutoff for distinguishing between pre- and post-setup locations. Similarly, the exact pipping and hatch times usually were unknown. We used 0600 as the cutoff for pre- vs. post-pipping and 1200 as the cutoff for pre- vs. post- hatching, as piping plover monitoring crews reported that nests typically hatched between 0600 and 1200. These restrictions resulted in considerable variation in the number of locations used to estimate short term home range centroids (Table 59).

Distances from centroids to nests varied among foxes and periods, as did the observed shifts in centroids before vs. after exclosure setup, pipping, and hatching (Table 60, Figure 54). On average, centroids were 43 m farther from nests after exclosure setup, 21m closer to nests after pipping, and 5m closer to nests after hatching. However, these home range shift differences were not statistically significant, with single-tailed *p*-values of 0.26 for exclosure setup, 0.77 for pipping, and 0.59 for hatching.

Table 59. Number of locations used to calculate short-term home range centroids for each red fox and period of interest on Fire Island, NY in 2016-2017.

			Exclosure		Pipping		Hatching	
Year	Fox	Nest Number	Pre	Post	Pre	Post	Pre	Post
2016	F-05	211	85	126	165	8	10	13
2016	F-07	204	54	140	107	155	185	125
2016	F-09	32	93	183	12	10	11	12
2016	F-10	32	105	184	12	9	12	12
2016	M-08	211	75	118	10	10	10	17
2017	F-11	203	149	24	184	155	8	12
2017	F-12	205	175	64	158	166	9	10
2017	F-13	211[a]	190	54	-	-	-	-
2017	M-08	214	6	128	20	0[b]	0[b]	0[b]
2017	M-17	211[a]	183	52	-	-	-	-

[a]Nest failed in between exclosure setup and estimated hatch

[b]Collar malfunction resulted in no usable data during these periods

Table 60. Distances from short-term fox home range centroids to plover nests for each period of interest, and shifts in home range centroid (Δ) following events, in meters. Percent change is also shown, and represents (Post-Pre)/Post.

			Exclosure				Pipping				Hatching			
Year	Fox	Nest	Pre	Post	Δ	% Δ	Pre	Post	Δ	% Δ	Pre	Post	Δ	% Δ
2016	F-05	211	440	615	175	0.40	459	449	-10	-0.02	332	407	75	0.23
2016	F-07	204	351	322	-29	-0.08	220	153	-67	-0.30	161	173	12	0.07
2016	F-09	32	1116	1157	40	0.04	1146	1206	60	0.05	1202	1191	-11	-0.01
2016	F-10	32	1349	1366	17	0.01	1849	1814	-35	-0.02	1824	1800	-24	-0.01
2016	M-08	211	439	455	16	0.04	443	295	-148	-0.33	241	310	68	0.28
2017	F-11	203	127	135	9	0.07	135	220	85	0.63	331	199	-131	-0.40
2017	F-12	205	563	500	-63	-0.11	496	466	-30	-0.06	445	418	-27	-0.06
2017	F-13	211[a]	491	545	53	0.11	-	-	-	-	-	-	-	-
2017	M-08	214	253	175	-78	-0.31	246	-[b]	-[b]	-[b]	-[b]	-[b]	-[b]	-[b]
2017	M-17	211[a]	357	643	286	0.80	-	-	-	-	-	-	-	-

[a]Nest failed in between exclosure setup and estimated hatch [b]Collar malfunction resulted in no usable data during these periods

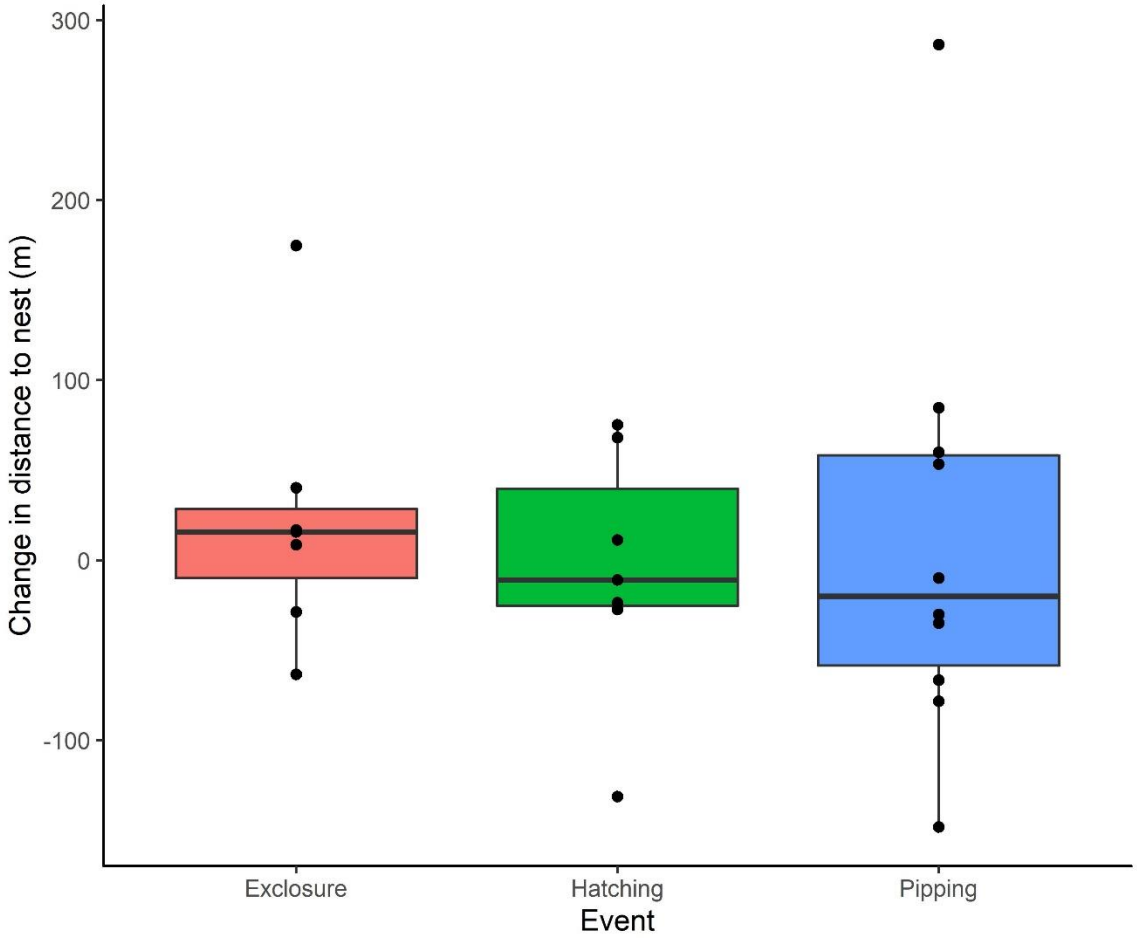


Figure 54. Changes in distances from short-term fox home range centroids to plover nests following exclosure setup, pipping, and hatching on Fire Island, NY in 2016-2017. Positive values indicate shifts away from the nest, and negative values indicate shifts towards the nest.

Discussion

We did not find clear evidence that red foxes are keying in on nests immediately following exclosure setup, pipping, or hatching. These findings differ from previous work that has used remote cameras to examine predator response to nest exclosures. For example, in eastern Canada, Beaulieu et al. (2014) found that foxes and avian predators visited exclosed nests more frequently than unexclosed nests, and that the duration of visits at exclosed nests was significantly longer than at unexclosed nests. Ongoing analyses will explore whether at a longer time scale, red foxes exhibited any attraction behavior to the exclosures.

Concurrent nest monitoring data (Part 2, Section 9) indicates piping plover nests within the monitoring areas in which our red fox location data was collected were visited by red foxes. In 2016, red foxes were the most commonly detected mammalian predator on nest cameras. Relative frequency of detection decreased in 2017, potentially related to the decline in fox abundance following the observed mange outbreaks. The foxes in these nest camera photographs did not appear to have GPS collars, although one had been GPS-collared

previously and was identifiable via ear-tag. Not all nest exclosures were monitored by camera, though, so it is also possible that GPS-collared foxes could have visited an exclosure but were not recorded in remote camera monitoring.

However, this portion of our monitoring and analyses had several limitations that impacted our ability to make inferences on the spatial response of red foxes to pipping and hatching. These limitations include small samples sizes and inconsistent location sampling across events and among foxes. We anticipated that our samples sizes would be limited by the number of GPS collars available, but there were several unanticipated factors that led to additional limitations. From our minimum estimated abundances (Part 3, Section 1), we know that there was likely a significant portion of the fox population each season that was not monitored by GPS collar. Of the foxes that were monitored, not all of them used home ranges that overlapped with piping plover nesting areas. Collar malfunctions and the death of both GPS-collared foxes prior to nest initiation in 2018 further limited our sample size.

Within the data that we used in our comparisons, the temporal resolution and overall amount of locations varied widely, ranging from 6–190 locations at 15-minute to >2-hour intervals (in the event of missed GPS fixes, Table 59). These issues were due in part to temporal mismatch between the estimated and actual event dates as well as collar malfunctions, and could have biased our findings. Our analytical methods may also have limited our ability to detect a potential response. Because we examined home range centroids as opposed to individual locations, any response would have to extend across a period >15 minutes, ideally lasting several hours, to cause a significant shift in home range centroids. Such a response also would have to be consistent across foxes. Given the variation in home range centroid shifts observed (Table 60, Figure 54), our results suggest that responses to exclosure set-up, pipping, and hatching varied among foxes, with some centroids shifting closer while others shifted farther away following a given event.

Despite these limitations, our results represent a key step forward in the application of spatial data to investigate predator response to nest exclosures and hatching. To our knowledge, this is the first study to attempt to use high frequency location data to examine behavioral responses of red foxes or other piping plover predators to exclosures. We are exploring methods to improve these analyses, including comparisons of individual locations and comparisons that exclude locations from daytime periods when foxes are most likely resting. A preliminary comparison of the proportion of locations within a 100-m buffer of nests before vs. after these events yielded similar results, with high individual variation and no statistical significance (*K. Black, unpublished data*).

Section 4: Dietary ecology of red foxes

This section describes data collection, analytical methods, and results related to the following objectives for red fox monitoring associated with the Fire Island Inlet to Moriches Inlet Fire Island Stabilization Project:

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

In Part 3, Section 3 of this report, we also describe the spatial interactions of red fox and piping plovers as part of this objective. We opted to break out the monitoring results on diet into this separate report section given the very different nature of the two types of data related to this monitoring objective.

Introduction

The diets of red foxes have been studied extensively, yielding a wide variety of findings (see Lloyd 1980, Ables 1983, Henry 1986, and Soulsbury et al. 2010 for detailed summaries). In general, red foxes are opportunistic omnivores that readily eat a wide range of items, including insects, fruit and plant matter, small rodents, rabbits and hares, small birds, fish, and other marine organisms when available. They will also scavenge carcasses of prey killed by other species (Soulsbury et al. 2010). Relative availability and ease of attainment both influence the proportion of fox diets comprised of specific items (Ables 1983, Lloyd 1980).

Previous research in coastal areas suggests that waterbirds and small mammals may comprise a significant portion of red fox diets in these systems. For example, Meisner et al. (2014) reported bird remains as the most frequently occurring item in fox scats collected in a wet coastal grassland area in Denmark, occurring in 43% of all scats examined and comprising nearly a third of estimated biomass consumed. Of these bird remains, the vast majority were from order Anseriformes, which includes ducks, geese, and swans. In the only published study of red fox diet on an U.S. Atlantic coast barrier island, Krim et al. (1990) reported mammal remains occurring most frequently (87%) in collected scats on Assateague Island, MD, with evidence of frequent crustacean (64%) and bird (46.3%) consumption as well. In that Assateague Island study, the authors did not report any further taxonomic information on bird remains in analyzed scats, although they mention observing willet remains at a fox den.

Red foxes are known to depredate piping plover nests on Fire Island and nearby Westhampton Island (Part 2, Section 9; Cohen et al. 2009, Monk et al. 2015), but it is unknown whether the depredation occurs by a few or many foxes, or what drives them to consume piping plover nests and chicks. Similarly, while red fox predation on piping plovers affected reproductive output in our monitoring areas (Part 2, Section 6), it is unknown whether plovers and other shorebirds make up a large portion of individual and overall fox diets in the area.

In addition, the significant human presence across much of the island raises the question of whether anthropogenic food sources might be subsidizing local fox populations and influencing how red foxes interact

with, and affect, piping plovers. In developed areas, anthropogenic food sources often represent the most common items in red fox diets (Soulsbury et al. 2010). Access to anthropogenic food sources has been shown to increase the abundance of red foxes and other potential piping plover predators (Shapira et al. 2008, Newsome et al. 2015). Changes in predator abundance due to human food subsidies can lead to increased pressure on prey species and altered behavior related to predator avoidance (Shapira et al. 2008, Newsome et al. 2015). While anthropogenic food may divert consumption away from prey species in the short term, in the long run these subsidies can increase the overall number of predators competing for a finite prey source. This increase in abundance can be particularly problematic when anthropogenic food availability decreases, which can cause predators to switch food sources (Yirga et al. 2012).

The objective of this section was to describe red fox dietary composition on Fire Island, New York. Specifically, we identified major prey species, investigated fox consumption of piping plovers and other shorebirds, and assessed their reliance on anthropogenic food sources.

Methods

Field methods

We collected potential red fox scat samples during regular transect surveys, den monitoring visits, and opportunistically while trapping and collecting other field data during April 2016–February 2019 (Part 3, Section 1). All potential red fox scats estimated to be ≤ 1 month old were collected in a paper bag and labeled by estimated freshness and a unique sample number corresponding to the recorded GPS waypoint. We also recorded an estimated age category and preliminary species identification category based on scat morphology and context clues, such as fresh fox tracks in the vicinity or deposition at features frequently scent marked by foxes (Table 61, Table 62). During den monitoring visits, we also 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. We used information on prey items found at dens to determine what types of prey adults were bringing back for kits and see how these items compared to what we observed in our fecal dietary analyses.

Table 61. Scat collection criteria and age category criteria and used during sign collection on Fire Island, NY from 2016–2019. "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	Collect?
Very fresh	Varied	Very moist	None	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

Table 62. Preliminary species identification categories and criteria used to describe collected scat samples on Fire Island, NY from 2016–2019.

Category	Criteria
Fox certain	Observed a fox depositing the sample, collected from a captured red fox during trapping, or collected at an active red fox den during den monitoring.
Fox likely	Sample morphology and deposition patterns typical of red fox scats: Tapered ends, <5/8" wide and 6" long, fully or partially segmented. Deposited above ground (rather than buried), often along trails and/or at noticeable landmarks. May have a musky odor (as opposed to the distinct odor of domestic dog scat).
Unknown	Sample morphology and/or deposition patterns ambiguous, no context clues to aid in species identification.

Genetic verification of source species

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 were sent to the Mammalian Ecology and Conservation Unit (MECU) in the Veterinary Genetics Lab at University of California, Davis for mitochondrial DNA sequencing to verify the source species (Aubry et al. 2009, Miles et al. 2015). Feral cats, domestic dogs, and raccoons are also present on Fire Island, and even experienced field biologists can have difficulty distinguishing among scats from these species. We sought to avoid making incorrect inferences on fox diets by verifying source species identification.

Fecal dietary analyses

Following DNA subsampling, we used the remaining portion of each scat sample for fecal dietary analysis. We freeze-dried samples for ≥48 hours to remove any remaining moisture, then separated out and sorted contents by hand. We identified contents to the lowest taxonomic level possible through comparison to a variety of reference materials (Spiers 1973, Debelica and Thies 2009, Dove and Koch 2010). We used a dissecting microscope to assist with identification of food items as needed, and we used a digital compound light microscope to assist with identification of hair and feathers found in samples. In addition to pre-existing reference materials, we used hairs from museum specimens in the Virginia Tech Department of Fish and Wildlife's Cheatham Hall Natural History Museum to create a hair microscopy guide of 23 species of mammals found in the eastern United States to assist in identifying hairs to genus and species (Black and Taylor 2019). We visually separated hairs and feathers into groups based on morphological similarity and examined 3-6 representative samples from each group. For hairs, we examined the internal medulla and external cuticular scales, selecting both guard hairs and underfur when present (Debelica and Thies 2009). For feathers, we focused our examinations on the barbs and barbules of plumulaceous feathers, as those sections contained the most useful diagnostic characteristics (Dove and Koch 2010).

We then calculated frequency of occurrence, or the proportion of all scats containing at least one item in a given category, for each diet item to assess reliance on different food sources (Larson et al. 2015). While many different indices can be used to describe dietary composition, frequency of occurrence is the most commonly reported index reported for carnivore fecal dietary analyses (Klare et al. 2011) and thus allows our monitoring results to be comparable to many other studies. We first calculated frequency of occurrence at the broad categorical level, then at each taxonomic level from class to species (Table 63). Items that were not identifiable to lower taxonomic levels were included in those analyses but labeled to the lowest identifiable level, i.e., "Unknown Rodentia." Ticks and other arachnids were grouped with "Insects" for broad categorical analyses and separated out for lower taxonomic level analyses. We excluded any red fox hair found in samples from our analyses, as it was likely ingested during grooming activities. We also excluded miscellaneous natural non-food items found in scats, such as rocks and beach debris, that were likely ingested unintentionally while eating other food items or unintentionally collected in sample bags during the scat collection process.

Table 63. Broad categories used to group red fox food items for preliminary analyses, and examples of common items within each category, on Fire Island, NY from 2016–2019.

Category	Examples
Anthropogenic	Human food waste, tin foil, plastic, paper
Avian	Feathers, beaks, avian bones
Fish	Fish scales, bones
Insect	Entire or partial remains of insects, ticks, or spiders
Mammalian	Mammal hair, bones, claws
Miscellaneous[a]	Any non-food item found in scat samples, e.g., rocks
Reptilian	Reptile skin, bones
Vegetation	Beach plum pits, twigs, leaves, grasses
Shellfish	Crustacean or mollusk shells, claws
Unknown	Any item not clearly identifiable as belonging to another category

[a]Miscellaneous non-food items were excluded from our frequency of occurrence calculations.

Results

Sample collection and genetic verification of source species

We collected 302 potential red fox scat samples during transect surveys, den monitoring, and opportunistically from 2016–2019. We collected DNA subsamples from 270 of these scat samples for genetic verification of source species. The remaining 32 samples were not suitable for DNA subsampling, due to age-based or environmentally-based degradation (Table 61, 'very old' categorization). Of the samples sent for genetic analysis, 220 of 270 yielded a source species identification, resulting in a genetic species identification success rate of 81.4% and an overall species identification rate of 72.8% for all samples collected. Of these samples, 97.7% (215 out of 220 samples) were identified as red fox. Of the 5 non-fox samples, 2 were identified as domestic dog, and 3 were identified as domestic cat. We conducted fecal dietary analysis of these non-fox samples, but did not include the results in our frequency of occurrence analyses due to small sample size. Given the high genetic source species verification rate of potential red fox samples examined, we chose to include the 32 samples that were not sent for source species verification and the 50 samples that failed to yield a source species during genetic analyses in our red fox dietary analyses. We refer to these as 'probable fox samples' hereafter, yielding 297 confirmed or probable fox samples.

Fecal dietary analyses

Red foxes on Fire Island, NY in 2016–2019 had a highly varied diet. At the broad categorical level, mammals (75% of samples), vegetation (73%) and insects (55%) were found in the highest proportion of samples examined (Table 64). We identified 27 unique taxonomic orders of prey items, as well as several additional broad taxonomic groups of which unknown vegetation (62% of samples), Rodentia (rodents, 40%), and Coleoptera (beetles, 38%) were the most frequently represented (Table 65). At the lowest identifiable taxonomic level, unknown vegetation (62% of samples), unknown mammals (27%), and unknown Decapoda (25%, an order including crabs and other crustaceans) were found in the highest proportions of samples examined (Appendix H). Unknown vegetation included partially and fully digested remains of various plant species. Unknown mammals consisted primarily of hair and mammal bone fragments that we could not further identify. Unknown Decapoda included numerous crab shell fragments. Of items that could be confidently identified to the species level, white-tailed deer were the most frequently occurring (10.9% of samples), followed by the American dog tick (10.3%), and eastern cottontail rabbit (7.4%; Appendix H). It is unclear whether ticks were consumed intentionally for nutritional value, or unintentionally during grooming activities and/or consumption of host remains.

Avian remains were found in 33% of confirmed and probable red fox scat samples. Roughly 2/3 of these remains were unidentifiable past order Aves (23% of samples). Of those that were further identifiable, Anseriformes (ducks, geese, swans, and similar waterfowl; 6% of samples), Passeriformes (perching birds, including songbirds; 5%), and Charadriiformes (shorebirds, including gulls, terns, plovers, 3%) were the most common orders. We did not find any clearly identifiable piping plover remains in confirmed or probable red fox samples.

Table 64. Frequency of occurrence for each broad category of food item from fecal dietary analysis of confirmed and probable fox scat samples (n=297), ranked from highest to lowest, from Fire Island, NY 2016–2019.

Category	Frequency of Occurrence
Mammal	0.75
Vegetation	0.73
Insect	0.55
Avian	0.33
Shellfish	0.33
Unknown	0.22
Anthropogenic	0.15
Fish	0.12
Reptile	0.01

Table 65. Frequency of occurrence for each order from fecal dietary analysis of confirmed and probable fox scat samples (n=297), ranked from highest to lowest, from Fire Island, NY 2016–2019.

Order	Frequency of Occurrence	Order	Frequency of Occurrence
Unknown vegetation	0.62	Unknown Malacostraca	0.03
Rodentia	0.40	Chiroptera	0.02
Coleoptera	0.38	Rosales	0.02
Decapoda	0.27	Soricomorpha	0.02
Unknown Mammalia	0.27	Didelphimorphia	0.02
Unknown Avian	0.23	Fagales	0.02
Unknown	0.22	Unknown Bivalvia	0.01
Unknown Insecta	0.15	Unknown insect	0.01
Anthropogenic	0.15	Galliformes	0.01
Ixodida	0.12	Amphipoda	0.01
Unknown fish	0.12	Falconiformes	0.01
Carnivora	0.11	Gruiformes	0.01
Artiodactyla	0.11	Hymenoptera	0.01
Poales	0.09	Unknown Squamata	0.01
Lagomorpha	0.07	Araneae	<0.01
Anseriformes	0.06	Columbiformes	<0.01
Passeriformes	0.05	Odonata	<0.01
Charadriiformes	0.03	Orthoptera	<0.01
Unknown shellfish	0.03	Pinales	<0.01
Eulipotyphla	0.03		

Mammals were the most frequently occurring category of item in domestic cat scats (3/3 samples), followed by birds (2/3), insects (2/3), and vegetation (2/3, Table 66). Insects (2/2 samples) and mammals (1/2) were the most frequently identified prey in domestic dog diets (Table 67). The only definitive piping plover remains found in our diet analyses included one set of adult leg bands found in a scat sample collected by piping plover monitoring crew at Smith Point County Park in June 2018. The scat sample containing these bands was one of the 3 samples identified as originating from a domestic cat via genetic analyses and also contained several feathers identified to family Charadriidae, which includes piping plovers as well as other plover species and

lapwings. Additional feathers identified to family Charadriidae were found in 2 other scat samples, but we were unable to identify these feathers to species. One of these samples was identified as originating from a red fox via genetic analyses, while the other failed to yield a genetic source species identification.

Table 66. Number of scats in which each broad category of food item was found during fecal dietary analysis of confirmed domestic cat scat samples (n=3), ranked from highest to lowest.

Category	Number of Scats
Mammal	3
Avian	2
Insect	2
Vegetation	2
Anthropogenic	0
Fish	0
Reptile	0
Shellfish	0
Unknown	0

Table 67. Number of scats in which each broad category of food item was found during fecal dietary analysis of confirmed domestic dog scat samples (n=2), ranked from highest to lowest.

Category	Frequency of Occurrence
Insect	2
Mammal	2
Avian	1
Shellfish	1
Vegetation	1
Anthropogenic	0
Fish	0
Reptile	0
Unknown	0

Anthropogenic food item remains composed 15% of confirmed and probable fox scat samples. Within this category, we found a wide variety of items including tin foil, paper, Styrofoam, stickers, and semi-digested

gum. We also found a variety of plastic food wrappers, part of a plastic bottle lid, and part of a plastic grocery bag in scat samples. Several samples contained small pieces of glass, and one sample contained a large amount of what appeared to be scented candle wax.

We recorded a wide variety of prey remains outside of 24 den entrances and in the immediate vicinity during our monitoring activities from April–August 2016 and 2017. No breeding dens were found in 2018, when only one fox was believed to be present sporadically East of Old Inlet and the mange outbreak had spread through all areas West of Old Inlet, leaving very few foxes and no evidence of successful reproduction (Part 3-Section 1). The remaining dens did not have any prey items in the vicinity, and were generally those used later in the season. The most commonly found natural food items included remains from skates (*Raja spp*., present at 19/24 or 79% of dens with prey items), clams (*Mya arenaria*, 17/24 or 71%), assorted fish species (12/24 or 50%), and white-tailed deer (*Odocoileus virginianus*, 11/24 or 46%). We observed a variety of anthropogenic food items (e.g., fast food bags and wrappers, hamburgers buns, cookie wrappers, chip bags, cat food cans) at 15 of 24 (63%) of dens with prey/food remains (Figure 55). Bird remains, including gulls (*Larus spp*.) and mallard duck (*Anas platyrhynchos*), were present less frequently (10/24 dens or 38%).



Figure 55. Some anthropogenic food items found at active red fox dens on Fire Island, NY, during den monitoring activities in May–July 2017.

Discussion

Our findings suggest that red fox diets on Fire Island are highly varied, including a wide range of natural and anthropogenic food items. Within this variety, vegetation, mammals, and insects appear to make up an important portion of fox diets on the island and were found in over half of scat samples examined. Birds and shellfish also appear to be regularly consumed and were found in approximately one third of samples. Within these broad categories, rodents made up more than half of mammal prey items, closely followed by unidentifiable mammal remains. Beetles made up nearly 70% of insect remains, and crabs made up over 80% of shellfish remains. Only about 1/3 of avian remains were identifiable beyond class Aves. Of the avian remains identifiable to order or lower, Charadriiformes represented a low proportion, accounting for <10% of all avian remains found in scats.

Previous examination of red fox diets in a barrier island ecosystem also identified small mammals as an important food source but differed in the observed proportions and species composition. Specifically, Krim et al. (1990) found mammal remains in 87% of scat samples examined, compared to our observation of mammal remains in 75% of samples. In their results, rabbit, meadow vole, and white-footed mouse were the most frequent mammal species represented, occurring in 44.4%, 31.5%, and 24.1% of scats, respectively, and they did not identify any other mammalian remains to species. We observed a wider variety of mammal species and much lower occurrence of rabbits (7% of samples). Interestingly, we found white-tailed deer remains in 11% of samples. Red fox predation on fawns has been known to occur but is generally thought to be infrequent and limited to times of year when fawns are less mobile (Epstein et al. 1983). We found deer remains in fox samples collected in almost all months of the year when we conducted transect surveys, which would suggest that most of these occurrences were from scavenged carcasses rather than depredated fawns.

While our analyses for non-fox samples were limited in scope, the discovery of adult piping plover leg bands in a domestic cat scat sample highlights the potential impacts of free-ranging cats in shorebird nesting areas and the potential need for including feral cats in predator management considerations. Both nest abandonment and loss of chicks have been attributed to feral cats previously on Fire Island (Winter and Wallace 2006). Moreover, on adjacent Westhampton Island, feline depredation of piping plovers and nests, as well as nest abandonment, has been reported (Winter and Wallace 2006, Cohen et al. 2009).

The presence of these leg bands on nearly all piping plover adults and rapid banding of chicks after hatching suggests a high probability of detection during fecal dietary analyses if consumed. Despite the likely red fox predation of exclosed and unexclosed nests at Smith Point County Park in 2015 and observed red fox disturbances at exclosures in 2018, we did not find any clearly identifiable piping plover remains in our examination of confirmed and probable fox scat samples collected in 2016–2019. This lack of remains in scat suggests that red fox predation of piping plover adults and chicks may be opportunistic and variable across

years, as opposed to habitual, targeted hunting. However, we recognize that egg fragments from predated nests may not be reliably detected using fecal dietary analysis methods. We recommend the use of nest cameras for more reliable identification of nest depredation events.

While direct predation of foxes on piping plover adults and chicks may not be a frequent occurrence on the island, even low levels of predation can have significant impacts on population viability. In addition, indirect effects of red fox presence and disturbance may have sub-lethal negative effects on piping plovers, warranting further investigation. For example, Doherty and Heath (2011) found the number of fox tracks counted on nearby transects to be the strongest predictor of nest abandonment at several sites across Long Island, NY, including Smith Point County Park.

The occurrence of anthropogenic items in 15% of scats suggests that foxes are regularly exploiting these additional food resources. Anthropogenic food items were more frequently found in samples collected during the spring and summer months, suggesting that availability likely varies with seasonal variations in human presence. Because many anthropogenic food items are fully digested, we believe that our results are likely an underestimate of actual consumption of these resources. For comparison, we found human food remains at 63% of red fox dens with prey remains in 2016 and 2017. We believe that cartilaginous fish, particularly skates (*Raja spp*.), may be similarly underrepresented in our analyses, as they were found even more frequently at dens (79% of dens with prey remains) but not found during our fecal dietary analyses.

Section 5: Red Fox Summary of Findings and Management Recommendations

This section describes the following objective for red fox monitoring associated with the Fire Island Inlet to Moriches Inlet Fire Island Stabilization Project:

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.

Red foxes (*Vulpes vulpes*) were important predators of piping plovers on Fire Island during our monitoring activities from 2015–2019. Red fox population dynamics, spatial ecology, and impacts on piping plovers varied dramatically across the monitoring period and were driven by high mortality and partial extirpation caused by sarcoptic mange infections. At the outset of this monitoring plan, we had expected to monitor the effects of predation management, as implemented by land managers, on red fox populations and piping plovers. Only one red fox was lethally removed during the monitoring period by USDA Wildlife Services at Smith Point County Park in the Winter of 2015-2016. This removal was in response to high plover nest mortality caused by likely one or a few foxes in 2015. A single red fox was removed near the end of the die-off of foxes at that site due to the mange outbreak. Non-lethal predator exclosures around piping plover nests were widely deployed and found to be the most important predictor of piping plover nest survival. Thus, even though sarcoptic mange did drive the population dynamics of red fox, the findings here are still highly informative to ongoing decisions about both non-lethal and lethal predation management on both Fire Island and at other sites along the U.S. Atlantic Coast. In the sections that follow, we summarize our key findings from our monitoring activities and then apply those findings to help guide ongoing predation management options.

Summary of Key Findings

Direct impacts of red foxes and other predators on piping plovers

Red fox predation was a strong driver of piping plover reproductive output on Fire Island during portions of our monitoring period. In 2015, prior to the observed mange outbreak, 18 plover nests were believed to have been depredated by red fox at Smith Point County Park based on fox sign found at the nest site. At least one fox learned to enter exclosures at Smith Point by tearing the netting at the top to get to the eggs inside, leading to the removal of exclosures from nests, all of which all were subsequently depredated.

Two outbreaks of sarcoptic mange (caused by the *Sarcoptes scabiei* mite) then occurred over the remainder of the monitoring period, allowing us to better isolate the impacts of red fox on piping plovers in these sites. In some ways, the natural mange-outbreak acted as a pseudo-experiment on the effects of complete

removal of foxes from a piping plover breeding area. Whether such management-driven extirpation of foxes would be feasible or recommended based on our findings is discussed below.

First, mange was detected in red foxes to the East of Old Inlet in late summer 2015; foxes were extirpated by mange and were apparently absent in that area from April 2016 to March 2017. Only sporadic sign from likely one individual was observed from April 2017 to the end of monitoring in 2019. Piping plover chick survival and overall reproductive output were negatively related to red fox trap success (an index of red fox activity based on the frequency of detections), such that plover chick survival and reproductive output were lower before versus after the mange die-offs. Our camera-based monitoring allowed us to simultaneously assess trap success of other known piping plover predators, but no relationship was found between feral cat or raccoon trap success and plover reproductive metrics. However, the bands of an adult piping plover were found in a feral cat scat, identified as such by genetic species-typing, at Smith Point County Park in 2018 after the mange-related die-off of red foxes at that site.

In 2016, prior to the mange-related die-off east of Old Inlet, red foxes were among the top 3 species captured on cameras deployed on piping plover nest exclosures throughout the study area. However, in 2017-2019 and across all years of monitoring combined, humans, gulls, unknown or unidentifiable birds, and white-tailed deer were the most frequently detected species at exclosures. Thus, in no years were red foxes the most frequent visitor to nest exclosures as detected by nest cameras.

Prior to, and during the mange-related die-offs of foxes, we collected scat samples and catalogued prey remains around den sites to quantify red fox diet and assess the role of piping plovers in their diet. Red foxes on Fire Island, NY in 2016-2019 had a highly varied diet. At the broad categorical level, mammals (75%), vegetation (73%) and insects (55%) were found in the highest percentage of samples examined. We identified 27 unique taxonomic orders of prey items. We did identify Charadriiformes remains in 3% of scat samples, but were unable to identify any prey remains down to the level of the piping plover, as the degradation of the feather remains after gut passage prohibited species-level identification. In analyses of 297 confirmed or probable red fox scat samples collected from 2016 –2019, no bands of piping plovers were found. However, piping plover bands were found in 1 of 3 feral cat scat samples.

In 2018, we captured photos of a peregrine falcon on a nest camera less than an hour before the carcass of one of the incubating adults was found inside the same exclosure. In 2019, a peregrine falcon was seen eating a plover chick while on its associated nest's exclosure, and the flagged leg of an adult was found at the base of the same exclosure less than a week later. Given how often we observed peregrine falcons on or near exclosures while surveying, we expected more camera detections than what were observed. Given these observations, plus the frequency of observations of gulls and other avian predators on nest cameras, additional monitoring and management of avian predators may be warranted.

Indirect impacts via habitat use, spatial overlap and behavior of red foxes and piping plovers

In addition to the quantifying the direct predation impacts of red foxes and other predators on piping plovers, we explored shared habitat use, spatial overlap and behavioral interactions as indirect metrics of predation risk by foxes on piping plovers and to best inform predation management.

Abundance and Density of Red Fox as Indirect Indicators of Risk

As noted above, the mange outbreak was the largest source of mortality for all marked red foxes and drove dramatic population declines and reproductive failure of foxes across our monitoring period. We quantified four lines of evidence documenting the change in red fox populations and the relation to the mange outbreaks. First, red fox camera trap success, which is an index of relative activity, declined dramatically in our remote camera grid surveys following the mange outbreaks in each monitoring area (Smith Point County Park and Old Inlet East: 13.99 Trap Success (TS) pre-mange in 2015-2016 to 5.08 TS in 2018-2019; Robert Moses State Park: 119.95 TS pre-mange in 2016-2017 to 4.99 TS post-mange in 2018-2019; Fire Island National Seashore Watch Hill to Old Inlet 103.36 TS pre-mange in 2016-2017 to 8.30 TS in 2018-2019). Second, red fox density, as quantified from both GPS-collared and ear-marked individuals and counts at den sites, declined dramatically following the mange outbreaks in each monitoring area (Smith Point County Park and Old Inlet East, post-mange data only, 0 foxes/km^2 in 2016, 0.21 foxes/km^2 in both 2017 and 2018; All areas west of Old Inlet, 5.88 foxes/km^2 in 2016, 3.48 foxes/km^2 in 2017 and 0.42 foxes/km^2 in 2018). Third, we saw similar declines following the mange outbreak in the number of unique genotypes detected in scat samples collected during twice regular transect sampling, from 32 unique individual genotypes in 2016 to 14 in 2018–early 2019. Fourth, we saw a decline in reproduction by foxes following mange outbreaks and a complete reproductive failure of red foxes in 2018 (17 dens and 8 family groups monitored in 2016 to 0 in 2018). All of these metrics combined quantify the collapse of red fox populations across the monitoring area and temporary extirpation of red fox in 2016 at Smith Point County Park; thus, red fox predation risk to plovers dramatically decreased as the monitoring progressed. As of the end of monitoring in early February 2019, we suspect only one fox was present to the east of Old Inlet. As of this same time, no red foxes were observed to the west of Old Inlet. We suspect the red fox will recolonize Fire Island, and it will be important to both monitor the recolonization and consider recolonization dynamics in future predation management efforts.

The densities of red foxes observed in our monitoring areas before the mange outbreaks were similar to those found in urban areas where foxes were heavily subsidized by anthropogenic food sources. Similar to urban fox populations, we found heavy use of anthropogenic food sources (15% of scat samples and 63% of den locations had anthropogenic food remains), even in the most remote parts of the Fire Island National Seashore Wilderness Area, near Old Inlet West. It is likely that these human food subsidies contributed to high

population densities of foxes, and active management of these subsidies should be considered in future predation management efforts in order to minimize fox densities and impacts to piping plovers.

Occupancy as an indicator of risk independent of abundance and density

Whereas indices of fox activity, fox abundance and density, and fox reproductive output all declined dramatically pre- to post-mange, red fox occupancy (i.e., probability of presence accounting for detection) remained high except during the window of fox extirpation in areas east of Old Inlet. For example, at Smith Point County Park, red fox occupancy was 0.86 in 2015-2016 before mange, 0 in 2016-2017, and 0.84 and 0.80 in 2017-2018 and 2018-2019. In the latter two years, trapping efforts and examination of photos led us to conclude that there was likely only 1 fox east of Old Inlet, yet that fox was present at 84% and 80% of camera stations over the monitoring area. Similarly, red fox occupancy was 1 at both Robert Moses State Park and Fire Island National Seashore, west of Old Inlet, in 2015-2017. Thus, red foxes were present at every camera station in those monitoring areas. Even after the mange-related declines in abundance, density and reproduction in these areas west of Old Inlet in 2017-2019, occupancy remained high (0.82 at Robert Moses State Park, 0.86 Fire Island National Seashore). Thus, despite dramatic population declines caused by mange, and reproductive failure in 2018-2019, we found red foxes were present across the monitoring areas at rates similar to before the mange impacts. These results, along with the habitat selection and home range behaviors of foxes described below, suggest that predation risk for piping plovers persists even at very low red fox population sizes.

Habitat and spatial overlap of red foxes and plovers

Our ability to track individually-marked and GPS collared red foxes was unique, and has never been conducted in a coastal or barrier island system. Our findings of high population density, as described above, were related to relatively small home ranges and core use areas for resident, breeding foxes. The home range sizes of resident, breeding foxes in our monitoring area were similar to other urban fox populations with high food subsidies. However, we also found that season and sex were strong drivers of home range sizes and movement patterns in red fox.

Males dispersed in the fall and transient, non-breeding and non-territorial males were present across the years and monitoring areas. Both dispersing and non-breeding males had very large home ranges, frequently moving between Old Inlet and both Moriches and Fire Island Inlets. However, no marked foxes were observed to cross Old Inlet. The observed dispersal and transient behavior of male red foxes suggests a high capacity for dispersal and recolonization should an occupied territory become available. These transient animals regularly moved between management areas, with most transient animals traveling the entire length of the island available to them. Thus, any predation management, lethal or non-lethal, must consider this recolonization potential and be sustained in its deployment.

Foxes on the island appear to be selecting areas closer to vegetation and wet sand edges, potentially including bayside marsh habitats, for daytime resting periods, while selecting more open habitats for nighttime periods of higher activity and movement. The fact that resident foxes used areas closer to vegetation, on average, while transients used areas further from vegetation, supports this idea of differential habitat use for movement. Hinton et al. (2015) found similar differences in selection for vs. against roads in transient coyotes compared to residents, and hypothesized that either more efficient movement paths or increased ability to detect residential animals and their territory boundaries could explain this difference in selection.

Within restoration areas, predicted relative suitability for inclusion in home ranges was generally higher for males, and was higher for both sexes in New Made (6.43 out of 10, 10 being highest suitability) compared to Great Gun (5.80 out of 10) due to differences in vegetation and open sand between the two. At Great Gun, suitability was highest along the main dune and in areas closest to vegetation. Similarly, suitability at New Made was highest in more vegetated areas. Both of these restoration areas were relatively more suitable than not (i.e., > 5 suitability). Continued vegetation removal and management will decrease suitability of the sites for daytime resting and also minimize the ability of concealment by foxes, and should be considered as part of a non-lethal management toolkit.

Behavioral responses of red fox to piping plovers

Nest exclosures should continue to be used, as they were demonstrated to be the strongest predictor of piping plover nest success. Further, we did not find evidence that GPS-collared red foxes were attracted to the exclosures in the 48 hours after setup, pipping or hatching. However, in 2015, a fox learned how to climb into and depredated nests in exclosures at Smith Point County Park. When managers subsequently removed the remaining exclosures throughout the park, those nests were immediately destroyed.

When foxes learn to enter exclosures, we think that the most effective management measure would be a targeted removal of the offending red fox, which also would protect chicks to some degree. In a high-density red fox population, as was present before the mange outbreak, most resident red foxes maintained a home range with radius of about 1.4 km (in narrower sections of island which averaged 150m ocean to bay width) or 523 m (in wider island sections which averaged 400m ocean to bay width). Therefore, removal of exclosures within that radius (either about 1.4 km in narrow island portions or 500 m in wide sections) might prevent adult plovers from interacting with foxes at exclosures, but likely would result in egg loss. Removal of exclosures from an entire management area may result in large-scale nest failures without reducing adult-fox interactions outside of the home range of the offending fox.

An extensive review of the adaptive process and options of lethal and non-lethal techniques for predation management is provided in Guidance and Best Practices for Coordinated Predation Management to Benefit Temperate Nesting Shorebirds (Hunt et al. 2019*a*). This document also recommends the use of a new

tool, PiperEx, to further guide decisions about when to remove nest exclosures (Cohen et al. 2016). Currently, our data are not incorporated into the structured decision-making process which guides PiperEx, but may be in future iterations and thus we recommend consideration of PiperEx alongside the recommendations summarized in this report. All predation management should be adaptive and collaborative between the land and predator managers as to the suite of options, and when possible, non-lethal techniques pursued before lethal.

We observed increased presence at exclosures and predation by raptors in later years of the monitoring. Further, gulls and other birds were the most frequent organisms, after humans, to be detected on camera traps deployed on nest exclosures. On the Missouri River, predation by great horned owls was density-dependent (Catlin et al. 2011). It is too early yet to ascertain if that is the case on Fire Island, but we do recommend targeted monitoring of avian predators and their effects as piping plover density increases on Fire Island. Hunt et al. (2019) review and provide pros and cons of both lethal and non-lethal management options specific to avian predators.

Management of anthropogenic food sources

We found high occurrences of anthropogenic food remains in both scat samples and at den sites of red foxes. We also frequently observed people feeding red fox and other wildlife throughout the monitoring areas, despite official policies prohibiting such behavior. Human food subsidies are known to increase density of commensal predators such as red fox, and thus the availability of human subsidies should be managed in order to reduce predation risk of red fox on piping plovers. We recommend that land managers on Fire Island consider launching a coordinated campaign to limit feeding of wildlife and to most efficiently manage trash from recreationists. Launching of a novel campaign now during the very low (east of Old Inlet) to potentially temporarily extirpated (west of Old Inlet) populations of red foxes could slow the population's rebound and thus continue to minimize predation risk. Under Best Practice 6, *Predation Management Outreach Tools* (Hunt et al. 2019*a*), communication best practices and examples of such are reviewed and related to predation management, in general. There are not yet any existing management campaigns related to food subsidies of predators along the Atlantic Flyway, so this task will be both a challenge for Fire Island managers and an opportunity for the entire flyway. Relevant discussion from Hunt et al. (2019) is quoted here for ease of access:

> "Many public beaches feature anti-littering signs and signs prohibiting feeding of wildlife, but few, if any, connect this message to the problem of artificially inflated predator abundances in important shorebird nesting areas. Programs like the Crumb Clean campaign in the Redwoods National and State Parks (https://www.nps.gov/ redw/learn/nature/crumb-clean.htm) and Leave No Trace (https://lnt.org/) are great examples of public outreach campaigns that target specific behaviors related to threatened species and conservation and preservation of natural areas, respectively. Similarly, the Fed Bear is a Dead Bear (https://lnt.org/blog/fed-bear-dead-bear) slogan is a popular campaign across the United

States and may be a model for predation management messaging if adapted to local situations. However, some states have discontinued the use of this slogan as the public does not always understand that the garbage itself does not kill the bears, but rather the feeding-related habituation of bears leads to the need for lethal removal. Thus, further work is needed. At Crane Beach in Ipswich, MA, the management entity does deliver a leave no trace campaign and tells users that they must carry in and carry out all food and trash when using the beach (http://www.thetrustees.org/ about-us/press-room/press-releases/some-important-updates-from.html); raccoon and skunk predation at the site is currently minimal (J. Denoncour, personal communication, Trustees)." - Hunt et al. 2019*a*

Harassment

Harassment may serve as an additional non-lethal management tool. Hunt et al. (2019) describe several harassment techniques and considerations, including auditory and visual scare devices and chemical repellents that can be used to target specific predators. While these techniques can assist in frightening predators away from nesting areas and/or aid in aversive conditioning, there is the potential for habituation. In addition, some techniques may also frighten the target species for conservation away from nesting and foraging areas. Thus, it is important to carefully consider potential options and regularly re-assess the effectiveness of harassment efforts.

Similarly, targeted manipulation of behavioral responses may be used to influence the distribution and behavior of predator species. There is a growing body of scientific literature surrounding the "landscape of fear" concept (Laundré et al. 2010), which describes variation in space use due to differences in perceived risk, and potential applications to mitigate human-wildlife conflict. While the concept was originally applied to predator-prey interactions, it has recently gained attention in the study of human-wildlife interactions. For example, Suraci et al. (2019) found that pumas avoided areas where recording of human voices were broadcast during playback experiments, and moved more slowly in areas with speakers, indicating more cautious behavior. In the context of pest management, it has been suggested that managers can mimic different cues of predation risk or human presence to induce avoidance of a specific area, and/or manipulate behavioral responses of pest species. For example, Cromsigt et al. (2013) proposed the use of hunting to induce a fear response in ungulates and discourage use of areas in close proximity to people. This concept could potentially be extended to the management of foxes and other piping plover predators by mimicking the presence of competing predators or artificially inflating the perceived risk of human encounters in a given area. As with more traditional harassment methods, it is important to ensure that these actions do not also elicit a fear response in target species for conservation.

Feral cat colonies and presence

We detected feral cats on camera stations every year (2015-2019) in every monitoring area on Fire Island. Further, we did find piping plover bands in 1 of 3 feral cat scats collected. We observed the general public maintaining discreet, and sometimes not discreet, feral cat structures at Smith Point County Park throughout the monitoring period, often near the parking areas and campgrounds. We did not observe such structures in Robert Moses State Park and Fire Island National Seashore, yet feral cats still were detected on cameras in every year of monitoring. We recommend prohibition of feeding of cats (and foxes) in all management areas in order to minimize population growth of these piping plover predators. Other management techniques (lethal removal, trap-neuter-release, TNR) often are controversial, but should be considered through collaborative discussions between land managers, predator managers, and the interested public in order to minimize predation risk to piping plovers by these non-native predators. In the development of the predation best practices document for the Atlantic Flyway (Hunt et al. 2019*a*), many managers mentioned the support of feral cat colonies in close proximity to beneficiary species as a major unaddressed challenge in need of a coordinated communications campaign. As in the case of development of a food subsidy reduction campaign, this will be both a challenge for Fire Island managers but a Flyway-wide opportunity if success can be achieved.

Vegetation management

We found proximity to vegetation to be a strong predictor of fox habitat selection, both within home ranges and across the landscape. Efforts to continue to minimize vegetation in areas used by nesting piping plovers and their chicks will minimize both daytime rest site suitability and the potential concealment of foxes during hunting activities and other movements.

Lethal predation management options

Our findings on occupancy, home range use, and movements suggest that traditional lethal removal techniques may not be an efficient predator management option on Fire Island, given the frequency of transient individuals and regular movements between management units, unless conducted on an intensive level across the entire island. Lethal removal of red foxes, whether via shooting or trapping or fumigation of dens (see Hunt et al. 2019*a* for critiques of lethal options), is challenging and time-consuming. Unless extirpation is achieved, our data on movements suggest that recolonization of vacated territories will be rapid. Thus, we recommend deployment of the suite of nonlethal options mentioned above as the primary focus of predation management efforts. In the case of an individual fox learning to depredate nest exlosures, we first recommend consideration of non-lethal harassment options before lethal removal given the difficulty in ensuring that the 'problem' animal is the one lethally removed. As recommended consistently throughout Hunt et al. (2019), we recommend land managers work collaboratively with USDA Wildlife Services or other professional predator management agencies or companies in adaptively developing and implementing management options.

Literature Cited

Ables, E. D. 1983. Ecology of the red fox in North America. Pages 216–236 in M. W. Fox, editor. The wild canids: Their systematics, behavioral ecology and evolution. Robert E. Krieger Publishing Company, Inc., Malabar, Florida.

Altmann, J. 1974. Observational study of behavior: sampling methods. Behaviour 49:227-267.

Alberico, J. A. R. 1995. Floating eggs to estimate incubation stage does not affect hatchability. Wildlife Society Bulletin 23:212–216.

Andersen, K. H., N. S. Jacobsen, T. Jansen, and J. E. Beyer. 2017. When in life does density dependence occur in fish populations? Fish and Fisheries 18:656–667.

Anteau, M. J., and M. H. Sherfy. 2010. Diurnal variation in invertebrate catch rates by sticky traps: potential for biased indices of piping plover forage. Wetlands 30:757–762.

Arnold, T.W. 2010. Uninformative parameters and model selection using Akaike's Information Criterion. Journal of Wildlife Management 74:1175–1178.

Association of Fish and Wildlife Agencies [AFWA]. 2014. Best management practices for trapping red fox in the United States. Association of Fish and Wildlife Agencies, Washington, D.C., USA.

Aubry, K.B., M.J. Statham, B.N. Sacks, J.D. Perrine, S.M. Wisely. 2009. Phylogeography of the North American red fox: vicariance in Pleistocene forest refugia. Molecular Ecology 18:2668–2686.

Bailey, L.L., T.R. Simons, K.H. Pollock. 2004. Estimating site occupancy and detection probability parameters for terrestrial salamanders. Ecological Applications 14:692–702.

Baker, P. J., C. P. J. Robertson, S. M. Funk, S. Harris. 1998. Potential benefits of group living in the red fox, *Vulpes*. Animal Behavior 56:1411–1424.

Baker, P.J., S.M. Funk, S. Harris, P.C.L. White. 2000. Flexible spatial organization of urban foxes, Vulpes, before and during an outbreak of sarcoptic mange. Animal Behaviour 59:127–146.

Baker, P. J., and S. Harris. 2004. Red foxes: The behavioral ecology of red foxes in urban Bristol. Pages 207–216 in D. W. MacDonald and C. Sillero-Zubiri, eds. Biology and Conservation of Wild Canids. Oxford University Press, Oxford, England, UK.

Baker, P.J., S. Harris. 2006. Does culling reduce fox (*Vulpes vulpes*) density in commercial forests in Wales, UK? European Journal of Wildlife Research 52:99–108.

Barber, C., A. Nowak, K. Tulk, and L. Thomas. 2010. Predator exclosures enhance reproductive success but increase adult mortality of piping plovers (*Charadrius melodus*). Avian Conservation and Ecology 5:6 DOI: 10.5751/ACE-00419-050206

Barker, R. J. 1997. Joint modeling of live-recapture, tag-resight, and tag-recovery data. Biometrics 53:666-677.

Beale, C. M, P. Monaghan. 2004. Human disturbance: people as predation-free predators? Journal of Applied Ecology. 41:335-343.

Beaulieu, G., D. Austin, M.L. Leonard. 2014. Do nest exclosures affect the behavior of Piping Plovers (*Charadrius melodus melodus*) and their predators? Canadian Journal of Zoology 92:105–112.

Bellman, H. 2018. Hurricane and human-induced habitat changes on Fire Island and Westhampton Island, New York and the effects on breeding piping plovers (*Charadrius melodus*). Thesis, Virginia Tech, Blacksburg, VA, USA.

Bergquist, D.C., M.V. Levisen, and L. Forbes. 2011. Kiawah Island East End erosion and beach restoration project: survey of changes in potential macroinvertebrate prey communities in piping plover (*Charadrius melodus*) foraging habitats. Final Report prepared by the South Carolina Department of Natural Resources, Marine Resources Research Institute, Charleston, SC for the Town of Kiawah Island. 64 pp.

Black, K. and H. Taylor. 2019. Creation of a microscopic hair identification guide using museum specimens. Final report for Natural History Collections and Curation.

Boyce, M.S., P.R. Vernier, S.E. Nielsen, F.K.A. Schmiegelow. 2002. Evaluating resource selection functions. Ecological Modelling 157:281–300.

Burnham K.P. and D.P. Anderson. 2007. Model selection and multimodel inference: a practical information–theoretical approach. 2nd edition. Springer-Verlag, New York, NY, USA.

Byers, C. R., R. K. Steinhorst, and P. R. Krausman. 1984. Clarification of a technique for analysis of utilization-availability data. Journal of Wildlife Management 48:1050–1053.

Calenge, C. 2006. The package "adehabitat" for the R software: A tool for the analysis of space and habitat use by animals. Ecological Modelling 197:516–519.

Calvert, A. M., D. L. Amirault, F. Shaffer, R. Elliot, A. Hanson, J. McKnight, and P. D. Taylor. 2006. Population assessment of an endangered shorebird: the piping plover (*Charadrius melodus melodus)* in Eastern Canada. Avian Conservation and Ecology 1:4. DOI: 10.5751/ace-00091-010304.

Catlin, D. H., J. H. Felio, and J. D. Fraser. 2011*a*. Effect of great-horned owl trapping on chick survival in piping plovers. Journal of Wildlife Management 75:458–462.

Catlin, D. H., J. D. Fraser, J. H. Felio, and J. B. Cohen. 2011*b*. Piping plover habitat selection and nest success on natural, managed, and engineered sandbars. Journal of Wildlife Management 75:305–310.

Catlin, D. H., O. Milenkaya, K. L. Hunt, M. J. Friedrich and J. D. Fraser. 2014. Can river management improve piping plover's long-term survival on the Missouri River? Biological Conservation, 180:196–205.

Catlin, D. H., J. D. Fraser, and J. H. Felio. 2015. Demographic responses of piping plovers to habitat creation on the Missouri River. Wildlife Monographs: 192:1–42.

Catlin, D. H., D. Gibson, K. L. Hunt, M. J. Friedrich, C. E. Weithman, S. M. Karpanty, and J. D. Fraser. 2019*a*. Direct and indirect effects of nesting density on survival and breeding propensity of an endangered shorebird. Ecosphere 10:e02740.

Catlin, D., D. Gibson, M. J. Friedrich, K. L. Hunt, S. M. Karpanty, and J. D. Fraser. 2019*b*. Habitat selection and potential fitness consequences of two early - successional species with differing life - history strategies. Ecology and Evolution. *In press.* DOI: 10.1002/ece3.5834

Cavallini, P. 1996. Variation in the social system of the red fox. Ethology Ecology & Evolution 8:323–342.

Cecere, J. G., C. Catoni, I. Maggini, S. Imperio, and G. Gaibani. 2013. Movement patterns and habitat use during incubation and chick-rearing of Cory's shearwaters (*Calonectris diomedea diomedea*) (Aves: Verebrata) from Central Mediterranean: influence of seascape and breeding stage. Italian Journal of Zoology 80: 82–89.

Cisek, C. 2013. Analysis of the foraging behavior of non-breeding piping plover (*Charadrius melodus*) and the association between foraging habitat characteristics and foraging behavior. College of Charleston, Charleston, South Carolina, USA.

Ciuti, S., Northrup, J. M., Muhly, T. B., Simi, S., Musiani, M., Pitt, J. A., and Boyce, M. S. 2012. Effects of humans on behaviour of wildlife exceed those of natural predators in a landscape of fear. PloS One, 7:e50611. DOI:10.1371/journal.pone.0050611

Cohen, J. B., J. D. Fraser, and D. H. Catlin. 2006. Survival and site fidelity of piping plovers on Long Island, New York. Journal of Field Ornithology 77:409–417.

Cohen, J. B., E. H. Wunker, J. D. Fraser. 2008. Substrate and vegetation selection by nesting piping plovers. The Wilson Journal of Ornithology 120:404–407.

Cohen, J. B., L. M. Houghton, and J. D. Fraser. 2009. Piping plover nesting density and reproductive success in response to storm and human-created habitat changes. Wildlife Monographs 173:1–24.

Cohen, J. B., and J. D. Fraser. 2010. Piping plover foraging distribution and prey abundance in the pre-laying period. The Wilson Journal of Ornithology 122:578–582.

Cohen, J. B., A. Hecht, K. F. Robinson, E. E. Osnas, A. J. Tyre, C. Davis, A. Kocek, B. Maslo, and S. M. Melvin. 2016. To exclose nests or not: structured decision making for the conservation of a threatened species. Ecosphere 7:e01499. DOI: 10.1002/ecs2.1499

Conner, L.M. and B.W. Plowman. 2001. Using euclidean distances to assess nonrandom habitat use. Pages 275–290 in J. J. Millspaugh, and J. M. Marzluff, eds. Radio tracking and animal populations. Academic Press, San Diego, CA, USA.

Conner, L.M., M.D. Smith, L.W. Burger. 2003. A comparison of distance-based and classification-based analyses of habitat use. Ecology 84:526-531.

Cooper, A. B., and J. J. Millspaugh. 2001. Accounting for variation in resource availability and animal behavior in resource selection studies. Pages 243-274 in J. J. Millspaugh, and J. M. Marzluff, eds. Radio tracking and animal populations. San Diego, CA: Academic Press.

Coulson, T., T. H. G. Ezard, F. Pelletier, G. Tavecchia, N. C. Stenseth, D. Z. Childs, J. G. Pilkington, J. M. Pemberton, L. E. B. Kruuk, T. H. Clutton-Brock, and M. J. Crawley. 2008. Estimating the functional form for the density? Dependence from life history data Ecology 89:1661–1674.

Cromsigt, J.P.G.M., D.P.J. Kujiper, M. Adam, R.L. Beschta, M. Churski, A. Eycott, G.I.H. Kerley, A. Mysterud, K. Schmidt, K. West. 2013. Hunting for fear: innovating management of human–wildlife conflicts. Journal of Applied Ecology 50:544–549.

Cuthbert, F. J., B. Scholtens, L. C. Wemmer, and R. McLain. 1999. Gizzard contents of piping plover chicks in northern Michigan. Wilson Bulletin 111:121–123.

Davis, K. L., K. L. Schoenemann, D. H. Catlin, K. L. Hunt, M. J. Friedrich, S. J. Ritter, J. D. Fraser and S. M. Karpanty. 2017. Hatch-year piping plover (*Charadrius melodus*) prospecting and habitat quality influence second-year nest site selection Auk 134:92–103.

Debelica, A. and M.L. Thies. 2009. An atlas and key to the hair of terrestrial Texas mammals. Texas Tech Museum Special Publication 55:1–102.

Dekker, J. J. A., A. Stein, I. M. A. Heitkönig. 2001. A spatial analysis of red fox (*Vulpes vulpes*) in the Dutch coastal dune area. Journal of Zoology 255:505–510.

DeRose-Wilson, A., K. Sloan, J. D. Fraser, D. H. Catlin, S. M. Karpanty. 2013. Response of piping plover and their Invertebrate Prey to Habitats Created by Hurricane Sandy. Annual Report prepared by the Virginia Tech Shorebird Program, Virginia Tech, Blacksburg, VA, USA. 50 pp.

DeRose-Wilson, A., J. D. Fraser, D. H. Catlin, S. M. Karpanty, S. J. Ritter. 2014. Response of piping plovers and their invertebrate prey to habitats created by Hurricane Sandy. Annual Report prepared by the Virginia Tech Shorebird Program, Virginia Tech, Blacksburg, VA, USA. 43 pp.

DeRose-Wilson, J. D. Fraser, S. M. Karpanty, M. D. Hillman. 2015. Effects of overflights on incubating Wilson's plover behavior and hear rate. Th Journal of Wildlife Management. 79:1246–1254.

DeRose-Wilson, A. L., K. L. Hunt, J. D. Monk, D. H. Catlin, S. M. Karpanty, and J. D. Fraser. 2018. Piping plover chick survival negatively correlated with beach recreation. Journal of Wildlife Management 82:1608–1616.

Díaz-Ruiz, F., J. Caro, M. Delibes-Mateos, B. Arroyo, and P. Ferreras. 2016. Drivers of red fox (*Vulpes vulpes*) daily activity: prey availability, human disturbance, or habitat structure? Journal of Zoology 298:128–138.

Doherty, P. J., A. H. Heath. 2011. Factors affecting piping plover hatching success on Long Island, New York. Journal of Wildlife Management 75:109-115.

Doncaster, C. P., and D. W. Macdonald. 1997. Activity patterns and interactions of red foxes (*Vulpes vulpes*) in Oxford city. Journal of Zoology 241:73–87. Dove, C.J. and S.L. Koch. 2010. Microscopy of feathers: A practical guide for forensic feather identification. Journal of American Society of Trace Evidence Examiners 1:15–61.

Elias, S. P., J. D. Fraser, and P. A. Buckley. 2000. Piping plover brood foraging ecology on New York barrier islands. Journal of Wildlife Management 64:346–354.

Elliott-Smith, E. and S. M. Haig (2004). Piping Plover (*Charadrius melodus*), version 2.0. In The Birds of North America (A. F. Poole, Editor). Cornell Lab of Ornithology, Ithaca, NY, USA DOI:10.2173/bna.2

Espie, H. M., R. M. Brigham, P. C. James. 1996. Habitat selection and clutch fate of piping plovers (*Charadrius melodus*) breeding at Lake Diefenbaker, Saskatchewan. Canadian Journal of Zoology 74:1069–1075.

Environmental Systems Research Institute (ESRI). 2018. ArcMap 10.6. Redlands, CA.

Epstein, M.B., G.A. Feldhammer, R.L. Joyner. 1983. Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies 37:161–172.

Evans, N. T., B. P. Olds, M. A. Renshaw, C. R. Turner, Y. Li, C. L. Jerde, A. R. Mahon, M. E. Pfrender, G. A. Lamberti, and D. M. Lodge. 2016. Quantification of mesocosm fish and amphibian species diversity via environmental DNA metabarcoding. Molecular Ecology Resources 16:29–41.

Farmer, A. H., and A. H. Parent. 1997. Effects of the landscape on shorebird movements at spring migration stopovers. Condor 99:698–707.

Fisher, R. A. 1925. Statistical Methods for Research Workers. Oliver & Boyd, Edinburgh, Scotland.

Fraser, J. D. and D. H. Catlin. 2019. Habitat ecology and conservation of *Charadrius* plovers. Pages 217–243 *in* M. A. Colwell and S. M. Haig, editors. The Population Ecology and Conservation of *Charadrius* plovers. CRC Press, Boca Raton, Florida, USA.

Frid, A. and L. Dill. 2002. Human-caused disturbance stimuli as a form of predation risk. Conservation Ecology 6:11 DOI: 10.5751/ES-00404-060111

Friedrich, M. J., K. L. Hunt, D. H. Catlin, and J. D. Fraser. 2015. The importance of site to mate choice: mate and site fidelity in piping plovers. The Auk 132:265–276.

Gamelon, M., V. Grøtan, S. Engen, E. Bjørkvoll, M. E. Visser, and B. E. Sæther. 2016. Density dependence in an age-structured population of great tits: Identifying the critical age classes. Ecology 97:2479–2490.

Gerik, D.E. 2018. Assessment and application of DNA metabarcoding for characterizing arctic shorebird chick diets. Thesis, University of Alaska Fairbanks, USA.

Getz, W.M., S. Fortmann-Roe, P.C. Cross, A.J. Lyons, S.J. Ryan, C.C. Wilmers 2007. LoCoH: Nonparameteric kernel methods for constructing home ranges and utilization distributions. PLoS ONE 2: e207. DOI:10.1371/journal.pone.0000207.

Gieder, K. 2015. Assessing the effects of sea-level rise on piping plover (Charadrius melodus) nesting habitat, and the ecology of a key mammalian shorebird predator, on Assateague Island. Ph.D. Dissertation, Virginia Tech.

Gill, J., Sutherland, W., & Watkinson, A. 1996. A method to quantify the effects of human disturbance on animal populations. Journal of Applied Ecology 33:786-792.

Goszczyński, J. 2002. Home ranges in red fox: territoriality diminishes with increasing area. Acta Theriologica 47, Suppl. 1:103–114.

Greentree, C., G. Saunders, L. McLeod, J. Hone. 2000. Lamb predation and fox control in south–eastern Australia. Journal of Applied Ecology 37:935–943.

Güthlin, D., I. Storch, and H. Küchenhoff. 2014. Is it possible to individually identify red foxes from photographs? Wildlife Society Bulletin 38:205–210.

Haffner, C. D., F. J, Cuthbert, and T. W. Arnold. 2009. Space use by Great Lakes piping plovers during the breeding season. Journal of Field Ornithology 80:270–279.

Harding, E.K., D.F. Doak, J.D. Albertson. 2001. Evaluating the effectiveness of predator control: the non–native red fox as a case study. Conservation Biology 15:1114–1122.

Haig, S. M., and L. W. Oring. 1988. Mate, site, and territory fidelity in piping plovers. Auk 105:268–277.

Heisey, D. M., T. L. Shaffer, and G. C. White. 2007. The ABC's of nest survival: theory and application from a biostatistical perspective. Studies in Avian Biology 34:1–21.

Henry, J.D. 1986. Red fox: The catlike canine. Smithsonian Institution Press, Washington, D.C.

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. Annual Report prepared by the Virginia Tech Shorebird Project, Virginia Tech, Blacksburg, VA, USA. 143pp.

Hijmans, R. J. (2016). geosphere: Spherical Trigonometry. R package version 1.5-5. https://CRAN.R-project.org/package=geosphere

Hillman, M. D., S. M. Karpanty,J. D. Fraser, A. DeRose-Wilson. 2015. Effects of aircraft and recreation on colonial waterbird nesting behavior. The Journal of Wildlife Management. 79:1192–1198.

Hines, J. E. 2006. PRESENCE- Software to estimate patch occupancy and related parameters. USGS–PWRC. http://www.mbr–pwrc.usgs.gov/software/presence.html. Accessed 10 October 2019.

Hinton, J. W, F.T. van Manen, M.J. Chamberlain. 2015. Space use and habitat selection by resident and transient coyotes (*Canis latrans*). PloS ONE 10: e0132203. DOI: 10.1371/journal.pone.0132203.

Holt, D. W., M. D. Larson, N. Smith, D. L. Evans, and D. F. Parmelee. 2015. Snowy Owl (*Bubo scandiacus*), version 2.0. In The Birds of North America (P. G. Rodewald, Editor). Cornell Lab of Ornithology, Ithaca, Ny, USA. DOI:10.2173/bna.10

Hunt, K. L., J. D. Fraser, M. J. Friedrich, S. M. Karpanty, and D. H. Catlin. 2018*a*. Demographic response of Piping Plovers suggests that engineered habitat restoration is no match for natural riverine processes. Condor 120:149–165.

Hunt, K., D. Gibson, D. Catlin. 2018*b*. Atlantic flyway disturbance project. Biological data collection report. Final report prepared by the Virginia Tech Shorebird Project, Virginia Tech, Blacksburg, VA, USA. 87pp.

Hunt, K. L, Karpanty, S. M., Davis, K. L., Wilke, A., Myers, N., Spiegel, C., Shulte, S., Catlin, D. H., Fraser, J. D. 2018*a*. Guidance and best practices for coordinated predation management to benefit temperate breeding shorebirds in the Atlantic Flyway. U.S. Fish and Wildlife Service and National Fish and Wildlife Foundation.

Hunt, K. L, D. Gibson, M. J. Friedrich, C. J. Huber, J. D. Fraser, S. M. Karpanty, and D. H. Catlin. 2019*b*. Using nest captures and video cameras to estimate survival and abundance of breeding Piping Plovers *Charadrius melodus.* Ibis. DOI:10.1111/ibi.12726

Houghton, L. M. 2005. Piping plover population dynamics and effects of beach management practices on piping plovers at West Hampton Dunes and Westhampton Beach, New York. Dissertation, Virginia Tech, Blacksburg, VA, USA.

Ivan, J. S. and R. K. Murphy. 2005. What preys on piping plover eggs and chicks? Wildlife Society Bulletin 33:113–119.

Johnson, D.H. 1980.The comparison of usage and availability measurements for evaluating resource preference. Ecology 61: 65–71.

Jones, D., and J. Theberge. 1982. Summer home range and utilization of the red fox (*Vulpes vulpes*) in a tundra habitat, northwest British Columbia. Canadian Journal of Zoology 60:807–812.

Keating, K.A. and S. Cherry. 2004. Use and interpretation of logistic regression in habitat-selection studies. Journal of Wildlife Management 68:774–789.

Kellner, K. 2018. jagsUI: A wrapper around 'rjags' to streamline 'JAGS' analyses. Retrieved from: https://cran.r-proje ct.org/web/packages/jagsU I/index.html

Kelly, M.J. and E.L. Holub. 2008. Camera trapping of carnivores: Trap success among camera types and across

species, and habitat selection by species, on Salt Pond Mountain, Giles County, Virginia. Northeastern Naturalist 15:249–262.

Klare, U., J.F. Kamler, D.W. Macdonald. 2011. A comparison and critique of different scat-analysis methods for determining carnivore diet. Mammal Review 41:294–392.

Kratzmann, M. G., and C. J. Hapke. 2012. Quantifying anthropogenically driven morphologic changes on a barrier island: Fire Island National Seashore, New York. Journal of Coastal Research 28:76–88.

Krim, P. M., T. L. Bashore, and J. G.L. Kirkland. 1990. Den site characteristics and food habits of the red fox (*Vulpes vulpes*) on Assateague Island, Maryland. Virginia Journal of Science 41:340–351.

Kwon, E., J. D. Fraser, D. H. Catlin, S. M. Karpanty, C. E. Weithman, and B. Muiznieks. 2018. Presence of ghost crabs and piping plover nesting success. The Journal of Wildlife Management 82:850–856.

Lampila, S., M. Orell, E. Belda, and K. Koivula. 2006. Importance of adult survival, local recruitment and immigration in a declining boreal forest passerine, the willow tit Parus montanus. Oecologia 148:405–413.

Larson, R.N., D.J. Morin, I.A. Wierzbowska, K.R. Crooks. 2015. Food habits of coyotes, gray foxes, and bobcats in a coastal southern California urban landscape. Western North American Naturalist 75:339–347.

Laundré, J.W., L. Hernández, W.J. Ripple. 2010. The landscape of fear: Ecological implications of being afraid. The Open Ecology Journal 3:1–7.

Lauro, B. and J. Tanacredi. 2002. An examination of predatory pressures on piping plovers nesting at Breezy Point, New York. Waterbirds: The International Journal of Waterbird Biology 25:401–409

Leewis, L., P. M. van Bodegom, J. Rozema, and G. M. Janssen. 2012. Does beach nourishment have long-term effects on intertidal macroinvertebrate species abundance? Estuarine, Coastal and Shelf Science 113:172–181.

Lengyel, S. 2006. Spatial differences in breeding success in the pied avocet *Recurvirostra avosetta*: effects of habitat on hatching success and chick survival. Journal of Avian Biology 37:381–395.

Lentz, E. E., and C. J. Hapke. 2011. Geologic framework influences on the geomorphology of an anthropogenically modified barrier island: Assessment of dune/beach changes at Fire Island, New York. Geomorphology 126:82–96.

Lloyd, H. G. 1980. The red fox. B.T. Batsford Ltd, London, England, UK.

Loegering, J. P., and J. D. Fraser. 1995. Factors affecting piping plover chick survival in different brood-rearing habitats. Journal of Wildlife Management 59:646–655.

Lucherini, M., and S. Lovari. 1996. Habitat richness affects home range size in the red fox (*Vulpes vulpes*). Behavioural Processes 36:103–106.

Lukacs, P. M., V. J. Dreitz, F. L. Knopf, and K. P. Burnham. 2004. Estimating survival probabilities of unmarked dependent young when detection is imperfect. Condor 106:926–931.

Lyons, A.J., W.C. Turner, W.M. Getz. 2013. Home range plus: A space-time characterization of movement over real landscapes. Movement Ecology 2013:1–2.

Lyons, A.J. 2014. T-LocoH for R: Tutorial and Users Guide. <http://tlocoh.r-forge.r-project.org/tlocoh_tutorial_2014-08-17.pdf>. Accessed 15 May 2019.

MacDonald, D.W. 1987. Running with the fox. Unwin Hyman, London, England.

MacDonald, D.W., S. Creel, M.G. Mills. 2004. Canid society. Pages 85–106 in D.W. MacDonald and C. Sillero-Zubiri, eds. The Biology and Conservation of Wild Canids. Oxford University Press, Oxford, England, UK.

MacKenzie, D.I., J.D. Nichols, G.B. Lachman, S. Droege, J.A. Royle, C.A. Langtimm. 2002. Estimating site occupancy rates when detection probabilities are less than one. Ecology 83:2248–2255.

Mackley, E. K., R. A. Phillips, J. R. D. Silk, E. D. Wakefield, V. Afanasyev, and R. W. Faness. 2011. At-sea activity patterns of breeding and nonbreeding white-chinned petrels *Procellaria aequinoctialis* from South Georgia. Marine Biology 158:429–438.

Majka, C. G., and F. Shaffer. 2008. Beetles (Coleoptera) in the diet of piping plovers in the Iles de la Madeleine, Québec, Canada. Wader Study Group Bulletin 115:77–83.

Manly, B.F.J., L.L. McDonald, D.L. Thomas, T.L. MacDonald, W.P. Erickson. 2002. Resource selection by animals: Statistical design and analyses for field studies. Kluwer Academic Publisher, London, England.

Marlow, N.J., P.C. Thompson, K. Rose, N.E. Kok. 2016. Compensatory responses by a fox population to artificial density reduction in a rangeland area in Western Australia. Conservation Science Western Australia 10:1–10.

McCabe, D. J., and N. J. Gotelli. 2000. Effects of disturbance frequency, intensity, and area on assemblages of stream macroinvertebrates. Oecologia 124:270–279.

McDonald, J.H. 2014. Handbook of Biological Statistics, 3rd ed. Sparky House Publishing, Baltimore, Maryland.

Meek, P., and G. Saunders. 2000. Home range and movement of foxes (*Vulpes vulpes*) in coastal New South Wales, Australia. Wildlife Research 27:663–668.

Meisner, K., P. Sunde, K. K. Clausen, P. Clausen, C. C. Fælled, M. Hoelgaard. 2014. Foraging ecology and spatial behaviour of the red fox (*Vulpes vulpes*) in a wet grassland ecosystem. Acta Theriologica 59:377–389.

Melvin, S. M., L. H. MacIvor, and C. R. Griffin. 1992. Predator exclosures: a technique to reduce predation at piping plover nests. Wildlife Society Bulletin 20:143–148.

Miles, K.A., M.N. Holtz, Z.T. Lounsberry, B.N. Sacks. 2015. A paired comparison of scat-collecting versus scat-swabbing methods for noninvasive recovery of mesocarnivore DNA from an arid environment. Wildlife Society Bulletin 39:797–803.

Million, A., X. Lambin, S. Devillard, and M. Schaub. 2019. Quantifying the contribution of immigration to population dynamics: a review of methods, evidences and perspectives in birds and mammals. Biological Reviews 94:2049–2067.

Monk, J. D., K. Miles, A. DeRose-Wilson, J. D. Fraser, D. H. Catlin, S. Ritter, K. M. Walker, S. M. Karpanty. 2015. Piping plover and red fox monitoring on Fire Island, NY. Annual Report prepared by Virginia Tech Shorebird Program, Virginia Tech, Blacksburg, VA, USA. 60pp.

Morin, D.J. and M.J. Kelly. 2017. The dynamic nature of territoriality, transience, and biding in an exploited coyote population. Wildlife Biology DOI: 10.2981/wlb.00335

Morris, L.R., K.M. Proffitt, J.K. Blackburn. 2016. Mapping resource selection functions in wildlife studies: concerns and recommendations. Applied Geography 76:173-183.

Murphy, A. J., Z. J. Farris, S. Karpanty, M. J. Kelly, K. A. Miles, F. Ratelolahy, R. P. Rahariniaina, and C. D. Golden. 2017. Using camera traps to examine distribution and occupancy trends of ground-dwelling rainforest birds in north-eaastern Madagascar. Bird Conservation International 28:567–580

Murphy, R. K., I. M. G. Michaud, D. R. C. Prescott, J. S. Ivan, B. J. Anderson, and M. L. French-Pombier. 2003. Predation on adult piping plovers at predator exclosure cages. Waterbirds 26:150–155.

Neuman, K. K., G. W. Page, L.E. Stenzel, J. C. Warriner, and J.S. Warriner. 2004. Effect of mammalian predator management on snowy plover breeding success. Waterbirds 27:247–376.

Nevoux, M., O. Gimenez, D. Arlt, M. Nicoll, C. Jones, and K. Norris. 2011. Population regulation of territorial species: both site dependence and interference mechanisms matter. Proceedings of the Royal Society B: Biological Sciences 278:2173–2181.

Newsome, T.M., J.A. Dellinger, C.R. Pavey, W.J. Ripple, C.R. Shores, A.J. Wirsing, C.R. Dickman. 2015. The ecological effects of providing resource subsidies to predators. Global Ecology and Biogeography 24:1–11.

Newton, I. 1998. Population Limitation in Birds. Academic Press. New York, New York, USA.

Nichols, J. D., and A. Kaiser. 1999. Quantitative studies of bird movement: A methodological review. Bird Study 46:S289–S298.

Niehaus, A.C., D. R. Ruthrauff, B. J. McCaffery. 2004. Response of predators to western sandpiper nest exclosures. Waterbirds 27:79-82.

O'Connell Jr., A.F., N.W. Talancy, L.L. Bailey, J.R. Sauer, R.C. Cook, A.T. Gilbert. 2006. Estimating site occupancy and detection probability parameters for meso– and large mammals in a coastal ecosystem. Journal of Wildlife Management 70:1625–1633.

Patterson, M. E., J. D. Fraser, and J. W. Roggenbuck. 1991. Factors affecting piping plover productivity on Assateague Island. Journal of Wildlife Management 55:525–531.

Phillips, M., and P. C. Catling. 1991. Home ranges and activity patterns of red foxes in Nadgee nature reserve. Wildlife Research 18:677–686.

Pietz, P. J. and A. G. Granfors. 2000. White-tailed deer (*Odocoileus virginianus*) predation on grassland songbird nestlings. The American Midland Naturalist 144:419–422.

Plissner, J. H., S. M. Haig, and L. W. Oring. 2000. Postbreeding movements of American avocets and implications for wetland connectivity in the Western Great Basin. The Auk 117:290–298.

Pollock, K.H., J.D. Nichols, T.R. Simon, G.L. Farnsworth, L.L. Bailey, J.R. Sauer. 2002. Large scale wildlife monitoring studies: statistical methods for design and analysis. Envirometrics 13:105–119.

Potts, J.R., S. Harris, L. Giuggioli. 2013. Quantifying behavioral changes in territorial animals caused by sudden population declines. The American Naturalist 182:E73–E82.

Pradel, R. 1996. Animal dispersal within subdivided populations: an approach based on monitoring individuals. Acta Oecologica 17:475–483.

Prugh, L. R., C. J. Stoner, C. W. Epps, W. T. Bean, W. J. Ripple, A. S. Laliberte, and J. S. Brashares. 2009. The Rise of the mesopredator BioScience 59:779–791.

Purvis, A., J. L. Gittleman, G. Cowlishaw, and G. M. Mace. 2000. Predicting extinction risk in declining species. Proceedings of the Royal Society B: Biological Sciences 267:1947–1952.

Quinn, C.B., P.B. Alden, B.N. Sacks. 2019. Noninvasive sampling reveals short-term genetic rescue in an insular fox population. Journal of Heredity 110:559–576.

R Core Team. 2016. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/.

Reinert, S. 1984. Use of introduced perches by raptors: Experimental results and management implications Raptor Research 18:25–29.

Rickenbach, O., Grüebler, M., Schaub, M., Koller, A., Naef-daenzer, B. & Schifferli, L. 2011. Exclusion of ground predators improves Northern Lapwing *Vanellus vanellus* chick survival. Ibis 153:531–542.

Ricklefs, R. E. 1984. Egg dimensions and neonatal mass of shorebirds Condor 86:711.

Rimmer, D. W., and R. D. Deblinger. 1990. Use of predator exclosures to protect piping plover nests. Journal of Field Ornithology 61:217–223.

Robinson, S., J. Fraser, D. Catlin, S. Karpanty, J. Altman, R. Boettcher, K. Holcomb, C. Huber, K. Hunt, and A. Wilke. 2019. Irruptions: evidence for breeding season habitat limitation in Piping Plover (Charadrius melodus) Avian Conservation and Ecology 14:19 DOI:10.5751/ACE-01373-140119

Robinson, S. G., C. E. Weithman, H. A. Bellman, S. P. Prisley, J. D. Fraser, D. H. Catlin, S. M. Karpanty. *In press*. Assessing error in locations of conspicuous wildlife using handheld GPS units and location offset methods. Wildlife Society Bulletin. DOI: 10.1002/wsb.1055.

Robinson, S. G., D. Gibson, T. V. Riecke, J. D. Fraser, H. A. Bellman, A. DeRose-Wilson, S. M. Karpanty, K. M. Walker, and D. H. Catlin. *In prep*. Piping plover population increase after Hurricane Sandy mediated by immigration and reproductive output. Condor 122:1–20.

Roche, E. A., T.W. Arnold, and F. J. Cuthbert. 2010. Apparent nest abandonment as evidence of breeding-season mortality in Great Lakes piping plovers (*Charadrius melodus*) The Auk 127:402–410.

Rodenhouse, N. L., T. W. Sherry, and R. T. Holmes. 1997. Site-Dependent Regulation of Population Size: a New Synthesis Introduction: the Problem of Population Regulation. Ecology 78:2025–2042.

Rotella, J. J., S. J. Dinsmore, and T. L. Shaffer. 2004. Modeling nest-survival data: a comparison of recently developed methods that can be implemented in MARK and SAS. Animal Biodiversity and Conservation 27:187–205.

Royama, T. 1977. Population persistence and density dependence. Ecological Monographs 47:1–35.

Sacks, B.N., M.J. Statham, J.D. Perrine, S.M. Wisely, K.B. Aubry. 2010. North American montane red foxes: Expansion, fragmentation, and the origin of the Sacramento Valley red fox. Conservation Genetics 11:1523–1539.

Sacks, B.N., M. Moore, M.J. Statham, H.U. Wittmer. 2011. A restricted hybrid zone between native and introduced red fox (*Vulpes vulpes*) populations suggests reproductive barriers and competitive exclusion. Molecular Ecology 20:326–341.

Saether, B.-E., S. Engen, and E. Matthysen. 2002. Demographic characteristics and population dynamical patterns of solitary birds. Science 295:2070–2073.

Saunders, S. P., F. J. Cuthbert, and E. F. Zipkin. 2018. Evaluating population viability and efficacy of conservation management using integrated population models. Journal of Applied Ecology 55:1380–1392.

Schaub, M., H. Jakober, and W. Stauber. 2013. Strong contribution of immigration to local population regulation: Evidence from a migratory passerine. Ecology 94:1828–1838.

Shaffer, F., and P. Laporte. 1994. Diet of piping plovers on the Magdalen Islands, Quebec. Wilson Bulletin 106:531–536.

Shaffer, T. L. 2004. A unified approach to analyzing nest success. Auk 121:526–540.

Shapira, I., H. Sultan, U. Shanas. 2008. Agricultural farming alters predator-prey interactions in nearby natural habitats. Animal Conservation 11:1–8.

Silva, M., K. M. Johnson, and S. B. Opps. 2009. Habitat use and home range size of red foxes in Prince Edward Island (Canada) based on snow-tracking and radio-telemetry data. Central European Journal of Biology 4:229–240.

Skórka, P., M. Lenda, and W. J. Sutherland. 2016. Response of young and adult birds to the same environmental variables and different spatial scales during post breeding period. Landscape Ecology 31:2063–2078.

Soulsbury, C.D., P J. Baker, G. Iossa, S. Harris. 2010. Red foxes (Vulpes vulpes). Pages 63–78 in S.D. Gerht, S.P.D. Riley, B.L. Cypher, editors. Urban carnivores: Ecology, conflict, and conservation. The Johns Hopkins University Press, Baltimore, MD, USA.

Speybroeck, J., D. Bonte, W. Courtens, T. Gheskiere, P. Grootaert, J. P. Maelfait, M. Mathys, S. Provoost, K. Sabbe, E. W. M. Stienen, V. Van Lancker, M. Vincx, and S. Degraer. 2006. Beach nourishment: An ecologically sound coastal defence alternative? A review. Aquatic Conservation: Marine and Freshwater Ecosystems 16:419–435.

Staine, K. J., and J. Burger. 1994. Nocturnal foraging behavior of breeding piping plovers (*Charadrius melodus*) in New Jersey. Auk 111:579–587.

Statham, M.J., B.N. Sacks, K.B. Aubry, J.D. Perrine, S.M. Wisely. 2013. The origin of recently established red fox populations in the United States: Translocations or natural range expansions? Journal of Mammalogy 93:52–65.

Sullins, D. S., D. A. Haukos, J. M. Craine, J. M. Lautenbach, S. G. Robinson, J. D. Lautenbach, J. D. Kraft, R. T. Plumb, J. H. Reitz, B. K. Sandercock, and N. Fierer. 2018. Identifying the diet of a declining prairie grouse using DNA metabarcoding. The Auk 135:583–608.

Suraci, J. P., M. Clinchy, L. Y. Zanette, and C. C. Wilmers. 2019. Fear of humans as apex predators has landscape-scale impacts from mountain lions to mice. Ecology Letters 22:1578–1586.

Telemetry Solutions. 2014. Collar SW 2.09. Concord, CA.

Towerton A.L., T.D. Penman, R.P. Kavanagh, C. R. Dickman. 2011. Detecting pest and prey responses to fox control across the landscape using remote cameras. Wildlife Research 38:208–220.

Trolliet, F., M. Huynen, C. Vermeulen, and A. Hambuckers. 2014. Use of camera traps for wildlife studies. A review. Biotechnology, Agronomy, Society and Environment 18:446–454.

Turchin, P. 1990. Rarity of density dependence or population regulation with lags? Nature 344:660–663.

Turner, M. G., 2010. Disturbance and landscape dynamics in a changing world. Ecology 91:2833–2849.

U.S. Army Corps of Engineers [USACE]. 1996. Breach Contingency Plan, Fire Island to Montauk Point, Long Island, New York. New York District, August 1995, revised January 1996.

U.S. Army Corps of Engineers [USACE]. 2002. Reformulation of the shore protection and storm damage reduction project. South Shore of Long Island, New York – Fire Island Inlet to Montauk Point. Final Small Mammal and Herpetile Survey Summary Report May–August 2002. NY District.

U.S. Army Corps of Engineers [USACE]. 2014. Fire Island Inlet to Moriches Inlet to Fire Island Stabilization Project. U.S. Army Corps of Engineers, New York, New York, USA.

U.S. Fish and Wildlife Service [USFWS]. 1995. Fish and Wildlife coordination act report section 2(b), Fire Island Inlet to Montauk Point, Long Island, New York, Breach Contingency Plan. USFWS, Long Island Field Office, Islip, NY, USA.

U.S. Fish and Wildlife Service [USFWS]. 2014. Biological opinion and conference opinion, Fire Island to Moriches Inlet, Fire Island Stabilization Project. USFWS, Suffolk County, NY, USA.

U. S. Fish and Wildlife Service [USFWS] 2019. Abundance and productivity estimates – 2018 update: Atlantic Coast piping plover population. Hadley, Massachusetts.

Voigt, D.R. 1987. Red fox. Pages 378–392 in M. Novak, J.A. Baker, M.E. Obbard, and B. Malloch, eds. Wild furbearer management and conservation in North America. Ministry of Natural Resources, Ontario, Canada.

Walker, K. M., J. D. Fraser, D. H. Catlin, S. J. Ritter, S. G., Robinson, H. A. Bellman, A. DeRose-Wilson, S. M. Karpanty, and S. T. Papa. 2019. Hurricane Sandy and engineered response created habitat for a threatened shorebird. Ecosphere 10:e02771. DOI:10.1002/ecs2.2771

Walter, S. T., P. L. Leberg, J. J. Dindo, and J. K. Karubian. 2014. Factors influencing Brown Pelican (*Pelecanus occidentalis*) foraging movement patterns during the breeding season. Canadian Journal of Zoology 92:885–891.

Weithman, C. E., D. Gibson, K. L. Hunt, M. J. Friedrich, J. D. Fraser, S. M. Karpanty, and D. H. Catlin. 2017. Senescence and carryover effects of reproductive performance influence migration, condition, and breeding propensity. Ecology and Evolution 7:11044–11056. DOI:10.1002/ece3.3533.

Weithman, C. E. 2019. Piping Plover (*Charadrius melodus*) demography, behavior, and movement on the Outer Banks of North Carolina. Thesis, Virginia Tech, Blacksburg, VA, USA.

Weithman, C. E., S. G. Robinson, K. L. Hunt, J. Altman, H. A. Bellman, A. L. DeRose-Wilson, K. M. Walker, J. D. Fraser, S. M. Karpanty, and D. H. Catlin. 2019. Growth of two Atlantic Coast Piping Plover populations. Condor 121:1–14.

Weston, M. A., and M. A. Elgar. 2005. Disturbance to brood-rearing hooded plover *Thinornis rubricollis:* responses and consequences. Bird Conservation International 15:193–209.

White, G. C., and K. P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46:S120–S139.

Whitfield, D. P. 2003. Density-dependent mortality of wintering Dunlins *Calidris alpina* through predation by Eurasian Sparrowhawks *Accipiter nisus*. Ibis 145:432–438.

Wilcox, L. 1959. A twenty year banding study of the piping plover. The Auk 76:129–152.

Wilcoxon, F. 1945. Individual comparison by ranking methods. Biometrics Bulletin 1:80–83.

Wiltermuth, M. T., M. J. Anteau, M. H. Sherfy, and A. T Pearse. 2015. Habitat selection and movements of piping plover broods suggest a tradeoff between breeding stages. Journal of Ornithology 156:999–1013.

Winter, L. and G.E. Wallace. 2006. Impacts of feral and free-ranging cats on bird species of conservation concern. A five-state review of New York, New Jersey, Florida, California, and Hawaii. American Bird Conservancy, Washington, D.C., USA.

Yirga, G., H.H.D. Iongh, H. Leirs, J. Gebrihiwot, J. Deckers, H. Bauer. 2012. Adaptability of large carnivores to changing anthropogenic food sources: diet change of spotted hyena (*Crocuta crocuta*) during Christian fasting period in northern Ethiopia. Journal of Animal Ecology 81:1052–1055.

Zeigler, S. L., E. R. Thieler, B. T. Gutierrez, N. G. Plant, M. Hines, J. D. Fraser, D. H. Catlin, and S. M. Karpanty. 2017. Smartphone technologies and Bayesian networks to assess shorebird habitat selection. Wildlife Society Bulletin 41:666–677.

Zonick, C. A. 2000. The winter ecology of piping plovers (*Charadrius melodus*) along the Texas Gulf Coast. Dissertation, University of Missouri, Columbia, MO, USA.

Appendix A. Supplementary table for Part 2, Section 3: Compare changes in numbers, density, and distribution of piping plovers in various habitats on Fire Island in 'Effectiveness monitoring and targeted studies in support of FIMI conservation measures.'

Table A-1. Nesting piping plover density of cross-section island subsites in which piping plover were monitored on Fire Island and Westhampton Island, 2013–2019. See Table 3 for subsite information.

Subsite	Year	Pairs	Ro (fledglings/pair)	Density (pairs/ ha)
BRE	2013	0	0.00	0.00
BRE	2014	0	0.00	0.00
BRE	2015	0	0.00	0.00
BRE	2016	0	0.00	0.00
BRE	2017	0	0.00	0.00
BRE	2018	0	0.00	0.00
BRE	2019	0	0.00	0.00
BRO	2013	0	0.00	0.00
BRO	2014	0	0.00	0.00
BRO	2015	0	0.00	0.00
BRO	2016	1	1.00	0.16
BRO	2017	1	3.00	0.14
BRO	2018	1	3.00	0.18
BRO	2019	1	4.00	0.16
BRW	2013	0	0.00	0.00
BRW	2014	0	0.00	0.00
BRW	2015	0	0.00	0.00
BRW	2016	0	0.00	0.00
BRW	2017	0	0.00	0.00
BRW	2018	0	0.00	0.00
BRW	2019	0	0.00	0.00
CSE	2013	0	0.00	0.00
CSE	2014	1	0.00	0.07
CSE	2015	2	0.00	0.14
CSE	2016	3	0.33	0.21
CSE	2017	1	0.00	0.07
CSE	2018	3	0.67	0.18
CSE	2019	6	1.17	0.37
CSW	2013	3	0.00	0.27
CSW	2014	3	2.00	0.32
CSW	2015	5	0.20	0.54
CSW	2016	3	1.67	0.32

CSW	2017	5	0.80	0.55
CSW	2018	4	1.75	0.49
CSW	2019	3	0.67	0.38
DMP	2013	10	1.20	0.30
DMP	2014	8	1.25	0.39
DMP	2015	7	0.14	0.32
DMP	2016	4	0.75	0.25
DMP	2017	4	1.75	0.24
DMP	2018	9	3.00	0.43
DMP	2019	15	1.20	0.48
FIL	2013	1	2.00	0.04
FIL	2014	0	0.00	0.00
FIL	2015	0	0.00	0.00
FIL	2016	1	0.00	0.04
FIL	2017	1	0.00	0.05
FIL	2018	1	0.00	0.05
FIL	2019	4	1.25	0.20
FLD3	2013	0	0.00	0.00
FLD3	2014	0	0.00	0.00
FLD3	2015	0	0.00	0.00
FLD3	2016	1	0.00	0.06
FLD3	2017	1	3.00	0.07
FLD3	2018	1	2.00	0.08
FLD3	2019	2	1.50	0.14
FLD4	2013	0	0.00	0.00
FLD4	2014	0	0.00	0.00
FLD4	2015	1	0.00	0.04
FLD4	2016	0	0.00	0.00
FLD4	2017	1	1.00	0.06
FLD4	2018	2	1.00	0.11
FLD4	2019	2	2.50	0.12
FLD5	2013	0	0.00	0.00
FLD5	2014	0	0.00	0.00
FLD5	2015	0	0.00	0.00
FLD5	2016	0	0.00	0.00
FLD5	2017	0	0.00	0.00
FLD5	2018	1	0.00	0.04
FLD5	2019	0	0.00	0.00
NBE	2013	1	3.00	0.05
NBE	2014	3	1.33	0.19
NBE	2015	2	0.50	0.09
NBE	2016	1	3.00	0.07
NBE	2017	2	1.50	0.17
NBE	2018	2	2.50	0.20
NBE	2019	2	1.00	0.25
NBW	2013	0	0.00	0.00

NBW	2014	1	0.00	0.09
NBW	2015	0	0.00	0.00
NBW	2016	3	2.33	0.28
NBW	2017	6	1.00	0.52
NBW	2018	4	1.50	0.42
NBW	2019	6	1.17	0.77
NMA	2013	0	0.00	0.00
NMA	2014	1	1.00	0.17
NMA	2015	4	0.00	0.27
NMA	2016	1	4.00	0.08
NMA	2017	1	3.00	0.08
NMA	2018	1	3.00	0.10
NMA	2019	0	0.00	0.00
OIE	2013	4	1.00	0.09
OIE	2014	4	0.75	0.14
OIE	2015	2	2.50	0.07
OIE	2016	5	2.40	0.20
OIE	2017	9	1.67	0.41
OIE	2018	8	2.88	0.48
OIE	2019	12	1.33	0.55
OIW	2013	0	0.00	0.00
OIW	2014	0	0.00	0.00
OIW	2015	0	0.00	0.00
OIW	2016	1	3.00	0.22
OIW	2017	0	0.00	0.00
OIW	2018	0	0.00	0.00
OIW	2019	1	0.00	1.16
OIWR	2013	2	0.00	0.07
OIWR	2014	1	2.00	0.05
OIWR	2015	0	0.00	0.00
OIWR	2016	0	0.00	0.00
OIWR	2017	0	0.00	0.00
OIWR	2018	1	3.00	0.05
OIWR	2019	6	3.17	0.30
PSE	2013	0	0.00	0.00
PSE	2014	0	0.00	0.00
PSE	2015	0	0.00	0.00
PSE	2016	0	0.00	0.00
PSE	2017	0	0.00	0.00
PSE	2018	1	3.00	0.15
PSE	2019	0	0.00	0.00
PSO	2013	2	0.00	0.07
PSO	2014	2	2.00	0.10
PSO	2015	4	0.75	0.19
PSO	2016	4	2.50	0.20
PSO	2017	4	1.50	0.21

PSO	2018	6	1.83	0.44
PSO	2019	5	2.20	0.37
SPTE	2013	3	0.00	0.06
SPTE	2014	1	3.00	0.03
SPTE	2015	2	0.00	0.04
SPTE	2016	2	2.50	0.04
SPTE	2017	5	2.20	0.09
SPTE	2018	8	1.38	0.16
SPTE	2019	14	1.29	0.31
SPTW	2013	1	0.00	0.05
SPTW	2014	0	0.00	0.00
SPTW	2015	1	0.00	0.05
SPTW	2016	2	1.00	0.13
SPTW	2017	2	2.50	0.12
SPTW	2018	3	2.00	0.18
SPTW	2019	4	0.75	0.21
OSSSTP	2013	2	0.00	0.04
OSSSTP	2014	2	0.00	0.03
OSSSTP	2015	2	1.00	0.03
OSSSTP	2016	4	1.75	0.07
OSSSTP	2017	2	1.50	0.04
OSSSTP	2018	2	1.50	0.03
OSSSTP	2019	3	1.33	0.06

Appendix B. Vegetation Report for 2013–2019 in Overwash and Restoration Areas at Smith Point County Park, Fire Island, NY

Introduction

The U.S. Fish and Wildlife Service's (USFWS) Biological Opinion (BO) for the U.S. Army Corps of Engineers' (USACE) Fire Island to Moriches Inlet Stabilization Project (FIMI) says, in part:

> "The three overwash areas, the Great Gun restoration site, and the New Made Dredge restoration site will be managed to inhibit vegetation growth from impairing the quality of these available habitats. The Corps will build these areas to specifications agreed to by the Service and the County will maintain the vegetation per the specifications. This includes devegetating the area, as necessary, to allow no more than 30 percent vegetation cover at any one time during the 10-year life of the project."

This report summarizes the estimation of vegetation cover in the named overwash and restoration areas for 2013 and 2015–2019. This report seeks only to address the estimated cover. The determination of the extent to which existing vegetation is "impairing the quality of these available habitats" will require additional analyses.

At Pattersquash and Narrow Bay (West) Breach, Hurricane Sandy caused sand to overwash the full width of the island. Narrow Bay East Management Area and Old Inlet East overwash both occur in areas where intertidal marsh lies on the bayside of the island. Hurricane Sandy caused sand to overwash the dune and interior of the island but the furthest extent of overwash deposited sand on top of the marsh vegetation rather than extending to bayside water. Vegetation removal at Great Gun Restoration Area and dredging operations at New Made Dredge Restoration Area had not been completed at the time the April 2013 imagery was obtained. The New Made Restoration Area was still an impoundment for dredge spoil in 2013.

Methods

We used multispectral imagery to determine vegetation cover in 2013, and 2015–2019. The imagery for each year is in New York State Plane coordinate system, Long Island Zone, NAD 83, US Survey Feet, 6-inch resolution, 4-bands. The imagery and elevation data (digital elevation model (DEM) and Light Detection and Ranging (LIDAR)) for dates between 2010 and 2019 were obtained from various sources listed in Table 1. We also used Google Earth imagery from May 11, 2016 to help assess the interim condition of part of the New Made Dredge site after vegetation was mechanically removed in the spring of 2016. The Geographical Positioning System (GPS) coordinates of this area were obtained by walking the disturbed area in May 2016 with a handheld Garmin GPSMAP 78.

The boundaries of the overwash areas and the New Made Dredge Restoration Area were taken from images in the Final Environmental Assessment for the FIMI project, attachment A, plans for construction http://www.nan.usace.army.mil/Portals/37/docs/civilworks/projects/ny/coast/fimp/FIMI_Docs/EA/Attachment_A_HSLRR_FIMI_Layouts_Final_.pdf. The boundaries of the Great Gun Restoration Area were hand-digitized from spring 2015 imagery, taken just weeks after completion of this restoration area. We obtained GPS locations of the string fence at Great Gun, by walking the fence line with a handheld Garmin GPSMAP 78. The Great Gun area encompasses the area between the string fencing and the foot of the dune at the north end of the site, determined by LIDAR imagery. To illustrate "starting conditions," we projected these boundaries on imagery obtained by New York State in the spring of 2013, after Hurricane Sandy and before the beginning of the plant growing season. The bayside extent of two overwashes, Narrow Bay Breach Management Area and Pattersquash Management Area, have experienced various degrees of inundation since 2013. Rather than

adjusting the bayside boundary each year, we used the "original" boundaries from 2013 for consistency in each year's imagery classification.

As a reference site, we created a monitoring area within the overwash just east of Old Inlet. There was no dune built in this area. The overwash extends into marsh vegetation and does not connect with the bay. It provides an indication of how rapidly vegetation may return at a site that does not extend across the island, similar to Narrow Bay East. The bayside boundary for the Old Inlet East overwash was determined by walking the spring high tide line in 2013. The oceanward boundary was estimated by determining the average distance from the oceanward boundary to the ocean spring high tide line for the USACE maintenance areas and measuring that distance from the ocean spring high tide mapped at OIE. For boundaries of the USACE and restoration areas, see Figures 1–3 (Pattersquash), 4–6 (Narrow Bay Breach), 7–9 (Narrow Bay East), 10–12 (Old Inlet East), 15–16 (Great Gun), and 17–18 (New Made).

Vegetation cover was determined using the 4-band, multispectral imagery obtained in the spring of 2013 and the fall of 2015–2019. Raster images were classified in ESRI ArcGIS 10.4.1/10.5/10.6 as sand or vegetation by performing interactive supervised classification. Using 4-band imagery, including the infrared band, training samples for the supervised classification were selected to differentiate vegetation from sand. We used visual inspection and ground truthing to ensure that dark sand was not inadvertently classified as vegetation.

The areas we evaluated can be divided into overwashes created by Hurricane Sandy and restoration areas created by the USACE. The overwashes can be further subdivided into overwashes which extended across the island (Pattersquash and Narrow Bay (West) Breach Management Areas) and those that overwashed the dunes but did not deposit sand all the way to bayside water (Narrow Bay East Management Area and Old Inlet East overwash). The vegetation regrowth in the overwash areas was compared to the areas of habitat creation at New Made Dredge and Great Gun Restoration Areas.

Results

Management Areas

In Spring of 2013, sand initially dominated each of the overwash areas as the percent of cells classed as vegetation ranged from 2% to 6% for Pattersquash (Table 2) and Narrow Bay West (Table 3), Narrow Bay East (Table 4) and Old Inlet East (Table 5). In subsequent years, substantial regrowth of vegetation occurred as expected.

At Pattersquash overwash, the percentage of vegetative cover increased from 3% to 23%, 46%, 65%, 61%, and 75% in 2013, 2015, 2016, 2017, 2018, and 2019, respectively (Table 2 and Figures 1.1–1.12). Elevation changes resulting from Hurricane Sandy were determined by comparing bare-earth DEM data measured immediately after the storm (November 2012) with the most recent pre-storm DEM which was 2010 (Figure 2). The elevation changes are overlain on imagery showing the extent of vegetation in September 2019. In Figure 2, it appears that areas where greater than 3 feet of sand was eroded or deposited between 2010–2012 remained relatively free of vegetation in 2019. Figure 3 shows the distribution of vegetation as classified on September 2019 imagery overlying imagery from April 2010. Even though significant erosion and deposition of sand has an impact on regrowth of vegetation, some vegetation-free areas in 2019 appear to result from the persistence of vegetation-free areas from 2010.

In the Narrow Bay (West) Breach Management Area, the percentage of vegetative cover increased from 2% to 35%, 59%, 64%, 75%, and 75% in 2013, 2015, 2016, 2017, 2018, and 2019, respectively (Table 3.1 and Figures 4.1-4.12). The variation in the rate of regrowth between the breached versus non-breached areas is demonstrated in Table 3.2. Vegetation in the breached area was 4%, 12%, 32%, 36%, 56%, and 52% in 2013, 2015, 2016,

2017, 2018, and 2019, respectively. For comparison, the non-breached area west of the Narrow Bay breach experienced more rapid regrowth on the order of 6%, 50%, 76%, 80%, 87%, and 87% for 2013, 2015, 2016, 2017, 2018, and 2019, respectively. The more rapid rates of immediate regrowth in the non-breached area are similar to those at Narrow Bay East and the Old Inlet East overwash.

Figure 5.1 shows the change in elevation that occurred in the Narrow Bay Breach Management Area during Hurricane Sandy. More than 3 feet of erosion occurred, especially in the breach. In addition, more than 3 feet of sand was deposited along the bayside shoreline. Between 2012 and 2014, the breach was filled with 3 to more than 6 feet of sand (Figure 5.2). By 2014, compaction and erosion resulted in subsidence of the newly deposited sand on the bayside shoreline. It again appears that in 2019, areas with substantial removal or addition of sand remain more sparsely vegetated. As with Pattersquash overwash, significant erosion and deposition of sand at Narrow Bay Breach has had a major impact on regrowth of vegetation, however, some vegetation-free areas in 2019 appear to result from the persistence of vegetation-free areas from 2010 (Figure 6).

Regrowth occurred quickly for both Narrow Bay East Management Area and Old Inlet East overwash. At Narrow Bay East, vegetation cover increased from 6% in 2013 to 33%, 61%, 69%, 78%, 80% in 2015, 2016, 2017, 2018, and 2019, respectively (Table 4, Figures 7.1–7.12). Elevation changes which occurred as a result of Hurricane Sandy are shown in Figure 8 for Narrow Bay East Management Area. Figure 9 shows the distribution of vegetation in 2019 over imagery for 2010. As evident on the 2010 imagery, significant erosion of the dune and trees immediately shoreward of the Narrow Bay East Management Area occurred during Hurricane Sandy. The distribution of vegetation-free areas in 2019 seems to be more influenced by the impact of erosion and deposition within the overwash area than the location of vegetation-free areas in 2010.

At Old Inlet East, vegetation increased from 4% in 2013 to 53%, 70%, 75%, 77%, and 88% in 2015, 2016, 2017, 2018, and 2019, respectively (Table 5, Figures 10.1–10.12). Elevation changes which occurred because of Hurricane Sandy are shown in Figure 11 for Old Inlet East. Areas which experienced greater than 3 feet of erosion or deposition of sand to remain relatively vegetation free on the September 2019 imagery. In addition, as shown in Figure 11, many of the vegetation-free areas from 2010 persisted in 2019.

A comparison of the rate of revegetation in Pattersquash and Narrow Bay Management Areas and Old Inlet East overwash for 2013–2019 is shown in Figure 13. Initially, revegetation occurred most quickly at Old Inlet East and Narrow Bay overwashes with each of these areas exceeding 30% cover by 2015. Regrowth of vegetation increased slightly more slowly at Pattersquash. By 2016, three years after Hurricane Sandy, the percent cover of vegetation had reached 60% for the Narrow Bay areas and Old Inlet East. Pattersquash vegetation cover exceeded 60% in 2017. After exceeding 60% cover, the rate of increase of revegetation slowed in all areas.

As noted above, regrowth of vegetation in the filled breach area at Narrow Bay occurred more slowly than the adjacent area (Table 3.2). The rate of regrowth for the breached and non-breached area is shown in Figure 14. This graph shows that revegetation occurred more slowly in the breach-fill area compared to the overwash west of the breach. The non-breached area experienced rapid revegetation with coverage exceeding 75% within three years of the hurricane. The rate of regrowth in the breached area has progressed at a much slower rate with coverage leveling out at just over 50% for the past two years.

Restoration Areas

Prior to creation of the Great Gun Restoration Area, the percent of vegetation in the defined area was 50% (Table 6, Figures 15.1–15.2). Excavation and grading of the area resulted in removal of all vegetation with only 1% of the area classified as vegetation in 2015 (Figures 15.3–15.4). Revegetation occurred more slowly at Great Gun than at the overwashes. In 2016, 2017, 2018, and 2019, vegetation increased from 13% to 25%, 23%, and

26%, respectively (Table 6, Figures 15.5–15.12). In 2019, Great Gun was the only area that remained below the 30% threshold of vegetation cover put forth in the BO. Figure 16 shows the elevation change that occurred at Great Gun between 2012 and 2015. Excavation and grading resulted in removal of greater than 3 feet of sand over most of the area. Some areas were lowered by more than 9 feet. The extensive removal of plant root and seed base is likely the reason for the slower rate of revegetation.

New Made Restoration Area was classified as 18% vegetation, 38% dry sand, 29% water, and 15% wet dredge material in 2013. (Table 7, Figures 17.1–17.2). Figure 18.1 shows the modification (mostly lowering) of the land surface which occurred between 2010 and 2012. Between 2012 and 2015, a substantial portion of the New Made Dredge Restoration Area had been filled with 3 feet to more than 6 feet of sand (Figure 18.2). These areas of fill remain largely vegetation free in 2019.

A comparison of the rate of revegetation for New Made and Great Gun restoration areas is presented in Figure 19. Nearly one third of the eastern part of New Made Restoration was vegetated and undisturbed when the western part of the area was filled and graded in 2015. To demonstrate the difference in the influence of extensive filling and grading, revegetation is presented separately for New Made East and West. Regrowth in New Made West and Great Gun remain significantly lower than that which occurred in the overwash management areas of Pattersquash and Narrow Bay.

In contrast to the effect of breaches washing away vegetative material, surficial disturbance where vegetation is present may actually facilitate the spread of vegetation. Earthmoving equipment was used at New Made Dredge Spoil Restoration Area in the spring of 2016 to scrape vegetation in that area. Imagery from February 2016 (Figure 20.1) shows the area just a few months before vegetation removal. Google Earth imagery from May 11, 2016 (Figure 20.2) shows the same area immediately after earthmoving equipment was used to scrap and grade the surface. In the September 2016 imagery, near the end of the 2016 growing season, it is evident that vegetation regrew in nearly the exact footprint of surface disturbance (Figure 20.3). This footprint continues to be the area of densest vegetative regrowth in the western portion of New Made Restoration Area.

Discussion

Many factors affect vegetation regrowth in these areas including the distribution and type of vegetation prior to overwash or restoration and the depth of sand eroded or deposited. Many plants are adapted to survive burial and readily grow upward through the sand. Other plants spread easily over the ground surface, especially if the surface has been stripped of competing vegetation. Both of these characteristics make them particularly suited to barrier island habitats. In contrast, some trees and shrubs may be more easily damaged when their roots are exposed or experience increased overburden.

Vegetation regrowth in the breach at Narrow Bay demonstrates the effect of whether or not rhizome/root or seed material is removed or destroyed. It is evident that regrowth in the Narrow Bay Breach Management Area is less dense in the area which was fully breached than in the adjacent overwashed area. Flow of water through the breach was effective at removing vegetation at the roots, as opposed to just covering vegetation with variable thicknesses of sand.

Some of the factors affecting persistence of vegetation-free areas in overwashes include presence of vegetation-free areas prior to the Hurricane Sandy overwash and the depth of erosion or deposition of sand which occurred during the storm. Areas subjected to greater than 2-foot loss or gain of elevation have a greater likelihood of removing or burying pre-Sandy vegetation. The adaptability of plants to survive exposure or burial varies with species.

A total breach of the island is more likely to occur in areas that are low-lying and narrow, especially areas with no bayside intertidal marsh. Island breaches result in connection to bayside water which, in turn, can provide a connection to bayside wet sand providing foraging habitat. Moreover, removal of vegetation roots and seed base in the breached area slows regrowth of vegetation. Overwashes in areas with bayside intertidal marsh are unlikely to provide persistent sandy habitat.

With the exception of the Great Gun Restoration area, vegetation growth has exceeded the 30 percent vegetation cover trigger specified in the BO. The western part of New Made, which underwent extensive dredge and fill, also experienced slower revegetation than the non-breached overwashes. Moreover, some of the imagery classified as sand in the overwashes occurs in isolated or highly fragmented patches surrounded by vegetation. These fragmented patches are unlikely to be used as nest sites, as plover nest camouflage depends on blending into a large expanse of sand. With vegetation covering 60–80% or more of most management and restoration areas, it is less likely that these vegetated areas will be used as nest sites by plovers. Data collected on how piping plovers utilize these areas during the coming breeding seasons may be used to develop a method for estimating carrying capacity in the face of a fragmented sand/vegetation landscape.

Future Management Implications

Open, sandy habitats are critical for the survival of many shorebird species. While these habitats have always occurred naturally on barrier island systems, creation of similar areas may be achieved through excavation and/or fill. Using a combination of naturally-occurring breaches and overwashes along with manmade disturbance may be the most effective way to maintain critical habitat.

Breaches typically occur in areas with narrow and/or low-lying beachfront. Allowing narrow and/or low-lying beachfront to occur in front of narrow sections of the island further enhances the likelihood for natural processes to remove vegetation and create a beneficial connection to bayside, wet-sand, foraging habitat. Bayside foraging habitat cannot be achieved in areas with extensive intertidal marshes. While excavation and fill can be effective at providing sandy habitat, locating these restoration projects in naturally-occurring breach sites and bayside overwashes may provide the greatest likelihood that suitable habitat would persist for more than two to three years. Evidence suggests that removing or adding sand to a depth of at least three feet is necessary to slow the rate of revegetation.

The BO calls for “devegetating the area, as necessary, to allow *no more* than 30 percent vegetation cover at any one time during the 10-year life of the project” (emphasis ours). It does not preclude devegetating before the 30% threshold is reached. By the time 30% cover (as measured herein) is reached, an extensive network of rhizomes and stems exist, such that vegetation removal is difficult and expensive. Mechanical removal without the use of herbicide is likely to promote the spread of vegetation by distributing rhizomes as apparently occurred at the New Made restoration area in 2016. It would likely be more cost effective, and of greater benefit to piping plovers, if, once a management site has been devegetated, it is subject to treatments required to keep vegetation well below the 30% level. This might require annual treatments with herbicide until root and rhizome systems are dead.

A good approach would be to spray with Glyphosate (Roundup ©) in early fall, immediately after plover season while plants still are growing. It is crucial to do this while plants are still growing so that the herbicide will have an effect. After plants have died back, they can be removed mechanically, say in December. In the last two weeks of March, Imazapyr (Groundclear ©), or a mix of Imazapyr and Glyphosate, can be applied. Imazapyr works as a pre-emergent, so it will reduce spring and summer emergence. The process could be repeated annually, as needed.

Table 1. Date and specifications of imagery and elevation data.

Date collected	Type	Resolution	Projection[1]	Source[2]
Aug, 2010	Lidar	6 feet	NAD83/NAVD88	NOAA_OCM
Nov, 2012	Lidar 4-band	1 meter	NAD83/NAVD88	NOAA_OCM
April 3, 2013	Imagery	6 inch	NAD83SPNY	NYS GIS
April 1, 2014	Lidar	6 feet	NAD83SPNY/D_NAD_1983	NOAA_OCM
April 16, 2015	Lidar 4-band	1 meter	NAD83SPNY/D_NAD_1983	Axis
Oct 21, 2015	Imagery	6 inch	NAD83SPNY	Axis
April 25, 2016	Lidar 4-band	1 meter	NAD83(2011)SPNY/D_NAD_1983_2011	Axis
Sept 13, 2016	Imagery	6 inch	NAD83(2011)SPNY/D_NAD_1983_2011	Axis
April 11, 2017	Lidar 4-band	1 foot	NAD83(2011)SPNY/D_NAD_1983_2011	Axis
Sept 23, 2017	Imagery	6 inch	NAD83(2011)SPNY/D_NAD_1983_2011	Axis
April 18, 2018	Lidar 4-band	1 foot	NAD83(2011)SPNY/NAVD88	Axis
Sept 30, 2018	Imagery	6 inch	NAD83(2011)SPNY/D_NAD_1983_2011	Axis
May 18, 2019	Lidar 4-band	1 foot	NAD83(2011)SPNY/NAVD88	Axis
Sept 17, 2019	Imagery	6 inch	NAD83(2011)SPNY/D_NAD_1983_2011	Axis

[1]**Projection:**

NAD83 = North American Datum of 1983
D_NAV83 = North American Datum of 1983 Vertical Datum
NAVD88 = North American Vertical Datum of 1988
NAD83SPNY =NAD_1983_StatePlane_New_York_Long_Island_FIPS_3104_Feet
NAD83(2011)SPNY =
NAD_1983_2011_StatePlane_New_York_Long_Isl_FIPS_3104_Ft_US

[2]**Source:**

NOAA = NOAA Office of Coastal Management (https://coast.noaa.gov)
NYSGIS = New York State GIS Clearinghouse (http://gis.ny.gov/gateway/)
Axis = Axis Geospatial, LLC, Easton, MD

Table 2. Area (ha) and percentage of cells classified in sand and vegetation cover for Pattersquash management/restoration area on Fire Island, New York, for 2013, 2015, 2016, 2017, 2018, and 2019

	2013	2015	2016	2017	2018	2019
Total Area (ha)	13.4	13.4	13.4	13.4	13.4	13.4
Sand						
Area (ha)	13.0	10.0	6.2	3.7	3.5	2.3
% of total Area	97%	75%	46%	28%	26%	17%
Veg						
Area (ha)	0.4	3.1	6.1	8.7	8.1	10.1
% of Total Area	3%	23%	46%	65%	61%	75%
Saturated Sand/Veg						
Area (ha)		0.3	1.1	0.6	1.8	0.2
% of Total Area		2%	8%	4%	13%	2%
Water						
Area (ha)				0.4	<0.1	0.8
% of Total Area				3%	0%	6%

Table 3.1. Area (ha) and percentage of cells classified in sand and vegetation cover for Narrow Bay Breach management/restoration area on Fire Island, New York, for 2013, 2015, 2016, 2017, 2018, and 2019

	2013	2015	2016	2017	2018	2019
Total Area (ha)	6.3	6.3	6.3	6.3	6.3	6.3
Sand						
Area (ha)	6.2	3.7	2.1	1.2	1.2	0.9
% of total Area	98%	59%	33%	19%	19%	15%
Veg						
Area (ha)	0.1	2.2	3.7	4.0	4.7	4.6
% of Total Area	2%	35%	59%	64%	75%	73%
Saturated Sand/Veg						
Area (ha)		0.4	0.3	0.7	0.4	0.4
% of Total Area		6%	5%	11%	6%	6%
Water						
Area (ha)			0.2	0.4		0.4
% of Total Area			3%	6%		6%

Table 3.2. Vegetation regrowth in breach versus non-breached areas. Area (ha) and percentage of cells classified in sand and vegetation cover for Narrow Bay Breach management/restoration area on Fire Island, New York, for 2013, 2015, 2016, 2017, 2018, and 2019

		Breach					West of Breach					
	2013	2015	2016	2017	2018	2019	2013	2015	2016	2017	2018	2019
Total Area (ha)	2.5	2.5	2.5	2.5	2.5	2.5	3.8	3.8	3.8	3.8	3.8	3.8
Sand												
Area (ha)	2.4	2.0	1.3	0.9	0.8	0.6	3.7	1.7	0.8	0.3	0.4	0.3
% of total Area	96%	80%	52%	36%	32%	24%	94%	45%	21%	8%	10%	8%
Veg												
Area (ha)	0.1	0.3	0.8	0.9	1.4	1.3	0.1	1.9	2.9	3.1	3.3	3.3
% of Total Area	4%	12%	32%	36%	56%	52%	6%	50%	76%	80%	87%	87%
Saturated Sand/Veg												
Area (ha)		0.2	0.3	0.4	0.3	0.3		0.2	<0.1	0.3	0.1	0.1
% of Total Area		8%	12%	16%	12%	12%		5%		8%	3%	2%
Water												
Area (ha)			0.1	0.3	<0.1	0.3			0.1	0.1	<0.1	0.1
% of Total Area			4%	12%		12%			3%	4%		3%

Table 4. Area (ha) and percentage of cells classified in sand and vegetation cover for Narrow Bay East management area on Fire Island, New York, for 2013, 2015, 2016, 2017, 2018, and 2019

	2013	2015	2016	2017	2018	2019
Total Area (ha)	10.5	10.5	10.5	10.5	10.5	10.5
Sand						
Area (ha)	9.9	7	3.8	2.7	1.9	2.1
% of total Area	94%	67%	36%	25%	18%	20%
Veg						
Area (ha)	0.6	3.5	6.4	7.2	8.2	8.4
% of Total Area	6%	33%	61%	69%	78%	80%
Saturated Sand/Veg						
Area (ha)			0.3	0.6	0.4	
% of Total Area			3%	6%	4%	0%

Table 5. Area (ha) and percentage of cells classified in sand and vegetation cover for Old Inlet East area on Fire Island, New York, for 2013, 2015, 2016, 2017, 2018, and 2019

	2013	2015	2016	2017	2018	2019
Total Area (ha)	26.5	26.5	26.5	26.5	26.5	26.5
Sand						
Area (ha)	25.1	12.1	6.7	5.2	5.7	2.8
% of total Area	95%	46%	25%	20%	22%	11%
Veg						
Area (ha)	1.1	14.1	18.6	20.0	20.5	23.4
% of Total Area	4%	53%	70%	75%	77%	88%
Saturated Sand						
Area (ha)	0.3	0.3	0.8	1.0	0.2	0.1
% of Total Area	1%	1%	3%	4%	1%	0%
Water						
Area (ha)			0.4	0.3	0.1	0.2
% of Total Area			2%	1%	0%	1%

Table 6. Area (ha) and percentage of cells classified in sand and vegetation cover for Great Gun restoration area on Fire Island, New York, for 2013, 2015, 2016, 2017, 2018, and 2019

Great Gun Restoration Area						
	2013	**2015**	**2016**	**2017**	**2018**	**2019**
Total Area (ha)	20.6	20.6	20.6	20.6	20.6	20.6
Sand						
Area (ha)	10.3	20.4	17.9	15.3	15.8	15.3
% of total Area	50%	99%	87%	74%	77%	74%
Veg						
Area (ha)	10.3	0.2	2.7	5.0	4.8	5.3
% of Total Area	50%	1%	13%	25%	23%	26%
Saturated Sand						
Area (ha)				0.3		

Table 7.1 Area (ha) and percentage of cells classified in sand and vegetation cover for New Made Dredge restoration area on Fire Island, New York, for 2013, 2015, 2016, 2017, 2018, and 2019

	2013	2015	2016	2017	2018	2019
Total Area (ha)	6.6	6.6	6.6	6.6	6.6	6.6
Sand						
Area (ha)	2.5	4.4	3.0	2.5	2.4	1.9
% of total Area	38%	67%	45%	38%	36%	29%
Veg						
Area (ha)	1.2	2.2	3.6	4.0	4.2	4.7
% of Total Area	18%	33%	55%	61%	64%	71%
Saturated Sand/Veg						
Area (ha)			0.1	0.1		
% of Total Area			1%	1%		
Wet Dredge						
Area (ha)	1.0					
% of Total Area	15%					
Water						
Area (ha)	1.9					
% of Total Area	29%					

Table 7.2. Area (ha) and percentage of cells classified as sand and vegetation cover for West (dredged) versus East (non-dredged) parts of New Made Dredge Restoration Area on Fire Island, New York, for 2013 and 2015–2019.

	New Made West						New Made East					
	2013	2015	2016	2017	2018	2019	2013	2015	2016	2017	2018	2019
Total Area (ha)	4.2	4.2	4.2	4.2	4.2	4.2	2.4	2.4	2.4	2.4	2.4	2.4
Sand												
Area (ha)	0.9	3.9	3.1	2.5	2.4	2.3	1.7	0.3	0.1	0.1	<0.1	<0.1
% of total Area	21%	93%	74%	60%	57%	55%	71%	12%	4%	4%	0%	0%
Veg												
Area (ha)	0.4	0.3	1.1	1.7	1.8	1.9	0.7	2.1	2.3	2.3	2.4	2.4
% of Total Area	10%	7%	26%	40%	43%	45%	29%	88%	96%	96%	100%	100%
Saturated Sand/Veg												
Area (ha)	1.0											
% of Total Area	24%											
Water												
Area (ha)	1.9											
% of Total Area	45%											



Figure 1.1. Pattersquash Management Area. Unclassified imagery from spring of 2013. Yellow polygon delineates designated maintenance area in 2013.

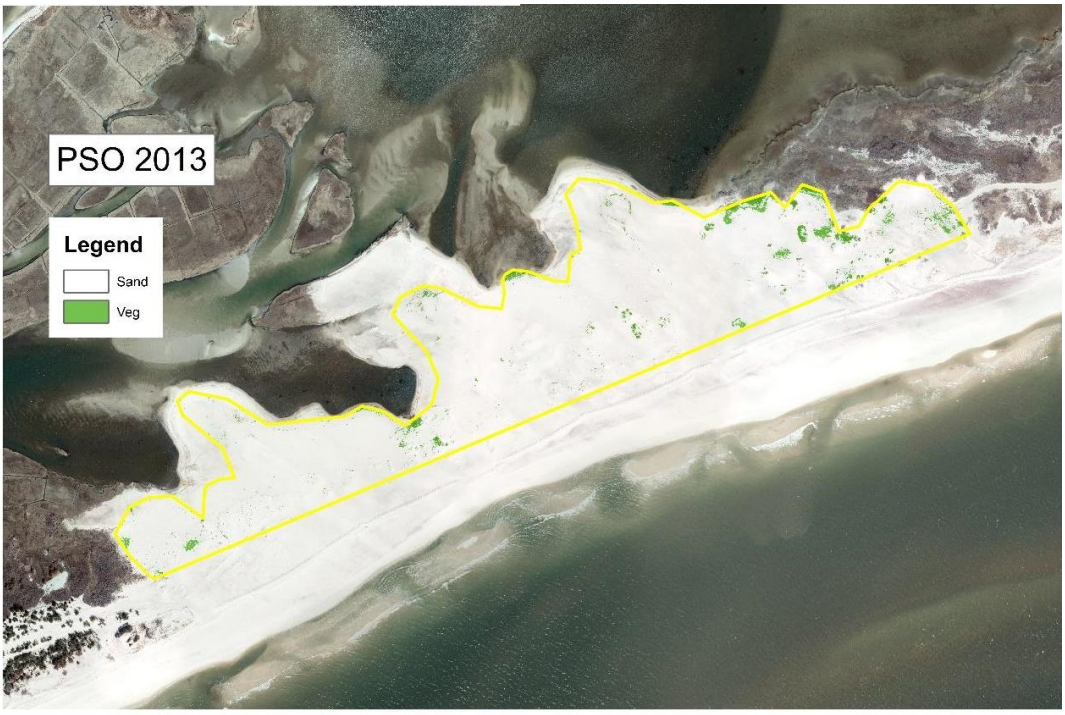


Figure 1.2. Pattersquash Management Area, classified imagery from spring 2013. Yellow polygon delineates designated maintenance area in 2013. Green is vegetation based on supervised classification; only 3% of pixels were classed as vegetation.



Figure 1.3. Pattersquash Management Area, unclassified imagery from October 2015. Yellow polygon delineates designated maintenance area in 2013.



Figure 1.4. Pattersquash Management Area. Yellow polygon delineates designated maintenance area in 2013. Imagery is from October 2015. Green is vegetation based on supervised classification; 23% of pixels were classed as vegetation.



Figure 1.5. Pattersquash Management Area, unclassified imagery from September 2016. Yellow polygon delineates designated maintenance area in 2013.



Figure 1.6. Pattersquash Management Area. Yellow polygon delineates designated maintenance area in 2013. Imagery is from September 2016. Green is vegetation based on supervised classification; 46% of pixels were classed as vegetation.



Figure 1.7. Pattersquash Management Area, unclassified imagery from September 2017. Yellow polygon delineates designated maintenance area in 2013.



Figure 1.8. Pattersquash Management Area. Yellow polygon delineates designated maintenance area in 2013. Imagery is from September 2017. Green is vegetation based on supervised classification; 65% of pixels were classed as vegetation.



Figure 1.9. Pattersquash Management Area, unclassified imagery from September 2018. Yellow polygon delineates designated maintenance area in 2013.



Figure 1.10. Pattersquash Management Area. Yellow polygon delineates designated maintenance area in 2013. Imagery is from September 2018. Green is vegetation based on supervised classification; 61% of pixels were classed as vegetation.



Figure 1.11. Pattersquash Management Area unclassified imagery from September 2019. Yellow polygon delineates designated maintenance area in 2013.



Figure 1.12. Pattersquash Management Area. Yellow polygon delineates designated maintenance area in 2013. Imagery is from September 2019. Green is vegetation based on supervised classification; 75% of pixels were classed as vegetation.



Figure 2. Pattersquash Management Area elevation change as a result of Hurricane Sandy. Unclassified imagery from September 2019. Yellow polygon delineates designated maintenance area in 2013.



Figure 3. Pattersquash Management Area. Green is vegetation based on supervised classification of September 2019 imagery. Underlying imagery is from April 2010. Yellow polygon delineates designated maintenance area in 2013.

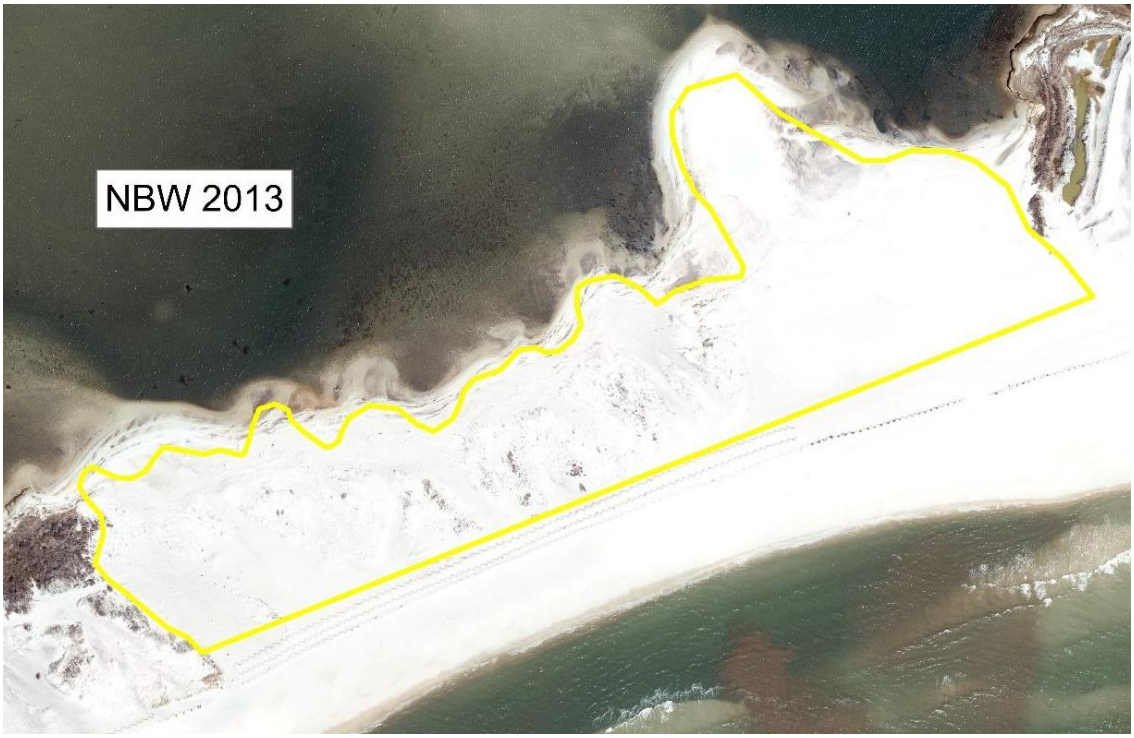


Figure 4.1. Narrow Bay Breach Management Area. Unclassified imagery from spring of 2013. Yellow polygon delineates designated maintenance area in 2013.

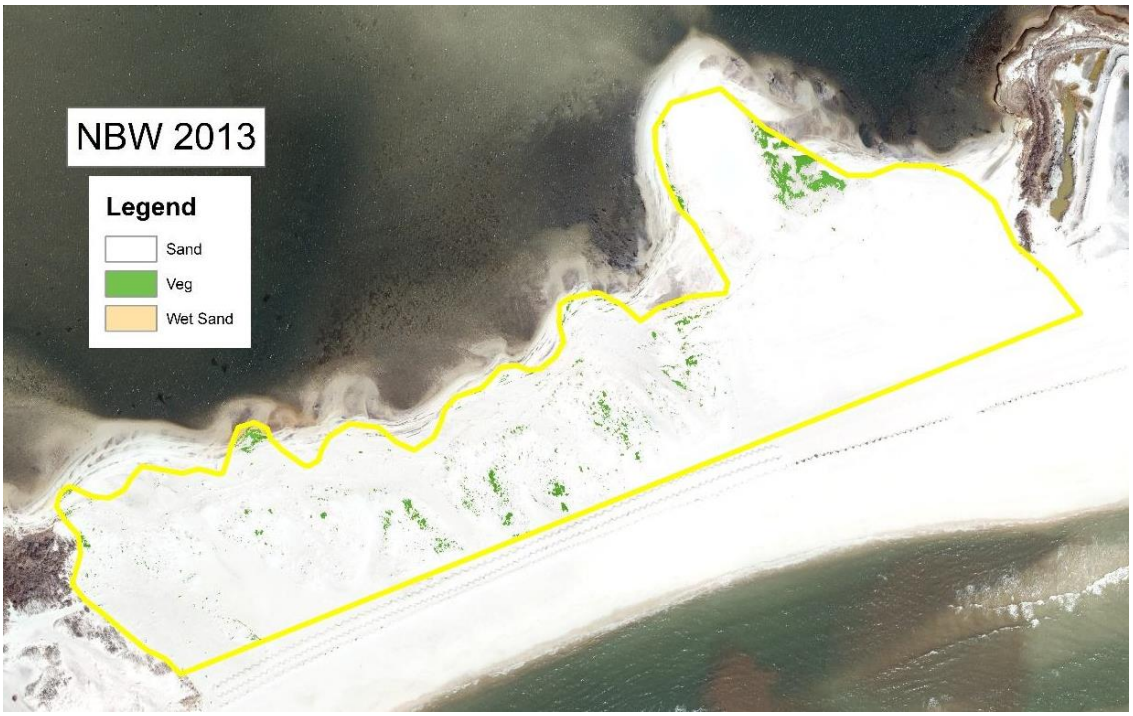


Figure 4.2. Narrow Bay Breach Management Area, classified imagery from spring 2013. Yellow polygon delineates designated maintenance area in 2013. Green is vegetation based on supervised classification; only 2% of pixels were classed as vegetation.



Figure 4.3. Narrow Bay Breach Management Area, unclassified imagery from October 2015. Yellow polygon delineates designated maintenance area in 2013.



Figure 4.4. Narrow Bay Breach Management Area. Green is vegetation based on supervised classification. Imagery is from October 2015; 35% of pixels were classed as vegetation. Yellow polygon delineates designated maintenance area in 2013.



Figure 4.5. Narrow Bay Breach Management Area, unclassified imagery from September 2016. Yellow polygon delineates designated maintenance area in 2013.



Figure 4.6. Narrow Bay Breach Management Area. Green is vegetation based on supervised classification. Imagery is from September 2016; 64% of pixels were classed as vegetation. Yellow polygon delineates designated maintenance area in 2013.



Figure 4.7. Narrow Bay Breach Management Area, unclassified imagery from September 2017. Yellow polygon delineates designated maintenance area in 2013.



Figure 4.8. Narrow Bay Breach Management Area. Imagery is from September 2017. Green is vegetation based on supervised classification; 64% of pixels were classed as vegetation. Yellow polygon delineates designated maintenance area in 2013.



Figure 4.9. Narrow Bay Breach Management Area, unclassified imagery from September 2018. Yellow polygon delineates designated maintenance area in 2013.



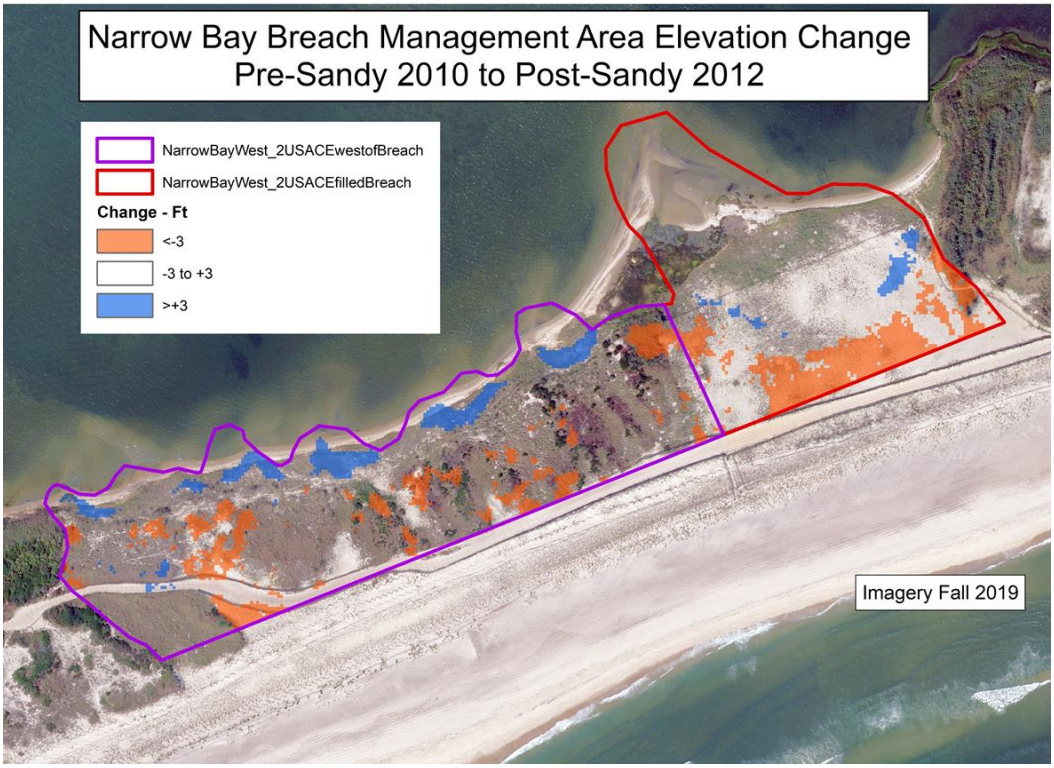
Figure 4.10. Narrow Bay Breach Management Area. Imagery is from September 2018. Green is vegetation based on supervised classification; 75% of pixels were classed as vegetation. Yellow polygon delineates designated maintenance area in 2013.



Figure 4.11. Narrow Bay Breach Management Area, unclassified imagery from September 2019. Yellow polygon delineates designated maintenance area in 2013.



Figure 4.12. Narrow Bay Breach Management Area. Imagery is from September 2019. Green is vegetation based on supervised classification; 75% of pixels were classed as vegetation. Yellow polygon delineates designated maintenance area in 2013.



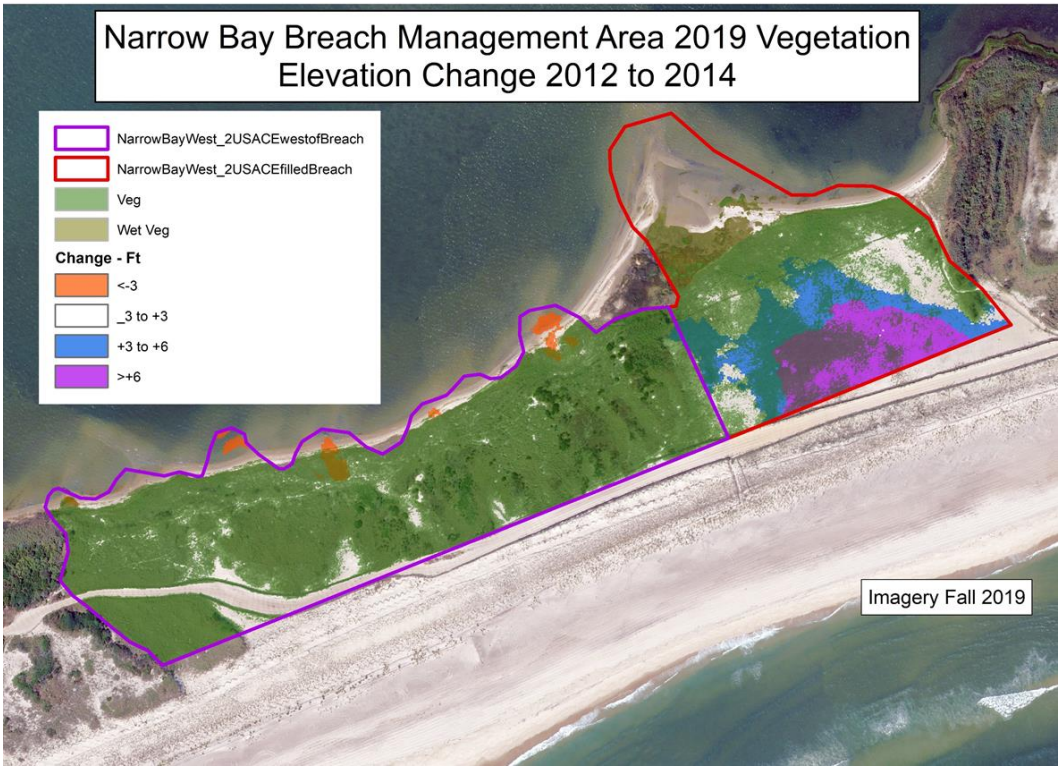


Figure 5.1/5.2. Narrow Bay Breach Management Area elevation changes 2010-2012 as a result of Hurricane Sandy and 2012-2014 showing breach fill. The gap in elevation data on the bayside shows where the 2012 lidar data was limited by the breach. Unclassified imagery from September 2019. Yellow polygon delineates designated maintenance area in 2013.



Figure 6. Narrow Bay Breach Management Area. Green is vegetation based on supervised classification from September 2019. Underlying imagery is from Spring 2010. Yellow polygon delineates designated maintenance area in 2013.



Figure 7.1. Narrow Bay East Management Area (yellow polygon delineates designated maintenance area in 2013). Unclassified imagery from spring of 2013.



Figure 7.2. Narrow Bay East Management Area (yellow polygon delineates designated maintenance area in 2013). Imagery is from spring 2013. Green is vegetation based on supervised classification; 6% of pixels were classified as vegetation.

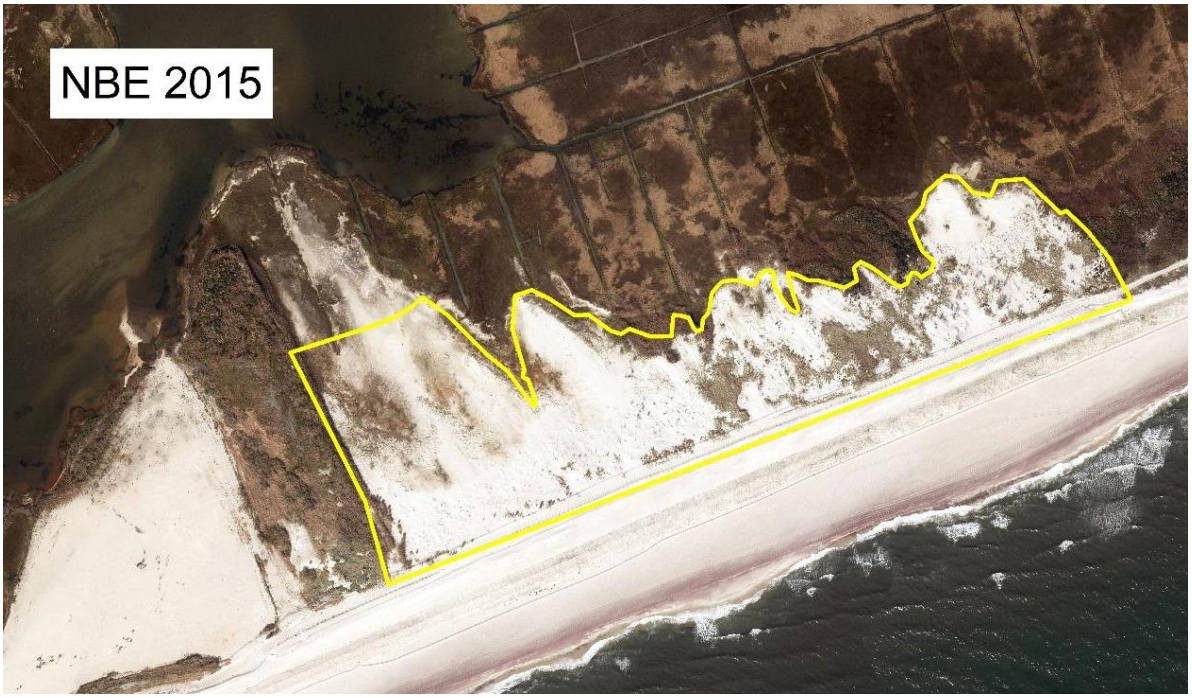


Figure 7.3. Narrow Bay East Management Area (yellow polygon delineates designated maintenance area in 2013). Unclassified imagery from October 2015.



Figure 7.4. Narrow Bay East Management Area (yellow polygon delineates designated maintenance area in 2013). Imagery is from October 2015. Green is vegetation based on supervised classification; 33% of pixels were classified as vegetation.



Figure 7.5. Narrow Bay East Management Area (yellow polygon delineates designated maintenance area in 2013). Unclassified imagery from September 2016.



Figure 7.6. Narrow Bay East Management Area (yellow polygon delineates designated maintenance area in 2013). Imagery is from September 2016. Green is vegetation based on supervised classification; 61% of pixels were classed as vegetation.



Figure 7.7. Narrow Bay East Management Area (yellow polygon delineates designated maintenance area in 2013). Unclassified imagery from September 2017.



Figure 7.8. Narrow Bay East Management Area (yellow polygon delineates designated maintenance area in 2013). Imagery is from September 2017. Green is vegetation based on supervised classification; 69% of pixels were classed as vegetation.



Figure 7.9. Narrow Bay East Management Area (yellow polygon delineates designated maintenance area in 2013). Unclassified imagery from September 2018.



Figure 7.10. Narrow Bay East Management Area (yellow polygon delineates designated maintenance area in 2013). Imagery is from September 2018. Green is vegetation based on supervised classification; 78% of pixels were classed as vegetation.



Figure 7.11. Narrow Bay East Management Area (yellow polygon delineates designated maintenance area in 2013). Unclassified imagery from September 2019.



Figure 7.12. Narrow Bay East Management Area (yellow polygon delineates designated maintenance area in 2013). Imagery is from September 2019. Green is vegetation based on supervised classification; 80% of pixels were classed as vegetation.

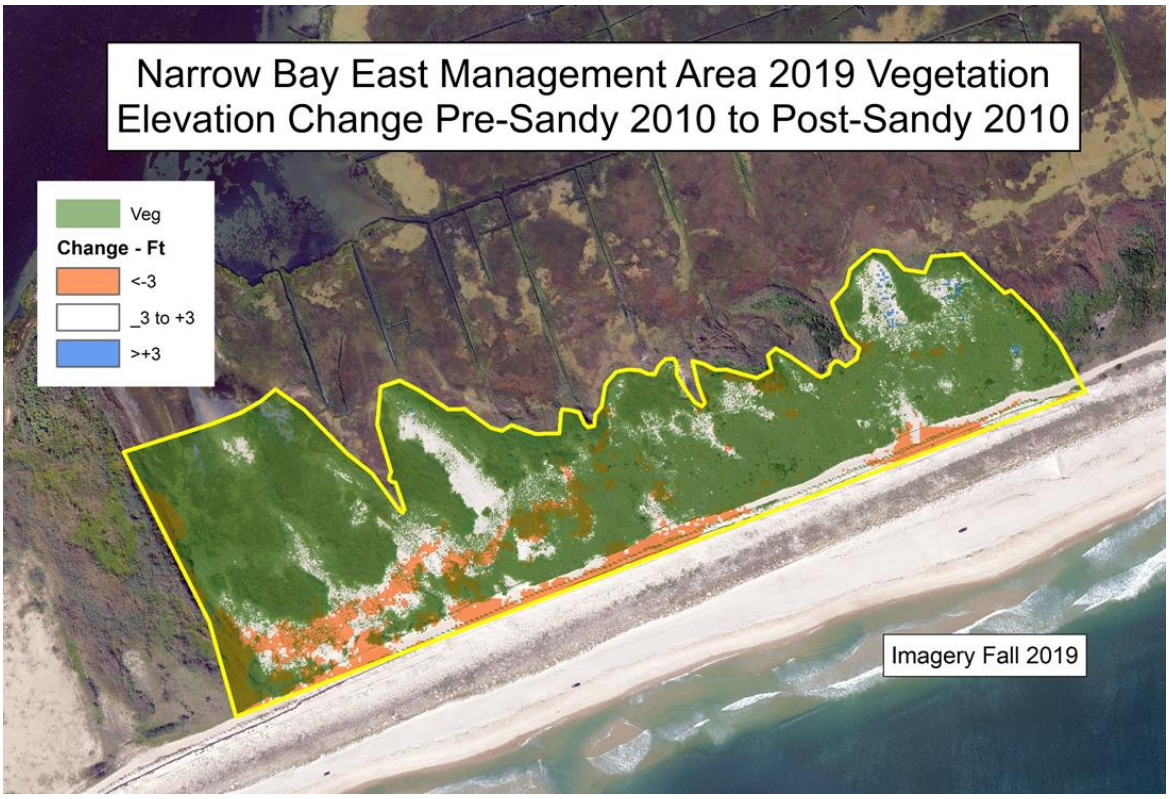


Figure 8. Narrow Bay East Management Area elevation changes 2010-2012 as a result of Hurricane Sandy. Unclassified imagery from September 2019. Yellow polygon delineates designated maintenance area in 2013.



Figure 9. Narrow Bay East Management Area. Green is vegetation based on supervised classification of September 2019 imagery. Underlying imagery is from April 2010. Yellow polygon delineates designated maintenance area in 2013.

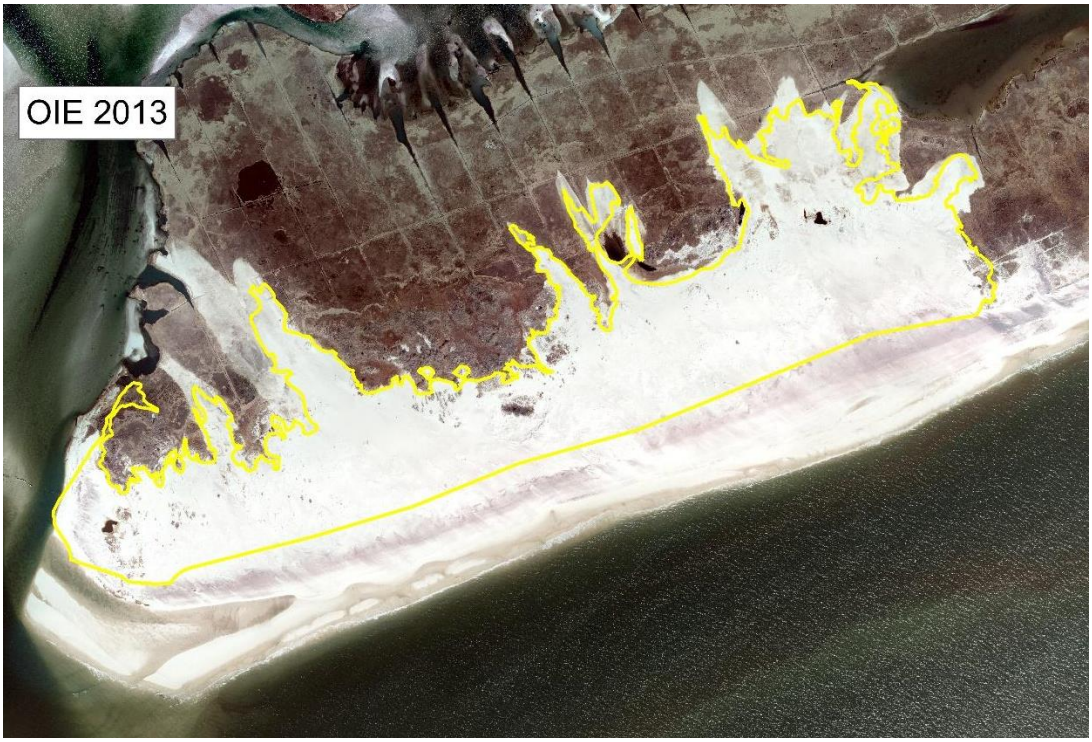


Figure 10.1. Old Inlet East Area. Unclassified imagery from spring of 2013. Yellow polygon delineates overwash area above spring high tide in 2013.

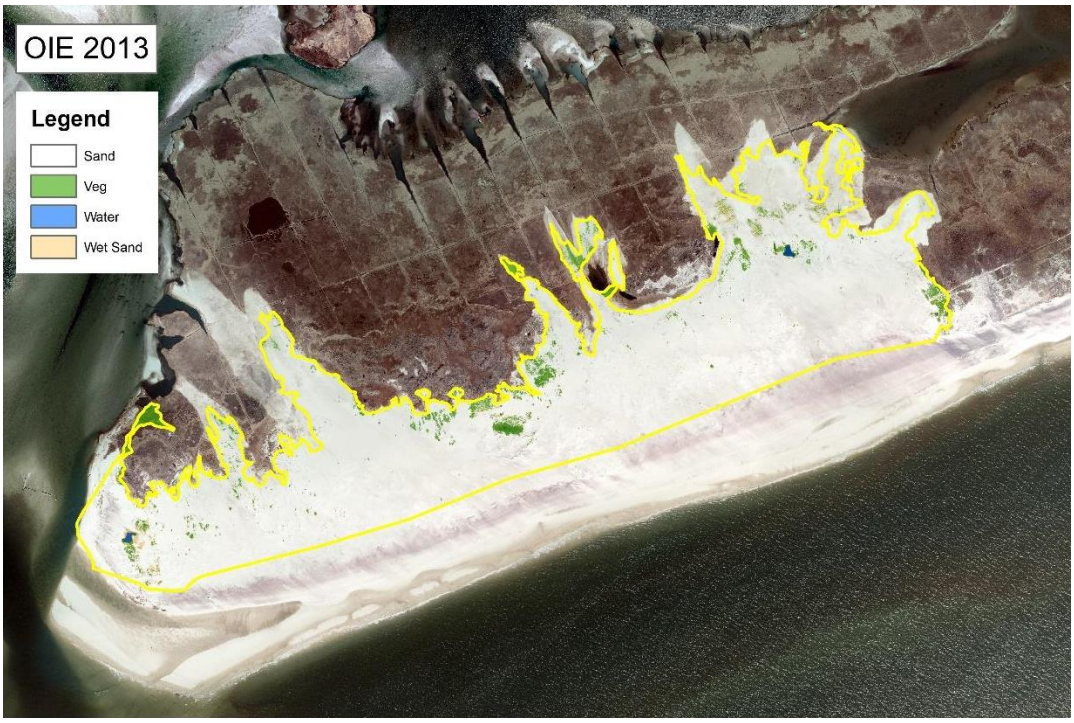


Figure 10.2. Old Inlet East Area, classified imagery from spring 2013. Yellow polygon delineates overwash area above spring high tide in 2013. Green is vegetation based on supervised classification; 5% of pixels were classed as vegetation.

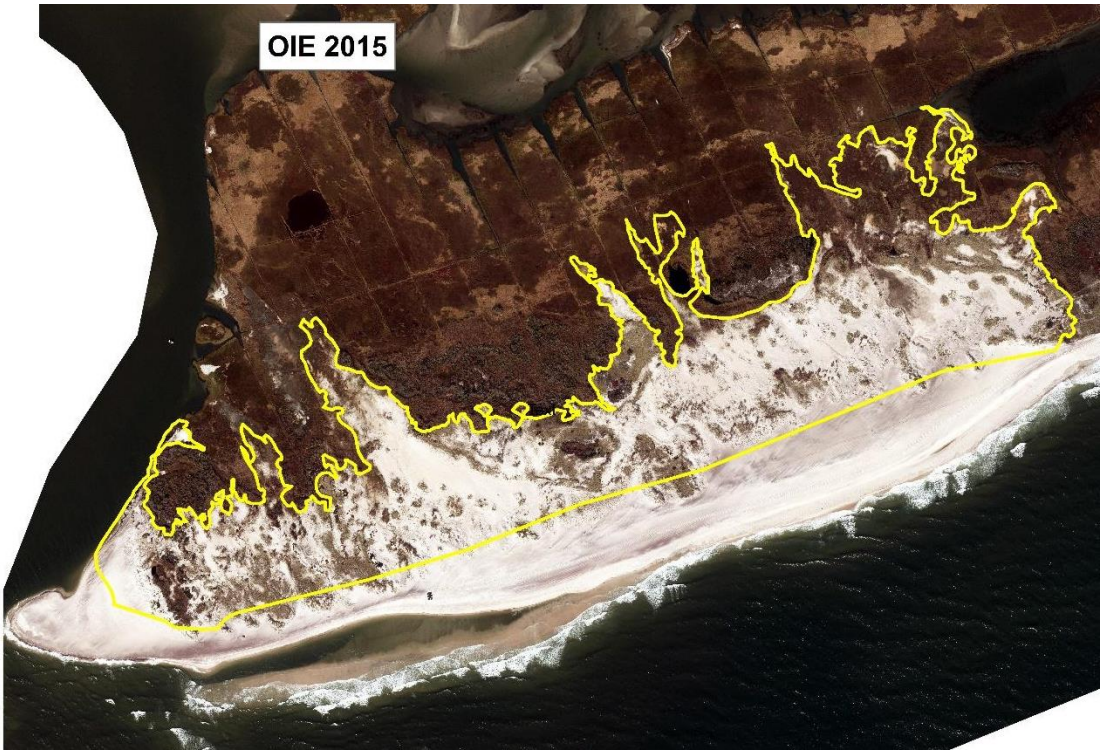


Figure 10.3. Old Inlet East Area, unclassified imagery from October 2015. Yellow polygon delineates overwash area above spring high tide in 2013.

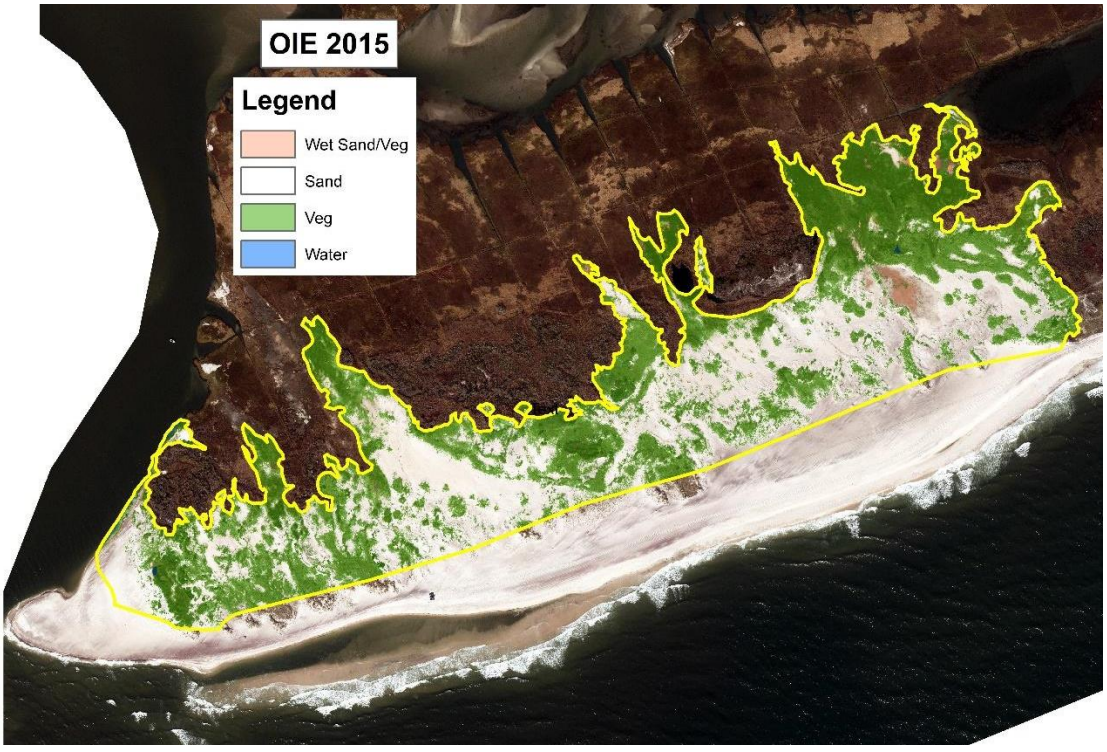


Figure 10.4. Old Inlet East Area. Green is vegetation based on supervised classification. Imagery is from October 2015; 53% of pixels were classed as vegetation. Yellow polygon delineates overwash area above spring high tide in 2013.

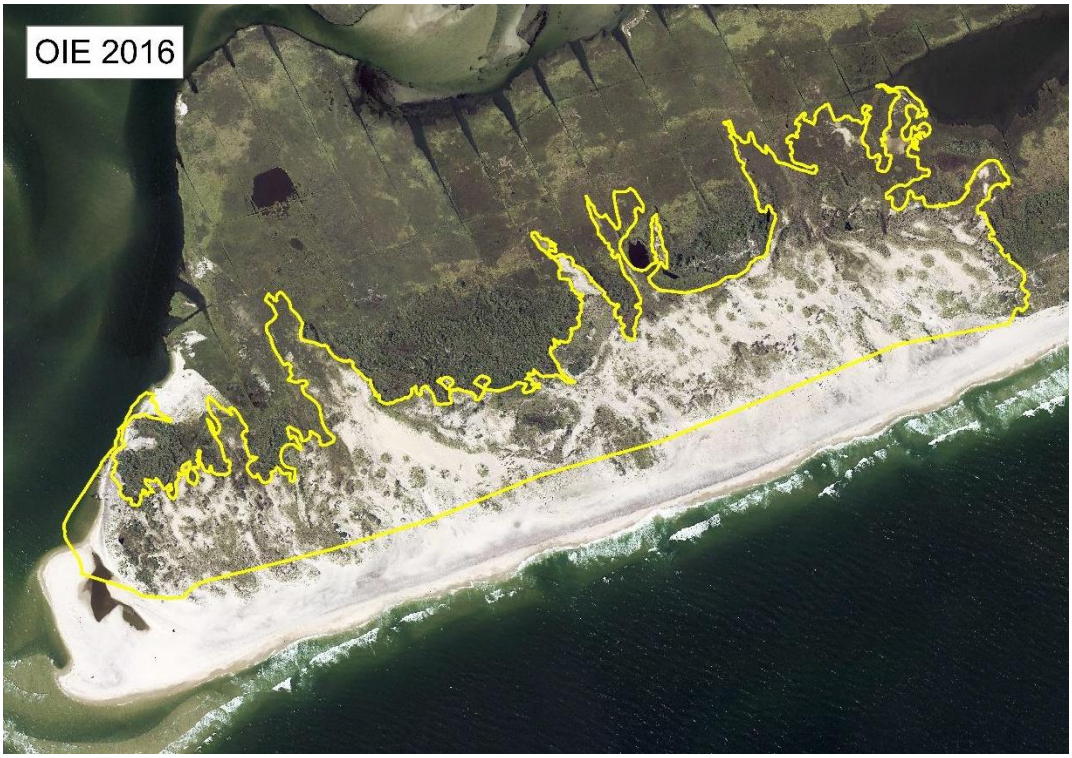


Figure 10.5. Old Inlet East Area, unclassified imagery from September 2016. Yellow polygon delineates overwash area above spring high tide in 2013.



Figure 10.6. Old Inlet East Area. Green is vegetation based on supervised classification. Imagery is from September 2016; 77% of pixels were classed as vegetation. Yellow polygon delineates overwash area above spring high tide in 2013.

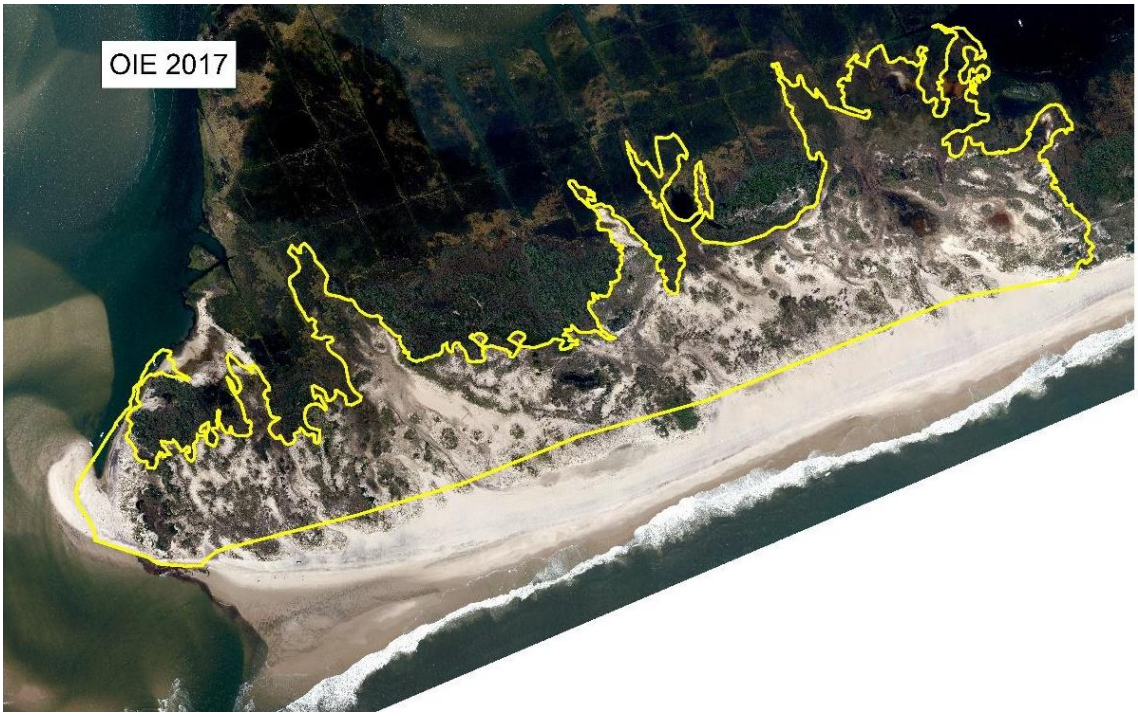


Figure 10.7. Old Inlet East Area, unclassified imagery from September 2017. Yellow polygon delineates overwash area above spring high tide in 2013.



Figure 10.8. Old Inlet East Area. Green is vegetation based on supervised classification. Imagery is from September 2017; 75% of pixels were classed as vegetation. Yellow polygon delineates overwash area above spring high tide in 2013.

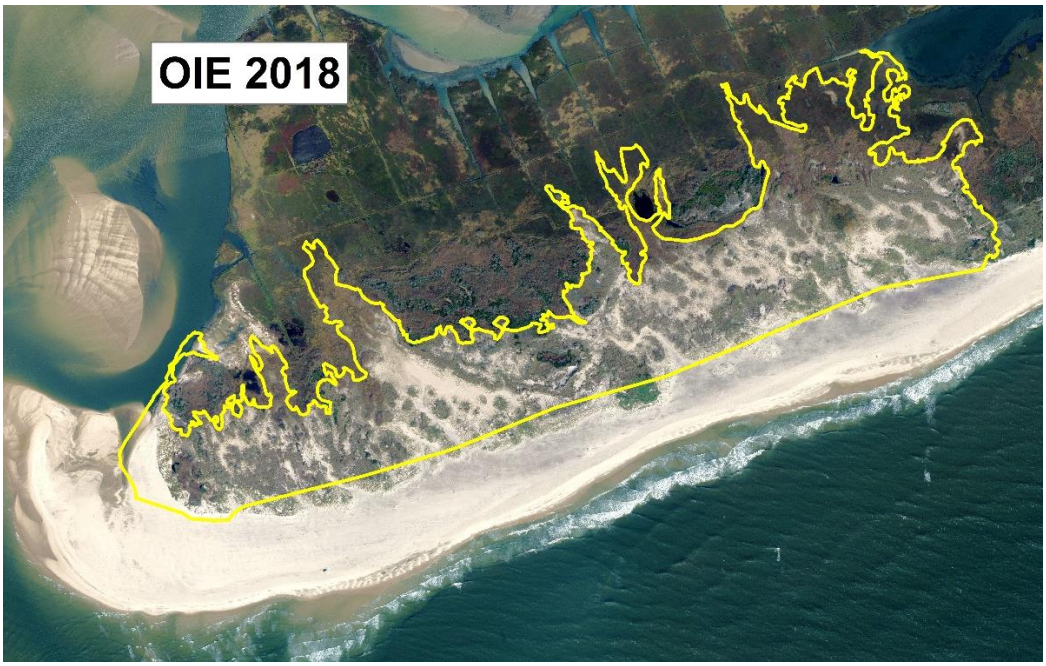


Figure 10.9. Old Inlet East Area, unclassified imagery from September 2018. Yellow polygon delineates overwash area above spring high tide in 2013.

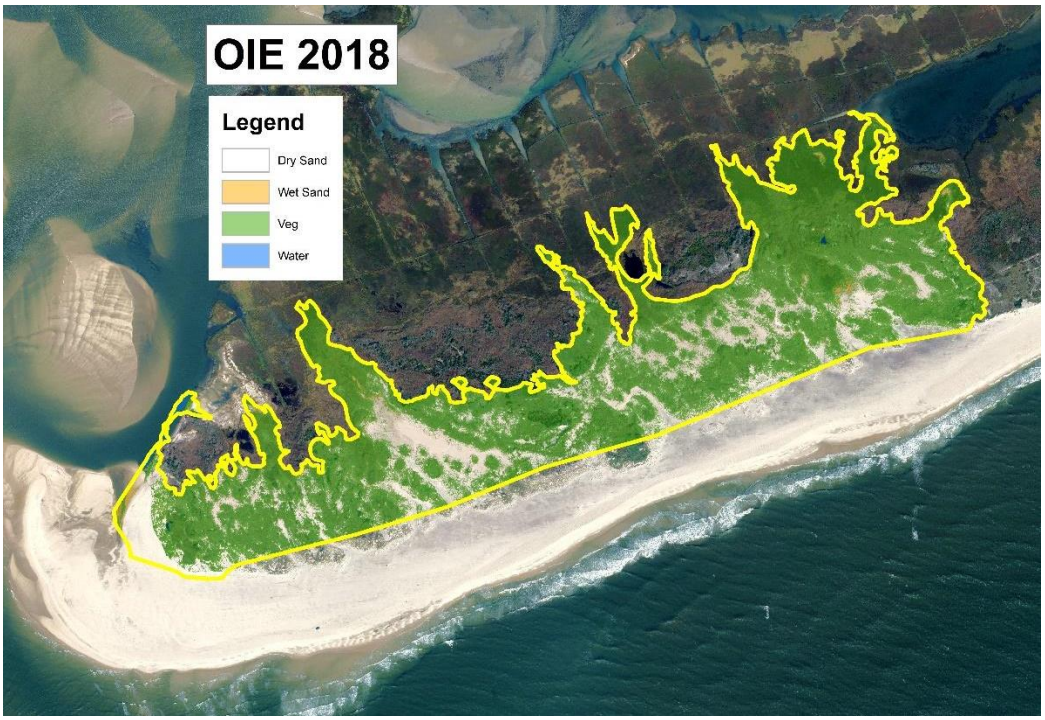


Figure 10.10. Old Inlet East Area. Green is vegetation based on supervised classification. Imagery is from September 2018; 77% of pixels were classed as vegetation. Yellow polygon delineates overwash area above spring high tide in 2013.

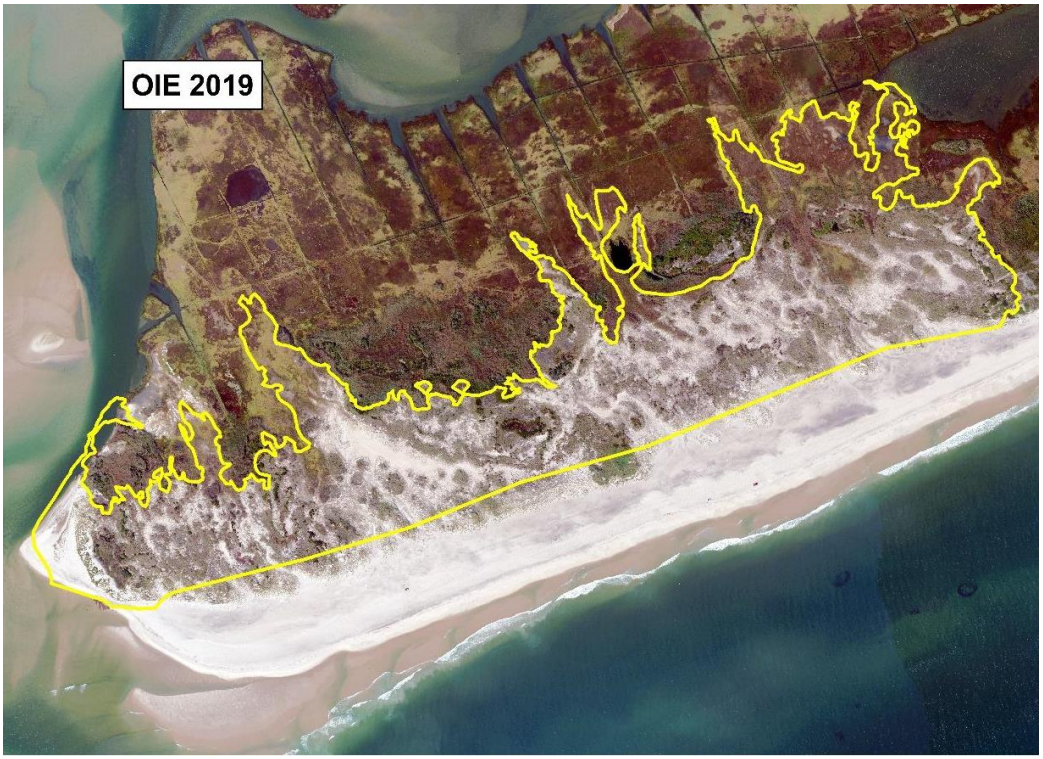


Figure 10.11. Old Inlet East Area, unclassified imagery from September 2019. Yellow polygon delineates overwash area above spring high tide in 2013.

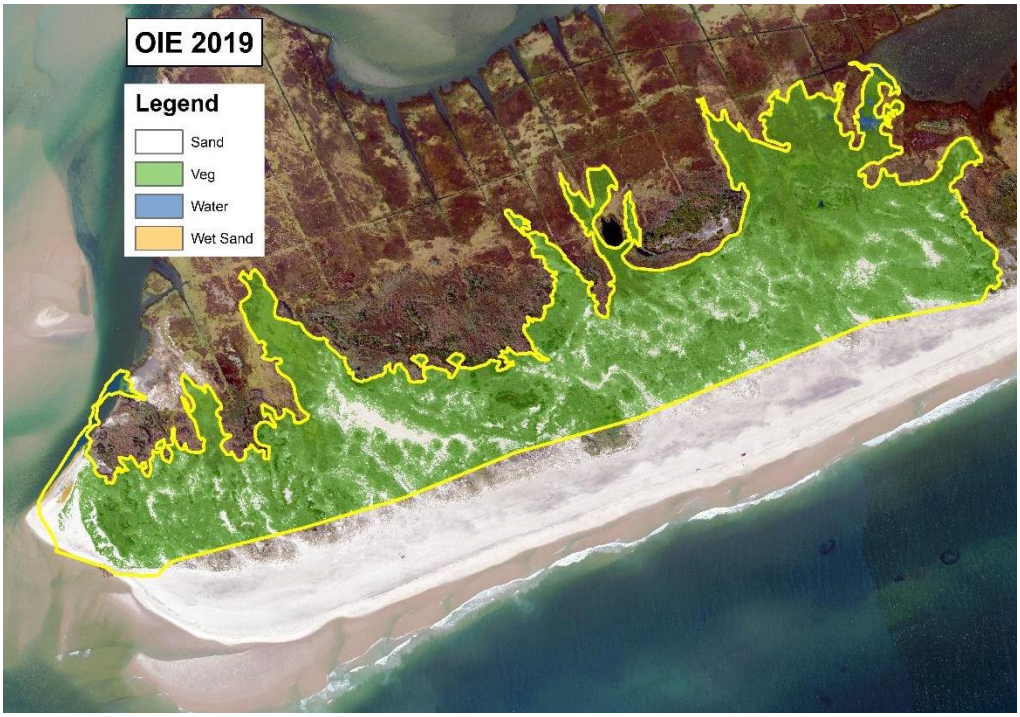


Figure 10.12. Old Inlet East Area. Green is vegetation based on supervised classification. Imagery is from September 2019; 88% of pixels were classed as vegetation. Yellow polygon delineates overwash area above spring high tide in 2013.

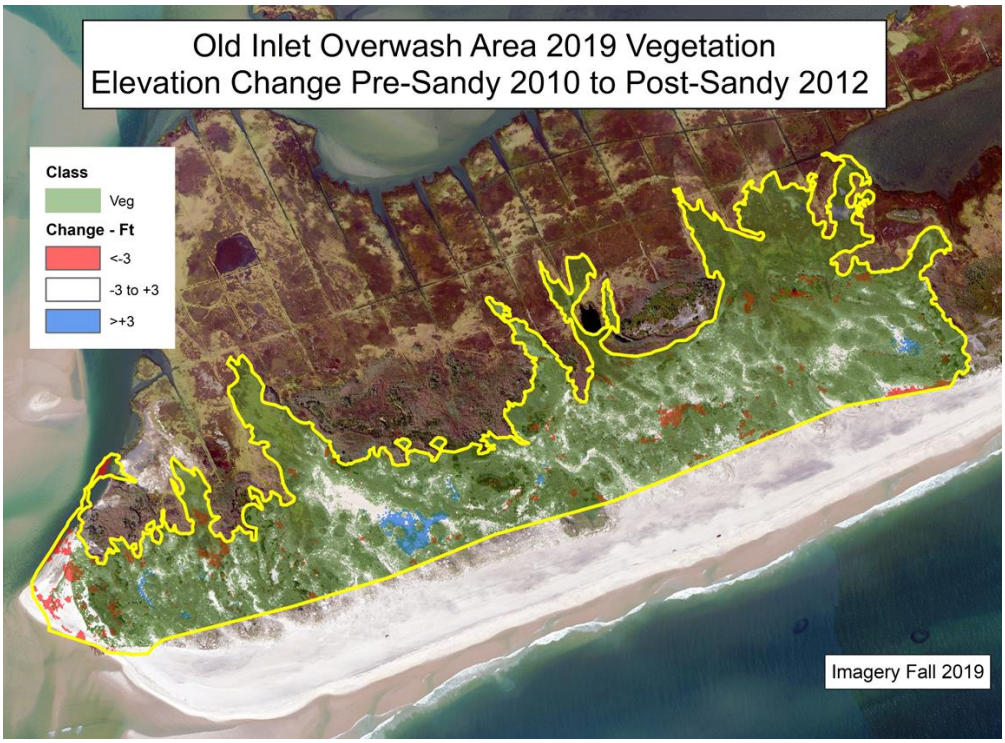


Figure 11. Old Inlet East overwash area elevation changes 2010-2012 as a result of Hurricane Sandy. Unclassified imagery from September 2019. Yellow polygon delineates designated maintenance area in 2013.

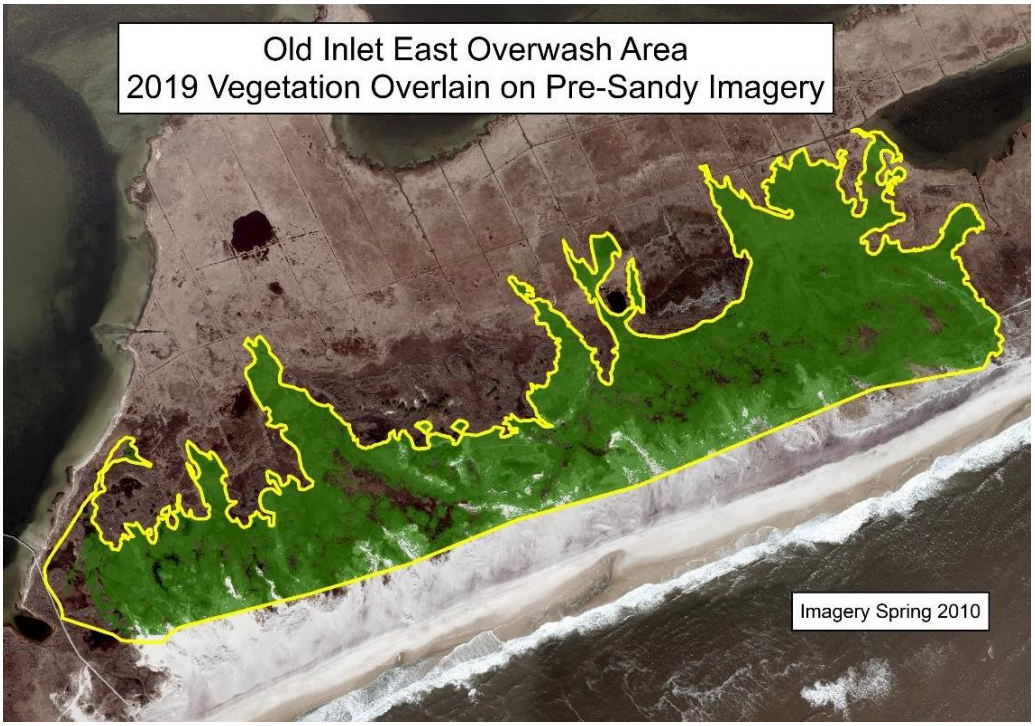


Figure 12. Old Inlet East overwash area. Green is vegetation based on supervised classification of September 2019 imagery. Underlying imagery is from April 2010. Yellow polygon delineates designated maintenance area in 2013.

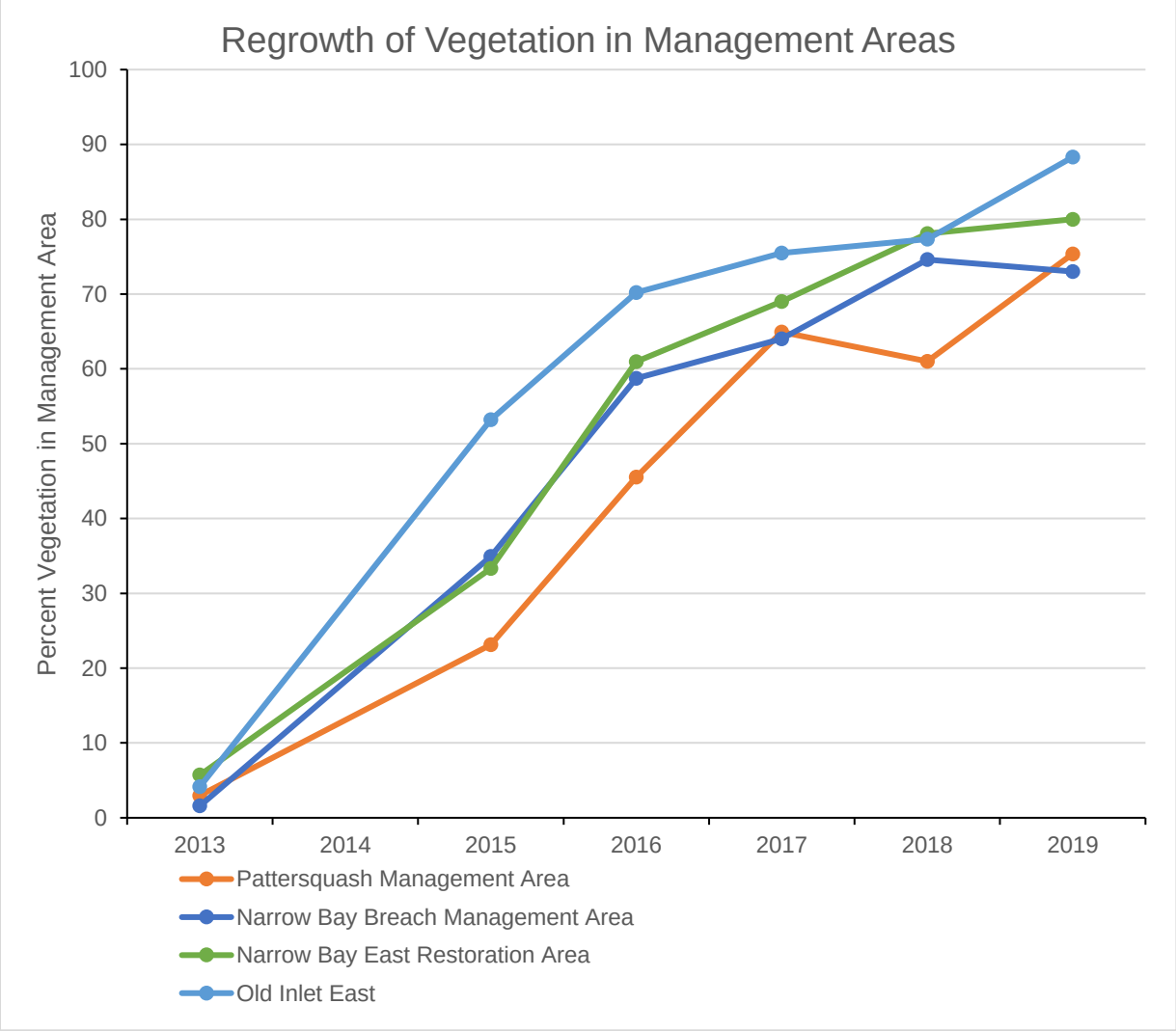


Figure 13. Graph of vegetation regrowth in Pattersquash and Narrow Bay Management Areas and Old Inlet East overwash. Percentage of cells classified as vegetation in overwash areas on Fire Island, New York, for 2013, 2015, 2016, 2017, 2018, and 2015–2019.2019

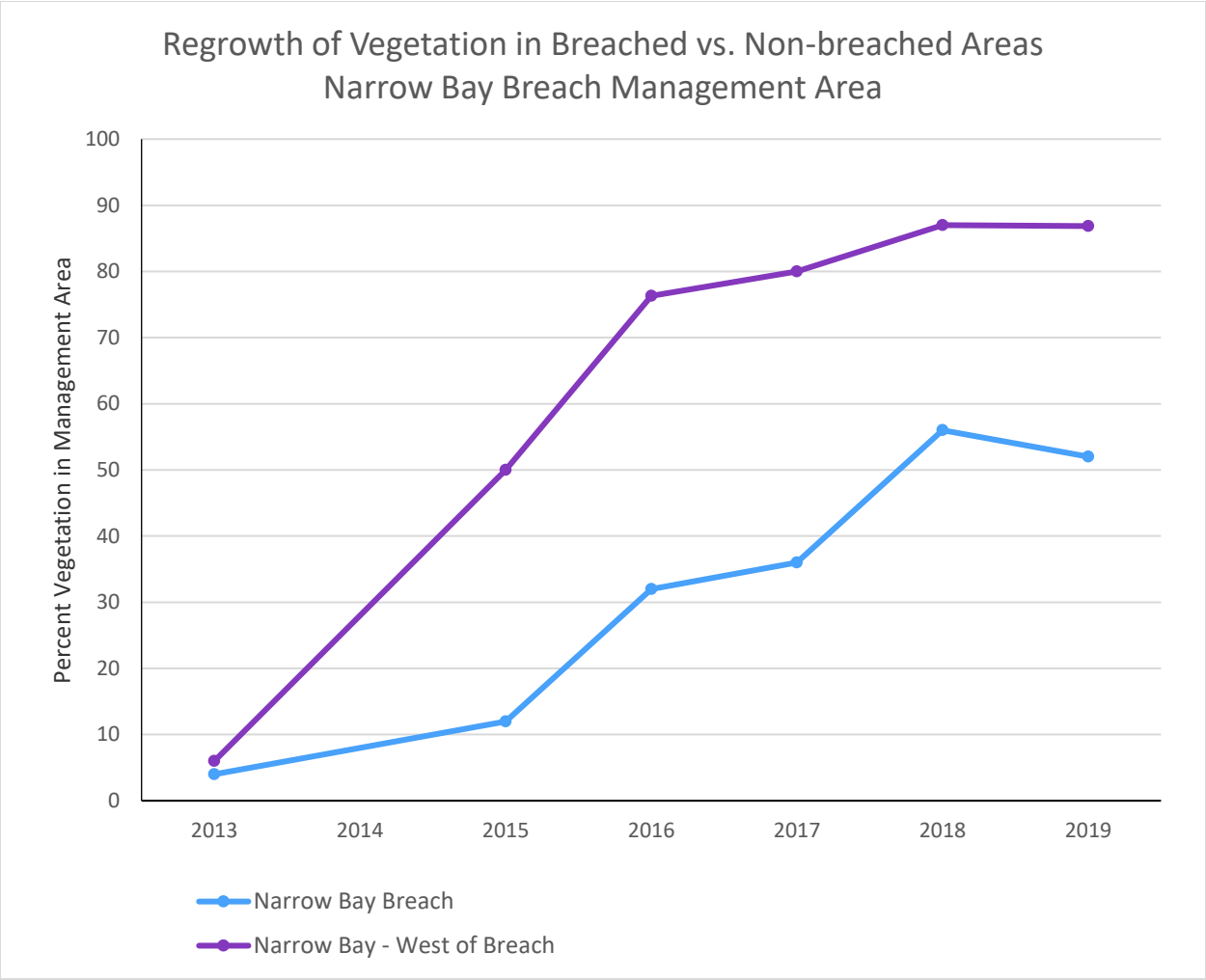


Figure 14. Graph of vegetation regrowth in breach versus non-breached areas. Percentage of cells classified as vegetation for Narrow Bay Breach management/restoration Management Area on Fire Island, New York, for 2013, 2015–2019.2015, 2016, 2017, 2018, and 2019

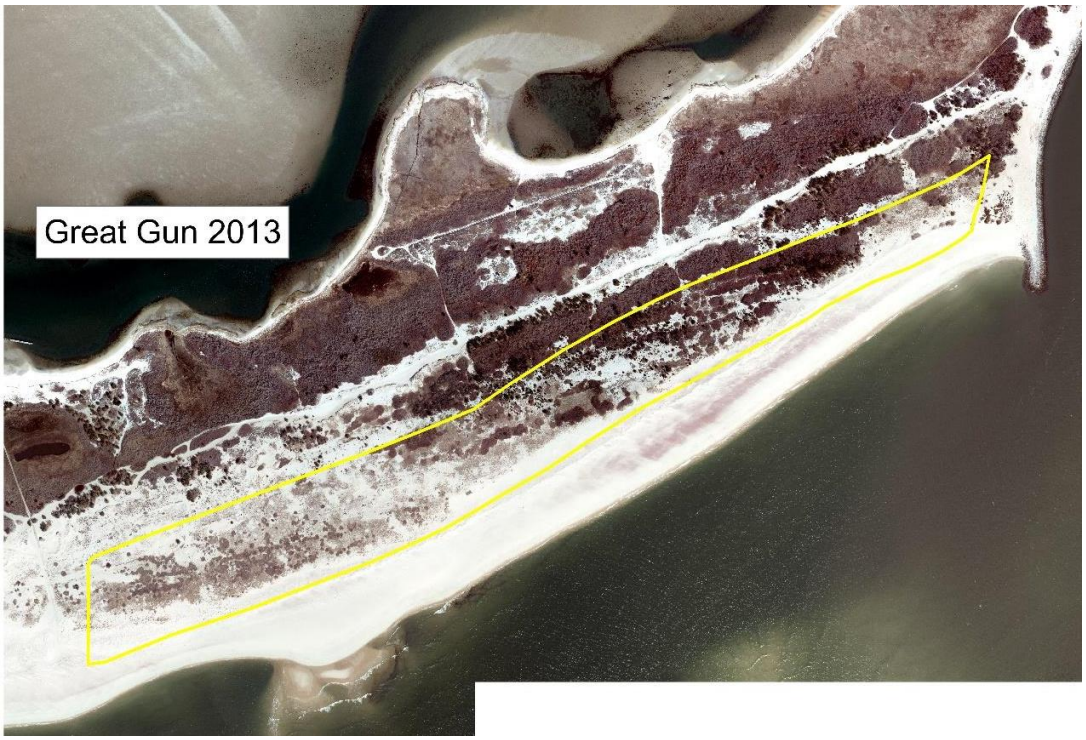


Figure 15.1. Great Gun Restoration Area. Unclassified imagery from spring of 2013. Yellow polygon delineates designated maintenance area in 2013.



Figure 15.2. Great Gun Restoration Area, classified imagery from spring 2013. Yellow polygon delineates designated maintenance area in 2013. Green is vegetation based on supervised classification; 50% of pixels were classed as vegetation.



Figure 15.3. Great Gun Restoration Area, unclassified imagery from October 2015. Yellow polygon delineates designated maintenance area in 2013.



Figure 15.4. Great Gun Restoration Area. Green is vegetation based on supervised classification. Imagery is from October 2015; 1% of pixels were classed as vegetation. Yellow polygon delineates designated maintenance area in 2013.



Figure 15.5. Great Gun Restoration Area, unclassified imagery from September 2016. Yellow polygon delineates designated maintenance area in 2013.



Figure 15.6. Great Gun Restoration Area. Green is vegetation based on supervised classification. Imagery is from September 2016; 13% of pixels were classed as vegetation. Yellow polygon delineates designated maintenance area in 2013.



Figure 15.7. Great Gun Restoration Area, unclassified imagery from September 2017. Yellow polygon delineates designated maintenance area in 2013.



Figure 15.8. Great Gun Restoration Area. Green is vegetation based on supervised classification. Imagery is from September 2017; 25% of pixels were classed as vegetation. Yellow polygon delineates designated maintenance area in 2013.



Figure 15.9. Great Gun Restoration Area, unclassified imagery from September 2018. Yellow polygon delineates designated maintenance area in 2013.



Figure 15.10. Great Gun Restoration Area. Green is vegetation based on supervised classification. Imagery is from September 2018; 23% of pixels were classed as vegetation. Yellow polygon delineates designated maintenance area in 2013.

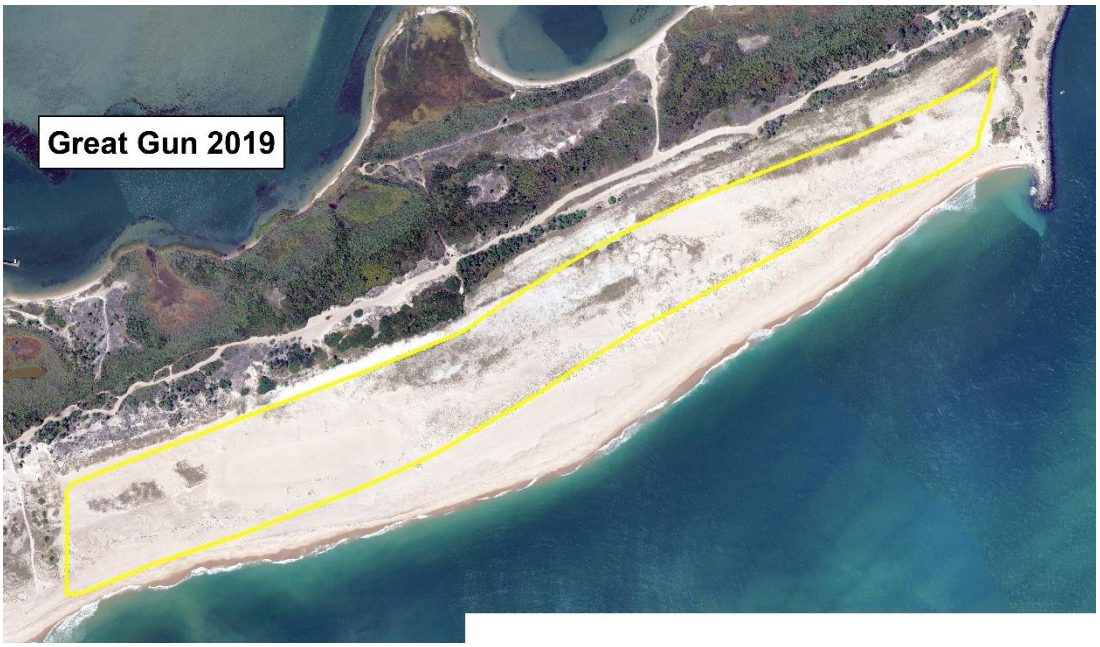


Figure 15.11. Great Gun Restoration Area, unclassified imagery from September 2019. Yellow polygon delineates designated maintenance area in 2013.



Figure 15.12. Great Gun Restoration Area. Green is vegetation based on supervised classification. Imagery is from September 2019; 26% of pixels were classed as vegetation. Yellow polygon delineates designated maintenance area in 2013.



Figure 16. Great Gun Restoration Area elevation changes 2012-2015 as a result of habitat creation. Classified imagery from September 2019. Yellow polygon delineates designated maintenance area in 2013.



Figure 17.1. New Made Dredge Restoration Area (orange polygon). Unclassified imagery from spring of 2013.

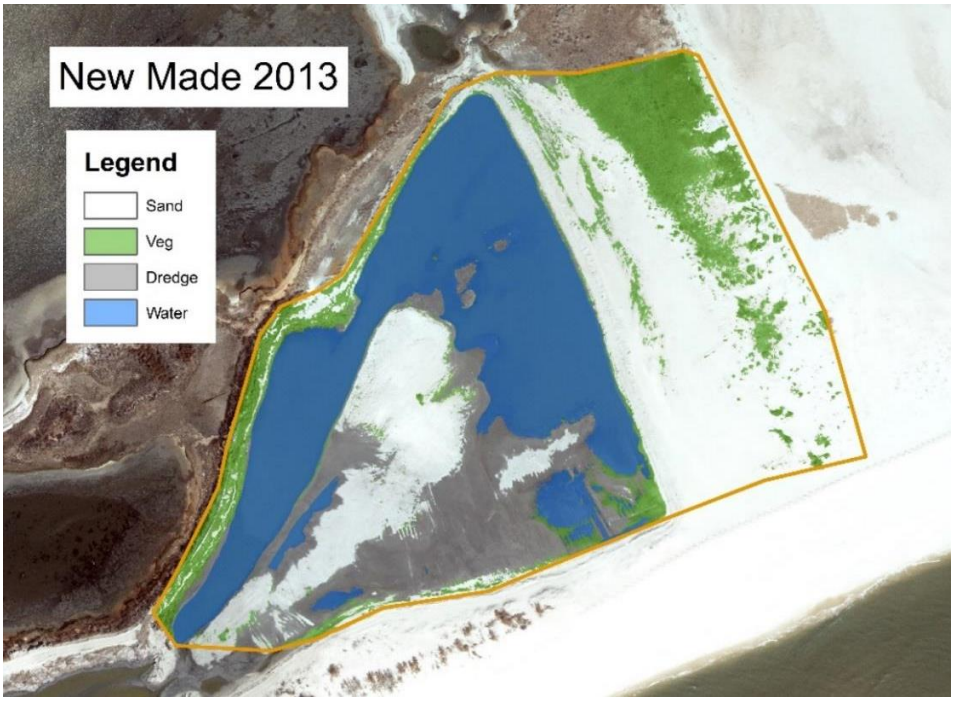


Figure 17.2. New Made Dredge Restoration Area (orange polygon). Imagery is from spring 2013. Green is vegetation based on supervised classification; 18% of pixels were classed as vegetation.

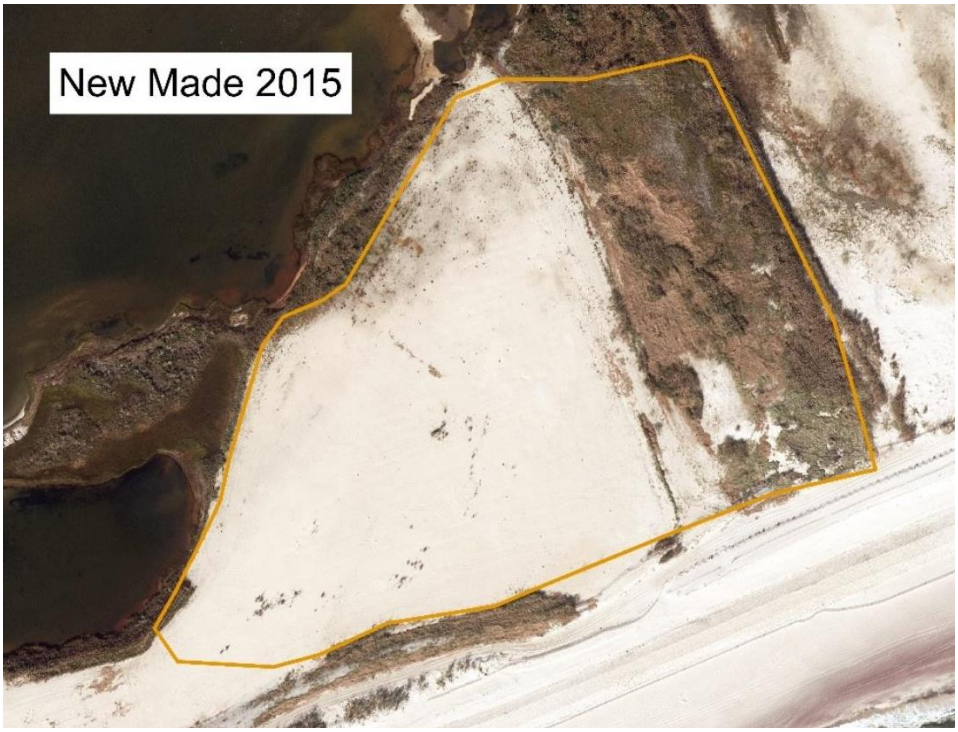


Figure 17.3. New Made Dredge Restoration Area (orange polygon). Unclassified imagery from October 2015.

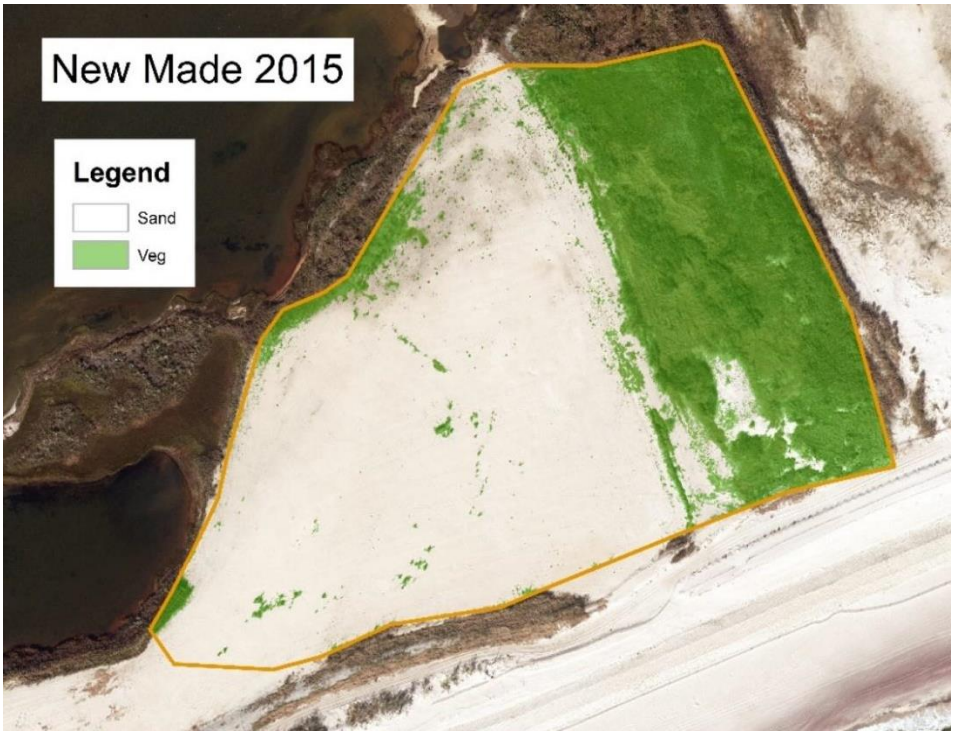


Figure 17.4. New Made Dredge Restoration Area (orange polygon). Imagery is from October 2015. Green is vegetation based on supervised classification; 33% of pixels were classed as vegetation.

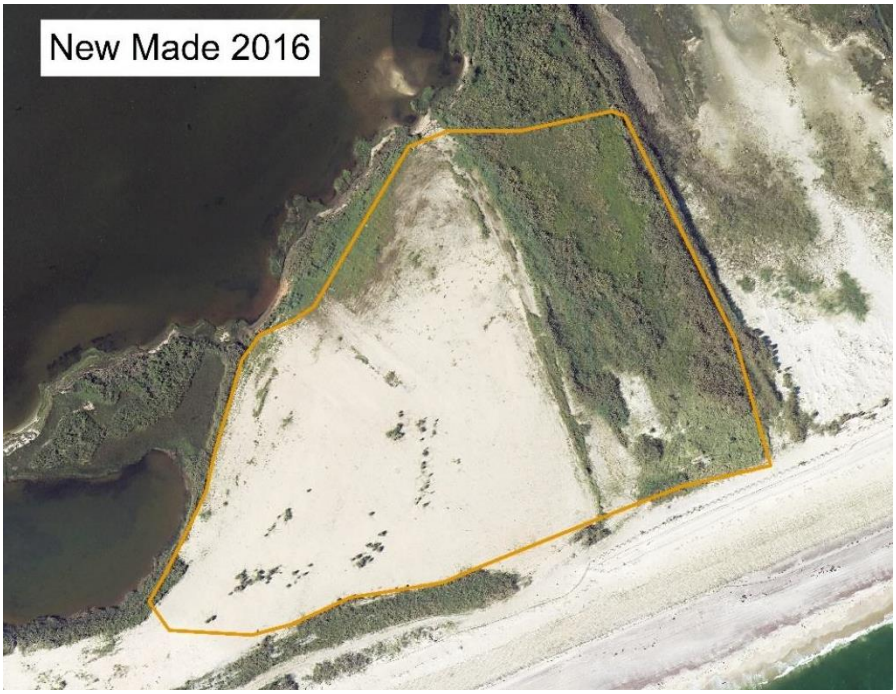


Figure 17.5. New Made Dredge Restoration Area (orange polygon). Unclassified imagery from September 2016.

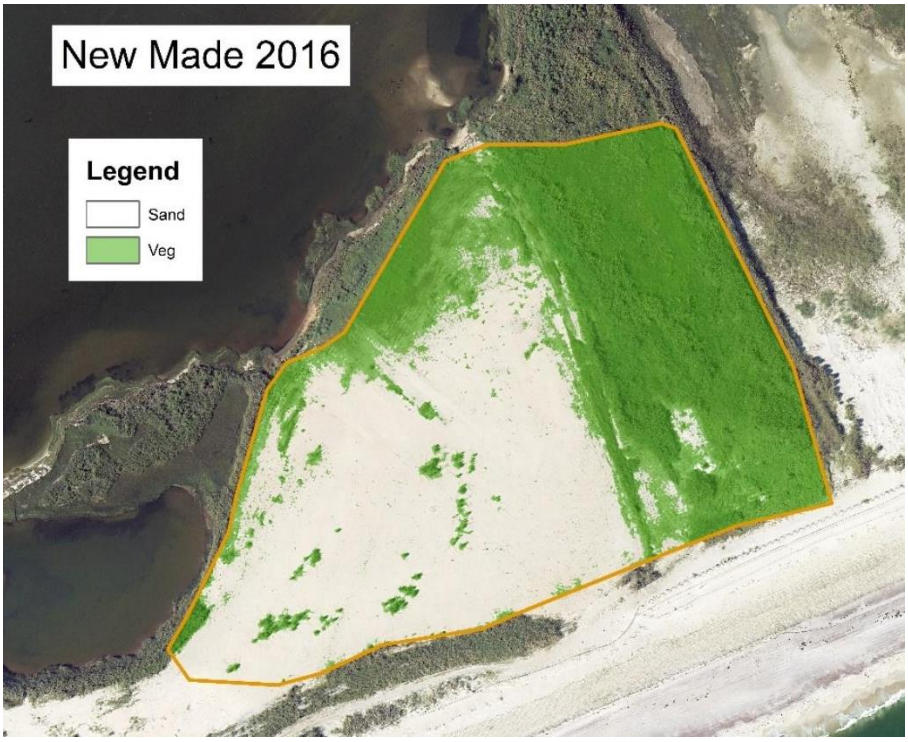


Figure 17.6. New Made Dredge Restoration Area (orange boundary). Imagery is from September 2016. Green is vegetation based on supervised classification; 55% of pixels were classed as vegetation.



Figure 17.7. New Made Dredge Restoration Area (orange boundary). Unclassified imagery from September 2017.



Figure 17.8. New Made Dredge Restoration Area (orange boundary). Imagery is from September 2017. Green is vegetation based on supervised classification; 61% of pixels were classed as vegetation.

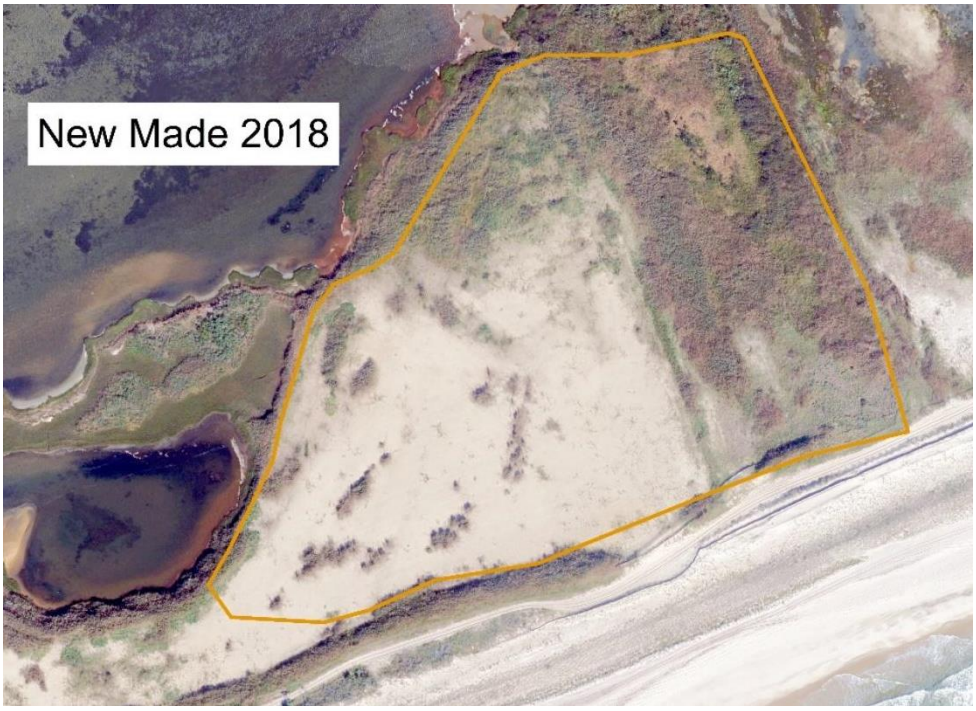


Figure 17.9. New Made Dredge Restoration Area (orange boundary). Unclassified imagery from September 2018.



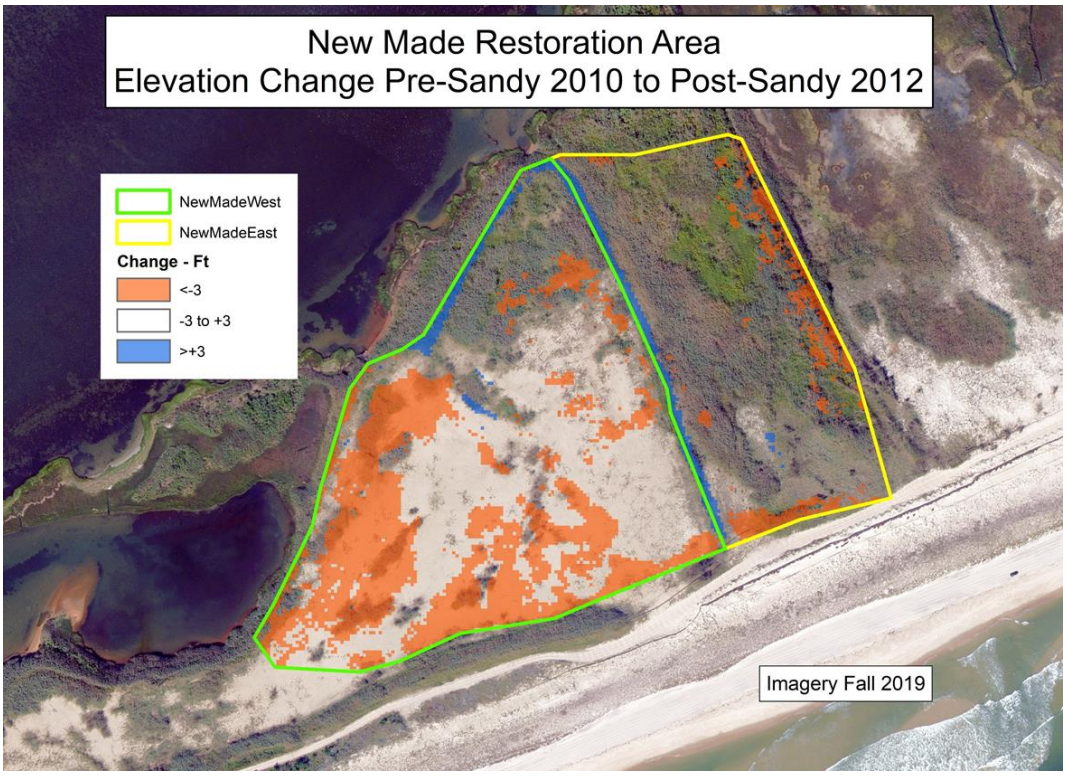
Figure 17.10. New Made Dredge Restoration Area (orange boundary). Imagery is from September 2018. Green is vegetation based on supervised classification; 64% of pixels were classed as vegetation.



Figure 17.11. New Made Dredge Restoration Area (orange boundary). Unclassified imagery from September 2019.



Figure 17.12. New Made Dredge Restoration Area (orange boundary). Imagery is from September 2019. Green is vegetation based on supervised classification; 71% of pixels were classed as vegetation.



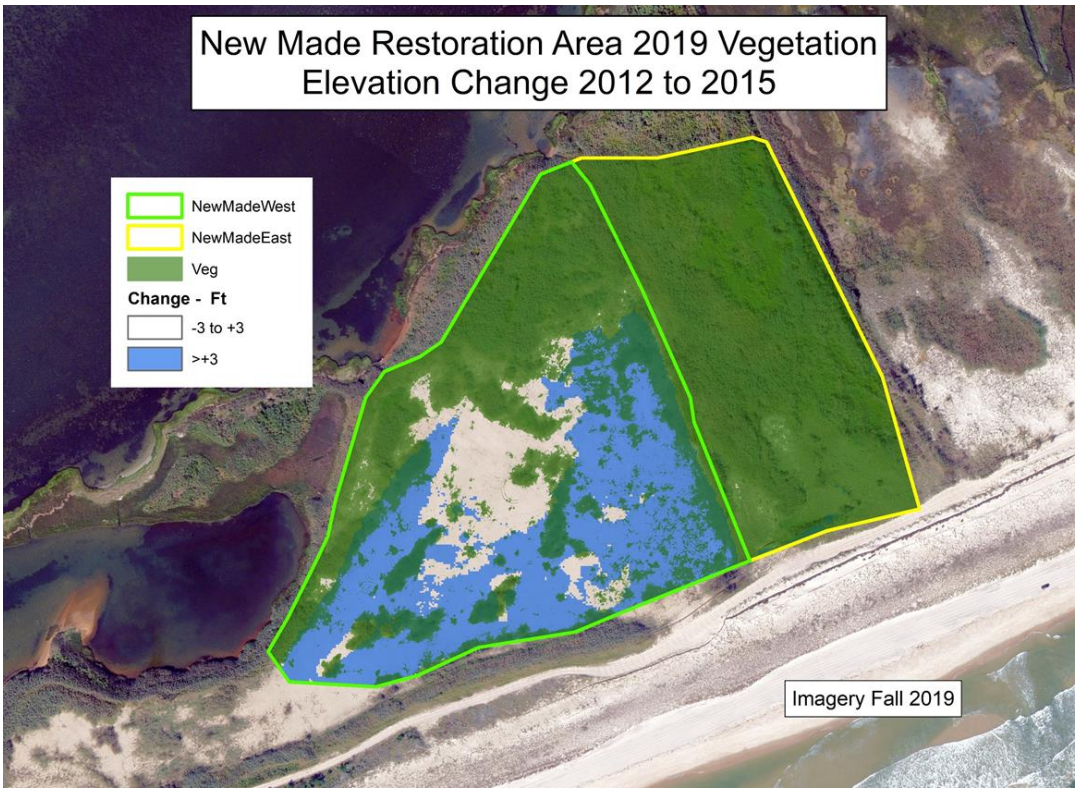


Figure 18.1/18.2. New Made Restoration Area elevation changes 2010-2012 showing excavations and 2012-–2015 showing subsequent fill. Unclassified imagery from September 2019. Yellow polygon delineates designated maintenance area in 2013.

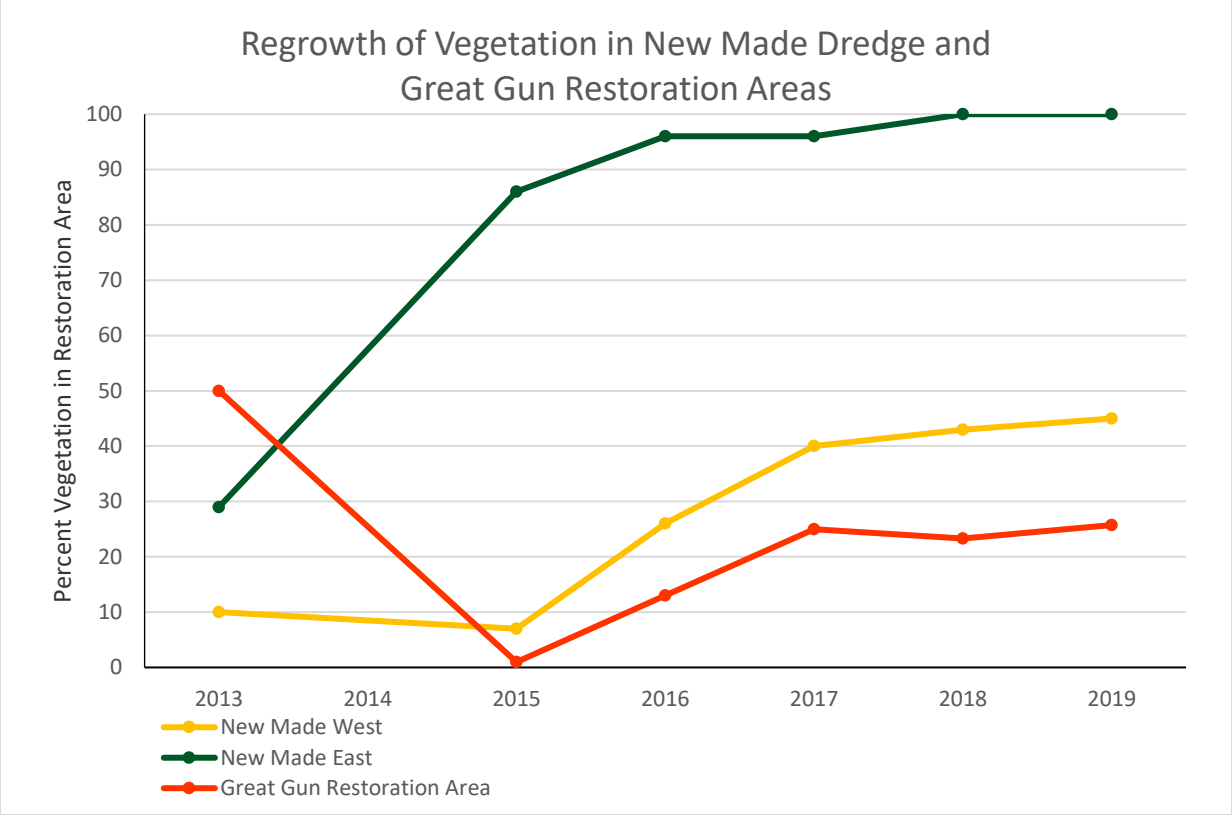


Figure 19. Graph of vegetation regrowth in New Made (East and West) and Great Gun Restoration Areas on Fire Island, New York. Percentage of cells classified as vegetation for 2013 and 2015–2019.



Figure 20.1. Impact of surficial disturbance on regrowth of vegetation - New Made Dredge Spoil Restoration Area. The red polygon outlines the area scraped of vegetation in May 2016. Imagery is from February 2016 showing the extent of vegetation immediately prior to removal.



Figure 20.2. Impact of surficial disturbance on regrowth of vegetation - New Made Dredge Spoil Restoration Area. Google Earth imagery from May 11, 2016 shows tracks of earthmoving equipment in the area where vegetation was scraped from the surface.



Figure 20.3. Impact of surficial disturbance on regrowth of vegetation - New Made Dredge Spoil Restoration Area. Imagery from September 2016 shows extensive vegetation regrowth within area of disturbance (red polygon).

Appendix C. Supplementary tables for Part 2, Section 6: Compare piping plover nest and chick survival rates in various habitats across Fire Island in 'Effectiveness monitoring and targeted studies in support of FIMI conservation measures.'

Table C-1. Estimates from a logistic exposure nest survival model testing the effects of management-unit and interval specific variables on nest survival. Model was developed using all available data (Full Study Area) and for Fire Island Only (i.e., No Cupsogue). The Fire Island only model was developed with the categorical mange variable comparing pre-and during the mange outbreak to after the mange outbreak.

Variable	Full Study Area					Fire Island Only - with fox				
	Mean	SD	LCI	UCI	f	Mean	SD	LCI	UCI	f
Exclosed	1.33	0.26	0.84	1.84	1.00	1.20	0.28	0.66	1.75	1.00
Age	-0.49	0.15	-0.79	-0.19	1.00	-0.59	0.16	-0.91	-0.27	1.00
Date	0.14	0.13	-0.12	0.40	0.85	0.15	0.15	-0.14	0.43	0.84
Eggs	0.21	0.10	0.02	0.39	0.98	0.22	0.10	0.01	0.42	0.98
Elevation	-0.11	0.10	-0.31	0.09	0.86	-0.17	0.11	-0.37	0.04	0.94
LC dist to bay	-0.08	0.13	-0.34	0.18	0.72	-0.06	0.15	-0.38	0.22	0.66
LC dist to ocean	0.09	0.13	-0.15	0.35	0.76	0.10	0.13	-0.16	0.37	0.77
Dist veg	-0.05	0.07	-0.19	0.09	0.77	-0.03	0.08	-0.18	0.12	0.68
Dist development	-0.04	0.12	-0.28	0.20	0.63	-0.03	0.13	-0.29	0.24	0.58
Temperature	0.20	0.11	-0.02	0.42	0.96	0.22	0.13	-0.03	0.48	0.96
Wind	-0.18	0.09	-0.36	0.00	0.98	-0.13	0.10	-0.33	0.06	0.91
Precip	-0.07	0.07	-0.20	0.06	0.87	-0.06	0.07	-0.18	0.08	0.81
Density	0.34	0.14	0.07	0.62	0.99	-0.16	0.22	-0.58	0.28	0.77
Post-mange	NA	-	-	-	-	1.49	0.43	0.68	2.37	1.00

Table C-2. Model estimates, standard deviation and lower and upper 95% credible intervals for logistic exposure nest survival models and Dail-Madsen adaptation of young-survival models. These models were used to estimate the effects of habitat type on nest and chick survival on Fire Island and Westhampton Island 2013–2019, and estimating the effects of predator trap success on a subset of nest and chick survival data from Fire Island, 2015–2018.

Model	Variable	Nest Survival*					Chick Survival				
		Mean	SD	LCI	UCI	f	Mean	SD	LCI	UCI	f
Habitat	Restoration	-0.25	0.29	-0.93	0.20	0.80	0.05	0.21	-0.36	0.48	0.59
	Manipulated	0.29	0.35	-0.68	0.69	0.72	0.55	0.27	0.04	1.10	0.98
	Natural	0.14	0.27	-0.45	0.61	0.68	0.39	0.18	0.04	0.75	0.99
	Breach	-0.20	0.26	-0.70	0.32	0.77	-0.64	0.22	-1.06	-0.19	1.00
Predator subset	Density	0.68	0.42	-0.13	1.53	0.95	-0.21	0.14	-0.48	0.06	0.94
	Fox Trap Success	0.11	0.28	-0.43	0.67	0.66	-0.29	0.11	-0.50	-0.08	1.00
	Raccoon Trap Success	0.25	0.36	-0.46	0.95	0.77	-0.02	0.14	-0.30	0.25	0.56
	Cat Trap Success	0.004	0.26	-0.51	0.51	0.51	0.10	0.11	-0.10	0.32	0.83

* Estimates from the predator subset nest survival model included all of the variables in Appendix 4 Table 1, but only relevant variables shown here for direct comparison to chick survival

Table C-3. Model estimates, standard deviations, 95% credible intervals and percentage of posterior-distribution that agrees with the sign of the mean (f) from a Dail-Madsen model modified into a piping plover young-survival model. We developed one model with all broods on Fire Island and Westhampton Island, 2013–2019 and one with all broods on Fire Island only with a categorical effect of pre- and during the mange outbreak as compared to after the mange outbreak.

Variable	Full Study Area					Fire Island Only – with fox				
	Mean	SD	LCI	UCI	f	Mean	SD	LCI	UCI	f
Max temperature	-0.02	0.06	-0.14	0.11	0.61	0.11	0.09	-0.07	0.30	0.89
Time	0.61	0.08	0.45	0.78	1.00	0.69	0.10	0.50	0.90	1.00
Average wind	0.04	0.07	-0.10	0.18	0.71	0.05	0.09	-0.12	0.22	0.72
Hatch day	-0.23	0.07	-0.37	-0.09	1.00	-0.34	0.10	-0.53	-0.16	1.00
LC dist bay	0.14	0.07	-0.01	0.29	0.97	0.00	0.08	-0.16	0.16	0.51
LC dist ocean	0.17	0.09	-0.01	0.35	0.97	0.08	0.10	-0.11	0.29	0.80
Dist veg	0.04	0.07	-0.10	0.18	0.71	-0.02	0.08	-0.17	0.14	0.61
Dist development	0.04	0.07	-0.09	0.17	0.71	-0.14	0.08	-0.30	0.01	0.96
Density	-0.10	0.07	-0.23	0.03	0.93	-0.05	0.10	-0.24	0.15	0.68
Post-mange	NA	-	-	-	-	0.29	0.22	-0.15	0.72	0.90

Table C-4. Model estimates from a linear regression assessing the effects of year, home range centroid habitat variables, and habitat type on brood home range size on Fire Island and Westhampton Island (left) and Fire Island only (right). The Fire Island only model included the categorical effect of pre-during and post-mange to test whether the reduction in

	Full Study Area					Fire Island Only			
Variable	Estimate	SE	Lower 95% CI	Upper 95% CI		Estimate	SE	Lower 95% CI	Upper 95% CI
Intercept	0.71	0.48	-0.22	1.65		1.92	0.55	0.84	2.99
2015	0.47	0.39	-0.30	1.24		0.26	0.43	-0.58	1.10
2016	0.03	0.34	-0.64	0.70		-0.10	0.40	-0.87	0.68
2017	-0.01	0.34	-0.67	0.66		-0.47	0.42	-1.28	0.35
2018	-0.15	0.35	-0.84	0.54		-0.78	0.46	-1.69	0.13
2019	-0.60	0.41	-1.40	0.20		-1.57	0.53	-2.60	-0.53
Distance to veg	-0.04	0.07	-0.19	0.10		-0.04	0.07	-0.18	0.11
Distance to development	0.07	0.07	-0.07	0.22		-0.03	0.09	-0.21	0.15
LC distance to bay	0.02	0.10	-0.18	0.23		-0.05	0.11	-0.27	0.18
LC distance to ocean	-0.13	0.08	-0.28	0.02		-0.14	0.08	-0.30	0.01
Number of points	0.04	0.01	0.01	0.06		0.02	0.01	0.00	0.05
Breach fill	-0.08	0.28	-0.63	0.47		-0.11	0.34	-0.78	0.56
Manipulated overwash	-0.26	0.21	-0.67	0.14		-0.46	0.22	-0.89	-0.03
Natural overwash	-0.20	0.30	-0.78	0.39		-0.67	0.36	-1.38	0.04
Restoration area	0.51	0.20	0.11	0.90		0.24	0.23	-0.21	0.68
Nesting density	-0.10	0.10	-0.31	0.10		0.28	0.15	-0.03	0.58
Post-mange	NA	-	-	-		-0.29	0.28	-0.84	0.27

Brood Home Range Figures 2014–2019

Figure C-1. 2014 brood home ranges on 2014 spring imagery. Robert Moses was not monitored in 2014, thus there are no brood home ranges for that year in 2014.





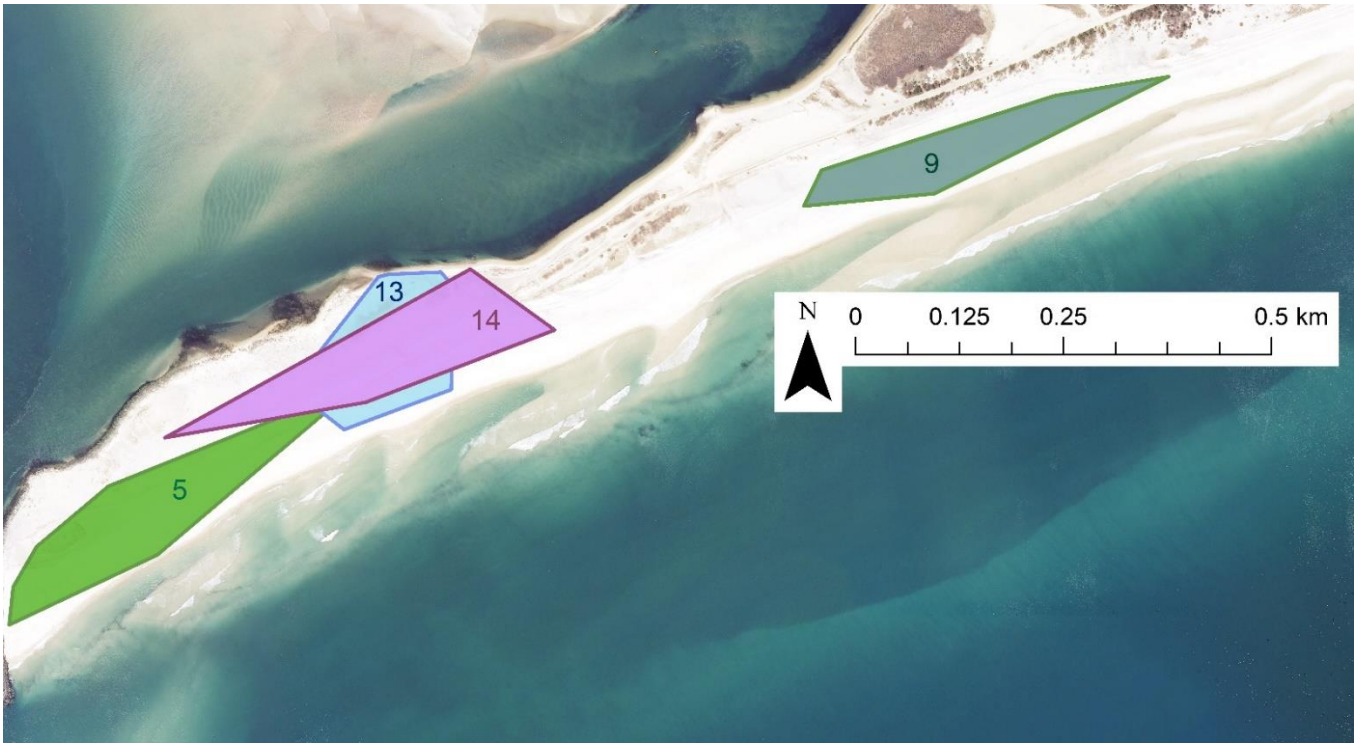
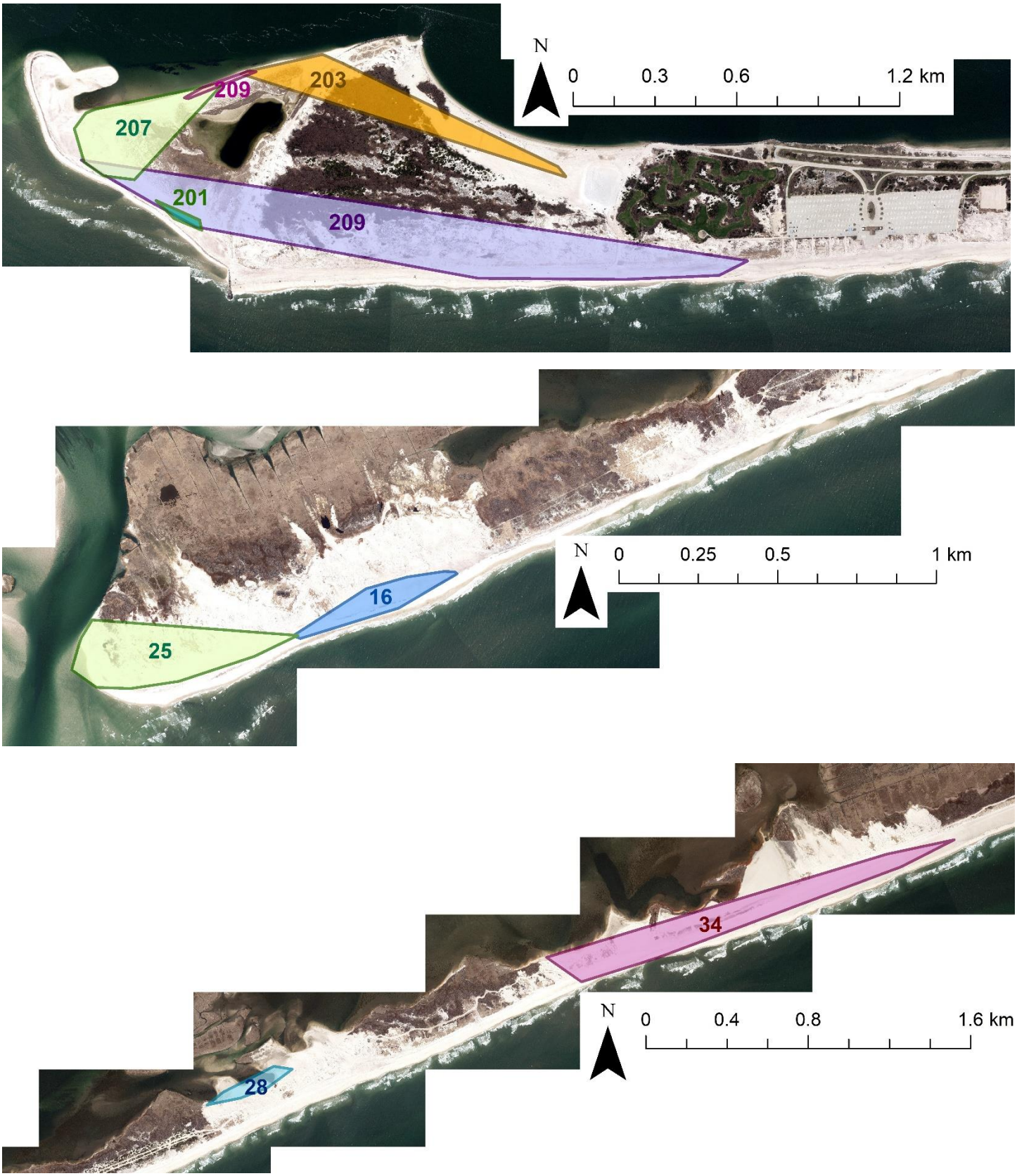

9
13
14
5
N
0
0.125
0.25
0.5 km

Figure C-2. 2015 brood home ranges on 2015 spring imagery



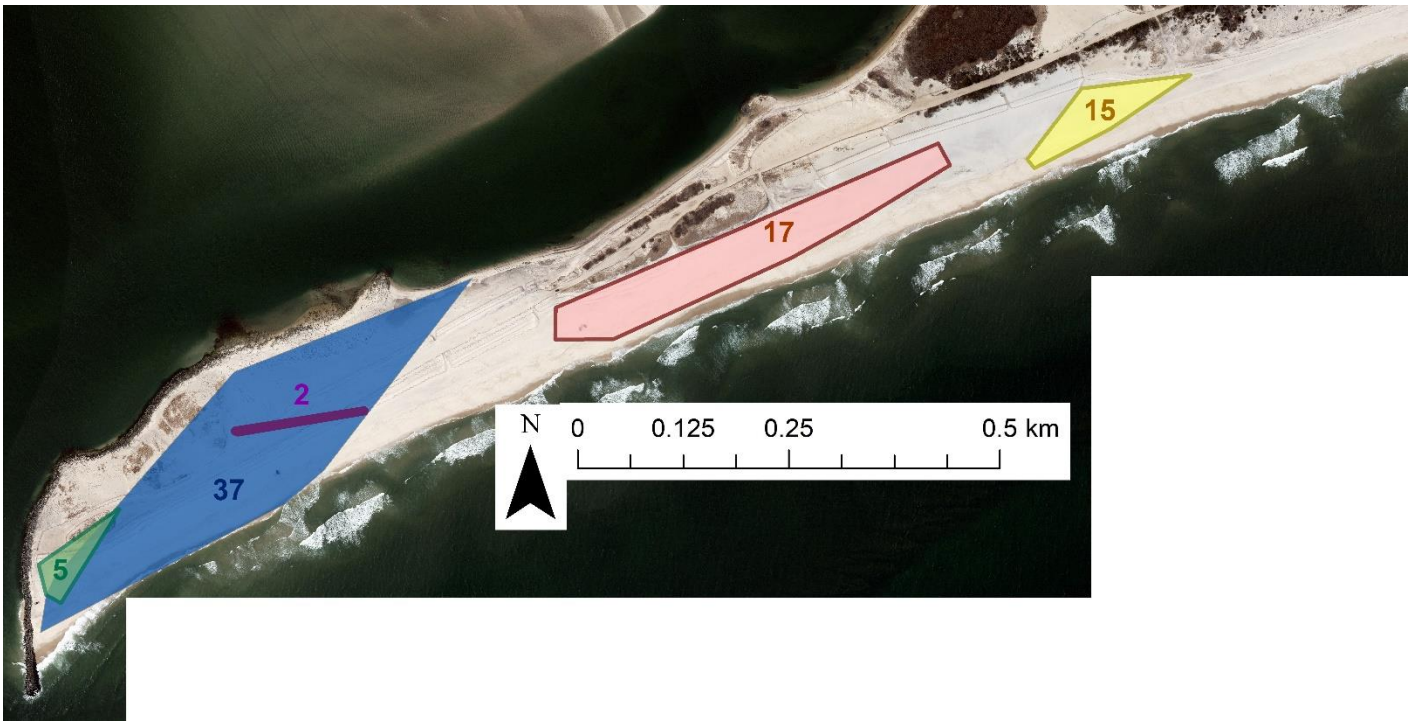
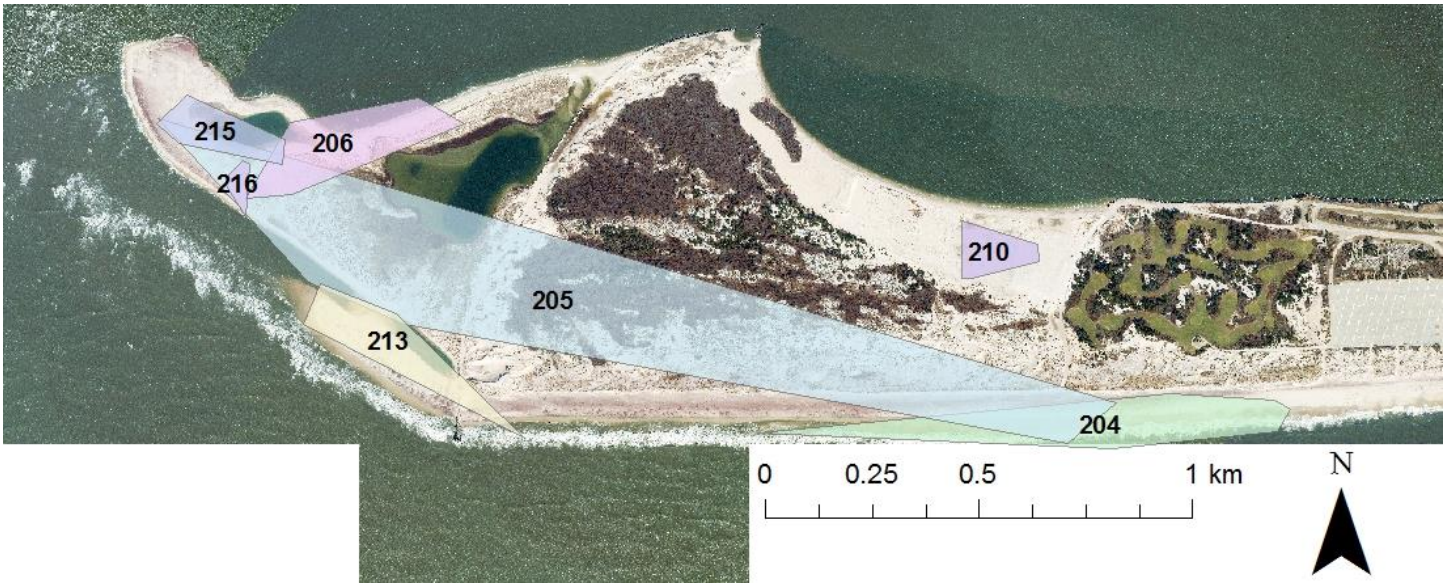
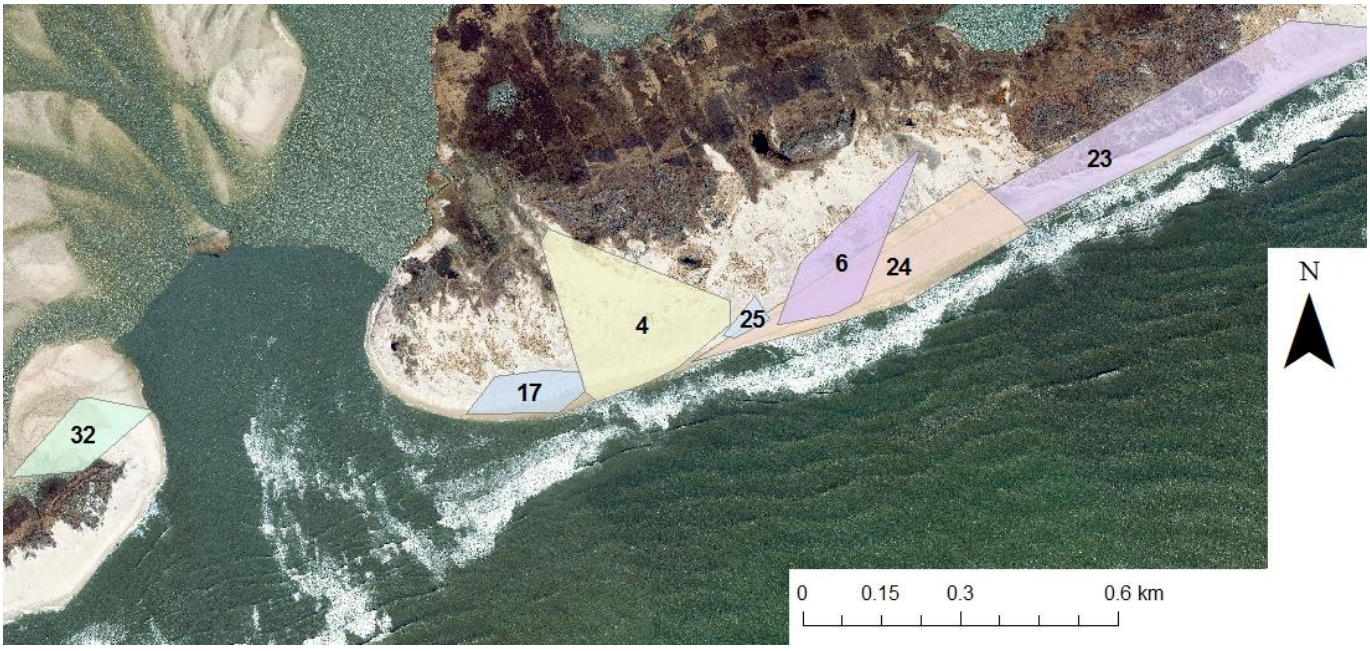
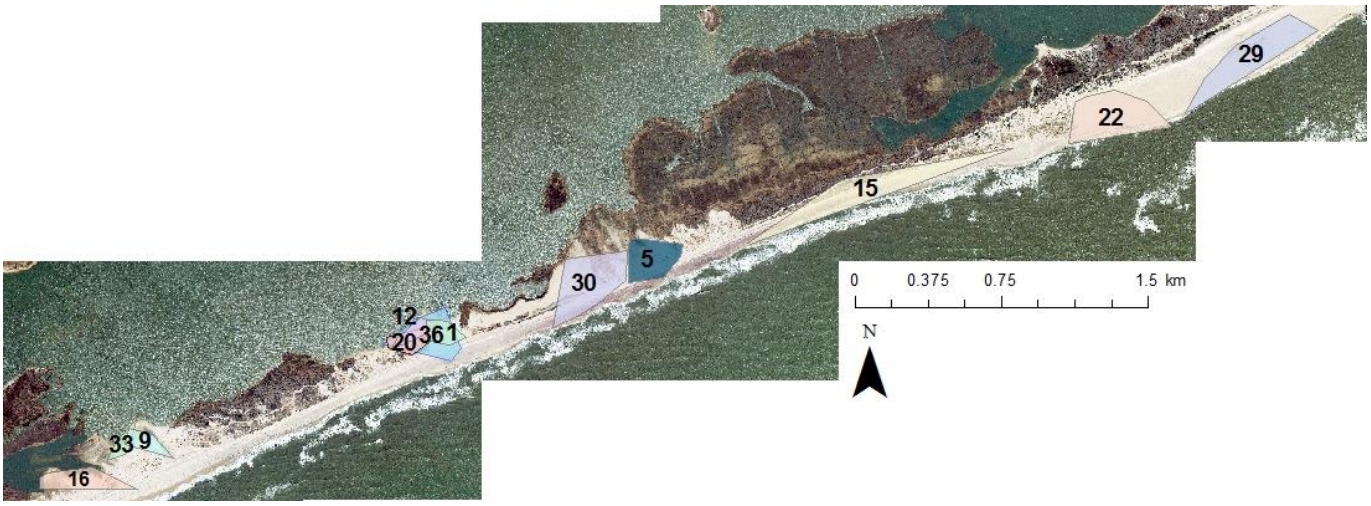

15
17
2
37
5
N
0
0.125
0.25
0.5 km

Figure C-3. 2016 brood home ranges






29
22
15
5
30
12
20
36
1
33
9
16
0 0.375 0.75 1.5 km
N

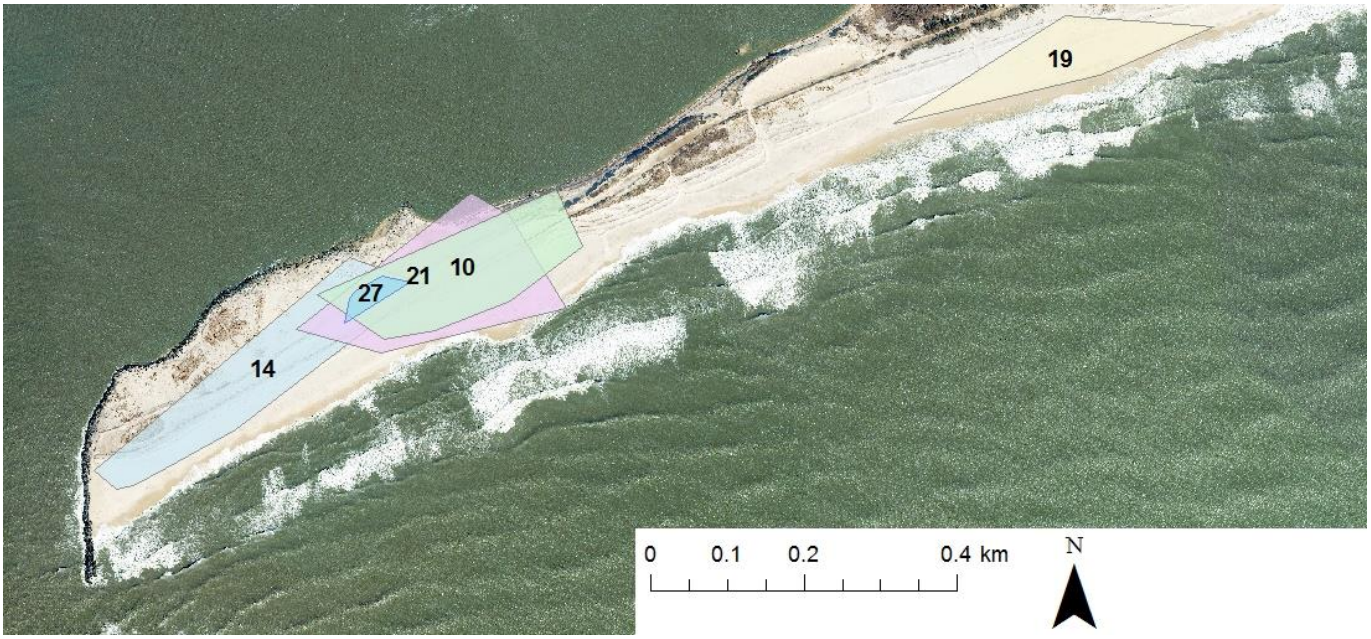
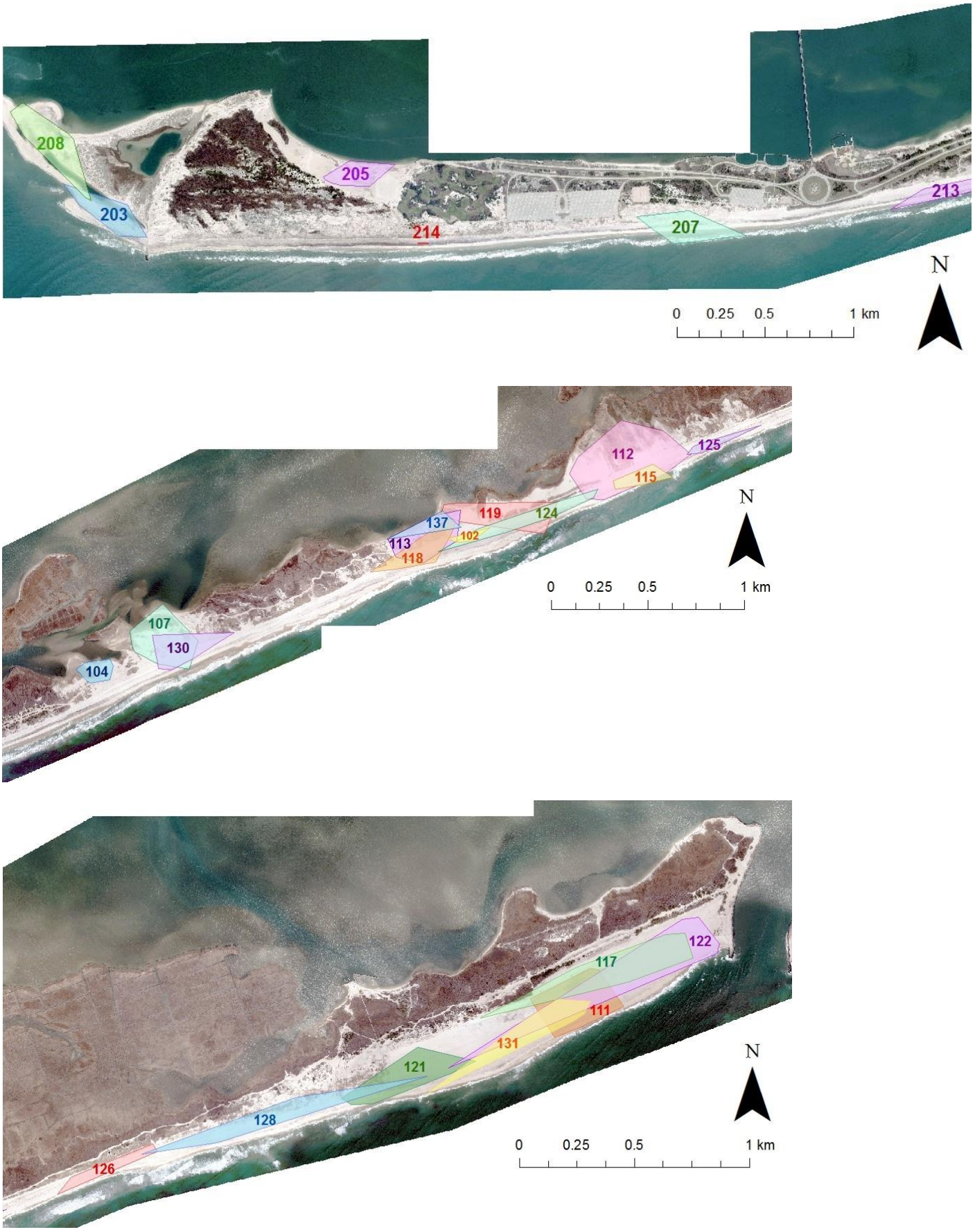
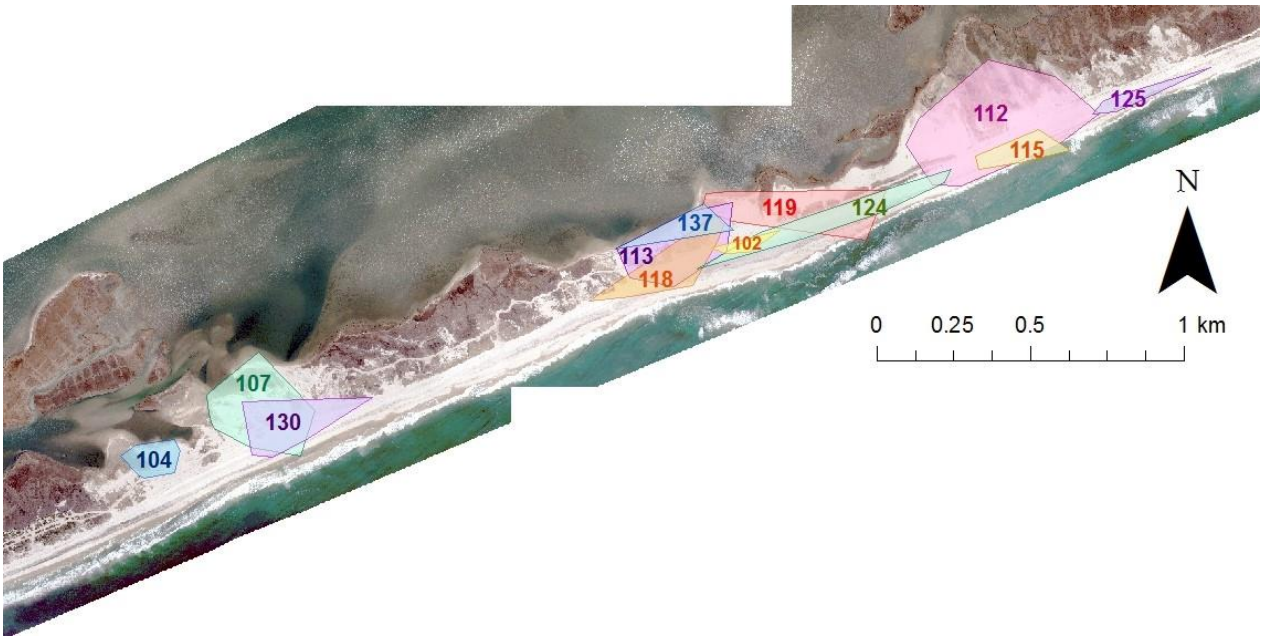

19
21
10
27
14
0 0.1 0.2 0.4 km
N

Figure C-4. 2017 Brood home ranges from west to east




112
125
115
N
119
124
137
102
113
118
0
0.25
0.5
1 km
107
130
104

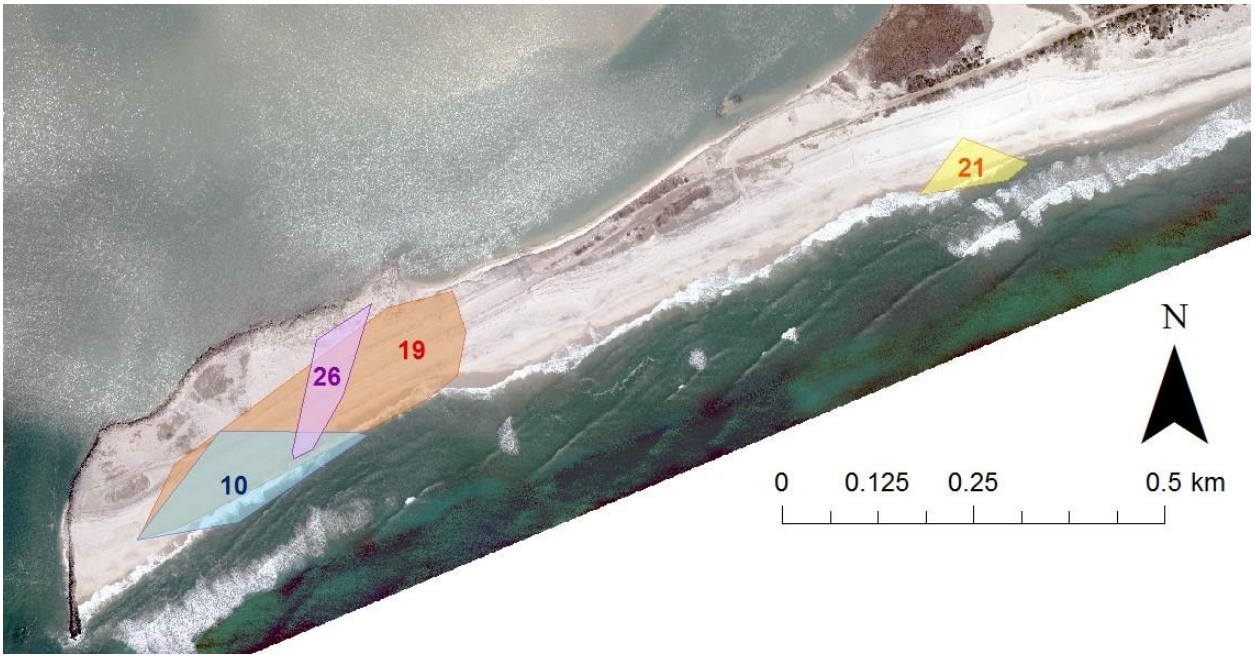
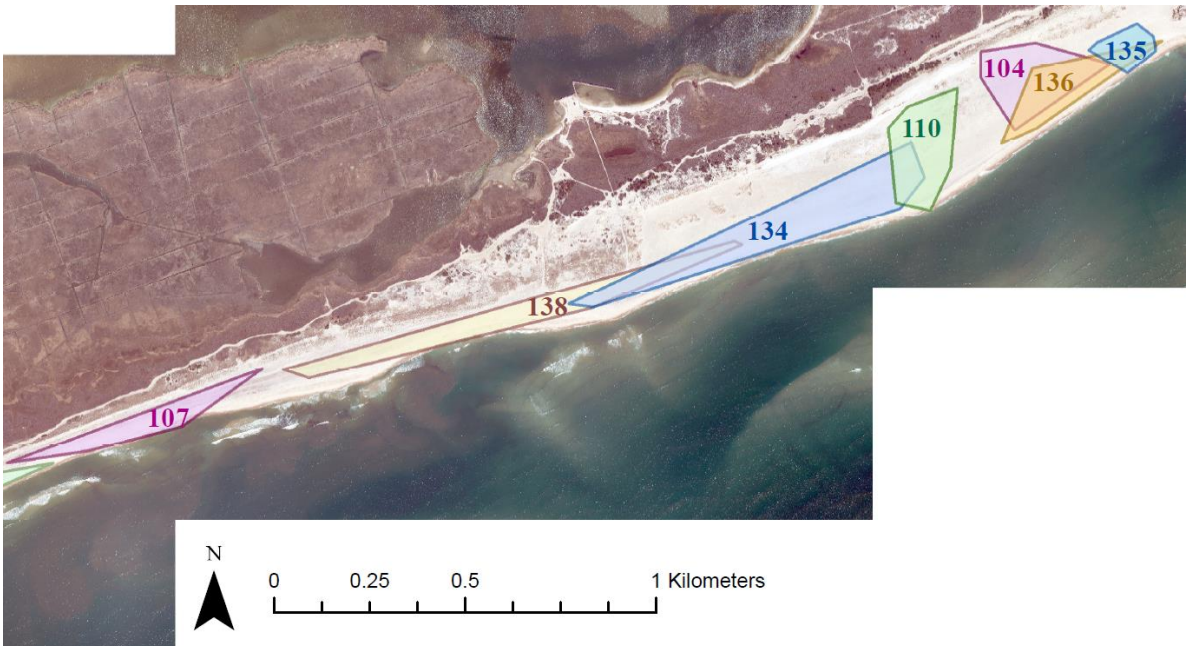

21
N
19
26
0
0.125
0.25
0.5 km
10

Figure C-5. 2018 brood home ranges from west to east.




104
135
136
110
134
138
107
N
0
0.25
0.5
1 Kilometers

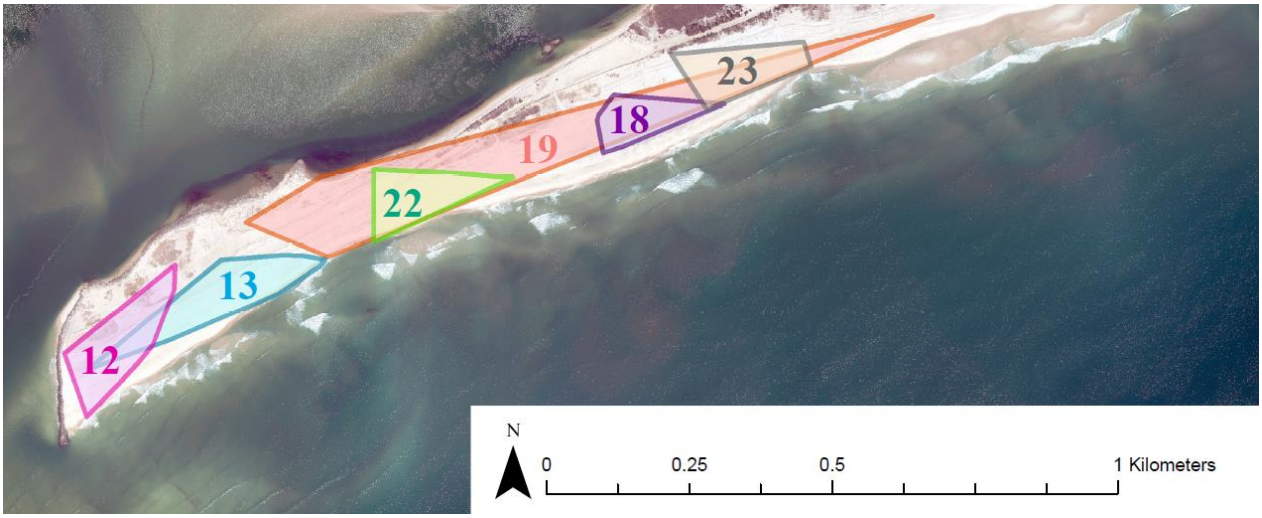
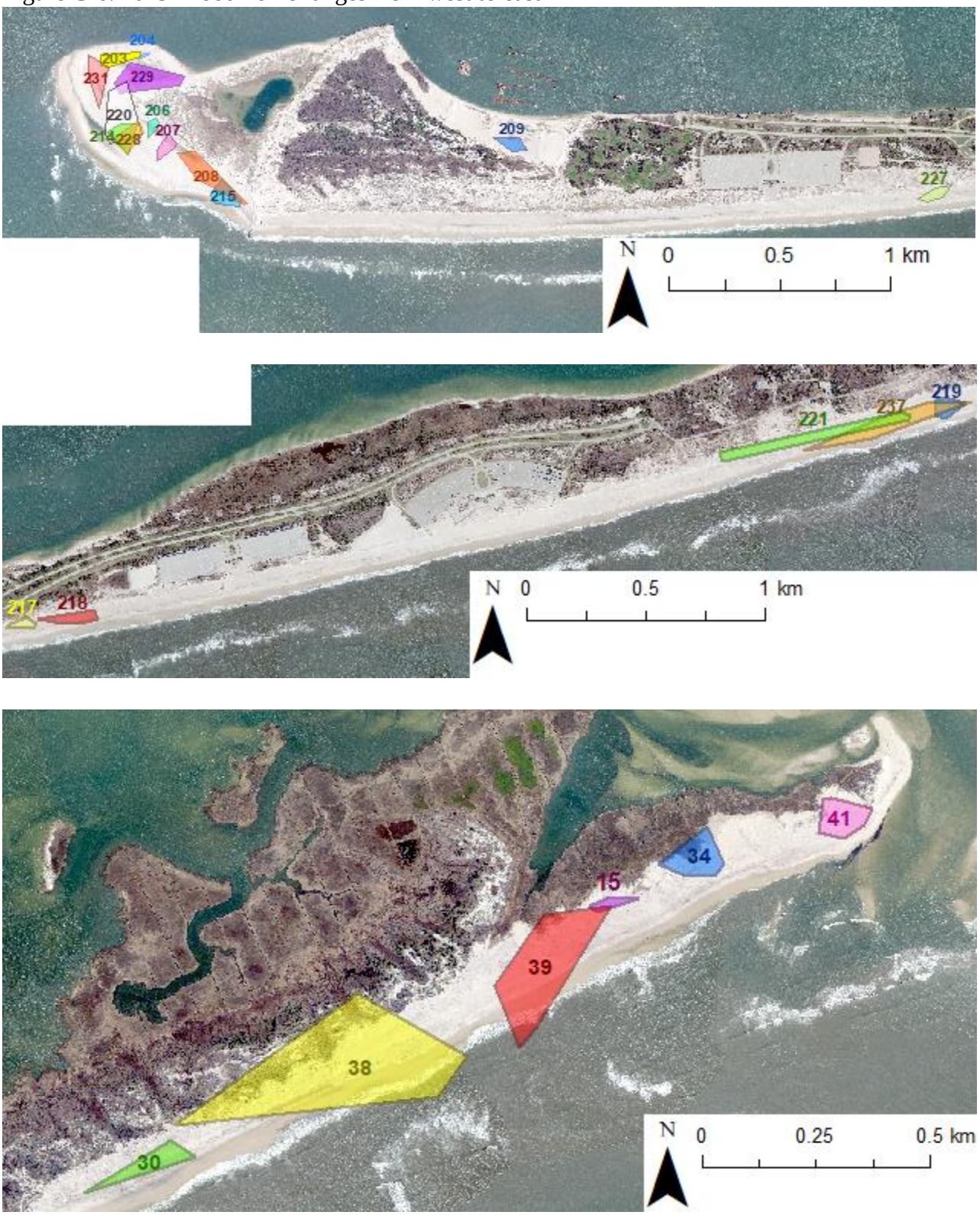
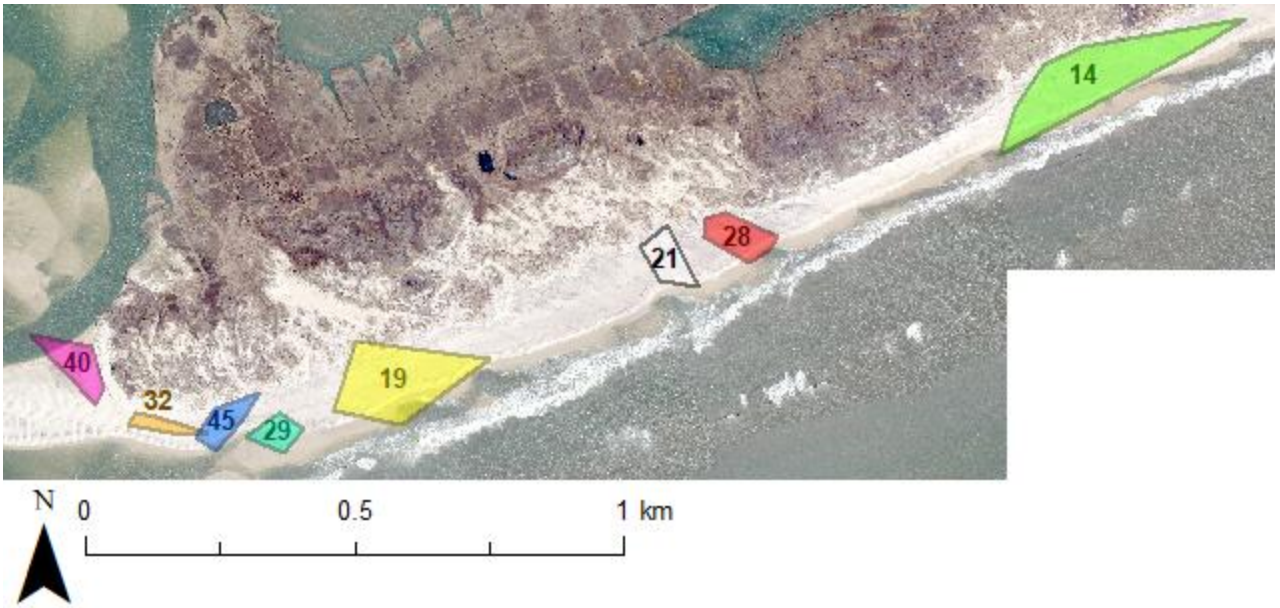
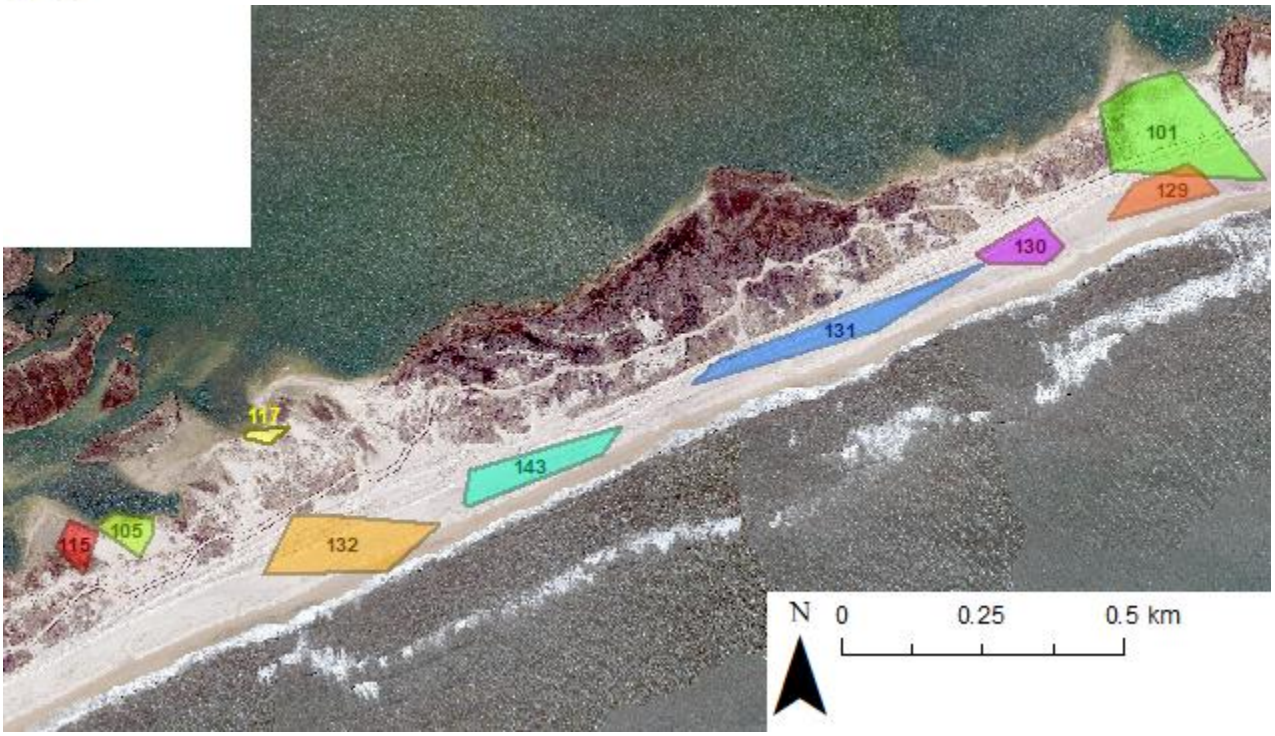
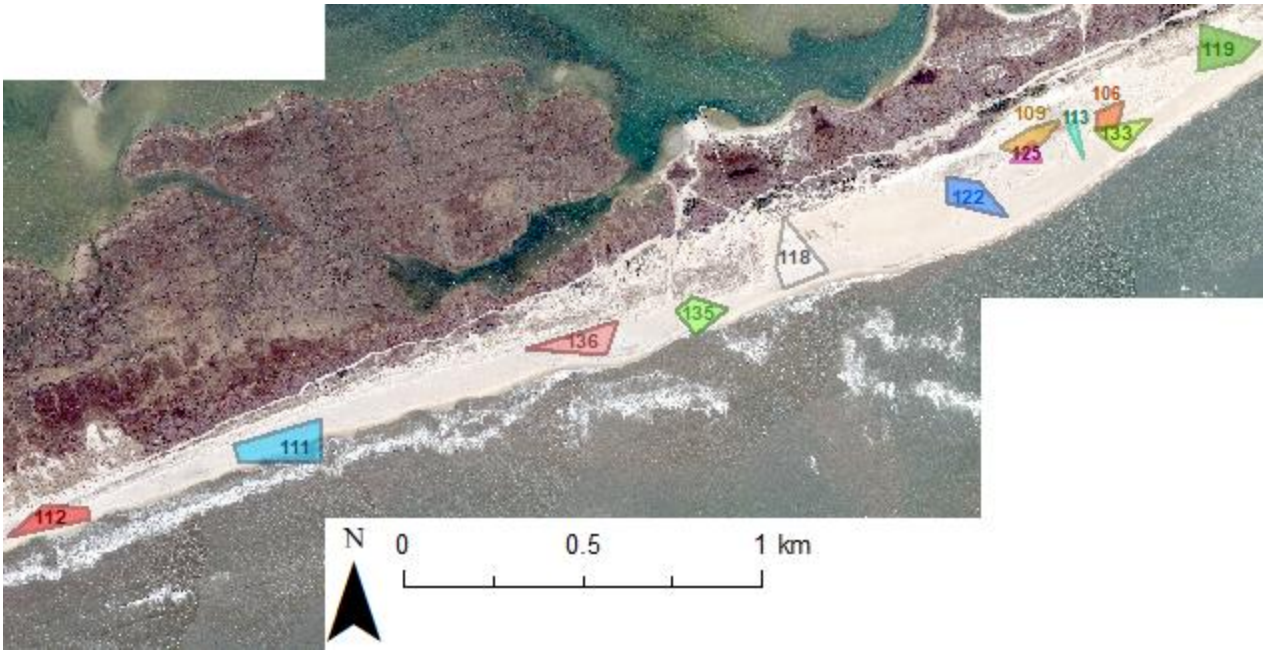

23
18
19
22
13
12
N
0
0.25
0.5
1 Kilometers

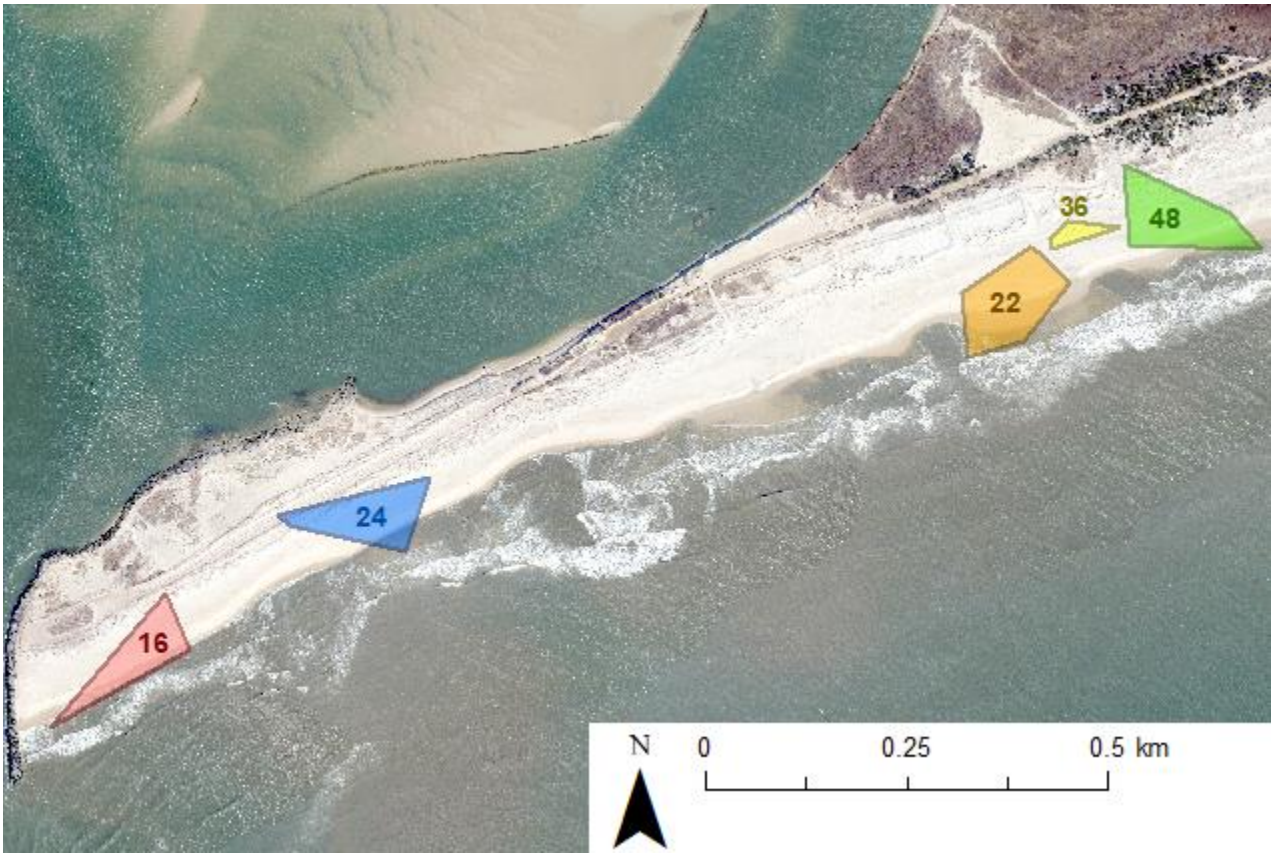
Figure C-6. 2019 Brood home ranges from west to east




14
28
21
40
32
45
29
19
N
0
0.5
1 km


101
129
130
131
117
143
105
115
132
N
0
0.25
0.5 km


119
106
109
113
133
125
122
118
135
136
111
112
N
0
0.5
1 km


36
48
22
24
16
N
0
0.25
0.5 km

Appendix D. Supplementary tables for Part 2, Section 7: Compare the relative contribution of local growth (natal recruitment) vs immigration in population growth associated with new habitats 'Effectiveness monitoring and targeted studies in support of FIMI conservation measures.'

Table D-1. Model ranking results for the Barker (1997) survival model of true survival (S), and fidelity (F) for after-hatch-year (AHY) and hatch- year (HY) Piping Plovers on Fire Island, New York, USA (FNS), 2013–2019, Models shown are those with ≥0.01 model weight (wi). For all models, resight rate (p), probability of emigrating and returning (F') were held constant, the supplementary period resight rate (R) was modeled as Age, the probability of being detected and dying during the supplemental period (R′) was Age + Time and the rate of dead recoveries (r) was fixed at 0.

S	F	K	ΔAICc	w_i	Deviance
HY:Year + Age	Age	22	0.00	0.54	555.07
Age+Year	Age	22	3.17	0.11	558.25
HY:Year + Age	Age + Year	27	3.20	0.11	548.09
Age	Age + Year	21	3.93	0.08	561.03
Age	Age	16	4.83	0.05	572.07
HY:Year + Age	HY:Year + Age	27	5.48	0.03	550.37
Age*Year	Age	28	5.69	0.03	548.54
Age	HY:Year + Age	21	6.91	0.02	564.02
Age + Year	Age + Year	27	7.69	0.01	552.59
Age + Year	HY:Year + Age	27	8.46	0.01	553.35

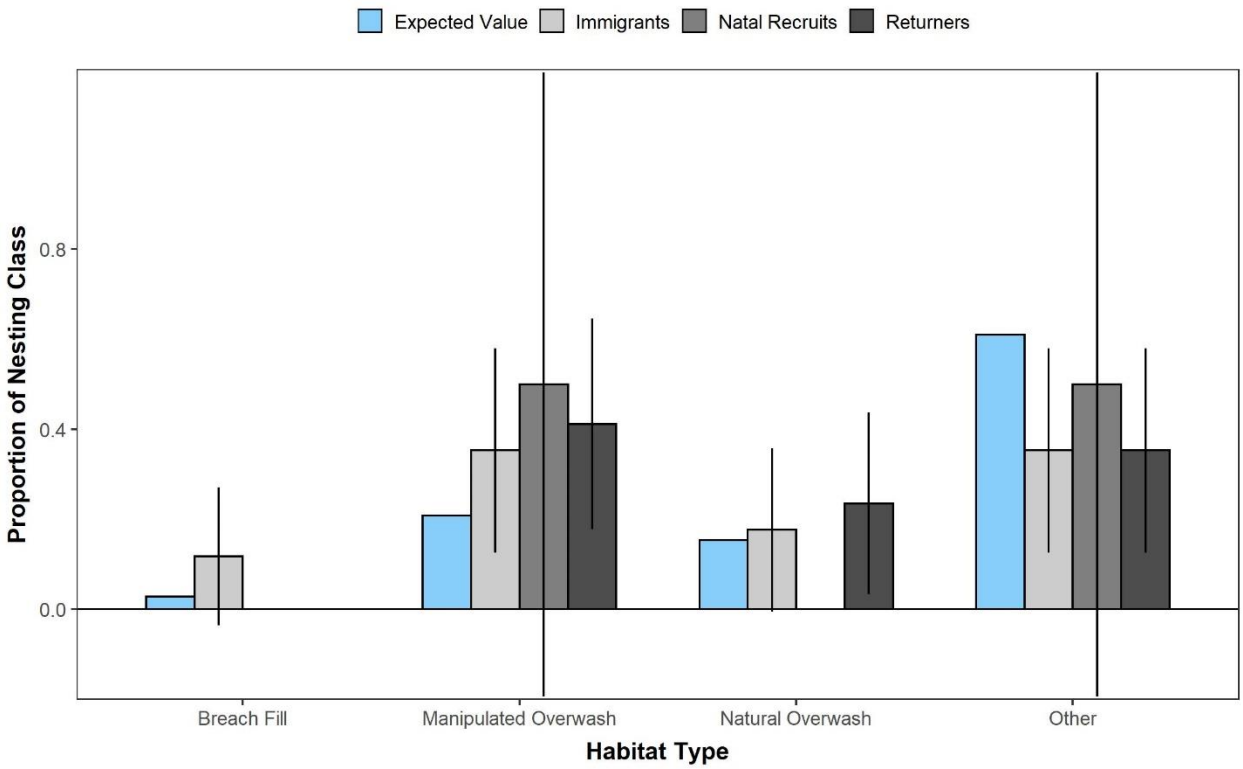


Figure D-1. Habitat used by immigrants, natal recruits, and returners in 2014 in relation to the expected usage based on availability of breach fill, manipulated overwash, natural overwash, and other habitats. Error bars represent the 95% confidence intervals and selection is evident when error bars do not overlap the expected habitat use value. (Immigrants: $\chi^2 = 8$, $P = 0.24$. Natal Recruits: $\chi^2 = 4$, $P = 0.26$. Returners: $\chi^2 = 12$, $P = 0.21$.)

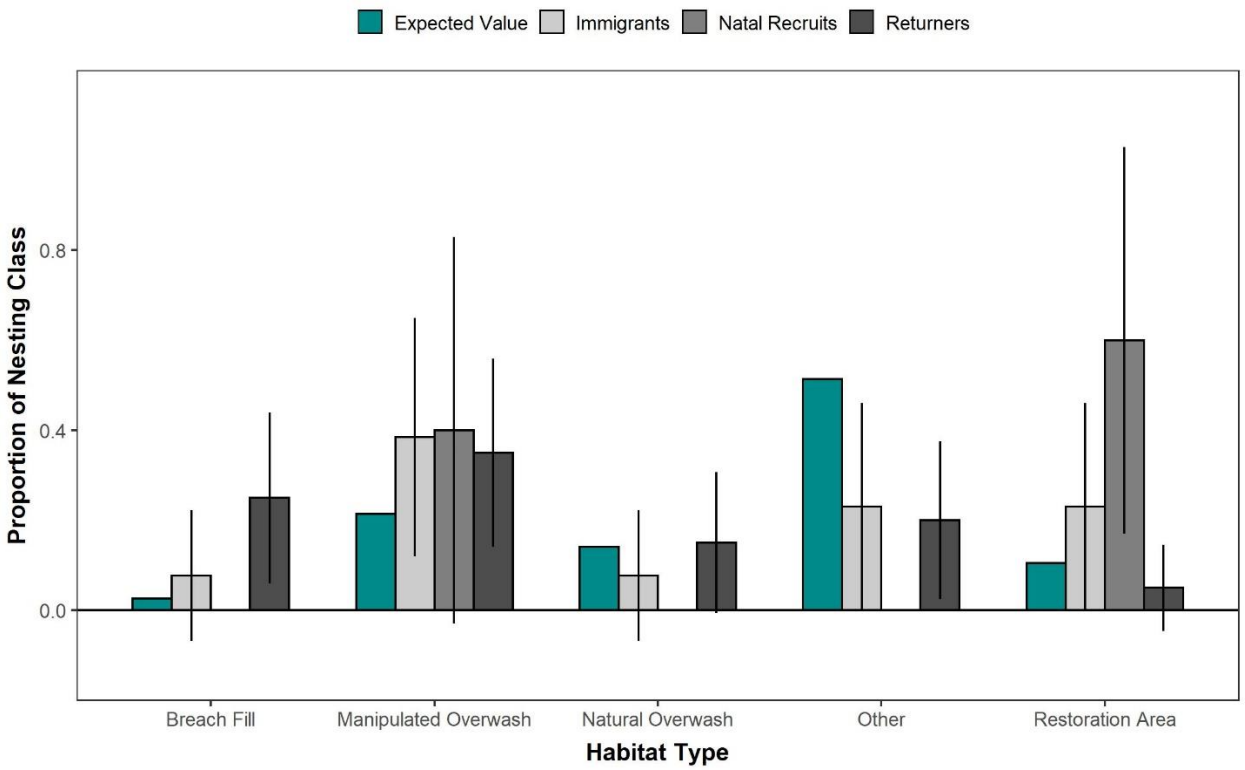


Figure D-2. Habitat used by immigrants, natal recruits, and returners in 2015 in relation to the expected usage based on availability of breach fill, manipulated overwash, natural overwash, other, and restoration area habitats. Error bars represent the 95% confidence intervals and selection is evident when error bars do not overlap the expected habitat use value. (Immigrants: $\chi^2 = 7.47$, $P = 0.11$. Natal Recruits: $\chi^2 = 15.83$, $P = 0.003$. Returners: $\chi^2 = 45.53$, $P < 0.001$.)

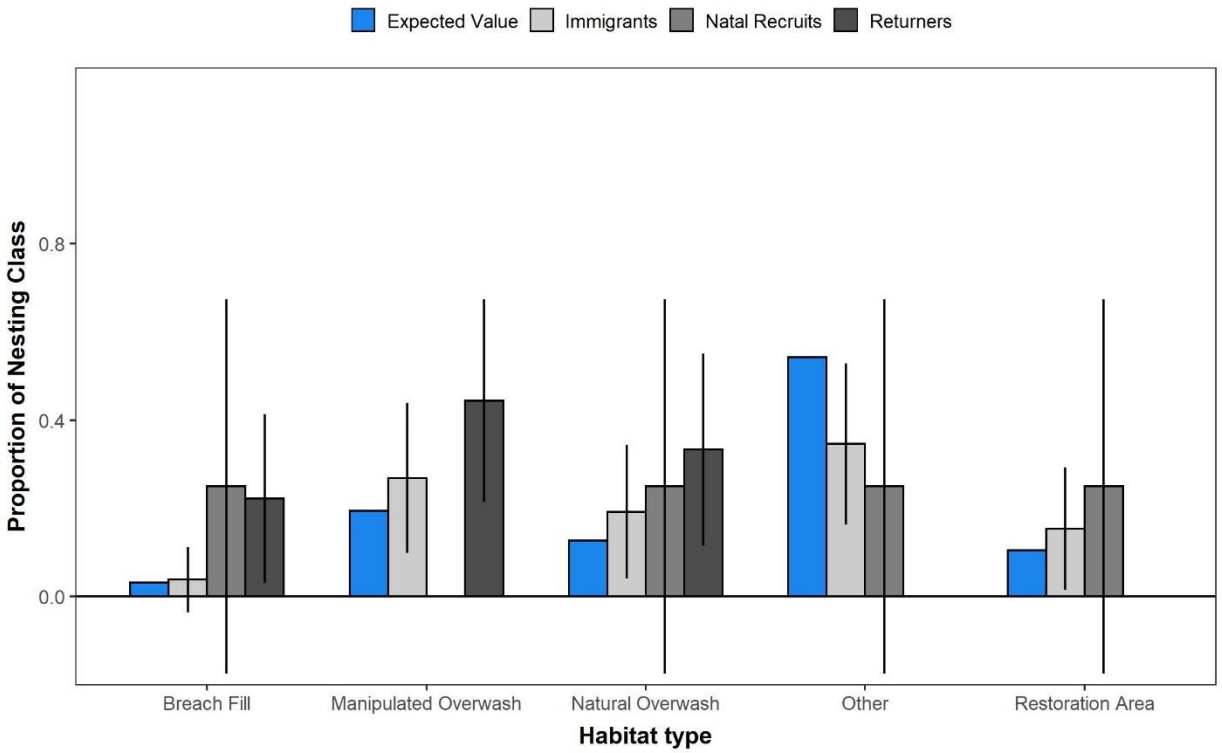


Figure D-3. Habitat used by immigrants, natal recruits, and returners in 2016 in relation to the expected usage based on availability of breach fill, manipulated overwash, natural overwash, other, and restoration area habitats. Error bars represent the 95% confidence intervals and selection is evident when error bars do not overlap the expected habitat use value. (Immigrants: $\chi^2 = 4.12$, $P = 0.39$, Natal Recruits: $\chi^2 = 8.74$, $P = 0.068$. Returners: $\chi^2 = 44.25$, $P < 0.001$.)

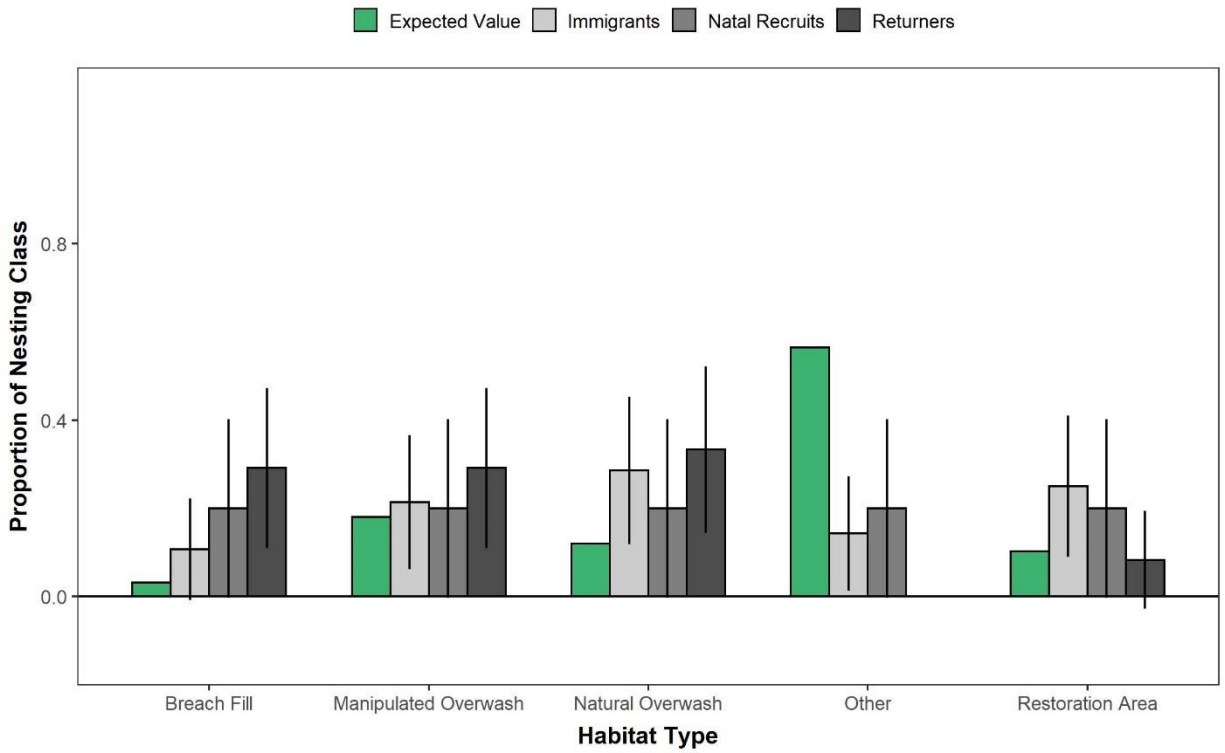


Figure D-4. Habitat used by immigrants, natal recruits, and returners in 2017 in relation to the expected usage based on availability of breach fill, manipulated overwash, natural overwash, other, and restoration area habitats. Error bars represent the 95% confidence intervals and selection is evident when error bars do not overlap the expected habitat use value. (Immigrants: $\chi^2 = 19.23$, $P < 0.001$. Natal Recruits: $\chi^2 = 19.23$, $P < 0.001$. Returners: $\chi^2 = 75.83$, $P < 0.001$.)

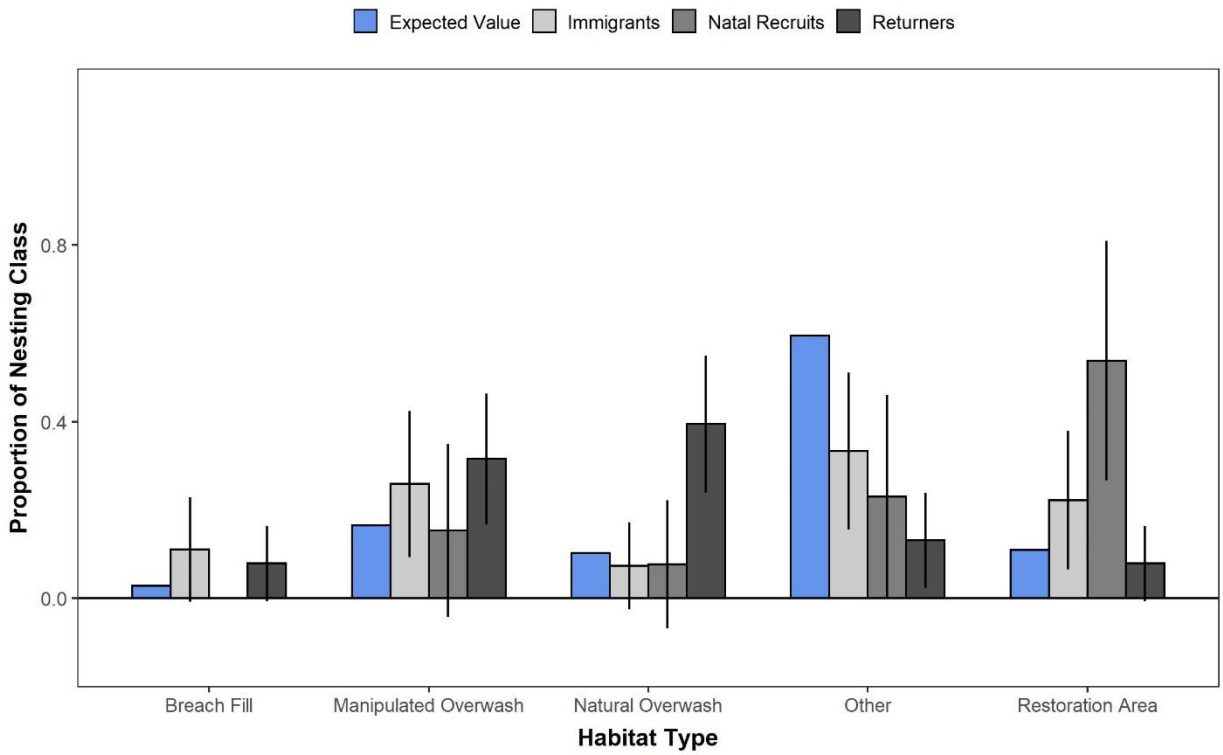


Figure D-5. Habitat used by immigrants, natal recruits, and returners in 2018 in relation to the expected usage based on availability of breach fill, manipulated overwash, natural overwash, other, and restoration area habitats. Error bars represent the 95% confidence intervals and selection is evident when error bars do not overlap the expected habitat use value. (Immigrants: $\chi^2 = 14.56$, $P = 0.006$. Natal Recruits: $\chi^2 = 25.11$, $P < 0.001$. Returners: $\chi^2 = 54.48$, $P < 0.001$.)

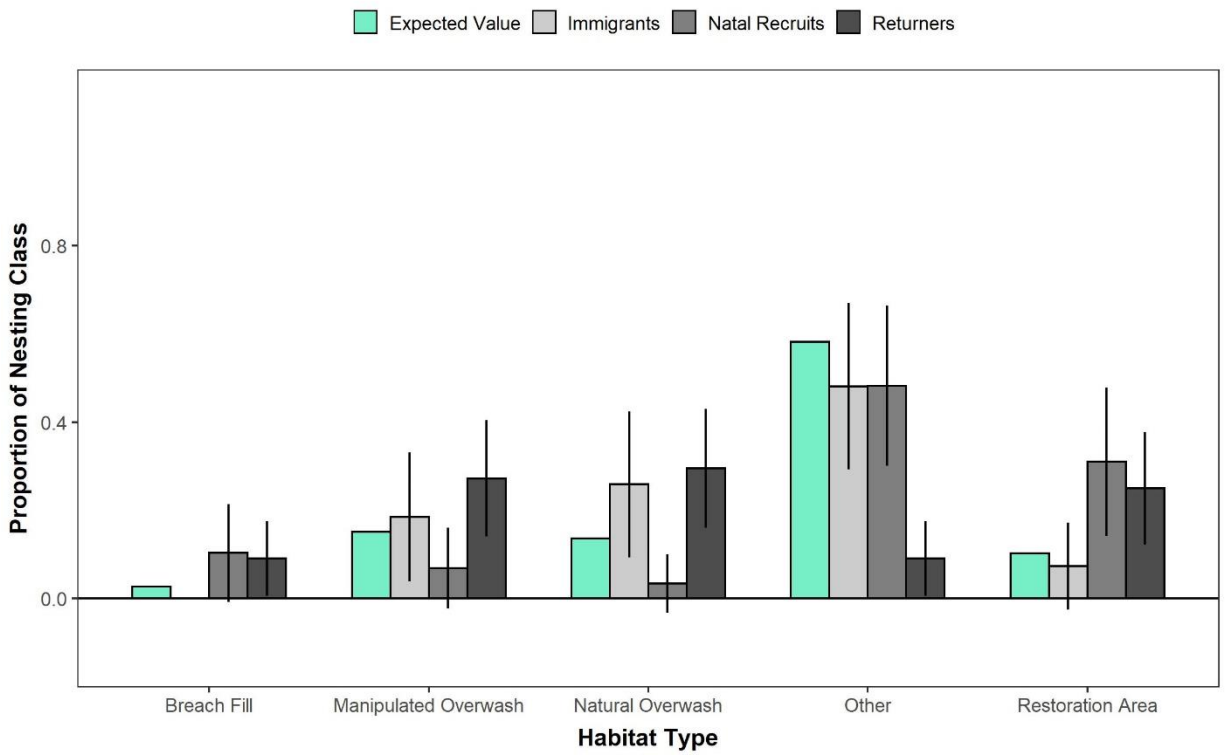


Figure D-6. Habitat used by immigrants, natal recruits, and returners in 2019 in relation to the expected usage based on availability of breach fill, manipulated overwash, natural overwash, other, and restoration area habitats. Error bars represent the 95% confidence intervals and selection is evident when error bars do not overlap the expected habitat use value. (Immigrants: $\chi^2 = 4.61$, $P = 0.33$. Natal Recruits: $\chi^2 = 22.35$, $P < 0.001$. Returners: $\chi^2 = 46.50$, $P < 0.001$.)

Appendix E. Supplementary material for Part 2, Section 8: Compare prey densities at several habitat types in 'Effectiveness monitoring and targeted studies in support of FIMI conservation measures.'

Response of Piping Plover Prey to Habitats Created by Hurricane Sandy

Final Report for 2013 and 2015

Submitted to

James Fraser

Virginia Tech

Wildlife Conservation

Blacksburg, VA 24061

Submitted by

Denise Sanger, Stacie Crowe, Sharleen Johnson, Martin Levisen

South Carolina Department of Natural Resources

Marine Resources Research Institute

217 Fort Johnson Road

Charleston, SC 29412

Introduction

The piping plover (*Charadrius melodus*) is a North American shorebird which is currently federally listed as threatened and endangered (USFWS 1985). Piping plover breeding grounds are distributed along the Atlantic Coast from Canada to North Carolina, the Northern Great Plains, and the Great Lakes (Johnson and Baldassarre 1988). Populations in all breeding regions are affected by a wide range of natural and anthropogenic factors such as habitat loss due to storm activity and coastal development, predation, and human disturbance (Plissner and Haig 2000).

Recent studies have found that physical habitat changes, such as sediment composition and tidal levels, may be associated with changes in infaunal macroinvertebrate distributions and subsequent reductions in shorebird utilization of the affected habitat (Peterson et al. 2006; Spruzen et al. 2008; Bergquist et al. 2011). Some studies

have suggested that loss or reductions in the densities of specific prey items can adversely affect the survival of a wide range of foraging shorebirds (Goss-Custard 1995; Stillman et al. 2005; West et al. 2007). Piping plover are visual predators whose dietary prey include infaunal and epifaunal marine invertebrates such as polychaete worms, crustaceans, and bivalve mollusks as well as insects (Nicholls 1989; Staine and Burger 1994; Zonick and Ryan 1996). Bergquist et al. (2011) investigated macroinfaunal invertebrate communities in piping plover foraging overwintering and migratory habitats and determined several organisms that may comprise the specific prey group of piping plover in South Carolina. Little evidence is available to determine the relationships between habitat modification, invertebrate prey communities and piping plover breeding-ground behavior.

In response to breaches created by Hurricane Sandy on Fire Island and Westhampton Island, NY in October 2012, a Breach Contingency Plan (USACE 1996) was implemented by the US Army Corps of Engineers (USACE) to evaluate and/or repair the damaged shoreline. Two of the breaches, including portions of the Narrow Bay West area and Cupsogue Beach County Park, were immediately filled which closed the breaches. The beaches of Fire Island and Westhampton Island, NY provide critical nesting habitat for the federally threatened/endangered piping plover (DeRose-Wilson et al. 2013).

The Virginia Tech Shorebird Project has been studying piping plover populations and their associated nesting/foraging habitat at Fire Island and Westhampton Island for close to 15 years. The goal of the most recent study was to better understand the effects of breaches and breach closures on piping plover populations and the invertebrate prey communities in their feeding habitats. Virginia Tech researchers collaborated with South Carolina Department of Natural Resources' (SCDNR) Environmental Research Section (ERS) on this study to assess the infaunal macrobenthic community. SCDNR has conducted previous research to investigate the potential prey communities of piping plover in overwintering and migratory habitats in South Carolina (Bergquist et al. 2011, Coastal Eco-Group, Inc. 2014a; 2014b; 2016). This report provides a summary of the Fire Island and Westhampton Island, NY infaunal macrobenthic community, as part of Virginia Tech's larger monitoring and assessment program on piping plover.

Methodology

The study area consisted of a 17.4 km stretch of barrier islands on Fire Island and Westhampton Island, NY. Piping plover nesting and foraging habitat in the area consists of sandy beaches backed by dunes and shoals, as well as habitat created by Hurricane Sandy. In 2013, intertidal habitats were sampled for macrobenthic infaunal invertebrates and sediment grain size at ten sites along the bay side of the islands (Table 1). The ten habitats sampled included two flood shoal sites - Little Flood Shoal (LFS) and Big Flood Shoal (BFS); one beach site - Cupsogue East (CSE); two filled breach sites - Narrow Bay West T1 (NBWT1) and Cupsogue West (CSW); three overwash sites - Bridge Overwash (BRO), Pattersquash Overwash (PSO), and Narrow Bay West T2 (NBWT2); and two inlet sites - Old Inlet West (OIW) and Old Inlet East (OIE) (Figures 1-5). Sampling occurred twice at each site approximately 0.5-1 month apart in June and July, 2013. Samples were collected within an hour before or after low tide and. Marine invertebrate sampling was conducted concurrently with bird and nest surveys.

In 2015, six sites were chosen for sampling: Cupsogue Mud Flats (CMF), Cupsogue Sandbar (CSB), Mussel Shoal (MUS, or Flood Shoals as identified in DeRose-Wilson et al. 2014), Narrow Bay West (NBWT1 and NBWT2), and Pattersquash Overwash (PSO) (Table 1, Figure 1) (DeRose-Wilson et al. 2014). CMF was identified as a mudflat (Figure 6). CSB and MUS were identified as flood shoals (Figures 7-8). NBWT1 was identified as a filled breach (Figure 9). NBWT2 and PSO were identified as overwash areas (Figures 9-10). Sampling occurred within an hour before or after low tide in April of 2015. Marine invertebrate sampling was conducted concurrently with bird and nest surveys. Only NBWT1, NBWT2 and PSO were sampled in both years.

At each site in 2013 and 2015, ten or thirty, respectively, sampling stations were established along a transect parallel to the water in the intertidal zone. GPS coordinates were obtained for each station. One push core (7.6 cm diameter X 10 cm deep; 0.004536 m^2 surface area) was collected at each station for characterization of the potential macrobenthic infaunal community. In 2013, an additional core was collected for the identification of sediment grain size.

The contents of each macrobenthic community core were washed through a 0.5 mm mesh sieve, and organisms and sediment retained on the sieve were preserved in a buffered solution of 10-20% formalin/seawater containing rose bengal stain. Benthic organisms were sorted from retained material under a magnifying lens and stored in alcohol.

In addition, fecal samples were opportunistically collected throughout Fire Island and Westhampton Island in 2013 (n=7) and 2014 (n=37) (Figure 11). GPS coordinates were obtained for most of the fecal sampling sites. Fecal samples were preserved in alcohol.

Potential prey organisms from macrobenthic community sediment cores and fecal samples were identified and enumerated by experienced SCDNR taxonomists. Organisms from the sediments were identified to the family level except for organisms from the classes of Insecta and Ostracoda, the order of Actinaria (sea anemones), and the phylum Platyhelminthes (flatworms). Meiofaunal species (such as nematodes and copepods that are not well quantified using a 0.5 mm sieve) were not recorded. Organisms from the fecal samples were identified to family, if possible. Fecal sample identification is very difficult and generally does not result in quantification of the organisms. Every attempt was made to quantify the number of organisms of each taxa in the fecal samples, if possible, but we have chosen to discuss the results as presence/absence of the taxon. A reference collection was created for the project and is maintained by the Environmental Research Section at the SCDNR Marine Resources Research Institute (Charleston, SC).

All sediment benthic samples processed for sorting (2015 only) and taxonomy (2013 and 2015) were subjected to a rigorous quality assurance (QA) process. Samples were processed in batches of ten with one random sample out of every ten samples being re-processed by an experienced SCDNR staff member (10% QA). For sorting and identification, if less than 90% of the organisms were sorted from the sieve-retained material (and more than one organism remained in the sample) or if more than 10% of the specimens in a sample were mis-

identified, the entire batch was re-processed until QA was passed. Sediment data was provided by Virginia Tech.

All data are stored in a Microsoft Access relational database. Multivariate statistics were used to assess the macrobenthic infaunal community. The study dataset was divided into three separate groups for analysis including: (1) total abundance of all organisms collected, (2) a group of taxa which are thought to be important prey items or resources for piping plover; and (3) a group of taxa which are thought to be important prey items or resources for piping plover excluding horseshoe crab eggs and larvae. The prey items for piping plover were selected based on Bergquist et al. (2011), fecal sample results from this study, and stomach sample results from piping plover in the northeast. The prey resources for this study were insects; horseshoe crab eggs and larvae; three large, errant polychaete families (Oenonidae, Glyceridae, Nereididae); two amphipod families (Haustoriidae and Corophiidae), and the clam *Donax* sp. The Corophiidae family was added for this report due to the finding of Corophiidae in the digestive tract of a piping plover from Maine (Bimbi and Levisen unpublished data). This family has not been considered prey in any previous reports.

Non-metric multidimensional scaling (MDS), hierarchical cluster analysis (CLUSTER), and similarity profile permutation tests (SIMPROF) were performed using Primer v6.1 (Primer-E Ltd. 2009). All of these tests use the taxonomic composition and relative abundances of macroinvertebrate infauna collected at each sampling site, and then evaluate similarities and differences among these communities. For analyses involving the total macroinvertebrate infaunal community, the species matrix consisted of all taxa identified during the entire study. For analyses only involving taxa considered to serve as prey for piping plover, the species matrix consisted of all taxa listed above. Analyses were performed on site level data: the abundance of each taxa was averaged across stations within each site. For 2013, this included both sampling events. Site level data were square-root transformed, a Bray-Curtis similarity resemblance matrix was calculated from the transformed data, and MDS analysis was performed on the resemblance matrix using 50 restarts (Clarke and Gorley 2006). Hierarchical cluster analysis (CLUSTER) was implemented on the same resemblance matrix to identify site community groupings, and CLUSTER similarity results were overlaid on the MDS plot (Clarke and Gorley 2006). In the MDS plots, similar communities appear closer together, while more distinct communities plot farther apart. Points that fall within a specific CLUSTER-based similarity grouping represent site communities that share at least that level of similarity with the other communities in that grouping.

To test for significant differences between the groups of communities identified in a CLUSTER analysis, similarity profile permutation tests (SIMPROF) with 999 simulations were applied (Clarke and Gorley 2006). For the SIMPROF test, the p-value was considered significant at $\alpha=0.05$; groupings of site communities that were determined to be significantly different from others are referred to as “clusters”.

Results

Fecal samples from Fire Island and Westhampton Island were found to contain representatives from phyla Arthropoda and Annelida (Table 2). Every sample contained an identifiable organism or fragment of an organism with at least one insect taxon observed in all 44 fecal samples. Marine-related taxa were found in only 14 (33%) of the samples. Six identifiable families and one unidentifiable Insecta group were observed in the

fecal samples. The Insecta group was found in 24 of the 44 samples. The insect families identified included two samples with Formicidae (ants), 22 samples with Coleoptera (beetles), and one each with Lepidoptera (butterflies/moths), Scarabaeidae (scarab beetles), Diptera (true flies), and Curculionidae (true weevils). Other arthropods found in the fecal samples included Limulidae (horseshoe crab) eggs (7 samples) and larva (1 sample) as well as evidence of Acarina (spider, 1 sample), and Natantia (shrimp-like decapod, 1 sample). Two polychaete families (Nereididae and Sabellidae) and a nondescript polychaete group were also observed in the fecal matter (Table 2). Nereididae is one of the polychaete families previously identified as prey items for piping plover in South Carolina (Bergquist et al. 2011) and was included in the prey items from this study. Worms from the family Sabellidae are relatively small tube-building polychaetes and not likely to be a major food resource. Therefore, they have not been included as a prey resource for this study.

If each piping plover fecal sample is considered separately then 34%, 39%, and 20% of the samples contained evidence of only one, two or three different taxa, respectively. Three fecal samples contained more than three taxa, with a maximum of four, five, and six taxa observed. These results may not comprehensively reflect piping plover prey resources as soft-bodied organisms were likely under-represented due to digestive processes. Sample sizes were different in 2013 and 2014; however, all seven of the samples from 2013 contained at least three taxa including the two samples that contained five and six taxa.

In 2013, a total of 58 macrobenthic taxa were identified in the 177 samples collected from the ten areas over various dates during June and/or July (Table 3). The sites and dates varied with regard to average density with two of the lowest values (154/m^2 and 397/m^2) observed at the overwash/inlet sites OIW and OIE from 7/8 and 6/23, respectively, and also at the LFS site from 7/19. The highest values (79,034/m^2 and 30,644/m^2) were observed at the overwash sites NBWT2 and PSO on 6/28 and 7/28, respectively (Table 3, Figure 12). At the NBWT2 site on 6/18, this was driven by a high abundance (78,461/m^2) of horseshoe crab eggs and larvae (HSC). HSC were also found to be present at BRO, PSO, BFS, LFS, CSW, and NBWT1. HSC abundances were in the thousands at NBWT2, LFS, and NBWT1 and were always higher during the first sampling event compared to the second sampling event at each site. The proportion of HSC at a site was a significant contribution to the community, particularly at BRO on 6/17, NBWT1 on 6/18, NBWT2 on 6/18, and LFS on 7/4 (Figure 13).

In June and July of 2013, the number of taxa was highest at the PSO (overwash) site on 7/28, with 24 unique taxa (family/class level) observed, followed by the PSO (overwash) site on 6/20 with 16 unique taxa. The number of unique taxa at NBWT1 and NBWT2 ranged from 5-15, while the other sites sampled were generally 10 or less (Table 3).

In April of 2015, a total of 51 macrobenthic taxa were identified in the 198 samples collected from the six areas (Table 3). The sites varied with regard to average density with the lowest values (3,743/m^2 and 7,216/m^2) observed at the two flood shoal sites (CSB and MUS, respectively) and the highest values (24,535/m^2 and 26,872/m^2) observed at the two overwash sites (PSO and NBWT2, respectively) (Table 3, Figure 12). The average densities at the mud flat site (CMF) and the filled breach site (NBWT1) were found to be 17,679/m^2 and 21,138/m^2, respectively. HSC abundances were rare (7/m^2) with only 2 out of 6 sites having any present.

In April of 2015, the number of taxa was highest at the CMF (mud flat) site, with 38 unique taxa (family/class level) observed, followed by the PSO (overwash) site with 30 unique taxa. NBWT1 and NBWT2 had the same number of unique taxa with 25 each. The number of unique taxa at MUS and CSB (flood shoal) sites was similar with 18 and 19, respectively (Table 3).

In June and July of 2013, the average densities of macrobenthic prey items (with HSC) for piping plover ranged from a high at NBWT2 of 78,704/m^2 on 6/18 compared to sites NBWT1 and OIW (on 7/16 and 7/8, respectively) with 0/m^2 (Table 4, Figure 14). The next highest site was LFS with a value of 26,367/m^2 on 7/4, followed by much smaller densities for the other sites (ranging from 44-7,848/m^2). HSC are well documented as a food source for shorebirds; however, the significant contribution and ephemeral nature of this food source led us to also summarize the prey item abundances excluding HSC to better understand the potential longer term food availability. If HSC were excluded, then four sites had no piping plover prey resources (LFS – 7/4; OIW – 7/18; NBWT1 – both dates; BRO – both dates) (Table 4, Figures 14-15). If HSC were excluded then the amphipods comprised 100 % of the prey taxa at all sites except for PSO on both sampling dates when polychaete were observed and OIW on 7/19 when the bivalve *Donax* sp. was observed (Figure 15).

In April of 2015, the average densities of piping plover prey items ranged from a high at NBWT1 of 4,636/m^2, driven by corophid amphipods, compared to the lowest at CSB and MUS of 33/m^2 and 100/m^2, respectively (Table 4, Figure 14). The next highest site was PSO with a value of 2,505/m^2, followed by similar densities of 935/m^2 and 862/m^2 at NBWT2 and CMF, respectively. The amphipods comprised the highest proportion of prey taxa at all but the CMF site; however, polychaetes also contributed to the proportion at CMF, PSO, NBWT1, and NBWT2 (Figures 14-15). Large polychaetes have been found to have higher caloric value than amphipods (USACE 1997).

The two years were sampled during different seasons and therefore any differences observed may be related to season or year differences. One of the differences between the two data sets was the larger presence of HSC in the 2013 compared to 2015. This was likely a seasonal effect considering HSC spawn from May to July in NY. In addition, the total abundance of the macrobenthos was generally higher in 2015 (when HSC were excluded) compared to 2013 except for the PSO site which showed similar abundance levels between the two years. Another difference was the larger presence of the polychaete prey families at the sites sampled in 2015 compared to the sites sampled in 2013. These polychaete families were found in all sites except MUS in 2015 and only one site (PSO) in 2013.

Multivariate analysis of the total macroinvertebrate infaunal communities showed the NBWT1_15, NBWT2_15, PSO_13, PSO_15, and CMF_15 were more similar to each other and clustered separately from all of the other 2013 sites as well as MUS_15 and CSB_15 (Figure 16). OIW_13 clustered away from all other sites indicating the community was unique at this location.

As observed among the total macroinvertebrate infauna communities, the piping plover prey communities (with HSC) at NBWT1_15, NBWT2_15, PSO_13, PSO_15, and CMF_15 were clustered separately from all of the sites-years (Figure 17). Two groups were also apparent with the NBWT1_13, NBWT2_13, BRO_13, and LFS_13 grouping together away from the remaining sites but this was not statistically significant. OIW_13 was removed from the analysis due to no prey items were present.

When horseshoe crab eggs and larvae were excluded from the prey communities then three sites did not contain any potential prey items and were removed from the analysis (BRO_13, OIW_13, and NBWT1_13). Two significant clusters and groups were apparent in the analysis. The NBWT1_15, NBWT2_2015, PSO_13, PSO_15, and CMF_15 sites had more similar communities and clustered away from all of the other sites (Figure 18).

Sediment grain size data was available for 2013 and in general there did not appear to be a clear relationship between the levels of pebbles, silt or grain size phi and habitat type or benthic community distribution (graphs not shown). This may be due to the small distribution in values.

Discussion

Piping plover distributions may be related to a wide range of variables including parental and social behavior, weather conditions, nesting success, disturbance, and predation (e.g., DeRose-Wilson et al. 2014; Murchison et al. 2016). Marine macrobenthic communities have been found to be related to a wide range of variables including location within the intertidal area, sediment composition, perturbation (e.g., wave energy), and predation. All of these variables are likely influencing both the macrobenthic community and the piping plover usage of an area. Understanding the availability of the macrobenthic community prey resources in relation to bird usage is critical to assessing habitat quality.

Big Flood Shoal, Little Flood Shoal, Old Inlet West, Old Inlet East and Mussel Shoal are all located in the same section of the Fire Island National Seashore on the Bellport Bay side. Mussel Shoal was sampled in 2015 and the other sites were sampled in 2013. The total macrobenthic community and prey resource abundances were low. These sites were classified as flood shoal and inlet habitat types. The other flood shoal sampled was Cupsogue Sandbar in 2015 which had similarly low levels of marine invertebrates and prey resources. These flood shoal and inlet sites were probably not good foraging areas for piping plover except possibly during horseshoe crab spawning time periods at LFS site.

The two filled breach sites (Narrow Bay West – Filled Breach and Cupsogue West) were both sampled in 2013 after they were filled following Hurricane Sandy. Both of these sites had no to very low macrobenthic prey resources for piping plover, except for the availability of horseshoe crab eggs at NBWT1. Therefore, these sites were probably not good foraging areas in June/July of 2013. However, the availability of prey resources was similar to other sites that were not filled in 2013 and it is difficult to know if the filling of the sites had a direct impact on the macrobenthic community.

Cupsogue Mud Flats, Narrow Bay West – Filled Breach, Narrow Bay West – Overwash, and Pattersquash Overwash in April of 2015 as well as Pattersquash Overwash in June/July of 2013 were found to have high macrobenthic prey resources for piping plover, particularly with the presence of large polychaetes and amphipods. Cupsogue East, the only beach site sampled in June/July of 2013, also had available macrobenthic prey resources for piping plover, namely the presence of amphipods. All of these sites were probably good foraging areas for piping plover.

Three sites including Narrow Bay West – Filled Breach, Narrow Bay West – Overwash, and Pattersquash Overwash were sampled in both 2013 and 2015. The total abundance and total community composition were very similar between the two years at Pattersquash Overwash despite the different seasons. The levels of available prey resources were different between 2013 and 2015. The macrobenthic prey abundances were higher in April of 2015 than in June/July of 2013; however, the prey community composition was similar between the two years. The total abundance, total community composition, prey resource abundance, and prey resource composition at Narrow Bay West – Filled Breach and Narrow Bay West – Overwash were different between the two years. Both sites had lower total abundances and prey resource abundances in June/July of 2013 compared to the abundances observed in April of 2015. The lower abundances in 2013 may be related to the filling of the breach at Narrow Bay West – Filled Breach site after Hurricane Sandy; however, the same pattern was observed at the Narrow Bay West – Overwash site which is considered an overwash site but is located immediately adjacent to Narrow Bay West – Filled Breach.

The invertebrates observed in piping plover fecal matter comprised representatives from two phyla, Arthropoda and Annelida. The arthropods were the dominant organisms (or fragments of organisms) observed, with insects being the most numerous type of arthropods. Horseshoe crab eggs and larvae, spiders, and shrimp-like decapods were the other arthropods observed in the fecal samples. Only two families of polychaetes, Nereididae and Sabellidae, were observed in the fecal samples. In general, these organisms have been identified as piping plover prey resources in other studies (e.g., Staine and Burger 1994; Cuthbert et al. 1999; Bergquist et al. 2011; Cisek 2012). Previous studies have generally identified polychaete worms, insects, crustaceans and mollusks as prey resources. No evidence of mollusks was found in the fecal matter of this study; however, evidence of *Donax* sp. have been observed in fecal samples from overwintering and migrating piping plover in SC (Bergquist et al. 2011).

Arthropods, particularly the insect taxa are likely over-represented in comparison to soft-bodied annelid taxa which may be under-represented in this study as many of their identifiable characteristics are destroyed by digestive processing. In addition, avian gizzards likely destroy polychaete setae (the only structures to survive digestion for some polychaete species) and amphipod integument beyond recognition. Future efforts to assess available piping plover stomach contents as well as using new techniques such as DNA barcode sequencing in combination with traditional taxonomic identifications will provide further information about their preferred prey resources and all potential prey resources. The macrobenthic behavior of the different species may also provide important information for identifying prey resources for piping plover considering they are visual feeders.

The impact on the abundances of macrobenthic organisms by foraging shorebird species including piping plover is unknown; however, it has the potential to be significant. For example, three Dunlin (*Calidris alpina*) stomachs obtained in SC were found to contain a significant number of nereid jaws, with 354 and 469 jaw halves observed in two of the stomachs (Sanders and Levisen unpublished data). This level of predation has the potential to be significant when considering the highest number of nereids, 776 per square meter, were found at the Cupsogue Mud Flats site. In other systems, large polychaetes have been found to have higher caloric value than amphipods (USACE 1997).

Staine and Burger (1994) observed the highest feeding rates of piping plover in their study were found around low tide and feeding occurred more frequently in areas with polychaetes than in other areas where polychaetes were less abundant. They did not assess the species or family of polychaetes available. Piping plover have also been observed more frequently in areas with higher numbers of large, errant polychaete abundance, particularly the Nereididae, Oenonidae, and Glyceridae, on the overwintering grounds of SC (Bergquist et al. 2011; Sanger unpublished data; Coastal Eco-Group, Inc. 2016). Therefore, large polychaetes are probably the preferred prey followed by haustorid and corophid amphipods.

In summary, macrobenthic prey resources for piping plover, particularly polychaetes and amphipods, were found to be highest in April of 2015 at Cupsogue Mud Flats, Narrow Bay West – Filled Breach, Narrow Bay West – Overwash, and Pattersquash Overwash as well as Pattersquash Overwash in June/July of 2013. Cupsogue East in June/July of 2013 had prey resources, particularly amphipods, available for piping plover. These sites appear to have been good foraging areas for piping plover. The other sites-years were found to have low prey resources for piping plover except for the ephemeral availability of horseshoe crab eggs and larvae at Little Flood Shoal, Narrow Bay West – Filled Breach, Narrow Bay West – Overwash, and Bridge Overwash sites in June/July of 2013. The other sites (Cupsogue Sandbar in 2015, Mussel Shoal in 2015, Big Flood Shoal in 2013, Cupsogue West in 2013, Old Inlet East in 2013, and Old Inlet West in 2013) had very low available prey resources and were probably not good foraging areas for piping plovers.

Literature Cited

Bergquist, D.C., M.V. Levisen, and L. Forbes. 2011. Kiawah Island East End Erosion and Beach Restoration Project: Survey of Changes in potential macroinvertebrate prey communities in piping plover (*Charadrius melodus*) foraging habitats. Final Report prepared by the South Carolina Department of Natural Resources, Marine Resources Research Institute, Charleston, SC for the Town of Kiawah Island. 64 pp.

Cisek, C. 2013. Analysis of non-breeding piping plover (*Charadrius melodus*) foraging behavior and the association between foraging habitat characteristics and foraging behavior. MS Thesis. College of Charleston. Charleston, SC.

Clarke, K.R. and R.N. Gorley. 2006. PRIMER v6: User Manual/Tutorial. PRIMER-E, Plymouth.

Coastal Eco-Group, Inc. 2014a. Piping plover foraging habitat evaluation, Year 1 (2012-2013) post construction monitoring results. Report submitted to Town of Hilton Head, SC. 58 pp.

Coastal Eco-Group, Inc. 2014b. Piping plover foraging habitat evaluation, Year 2 (2013-2014) Post-Construction Monitoring Results. Report submitted to Town of Hilton Head, SC. 77 pp.

Coastal Eco-Group, Inc. 2016. Port Royal Sound Shoreline Restoration and Stabilization Project: Piping plover foraging habitat evaluation Year 3 (2014-2015) Post-construction monitoring results. Report prepared by Coastal Eco-Group, Inc., Deerfield Beach, FL for the Town of Hilton Head Island. 95 pp.

Cuthbert, F.J., B. Scholtens, L.C. Wemmer, and R. McLain. 1999. Gizzard contents of piping plover chicks in Northern Michigan. The Wilson Bulletin 111(1): 121-123.

DeRose-Wilson, A., K. Sloan, J.D. Fraser, D.H. Catlin, and S.M. Karpanty. 2013. Response of piping plovers and their invertebrate prey to habitats created by Hurricane Sandy. Annual Report prepared by the Virginia Tech Shorebird Project. 47 pp.

DeRose-Wilson, A., J.D. Fraser, D.H. Catlin, S.M. Karpanty, and S.J. Ritter. 2014. Response of piping plovers and their invertebrate prey to habitats created by Hurricane Sandy. Annual Report prepared by the Virginia Tech Shorebird Project. 43 pp.

Goss-Custard, J.D., R.T. Clarke, S.E.A., Durell, le V. dit, R.W.G. Caldow, and B.J. Ens. 1995. Population consequences of winter habitat loss in a migratory shorebird. II. Model predictions. Journal of Applied Ecology 32: 337-351.

Johnson, C.M. and G.A. Baldassarre. 1988. Aspects of the wintering ecology of piping plovers in coastal Alabama. Wilson Bulletin 100(2) pg. 214-223.

Nicholls, J. 1989. Distribution and other ecological aspects of piping plovers (*Charadrius melodus*) wintering along the Atlantic and Gulf coasts. Auburn Univeristy. Auburn, Alabama.

Murchison, C.R., Y. Zharikov, and E. Nol. 2016. Human activity and habitat characteristics influence shorebird habitat use and behavior at a Vancouver Island migratory stopover site. Environmental Management 58: 386-398.

Peterson C.H., M.J. Bishop, G.A Johnson, L.M. D'Anna, and L.M. Manning. 2006. Exploiting beach filling as an unaffordable experiment: benthic intertidal impacts propagating upwards to shorebirds. Journal of Experimental Marine Biology and Ecology. 338(2): 205-21.

Plissner, J.H. and S.M. Haig. 2000. Viability of piping plover *Charadrius melodus* metapopulations. Biological Conservation 92: 163-173.

Spruzen, F., A.M.M. Richardson, and E.J. Woehler. 2008. Spatial variation of intertidal macroinvertebrates and environmental variables in Robbins Passage wetlands, NW Tasmania. Hydrobiologia 598: 325-342.

Staine, K. J. and J. Burger. 1994. Nocturnal foraging behavior of breeding piping plovers (*Charadrius melodus*) in New Jersey. Auk 111: 579–587.

Stillman, R.A., A.D. West, J.D. Goss-Custard, S. McGrorty, N.J. Frost, D.J. Morrisey, A.J. Kenny, and A. Drewitt. 2005. Predicting site quality for shorebird communities: a case study on the Humber Estuary, UK. Marine Ecology Progress Series 305: 203-217.

USACE. 1996. Fire Island to Montauk Island Long Island, NY Breach Contingency Plan: An evaluation of the benefit of rapid response to breaches along the Federally authorized project shoreline. Executive Summary and Environmental Assessment prepared by the US Army Corps of Engineers New York District North Atlantic Division. 582 pp.

USACE. 1997. Townsends Inlet to Cape May Inlet Feasibility Study, Volume 3, Appendices A through C and E through H. U.S. Army Corps of Engineers Philadelphia District.

USFWS. 1985. Endangered and threatened wildlife and plants; determination of endangered and threatened status for the piping plover. United States Fish and Wildlife Service. Federal Register 50: 50726-50734.

West, A.D., M.G. Yates, S. McGrorty, and R.A. Stillman. 2007. Predicting site quality for shorebird communities: A case study on the Wash embayment, UK. Ecological Modelling 202: 527-539.

Zonick, C. and M. Ryan. 1996. The ecology and conservation of piping plovers (*Charadrius melodus*) wintering along the Texas Gulf Coast. Annual Report. Department of Fisheries and Wildlife, University of Missouri, Columbia, Missouri. 49pp.

Table 1. Sampling site name, general habitat description, and transect length for macrobenthic sampling for each site sampled in the 2013 and 2015 macrobenthic study.

Name	Subsite Code	Description	Habitat Type	Habitat Type Year	Year Sampled	Transect Length (m)
Cupsogue Sandbar	CSB	West Beach and narrow strip of bayside intertidal zone at east end of the Cupsogue Beach County Park.	Flood shoal	2001	2015	333
Mussel Shoal	MUS	Flood shoal just west of the old inlet, on the bay side. A component of the Flood Shoals (FS) system.	Flood shoal	2012	2015	361
Big Flood Shoal	BFS	Flood shoal associated with old inlet. A component of the Flood Shoals (FS) system.	Flood shoal	2012	2013	522
Little Flood Shoal	LFS	Flood shoal associated with old inlet. A component of the Flood Shoals (FS) system.	Flood shoal	2012	2013	371
Cupsogue Mud Flats	CMF	Mud flats north of the Cupsogue parking lot, and south of the high bare island in Moriches Bay.	Mud flat	2001	2015	310
Cupsogue East	CSE	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	2012	2013	429
Narrow Bay West - Filled Breach	NBWT1	Small breach-fill area west of the Hazardous Area.	Filled breach	2012	2013 2015	289 322
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.	Filled breach	2012	2013	304
Narrow Bay West - Overwash	NBWT2	Overwash west of the Hazardous Area.	Overwash	2012	2013 2015	381 331
Pattersquash Overwash	PSO	Overwash west of Pattersquash East, has marshy spots, on the bay side.	Overwash	2012	2013 2015	1021 322
Bridge Overwash	BRO	Overwash in the Otis Pike Wilderness, just west of the NPS Visitor Center.	Overwash	2012	2013	117
Old Inlet East	OIE	Begins where the primary duneline ends and extends west to Old Inlet.	Inlet	2012	2013	302
Old Inlet West	OIW	Begins at west side of Old Inlet and extends to the point where large primary dunes begin.	Inlet	2012	2013	109

Table 2. The number of fecal samples containing each taxon observed in the fecal samples (n=44).

Category	Phylum	Taxon	# of Fecal Samples Containing Taxon
polychaete	Annelida	Nereididae	3
polychaete	Annelida	Polychaeta	2
polychaete	Annelida	Sabellidae	1
decapod (shrimp-like)	Arthropoda	Natantia (shrimp-like decapod)	1
horseshoe crab	Arthropoda	Limulidae egg	7
horseshoe crab	Arthropoda	Limulidae larva	1
arachnid	Arthropoda	Acarina	1
insect (no distinguishing parts)	Arthropoda	Insecta (insect like)	24
insect (ants)	Arthropoda	Formicidae	2
insect (beetles)	Arthropoda	Coleoptera	22
insect (butterflies/moths)	Arthropoda	Lepidoptera	1
insect (scarab beetles)	Arthropoda	Scarabaeidae	1
insect (true flies)	Arthropoda	Diptera	1
insect (true weevils)	Arthropoda	Curculionidae	1

Table 3. The average density (#/m^2) of each macrobenthic taxon collected for each study site and date collected. The samples from April are all from 2015 and the samples from June/July are all from 2013.

Phylum	Type of Organism	Taxon	Average Density (#/m^2) by Habitat Type, Site, and Sampling Date													
			Flood Shoal						Mud Flat	Beach		Filled Breach				
			CSB	MUS	BFS		LFS		CMF	CSE		NBWT1	CSW		NBWT1	
			4/14/15	4/16/15	7/4/13	7/19/13	7/4/13	7/19/13	4/14/15	6/24/13	7/17/13	4/15/15	6/24/13	7/17/13	6/18/13	7/16/13
			April		July				April	June/July		April	June/July			
Annelida	oligochaete	Enchytraeidae	154	7	0	0	0	0	3,768	287	0	20	0	44	0	0
		Tubificidae	20	94	132	88	44	0	281	2,116	0	4,296	198	22	287	441
	polychaete	Ampharetidae	20	7	0	0	0	0	67	0	0	0	0	0	0	0
		Capitellidae	73	67	0	0	22	0	120	0	0	147	0	0	110	1,190
		Chaetopteridae	0	0	0	0	0	0	7	0	0	7	0	0	0	0
		Cirratulidae	0	0	0	0	0	0	187	0	0	127	0	0	0	0
		Glyceridae	0	0	0	0	0	0	13	0	0	0	0	0	0	0
		Lumbrineridae	13	0	0	0	0	0	2,819	0	0	815	0	0	0	220
		Maldanidae	20	0	0	0	0	0	1,937	22	0	7	0	0	0	0
		Nephtyidae	0	27	0	0	44	0	0	0	0	20	0	0	0	0
		Nereididae	0	0	0	0	0	0	775	0	0	187	0	0	0	0
		Oenonidae	7	0	0	0	0	0	0	0	0	0	0	0	0	0
		Orbiniidae	301	73	132	331	22	44	94	132	0	615	0	66	0	88
		Paraonidae	67	20	0	0	0	0	20	0	0	7	0	0	0	0
		Pectinariidae	0	0	0	0	0	0	7	0	0	0	0	0	0	0
		Phyllodocidae	94	0	0	0	0	0	294	0	0	134	0	0	110	551
		Polychaeta	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Sabellidae	0	0	0	0	0	0	0	44	0	0	0	22	0	0
		Spionidae	0	13	0	44	44	0	735	0	0	107	0	0	0	22
		Syllidae	40	13	0	0	0	0	2,519	309	0	842	0	66	0	88
Arthropoda	amphipod	Ampeliscidae	0	0	0	0	0	0	20	22	0	0	0	0	0	0
		Aoridae	0	0	0	0	0	0	0	0	0	7	0	0	0	44
		Corophiidae	0	20	0	0	0	0	60	0	0	4,416	0	0	0	0
		Gammaridae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Haustoriidae	20	80	22	66	0	44	0	794	1,127	0	22	66	0	0
		Liljeborgiidae	0	0	0	0	0	0	13	0	0	0	0	0	0	0
		Lysianassidae	0	0	0	0	0	0	7	0	0	7	0	0	0	0
		Oedicerotidae	0	0	0	0	0	0	7	0	0	0	0	0	0	0
		Phoxocephalidae	0	0	0	0	0	0	40	0	0	0	0	0	0	0
	cumacean	Cumacea	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Diastylidae	0	0	0	0	0	0	7	0	0	0	0	0	0	0
	decapod (mole crab)	Hippidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	decapod (mud shrimp)	Upogebiidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	decapod (true crab)	Pinnotheridae	0	0	0	0	0	0	7	0	0	0	0	0	0	0
	horseshoe crab	Limulidae	0	0	243	44	26,367	132	7	0	0	7	22	0	3,704	0
	insect	Insecta	7	0	0	0	0	0	7	0	0	27	0	0	0	0
	isopod	Idoteidae	0	0	88	0	0	0	20	22	0	0	0	0	0	88
		Sphaeromatidae	0	0	0	0	22	22	0	22	73	0	0	66	0	22
	ostracod (seed shrimp)	Ostracoda	27	441	0	0	0	0	20	0	0	100	0	0	0	0
	pycnogonid (sea spider)	Phoxichilidiidae	7	0	0	0	0	0	0	0	0	0	0	0	0	0
	tanaid	Leptocheliidae	0	0	0	0	0	0	0	0	0	7	0	0	0	0
		Tanaidacea	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cnidaria	cnidarian	Actiniaria	0	0	0	0	22	0	7	0	0	40	0	22	0	0
		Cerianthidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Mollusca	bivalve	Bivalvia	0	40	22	22	88	110	147	0	0	0	0	154	0	0
		Cyrenidae	915	27	0	0	44	22	3,334	0	0	9,172	0	0	66	1,301
		Donacidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Mytilidae	1,951	6,226	0	0	0	0	94	1,080	122	13	220	110	0	0
		Solemyidae	0	0	0	0	0	0	7	0	0	0	0	0	0	0
		Solenidae	0	7	0	0	0	0	0	0	0	0	0	0	0	0
		Tellinidae	7	27	0	0	0	0	13	0	0	0	22	0	0	0
	gastropod	Acteonidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Calyptraeidae	0	0	0	0	0	0	0	0	0	13	0	0	0	0
		Hydrobiidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Littorinidae	0	0	0	0	0	0	40	0	0	0	0	0	0	0
		Nassariidae	7	0	0	0	0	0	60	0	0	0	0	0	0	0
Nemertea	nemertean worm	Nemertea	0	33	0	0	0	22	13	0	0	0	0	0	0	0
Platyhelminthes	flatworm	Platyhelminthes	0	0	0	0	0	0	114	0	0	0	0	0	0	0
		Total (#/m^2)	3,748	7,222	639	595	26,720	397	17,683	4,850	1,323	21,137	485	639	4,277	4,056
		Number of Taxa	19	18	6	6	10	7	38	11	3	25	5	10	5	11

Table 3. Continued.

Phylum	Type of Organism	Taxon	Average Density (#/m2) by Habitat Type, Site, and Sampling Date											
			Overwash								Overwash/Inlet			
			NBWT2	PSO	BRO		NBWT2		PSO		OIE		OIW	
			4/15/15	4/15/15	6/17/13	7/18/13	6/18/13	7/16/13	6/20/13	7/28/13	6/23/13	7/18/13	7/8/13	7/19/13
			April		June/July						June/July			
Annelida	oligochaete	Enchytraeidae	33	80	0	0	0	0	88	88	0	0	0	0
		Tubificidae	855	2,819	110	0	88	1,345	10,053	1,830	176	441	0	0
	polychaete	Ampharetidae	0	0	0	0	0	0	0	0	0	0	0	0
		Capitellidae	53	154	0	0	0	287	2,425	8,532	0	0	0	0
		Chaetopteridae	7	7	0	0	0	0	0	0	0	0	0	0
		Cirratulidae	20	134	0	0	0	0	176	1,058	0	0	0	0
		Glyceridae	7	13	0	0	0	0	0	22	0	0	0	0
		Lumbrineridae	1,116	601	0	0	0	132	0	331	0	0	0	0
		Maldanidae	0	0	0	0	0	0	0	22	0	0	0	0
		Nephtyidae	0	0	0	0	0	0	0	0	0	0	0	0
		Nereididae	180	421	0	0	0	0	0	22	0	0	0	0
		Oenonidae	0	0	0	0	0	0	66	66	0	0	0	0
		Orbiniidae	942	127	0	22	0	0	88	132	0	0	0	0
		Paraonidae	7	0	0	0	0	0	0	0	0	0	0	0
		Pectinariidae	0	0	0	0	0	0	0	0	0	0	0	0
		Phyllodocidae	33	134	0	0	0	529	88	441	0	0	0	0
		Polychaeta	0	0	0	0	0	0	0	44	0	0	0	0
		Sabellidae	0	0	0	0	0	88	0	0	0	529	0	22
		Spionidae	20	27	44	22	0	88	88	132	0	44	0	0
		Syllidae	428	561	0	0	0	132	485	419	0	0	0	0
Arthropoda	amphipod	Ampeliscidae	7	7	0	0	0	0	44	22	0	0	44	0
		Aoridae	13	114	0	0	0	0	0	353	0	44	0	0
		Corophiidae	748	2,064	0	0	0	44	44	353	0	0	0	0
		Gammaridae	0	0	0	0	0	0	0	198	0	0	0	0
		Haustoriidae	0	0	0	0	243	0	0	0	44	66	0	0
		Liljeborgiidae	0	0	0	0	0	0	0	0	0	0	0	0
		Lysianassidae	0	7	0	0	22	0	0	0	0	0	0	22
		Oedicerotidae	0	0	0	0	0	0	22	0	0	0	0	0
		Phoxocephalidae	27	174	0	0	0	0	0	0	0	0	0	0
	cumacean	Cumacea	7	0	0	0	0	0	0	0	0	0	0	0
		Diastylidae	0	0	22	0	22	88	309	22	0	0	0	0
	decapod (mole crab)	Hippidae	0	0	0	44	0	44	0	0	44	44	66	463
	decapod (mud shrimp)	Upogebiidae	0	0	0	0	0	0	0	0	0	0	22	0
	decapod (true crab)	Pinnotheridae	0	0	0	0	0	0	0	0	0	0	0	0
	horseshoe crab	Limulidae	0	0	7,848	220	78,461	110	243	132	0	0	0	0
	insect	Insecta	0	7	0	0	0	0	0	0	0	0	0	0
	isopod	Idoteidae	7	127	0	22	0	88	44	507	0	0	0	0
		Sphaeromatidae	13	2,332	0	66	44	110	0	22	0	22	0	0
	ostracod (seed shrimp)	Ostracoda	73	154	0	0	0	0	0	22	0	0	0	0
	pycnogonid (sea spider)	Phoxichilidiidae	0	94	0	0	0	0	0	0	0	0	0	0
	tanaid	Leptocheliidae	0	982	0	0	0	0	0	0	0	0	22	0
		Tanaidacea	0	7	0	0	0	0	0	0	0	0	0	0
Cnidaria	cnidarian	Actiniaria	0	80	0	22	0	0	0	0	0	0	0	0
		Cerianthidae	0	0	0	0	22	0	0	0	0	0	0	0
Mollusca	bivalve	Bivalvia	0	0	0	132	0	0	0	0	0	22	0	0
		Cyrenidae	22,253	13,214	375	44	132	419	3,020	15,873	0	0	0	22
		Donacidae	0	0	0	0	0	0	0	0	0	0	0	44
		Mytilidae	7	13	0	0	0	0	0	0	110	0	0	22
		Solemyidae	0	0	0	0	0	0	0	0	0	0	0	0
		Solenidae	0	0	0	0	0	0	0	0	0	0	0	0
		Tellinidae	0	0	0	0	0	22	0	0	0	0	0	22
	gastropod	Acteonidae	7	0	0	0	0	0	0	0	0	0	0	0
		Calyptraeidae	0	0	0	0	0	0	0	0	0	0	0	0
		Hydrobiidae	0	7	0	0	0	0	0	0	0	0	0	0
		Littorinidae	0	0	0	0	0	0	0	0	0	0	0	0
		Nassariidae	13	47	0	0	0	0	0	0	22	0	0	0
Nemertea	nemertean worm	Nemertea	0	27	0	0	0	0	0	0	0	0	0	0
Platyhelminthes	flatworm	Platyhelminthes	0	0	0	0	0	0	0	0	0	0	0	0
		Total (#/m^2)	26,876	24,531	8,399	595	79,034	3,527	17,284	30,644	397	1,213	154	617
		Number of Taxa	25	30	5	9	8	15	16	24	5	8	4	7

Table 4. The average density of each macrobenthic taxon considered to be a Piping Plover prey item, by study site and date collected. The samples from April are all from 2015 and the samples from June/July are all from 2013.

Phylum	Type of Organism	Taxon	Average Piping Plover Prey Density (#/m^2) by Habitat Type, Site, and Sampling Date																									
			Flood Shoal						Mud Flat	Beach		Filled Breach					Overwash										Overwash/Inlet			
			CSB	MUS	BFS		LFS		CMF	CSE		NBWT1	CSW		NBWT1		NBWT2	PSO	BRO		NBWT2		PSO		OIE		OIW	
			4/14/15	4/16/15	7/4/13	7/19/13	7/4/13	7/19/13	4/14/15	6/24/13	7/17/13	4/15/15	6/24/13	7/17/13	6/18/13	7/16/13	4/15/15	4/15/15	6/17/13	7/18/13	6/18/13	7/16/13	6/20/13	7/28/13	6/23/13	7/18/13	7/8/13	7/19/13
			April		July				April	June/July		April	June/July				April		June/July						June/July			
Annelida	polychaete	Glyceridae	0	0	0	0	0	0	13	0	0	0	0	0	0	0	7	13	0	0	0	0	0	22	0	0	0	0
		Nereididae	0	0	0	0	0	0	775	0	0	187	0	0	0	0	180	421	0	0	0	0	0	22	0	0	0	0
		Oenonidae	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	66	66	0	0	0	0
Arthropoda	amphipod	Corophiidae	0	20	0	0	0	0	60	0	0	4,416	0	0	0	0	748	2,064	0	0	0	44	44	353	0	0	0	0
		Haustoriidae	20	80	22	66	0	44	0	794	1,127	0	22	66	0	0	0	0	0	0	243	0	0	0	44	66	0	0
	horseshoe crab	Limulidae	0	0	243	44	26,367	132	7	0	0	7	22	0	3,704	0	0	0	7,848	220	78,461	110	243	132	0	0	0	0
	insect	Insecta	7	0	0	0	0	0	7	0	0	27	0	0	0	0	0	7	0	0	0	0	0	0	0	0	0	0
Mollusca	bivalve	Donacidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	44
Total (#/m^2)			33	100	265	110	26,367	176	862	794	1,127	4,636	44	66	3,704	0	935	2,505	7,848	220	78,704	154	353	595	44	66	0	44
Total without Limulidae (#/m^2)			33	100	22	66	0	44	855	794	1,127	4,630	22	66	0	0	935	2,505	0	0	243	44	110	463	44	66	0	44
Number of Taxa			3	2	2	2	1	2	5	1	1	4	2	1	1	0	3	4	1	1	2	2	3	5	1	1	0	1

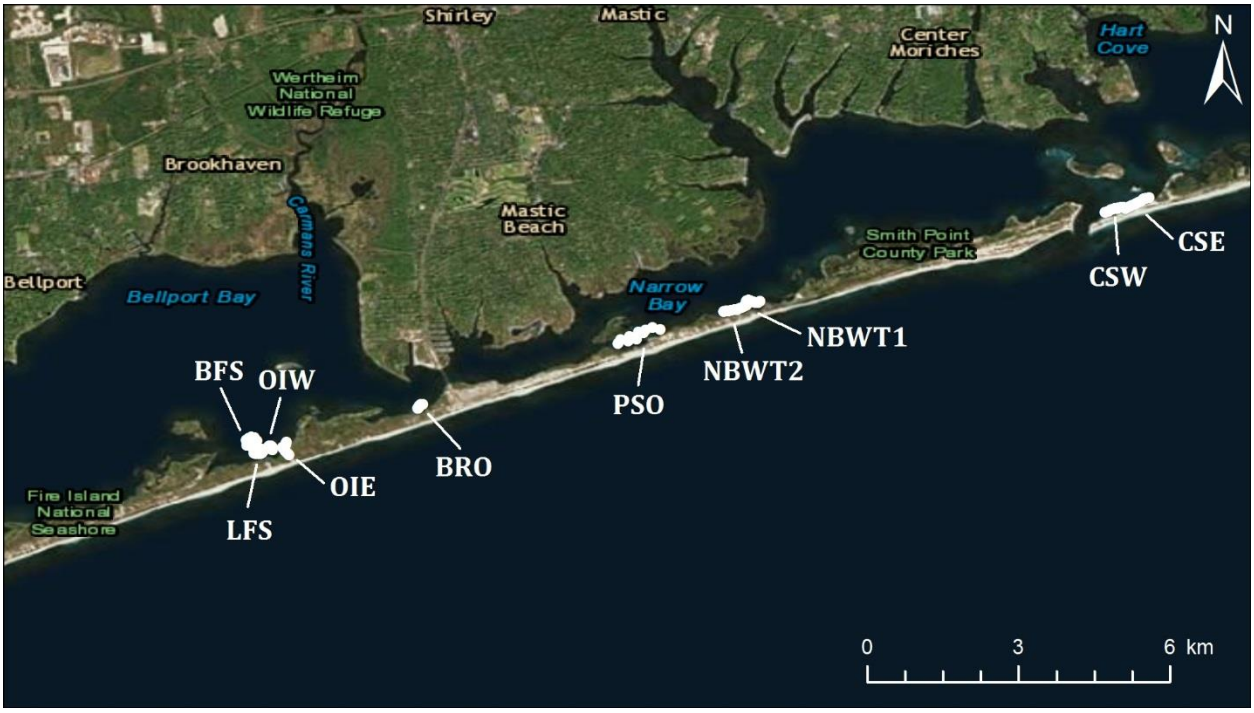




Figure 1. Map of the study area with 2013 (top) and 2015 (bottom) sampling sites used for this study. Base map source: 2013 NYGIS Orthoimagery.

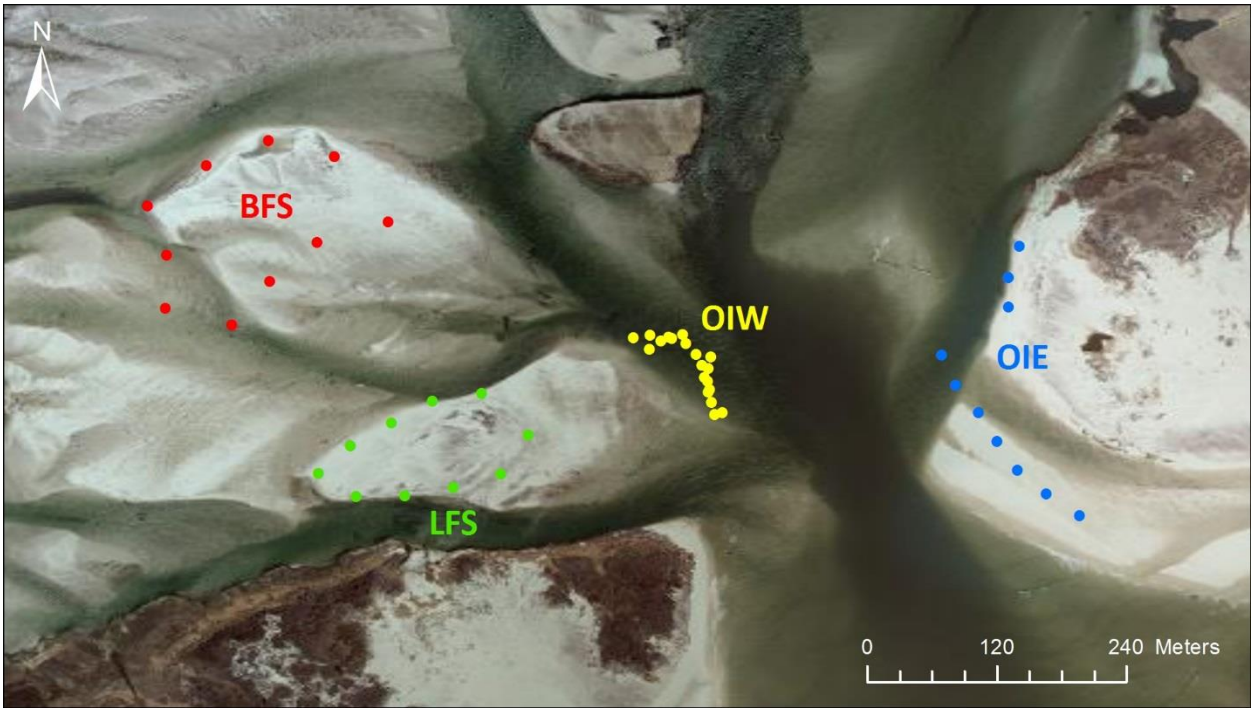


Figure 2. Map of the Big Flood Shoal (BFS), Little Flood Shoal (LFS), and Old Inlet East (OIE) and West (OIW) study area and 2013 sampling stations used for this study. Base map source: 2015 NAIP USDA Farm Service Imagery.

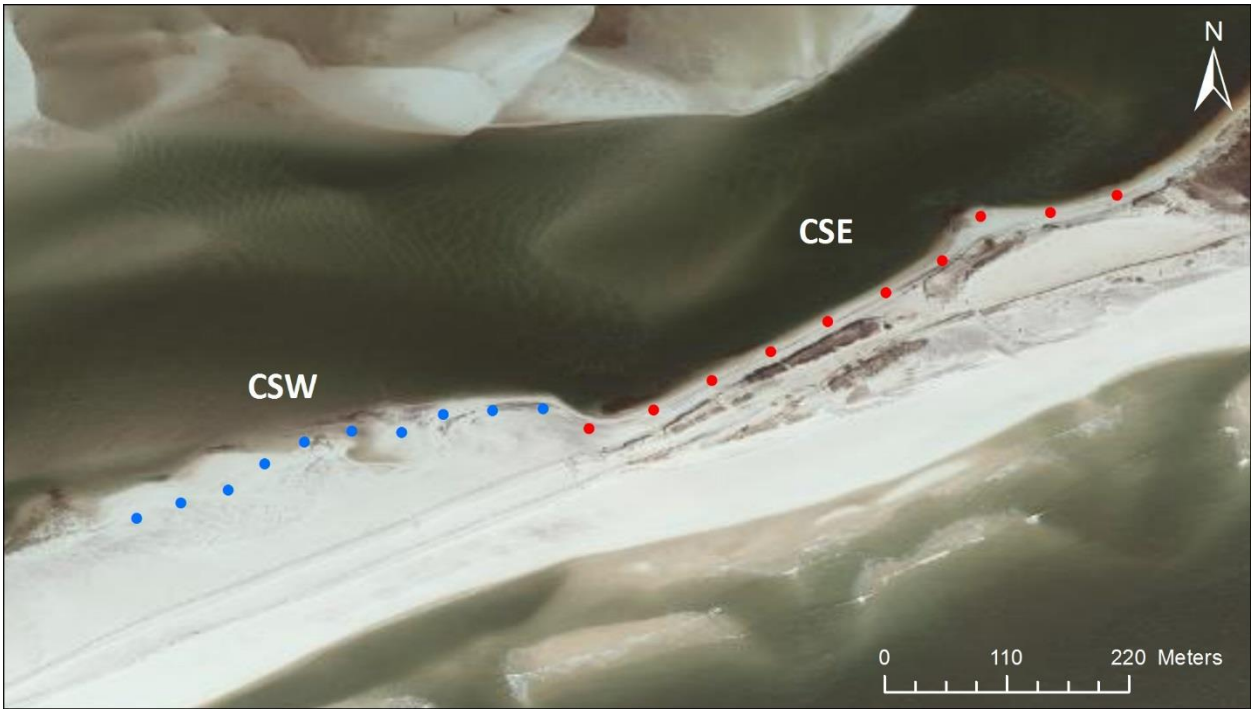


Figure 3. Map of the Cupsogue East (CSE) and Cupsogue West (CSW) study area and 2013 sampling stations used for this study. Base map source: 2015 NAIP USDA Farm Service Imagery.

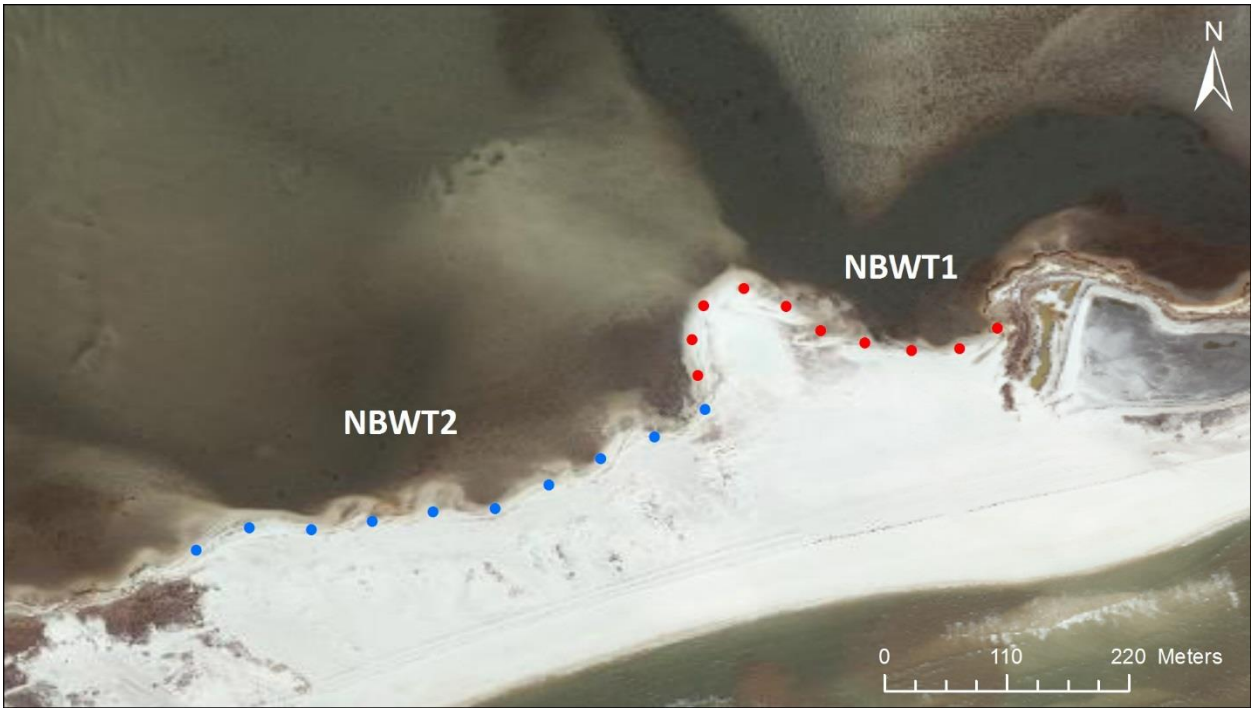


Figure 4. Map of the Narrow Bay West (NBWT1, NBWT2) study area and 2013 sampling stations used for this study. Base map source: 2015 NAIP USDA Farm Service Imagery.



Figure 5. Map of the Pattersquash Overwash (PSO) study area and 2013 sampling stations used for this study. Base map source: 2015 NAIP USDA Farm Service Imagery.

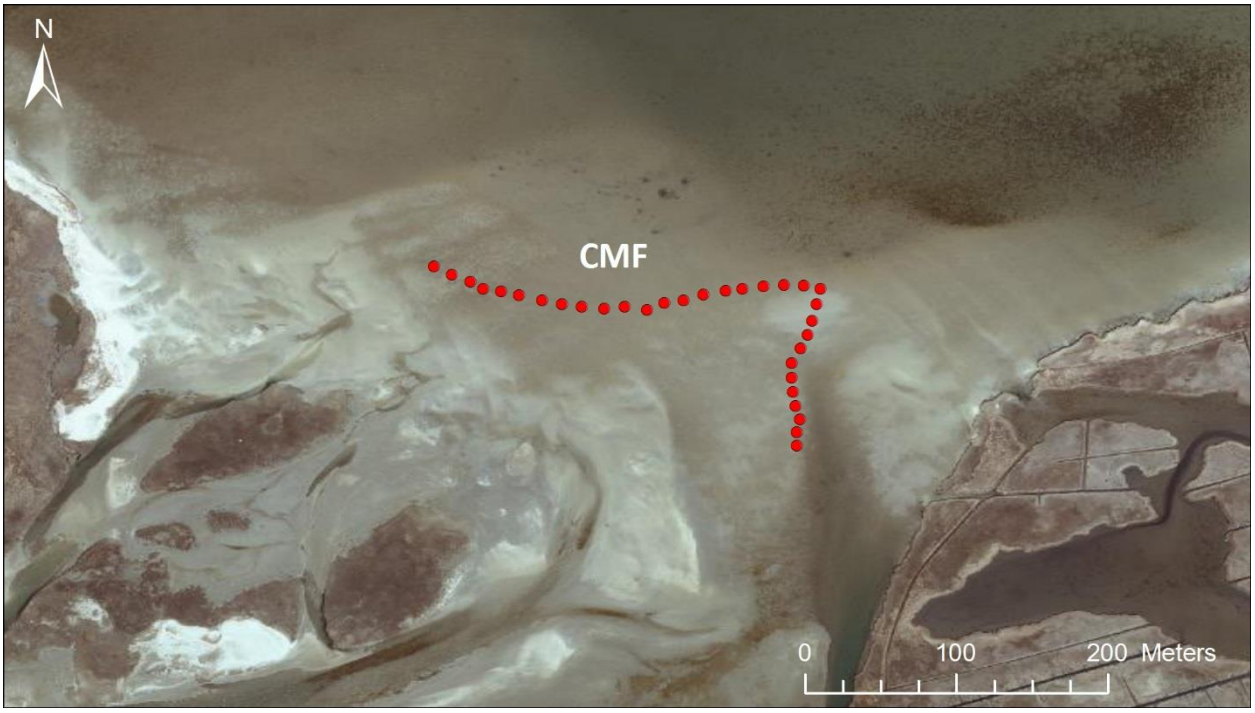


Figure 6. Map of the Cupsogue Mud Flats (CMF) study area and 2015 sampling stations used for this study. Base map source: 2015 NAIP USDA Farm Service Imagery.

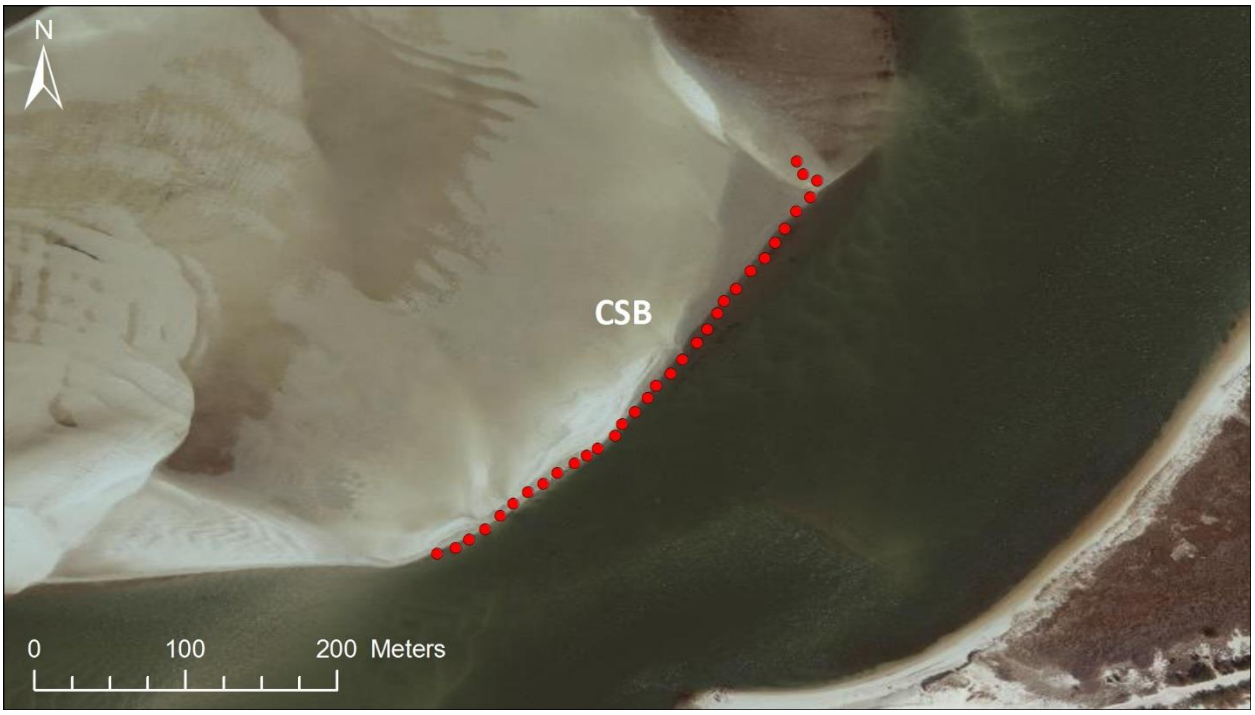


Figure 7. Map of the Cupsogue Sand Flats (CSB) study area and 2015 sampling stations used for this study. Base map source: 2015 NAIP USDA Farm Service Imagery.



Figure 8. Map of the Mussel Shoal (MUS) study area and 2015 sampling stations used for this study. Base map source: 2015 NAIP USDA Farm Service Imagery.

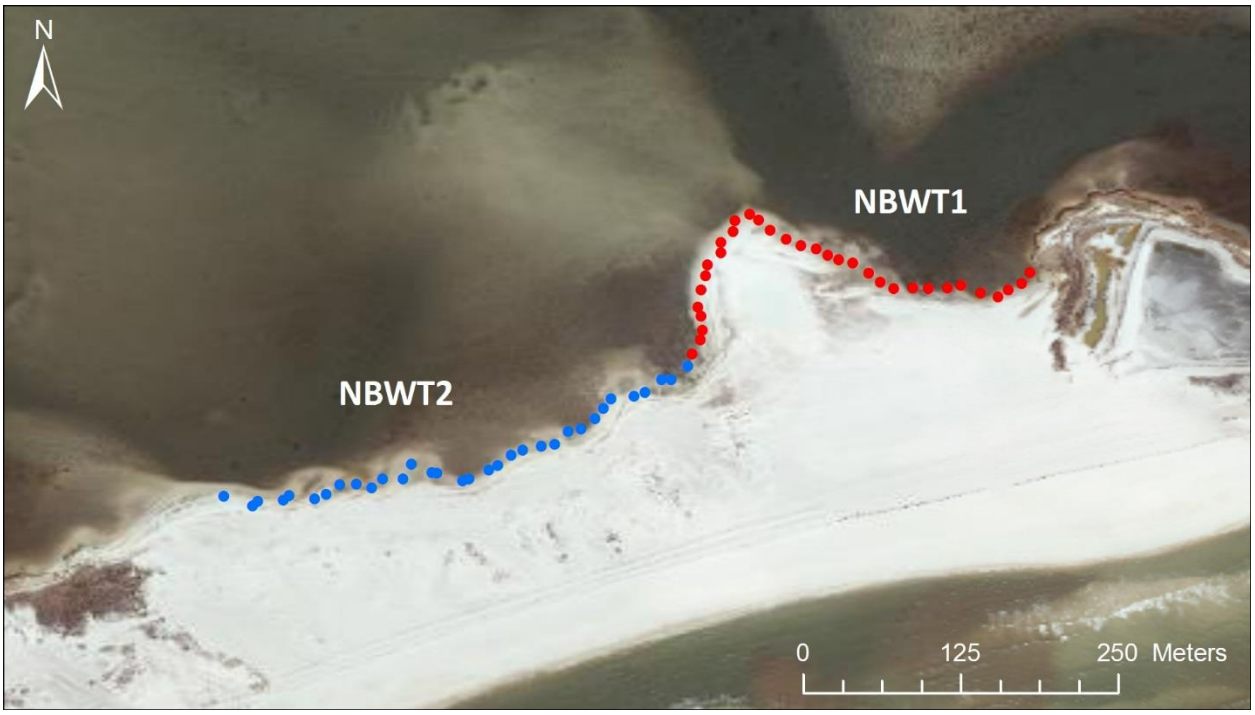


Figure 9. Map of the Narrow Bay West Filled Breach (NBWT1, red circles) and Overwash (NBWT2, blue circles) study area and 2015 sampling stations used for this study. Base map source: 2015 NAIP USDA Farm Service Imagery.

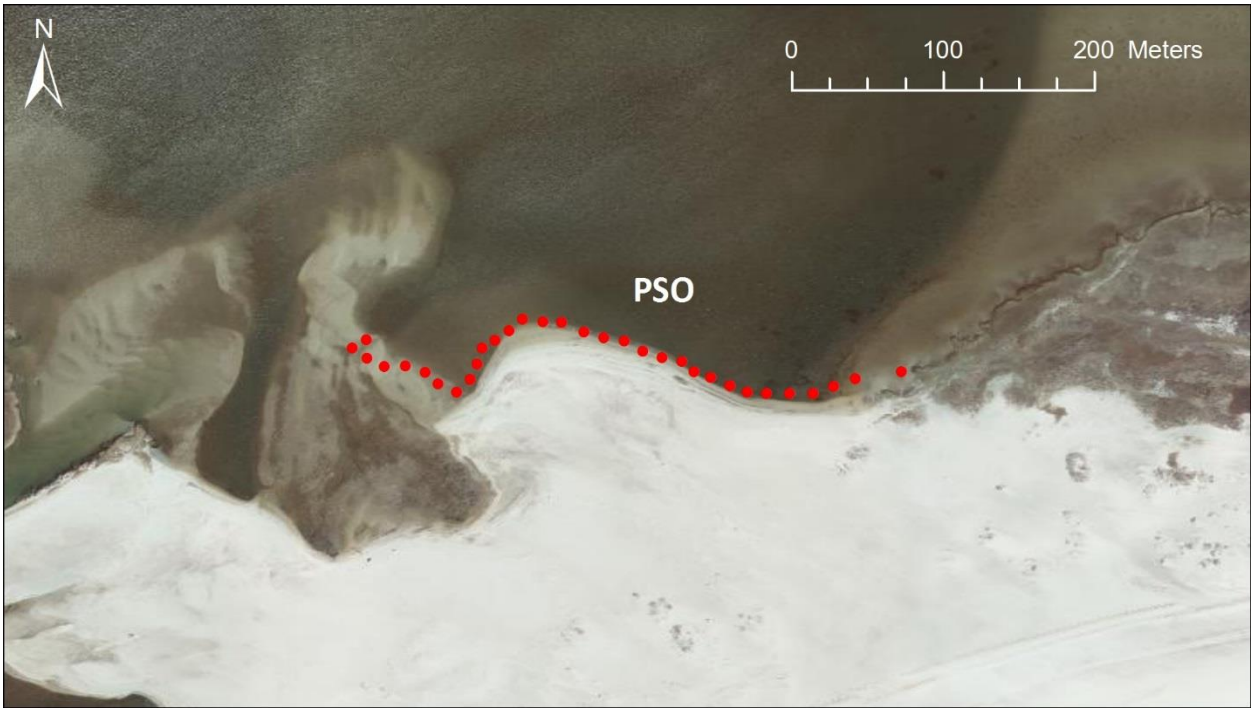


Figure 10. Map of the Pattersquash Overwash (PSO) study area and 2015 sampling stations used for this study. Base map source: 2015 NAIP USDA Farm Service Imagery.

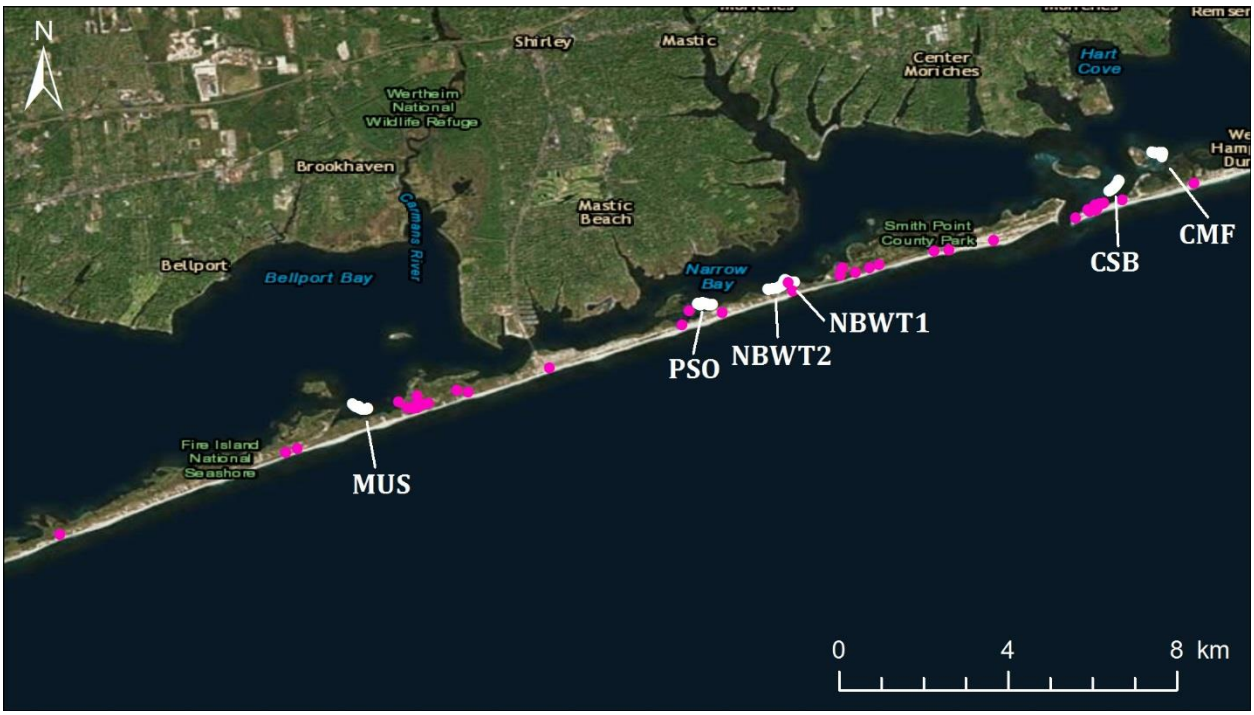


Figure 11. Map of the macrobenthic sampling sites (clusters of white circles) and fecal sampling sites (pink circles) used in this study. Base map source: 2015 NAIP USDA Farm Service Imagery.

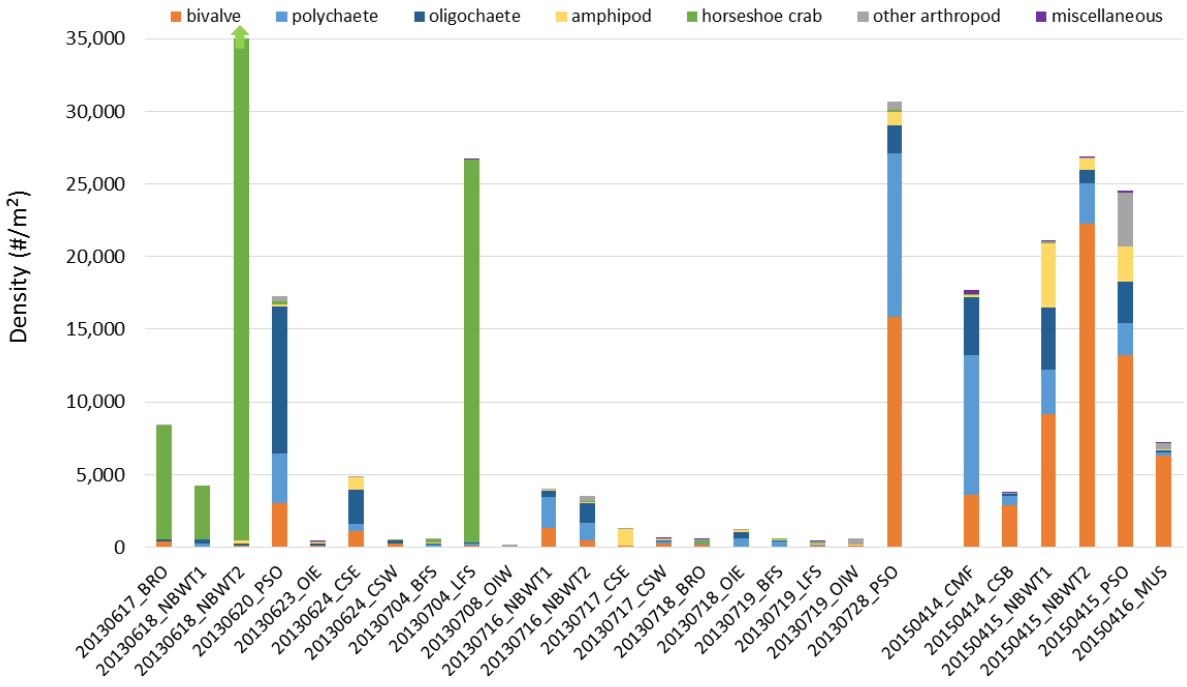


Figure 12. The average density (#/m^2) of macrobenthic taxa by category for the 2013 and 2015 study sites. Sites are ordered by date sampled.

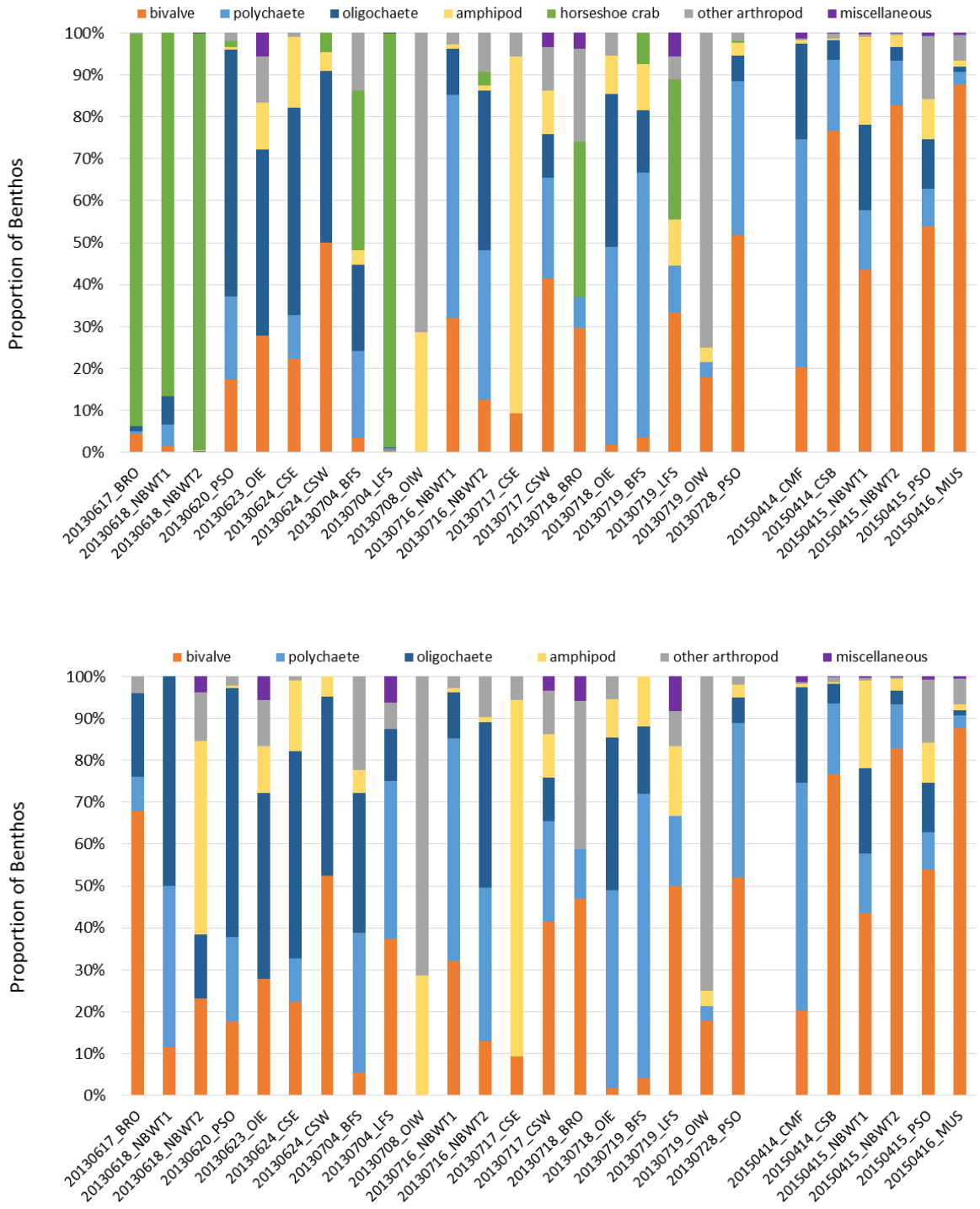


Figure 13. The proportion of each benthic group comprising the macrobenthic community for the 2013 and 2015 study sites. The top graph is the whole community and the bottom graph excludes the horseshoe crab data considering the seasonality of their abundances. Sites are ordered by date sampled.

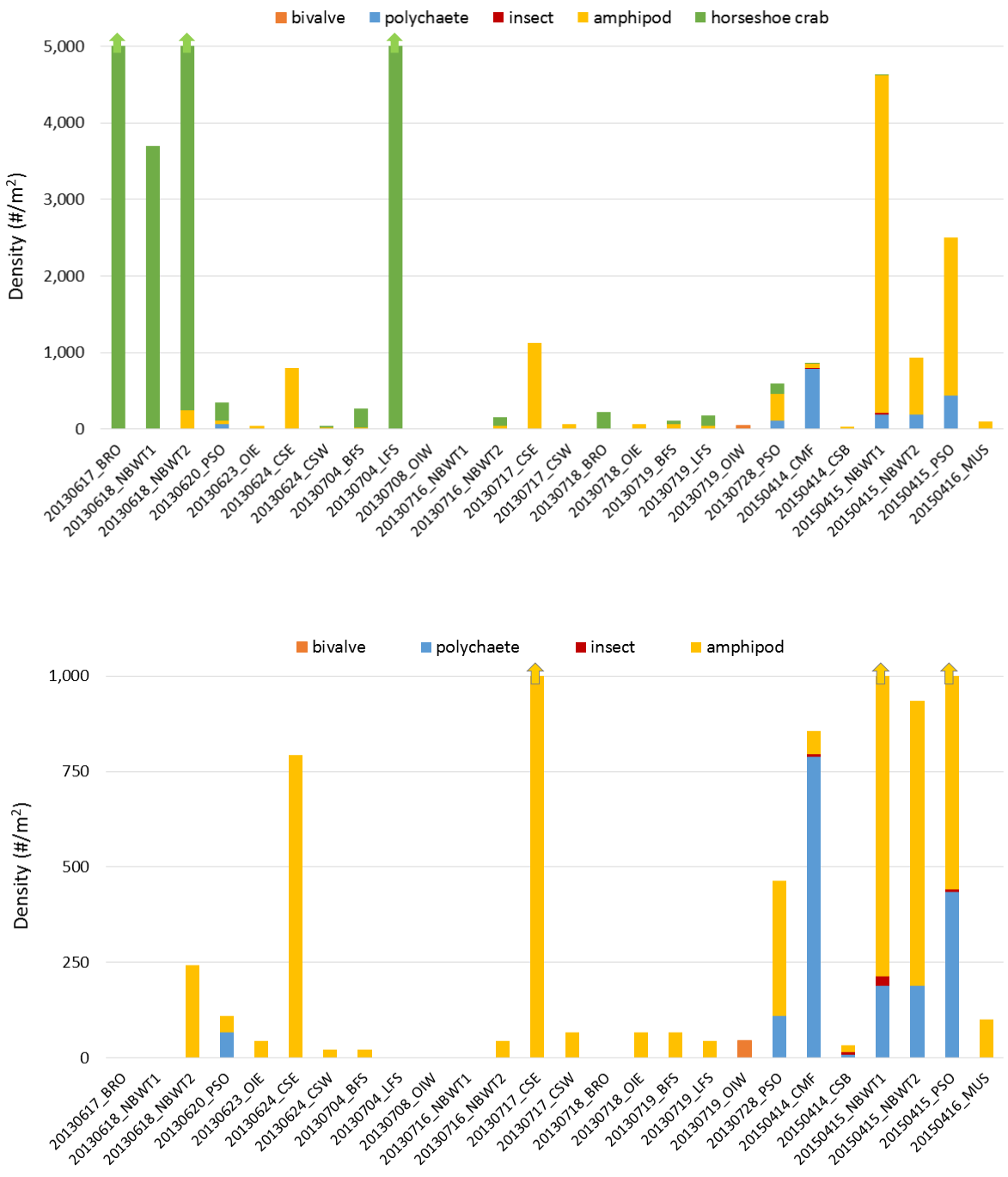


Figure 14. The average density (#/m^2) for each piping plover prey item for the 2013 and 2015 study sites. The top graph is the whole community and the bottom graph excludes the horseshoe crab data considering the seasonality of their abundances. Sites are ordered by date sampled.

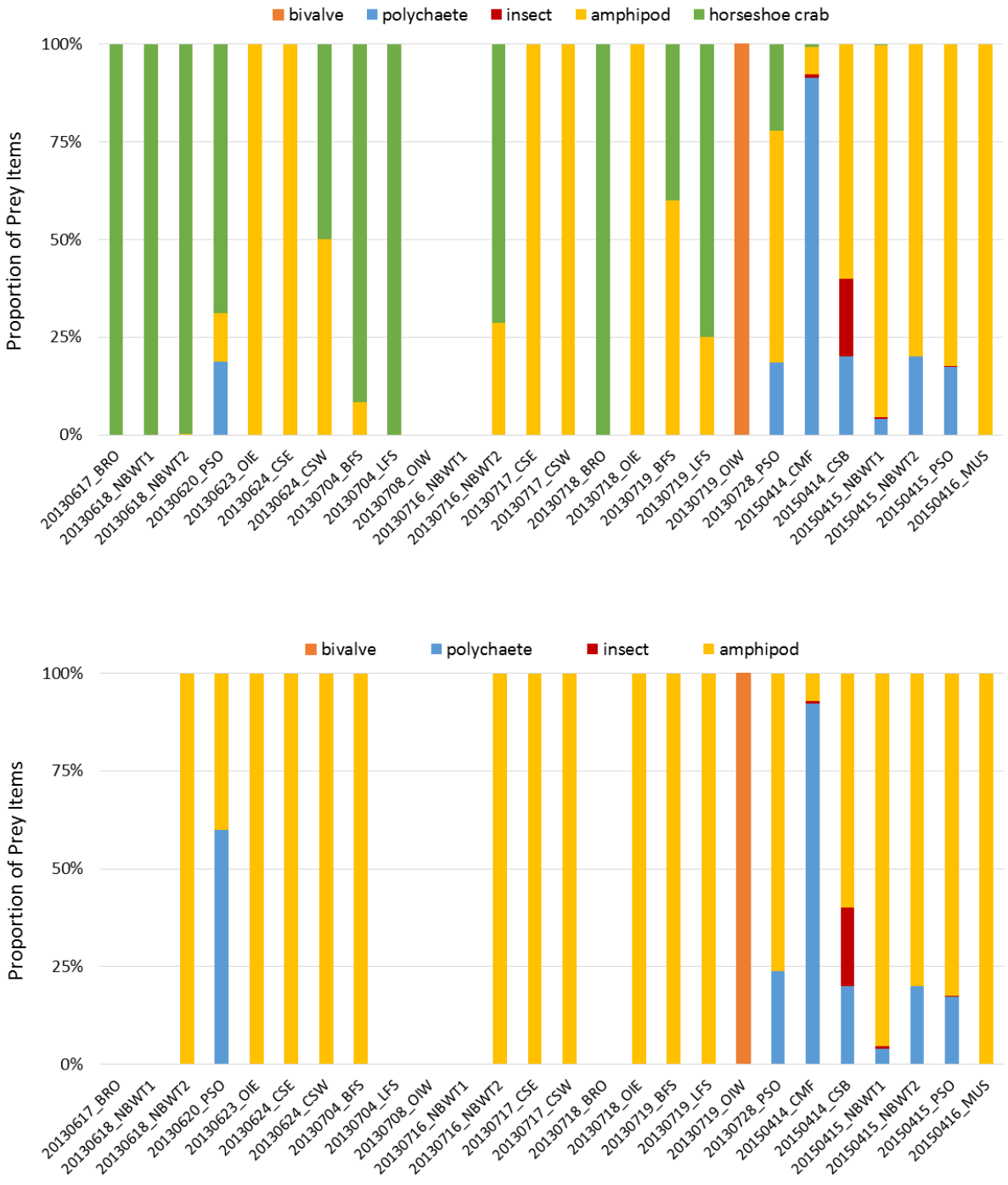


Figure 15. The average density (#/m^2) for each piping plover prey item for the 2013 and 2015 study sites. The top graph is the whole community and the bottom graph excludes the horseshoe crab data considering the seasonality of their abundances. Sites are ordered by date sampled.

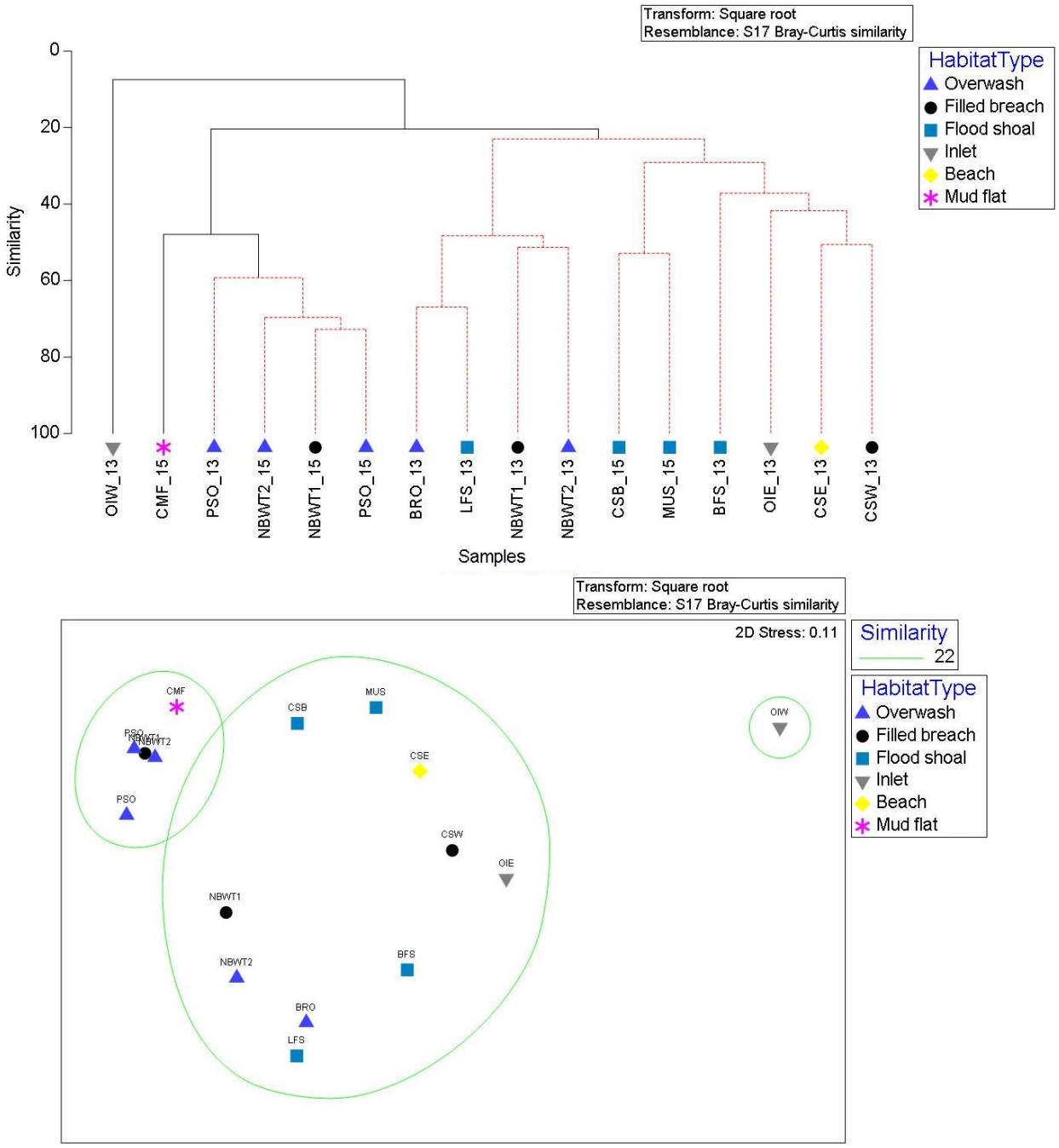


Figure 16. CLUSTER tree (top) and MDS ordination plots (bottom) of the macroinvertebrate infaunal community structure averaged for each of the sites sampled in 2013 and 2015. MDS similarity lines are based on CLUSTER results.

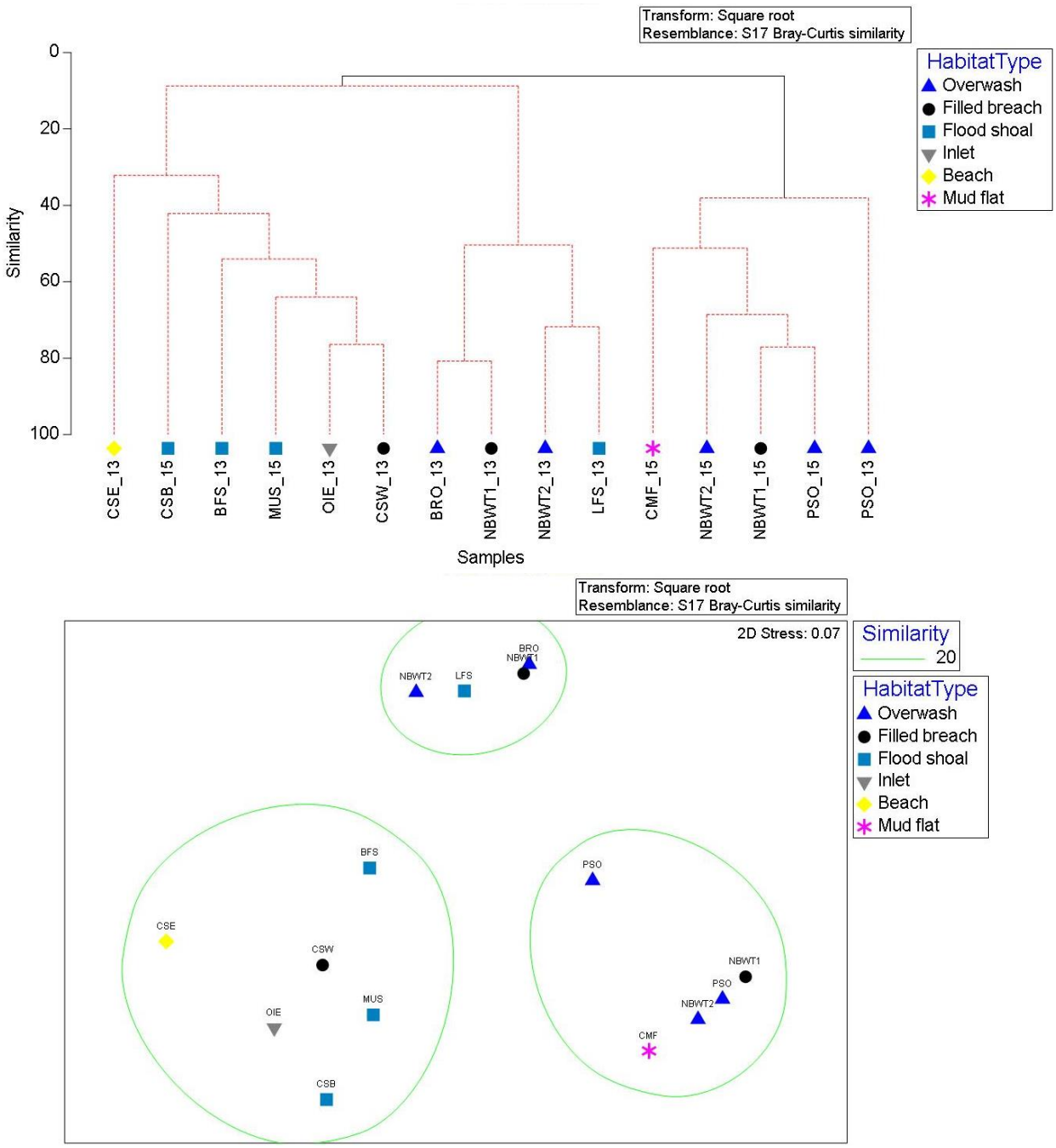


Figure 17. CLUSTER tree (top) and MDS ordination plots (bottom) of the piping plover prey subset of macroinvertebrate infaunal community structure averaged for each of the sites sampled in 2013 and 2015. OIW was removed from the analysis due to no prey items being present. MDS similarity lines are based on CLUSTER results.

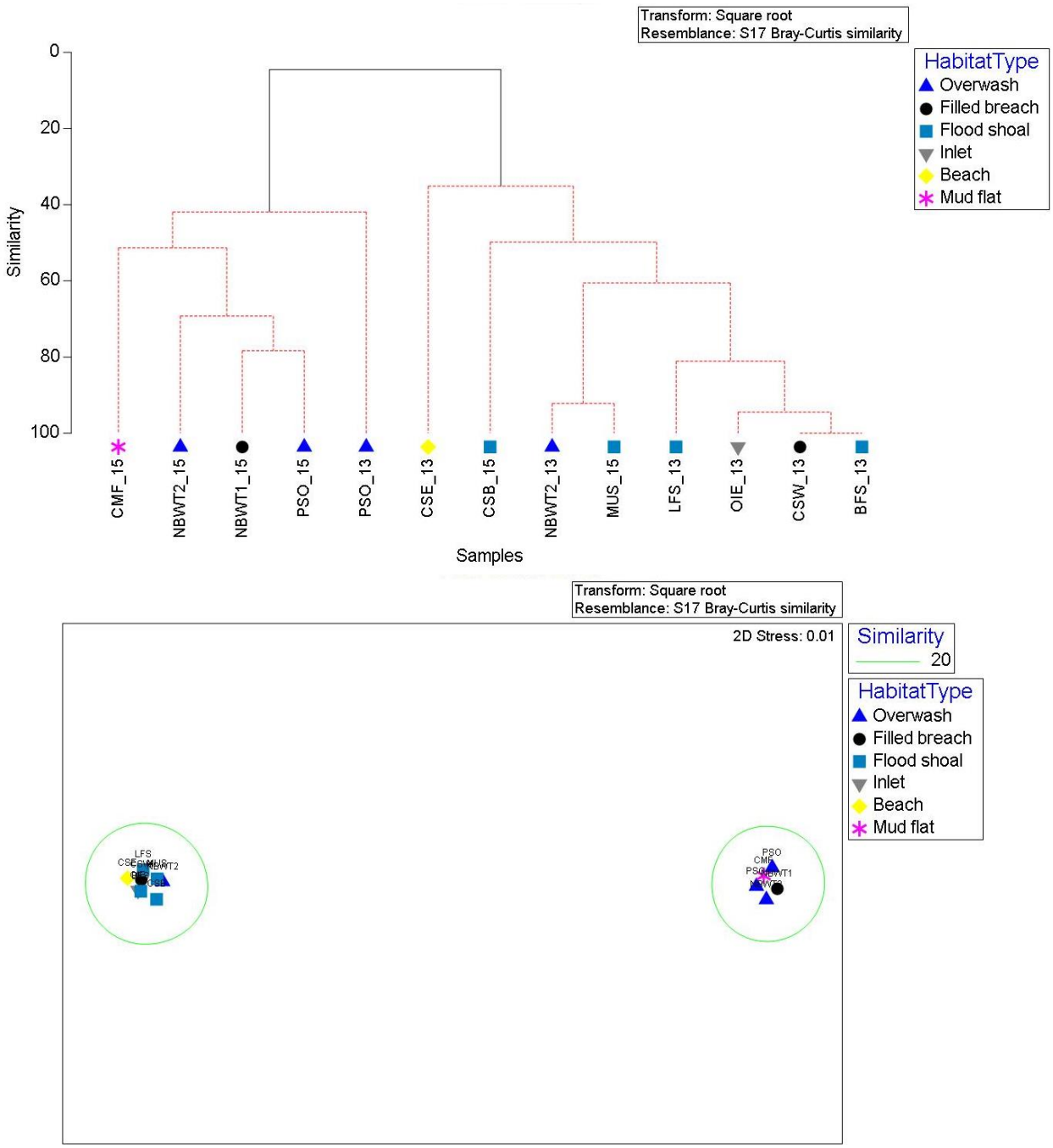


Figure 18. CLUSTER tree (top) and MDS ordination plots (bottom) of the piping plover prey subset of macroinvertebrate infaunal community structure excluding horseshoe crabs averaged for each of the sites sampled in 2013 and 2015. OIW, BRO, and NBWT1_13 were removed from the analysis due to no prey items being present. MDS similarity lines are based on CLUSTER results.

Appendix F. Supplementary material for Part 2, Section 8: Compare prey densities at several habitat types in 'Effectiveness monitoring and targeted studies in support of FIMI conservation measures.'

Introduction

Piping plover diet is relatively understudied, particularly as the collection of individuals to explore stomach or gizzard contents is limited due to their status as a species of conservation concern (Elliot-Smith and Haig 2004). Historically, piping plovers were thought to feed only on arthropods and substrate invertebrates. Past investigations into piping plover diet have used gizzard samples of chicks or visually identified invertebrates in fecal samples, and they have found mainly invertebrates from the orders Hymenoptera, Coleoptera, Diptera, Decapoda, and Amphipoda, but some sources may be difficult to identify (e.g., mollusks, annelids, etc.; Shaffer and Laporte 1994, Cuthbert et al. 1999, Majka and Shaffer 2008, Cisek 2013). However, on Fire Island, piping plovers have been documented eating small fish, which would not be apparent from invertebrate samples (Elliot-Smith and Haig 2004, Monk et al. 2016). Visually identifying prey in fecal samples may under-represent soft bodied prey, and polycheates are likely a prey source for plovers in certain areas (Zonick 2000). An alternative method, DNA metabarcoding, amplifies prey DNA from fecal samples using primers specific to the taxonomic prey group of interest to obtain data about consumed prey species. For example, a recent analysis of fecal samples of semipalmated sandpipers (*Calidris pusilla*) using DNA metabarcoding approaches found that in the Bay of Fundy, semipalmated sandpipers were opportunistic generalists, despite previous assumptions that they selected a single prey item (Gerwing et al. 2016).

Understanding invertebrate availability, diet, and prey selection can assist in understanding habitat selection and managing foraging habitat for plovers. If, using DNA metabarcoding approaches, we can develop a framework to noninvasively investigate use and abundance of prey species, then we can advise managers on the best way to proceed following habitat change. However, a comparison between DNA metabarcoding and traditional methods is needed to determine if secondary consumption may be apparent or if one method outperforms the other for specific prey taxa. We could then potentially manage for maximizing plover fitness by targeting management actions in a way that could maximize the prey piping plovers are selecting for. Using invertebrate availability as determined in Part 2, Section 8, and Appendix E, our objectives were to determine 1) what Fire Island and Westhampton Island plovers are eating using both visual identification and DNA metabarcoding methods in fecal samples, 2) compare the two methods of prey identification, and 3) determine if we can evaluate prey selection by piping plovers using diet tools.

Methods

We evaluated piping plover diet using two methods. The first was using visual identification, which was conducted on fecal samples collected in 2013 and 2014. Visual identification in fecal samples was conducted by the South Carolina DNR, who also identified invertebrates in core samples (Appendix E). The second method was DNA metabarcoding, which was conducted by Jonah Ventures, Boulder, CO (https://jonahventures.com/, Boulder, CO, USA).

We used four primers to amplify specific genes to evaluate prey in fecal samples. The first, mitochondrial cytochrome-c-oxidase subunit I (hereafter; COI), was primarily used for detecting arthropod prey. To detect polycheates, we used the nuclear hypervariable region V8 of 18S rDNA (hereafter; 18S) primer, which is a general Eukaryote primer, and due to previous detections of plovers eating fish (Monk et al. 2016), we used the Ac12S and 16S primers to detect fish (Evans et al. 2015). DNA metabarcoding is a developing science and in particular, the use of the 18S, Ac12S, and 16S primers are still in the developmental stages with Jonah Ventures. We received a file from Jonah Ventures for each primer-year combination, which identified which taxonomic units were detected in each fecal sample. Each sample had a certain number of hits per taxonomic unit, and we subset detections to those with at least 99 hits when summed by Order, and at least 50 detections within a single sample, and we removed any plover fecal samples that did not have any prey detected (J. Craine, Jonah Ventures, personal communication).

We compared the orders detected in substrate invertebrates samples (See Appendix E) and terrestrial invertebrate samples (Part 2, Sections 1 and 8) to the orders detected in fecal samples. Given that collection years for substrate and terrestrial invertebrate collection did not completely overlap with fecal sample data collection, we lumped both years of substrate invertebrate sampling and all years six of terrestrial invertebrate sampling to compare frequency of occurrence in availability samples to use.

Results

Using the COI primer, we detected prey items from 16 orders in 70 fecal samples (Table 1). The 18S primer only detected prey from the 2017 samples, in which we detected prey items from 73 orders in 34 fecal samples. We only retained 24 orders, from Arthropod, Polycheate, Mollusk or Fish phyla, as potential prey. We removed detections of birds, humans, algae, plants, molds, and parasites, as they were unlikely to be plover prey items.

At least one prey taxa was detected by visual identification in all 44 fecal samples collected between 2013 and 2014, however some prey only were identifiable to broader taxonomic groups (e.g., Arthropoda). In those cases, we lumped the broader group with all prey taxa that would fall under that group for a coarser prey comparison. When comparing the fecal sample techniques at the phylum level, the 18S primer detected

polycheates more often, whereas visual identification detected arthropods more often (Table 4). The COI primer detected both phyla least among the three methods.

We detected six taxa of fish in using the the 18S, Ac12s and 16S primers. These were Sparidae [sea breams and porgies], Ophichthidae [snake eels], Pomatomidae [blue fish], Ammodytidae [sand lance], Gadiformes [cod and allies], and Rajiformes [skates and rays].

Table 1. Invertebrate taxa detected in piping plover fecal samples using DNA metabarcoding methods, and the 18S and COI primers. Orders were retained if detected using a specific primer at least 99 times, and detected within a sample at least 50 times.

Order	Common Name	Frequency of occurrence in fecal samples	
		18S	COI
Amphipoda	Crustaceans (aquatic and "sand hoppers")	0.74	0.01
Anaspidacea	Crustaceans	0.06	-
Araneae	Spiders and mites	0.12	0.11
Calanoida	Copepods (zooplankton)	0.12	-
Coleoptera	Beetles	0.47	0.41
Cumacea	Crustaceans (hooded shrimp, comma shrimp)	0.09	0.04
Decapoda	Crustaceans (crabs, crayfish, lobsters, shrimp)	0.12	0.20
Dermaptera	Earwigs	0.03	-
Diptera	Flies	0.32	0.63
Ephemeroptera	Mayflies	-	0.07
Euphausiacea	Krill	0.09	-
Hemiptera	True bugs	0.06	0.11
Hymenoptera	Wasps, bees, ants, sawflies	0.03	0.09
Isopoda	Crustaceans (includes woodlice)	0.12	0.06
Lepidoptera	Butterflies and moths	-	0.40
Mytiloida	True mussels	0.12	-
Neuroptera	Lacewings, Mantidflies, antlions, etc.	0.18	0.09
Phyllodocida	Polychaete	0.15	0.01
Spariformes	Ray-finned fishes	0.09	-
Spionida	Marine polychaete	0.09	0.03
Stylommatophora	Snails and slugs	-	0.03
Trombidiformes	Mites	0.03	-
Veneroida	Bivalve mollusks	0.03	-
Xiphosura	Horseshoe crabs	0.12	0.13

Table 2. Invertebrate taxa detected in piping plover fecal samples using visual identification methods

Order	Common Name	Frequency of occurrence in fecal samples
Araneae	Spiders and mites	0.02
Arthropoda (excluding decapoda)	Unidentifable arthropods	0.55
Coleoptera	Beetles	0.50
Decapoda	Crustaceans (crabs, crayfish, lobsters, shrimp)	0.02
Diptera	Flies	0.02
Hymenoptera	Wasps, bees, ants, sawflies	0.05
Lepidoptera	Moths and butterflies	0.02
Phyllodocida	Polycheate	0.07
Polychaeta	Polycheate	0.05
Sabellida	Polycheate	0.02
Xiphosurida	Horseshoe crabs	0.18

Table 3. Comparison of phylum-level identification between the two primers used to detect prey in piping plover fecal samples by DNA metabarcoding, and traditional visual identification in fecal samples. 18S was only successful on samples from 2017, COI was used in 2016 and 2017, and visual identification was used on fecal samples from 2013 and 2014.

	18S	COI	Visual Identification
Polychaeta	23.5%	4.3%	13.63%
Arthropoda	94.1%	74.3%	100%

Eight invertebrate orders were detected using DNA metabarcoding and visual identification that were detected in the substrate invertebrate samples (Table 4). All eight orders were detected more frequently in core samples than in fecal samples. The 18S primer detected more polycheates in samples than either COI or visual identification.

Seven invertebrate orders were detected in fecal samples that also were found in terrestrial invertebrate samples (Table 6). All orders except Amphipoda were detected at equal or greater frequency in the fecal samples as compared to terrestrial invertebrate samples.

Table 4. Frequency of occurrence of prey in plover fecal samples, for which prey taxa were also observed in substrate invertebrate samples.

Order	Taxa Type	Substrate Invertebrate Samples (Visual)	Fecal Samples		
			18S	COI	Visual Identification
Amphipoda	Amphipod	84.6%	73.6%	1.43%	NA
Cumacea	Hooded Shrimp	26.9%	8.8%	4.3%	NA
Decapoda	Crustacean	26.9%	11.8%	20%	2.3%
Isopoda	Crustacean	61.5%	11.8%	5.7%	NA
Phyllodocida	Polycheate	53.8%	14.7%	1.43%	6.8%
Spionida	Polycheate	53.8%	8.8%	2.86%	NA
Xiphosura	Horseshoe Crab	53.8%	11.8%	12.9%	18.2%
Sabellida	Polycheate	19.2%	NA	NA	2.3%

Table 5. Invertebrate taxa detected in both fecal samples and terrestrial invertebrate samples. COI and 18S were primers used to amplify prey species in fecal samples 2016–2017, and visual identification was the identification of morphometric body parts in fecal samples 2013–2014. The measured values are frequency of occurrence, which is the % of samples which contained each taxa. Substrate invertebrate samples were used as a measure of availability, whereas 18S, COI and Visual Identification were used as measures of consumption in fecal samples. NA implied nondetection of a prey taxon.

Order	Taxa Type	Terrestrial invertebrate samples - zero detections included	Fecal Samples		
			COI	18S	Visual
Araneae	Spider	0.58%	11.42%	NA	2.3%
Amphipoda	Amphipod	4.82%	1.43%	NA	NA
Coleoptera	Beetle	2.39%	41.43%	47.1%	50%
Diptera	Flies	44.38%	62.86%	32.3%	2.3%
Hymenoptera	Wasps, bees, ants	1.81%	8.6%	2.9%	4.6%
Hemiptera	True bugs	0.96%	11.43%	5.9%	NA
Ephemeroptera	Mayfly	0.04%	7.14%	NA	NA

Discussion

Coleoptera, Diptera, Xiphosura, Lepidoptera, Polychaeta, Amphipoda, and Decapoda constituted the majority of invertebrate orders detected in fecal samples, suggesting that both terrestrial and substrate invertebrate samples are needed to characterize the breadth of available terrestrial and benthic invertebrates for piping plover. We found high variability in detection between the DNA metabarcoding primers and the visual identification methods. For example, Arthropods were visually detected in 100% of fecal samples but only detected in 94 and 74% by the 18S and COI primers, respectively. Amphipoda also was the most frequently detected prey order using the 18S primer, but was only detected in one sample using the COI primer. There may be a genetic detection bias between polycheates and arthropods, as detection can even vary within broader taxa (Gerik 2018). Bias also could be apparent if specific taxa are not available in the taxonomic database used to identify prey.

Primer bias and a lack of a controlled study limits our direct comparison between what plovers consumed and what was available but we can draw some inferences about availability. Diptera was the most

frequently detected taxa in terrestrial invertebrate samples, such that they were detected in approximately 45% of terrestrial invertebrate samples and their frequency of occurrence in fecal samples was nearly 50%, suggesting that they are foraging on flies relatively equally in relation to abundance. Alternatively, Coleoptera were detected in only 6% of terrestrial invertebrate samples, but the frequency of occurrence was 30.5% in our DNA metabarcoded fecal samples and 50% in our visual identification samples, which suggests selection for beetles, assuming no detection bias. Previous studies that tested for plover selection of prey have found that substrate invertebrates were more numerous in brood rearing transects than random transects (Loegering and Fraser 1995) and that prey selection varied by habitat type (Zonick 2000), so evaluating prey consumption and use at the brood level may be useful if diet studies continue.

Some invertebrate taxa detected using DNA metabarcoding and visual identification [i.e., Neuroptera and Lepidoptera] were not detected in terrestrial invertebrate samples or substrate samples, suggesting that the current level of invertebrate availability sampling may be limited to successfully capturing smaller invertebrates. Plovers also may be eating larvae of species before they are flighted (Sullins et al. 2018), leading to additional challenges comparing availability to use in this system. This discrepancy is notable in horseshoe crabs, which we were able to detect in some core samples and fecal samples, but did not find in terrestrial invertebrate samples. In addition, to assess frequency of occurrence in prey taxa from the substrate invertebrate core samples, we treated each transect as a single independent sample. However, it is likely that there is dependency within transects between sampling occasions, and thus we may need to group the within-year samples in future analyses of these data.

Observation of plovers would suggest far more consumption of polychaetes than we could detect either with visual identification or DNA metabarcoding, and past studies have shown evidence of selection for areas of high polycheate abundance (Staine and Burger 1994, Cohen et al. 2010). This discrepancy could lie in temporal use or availability of polychaetes, in the primers we used to evaluate their consumption, or in their detectability using visual identification methods – although their mouthparts and chaeta are calciferous and thus should be detectable if present. We also did not assess biomass, as DNA metabarcoding only is able to evaluate presence/absence.

As previously noted in Monk et al. (2016), consumption of fish by piping plovers is likely opportunistic and we did not evaluate availability of fish in our study area. Fire Island supports nesting Least Terns (*Sternula antillarum*), obligate piscivores that share habitat with plovers and return to nesting areas to feed their young. Further, anglers frequently catch skates and do not always return them to the water. Both of these occurrences could make fish available for plovers to scavenge. However, there is the possibility that DNA barcoding of plover fecal samples could be detecting secondary consumption (Gerwick 2018), when a plover prey item consumed something else prior to being consumed by the plover.

Our study was an opportunistic look into invertebrate use by plovers on Fire Island and Westhampton Island. Overall, DNA metabarcoding seems to be an effective way to characterize arthropod prey in piping plover fecal samples if 18S and COI primers are both used, as with visual identification many prey fragments were only identified to Phylum. However, we still need to assess detection of prey species within samples using different primers from DNA metabarcoding for a direct comparison as frequency of occurrence varied among the primers and taxa identification methods. Comparison across primers between the same samples may suggest whether one can be used exclusively, or if both primers amplify prey differently enough that they should both be used for identification. Future work should focus on evaluating invertebrate availability within plover foraging territories or immediately following prey consumption. More research also is needed to identify adequate primers for polychaetes in plover fecal samples.

Appendix G. Supplementary maps for Part 3, Section 2: Red fox spatial ecology in 'Effectiveness monitoring and targeted studies in support of FIMI conservation measures.'

Predicted habitat suitability maps for red foxes using 2016 landcover data

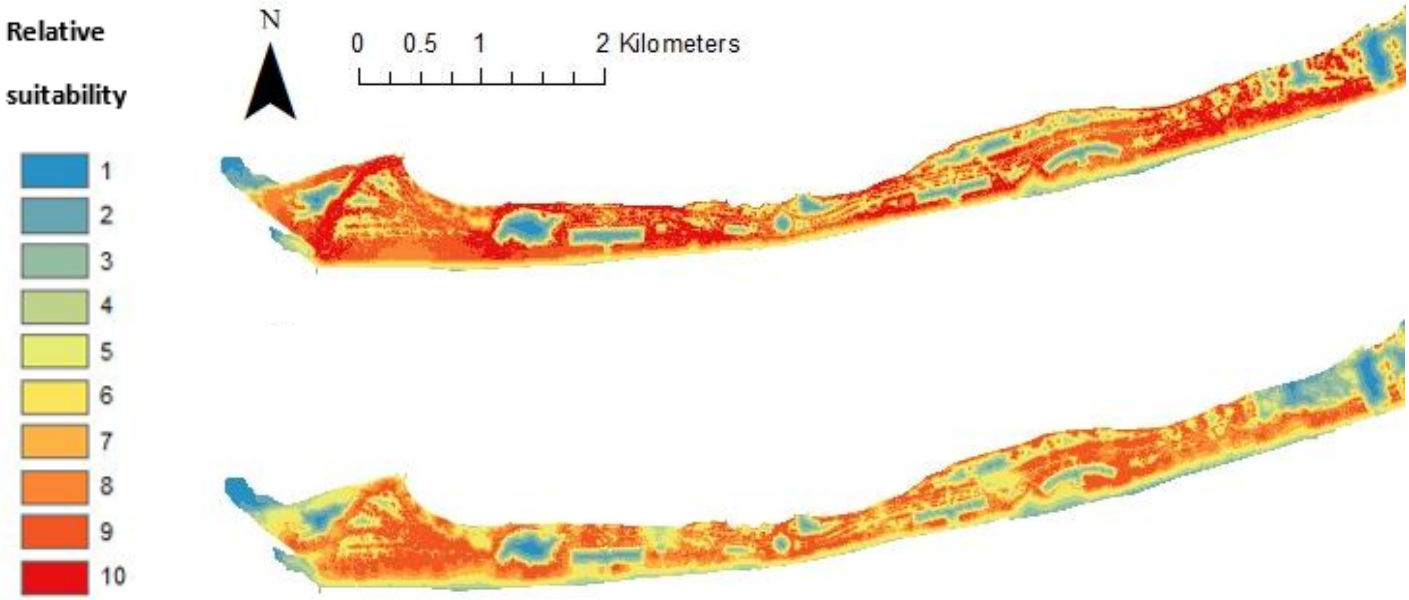


Figure G-1. Predicted habitat suitability (10 = highest suitability to 1 = lowest suitability) maps for male (top) and female (bottom) red foxes at Robert Moses State Park and Fire Island National Seashore's Lighthouse Tract, using 2016 landcover classification data.

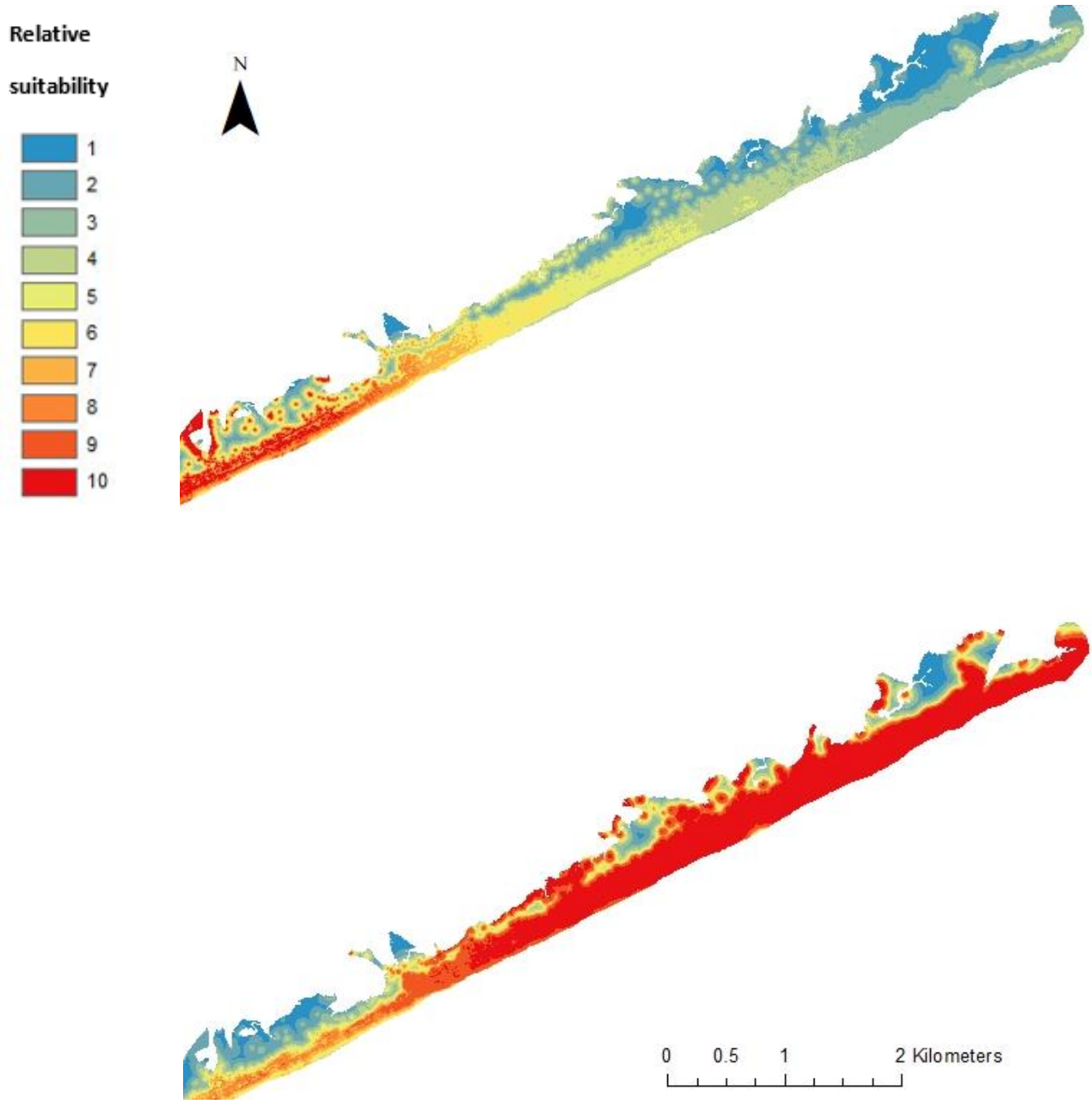


Figure G-2. Predicted habitat suitability (10 = highest suitability to 1 = lowest suitability) maps for male (top) and female (bottom) red foxes at Fire Island National Seashore (Watch Hill to Old Inlet), using 2016 landcover classification data.

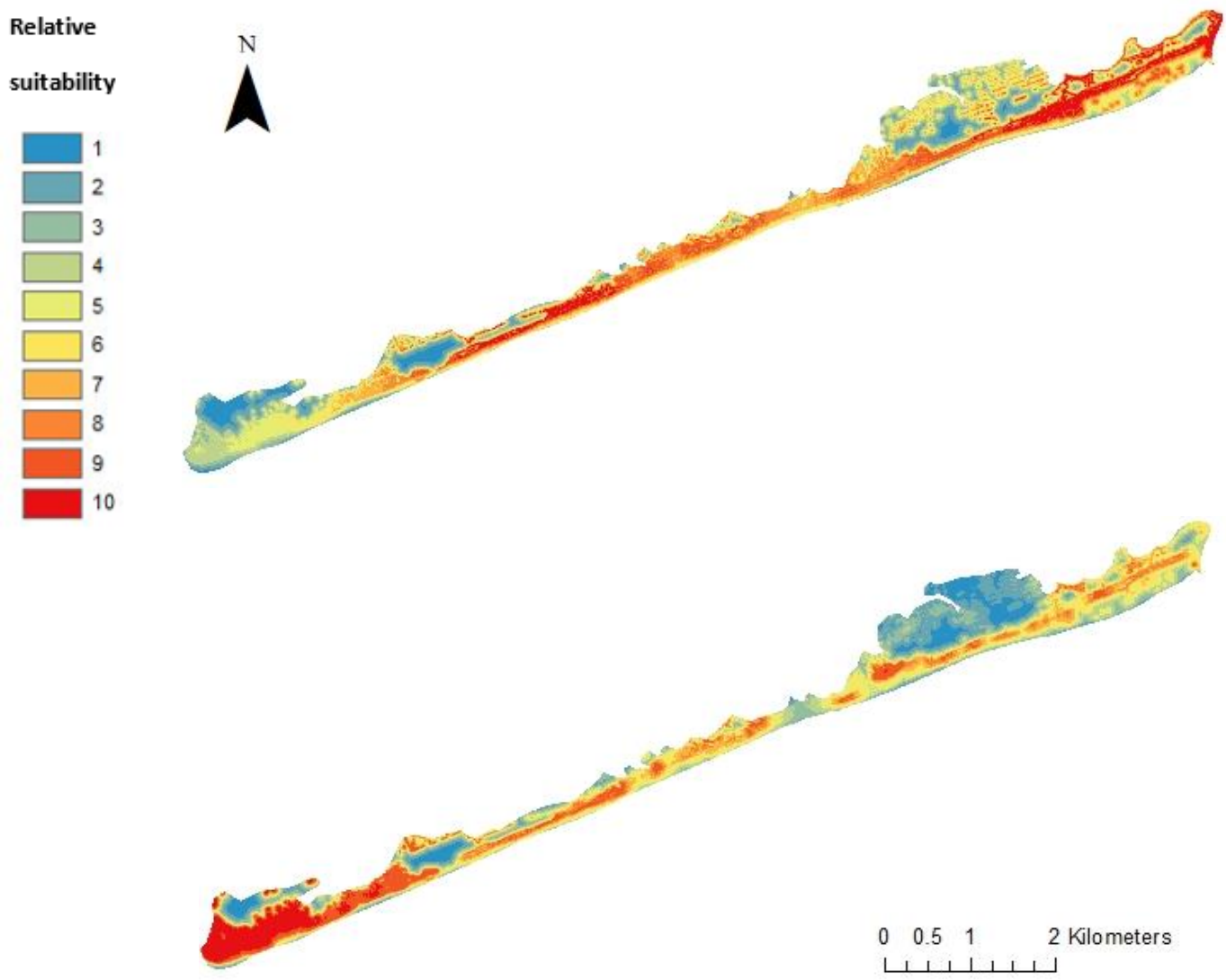


Figure G-3. Predicted habitat suitability (10 = highest suitability to 1 = lowest suitability) maps for male (top) and female (bottom) red foxes at Fire Island National Seashore (east of Old Inlet) and Smith Point County Park, using 2016 landcover classification data.

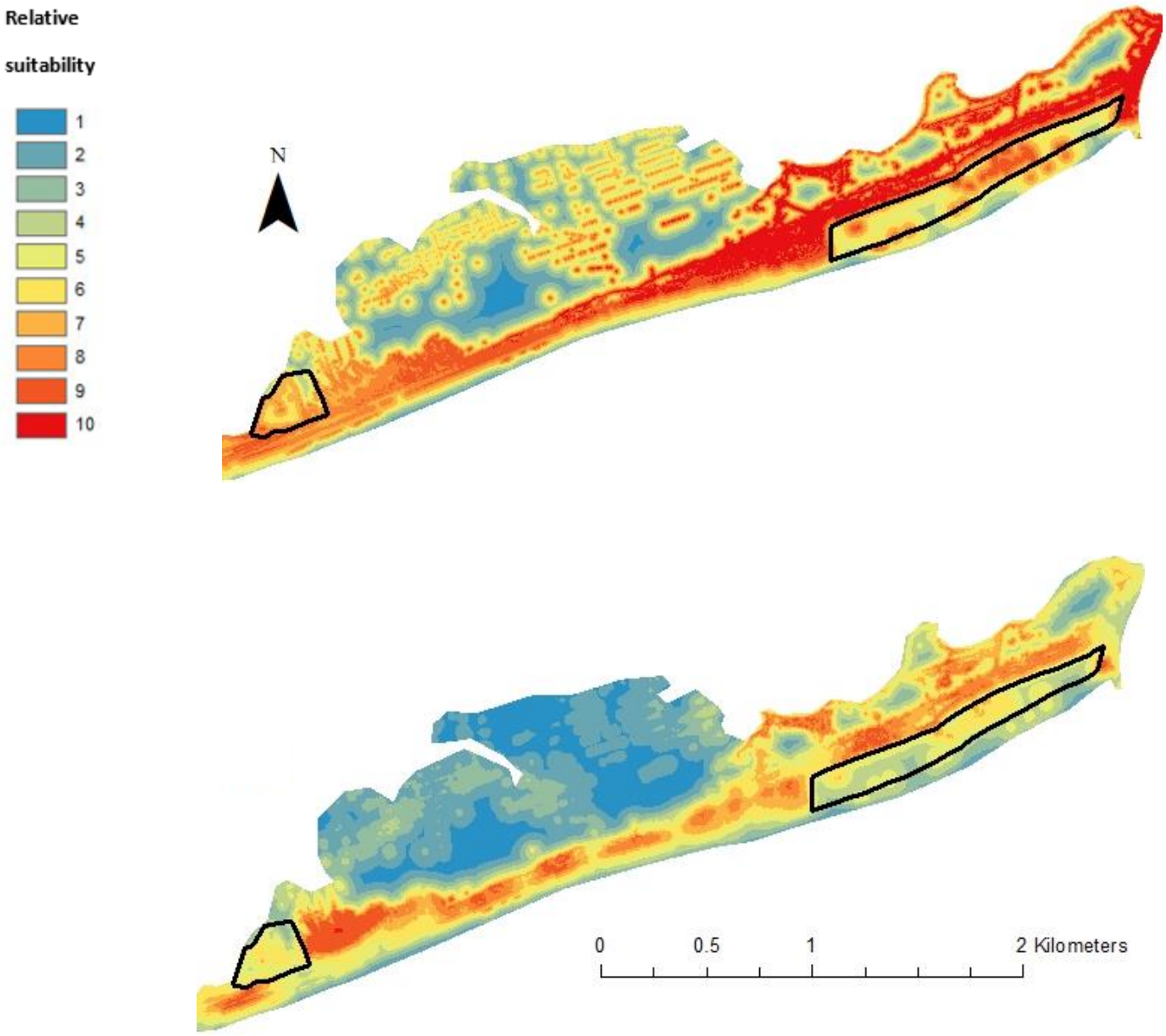


Figure G-4. Predicted habitat suitability (10 = highest suitability to 1 = lowest suitability) maps for male (top) and female (bottom) red foxes within the USACE restoration areas (outlined in black) at Smith Point County Park, using 2016 landcover classification data.

Appendix H. Supplementary material for Part 3, Section 4: Red fox dietary ecology in 'Effectiveness monitoring and targeted studies in support of FIMI conservation measures.'

Table H-1. Frequency of occurrence for each item at the lowest identifiable taxonomic level from fecal dietary analysis of confirmed and probable fox scat samples (n=297), ranked from highest to lowest, from Fire Island, NY 2016–2019.

Lowest identifiable taxonomic level	Frequency of Occurrence
Unknown vegetation	0.62
Unknown Mammalia	0.27
Unknown Decapoda	0.25
Unknown Aves	0.23
Unknown Coleoptera	0.22
Unknown	0.22
Unknown Insecta	0.15
Anthropogenic	0.15
Unknown fish	0.12
Odocoileus virginianus	0.11
Phyllophaga spp.	0.11
Dermacentor variabilis	0.10
Unknown Poaceae	0.09
Microtus spp.	0.08
Sylvilagus floridanus	0.07
Unknown Rodentia	0.07
Procyon lotor	0.06
Unknown Passeriformes	0.05
Unknown Anatidae	0.05
Rattus spp.	0.04
Microtus pennsylvanicus	0.04
Ondatra zibethicus	0.04
Unknown Scarabaeidae	0.04
Napaeozapus insignis	0.03
Unknown shellfish	0.03
Peromyscus spp.	0.03
Sciurus carolinensis	0.03
Unknown Carabidae	0.03
Unknown Malacostraca	0.03
Peromyscus leucopus	0.02
Rattus norvegicus	0.02
Unknown Chiroptera	0.02
Didelphis virginiana	0.02
Larus spp.	0.02
Libinia emarginata	0.02
Mustela spp.	0.02
Prunus maritima	0.02

Lowest identifiable taxonomic level	Frequency of Occurrence
Unknown Fagaceae	0.02
Scalopus aquaticus	0.01
Unknown Bivalvia	0.01
Unknown Carnivora	0.01
Unknown Cricetidae	0.01
Unknown insect	0.01
Unknown Ixodidae	0.01
Unknown Soricidae	0.01
Amblyomma americanum	0.01
Blarina brevicauda	0.01
Melanotus spp.	0.01
Mustela vison	0.01
Rattus rattus	0.01
Unknown Anseriformes	0.01
Emerita talpoida	0.01
Ixodes scapularis	0.01
Mus musculus	0.01
Peromyscus maniculatus	0.01
Sorex cinereus	0.01
Unknown Charadriidae	0.01
Unknown Galliformes	0.01
Unknown Gruiformes	0.01
Unknown Hymenoptera	0.01
Unknown Laridae	0.01
Unknown Lucanidae	0.01
Unknown Muridae	0.01
Unknown Squamata	0.01
Zapus hudsonius	0.01
Acritus spp.	<0.01
Blarina spp.	<0.01
Calasoma scrutator	<0.01
Colinus virginianus	<0.01
Dendroctonus spp.	<0.01
Dermacentor spp.	<0.01
Dysdera crocata	<0.01
Epilnacha borealis	<0.01
Felis cattus	<0.01
Mustela frenata	<0.01
Pinus spp.	<0.01
Prunus spp.	<0.01
Pterostichus spp.	<0.01
Unknown Accipitridae	<0.01
Unknown Amphipoda	<0.01

Lowest identifiable taxonomic level	Frequency of Occurrence
Unknown Charadriiformes	<0.01
Unknown Columbiformes	<0.01
Unknown Curculionidae	<0.01
Unknown Dipodidae	<0.01
Unknown Elateridae	<0.01
Unknown Falconiformes	<0.01
Unknown Formicidae	<0.01
Unknown Histeridae	<0.01
Unknown Mustelida	<0.01
Unknown Ocypode	<0.01
Unknown Odonata	<0.01
Unknown Orthoptera	<0.01
Unknown Parulidae	<0.01
Unknown Talitridae	<0.01