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Natural Resource Stewardship and Science

Technical Synthesis Report for Physical and Ecological Resources at Fire Island National Seashore

Natural Resource Report NPS/FIIS/NRR—2017/1415

ON THIS PAGE
Photograph of the breach at Fire Island National Seashore.
Photograph courtesy of Charles Flagg, June 24, 2014.

ON THE COVER
Photograph of the breach at Fire Island National Seashore.
Photograph courtesy of John Vahey, April 18, 2013.

Technical Synthesis Report for Physical and Ecological Resources at Fire Island National Seashore

Natural Resource Report NPS/FIIS/NRR—2017/1415

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Executive Summary

In October 2012, a breach was formed in the National Park Service (NPS) Otis Pike Fire Island High Dune Wilderness as a result of storm effects from Hurricane Sandy. The NPS is preparing an environmental impact statement (EIS) to evaluate alternatives for managing the breach. Because the breach had existed for less than 3 years at the initiation of the EIS project, much of the research relating to the breach was or is still underway. To support the development of the EIS, existing and ongoing research pertaining to the pre-breach and post-breach conditions in Great South Bay and surrounding areas was collected, compiled, and synthesized into this technical synthesis report. This report is a compilation of the best available information and describes the current state of the science for the physical and natural resource issues specific to Great South Bay and surrounding areas, as identified by NPS.

This technical synthesis report presents a synthesis and summary of published and unpublished studies, and information gained from discussions with subject matter experts. The report is divided into two main resource areas: (1) a physical resources section, and (2) a marine and estuarine resources section. The physical resources section summarizes information regarding hydrodynamic processes, water quality properties, waves, sediment transport, breach geomorphology, and long-term issues such as climate change and sea level rise. The marine and estuarine resources section provides a summary of information on water quality and phytoplankton, wetlands, submerged aquatic vegetation, benthic communities, hard clams, finfish and decapods, and ecosystem structure and processes. The resource areas covered under each chapter were identified as the resource areas most likely to experience and elicit responses to potential breach-related conditions. The information contained in this synthesis report will provide the scientific foundation for the EIS.

Effects of the breach on high tide water levels in Great South Bay from daily tidal fluctuations and small surge events were evaluated using hydrodynamic modeling and analyses of tide gage data. Both modeled and measured data indicate that high tide water levels follow a gradient, with small increases in water levels in the western and central parts of Great South Bay and minimal changes in the eastern parts of the bay. The greatest change in high tide water levels was observed near Lindenhurst, in western Great South Bay, where modeling and tide gage data indicate elevations have increased between 2.0 and 2.5 centimeters (0.8 and 1.0 inches) since the breach occurred. Elsewhere in central and eastern Great South Bay, modeling and measured data show that high tide water level increases have been very small at less than 0.8 centimeters (0.3 inches).

Stage frequency curves generated from numerical modeling of pre- and post-breach conditions indicated a maximum water level increase likely under the 100-year return period scenario (1% annual chance) of 60 centimeters (24 inches) for the Connetquot River area of the central Great South Bay. Under the same 100-year return period scenario, stage frequency curve data exhibited increases of 20 to 40 centimeters (7.9 to 15.7 inches) in other areas of Great South Bay and Moriches Bay. The stage frequency curves, originally based on model simulations with a breach nearly 2.5 times larger than the current opening, were adjusted using model runs on a breach configuration measured in June 2014. The 100-year return period water levels are predictions based on statistical

analyses of numerical model simulations of a dynamic coastal system. While absolute values of storm related water levels are difficult to predict, the order of magnitude and spatial distribution of the modeled increases is considered reliable.

Geomorphic impacts due to waves and sediment transport as a result of the breach are localized to a zone approximately 0.5 to 1.0 kilometer (0.3 to 0.6 mile) from the opening, primarily on the ocean facing western or down-drift side of the breach. Shoreline change data suggest that the breach causes little interruption in longshore sediment transport processes, although studies regarding directions of sand transport through the breach have not been conducted and model results regarding ebb/flood dominance are contradictory.

Wave conditions inside Great South Bay have not been impacted by formation of the breach as the limited width of the breach and the shallow nature of the flood-tidal delta complex do not allow wave propagation from the ocean through to Great South Bay. Wind generated waves within the bay have not been impacted because increases in water levels due to the breach are very small. The cross-sectional area of the breach increased during the first two years after formation, with only small variations in width occurring beyond the two-year time frame. Following initial formation, the main channel of the breach migrated gradually to the west. It is expected that the extent of westward migration will be limited by the location of erosion resistant geologic deposits located approximately 1.5 kilometers (0.9 miles) west of the May 2015 breach centerline. Further, the breach resulted in a change in general circulation patterns in eastern Great South Bay from stationary eddies to a pattern of mean through-flow moving from east to west. This shift in general circulation patterns has resulted in a decrease in water residence times for Great South Bay as a whole, and Bellport Bay in particular.

Water quality in Great South Bay has improved since formation of the breach. Salinities have increased between 2 and 5 practical salinity units (psu) in the central and eastern parts of the bay. Water temperature in the summer has decreased by 3°C, dissolved nitrogen has decreased by 0.2 milligrams per liter (mg/L), water clarity has improved, and chlorophyll-a concentrations have decreased from a range of 18 to 42 micrograms per liter (µg/L) prior to the breach, to a range of 6 to 15 µg/L in 2014, and a range of 26 to 35 µg/L in 2015. The frequency and intensity of brown tide has decreased in areas of eastern Great South Bay where water quality has improved and following bloom events, brown tide cells are cleared from the bay more quickly by transport to the ocean through the breach. Species composition of phytoplankton shifted after the breach occurred, and now is dominated by chain-forming diatoms with larger cell sizes. Central Great South Bay has experienced increased brown tide bloom frequency and intensity since the breach formed, potentially a result of changes observed in general water circulation patterns in Great South Bay.

Coastal habitat availability for Great South Bay fauna has changed since the formation of the breach. Field surveys indicate that beds of eelgrass have become established east of the breach in response to improvements in water quality, and new sand platforms have formed in the flood-tidal delta as a result of the accumulation of sandy sediment in overwash areas. The newly formed sand platforms provide opportunities for future establishment of marsh habitats, the type typically dominated by the marsh plant *Spartina alterniflora.* The potential for colonization by marsh plants would be more favorable if the flood-tidal delta becomes less dynamic (i.e., less shifting sediment). The post-breach improvements observed in water quality could further encourage marsh and seagrass bed development. Coastal vegetation is important for nutrient and carbon cycles, food resources for fish and waterfowl, nursery habitat and refuge space for juvenile fish, sediment stabilization, and water quality; benefits provided by sequestering nutrients and trapping suspended sediments and organic matter.

Changes in the faunal community of Great South Bay have also been observed since the breach formed, particularly in Bellport Bay, Narrow Bay, and western Moriches Bay, where water quality has improved. Post-breach surveys indicate that total abundance and diversity of finfish species has generally increased. The observed changes are attributed to a response to improvements in water quality, moderated summer and winter water temperatures, and increased habitat availability (i.e., increased submerged aquatic vegetation density and eelgrass). However, declines observed for some species, such as blue crab (*Callinectes sapidus*), are a potential result of increased salinities; while declines observed for grass shrimp (*Palaemonetes vulgaris*) were attributed to increased predation by fish foraging in seagrass habitats near the breach. Hard clam growth rate and condition index have both improved in Bellport Bay in response to post-breach improvements in water quality and food quality, and the ameliorating effect that the breach has had on brown tide blooms there. However, in central Great South Bay the post-breach circulation patterns create conditions that favor brown tide blooms and have negatively affected hard clams.

Post-breach changes in the benthic community are likely to have occurred, but have not yet been studied. Benthic communities are sensitive to sediment grain size and water quality. Overwash and development of flood shoals consisting predominantly of sand have likely increased the sediment grain size in the vicinity of the breach. The environmental conditions resulting from the breach favor high-flow, high-salinity adapted benthic communities comparable to those observed in the vicinity of other Great South Bay inlets.

Prior to breach formation, the Great South Bay was characterized as an immature ecosystem with lower connectivity to the ocean, thus limiting the health and stability of the bay system. The post-breach changes documented in the Great South Bay and resulting from increased ocean connectivity include improvements in water quality, temperature moderation, increased species richness, and abundance, and formation of new aquatic habitats, including eelgrass beds. Further, marsh grass is more abundant and could potentially colonize overwash areas and the flood-tidal delta, which would support increased fish and invertebrate production. Other metrics of ecosystem maturity, such as food web complexity and the abundance of suspension feeders and upper trophic levels have not been evaluated post-breach and are not known at this time, but may have also been affected by the breach. As a result of the post- breach changes in the Great South Bay, the ecosystem is more characteristic of a mature ecosystem, exhibiting increased species richness and abundance, and more desirable characteristics such as improved system health, stability, and resilience to disturbance.

List of Acronyms

ADCIRC	ADvanced CIRCulation model
BOC	breach open condition
CI	condition index
EIS	environmental impact statement
FIMP	Fire Island Inlet to Montauk Point Reformulation Study
FVCOM	Finite Volume Coastal Ocean Model
HWM	high water mark
LISHORE	Long Island SHORE monitoring stations
µg/L	microgram per liter
mg/L	milligram per liter
mL	milliliter
mm/yr	millimeters per year
NOAA	National Oceanic and Atmospheric Administration
NPS	National Park Service
NYSDEC	New York State Department of Environmental Conservation
ppt	parts per thousand
psu	practical salinity units
SAV	submerged aquatic vegetation
SBEACH	Storm-induced BEAch CHange model
SET	surface elevation table
SME	subject matter expert
SoMAS	School of Marine and Atmospheric Sciences
SWAN	Simulating WAves Nearshore model
TNC	The Nature Conservancy
USACE	United States Army Corps of Engineers
USGS	United States Geological Survey

Introduction

In October 2012, Hurricane Sandy made landfall in the United States near Brigantine, New Jersey. The storm caused wave and flood related damages in New Jersey and New York, and formed a breach in the National Park Service (NPS) Otis Pike Fire Island High Dune Wilderness located on Long Island (Figure 1). The existing Breach Contingency Plan (USACE 1996) guides the response for breaches that form along Fire Island, with breach closure typically occurring immediately after the formation of the breach; however, the Breach Contingency Plan provides an exception for breaches that form in the wilderness. These breaches are to be studied to determine if natural closure will occur. Federal wilderness areas are wild, undeveloped federal lands that have been designated and protected by congress. The legislation creating the wilderness area (Public Law 96-585) directs the Seashore to manage this area to preserve the wilderness character. The legislation also directs the NPS to refrain from interfering with natural processes that would typically occur within a barrier island, including breaches. The legislation does not preclude closing a breach in the wilderness if there is a need to do so; however, the *Fire Island National Seashore Wilderness Management Plan* (NPS 1983) stipulates that an environmental impact statement (EIS) must be prepared and public review and comment on alternatives must be conducted before such a decision would be made. For this reason, the NPS is preparing an EIS to evaluate alternatives for managing the breach.

As a result of the breach, numerous studies were initiated by researchers to better understand the dynamics of the breach and the effects of the breach on various elements of the Great South Bay ecosystem. As noted above, breaches along Fire Island are typically closed immediately, so this offered researchers a rare opportunity to study the dynamics of the breach following its formation and the effects of the open breach on the bay ecosystem. Much of the research relating to the breach is still underway. In order to access the most current scientific information and to reach consensus among researchers on resource issues, NPS elected to prepare this technical synthesis report to compile and document the best available data and describe the current state of the science for the physical and natural resource issues, as identified by NPS. This information will provide the scientific foundation for the EIS.

To collect the information needed for this technical synthesis report, NPS researchers and consultants developed a program designed to collaborate with subject matter experts (SMEs) and document ongoing research. SMEs consisted of university professors, student scientists, and postdoctoral researchers; federal and state agency staff, listed in Table 1. The program consisted of initial data requests, review of available information provided by SMEs and obtained from the literature, and a workshop to process and discuss the information obtained, all leading to the development of this report.

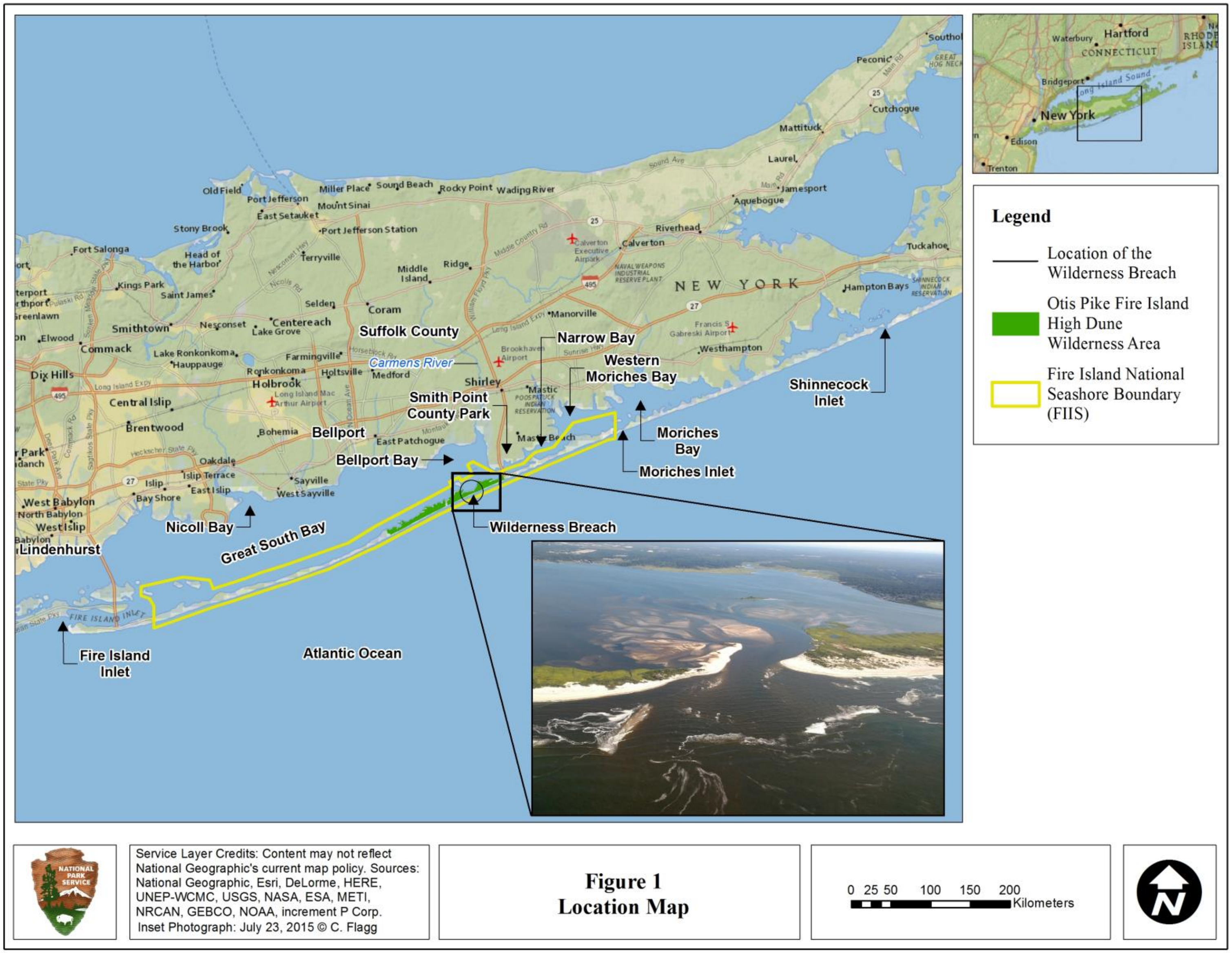

Figure 1. Location Map.

Table 1. Names and affiliations of workshop attendees.

Name	Affiliation	Name	Affiliation
Alan Fuchs	New York State Department of Environmental Conservation	Karl Nordstrom	Rutgers University
Anita Struzinski	EA Engineering, Science, and Technology, Inc., PBC	Kelly Fellner	National Park Service
Carrie McCabe	US Army Corps of Engineers	Kim McKown	New York State Department of Environmental Conservation
Charles Flagg	Stony Brook University	Kirk Bosma	Woods Hole Group
Charles Roman	National Park Service	Lee Terzis	National Park Service
Cheryl Hapke	US Geological Survey	Leslie Fields	Woods Hole Group
Chip Paterson	Industrial Economics, Inc.	Lindsay Ludwig	Industrial Economics, Inc.
Chris Gobler	Stony Brook University	Lisa Methratta	EA Engineering, Science, and Technology, Inc., PBC
Chris Olijnyk	National Park Service	Lynn Bocamazo	US Army Corps of Engineers
Chris Soller	National Park Service	Lynn Koontz	National Park Service
Claudia Hinrichs	Stony Brook University	Maarten van Ormondt	Deltares
Dawn McReynolds	New York State Department of Environmental Conservation	Mary Foley	National Park Service
Debra Barnes	New York State Department of Environmental Conservation	Michael Frisk	Stony Brook University
Elizabeth Rogers	National Park Service	Mike Bilecki	National Park Service
Heidi Clark	Woods Hole Group	Morgan Elmer	National Park Service
Howard Ruben	US Army Corps of Engineers	Morgan Gelinas	EA Engineering, Science, and Technology, Inc., PBC
Jacki Katzmire	National Park Service	Patti Rafferty	National Park Service
Janet Nye	Stony Brook University	Rafael Canizares	Moffatt & Nichol
Jill Olin	Stony Brook University	Robert Cerrato	Stony Brook University
Jim Neumann	Industrial Economics, Inc.	Steve Heck	Stony Brook University
John Stewart	National Park Service	Suzie Boltz	EA Engineering, Science, and Technology, Inc., PBC
Kaetlyn Jackson	National Park Service		

In January 2016, NPS hosted the workshop, bringing together the SMEs and providing an opportunity for the SMEs to discuss the current science in the context of the issues that would potentially drive the EIS decision. Results from discussions were compiled into daily meeting summaries, which were used in the development of the draft technical synthesis report. The final draft provides the technical details on the current state of the science for physical and ecological resources related to the breach, and, as noted above, provides the scientific foundation for the EIS.

Physical Resources

Physical processes in the vicinity of the Fire Island wilderness breach and Great South Bay play an important role in defining impacts of the breach on the ecological resources, the surrounding natural and developed lands, and the communities that inhabit the south shore of Long Island. Prior to Hurricane Sandy, open ocean and inlet dominated physical processes were active along the barrier beach and at the two tidal inlets, Fire Island and Moriches. Storm generated overwash processes were active along the lower more vulnerable areas of the barrier. Conditions within Great South Bay were controlled primarily by estuarine processes, with tidal exchange controlled by the two established inlets located approximately 50 kilometers (31 miles) apart (Figure 1). With formation of the Fire Island wilderness breach, a dynamic new environment was formed that increased the connectivity between Great South Bay, western Moriches Bay, and the Atlantic Ocean, allowing greater influence to the bays from ocean and inlet processes.

The breach has resulted in changes to physical factors that affect the barrier beach, nearby shoals, and the Great South Bay estuary. The primary physical factors include hydrodynamics, water quality properties, waves, sediment transport, and island geomorphology. Superimposed on the short-term changes to the system caused by the breach are the longer-term impacts caused by climate change and sea level rise. The information presented in this technical synthesis report addresses the current scientific knowledge regarding changes to these physical factors as a result of the breach, as well as impacts to Great South Bay and the surrounding areas.

Hydrodynamics

Changes in the hydrodynamics of Great South Bay and Moriches Bay as a result of breach formation have the potential to affect water levels in the bays. This includes daily changes in water levels caused by astronomical tides as well as storm-generated water levels. The tidal prism of the breach, or the volume of water exchanged during a tidal cycle excluding any contributions from freshwater inflows, is an important component of the hydrodynamics that can also influence bay water levels. Prior to formation of the breach, the hydrodynamics and water levels were controlled primarily by water exchange through Fire Island and Moriches Inlets. However, with the addition of the Fire Island wilderness breach, there is potential for alteration to these processes. General circulation patterns in the bays may also be affected, and this in turn can influence basic water quality parameters such as salinity, water temperature, water clarity, and nutrient concentration.

Synthesis of Hydrodynamics Pre- versus Post-Breach

Effects of the breach on hydrodynamics have been studied by three primary groups: the US Army Corps of Engineers (USACE), US Geological Survey (USGS) in cooperation with Deltares, and the State University of New York Stony Brook University. While the goals of the three studies were different, they each provide valuable information on changes to the hydrodynamics of the bays along the south shore of Long Island (Great South Bay, Moriches Bay, Shinnecock Bay) as a result of the breach. A brief summary of each study is provided below followed by a summary of findings and conclusions regarding impacts of the breach on water levels and circulation in the bays.

US Army Corps of Engineers Numerical Model Studies

Numerical modeling of physical processes for the south shore of Long Island was undertaken by the USACE New York District in support of the Fire Island Inlet to Montauk Point Reformulation Study (FIMP). The purpose of the modeling was to determine storm impacts and bay flood levels along the south shore of Long Island between Fire Island Inlet and Montauk Point to inform decisions on long-term management of the FIMP area for storm damage reduction. Information on initial modeling conducted for FIMP is contained in a draft report entitled *Baseline Conditions Storm Surge Modeling and Stage Frequency Generation: Fire Island to Montauk Point Reformulation Study* (USACE 2006a). The work is also summarized by Canizares and Irish (2008), the primary modelers.

The modeling approach for FIMP incorporated a wide array of physical processes including winds, barometric pressure, astronomic tides, waves, morphologic response, and localized wind and wave setup. Specialized numerical models capable of simulating hydrodynamics, waves, and sediment transport were merged and used to evaluate surge elevations in Great South Bay and surrounding areas under a range of storm and breach scenarios. The modeling approach used the following four process models:

- WAVAD (i.e., WISWAVE) to determine extreme storm wave conditions,
- ADvanced CIRCulation model (ADCIRC) to simulate storm water levels in the ocean, nearshore, and areas seaward of the surf zone,

- Storm-induced BEAch CHange model (SBEACH) to estimate pre-inundation dune lowering, and
- Delft3D model suite to simulate storm water levels in the bay accounting for contributions from storm surge, waves, winds, overwash, and/or breaching.

Baseline conditions evaluated in initial modeling efforts were representative of the FIMP area topography in 2000. At that time three inlets (Fire Island, Moriches, and Shinnecock) connected the ocean with bays in the FIMP area (Figure 1). Both ADCIRC and Delft3D models were calibrated to astronomical tide and storm water levels collected at established gage locations. The ADCIRC calibration used four National Oceanic and Atmospheric Administration (NOAA) stations and one Long Island SHORE (LISHORE) station, and the Delft3D model was calibrated using 13 bay area measurement locations (6 in Great South Bay, 4 in Moriches Bay, and 3 in Shinnecock Bay). The Delft3D model was then validated by comparing model results with available high water marks (HWMs) and overwash and breaching data for two of the most significant storms of record: the September 1938 hurricane and the December 1992 nor'easter. The 1938 hurricane made landfall as a Category 3 hurricane on Long Island and represents one of the most powerful and deadliest hurricanes in New England history. The storm caused widespread overwash and created several breaches across the barrier island. The December 1992 nor'easter produced record high water levels across the northeastern United States. On Long Island the storm created two breaches east of Moriches Inlet. Model simulations of the 1938 hurricane produced breaching and overwash in areas similar to the observed storm impacts. Peak water levels from the 1938 simulation matched closely with HWMs recorded in eastern Shinnecock Bay, South Oyster Bay, and central Great South Bay (USACE 2006a). Model skill was lower in eastern Great South Bay and Moriches Bay where simulated water levels were 30 to 60 centimeters (12 to 24 inches) lower than the HWMs. Modeling of the 1992 nor'easter produced two breaches in the same area as the observed storm impacts. Peak water levels from the 1992 simulation either matched or underestimated by 60 centimeters (24 inches) the HWM data recorded in Moriches Bay and Great South Bay. Further to the west in Patchogue and Lindenhurst, the model results were within the reported range of HWMs, although the average reported HWMs were underestimated by approximately 60 centimeters (24 inches). Overall, the model simulations for these two historic storms were considered to provide realistic results, particularly when considering the uncertainty in the input hydrodynamic conditions and, more importantly, the pre-storm topographies.

The numerical models were used to simulate water levels in the bays under baseline conditions during 36 historical storms (14 hurricanes and 22 nor'easters), as well as 21 additional storms. These simulations showed that peak water levels in Great South Bay were produced by extratropical storms, while tropical storms generated the highest water levels in Moriches and Shinnecock Bays. Peak simulated water levels in Great South Bay from the tropical storms were on the order of 20 centimeters (7.9 inches) lower than those from the extratropical storms. This was attributed to the lower efficiency transference characteristics of Fire Island Inlet. Surge associated with tropical storms that tend to pass through the area more quickly than nor'easters is significantly dampened at Fire Island Inlet due to its lower efficiency, while the surge generated by extratropical storms that typically last several tidal cycles increases the total volume of water passing through the inlet, resulting in higher peak water levels. Moriches Bay fills more rapidly with storm surge flows than Great South Bay because of its smaller size and more efficient inlet, and therefore Moriches Bay responds more quickly to fast moving extratropical storms.

For baseline conditions (pre-Sandy), only a small number of storm simulations resulted in significant contributions to water levels in the bays due to overwash/inundation of the barrier island. These same storms were the only ones to produce full breaches or partial breaches of the barrier. Full breaches were considered to be storm-induced cuts through the barrier where scour depth was at or below mean low water. Partial breaches were considered where the scour depth was between mean high water and mean low water. The wilderness area, and in particular the site of the breach that formed during Hurricane Sandy, was found to be the most vulnerable spot for full breaching in the FIMP study area, due in large part to the lower elevation of the dunes under baseline conditions. The storm modeling for the 1938 event also resulted in partial breaching at Smith Point County Park, Tiana Beach, and West of Shinnecock Inlet.

Stage-frequency relationships using the modeled water levels for all storm simulations were developed using the one-dimensional Empirical Simulation Technique. Peak water levels (storm surge and tides) at 28 stations in Great South Bay and Moriches Bay for the 6-, 10-, 25-, 50-, 73-, and 100-year return periods were developed for the pre-Sandy baseline conditions (Figure 2). For Great South Bay the 6-year return period peak water levels were found to be between 1.1 and 1.4 meters (3.6 and 4.6 feet), the 50-year return period peak water levels were between 1.2 and 1.5 meters (3.9 and 4.9 feet), and the 100-year return period levels were between 1.2 and 1.8 meters (3.9 and 5.9 feet) (NGVD29 adjusted for sea level rise to 2000) (Figures 3, 4, and 5). Stage frequency results in Moriches Bay were generally higher than those in Great South Bay since Moriches more readily responds to ocean conditions. Peak water levels in Moriches Bay were between 1.2 and 1.5 meters (3.9 and 4.9 feet) for the 6-year return period, between 1.8 and 2.1 meters (5.9 and 6.9 feet) for the 50-year return period, and between 2.0 and 2.3 meters (6.6 and 7.5 feet) for the 100-year return period (Figures 3, 4, and 5). Spatial variations in water levels within the bays were found to be consistent with the bay's geometry, inlet configurations, and exchange with adjacent water bodies.

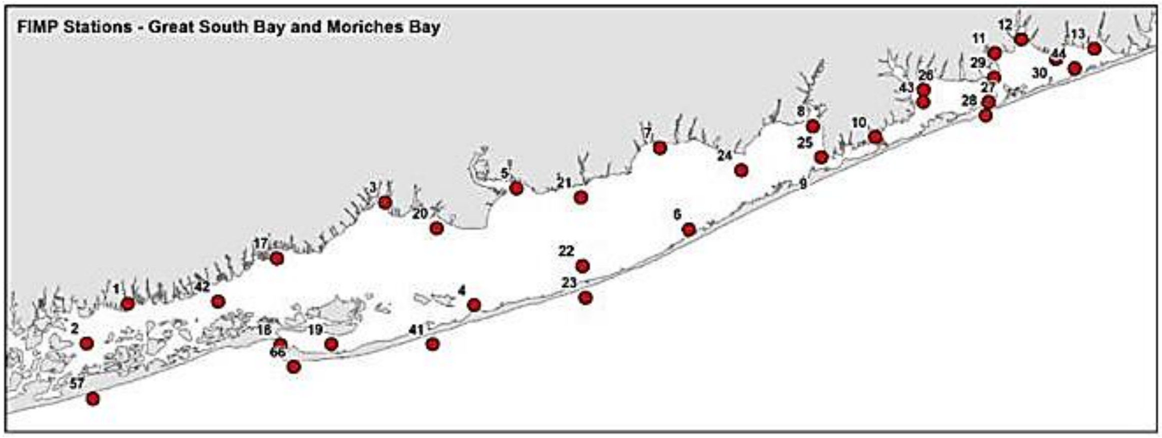

Figure 2. Storm water level output stations in Great South Bay and Moriches Bay from the USACE modeling (Moffatt & Nichol and Canizares 2015).

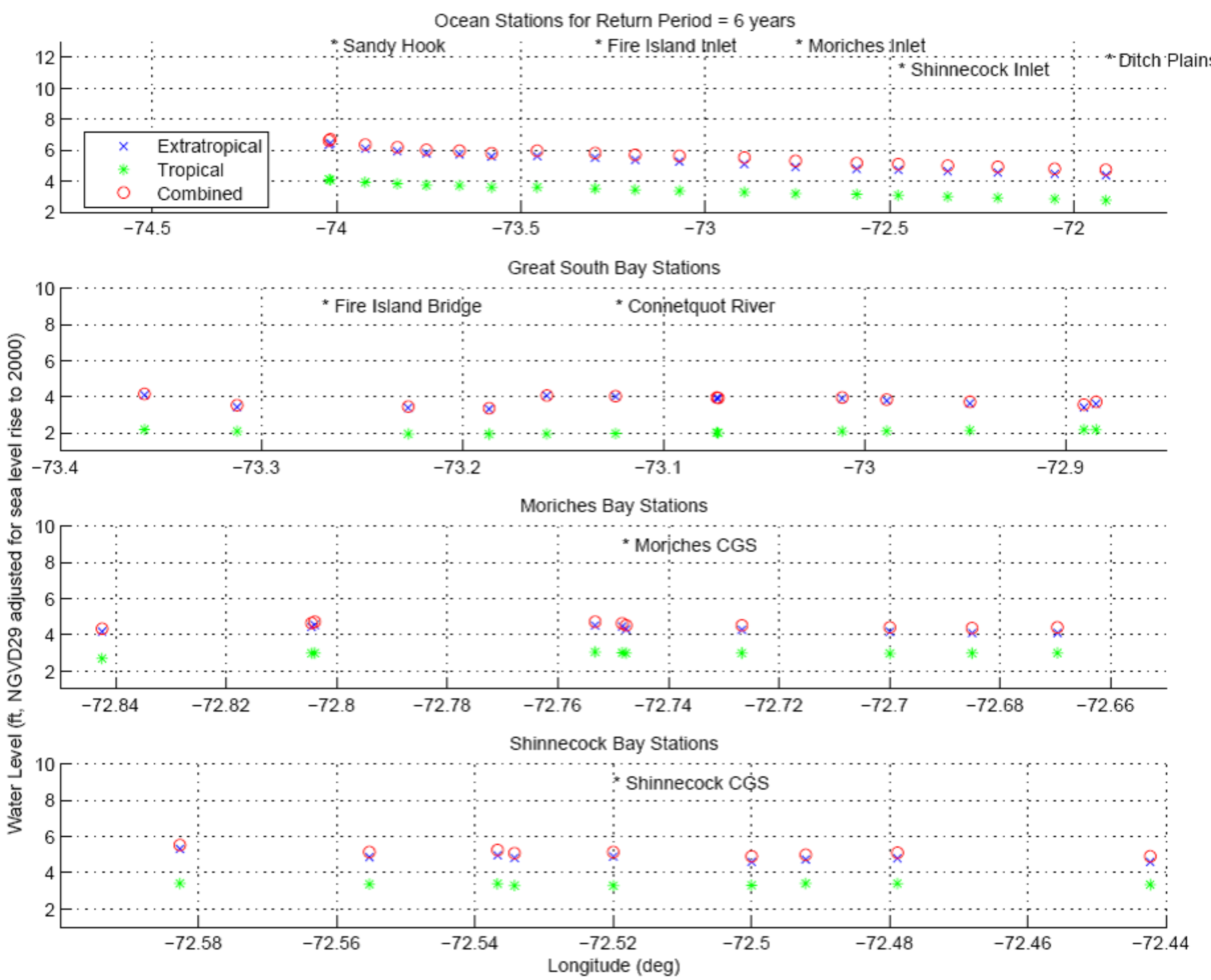

Figure 3. Spatial distribution of 6-year return period peak water levels (in feet) for baseline conditions (pre-Sandy) from USACE modeling (USACE 2006a).

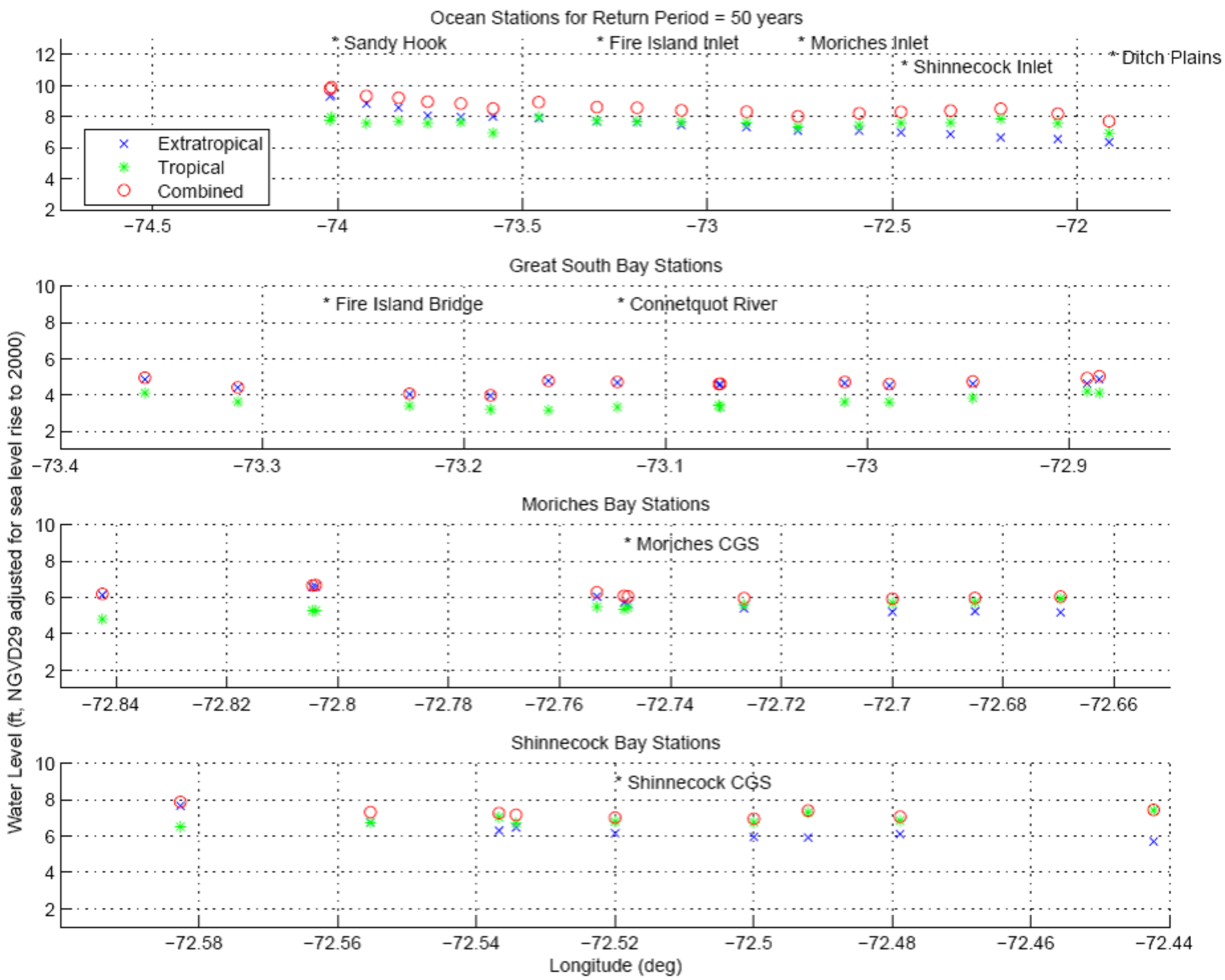

Figure 4. Spatial distribution of 50-year return period peak water (in feet) levels for baseline conditions (pre-Sandy) from USACE modeling (USACE 2006a).

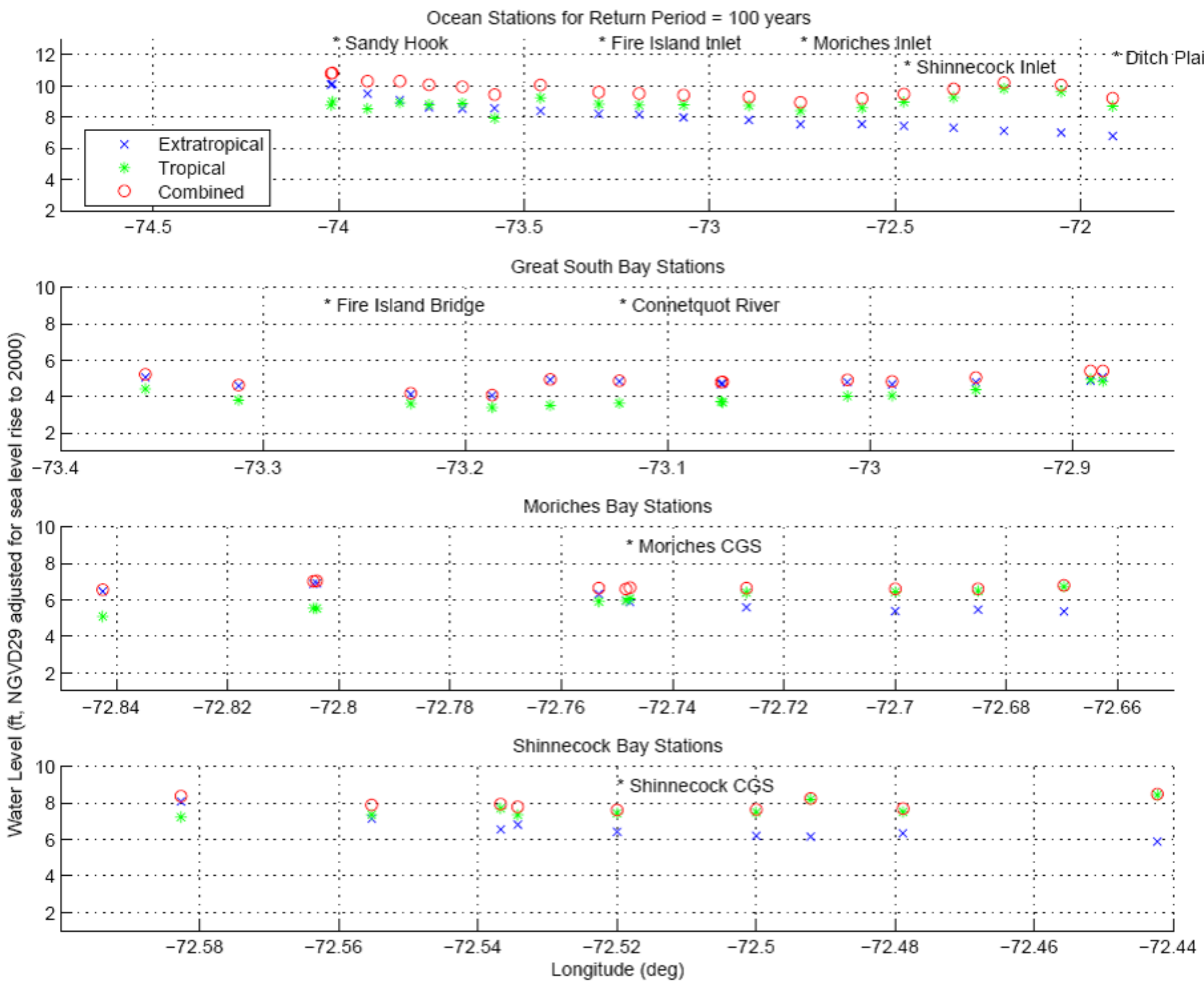

Figure 5. Spatial distribution of 100-year return period peak water levels (in feet) for baseline conditions (pre-Sandy) from USACE modeling (USACE 2006a).

Additional modeling scenarios were also conducted by the USACE to evaluate the effects of barrier island breaches on maximum water levels in the bays. The methodology used to select the breach locations and the different modeling scenarios is summarized in a February 2, 2006, document entitled *Summary of Development Approach and Draft Results for Baseline, Future Vulnerable, and Breach Closed Conditions Breach/Overwash-Frequency Relationships* (Moffatt & Nichol, Canizares, and Alfageme 2006), as well as a March 22, 2006 Memorandum prepared by Moffat & Nichol (Author Unknown 2006). At this time, 12 representative breach open scenarios were evaluated for six select storms. The smaller number of storms was considered sufficient to provide enough information to create stage frequency curves for the bays under the different breach open scenarios. The 12 breach open scenarios were comprised of 4 representative location-based scenarios with three possible breach sizes corresponding to estimated widths at 3, 6, and 12 months from breach formation. The widths were 762 meters (2,500 feet), 1,128 meters (3,700 feet), and 1,433 meters (4,700 feet) for the 3-, 6-, and 12-month scenarios, respectively. All scenarios assumed a breach depth of 2.1 meters (6.9 feet) (mean sea level), yielding cross-sectional areas of 1,600, 2,370, and 3,009 square meters for the 3-, 6-, and 12-month scenarios (17,250, 25,530, and 32,430 square feet).

The breach open condition (BOC)-1 scenario included a breach in eastern Great South Bay in the vicinity of the current wilderness breach and a second breach in western Shinnecock Bay. The other scenarios considered multiple breaches or openings in western and central Great South Bay, eastern Moriches Bay, and Shinnecock Bay (Table 2). Results from the model simulations were used to prepare stage frequency curves for bay stations in the FIMP area, given the different breach scenarios.

Table 2. Four location-based scenarios considered for breach open modeling simulations. Wilderness breach is represented by the Eastern Great South Bay location.

Breach Open Conditions for Numerical Simulations						
Breach Open Scenario	**Western Great South Bay**	**Central Great South Bay**	**Eastern Great South Bay**	**Eastern Moriches Bay**	**Western Shinnecock Bay**	**Shinnecock Bay**
BOC-1			X		X	
BOC-2	X			X	X	
BOC-3		X				X
BOC-4	X		X	X		X

Following formation of the wilderness breach in 2012, the USACE modeling was updated to validate the integrity of the earlier modeling efforts and to examine applicability of the modeling approach to the wilderness breach. This recent work was documented in a Draft Memorandum from Moffatt & Nichol dated September 11, 2015. The updated work included revalidation of the model with breach closed conditions and validation to conditions with the wilderness breach. Model simulations were conducted to evaluate impacts on tides and storm tides with various BOCs, and the stage frequency curves for bay water levels were updated to reflect the influence of the wilderness breach.

Post- Hurricane Sandy modeling was performed using updated versions of the Delft3D software and the updated Simulating WAves Nearshore (SWAN) model (Moffatt & Nichol and Canizares 2015). Revalidation was conducted using baseline conditions (breach closed) and the model was found to accurately reproduce tidal propagation in the bays, flow through the inlets (Fire Island and Moriches), and the effects of winds, waves, and surge propagation during the blizzard of 2003.

The updated model was validated by running a 2-year simulation from November 1, 2012, to November 1, 2014, using breach bathymetry from a June 2014 USGS survey (Nelson et al. 2016a). At the time of the 2014 survey the breach was approximately 305 meters (1,001 feet) wide at its narrowest point. The 2014 surveyed breach condition was used in a separate modeling study by the USGS (discussed later in this section), allowing for comparison of results from the two model studies. Model validation was performed for a 2-month period in early 2014 by comparing tidal constituents from four water level gages in Great South Bay with tidal constituents predicted by the model. The validation showed good agreement between the observed and modeled data. For storm conditions, the model showed a slight over-prediction in peak bay water level, by as much as 25 centimeters (9.8 inches) at one location, but was considered representative given uncertainties in model bathymetry and boundary conditions. It was noted that differences between modeled and observed storm water levels were consistent with those found with the Deltares model (van Ormondt et al. 2015). It was also noted that model bathymetry from the June 2014 survey could have caused the over-prediction in peak bay water levels during the first months of the simulation, since the breach grew in size, shape, and orientation rapidly between Hurricane Sandy and the time of the survey.

To assess impacts of the wilderness breach on tides and small storm tides in the bay, the 2-year simulation (November 1, 2012 to November 1, 2014) was repeated with "breach closed" conditions. The breach was found to have a very small effect on daily tidal fluctuations and small storm tides. Changes to the daily tide at Fire Island Inlet, Tanner Park, and Bellport (Figure 1), as determined by absolute changes in the modeled M2 tidal constituent and mean high water, were all less than 0.8 centimeters (0.3 inches). At Lindenhurst (Figure 1) the increase was slightly greater at approximately 2.7 centimeters (1.1 inches) (Table 3). These results are consistent with the Deltares modeling performed by van Ormondt et al. (2015).

Table 3. Impact to tides of breach open at the wilderness breach.

	Absolute Change in Centimeters (Inches)		Percent Change	
Station	**M2**	**Mean High Water**	**M2**	**Mean High Water**
Fire Island Inlet	0.2 (0.08)	0.2 (0.08)	0.9%	0.8%
Tanner Park	0.8 (0.3)	0.8 (0.3)	3.5%	3.5%
Bellport	-0.2 (-0.08)	-0.2(-0.08)	-1.3%	-1.2%
Lindenhurst	2.7 (1.1)	2.87 (1.13)	19%	19.0%

Model results showed that the wilderness breach had a similar effect on small storm tides (i.e., tides plus storm surge). Simulations on two small storms in December 2012 showed that peak storm tides at Lindenhust and Bellport were 2.5 to 7.6 centimeters (1.0 to 3.0 inches) higher with the breach open. Overall however, linear regression analyses on the entire 2-year simulation showed an increase in small storm peak water levels at Lindenhurst, and a slight decrease at Bellport.

Impacts of the breach on water levels during large storm events were evaluated by simulating six storm events. The storms selected were considered sufficient to update the stage frequency curves produced during the original 2006 modeling for the FIMP bay areas using the different breach open scenarios (i.e., 3-, 6-, and 12-month scenarios). Increases in maximum water level between the breach closed and the June 2014 BOC for these storm events were found to be as high as 25.4 centimeters (10 inches) during the smaller storm events and up to 55.9 centimeters (22.0 inches) during the larger storm events (Figures 6 and 7). The highest water levels generally occurred near the center of Great South Bay. The model predictions of increased water levels during storms represent order of magnitude increases rather than absolute values. The dynamic and constantly changing nature of the breach, including variations in morphology since the June 2014 survey which have decreased the width, hinder predictions of absolute water levels with a numerical model; however, the order of magnitude for water level increases shown by the model is considered reliable.

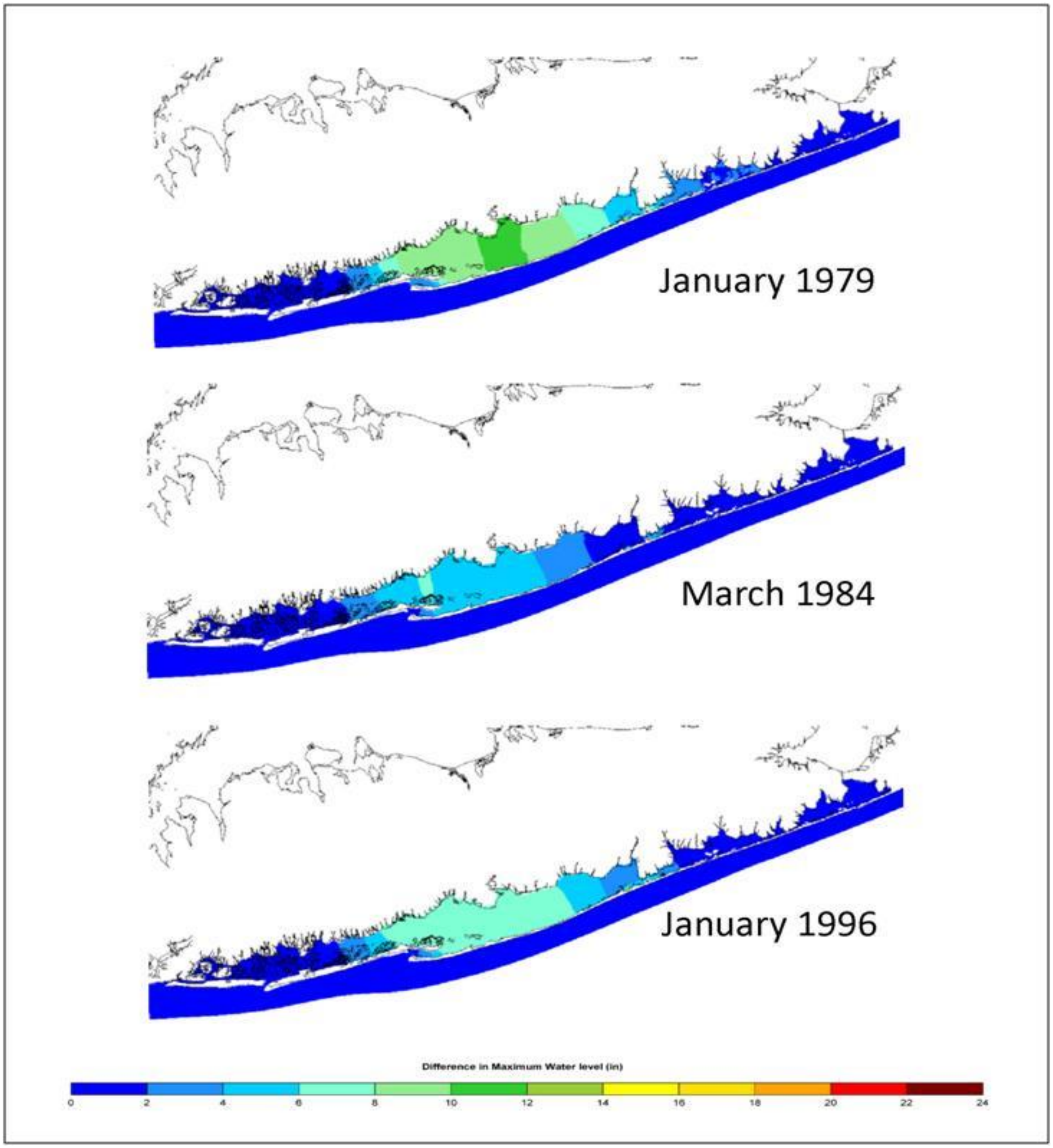

Figure 6. Difference in modeled peak water levels during small storms with and without breach (Moffatt & Nichol and Canizares 2015). Extratropical storms: January 1979, March 1984, and January 1996.

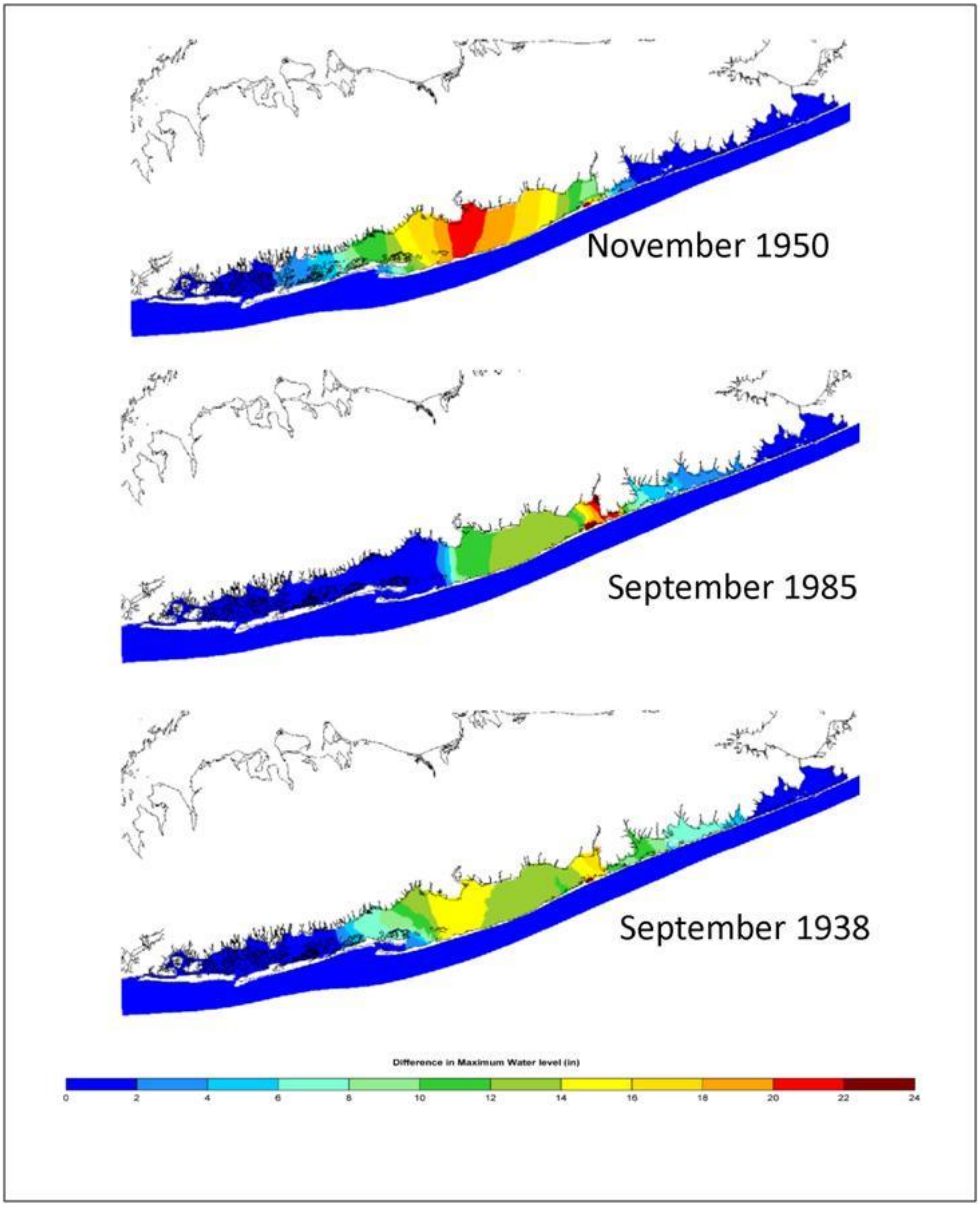

Figure 7. Difference in modeled peak water levels during large storms with and without breach (Moffatt & Nichol and Canizares 2015). Tropical storms: September 1938 and September 1985; Extratropical storm: November 1950.

Given the persistence of the wilderness breach since initial formation in 2012, the USACE revised the pre-Sandy baseline conditions to reflect the wilderness breach as of June 2014. Revised baseline stage frequency curves were based on the BOC-1 (3-month) curves (developed in 2006). Although the cross-sectional area of the BOC-1 (3-month) breach was nearly 2.6 to 3.2 times larger than the June 2014 breach, the model simulations showed similar impacts on bay water levels in terms of magnitude and spatial extent. Analyses comparing relative increases in bay water levels caused by the BOCs versus return period showed slightly greater water levels at higher return periods for the June 2014 breach. This was thought to be related to increased conveyance through the deeper and better defined channels represented in the June 2014 breach and model bathymetry, as compared to the simple rectangular breach geometry used to represent the BOC-1 (3-month) condition. Comparative analyses for all model stations in Great South Bay and Moriches Bay were used to identify adjustments for shifting the BOC-1 (3-month) stage frequency curves. Adjustments to the BOC-1 (3-month) curves varied by station and generally ranged between 0.9 and 10.1 centimeters (0.4 and 4 inches), with the greatest water level increases occurring in the vicinity of the wilderness breach (Figure 8).

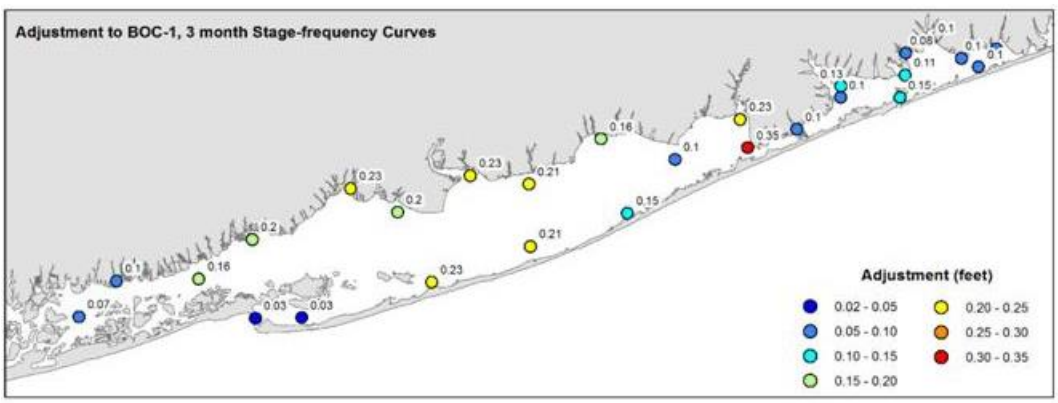

Figure 8. Adjustment values by station to BOC-1, 3-month stage frequency curves to reflect June 2014 breach open conditions.

Comparison between stage frequency curves for the 2006 and 2014 baseline conditions at stations throughout Great South Bay and Moriches Bay showed a maximum 100-year return period water level increase of 60 centimeters (23.6 inches) at Station 5 near the mouth of the Connetquot River (Figure 9). Increases elsewhere in Great South Bay and Moriches Bay for the 100-year return period were between 20 and 40 centimeters (7.9 and 15.7 inches).

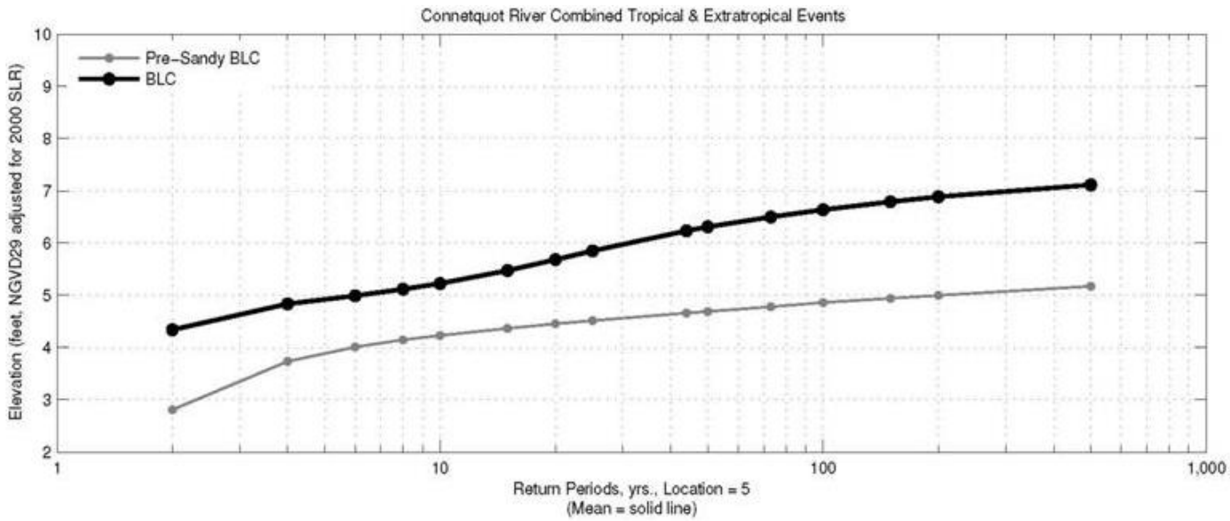

Figure 9. Stage frequency curve for Station 5 near the Connetquot River showing differences in water levels between the 2006 Pre-Sandy baseline condition (no breach) and the 2014 baseline condition (with breach) conditions (Moffatt & Nichol and Canizares 2015).

Geospatial data developed by the USACE from stage frequency curves for the 2-, 10-, and 100-year return period storms using the 2006 and 2014 baseline conditions show areas of increased flooding around Great South Bay and Moriches Bay that may result from the breach. When comparing baseline conditions for the 2-year storm event, the model predicts a potential increase in flooded area of approximately 3,825 acres (2014 baseline with 2-year storm minus 2006 baseline with 2-year storm). During the 10-year storm event the flooded area with the breach is approximately 970 acres greater than the pre-breach condition; 100-year event the flooded area with the breach is 2,790 acres greater than the pre-breach condition. The affected areas likely contain a mixture of land use types (e.g., residential, open space, commercial). A summary of impacts to the various land use types from increased flooding using the USACE model results is provided in the EIS.

To evaluate potential impacts to water levels from wider breaches, the USACE ran model simulations with the revised BOC-1 conditions for 6- and 12-month conditions. The 6-month condition used a breach width of 1,128 meters (3,700 feet) and the 12-month condition used a breach width of 1,433 meters (4,700 feet). All scenarios assumed a breach depth of 2.1 meters (6.9 feet) (mean sea level), yielding cross-sectional areas of 2,370 and 3,009 square meters, for the 6- and 12-month conditions respectively (25,530 and 32,430 square feet). Comparative analyses for all model stations in Great South Bay and Moriches Bay were used to identify adjustments for shifting the 6- and 12-month stage frequency curves developed previously in 2006. This approach made use of extensive modeling results from the 2006 study by identifying relative increases in bay water levels caused by the 6- and 12-month BOCs versus return period. Results from these simulations showed maximum water level increases of 80 centimeters (31 inches) for the 100-year return period (Bocamazo pers. comm. 2016). The 6- and 12-month breach scenarios represent openings 1.5 to 2.0 times larger than the BOC-1 (3-month) condition, and approximately 3.6 to 4.6 times larger than the June 2014 breach. Given the history since formation in 2012, enlargement of the breach 3.6 to 4.6

times its June 2014 width would represent a dramatic change in breach evolution. The modeled scenarios may be more representative of future storm water levels under climate change and sea level rise conditions.

US Geological Survey/Deltares Model Studies

Post- Hurricane Sandy modeling of Fire Island and the surrounding areas has been conducted by Deltares with geospatial data required for model runs provided by the USGS. The work has focused on studying the changes in morphology of the breach and its impacts on the neighboring areas. Modeling capabilities for predicting stability of future breaches on Fire Island or other similar environments were also developed. Information on the numerical modeling aspect of the work is contained on the Deltares Fire Island web site (www.cosmos.deltares.nl/FireIsland/index.html) and in a conference proceedings article entitled *The Effects of Geomorphic Changes during Hurricane Sandy on Water Levels in Great South Bay* (van Ormondt et. al. 2015).

The model approach included a series of nested hydrodynamic and spectral wave models in combination with a model for simulating sediment transport and morphological change. The following combinations of numerical models were used:

- DFlow-FM to simulate tidally generated and storm induced water levels in the ocean and nearshore areas,
- SWAN to develop 2D spectral wave conditions,
- XBeach to simulate dune and beach erosion and overwash volumes, and
- Delft3D-FLOW/SWAN to simulate water levels, waves, and changes in morphology in Great South Bay (referred to as Great South Bay model).

Forcing at the boundaries was accomplished by imposing water levels and waves from a series of larger scale models. The Delft3D models were used during relatively calm periods when offshore waves heights were less than 3.5 meters (11.5 feet), and the XBeach model was used during storms. Model simulations were initially run to reproduce conditions during Hurricane Sandy. Subsequent model runs were conducted for a 2-year period from November 2012 (post- Hurricane Sandy) to October 2014.

Model results for the Hurricane Sandy simulation showed agreement between computed and observed water levels at the Battery, New York, NOAA station, although the peak of the surge was underestimated by 27 centimeters (10.6 inches). The storm simulation also showed the highest volumes of overwash (water) at the western end of Fire Island, the eastern portion of the island near the wilderness breach, and just west of Moriches Inlet. Overall however, the total volume of water flowing over the barrier beach was small compared to the volume flowing through the main inlets.

Several experiments were conducted using the Delft3D model of Great South Bay to evaluate the effect of different processes on water levels in Great South Bay during Hurricane Sandy. The tests were designed to assess the influence of short waves and overwash contributions on bay water levels.

The sensitivity testing showed that peak water levels in Great South Bay were increased by approximately 20–50 centimeters (7.9 to 19.7 inches) by adding wind-generated waves, and this improved the match between modeled and observed water levels. Addition of overwash volumes added approximately 20 centimeters (7.9 inches) to the peak water levels, and this resulted in an over-prediction in water levels west (Lindenhurst) and east (Narrow Bay) of the breach.

Additional sensitivity testing has been conducted using Hurricane Sandy forcing to assess the effects of a significantly wider breach configuration on water levels in Great South Bay. These preliminary results are expected to be released for publication in 2016.

To evaluate effects of the wilderness breach on water levels in the bays, the Great South Bay model was run both with and without the breach over the 2-year period immediately following Hurricane Sandy to October 2014. The model for the with breach simulations was run with a fixed bathymetry based on a June 2014 USGS survey of the breach. Linear regression analyses of twice-daily high water levels and small storm surge levels, as well as tidal analysis of the computed time series from the model were performed to evaluate effects of the breach. Daily peak water and surge levels in Hudson Bay at the far western end of the system were not affected by the breach. At Lindenhurst near the western end of Great South Bay, the model showed an increase of 2.6 centimeters (1.0 inch) in daily high water levels with the breach, and at Bellport (eastern Great South Bay) there was an increase of 0.3 centimeters (0.11 inches) (Table 4 and Figure 1). Surge levels with the breach increased by 4.2% at Lindenhust and 1.3% at Bellport. A comparison between computed water levels with and without the breach at each station for a 9-day period in December 2012 illustrates the effects of the breach on daily water levels and small surge events (Figure 10).

Table 4. Impact on tidal amplitude and phase with and without the breach.

	Observed		Modeled without Breach		Modeled with Breach	
Station	Amplitude (m / ft)	Phase (°)	Amplitude (m / ft)	Phase (°)	Amplitude (m / ft)	Phase (°)
Hudson Bay	0.566 / 1.9	9.1	0.620 / 2.0	10.9	0.618 / 2.0	11.0
Lindenhurst	0.172 / 0.6	71.8	0.169 / 0.6	83.4	0.195 / 0.6	85.2
Bellport	N/A	N/A	0.166 / 0.5	108.0	0.163 / 0.5	91.0

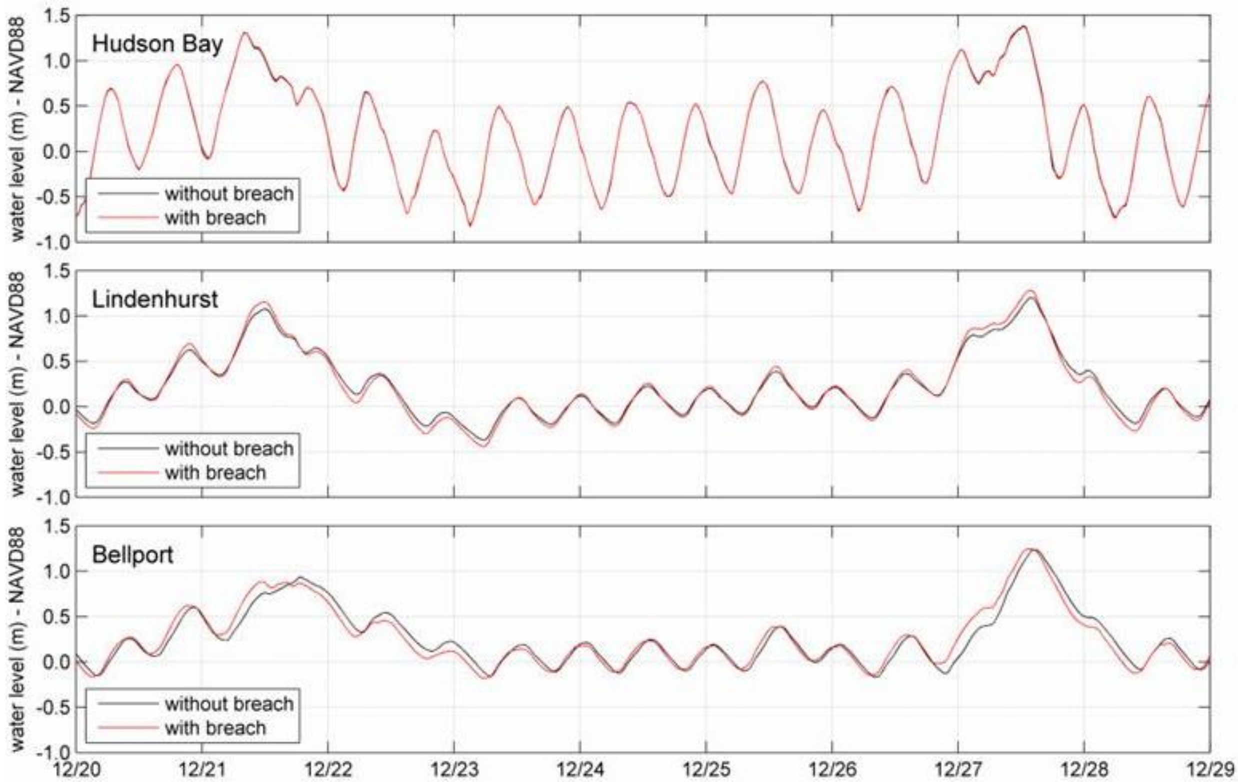

Figure 10. Comparison of computed (modeled) water levels with and without the breach for Hudson Bay, Lindenhurst, and Bellport for the period of December 20–29, 2012 (van Ormondt et al. 2015).

Tidal constituent analyses on the model results showed that tides were unaffected by the breach at the far western end of the system in Hudson Bay. At Lindenhurst in western Great South Bay, the amplitude of the M2 component increased by 15% and there was a small phase shift of +2°. The tidal analysis at Bellport showed a 2% increase in M2 amplitude and a phase shift of -17°. The modeling indicated that high and low waters in Bellport occur approximately 35 minutes sooner as a result of the breach (Table 4).

Results of the 2-year model simulation showed that daily high tides and small storm surge levels in Great South Bay have been minimally impacted by the breach. While both changes are of similar order of magnitude, the changes are small relative to total water level variations in the bay. In the period between Hurricane Sandy and June 2014, the model suggests that the breach did not increase peak water levels by more than 10 centimeters (3.9 inches) at any time (van Ormondt et al. 2015).

US Geological Survey Data Analyses

The USGS working in conjunction with Integrated Statistics conducted a study to evaluate whether the wilderness breach influenced maximum water levels in Great South Bay following Hurricane Sandy (Aretxabaleta, Butman, and Ganju 2014). The study used offshore water level data measured at Sandy Hook, New Jersey, The Battery, New York, and bay water levels measured at Lindenhurst, New York. Data spanning the period from October 1, 2007, to December 31, 2013, were obtained to cover pre- and post-breach conditions. Analyses were performed for tidal amplitudes, spectra of water level fluctuations, and spectral coherence and transfer functions between offshore and bay water levels before and after Hurricane Sandy. Comparisons between water levels before and after Hurricane Sandy at Lindenhurst and offshore stations showed no significant differences in the transfer of sea level fluctuations from offshore to Great South Bay. High water levels in the bay were attributed to winter storms and not the breach or geomorphic changes in Great South Bay caused by Hurricane Sandy. Coherence, transfer coefficients, and regression between water levels in the bay and offshore suggested that water levels in the bay were mostly damped co-oscillations driven by offshore sea level, modified by the duration of offshore events and by the breach and bay geometry.

Stony Brook University – School of Marine and Atmospheric Sciences Model Studies

Stony Brook University's School of Marine and Atmospheric Sciences (SoMAS) runs the Great South Bay Observatory, which monitors water levels, temperature, and salinity at several stations around the bay and at the buoy anchored in the central bay. The buoy also measures meteorological conditions. The Great South Bay Project at SoMAS has been studying the hydrodynamics, biochemistry, and benthic and pelagic ecology before and after the opening of the breach. The work combines field observations with numerical modeling to develop an ecosystem based management approach for addressing ecological problems in Great South Bay. As part of this program the Finite Volume Coastal Ocean Model (FVCOM) was set up to examine the structure of tidal and wind-driven circulation in Great South Bay and to evaluate the impacts that breaches in the barrier island could have on the ecology of the bay (Yang 2014; SoMAS 2016).

Early model development concentrated on replicating tidal and salinity measurements for baseline conditions (no breach) which included the four tidal inlets from East Rockaway to Moriches Inlet. The model was forced with the following data to quantify the physical processes:

- Tidal forcing with six constituents from the Oregon State University tidal model,
- Freshwater inflow from 61 rivers, streams and outfalls along the north shores of Great South Bay and Moriches Bay,
- Groundwater discharge along the north shores of the bays, and
- Water temperature and salinity measured at 21 stations in Great South and Moriches Bays and 1 long-term station just outside Fire Island Inlet.

For the baseline condition (no breach) the model was used to identify tidal- and depth-averaged velocities and residual currents (tidal-mean transport). The mean current data showed the largest residual currents in the inlets, the channels along the north shore of the western bay, and in Smith Point Channel. Residual currents were much smaller in the open central portion of the bay. A mean inflow was identified in the three smaller inlets (East Rockaway, Jones, and Moriches) with an outflow through Fire Island Inlet. Thus, the model showed that the western and eastern ends of the bay were supplying more saline water to the central bay, in order to maintain the salinity balance against the influence of the larger rivers. The tidal-mean transport for the baseline condition, represented by transport streamlines in Figure 11, showed a number of residual eddies in the open portion of central and eastern Great South Bay. Most of the headlands were also shown to have an associated eddy, and a large counterclockwise eddy was identified south of Patchogue (~33 to 34 kilometers (21 miles) on Figure 11).

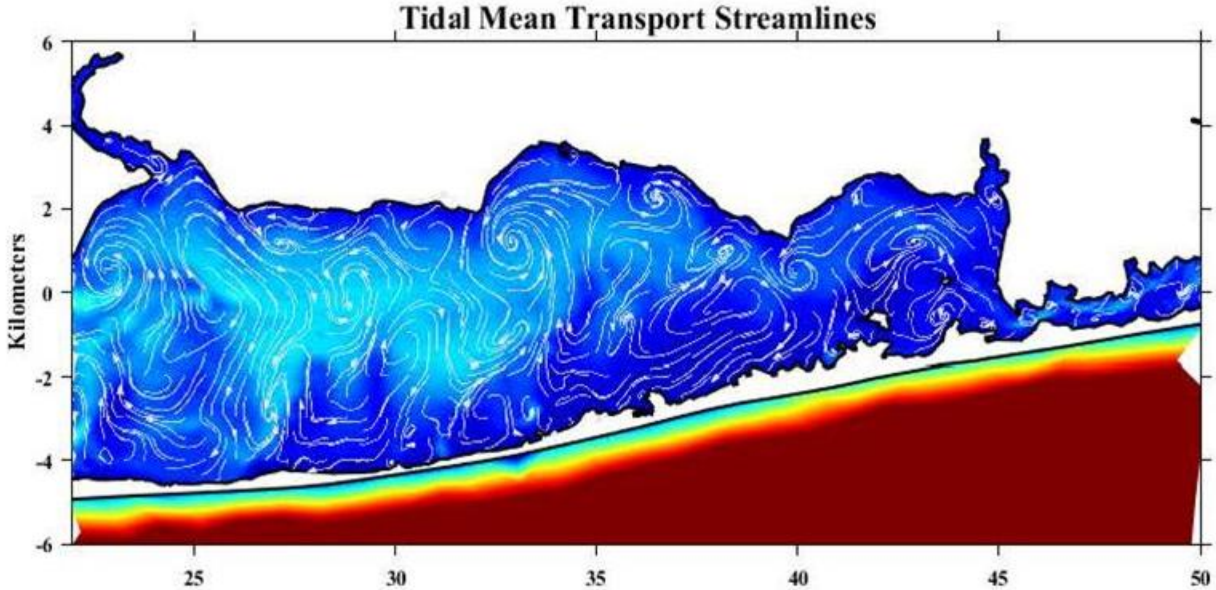

Figure 11. Tidal mean transport in eastern Great South Bay for the baseline condition (no breach) as simulated by the School of Marine and Atmospheric Sciences Model Studies Finite Volume Coastal Ocean Model for Great South Bay (SoMAS 2016).

Additional FVCOM simulations were conducted on the baseline conditions by adding constant along-bay wind forcing from the west and east. Under westward winds with baseline conditions, a large clockwise eddy was created in eastern Great South Bay, with a smaller connected eddy in Bellport Bay. Eastward winds under the no breach baseline condition showed a series of clockwise eddies along the northern side of eastern Great South Bay, and larger counterclockwise eddies on the south side near the barrier beach. Flow was strongly to the east immediately adjacent to the barrier and into Smith Point Channel. The addition of winds to the FVCOM simulations showed the strong influence that wind forces can have on the circulation patterns. In general, westward winds produce a clockwise eddy while the eastward winds produce a stronger east flowing current in the lower portion of eastern Great South Bay.

Impacts of the breach on tide range and residual currents were evaluated with the FVCOM model via tide and wind forced simulations using bathymetry and flood-tidal delta morphology surveyed in July 2015. These simulations showed the breach to cause a maximum increase in tidal amplitude in the bay of approximately 2.5 centimeters (1.0 inch), very near what has been observed, while also advancing the tidal phase in the eastern bay by about 20 minutes, again very similar to observations. Changes in the general circulation patterns in central and eastern Great South Bay were also seen, with fewer small eddies, and a mean through-flow directed to the west out through Fire Island Inlet (Figure 12). Model simulations with eastward and southward winds showed water levels in the bay to be lower with the breach than without, and under westward and northward winds, the breach simulations produced higher water levels. The eastern portion of the bay (Bellport, Blue Point, Barrett Beach) was shown to be slightly more responsive than the western bay, but the maximum difference in water level for the high wind simulations did not exceed a few centimeters.

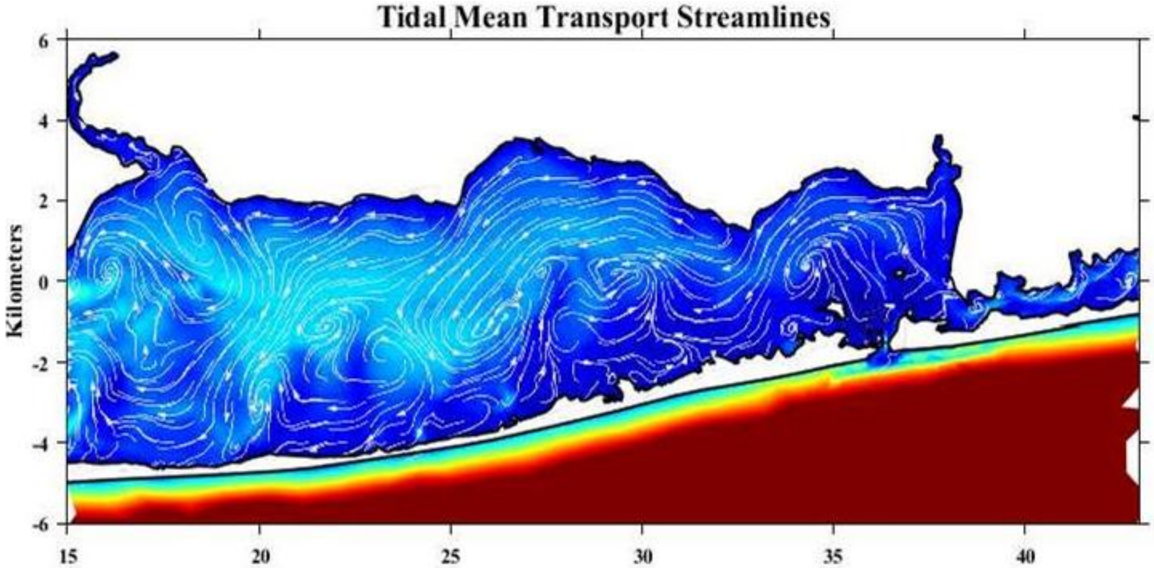

Figure 12. Tidal mean transport in eastern Great South Bay with the breach open as simulated by the School of Marine and Atmospheric Sciences Model Studies Finite Volume Coastal Ocean Model for Great South Bay (SoMAS 2016).

Stony Brook University – School of Marine and Atmospheric Sciences Data Analyses

SoMAS analyzed tide gage records from Bellport and Lindenhurst to evaluate changes in tidal amplitude and phase since before the breach was opened (Flagg et al. 2016). A tidal constituent analysis was performed on data from October 2012 through May 2016 to estimate the primary tidal parameters. Changes in the M2 tidal amplitude and phase for the two stations were compared (Figure 13). The data show no significant changes in tidal amplitude at Bellport since before Hurricane Sandy, whereas high tide has advanced by about 15 minutes relative to conditions prior to Hurricane Sandy. At the western end of the bay at Lindenhurst, the amplitude of the M2 tidal constituent has increased by about 2.0 centimeters (0.8 inches) which translates to an increase in tidal range of about 4.0 centimeters (1.6 inches). The data do not show a change in tidal phase for this western part of the bay.

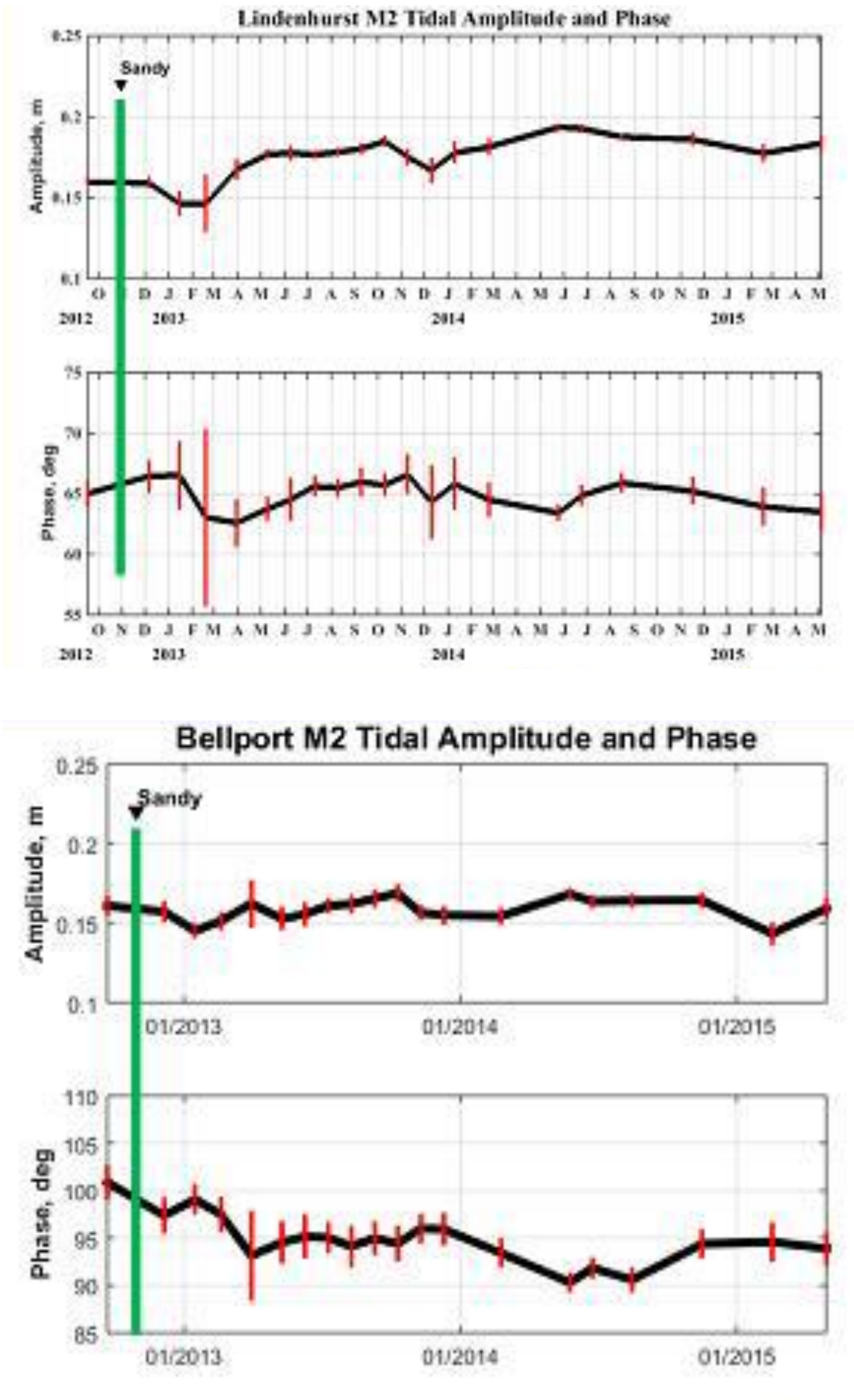

Figure 13. Temporal variation in the M2 tidal amplitude (in meters) and phase (in degrees) since Hurricane Sandy at Bellport and Lindenhurst (Flagg et al. 2016).

Summary of Breach Impacts on Water Levels and Circulation

Effects of the breach on high water levels in Great South Bay from daily tidal fluctuations and small surge events have been evaluated using hydrodynamic modeling and analyses of tide gage data. Both modeled and measured data show a small increase in high tide water levels in the western and central parts of Great South Bay and minimal changes in the eastern parts of the bay.

The greatest changes in tidal range are seen near Lindenhurst in western Great South Bay, where modeling and tide gage data indicate high tide water levels have increased between 2.0 and 2.5 centimeters (0.8 and 1.0 inches). Elsewhere in central and eastern Great South Bay, increases in the high tide water level as a result of the breach, as shown by modeled and measured data, have been less than 0.8 centimeters (0.3 inches). Daily water levels at the far western end of Great South Bay and Hempstead Bay have not been affected by the breach. Overall, the changes in daily high water levels are small relative to the total water level variations.

Stage frequency curves generated from numerical modeling with and without the breach show an increase in the 100-year return period water level (1% annual chance water level) of 60 centimeters (23 inches) for the Connetquot River area in central Great South Bay for the with breach conditions. Elsewhere in Great South Bay and Moriches Bay, the stage frequency curves show increases of 20 to 40 centimeters (16 inches) for the 1% annual chance water levels. The stage frequency curves, originally based on model simulations with a breach nearly 2.5 times larger than the current opening, were adjusted using model runs on a breach configuration measured in June 2014. The predicted 1% annual chance water levels are based on statistical analyses of numerical model simulations that may over predict peak storm water levels by as much as 25 centimeters (9.8 inches). Although absolute values of water level increases during storms are difficult to predict with a dynamic coastal system, the order of magnitude spatial distribution of the modeled increases is considered reliable.

Numerical model studies and analyses of measured water level data show that the breach has resulted in a phase shift in the tide and surge in the easternmost part of Great South Bay, causing high and low water in Bellport to arrive 20 to 35 minutes sooner as a result of the breach.

Hydrodynamic modeling simulations of wider breach scenarios in the wilderness area of 1,128 and 1,433 meters (3,701 and 4,701 feet) and 2.1 meters (6.9 feet) depth (6- and 12-month conditions) show a maximum water level increase of 80 centimeters (31 inches) for the 100-year return period event. The breach geometry in these two scenarios is considerably larger than the modeled 3-month condition and the June 2014 breach, and as such, the model results may be more representative of future storm water levels under climate change and sea level rise conditions.

Hydrodynamic modeling indicates that the breach has altered the general circulation patterns in central and eastern Great South Bay (Figures 11 and 12). Prior to the breach the circulation was characterized by a number of smaller localized eddies. Since the breach the circulation has become a mean through-flow directed from Bellport Bay to the west out through Fire Island inlet. This change in residual circulation suggests a reduction in residence time in eastern Great South Bay which is an important factor that affects bay water quality.

Physical Water Quality

Physical water quality in Great South Bay is influenced by mixing between fresh and marine waters through the tidal inlets. The wilderness breach in Fire Island has the potential to change bay water quality by increasing tidal and subtidal flushing. Increased tidal and subtidal flushing can lead to water quality improvements by reducing residence times for water parcels in the bay, which has a history of impaired ecological function.

Key estuarine water quality parameters such as temperature and salinity are partially controlled by the extent of tidal and subtidal flushing, and these parameters are important factors that influence the bay ecology. Water temperature is a driving factor in the physiochemical and biological processes that determine how well the estuary can support aquatic life. Variations in salinity according to location in the bay, tidal fluctuations, and volumes of freshwater input influence the distributions of estuarine species.

For the purposes of this technical report, the discussion of water quality within the Great South Bay and Moriches System will focus primarily on the physical characteristics of residence time, temperature, and salinity that impact water quality. Additional water quality parameters relating to nutrients, chlorophyll a, bacteria, and certain suspended algal communities are discussed in the marine and estuarine resources technical report.

Synthesis of Water Quality Pre- versus Post-Breach

Water quality monitoring in Great South Bay has been conducted by Suffolk County and Stony Brook University's SoMAS. Combined, these datasets cover an extensive period before Hurricane Sandy, and therefore offer excellent sources of information to evaluate impacts of the breach on water quality. Suffolk County monitoring of Great South Bay began in 1976 and includes regular sampling throughout the bay for various physical parameters including salinity and temperature. SoMAS has maintained a network of observation stations in the bay since 2005, measuring a full suite of physical parameters for tracking water quality and meteorological conditions (Figure 14). More recent studies since formation of the breach looking at the plankton community in Great South Bay have used water quality parameters from the SoMAS observation stations as well as cruise data to evaluate effects of the breach on temperature and salinity (Gobler, Collier, and Lonsdale 2014). Residence times in Great South Bay have been calculated using salinity and freshwater input data, and more recently SoMAS modeled residence times using the FVCOM model. A brief summary of findings and conclusions regarding impacts of the breach on water temperature, salinity, and residence time is provided below.

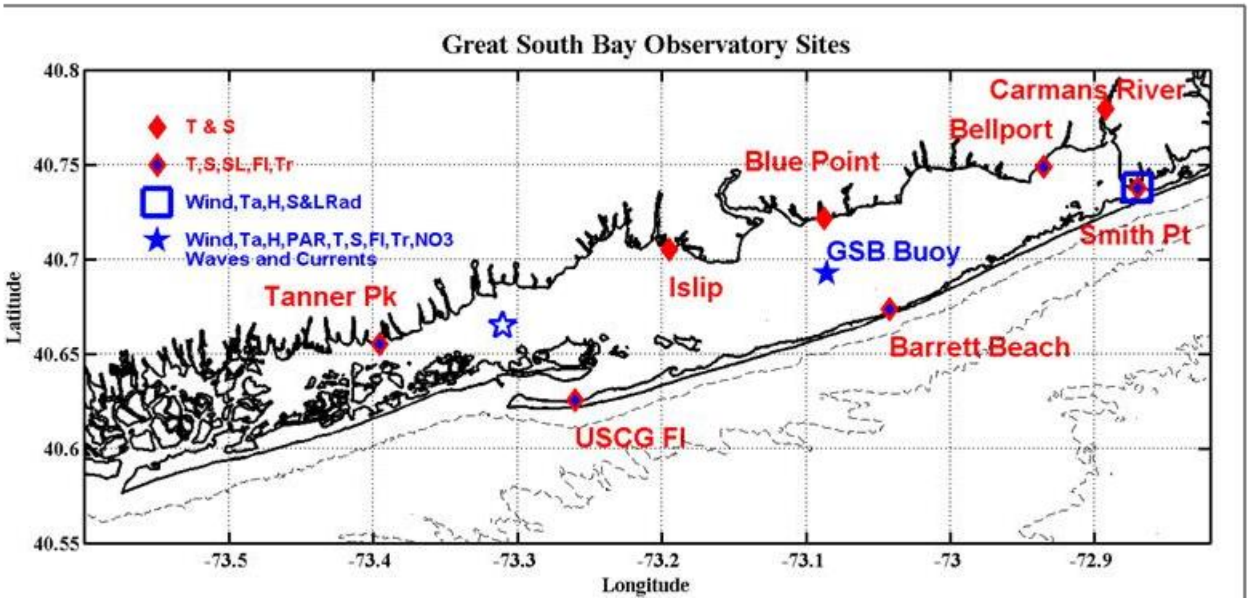

Figure 14. Great South Bay observatory locations maintained by School of Marine and Atmospheric Sciences Model Studies (SoMAS 2016).

Residence times for Great South Bay before the wilderness breach were calculated for the NPS by Hinga (2005). The volume of freshwater in the estuary, taken as the difference between ocean and bay salinities, was divided by the rate of freshwater input to determine an average residence time of 50 days. Using somewhat different assumptions for the dimensions of Great South Bay, an average residence time of 96 days was calculated by Conley (2000) prior to the breach. Predictions on potential changes in residence time with a breach at Old Inlet showed a reduction to 40 days (Conley 2000). This estimate suggests that flushing characteristics in Great South Bay would be enhanced by the breach. However, it was noted that flushing would not be uniform across the bay, with potential residence times considerably greater in the northern portions of the bay near the mainland and lower in the southern reaches. More recent calculations of residence time conducted by SoMAS for the Bellport Bay area near the wilderness breach showed a decrease from 25 to 10 days (Gurdon et al. 2015; Flagg pers. comm. 2016).

Water temperatures in Great South Bay vary seasonally. Synthesis of Suffolk County data before the breach showed summer surface water temperatures of 25 to 26°C, with occasional measurements up to 29°C. Wintertime data were not collected as regularly, but temperatures of 0 to 2°C were common (Hinga 2005). These values are consistent with pre-breach water temperatures measured at the SoMAS observation stations (SoMAS 2016). Comparison with data collected after the breach shows that summer temperatures are somewhat cooler, while winter temperatures do not seem to be impacted by the breach. More specifically, Gobler, Collier, and Lonsdale (2014) found a decrease in summer temperatures in Bellport Bay, Narrow Bay, and Moriches Bay by as much as 3°C. Despite findings that the breach has resulted in a small decrease in summertime water temperatures, in general, water temperatures in Great South Bay are mostly dependent on air-sea interactions rather than bay-ocean exchange. Changes in the heat budget of the bay due to additional water exchange through the breach are planned by SoMAS for future hydrodynamic model experiments.

Salinities in the bay are greatly influenced by the influx of groundwater, rainfall, wind stress, and location. Areas closest to the inlets have the highest salinities and areas along the northern shoreline closest to streams and areas of groundwater influx have the lowest salinities. In general, salinities are the lowest in the northeast and north central areas of the bay, and increase toward the western end of the bay and Fire Island Inlet. Before formation of the wilderness breach, average salinities typically ranged from 25 to 30 practical salinity units (psu) (Hinga 2005), except near Bellport where values were lower: between 20 and 25 psu. Since formation of the breach, average salinities in the eastern half of the bay have increased. Data collected by SoMAS at Bellport and the Great South Bay #1 buoy show a sharp increase in salinity at both stations following the breach (Figure 15; Flagg pers. comm. 2016). Bellport Bay has seen the greatest increase in average salinity since the breach (+5 psu). Barrett Beach on Fire Island to the west of the breach has seen an increase of +2 psu and measurements from the US Coast Guard Station at Fire Island Inlet show a negligible change in salinity since the breach (Figure 16 a-c; Flagg pers. comm. 2016).

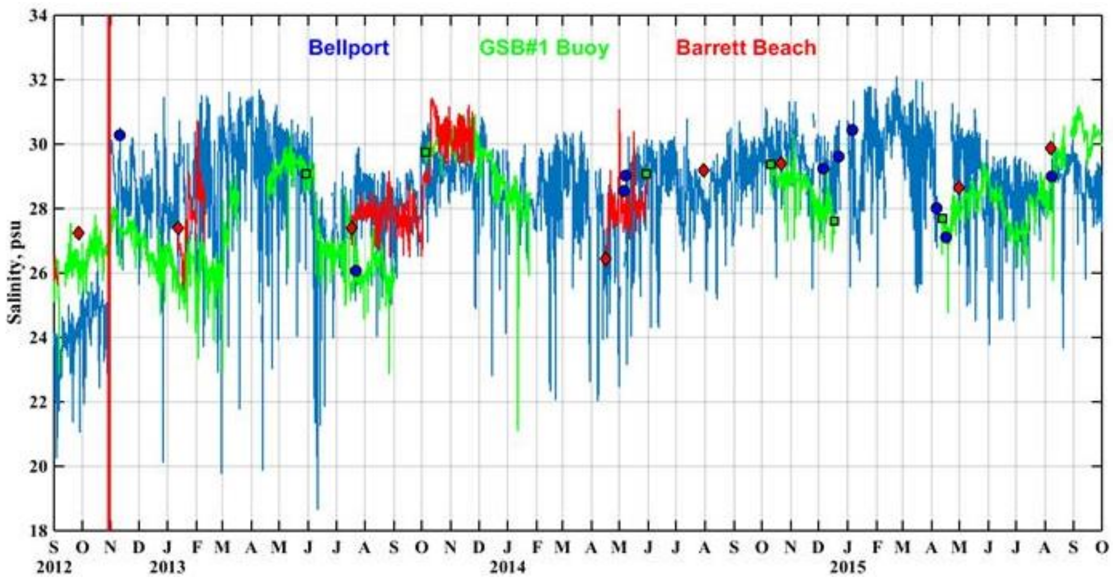

Figure 15. Salinity data in Great South Bay before and after the wilderness breach (Flagg pers. comm. 2016 (figure provided at workshop)).

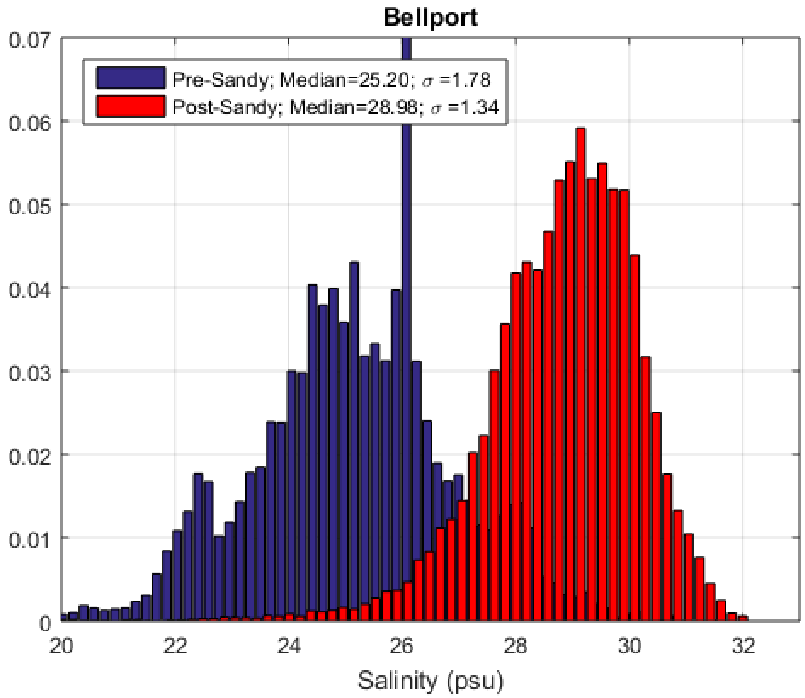

Figure 16(a). Changes in salinity in Bellport Great South Bay station following formation of the wilderness breach (Flagg pers. comm. 2016 (provided at workshop)). Figure shows the frequency distribution of salinity pre-Sandy and post-Sandy.

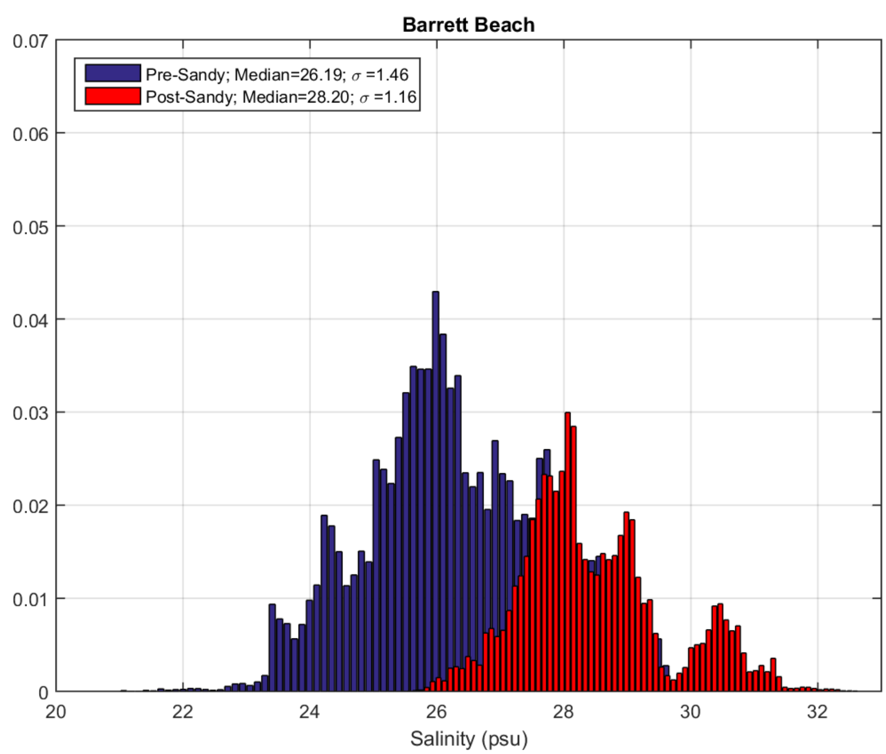

Figure 16(b). Changes in salinity in Barrett Beach Great South Bay station following formation of the wilderness breach (Flagg pers. comm. 2016 (provided at workshop)). Figure shows the frequency distribution of salinity pre-Sandy and post-Sandy.

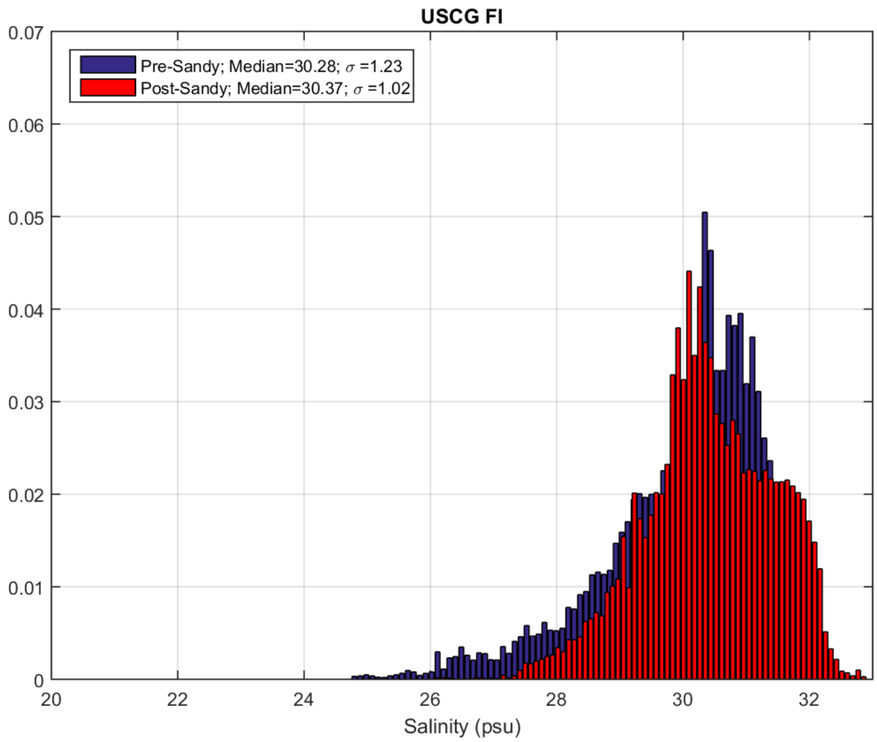

Figure 16(c). Changes in salinity in USCG Fire Island Inlet Great South Bay station following formation of the wilderness breach (Flagg pers. comm. 2016 (provided at workshop)). Figure shows the frequency distribution of salinity pre-Sandy and post-Sandy.

Summary of Breach Impacts on Residence Time, Temperature, and Salinity

Effects of the breach on key water quality parameters have been evaluated through the use of numerical modeling and analyses of long-term water quality data measured at various locations in the bay. In general, water quality in the bay has improved since formation of the breach.

Residence time for Great South Bay prior to the breach was calculated to be 50 to 100 days. Predictions on potential changes in residence time with a breach at Old Inlet showed a reduction to 40 days. Long-term observations suggest a reduction in residence time calculated based on a freshwater fraction method from 25 to 10 days locally for the Bellport Bay area.

Data analyses on water temperature changes since the breach indicate a reduction by as much as 3°C during the summer months in Bellport Bay, Narrow Bay, and Moriches Bay. However, water temperatures in Great South Bay are mostly dependent on air-sea interactions and to a lesser extent on water exchange through the breach. Hydrodynamic modeling is planned to further evaluate the influence of additional water exchange through the breach on the heat budget of the bay. Salinities in Great South Bay have increased by 2 psu in the eastern central portion of the bay, and by as much at 5 psu in the Bellport area.

Waves

Tidal inlets through barrier beaches can alter wave action along the outer coastline and can also allow the transmission of increased wave energy through the inlet into more sheltered back bay areas. The propagation and breaking of incident waves along the outer coastline is affected by the morphology of the ebb-tidal delta and the strength and structure of the ebbing currents. For example, wave refraction over and around the ebb-tidal delta complex can generate currents that flow back toward the inlet along the adjacent shorelines, particularly on the downdrift side of the inlet. This process can result in erosion, especially at inlets with large ebb-tidal delta complexes. Ocean waves entering an inlet against the ebb current tend to steepen as their wave heights increase and wavelengths decrease. This wave current interaction can influence channel shoaling and pose a threat to navigation. In cases where tidal inlets are wide and deep enough to allow the propagation of wave energy through to the estuarine environment, the increased energy can cause shoreline erosion and changes in sedimentation. Potential impacts of the wilderness breach on outer coast and bay area wave conditions are described in the following section.

Synthesis of Wave Information Pre- versus Post-Breach

The wave climatology offshore of Long Island is characterized by moderate Atlantic waves typically from the southeast quadrant. There is a relatively strong seasonal component of mild waves during summer, severe waves associated with extratropical storms frequent during winter and spring, and severe waves associated with tropical storms during fall. Mean significant wave height over a 6-year period at NOAA National Data Buoy Center Buoy #44017 is approximately 1.5 meters (4.9 feet) with a mean wave period of 5 seconds (NOAA 2016). The majority of waves are from the southeast and the more severe storms associated with extratropical storms are from the east-southeast. This results in a net westerly longshore transport direction (Leatherman 1985; Smith et al. 1999).

The inner shelf at the eastern end of the Fire Island (east of Watch Hill) is characterized by relatively straight and parallel contours (Figure 17; Schwab, Denny, Baldwin 2014). This means that the bathymetric contours are parallel with the shoreline, forming a uniformly sloping shelf. The consistent nature of the bathymetry causes the incident waves to act uniformly across the inner shelf. As waves approach the nearshore area and enter the surf zone, they interact with the nearshore bar and this causes variations in wave energy along the shoreline (Nelson et al. in review). Since formation of the wilderness breach, the ebb-tidal delta has caused a perturbation in the nearshore bar system.

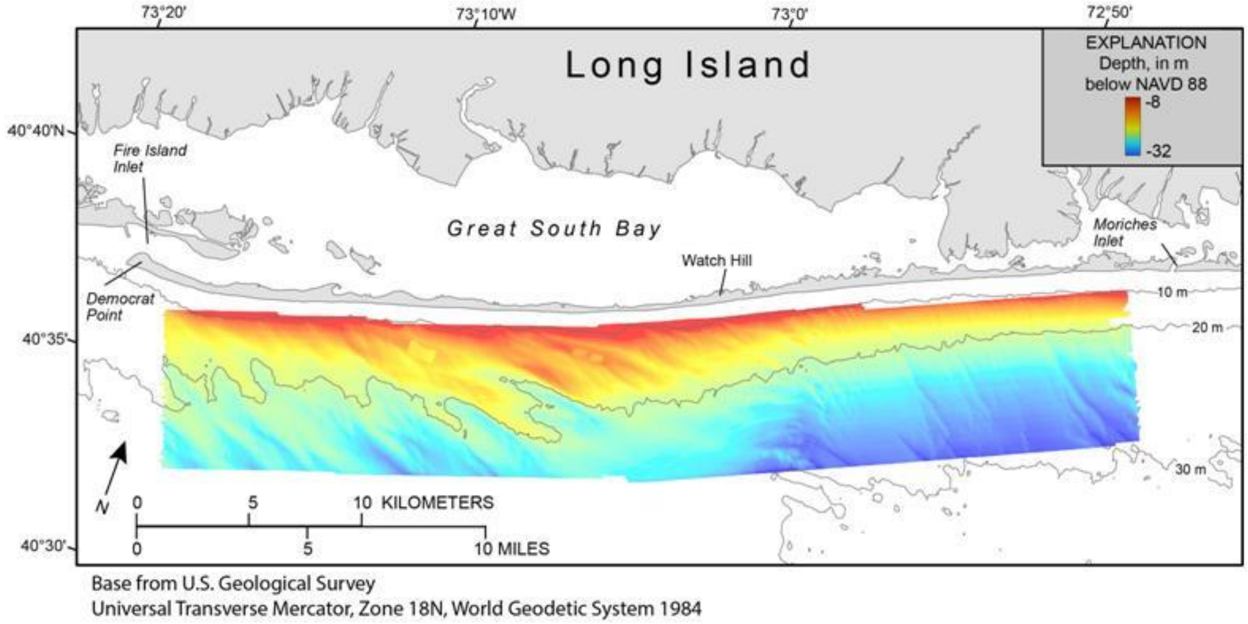

Figure 17. US Geological Survey bathymetric map shore-connected ridges along the western end of Fire Island and a more planar shelf with straight and parallel contours on the eastern end.

Studies specifically looking at impacts of the breach on the nearshore wave climatology have not been conducted. The USACE and Deltares model studies have simulated wave conditions in the breach; however, the results have not been extracted at the fine resolution needed to evaluate localized wave changes and resulting impacts on sediment transport. Data collected to date suggests that shoreline erosion resulting from wave interaction with the ebb-tidal delta is small and localized to the downdrift or western side of the inlet within the wilderness area. Supporting evidence includes a relatively small ebb-tidal delta complex and evidence of localized downdrift increased shoreline erosion since formation of the breach.

The USGS collected bathymetric surveys of the breach and associated flood- and ebb-tidal deltas in June 2013, June 2014, October 2014, and May 2015 (Figure 18; Brownell et al. 2015; Nelson et al. 2016a, 2016b; and Nelson et al. in review). These surveys show a significant increase in the size of the ebb-tidal delta between June 2013 and June 2014, followed by relatively little growth through October 2014. This suggests that ebb-tidal delta growth has slowed. Current photographs show the delta to extend approximately 0.8 kilometer (0.5 mile) on either side of the breach centerline. Shoreline change data analyzed by the USGS for the period October 2012 to September 2015 show effects of the breach to be localized to an area approximately 0.5 kilometer (0.3 mile) downdrift of the opening (Figure 19; USGS 2016a). Since September 2013 the mean high water shoreline immediately downdrift of the breach has eroded approximately 60 meters (197 feet). This is in sharp contrast to shorelines further to the west that have eroded between 20 and 30 meters (66 and 98 feet). The relatively small size of the ebb-tidal delta and lack of recent growth, in combination with localized downdrift erosion were cited as evidence that the breach has a limited impact on the adjacent open coast shorelines. It was not determined whether the localized downdrift erosion was entirely due to wave interaction with the ebb-tidal delta; however, it was concluded that changes in the nearshore waves as a result of the breach are not likely to result in erosion much beyond the opening. Therefore, developed areas on Fire Island west of the breach will not be affected by changes in wave climatology due to the breach.

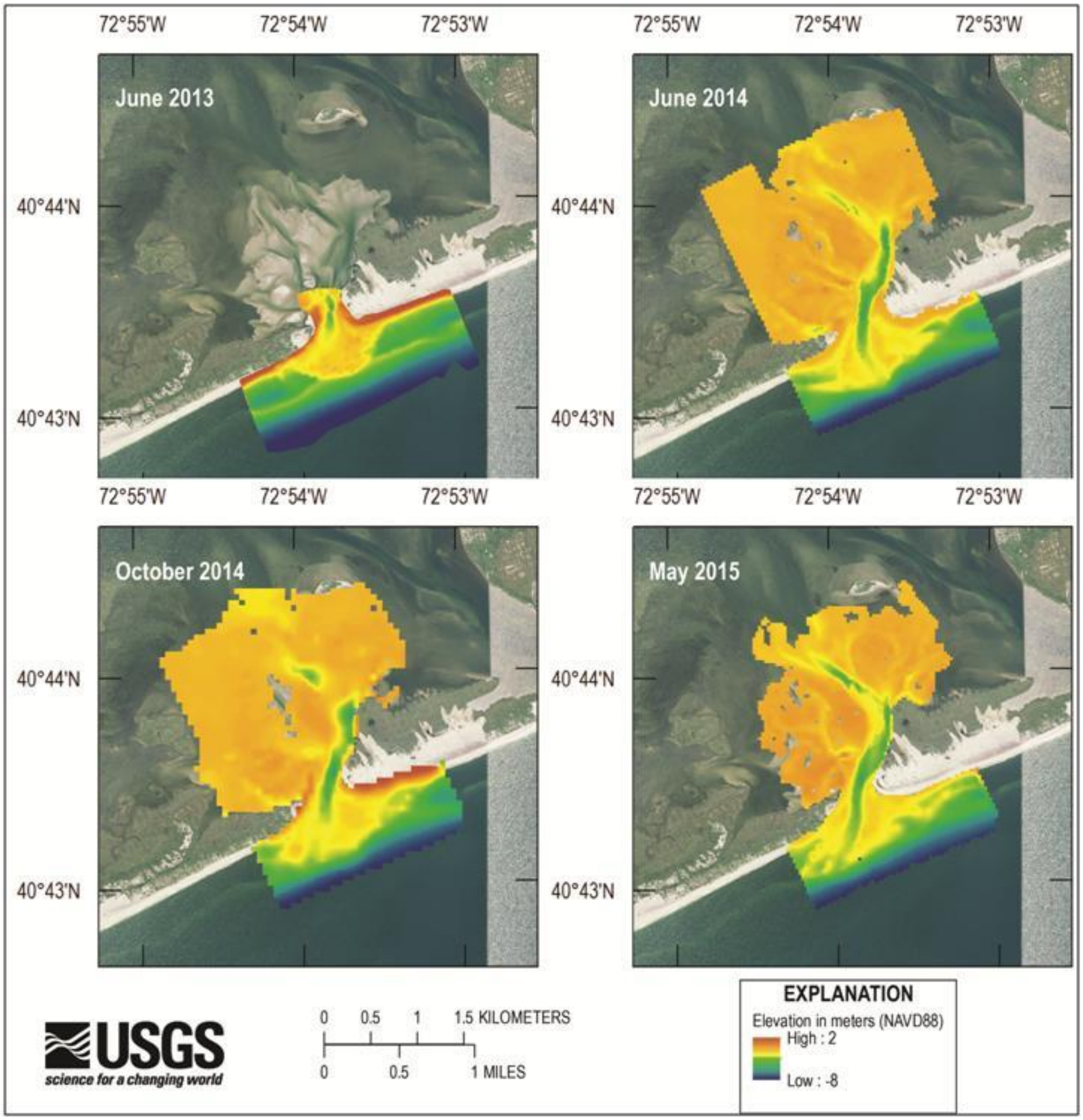

Figure 18. US Geological Survey bathymetric surveys of the breach and associated shoals from June 2013, June 2014, and October 2014 (Brownell et al. 2015; Nelson et al. 2016a and 2016b).

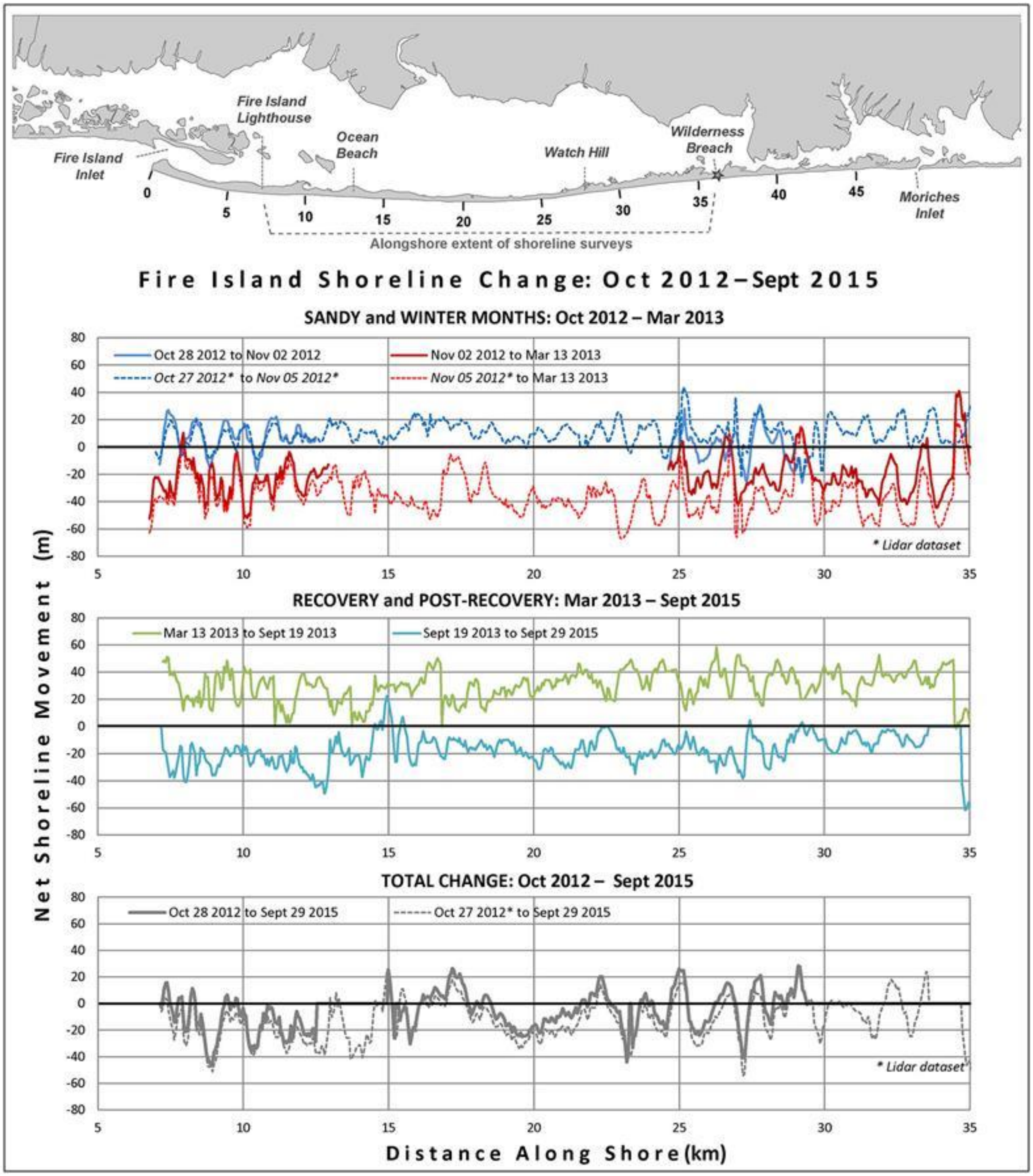

Figure 19. US Geological Survey shoreline change data in the Fire Island National Seashore for the period October 2012 to September 2015 (USGS 2016a).

The width of the breach and shallow nature of the flood-tidal delta are primary factors that limit wave propagation from the open ocean through the breach to Great South Bay. Specific studies have not been conducted to evaluate this process, but general consensus at the January workshop was that wave activity in the bay has not increased as a result of the breach. The current width of the breach at approximately 500 meters (1,640 feet) provides a narrow window of exposure for Great South Bay (Figure 16; SoMAS 2016), and any waves that enter the breach will be broken by the shallower waters over the flood-tidal delta (Figure 20). The potential for increased wave activity in Great South Bay given a wider breach was also considered to be low, since the extent and shallow nature of the flood shoals will provide protection. Based on these discussions, it was concluded that the potential for increased shoreline erosion in Great South Bay as a result of wave propagation through the beach was extremely low. Wind generated waves within the bay have not been impacted because increases in water levels due to the breach are very small.

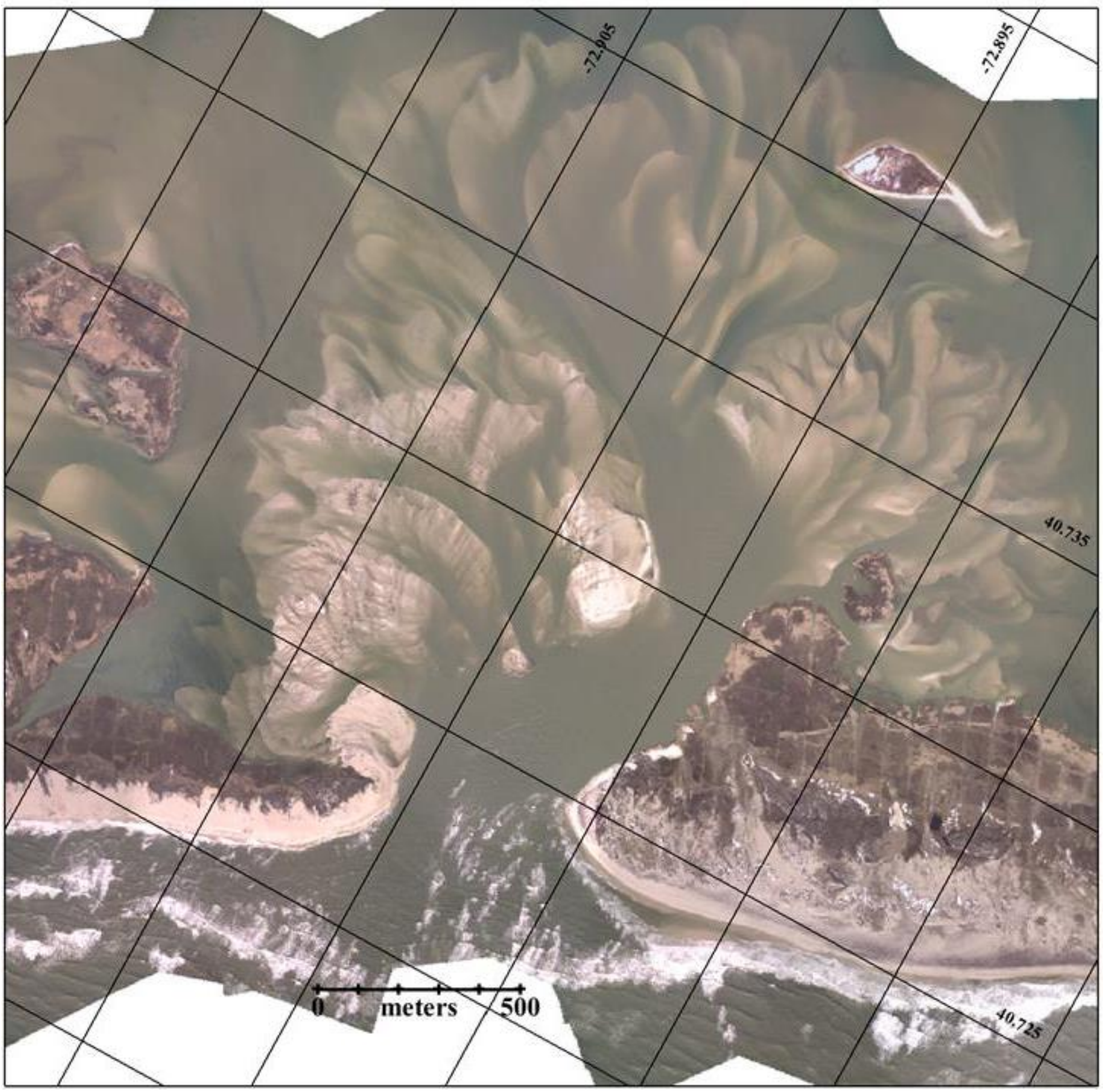

Figure 20. January 31, 2016, aerial photo showing the configuration of the breach and the approximate width of 500 meters (1,640 feet) (SoMAS 2016).

Summary of Breach Impacts on Waves

Impacts to waves along the south shore of Fire Island as a result of the breach are localized to an area within 0.5 to 1.0 kilometer (0.3 to 0.6 mile) of the breach. Wave interaction with the ebb-tidal delta may be partly responsible for increased shoreline erosion within an area 0.5-kilometer (0.3-mile) downdrift of the breach.

Wave propagation from the ocean through the breach and into Great South Bay is limited by the width of the breach and the shallow nature of the flood-tidal delta complex. Consequently, the potential for increased shoreline erosion in the bay as a result of wave propagation through the breach is extremely low.

Sediment Transport and Breach Geomorphology

Sediment transport processes along open coast shorelines are often interrupted by tidal inlets. Inlets provide conduits for the transport of littoral drift into flood- and ebb-tidal delta complexes, back barrier bays, and salt marsh systems. Inlets that remove material from the longshore transport system are sediment sinks. In cases where a significant percentage of the annual littoral drift is lost through landward transport into the inlet, erosion of the downdrift shorelines can be an issue. Inlets that are not sediment sinks typically allow sediment bypassing through one or more mechanisms that help to feed the downdrift shorelines with sediment. The geomorphology of inlet systems is therefore influenced by sediment transport processes that are driven by a combination of waves and tidal currents. Potential impacts of the wilderness beach on sediment transport processes are described in the following section. Changes in breach geomorphology are also discussed.

Synthesis of Sediment Transport and Geomorphology Information Pre- versus Post-Breach

The dominant direction of longshore transport from east to west along Fire Island has been documented based on spit growth and inlet dredging records (Leatherman 1985; Kana 1995; Smith et al. 1999). Rates of longshore sediment transport have been estimated based on calculations at Moriches and Fire Island Inlets. Most studies have found higher rates of transport leaving from Fire Island Inlet than entering at Moriches Inlet, suggesting that additional sand is added to the overall budget somewhere along Fire Island (Panuzio 1969; Kana 1995; Rosati et al. 1999). Hapke et al. (2010) reviewed published sediment budgets for the area and found an average increase of 217,000 cubic meters (283,800 cubic yards) per year in the transport rates between Moriches and Fire Island Inlets. Updrift shoreline erosion and redistribution of beach nourishment material have been cited as potential sources for the additional sediment. Shoreface-connected sand ridges have also been proposed as a possible source of sediment required to balance the sediment budget (Schwab et al. 2000). These are linear shoals that tend to be oriented parallel to the direction of the dominant storm wave approach. More recent work suggests the source of sediment is from erosion of the inner shelf with the sand ridges and beach both composed of the reworked shelf material (Schwab et al. 2013; Schwab et al. 2014). Warner et al. 2014 use modeling results to suggest the net onshore flux is from the troughs between the sand ridges. Hapke et al (2010) used previously published sediment budgets to develop a single conceptual model to balance existing sediment budget estimates for Fire Island by considering these onshore sediment transport contributions. The conceptual model shows that onshore sediment transport via the shoreface-connected ridges is an important process along the western end of Fire Island, some 16 kilometers (9.9 miles) west of the wilderness breach (Figure 21). Longshore transport rates in the vicinity of the wilderness breach are likely similar to those estimated as influx at Moriches Inlet.

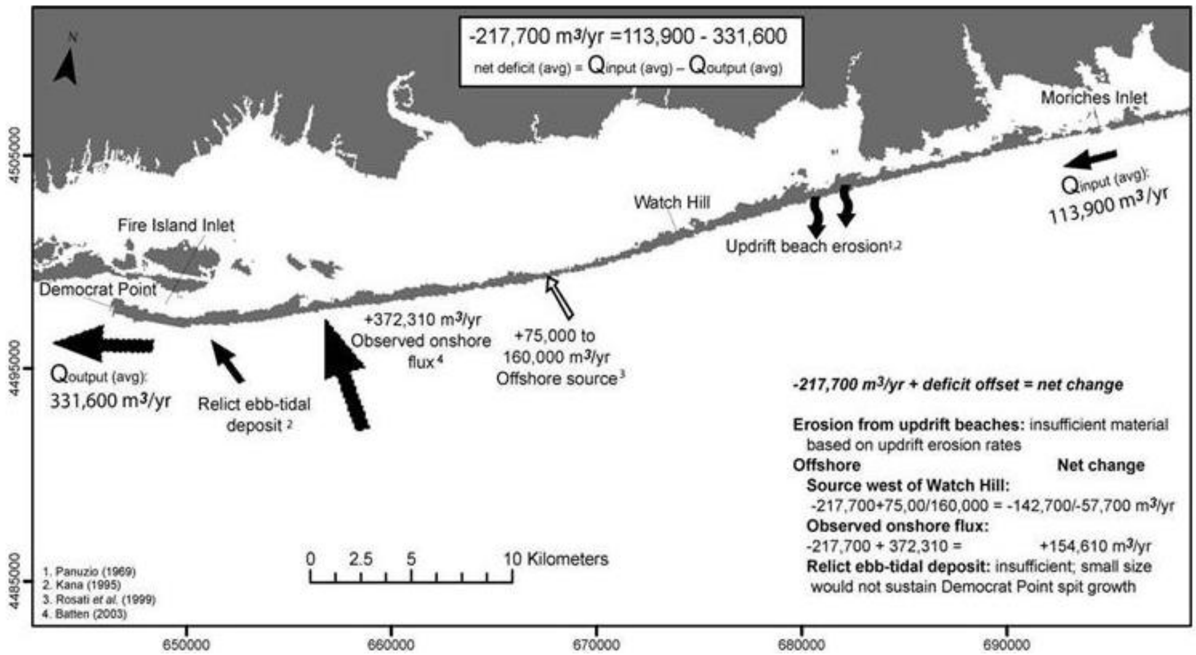

Figure 21. Conceptual model to balance the Fire Island sediment budget between Moriches and Fire Island Inlets (Hapke et al. 2010 and references therein). The breach formed approximately 10 km west of Moriches Inlet, near where the arrows are located to show "updrift beach erosion."

There have been no updates to the Fire Island sediment budget since formation of the wilderness breach; although existing data suggest that the inlet is not causing a significant interruption in longshore sediment transport. Supporting data include analyses of changes in mean high water shoreline position downdrift of the breach following Hurricane Sandy (Henderson et al. 2015), analyses of flood-tidal delta growth, and evaluation of ebb-tidal delta breaching processes.

Beach surveys have been conducted by the USGS to evaluate shoreline changes caused by Hurricane Sandy and to monitor the continued response and recovery of the barrier beach (USGS 2016a). Surveys have been collected along shore parallel and shore normal tracks to capture the base of the dune, the mid-beach, and the upper and lower foreshore along the length of Fire Island west of the wilderness breach. The dataset includes measurements before and after Hurricane Sandy, monthly data between December 2012 and April 2013, and a September 2015 survey. LiDAR data have also been used to supplement the USGS field measurements. Plots of net shoreline movement (shown in Figure 19) were developed to show various stages of shoreline response, where the wilderness breach is located at the far right of the x-axis (USGS 2016a). Data from the winter months following Hurricane Sandy showed significant erosion along most of the shoreline (November 2012 to March 2013; red line top plot). A 6-month recovery period was then identified during which time the shoreline accreted an average of 30 meters (98 feet) (March 2013 to September 2013; green line middle plot). Over the next two years the system transitioned to a state of increased or sustained erosion with an average change of -17 meters (-56 feet) (September 2013 to September 2015; blue line middle plot). The average net shoreline movement from immediately before Hurricane Sandy to the most recent September 2015 survey was erosional, with an average movement of -12 meters (-39 feet) (bottom plot). While the net shoreline response shows distinct zones of erosion and accretion along the length of Fire Island, there is no indication that the wilderness breach is creating a regionalized downdrift zone of erosion caused by trapping of the littoral drift. As discussed previously, the breach appears to be responsible for localized erosion immediately downdrift of the opening, but there is no indication that this erosion extends more than 1 kilometer (0.6 mile) west of the breach.

Bathymetric and topographic surveys of the wilderness beach and associated shoals have also been used by the USGS and Deltares to track changes in the morphology of the system. Analyses using these data suggest that the breach is not currently acting as a sediment sink, and therefore not causing a major interruption in littoral drift. A total of four surveys were collected in June 2013, June 2014, October 2014, and May 2015 using a combination of the USACE Lighter Amphibious Resupply Cargo and USGS personal watercraft equipped with a backpack GPS unit (Nelson et al. 2016a). Flood shoals that formed inside Great South Bay as a result of the breach showed fast initial growth in the first winter after Hurricane Sandy, importing large amounts of sediment from erosion of the adjacent barriers. During this initial period the breach acted as a sediment sink. Following the winter of 2013, growth of the flood shoals stabilized and the system is reported by Deltares to be exporting sediment (Deltares 2016). Volumetric change analyses on bathymetric data from the flood-tidal delta suggests that the increased size of the delta seen in aerial photographs may be the result of reworking of deposits and addition of sediment derived from channel deepening rather than the import of sediment from the ocean side of the system (van Ormondt pers. comm. 2016).

Review of aerial photographs provides further support that the breach is relatively efficient at bypassing sediment to the downdrift shoreline. Continuous bypassing around the outer edge of the ebb-tidal delta driven by waves and currents is a mechanism for transport around the breach. This method delivers sediment to the western shoreline approximately 1.0 kilometer (0.6 mile) downdrift of the breach where the ebb-tidal delta merges with the nearshore bathymetry. Sediment is also transported via shallow channels into the main breach channel from the east and moves out to the west through ebb-tidal delta channels, resulting in negligible net influx to the flood-tidal delta complex. The migration of large bar complexes to the downdrift side of the breach also bypasses sediment, as the main channel switches from a northeast-southwest orientation through the ebb-tidal delta to a more direct north-south orientation, usually initiated by a storm.

Changes in the geomorphology of the wilderness breach have been documented by SoMAS (2016) and the USGS (Nelson et al. 2016b; Nelson et al. in review) using surveys of the breach cross-sectional area, width, depth, and location. The geomorphologic parameters fluctuated along with the patterns of erosion and deposition since the breach opened. Generally, the breach widened and the cross-sectional area at mean sea level increased during the first 2 years after the breach formed, then decreased during the third year, 2015.

Flagg et al. (2015) and Flagg and Flood (2013) describe the changes in breach cross-sectional area and nearby features based on bathymetric surveys conducted between December 2012 and August 2015. These data show that the cross-section at mean sea level increased during the first 2 years after the breach formed, then decreased somewhat during 2015. Figure 22 shows the spatial patterns and extent of change. Figure 22a (left panel) shows variations in the cross-sectional profile (width, depth, and location) over time and Figure 22b (right panel) shows changes in the breach cross-sectional area since December 2012. The inlet depth profiles show maximum depths have ranged from 3 to 7.5 meters (10 to 25 feet) NAVD88 and the location of the breach centerline has migrated approximately 200 meters (656 feet) to the west since initial formation. The mean sea level cross-sectional area increased from about 100 square meters (1,076 square feet) in 2012 to about 300–400 square meters (3,229–4,306 square feet) during 2013, further increased to about 500–600 square meters (5,382–6,458 square feet) in 2014, and then decreased steadily during 2015. The most recent data (July 2015) from Flagg et al. (2015) indicate the cross-sectional area is about 450 square meters (4,844 square feet).

Additional cross-sectional data at mean high water are available from the June 2013, June 2014, October 2014, and May 2015 surveys collected by the USGS (Brownell et al. 2015; Nelson et al. 2016a, 2016b; and Nelson et al. in review). These survey data corroborate the trends seen in the SoMAS data (Figure 22, right panel). Combination of the two data sets creates a more robust series that could be updated in the future with regular measurements of cross-section to identify changes in breach morphology indicative of widening.

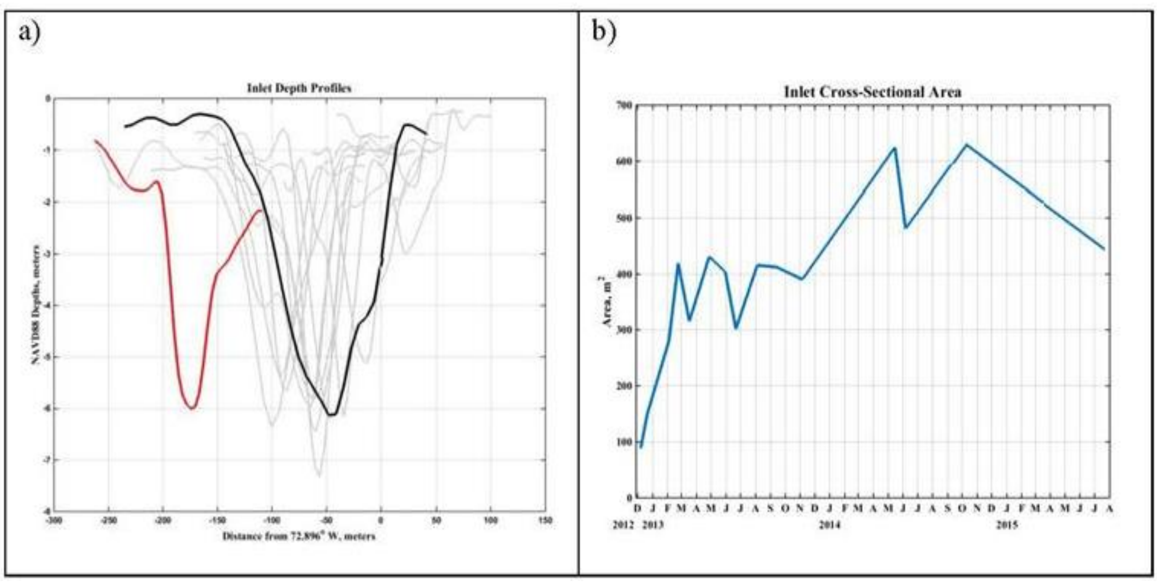

Figure 22. (a) Cross-sectional area of the breach from 15 bathymetric surveys (October 2014 survey in black; July 2015 survey in red); (b) time series of minimum cross-sectional area (Flagg et al. 2015).

Similar changes in breach geomorphology have been documented by Nelson et al. (2016b) and Nelson et al. (in review). Shoreline surveys conducted immediately following Hurricane Sandy, and again in April 2013 and October 2014 show the width of the breach at mean high water has increased significantly since initial formation (Figure 23). Results and analysis from a series of USGS and NPS surveys between November 2012 and April 2015 show the width of the breach, measured at the narrowest point between the two adjacent shorelines, increased steadily during the first year (Figure 24, top panel). This was followed by small narrowing of the breach which continued with few fluctuations to April 2015. The fourth panel in Figure 24 shows the breach has migrated west since 2012, primarily through erosion of the western shoreline and little change in the eastern shoreline.

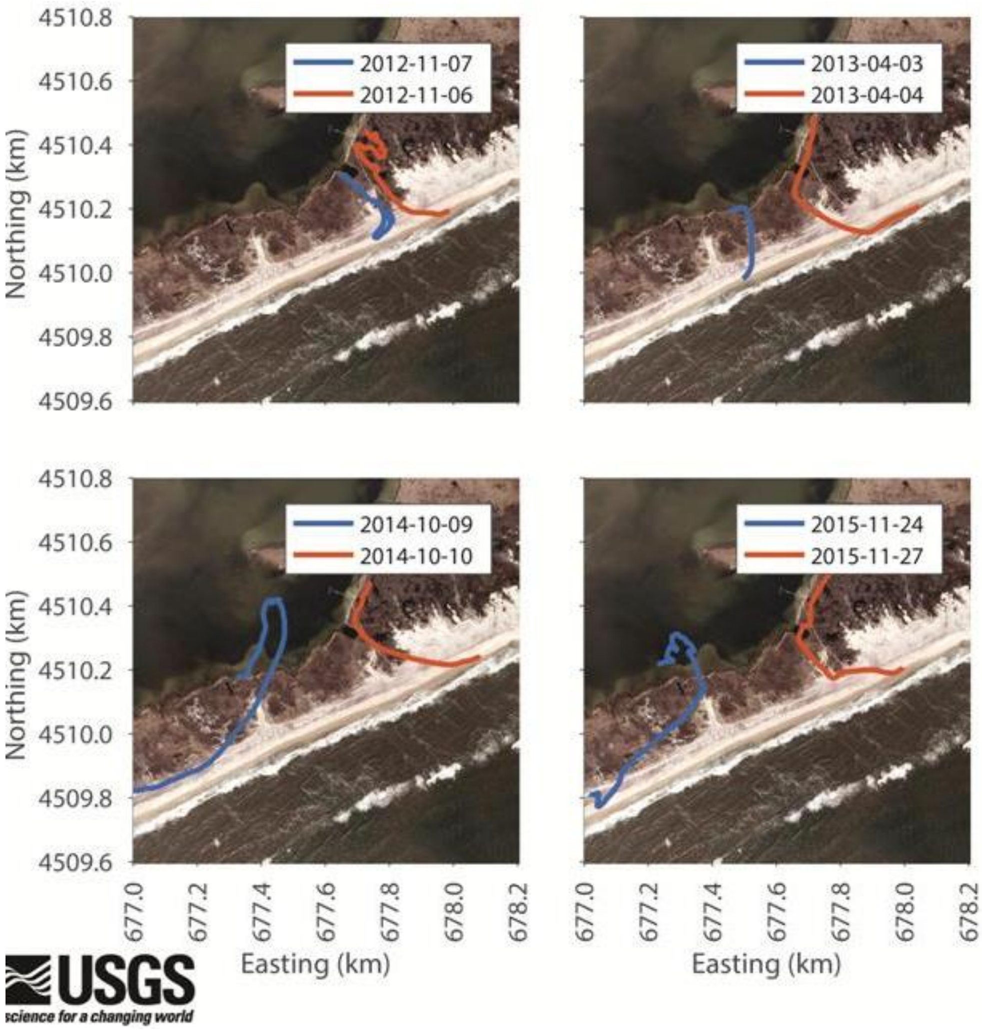

Figure 23. Breach shoreline change and width change between 2012 and October 2014 (Nelson et al. 2016b).

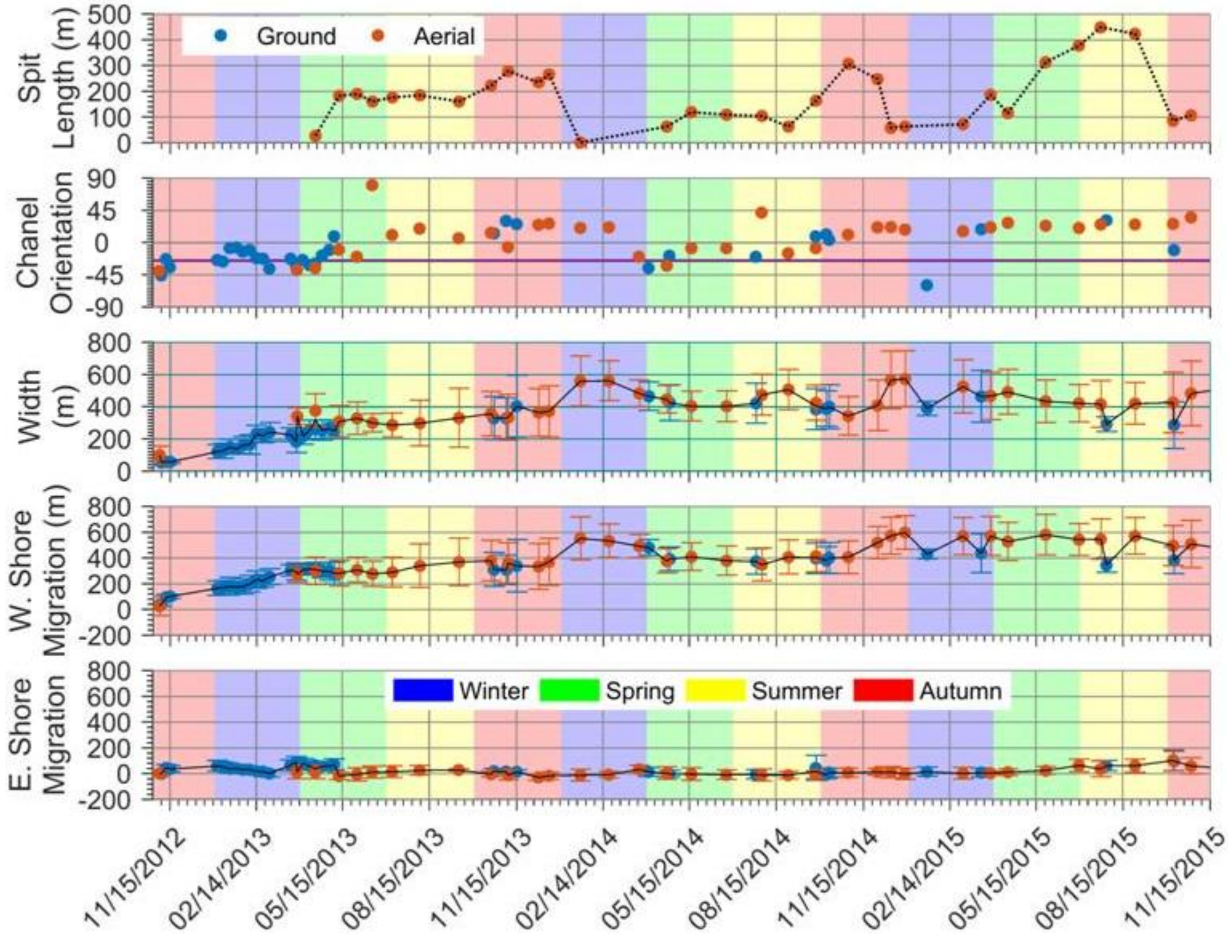

Figure 24. Changes in breach width and migration patterns between November 2102 and November 2015 (Hapke 2016).

The potential for continued changes in breach geomorphology was discussed at the January workshop, particularly with regard to breach width and continued westerly migration. There was general agreement that storm activity could result in widening, but uncertainty as to the ability of Great South Bay to maintain the breach in a wider configuration, especially since the two inlets at Fire Island and Moriches are stabilized through dredging. Primary controls on inlet migration were thought to be based on barrier beach stratigraphy documented in USGS unpublished sediment cores, which show erosion resistant materials located approximately 1.5 kilometers (0.9 miles) west of the breach centerline and in the marsh resource 0.5 kilometers (0.3 miles) east of the breach (Figure 25; Hapke 2016).

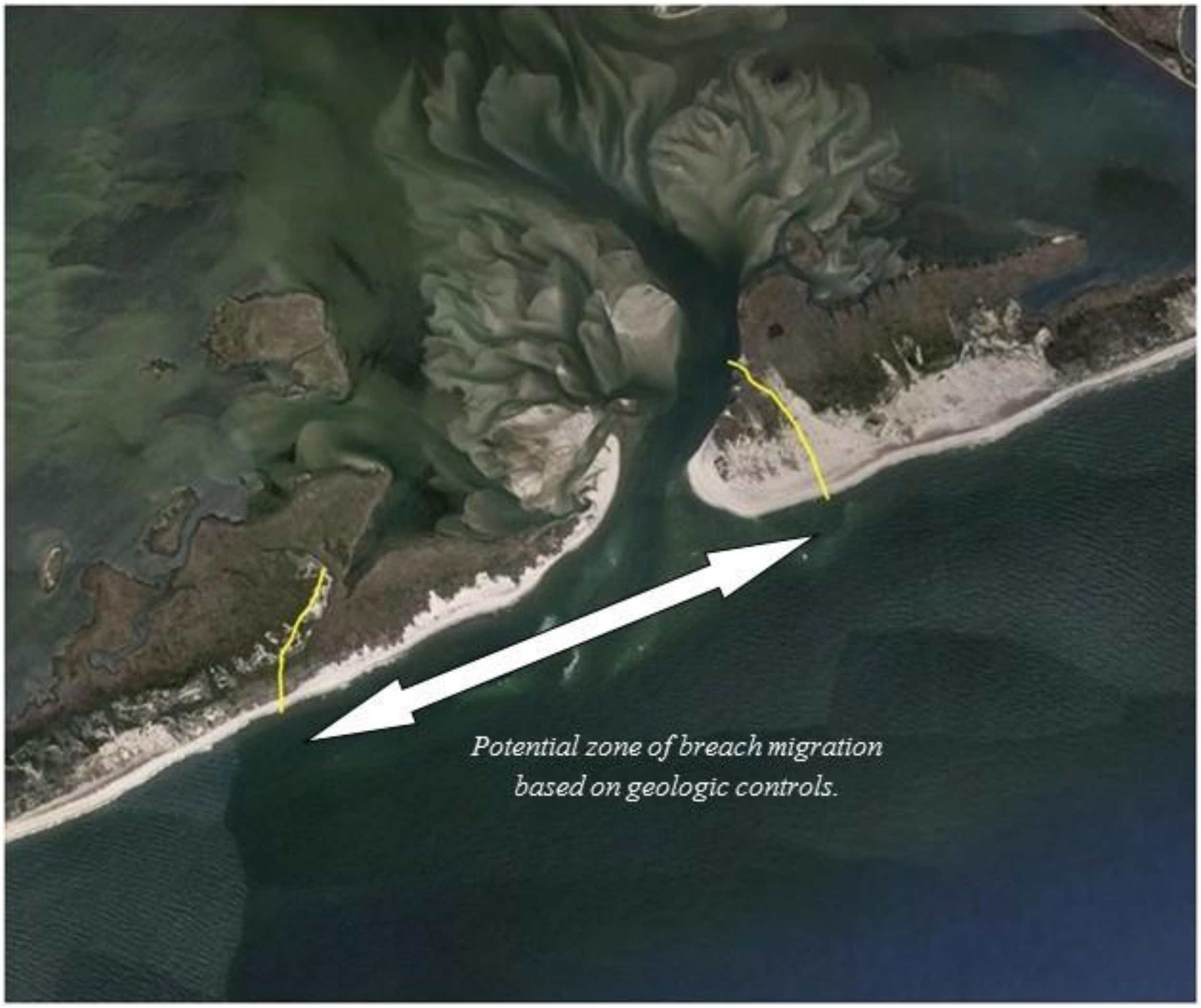

Figure 25. Projected zone of breach migration based on geologic controls identified through coring by USGS. Photo from GoogleEarth, May 23, 2015.

Summary of Breach Impacts on Sediment Transport and Geomorphology

The breach has not resulted in a significant interruption to longshore transport processes and therefore is not responsible for downdrift erosion beyond the immediate vicinity of the breach.

The breach no longer functions as a sediment sink and bypasses material relatively efficiently via ebb-tidal delta processes.

The cross-sectional area of the breach increased steadily for the first 2 years following formation, and has remained relatively stable since this time.

The centerline of the breach has migrated west approximately 200 meters (656 feet) since formation in 2012. Continued westerly migration is thought to be controlled by erosion resistant materials located approximately 1.5 kilometers (0.9 feet) west of the current breach centerline.

Climate Change

The changing climate is altering the climatic conditions upon which traditional engineering design has been based. Essentially, climate change is redefining risk for engineering projects. Historic events are no longer reliable proxies for future conditions. This creates a challenging environment for property owners, policy makers, investors, designers, insurers, and the general public when evaluating potential projects and management options. Additionally, the impacts of climate variability and extreme weather events are often felt more intensely in coastal areas because the coastal zone defines the confluence of marine and terrestrial processes. For instance, coastal communities are more vulnerable to increased flooding due to both sea level rise and projected increases in precipitation and river flows as a result of climate change (Kirshen et al. 2008; USCCSP 2008; Bosma et al. 2015). Flooding probabilities are also expected to increase in the coming decades due to climate change and probable increases in the intensity and frequency of coastal storms (Thomas, Melillo, and Peterson 2009). This is especially true in the northeast, where ocean dynamical mechanisms have the potential to further exacerbate relative sea level rise (Yin 2012). This increasing vulnerability along coastlines is further heightened by the density of people residing in coastal areas. It is estimated that over 50% of the population in the United States now live in coastal zones, and this number is projected to increase (Wilbanks et al. 2008). Populations and infrastructure are not the only elements being influenced by climate change. Natural resources and coastal processes will also be affected by the projected sea level increases and extreme storm conditions. As such, climate change should be a key consideration for any project located along the coast.

The changing climate, and specifically projected sea level rise and increased storm intensity and frequency, is a significant factor that is expected to impact the Fire Island area and the potential breach formation and dynamics in the future. Currently, the existing technical studies evaluating the post-breach conditions have not focused on the potential impacts of climate change. Rather the focus has been more geared towards the short-term impacts (1–5 years) associated with the breach opening, as well as potential mid-term changes (5–15 years) expected to occur if the breach is allowed to stay open. This is likely a reasonable approach when considering near term risk and comparing conditions existing pre- and post-breach.

However, from an economic perspective and when considering more mid- to long-term conditions, it is advisable to consider management alternatives and associated impacts under the lens of a changing climate. For example, it is likely that due to climate change, future breaches at the location will become more frequent and prevalent. Longer-term management strategies for the Seashore will likely be shaped by changes to the natural environment brought about by climate change.

Ecological Resources

The marine and estuarine resources in the vicinity of the Fire Island wilderness breach and Great South Bay are affected by physical, chemical, and biological processes. Prior to the formation of the breach, the ecological community in the area of the breach was primarily composed of estuarine habitats and species. Existing inlets, including Fire Island and Moriches Inlets, provided the only connectivity for bay water and organisms with the ocean. The formation of the breach created a new connection between Great South Bay, western Moriches Bay, and the Atlantic Ocean, providing a new portal through which water and organisms could transit, and allowing a greater influence of the ocean on the bay's ecosystem.

The breach has resulted in changes to marine and estuarine resources in Great South Bay, Narrow Bay, and western Moriches Bay. The affected resources include water quality, wetlands, submerged aquatic vegetation (SAV), benthic communities, hard clams, finfish and decapods, and ecosystem structure and processes. The information presented in this technical synthesis report for marine and estuarine ecological resources addresses the current science regarding ecological changes as a result of the breach in Great South Bay.

Biological Water Quality and Phytoplankton

The term "water quality" typically describes both the physicochemical and biological characteristics of a waterbody that influence the abundance and distribution of aquatic organisms, including those at higher trophic levels (i.e., fish). The key physicochemical drivers of water quality include nutrients, salinity, temperature, and dissolved oxygen levels. Biological parameters important to water quality determinations include measures of the density, species composition, and distribution of phytoplankton, harmful algae, and coliform bacteria. These biological organisms are sensitive to nutrient levels, and other physicochemical parameters, and therefore provide a natural indicator of water quality. Fecal coliform bacteria density and species composition are water quality measures that are very useful for assessing the potential risk posed to human health from direct uses of a waterbody, such as swimming or shellfish consumption. Water quality monitoring is a common scientific practice used by local, state, and federal agencies throughout the United States to evaluate and monitor the health of aquatic systems. Changes in water quality can be observed in the physicochemical and biological characteristics of a waterbody and are easily evaluated with laboratory analyses of water samples. Water quality and phytoplankton communities were included in this data synthesis due to their ability to detect short- and long-term changes in water quality as a response to perturbations, such as the breach documented at the Fire Island National Seashore.

The following section provides a synthesis and summary of available water quality and phytoplankton data relevant to the Great South Bay. General trends for temperature and salinity, and other physicochemical parameters, are briefly discussed in this section but are discussed in greater detail in the "Physical Resources" section of this report.

Synthesis of Information: Comparison of Pre- versus Post-Breach

Nitrogen

Prior to the breach at Fire Island National Seashore, Great South Bay was ranked low for eutrophication compared to other estuarine ecosystems (Kinney and Valiela 2011). The absence of nutrient extremes in the bay, according to Kinney and Valiela (2011), was attributed to the ability of the surrounding watershed to retain land-derived nutrients. Prior to the breach, two inlets existed along the seashore, which provided some exchange of oligotrophic ocean water with the Great South Bay. Areas of the bay system located at a greater distance from the two inlets generally exhibited higher nutrient levels than areas closer to the inlets as a result of the water exchange. Data collected from Bellport Bay marina, which is located approximately 20 miles from Fire Island Inlet and approximately 9 miles from Moriches Inlet, showed some of the highest nutrient concentrations when compared to data collected closer to Fire Island Inlet (Flagg 2013). While the nutrient concentrations in Great South Bay may be lower than those found in similar estuarine ecosystems, recent research suggests that farms within the watershed of the bay contribute high nitrogen loads that influence nutrient concentrations, particularly in areas of Great South Bay that are far removed from oceanic water exchange.

More recent data collections (analysis based on Suffolk County data reported in Gobler, Collier, and Lonsdale 2014), performed after the breach formation, found decreased total nitrogen concentrations in the areas of Bellport Bay, Narrow Bay, and western Moriches Bay (figure 26). Formation of the breach shortened residence time in Bellport Bay, Narrow Bay, and western Moriches Bay, and thus can reduce nutrient concentrations (Flagg 2013). The decreased nutrient levels are likely a response to dilution of the nutrient rich estuarine water, a result of the post-breach increase in volume of ocean water moving into the bay.

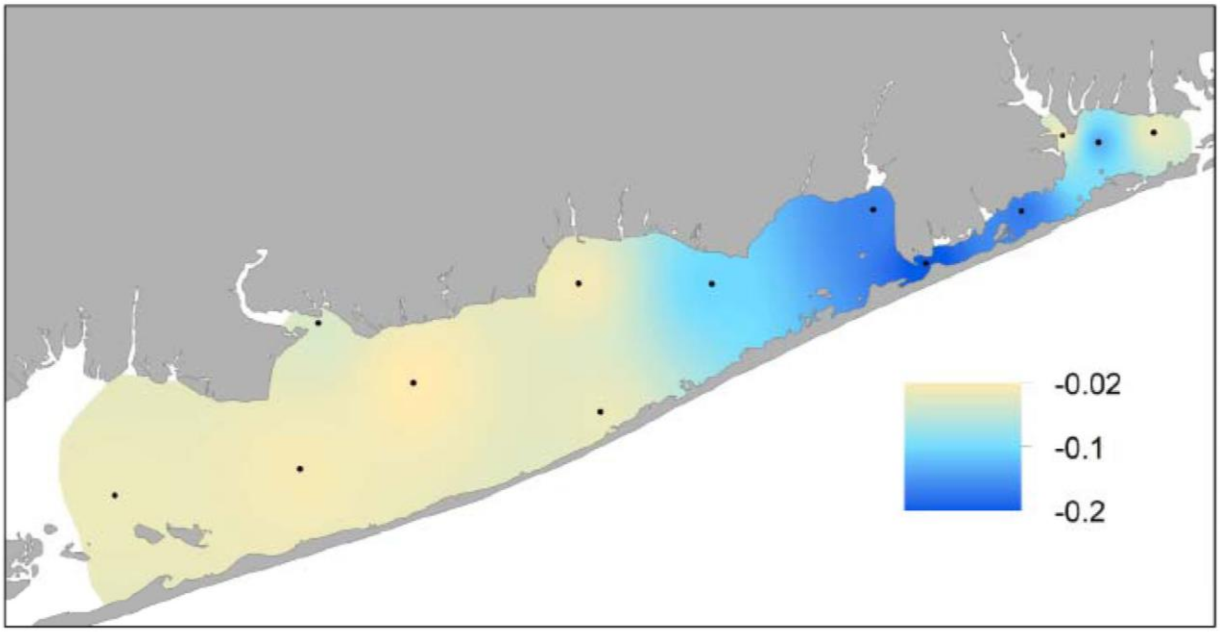

Figure 26. Change in total nitrogen (mg/L) from before the breach (average of 2000 and 2008 values) and after the breach (2013) (Gobler, Collier, and Lonsdale 2014).

Salinity

Data collected for this report indicate that salinity levels have increased approximately 5 psu in the Bellport Bay, Narrow Bay, and western Moriches Bay areas (Figures 27 and 28) since the breach formed (analysis based on data collected during a 4-week summer index period reported by Peterson 2014 and Heck and Peterson 2016; Gobler pers. comm. 2016). The increase was attributed to the influx of seawater coming through the breach. For a more detailed discussion on salinity, see the "Physical Resources" section of this report.

Temperature

An analysis of data collected during a 4-week summer index period (as reported by Peterson 2014 and Heck and Peterson 2016; Gobler, Collier, and Lonsdale 2014) shows summer water temperatures have decreased as much as 3°C in the Bellport Bay, Narrow Bay, and western Moriches Bay since the breach formed (Figures 29 and 30). Despite findings that in the vicinity of the breach and Moriches and Fire Island Inlets, there is a small decrease in summertime water temperatures (Figure 30), in general, water temperatures in Great South Bay are mostly dependent on air-sea interactions rather than bay-ocean exchange. Changes in the heat budget of the bay due to additional water exchange through the breach are planned by SoMAS for future hydrodynamic model experiments.

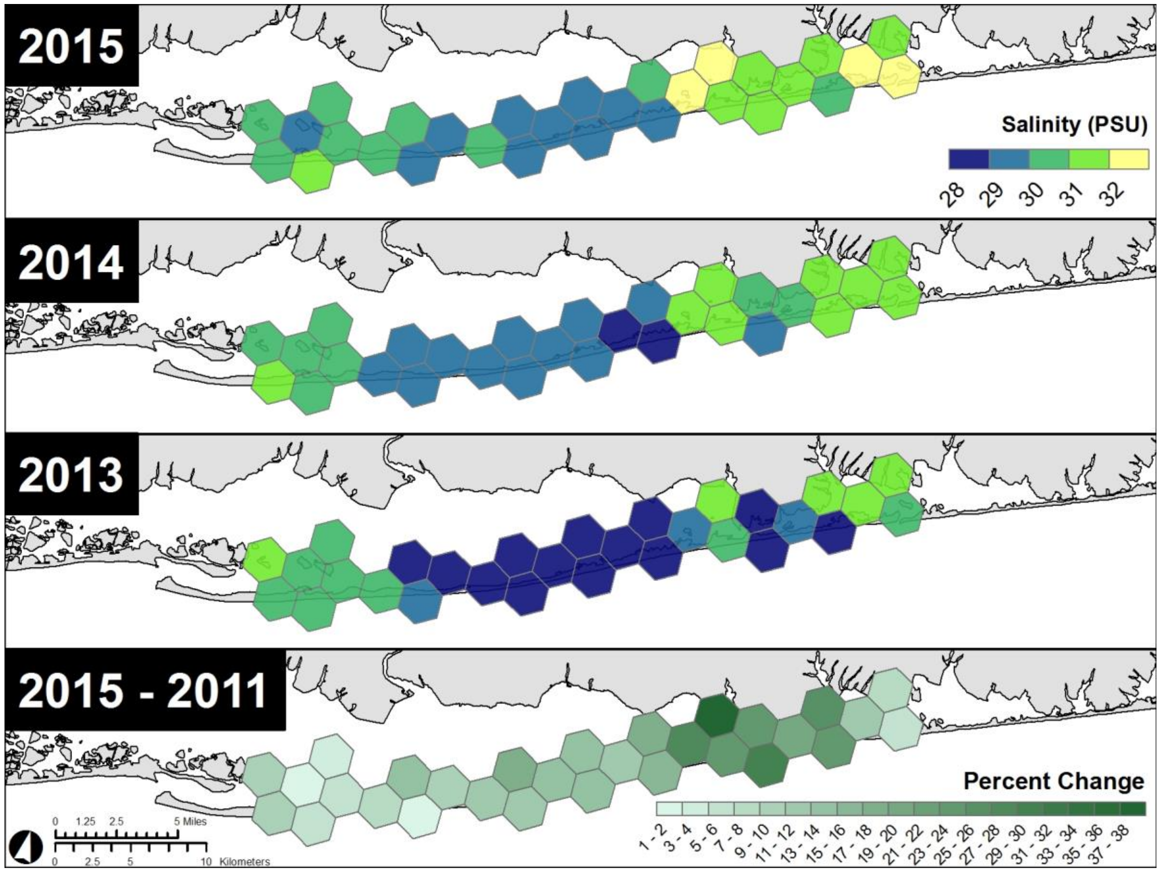

Figure 27. Salinity (2013, 2014, 2015) and change in salinity (2015–2011) (Heck and Peterson 2016).

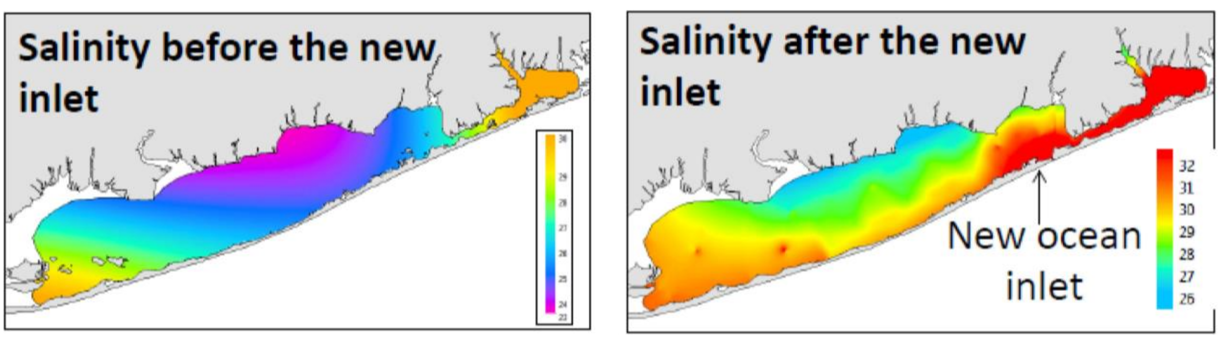

Figure 28. Salinity pre-breach (1976–2011, March) and post-breach (March 2013) (Gobler, Collier, and Lonsdale 2014).

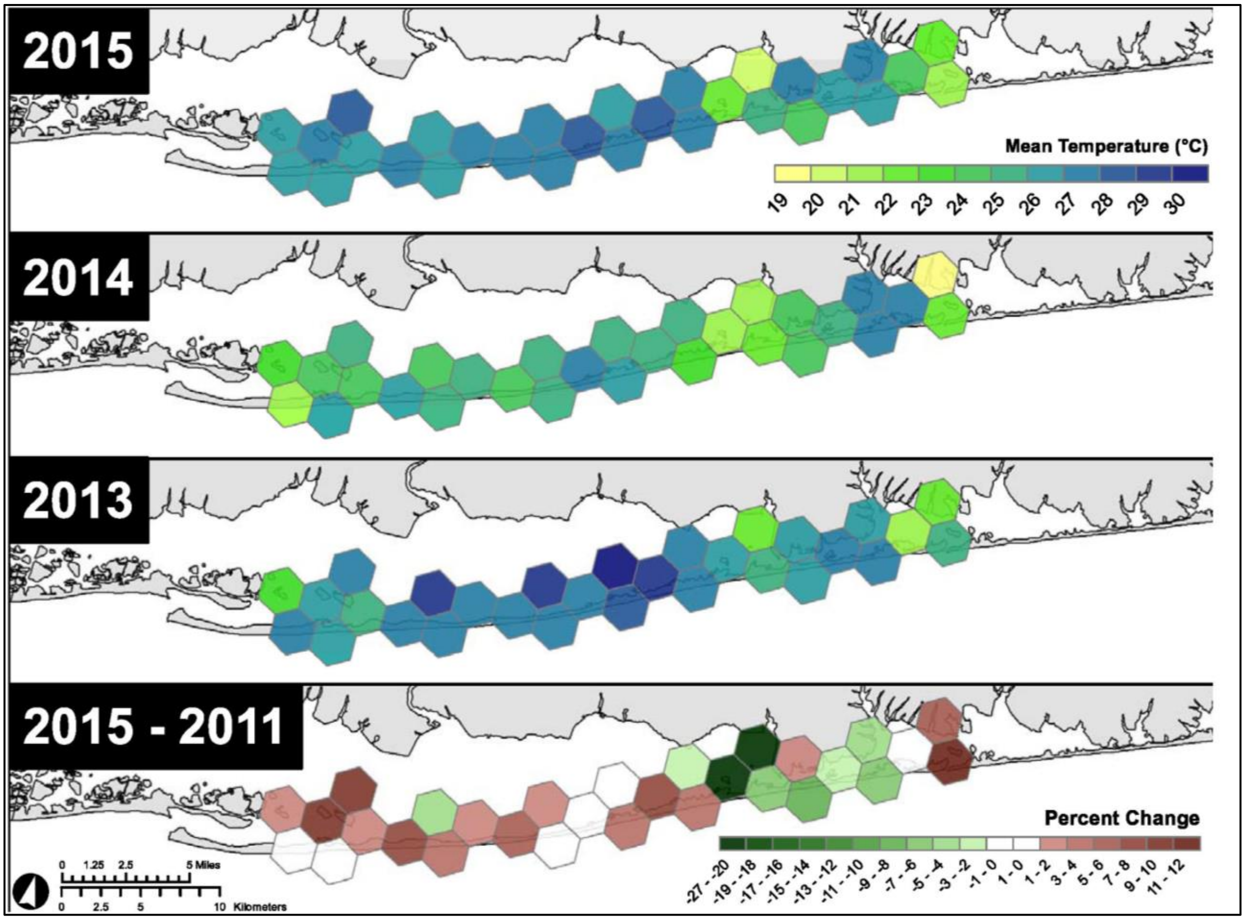

Figure 29. Temperature (2013, 2104, 2015) and change in temperature (2015–2011) (Heck and Peterson 2016).

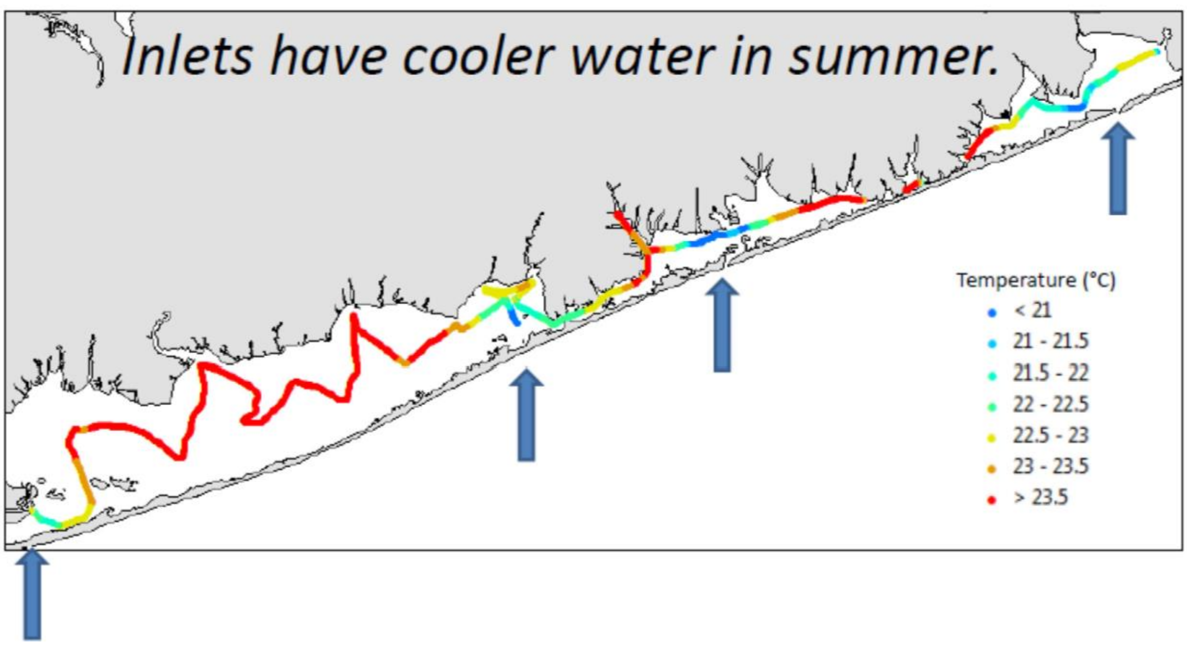

Figure 30. Temperature post-breach (August 2013) (Gobler, Collier, and Lonsdale 2014).

Dissolved Oxygen

Prior to the breach, dissolved oxygen data were collected from six sampling stations from the Fire Island Inlet to Bellport Bay (USACE 2006b). Overall, no significant differences in dissolved oxygen existed among the six sites and no distinct patterns of increase or decrease in dissolved oxygen concentrations were observed during the months of sampling (June through November). Surface dissolved oxygen levels typically followed expected seasonal trends documented at the stations in Great South Bay, with mean dissolved oxygen levels of 9.5 milligrams per liter (mg/L) during the May to November sampling period. Post-breach formation, trends in dissolved oxygen concentration at the immediate area of the breach have been variable. Gobler, Collier, and Lonsdale (2014) noted a net increase of approximately 1 to 1.2 mg/L in dissolved oxygen after the breach formed. Gobler (pers. comm. 2015) suggested that this increase in dissolved oxygen could potentially alleviate overnight periods of hypoxia (Figure 31). In contrast, Peterson (2014) found an overall net decrease of between 1% and 5% dissolved oxygen in Great South Bay (corresponding to concentrations of approximately 7 to 10 mg/L pre-breach and 6 to 8 mg/L post-breach) from 2011 to 2015 (Figure 32). However, areas directly adjacent to the breach showed a small net increase (1% to 3%) in dissolved oxygen concentrations during the same time period, suggesting that dissolved oxygen concentrations were variable in the years following the breach at both the immediate breach area and areas surrounding the breach (Peterson 2014). Despite these improvements in daytime dissolved oxygen, nighttime dissolved oxygen levels are still capable of reaching anoxic levels in North Bellport Bay (based on monitoring at the Bellport Bay Yacht Club) since the breach formed (Gobler pers. comm. 2016; USGS 2016b).

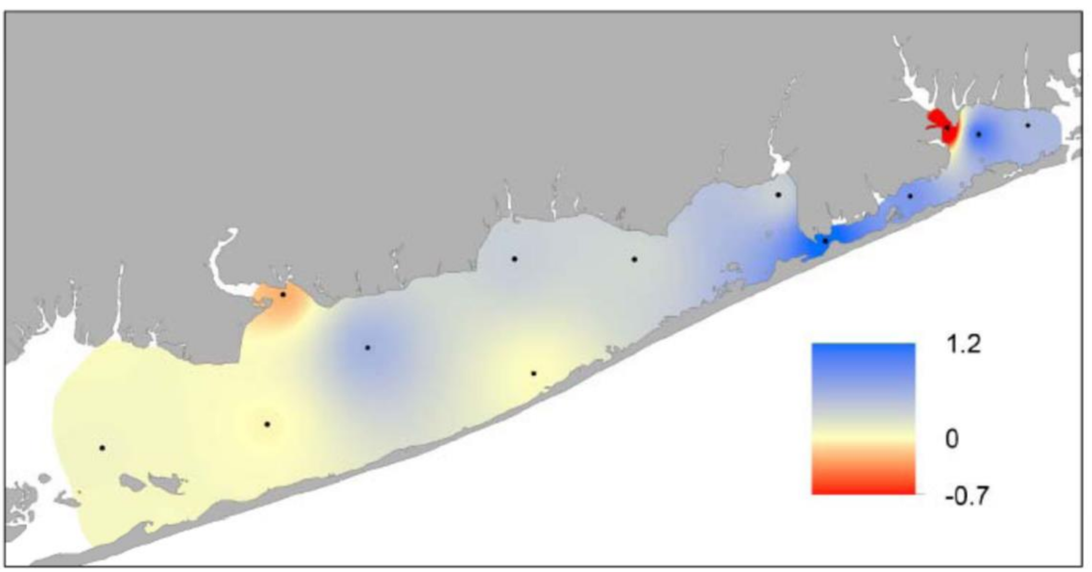

Figure 31. Change in dissolved oxygen (mg/L) from before the breach (average of 2000 and 2008 values) to after the breach (2013) (Gobler, Collier, and Lonsdale 2014).

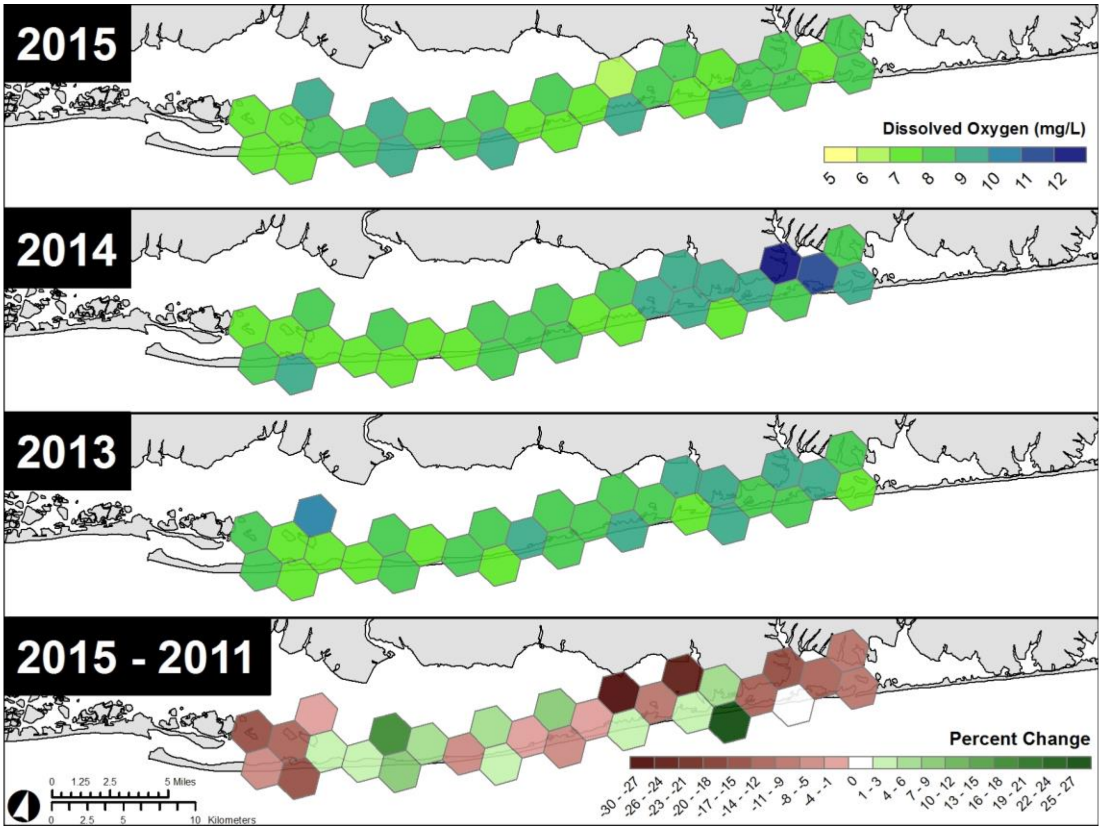

Figure 32. Dissolved oxygen (2013, 2014, 2015) and change in dissolved oxygen (2015–2011) (Heck and Peterson 2016).

Phytoplankton

Water column photosynthesis (primary production) drives many biogeochemical and ecological processes in coastal waters. At the base of aquatic food chains, phytoplankton drive gross primary production, the rate of conversion of carbon dioxide to organic carbon and measured as grams carbon per square meter per year (g C/m2/yr). Phytoplankton production, species composition and timing, and distribution within aquatic systems are important determinants of the quality and quantity of food available for consumer organisms, and under certain conditions can have a profound negative effect on physicochemical processes and water quality. Major phytoplankton blooms restrict sunlight availability and reduce the photic zone depth (area from surface of water to maximum depth of sunlight penetration) in aquatic systems. The photic zone is the location where photosynthesis and gross primary production occur; reducing this zone can result in mortality of seagrass or other organisms.

The die-off of phytoplankton can result in hypoxic (insufficient dissolved oxygen) conditions. Phytoplankton die-off is driven by an excess of nutrients in the water column that subsequently increases phytoplankton abundance. When phytoplankton become too dense in the water column, mortality occurs. Bacteria then consume the phytoplankton, and through that process, respire and deplete oxygen in the water column. Hypoxic conditions can cause widespread mortality in fish and invertebrate populations, the extent of which is highly dependent upon the timing and severity of the phytoplankton bloom. The community composition and densities of a phytoplankton bloom can be highly variable, as their rapid life cycle allows them to respond quickly to changes in light, temperature, and water quality. Several historical studies of Great South Bay chlorophyll and phytoplankton productivity have been conducted (e.g., Lively, Kaufman, and Carpenter 1983; USACE 2004c; Hinga 2005; McElroy et al. 2009), and those with most relevant data with regard to understanding the influence of the new breach are discussed here.

Areas of an estuary near an inlet tend to have lower phytoplankton production because of active mixing of estuarine waters with ocean water (Lively, Kaufman, and Carpenter 1983). A study of primary production by the USACE (2004c) found that mean chlorophyll levels substantially increased with distance from Fire Island Inlet and Moriches Inlet, where concentrations were typically less than 5 μg/L. In contrast, locations furthest from inlets averaged greater than 10 μg/L chlorophyll-*a*. The same study found that Moriches Inlet exhibited lower phytoplankton production than all other sites in 2001 and 2003. Similar findings were reported by Caldwell et al. (2015) who observed that water clarity was greatest near existing inlets (Figure 33), and by USACE (2004c) who found that total chlorophyll concentration at Moriches Inlet was significantly lower than all other sites in 2001 and 2003.

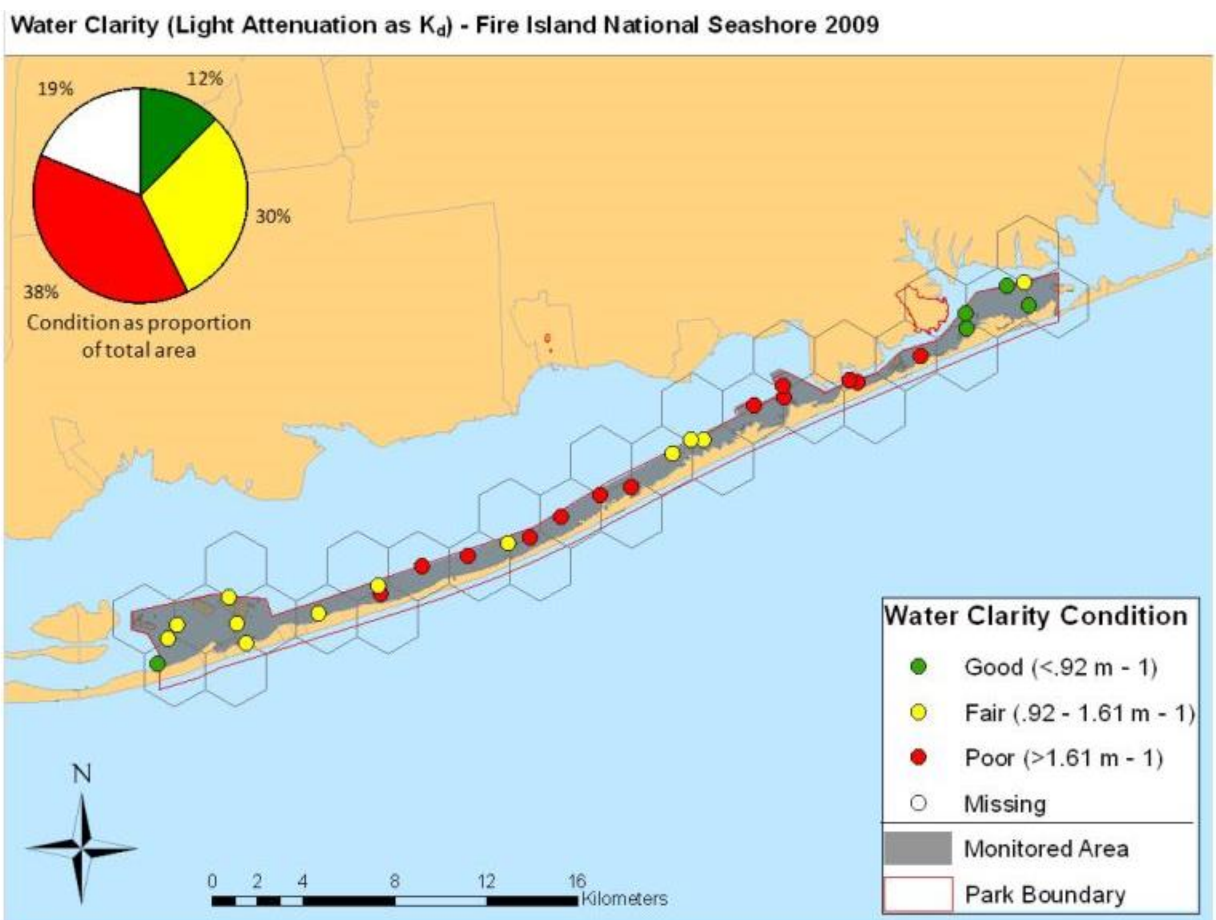

Figure 33. Water clarity conditions pre-breach at Fire Island National Seashore (Caldwell et al. 2015).

Proximity to inlets is also associated with larger form algae which provide higher quality food resources for suspension feeders (Weiss et al. 2007). A study by USACE (2004c) found that stations in Great South Bay located furthest from the inlets had the greatest mean abundance of smaller phytoplankton, while mid-bay sites had a greater mean abundance of larger phytoplankton relative to other sites, especially stations near inlets. The same study found that a station at Moriches Inlet, had more large form phytoplankton (>5 μm in diameter) than other stations regardless of flooding or ebbing tide. However, samples collected on the flood tide at Moriches Inlet generally contained fewer small form species, had less abundant phytoplankton, and lower chlorophyll concentrations.

Data collected post-breach in Great South Bay show a net decrease in phytoplankton production in the Bellport Bay (Peterson 2014, Gobler, Collier, and Lonsdale 2014). Prior to the breach, chlorophyll-*a* concentrations in the vicinity of the breach averaged 18 to 42 μg/L. Post-breach water quality monitoring in 2013 showed an initial drop in chlorophyll-*a* (16 to 20 μg/L) from pre-breach concentrations (Peterson 2014). In 2014, water quality data indicated that chlorophyll-*a* concentrations ranged between 6 and 15 μg/L, and were between 26 and 35 μg/L in 2015. Comparison of data from 2011 (pre-breach) and 2015 (post-breach) supports the overall observation of a net decrease in chlorophyll-*a* (Figure 34) in the water column near the breach (Peterson 2014). Figure 35 depicts the decline in chlorophyll-a since the breach formed (Gobler 2014). Note that the comparison shown in Figure 35 is between the *average concentrations during March for 25 years prior to the breach*, and the *average March 2013 concentration*. Comparison of years in which brown tide occurred before (2000 and 2008) and after (2013) the breach in Suffolk County also indicates a substantial reduction in chlorophyll in the Bellport Bay region (Figure 36) where seawater exchange has increased (Gobler, Collier, and Lonsdale 2014).

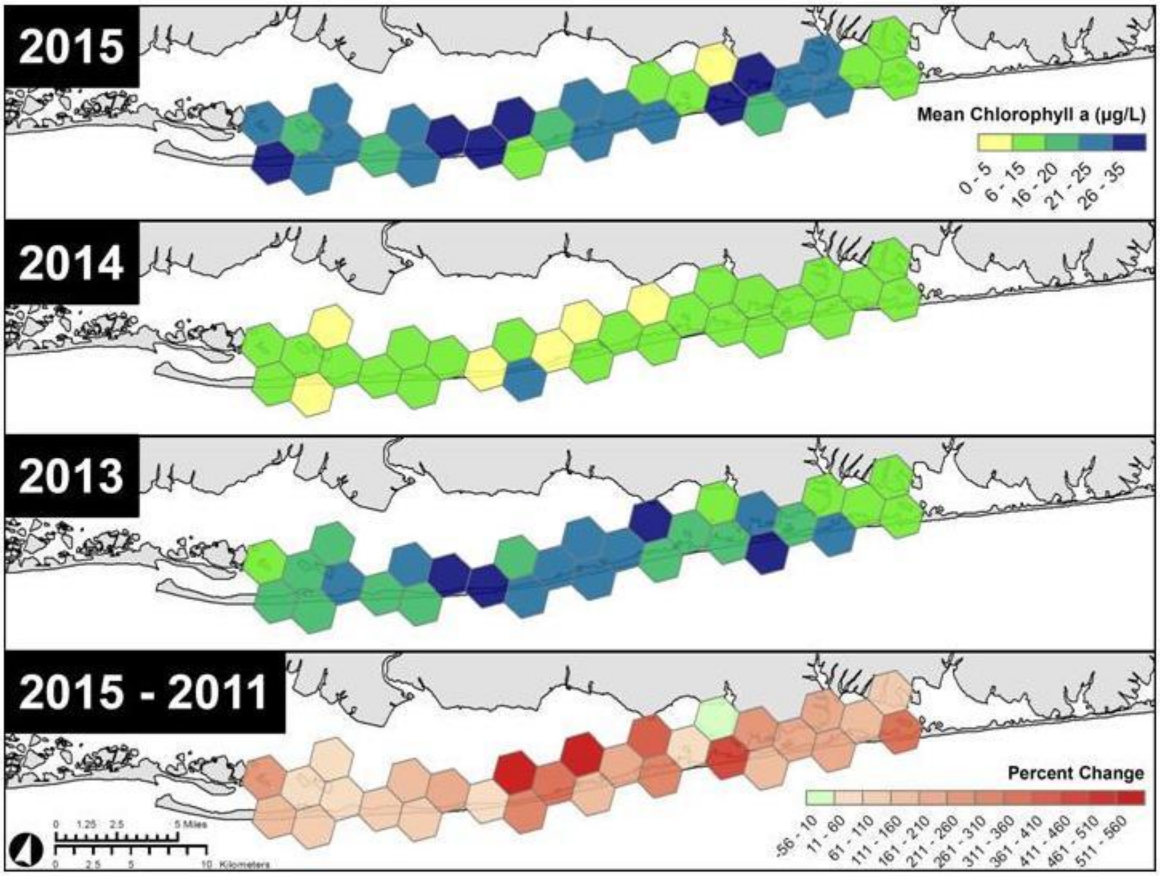

Figure 34. Changes in chlorophyll-*a* concentration in Great South Bay pre- and post-breach. Pre-breach is 2011; all other years are post-breach (Heck and Peterson 2016).

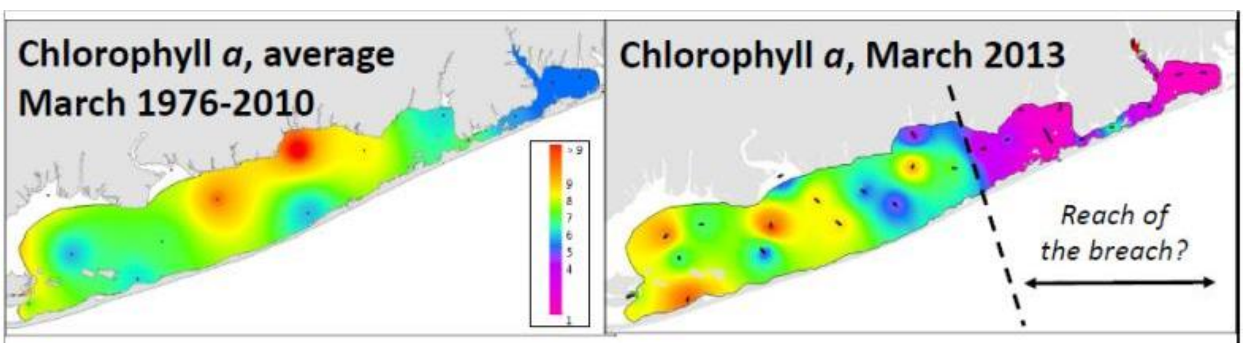

Figure 35. Chlorophyll-*a* concentrations pre-breach (average value for month of March, 1976–2010) and post-breach (average value during March, 2013) (Gobler, Collier, and Lonsdale 2014).

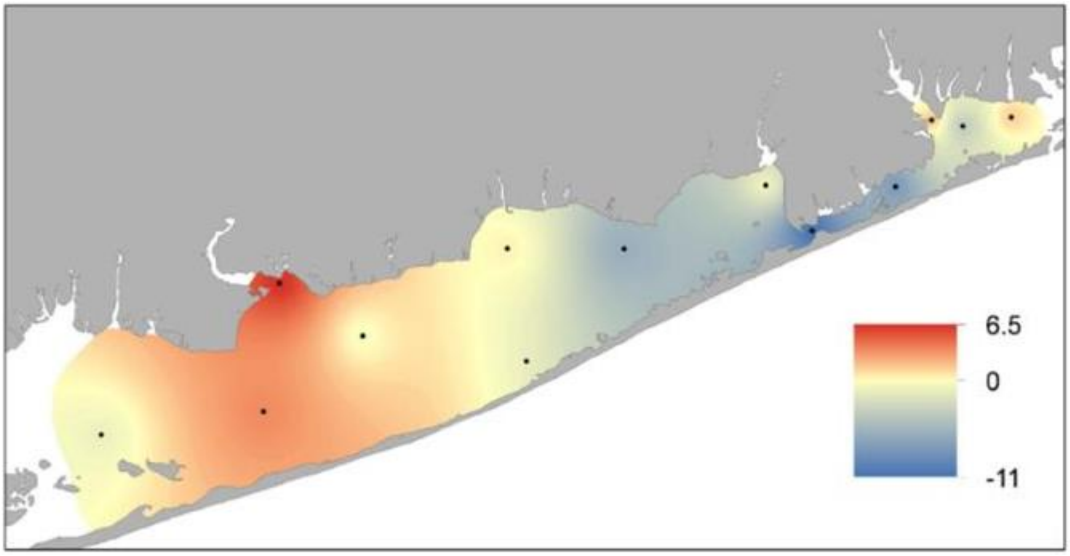

Figure 36. Reduction in chlorophyll-*a* concentration (ug/L) in brown tide year, post-breach (2013) versus pre-breach (average of 2000 and 2008 values) (from Gobler, Collier, and Lonsdale 2014, based on Suffolk County water quality data from pre-breach brown tide years 2000 and 2008, and one post-breach year, 2013).

Brown Tide

Blooms of harmful brown tide algae occur periodically in Great South Bay. Brown tides are considered harmful because they can cause an overabundance of water column chlorophyll, considerable reduction in light penetration, a reduction in dissolved oxygen in the water column, and are a poor source of nutrition for suspension feeders (Gobler, Collier, and Lonsdale 2014; Peterson 2014; Weiss et al. 2007). Brown tide-induced water quality impacts have resulted in decreased SAV biomass and reduced hard clam landings in Long Island bay systems (Gobler, Collier, and Lonsdale 2014). During very large blooms, *Aurecoccus anophagefferns* becomes nearly the only phytoplankton species present. Brown tide incidence appears to be related to nutrient and dissolved organic matter in the water column (Hinga 2005). First observed in Great South Bay in the 1950s (Ryther 1954), harmful blooms were infrequent for approximately 30 years after Moriches Inlet opened and duck farming practices were changed (USACE 2009). Starting in the summer of 1985, the brown tide species *A. anophagefferens* began to experience intense periodic blooms. Although brown tide organisms have been studied extensively and the genome of *A. anophagefferns* has been mapped (Gobler et al. 2011), the ability to predict the bloom cycle in any given year is still not possible.

Brown tides are characterized by frequency, occurrence, and intensity. Intensity can be described as the density or concentration of brown tide cells in a bloom. Pre-breach studies of brown tide intensity showed that brown tide cell concentrations were highly variable during blooms, ranging from less than 50,000 cells per milliliter (cells/mL) to more than 1 million cells/mL. In 2008, brown tide cells in the vicinity of the current breach location (Figure 37) were documented at concentrations between 300,000 and 475,000 cells/mL (Gobler, Collier, and Lonsdale 2014).

Figure 37. Brown tide (cells/mL) in Great South Bay, June 2013 (Gobler, Collier, and Lonsdale 2014)

Since breach formation, there has been a reduction in the intensity of brown tide in eastern Great South Bay, in the areas of Bellport Bay and the Narrow Bay (Gobler, Collier, and Lonsdale 2014). Likewise, an increase in the frequency and intensity of brown tide in central Great South Bay has been documented. This is indicated in 2013 brown tide sampling which showed cell densities averaged ~400,000 cells/mL, with some sampling stations in the central Great South Bay area having densities over 1.2 million cells/mL (Gobler, Collier, and Lonsdale 2014). In the vicinity of the breach (Figure 37), cell densities were lower (ranging from about 22,800–75,900 compared with an average of 400,000) during the 2013 bloom. Figure 38 shows the 2013 brown tide cell densities in both a bar graph, and a sampling station map. The figure shows that cell densities were lower at stations close to inlets and the breach (blue bars on graph), when compared with more distant stations (brown bars on graph). Arrows on the lower figure point to sampling stations located near the inlet and breach, where cell densities are lower. Frequency of brown tide blooms has also increased in Great South Bay overall; Brown tide blooms occurred in 4 of the 13 years preceding the breach (2000–2012) but occurred in each year since the breach: 2013, 2014, and 2015 (Gobler, Collier, and Lonsdale 2014).

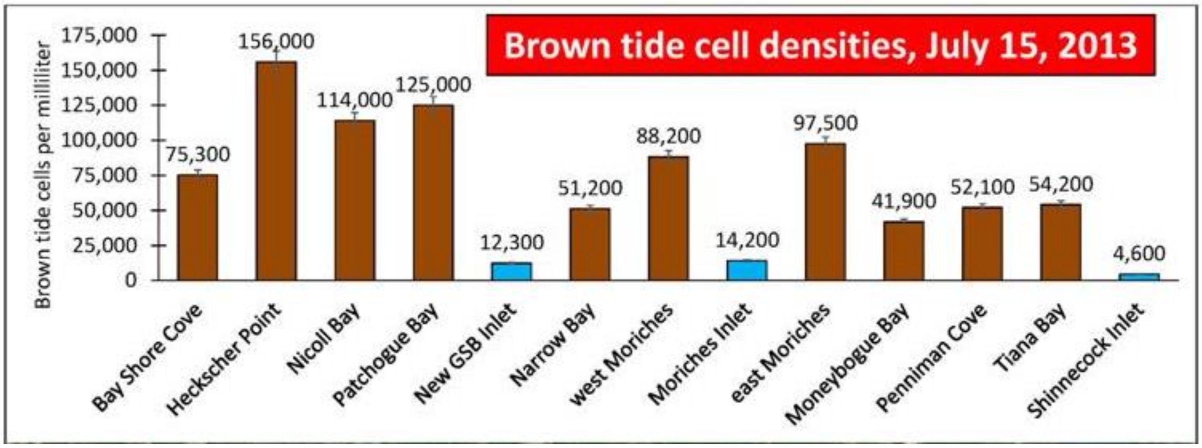

Figure 38. Brown tide cells in the Great South Bay lagoon system during the brown tide bloom in July 2013 (Gobler, Collier, and Lonsdale 2014).

More rapid reduction in the cell density of brown tide following the 2013 bloom occurred in areas closer to all inlets and the wilderness breach, with cell densities between 1 and 2 orders of magnitude lower near the inlets. Post-breach cell densities of the 2013 brown tide bloom were substantially lower than those measured during two bloom years (2000 and 2008) that occurred prior to the breach (Figure 39). Plankton community composition also differed spatially during 2013–2014, with large chain-forming diatoms dominant at the inlets, distinct from species found at central bay sites where ciliates and *Thallasoria* species were more common (Figure 40). This is notable because large form phytoplankton are a better food source for filter feeders, relative to small form species. Gobler noted a decline in the toxic dinoflagellate, *Dinophysis acumanata*, and pathogenic strains of bacteria in Great South Bay. Aerial photographs of the breach site (Figure 41) captured in 2014 showed a "flushing out" of brown tide cells resulting from water exchange between the estuarine and oceanic environments.

The occurrence of higher intensity brown tides in central Great South Bay compared to eastern Great South Bay may be attributable to increased water retention time in this portion of the bay brought about by new circulation patterns associated with the breach (Hinrichs 2016; Flagg pers. comm. 2015; Gobler pers. comm. 2015). However, water quality conditions in Central Great South Bay, including pathogen numbers and brown tide intensity and frequency, have been getting worse since the mid-2000s (Gobler, Collier, and Lonsdale 2014).

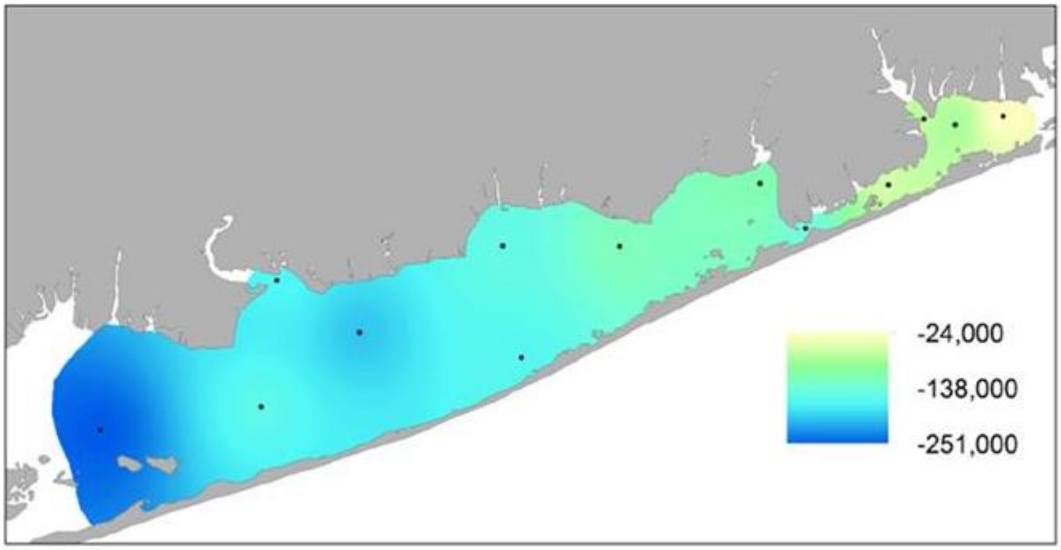

Figure 39. Average decrease in brown tide cell density (cells/mL) in Great South Bay lagoon system in brown tide years, post-breach (2013) versus pre-breach (average of 2000 and 2008 values).

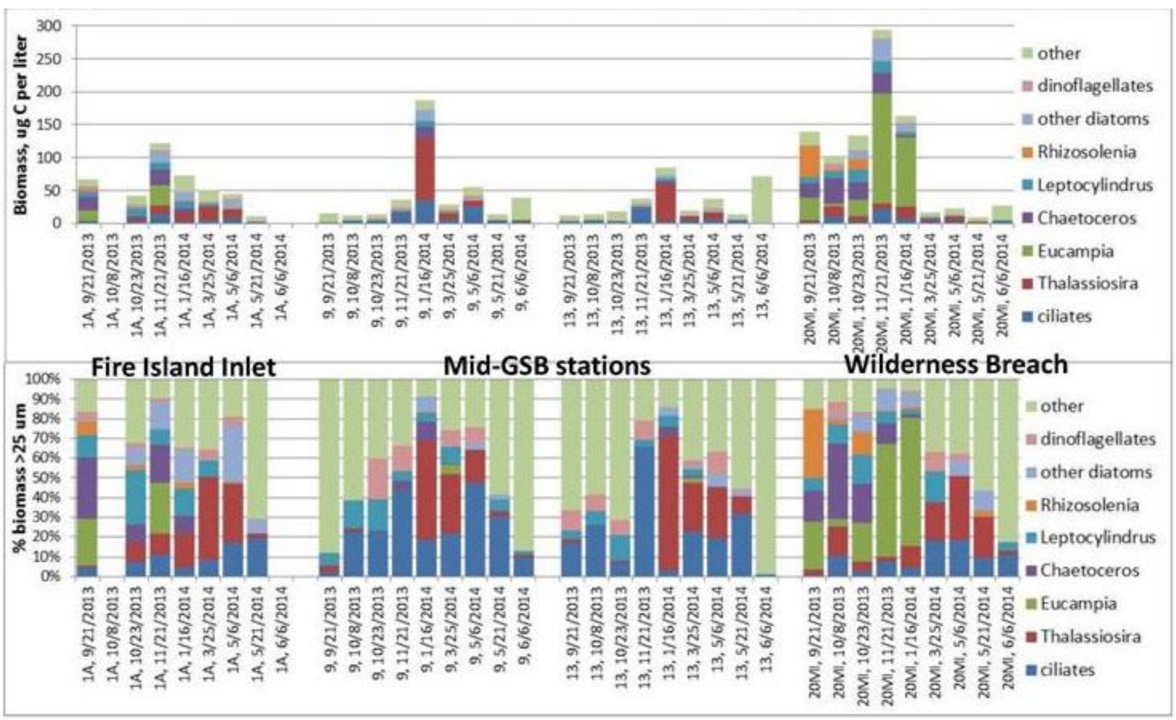

Figure 40. Phytoplankton community composition at inlets and mid-bay stations, Fall–Winter 2015–2015. FlowCAM data collection and analysis by Yuriy Litvinenko and Jackie Collier.

Figure 41. Wilderness breach exports brown tide (Fall 2014) (Gobler, Collier, and Lonsdale 2014).

Fecal Coliform Bacteria

Fecal coliform data are used by the New York State Department of Environmental Conservation (NYSDEC) to evaluate water quality in Great South Bay and to ensure that shellfish lands meet the sanitary criteria for certification during the period when they are certified. Post-breach changes in water quality have not affected the classification of seasonally certified[1] or uncertified shellfish lands (NYSDEC 2014a, 2015a). Post-breach survey data (from most recent triennial evaluation) indicates that certified and seasonally certified beds of hard clams are correctly classified as such; therefore, no changes in classification are necessary at this time (NYSDEC 2015a). There may however, be potential for changing the status in the southeastern area near Narrow Bay (currently closed year-round uncertified) as a result of a post-breach improvement in fecal coliform levels (Barnes pers. comm. 2016b). It is not yet known if this improvement is related to the breach. There will be no decision on this status change until 2018, when the next triennial evaluation is conducted.

Data Gaps

There were no long-term or systematic pre-breach phytoplankton data sets collected in the immediate vicinity of the breach prior to 2008. The ecological consequences of the breach-related changes in water clarity and quality are just beginning to be quantified, so it is uncertain whether the observed changes will remain over the long term. Decreasing intensity of brown tide blooms coinciding with the post-breach change in circulation patterns, are hypothesized to be associated with the formation of the breach. However, both the pathogen and nitrogen loading to central Great South Bay have been increasing for quite some time (prior to breach formation). Further, an increase in frequency and

[1] Seasonally certified and uncertified are designations assigned to shellfish lands by the NYSDEC. Hard clams may be taken only from areas designated as certified (or open) for the harvest of hard clams.

intensity of harmful algae blooms, and subsequent decrease in water quality, and failure to meet sanitary regulatory thresholds for shellfish consumption was an ongoing issue pre-breach occurrence. Several factors, in particular the ongoing trend in water quality reduction, natural variability in water quality and phytoplankton communities, and the relatively short time period over which breach effects have been evaluated, are sources of uncertainty. This uncertainty limits our understanding of the dynamic long-term effects of the breach on phytoplankton, water quality, algal blooms, or bacterial blooms.

Summary of Changes since the Formation of the Breach

The formation of the breach created a new portal through which ocean water can enter Great South Bay. In Bellport Bay, Narrow Bay, and western Moriches Bay, this exchange of ocean water has resulted in increased salinity, moderated water temperature, increased water clarity, decreased nitrogen, decreased chlorophyll-*a*, decreased brown tide frequency, lower brown tide cell densities during brown tide events, faster clearing out of brown tide cells following bloom events, and a change in species composition toward larger, chain-forming diatoms.

Wetlands

Wetlands occurring in the vicinity of the breach are primarily tidal marshes—beds of intertidal salt-tolerant grasses that are flooded and drained by the tide. These accretional environments accumulate terrigenous and biogenic sediments in response to tidal flooding and plant growth. Low saltmarsh is the most abundant wetland cover type on Fire Island, encompassing 670 hectares (26%) of the total area (McElroy et al. 2009). Most of these marshes are located along the north shore of the barrier island and consist of discontinuous patches of back barrier tidal fringe marsh. The most extensive salt marshes of the Fire Island barrier are located to the west of Watch Hill and extend to Moriches Inlet (Roman and Lynch pers. comm. 2016).

Synthesis of Wetland Vegetation Information

Fire Island supports a total of eight distinct salt marsh habitat subtypes, each with its own characteristic vegetation (McElroy et al. 2009):

- **Low salt marsh** is dominated by cordgrass (*Spartina alterniflora*) and occurs at the seaward border of the high marsh, along the edges of saltwater tidal creeks, and along mosquito ditches that drain the high salt marsh. This is the most abundant wetland vegetation type on Fire Island.
- **High salt marsh** is dominated by saltmeadow cordgrass (*Spartina patens*) or the dwarf form of cordgrass; large areas dominated by spikegrass (*Distichlis spicata*), black-grass (*Juncus gerardii*), and glassworts (*Salicornia* spp.) are also common.
- **Salt pannes** are dominated by the dwarf form of cordgrass and glassworts in shallow depressions within the marsh; salinity is higher in these areas due to trapping of salt water.
- **Northern salt shrub** is dominated by groundseltree (*Baccharis halimifolia*) and/or saltmarsh-elder (*Iva frutescens*) at the upland border of the high salt marsh.
- **Brackish meadow** is dominated by switch grass (*Panicum virgatum*) and saltmeadow cordgrass occurring at the upland border of the high salt marsh.
- **Oligohaline tidal marsh** occurs as a narrow band between high salt marsh and salt shrub vegetation, dominated by spikerush (*Eleocharis rostellata*) and twig-rush (*Cladium mariscoides*).
- **Brackish tidal marsh** is dominated by narrow-leaved cattail (*Typha angustifolia*).
- **Reedgrass marsh** is dominated by common reedgrass (*Phragmites australis*).

Marshes are dynamic ecosystems. Several processes including sediment transport from the ocean to the bay, overwash events, and the formation of flood-tidal deltas contribute to the development of new platforms for the colonization and establishment of salt marsh communities (Leatherman 1979; Roman and Nordstrom 1988; Donnelly et al. 2004). Overwash delivers sediment to the marsh surface and can increase the elevation of the marsh (Figure 42). The areas east of Watch Hill and extending to Moriches Inlet where salt marshes are most common on Fire Island, are known to have had several inlets and overwash events during the past two centuries (Leatherman and Allen 1985). In the area of the current breach and to the east (Great Gun), there is evidence from sediment cores, dating back to the late 1700s, that marsh establishment historically has coincided with the formation of inlets (Roman et al. 2007).

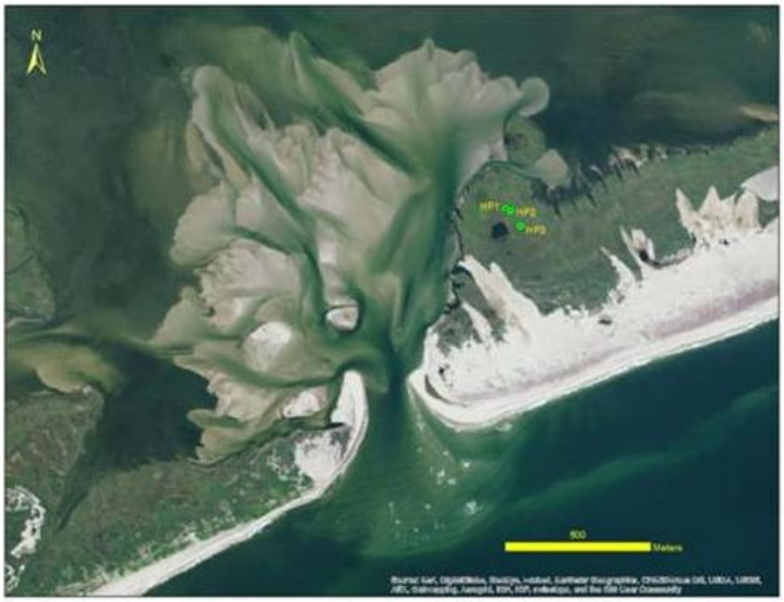

Figure 42. Aerial image (June/July 2013) showing location of the Hospital Point Marsh SET monitoring sites. Note the extensive flood-tidal delta associated with the Wilderness Breach and overwash sand deposits on the marsh surface in the vicinity of the SET sites (Roman and Lynch pers. comm. 2016).

Regionally, salt marshes are in decline due to several factors including sea level rise, development, sediment regime alteration, dredge and fill activities, wave action, coastal development, and nutrient enrichment (Cameron Engineering 2015; Wigand et al. 2014). An accelerated loss throughout Long Island has been observed in recent decades, based on comparisons of aerial images from 1975 and 2008 (Cameron Engineering 2015). From the mid-1970s to the mid-2000s, marshes in the South Shore estuaries declined over 10%. During that same time period in Great South Bay between the Fire Island Inlet and Smith Point County Park, over 200 acres of marsh were lost; and within the Fire Island National Seashore, 22 acres or 4.6% of the total marsh acreage was lost (Cameron Engineering 2015).

Nitrogen enrichment is one of the causes associated with marsh loss. Excess nitrogen enrichment can cause marsh plants to grow more aboveground biomass (leaves and stems) and less belowground biomass (roots and rhizomes), resulting in poor stability. The poorly rooted grasses eventually are unable to remain in place, and their loss destabilizes the creek-edge and bay-edge marsh resulting in slumping and exposing soils to erosive forces (NYSDEC 2014b). The destabilization of creek-edge and bay-edge marshes makes these areas susceptible to erosion and loss of marsh habitat. Excess nitrogen also stimulates microbial decomposition of organic matter within the underlying soils, which in turn can lead to increased production of hydrogen sulfide, reduction in pH, and further destabilization of marsh soils and vegetation (Deegan et al. 2007; Deegan et al. 2012; Wigand et al. 2014). It is possible that nitrogen enrichment could play a role in marsh loss at Fire Island (Roman and Lynch pers. comm. 2016); however, there are no site-specific studies demonstrating this.

For long-term sustainability, increases in salt marsh elevation must keep pace with sea level rise. If sea level rise exceeds marsh elevation increases, the marsh could become wetter and convert to unvegetated mudflat or open water. The NPS has been using surface elevation tables (SETs) and feldspar marker horizons to monitor the relationship between sea level rise and marsh elevation since 2002. Monitoring sites are located to the immediate east of the breach (Hospital Point Marsh, Figure 42) and 8 kilometers (5.0 miles) west of the breach (Watch Hill Marsh; Roman and Lynch pers. comm. 2016). The rate of increase in marsh elevation was measured as (mean ± standard error) 4.2± 0.24 millimeters per year (mm/yr) (Watch Hill) and 3.9± 0.12 mm/yr (Hospital Point) (Figure 43). These rates are slower than those estimated for sea level rise during the recent period of 2002 to 2015 in the region of Fire Island which ranged from 4.74 ±2.45 mm/yr at Sandy Hook, New Jersey, to 5.93± 2.32 mm/yr at Battery Park, New York (Roman and Lynch pers. comm. 2016). These rates are slower than those estimated for sea level rise during the recent period of 2002 to 2015 (the same duration of the SET monitoring) in the region of Fire Island which ranged from 4.19 + 2.47 mm/yr (0.16 + 0.097 inches/yr) at Montauk Point, 4.74 + 2.45 mm/yr (0.19 + 0.096 inches/yr) at Sandy Hook, and 5.93 + 2.32 mm/yr (0.23 + 0.091 inches/yr) at Battery Park, New York City (Roman and Lynch pers. comm. 2016). The authors of the study caveat that these rates have high variability (both sea level rise and marsh elevation change) and should be evaluated with caution. However, this work suggests that the short-term deficit in marsh elevation could lead to marsh submergence or alternatively, marsh elevation could keep pace if the rate of sea level rise were to slow.

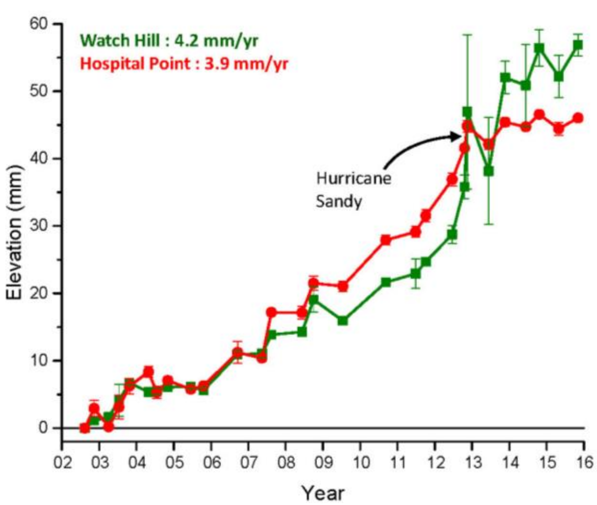

Figure 43. Marsh surface elevation change monitored using the SET method. Note the passing of Hurricane Sandy, October 28, 2012. Sample size is three SETs at each site with standard error presented. Rates determined by linear regression (p<0.0001).

Prior to the breach there was little salt marsh vegetation located where the breach channel was initially formed; however, as the breach migrated to the west and may continue (Figure 25), salt marsh will be lost to the breach channel (Figure 44). Small portions of salt marsh were smothered by overwash sand deposition onto the marsh surface (Figure 42). These overwash areas may be platforms for future salt marsh establishment as sea-level continues to rise.

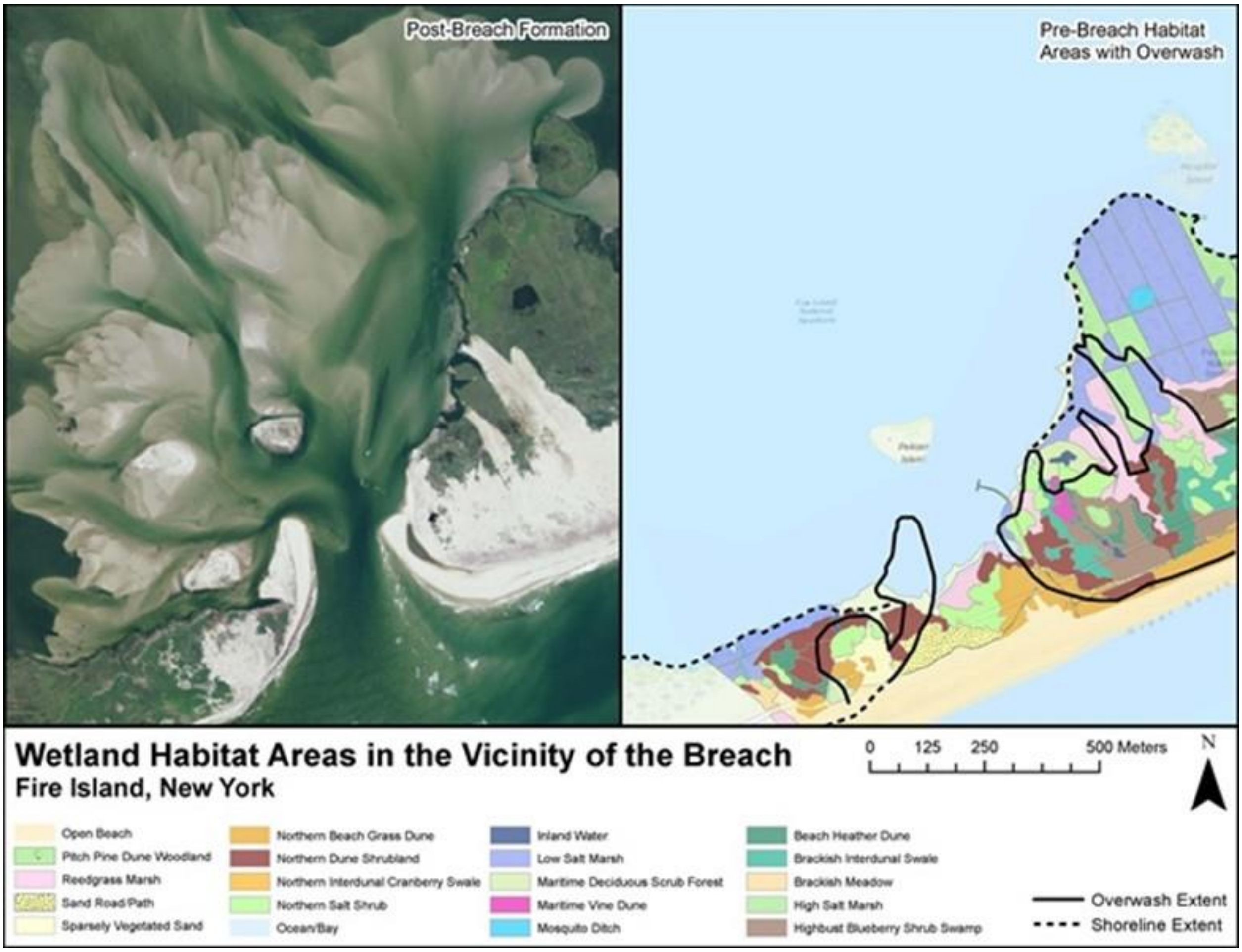

Figure 44. Breach area overflight and vegetation map showing overwash areas and affected habitats, including salt marsh. Figure shows the small amount of marsh vegetation affected by overwash.

Post-breach changes in sediment dynamics have occurred in the vicinity of the breach, and include the formation of the flood-tidal delta and overwash (Roman and Lynch pers. comm. 2016). Aside from localized overwash sand deposition on the marsh in the vicinity of the breach, there did not appear to be any widespread storm-related deposition onto the marsh surface, as noted from the lack of sediment accumulation above SET feldspar marker horizons (Figure 45) and the vegetation map with overwash areas indicated (Figure 44). The aerial extent of salt marsh affected by overwash was calculated using a GIS mapping tool that allowed the overwash areas (shown in the aerial photo on the left in Figure 44) to be superimposed on the vegetation map. Affected area was calculated as the sum of the area within polygons where salt marsh vegetation and overwash intersect. This analysis indicates a very small area, approximately 25,950 m^2 (279,323 $feet^2$) of salt marsh was affected by overwash.

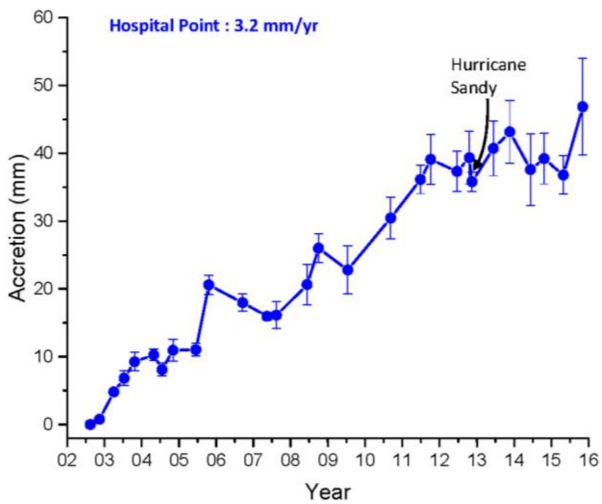

Figure 45. Marsh vertical accretion derived from the feldspar marker horizons. Sample size is three marker horizons at each site with standard error presented. Rates determined by linear regression ($p<0.0001$).

There was a slight increase in marsh elevation following the storm (Figure 46), but it was probably due to peat expansion from water saturation as opposed to sediment deposition (Roman and Lynch pers. comm. 2016). Marsh migration and the health of marsh plants depend in part on water quality. Improvements in water quality due to the mixing of estuarine with ocean water occurred with the formation of the breach. Increased mixing reduces nitrogen concentrations in surface water, potentially benefiting marsh plants. Improved water quality will provide additional benefit for the development of new marsh areas on the flood-tidal deltas as noted above.

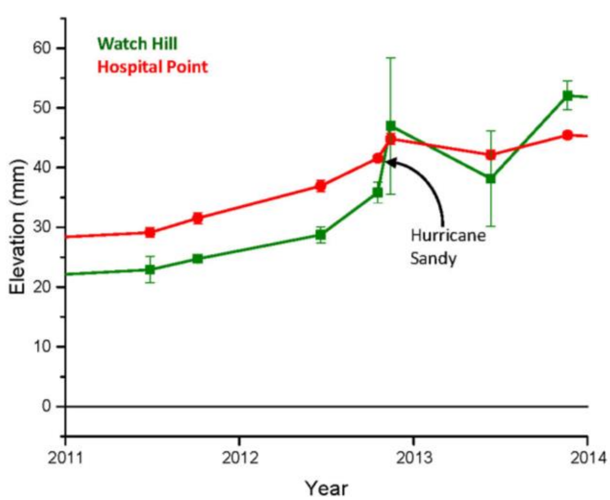

Figure 46. Marsh surface elevation change monitored using the SET method, focused on a few years pre- and post-Hurricane Sandy.

Data Gaps

Based on a review of post-breach data, no estimations have been performed to determine the acreage or type (e.g., high marsh, low marsh) of marsh habitat lost due to overwash and channel formation; although vegetation maps (McElroy et al. 2009 and NPS online mapping tool) suggest little or no marsh acreage exists in the immediate vicinity of the breach. Therefore, overwash-related burial would be very limited. To date, no post-breach study of the expected marsh development on new flood shoal and overwash area has been performed; development of new marsh is more likely to occur in the future and depends on the stabilization of the newly formed flood-tidal delta. Understanding of the dynamic long-term effects of the breach on wetlands in the vicinity is limited. However, the breach occurred recently, and marsh development takes place over longer time periods.

Summary of Changes since the Formation of the Breach

Changes in sediment deposition brought about by the overwash of sediment and the formation of the flood-tidal delta have created new platforms where new low marsh could develop in the future. Currently, the dynamic shifting sediment in the immediate vicinity of the breach is not conducive to marsh plant colonization. However, if this area becomes more quiescent, the low marsh plant, *Spartina alterniflora*, would be expected to become established. Post-breach improvements in water quality that have been observed will continue to encourage marsh development. Some sites immediately to the east of the breach experienced a post-breach increase in elevation, which may have been a result of peat expansion due to water saturation.

Submerged Aquatic Vegetation

Submerged aquatic vegetation (SAV) or seagrass beds function as vital habitat for numerous commercially, recreationally, and ecologically important fish and shellfish. Seagrasses play a major role in the nutrient and carbon cycles, provide an important food source, nursery habitat, and foraging area for various species, stabilize sediments, and improve water quality both by sequestering nutrients and trapping suspended sediments and organic matter. The presence of SAV is often used as an indicator of estuarine health and water quality.

Synthesis of Information: Comparison of Pre- versus Post-Breach

The two SAV species in south shore estuaries of New York are eelgrass (*Zostera marina*) and widgeongrass (*Ruppia maritima*). Eelgrass, the more common of the two species, is a perennial species in New York (with a few exceptions; NYS Seagrass Task Force 2009), and commonly found in salinity ranges from 10 to 36 parts per thousand (ppt) (NYS Seagrass Task Force 2009). Widgeongrass is a euryhaline, pioneering species that occurs sporadically in marine environments, withstands abrupt salinity pulses, has a broader temperature and salinity tolerance, and grows better in nutrient enriched environments that can be stressful to other seagrasses (Cho, Biber, and Nica 2009). Both eelgrass and widgeongrass are marine vascular flowering plants capable of sexual (flowers/seeds) and asexual (clonal) reproduction. Both are commonly found in shallow water where light levels are sufficient for photosynthesis, therefore the optimal depth for these species depends in part on water clarity. Compared to *Zostera*, *Ruppia* has lower peak biomass and lower value as habitat for SAV-associated species.

Areas of SAV are variable in structure (USACE 2004b). A 2003 survey found mean shoot heights ranged from 7.62 centimeters to 53.34 centimeters (3 inches to 21 inches). Average percent coverage was 52% in Great South Bay, 58% in Moriches Bay, and 43% in Shinnecock Bay (USACE 2004b). Macroalgae, which competes with and can be detrimental to seagrasses, is often present in SAV beds (USACE 2004b). Fifteen types of macroalgae were identified in association with the SAV beds in the USACE study (2004b). Dominant types included wire weed (*Ahnfeltia*), the red algae *Ceramium*, and the green algae *Chaetomorpha*.

Eelgrass provides 3-dimensional structure that serves as refugia for small fish and crustaceans, substrate for epiphytes and grazers, and preferred habitat for economically important species including bay scallops (*Argopecten irradians*). Eelgrass is also officially designated as Essential Fish Habitat for several interstate and federally managed fish species including summer flounder (*Paralichthys dentatus*), which supports the most economically important recreational fishery in New York. It provides forage for a number of waterfowl including brant and black ducks. Field studies of eelgrass communities show that both distance from, and biomass of, eelgrass beds in estuaries has a pronounced effect on the composition of the associated community of fishes, decapods, and crustaceans (McElroy et al. 2009 and citations therein).

Seagrass is declining in Great South Bay (McElroy et al. 2009). This is well demonstrated by comparisons of aerial photographs from 2002 with those from the mid-1970s that indicate a loss of SAV beds fringing the mainland south shore (north side) of Great South Bay from Howell’s Point in

Bellport west to the Robert Moses Causeway (USACE 2004b). Aerial photographs also show seagrass coverage in the South Shore Estuarine Reserve was 90% lower in 2003 compared to the 1930s when SAV thrived in Great South Bay (NYS Seagrass Task Force 2009; USACE 2004b). In the 1930s, 200,000 acres of seagrass were present (NYS Seagrass Taskforce 2009) in Great South Bay. However, in 2003 SAV beds totaled just over 20,000 acres and were limited to shallow areas (<2 meters (0.8 inch)) along the north shore of Fire Island and in the vicinity of the Fire Island Inlet (Figure 47).

Field studies also demonstrate the loss of SAV throughout Great South Bay. Currently, most of the eastern bay in the town of Brookhaven is devoid of any significant eelgrass beds, including the areas where eelgrass was historically abundant along the Fire Island shoreline (Cashin Technical Services Inc. 2011). Shallow areas between Watch Hill and Smith Point and formerly occupied by eelgrass were partially covered by patches of rooted widgeon grass and attached macroalgae, and the widgeon grass appeared to be taking over habitat previously occupied by eelgrass. Eelgrass beds were substantially reduced or nonexistent in the shallows around east and west Fire Island. Some eelgrass was observed in Babylon waters, but baymen reported a significant decrease in this area as well.

Declines of eelgrass have been attributed to multiple factors. In the 1930s, eelgrass nearly disappeared from its range due to "wasting disease" caused by a pathogenic strain of a marine slime mold, *Labyrinthula* (Short, Muehlstein, and Porter 1987). Although wasting disease was the major cause of eelgrass decline, other environmental stressors including warmer water temperatures may have been involved (Short and Neckles 1999). The wasting disease outbreak of the 1930s brought global attention to the importance and value of seagrass, as its loss was accompanied by declines in economically valuable species that use eelgrass as habitat. Although not entirely wiped out in the 1930s, eelgrass beds did not fully recover in the Great South Bay area until the 1950s (Hinga 2005). Losses since the mid-1970s were associated with a variety of stressors, including large- scale nutrient enrichment, organic enrichment, temperature increase, and sedimentation (NYS Seagrass Taskforce 2009).

Post-breach changes in both water quality and substrate, two factors that influence SAV communities, have been documented. The water in the vicinity of the breach is currently more marine in nature due to mixing with seawater, more moderate in temperature, and contains more oxygen. The more ocean-like conditions including clearer water with better light penetration, higher salinity, more moderate temperatures including cooler summer temperatures, favor eelgrass. The development of new flood shoals during Hurricane Sandy buried some SAV beds. However, the newly created sandy shoal areas provide a platform for development of new seagrass beds in locations where water depths are appropriate to support SAV. According to Peterson (2014; 2015a, 2015b) initially there was significant sand overwash that smothered some SAV sites near the breach and that sandbars shifted over other seagrass areas. Although the overwash areas recovered the following year, the sand bar areas did not recover (Peterson 2014, 2015b).

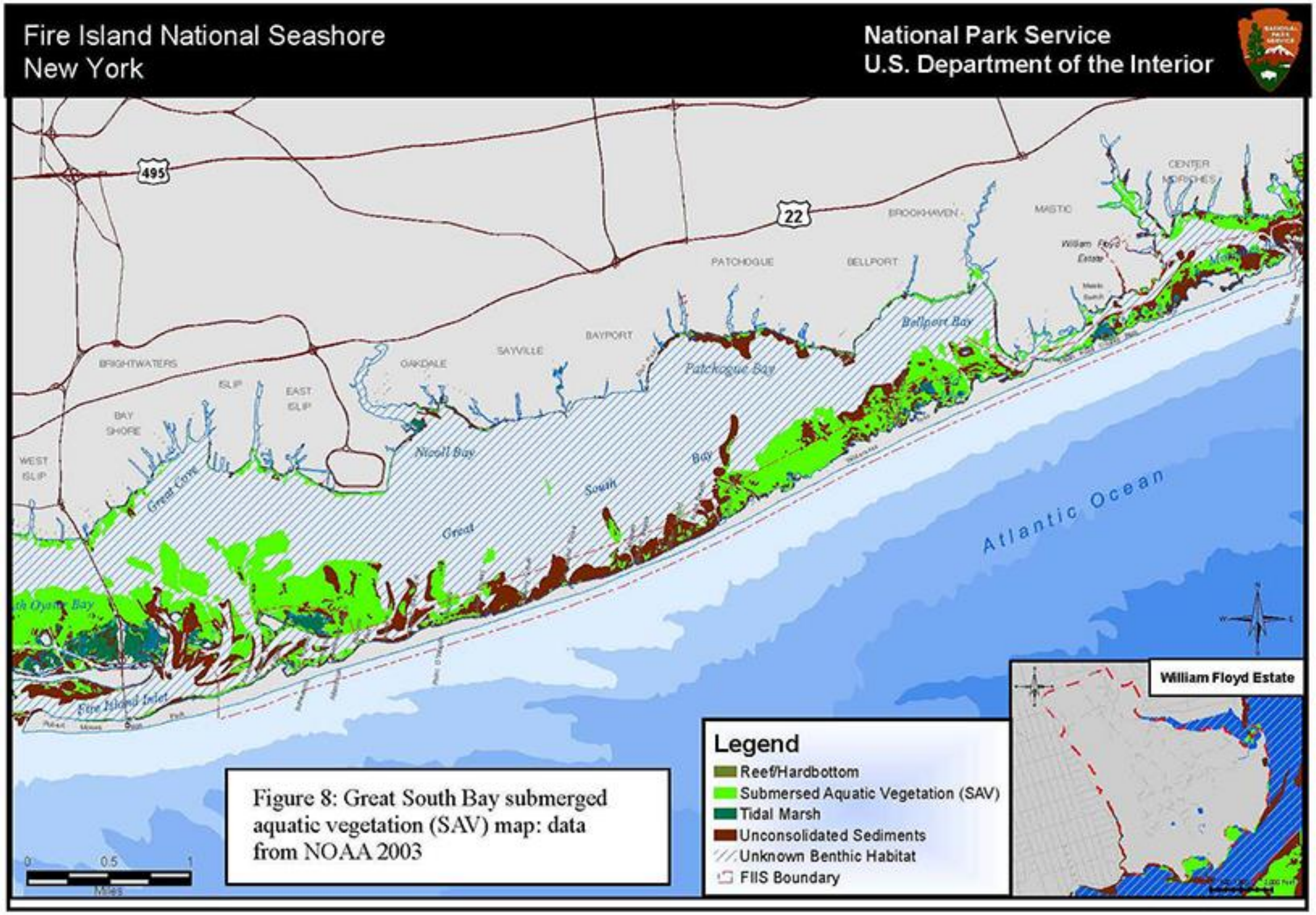

Figure 47. Submerged aquatic vegetation distribution in south shore estuaries (based on data from NOAA 2003 and published in McElroy et al. 2009).

Since the formation of the breach, eelgrass has increased in some areas east of the breach, despite declines in other areas. This is evident from systematic field surveys conducted by researchers at Stony Brook University. Initiated by NPS, this probability-based survey studied water quality and SAV in 2007, 2009, 2013, and 2014 at Fire Island National Seashore (Peterson 2015a; Heck and Peterson 2016; see the vegetation monitoring grid in Figure 48). This survey found an increase in eelgrass extent and percent cover throughout the study area between 2007 and 2009, followed by a noticeable reduction between 2009 and 2013 (Peterson 2015a, 2015b; Heck and Peterson 2016) (Figure 49, left side). In the vicinity of the breach, the survey found an overall loss of seagrass between 2009 and 2015, but a notable increase in eelgrass percent cover in certain areas east of the breach (Figure 49, right side) where water quality has improved. At one station approximately 1 kilometer (0.6 mile) west of the breach, eelgrass moved into shallow water but had later died out (Peterson 2015a, 2015b; and Heck and Peterson 2016). Widgeongrass was less common throughout the survey, and density changed at many locations, but the direction of change was not uniform throughout the study area. Between 2009 and 2015, widgeongrass density increased at 14 sites and decreased at 16 sites (Figure 50).

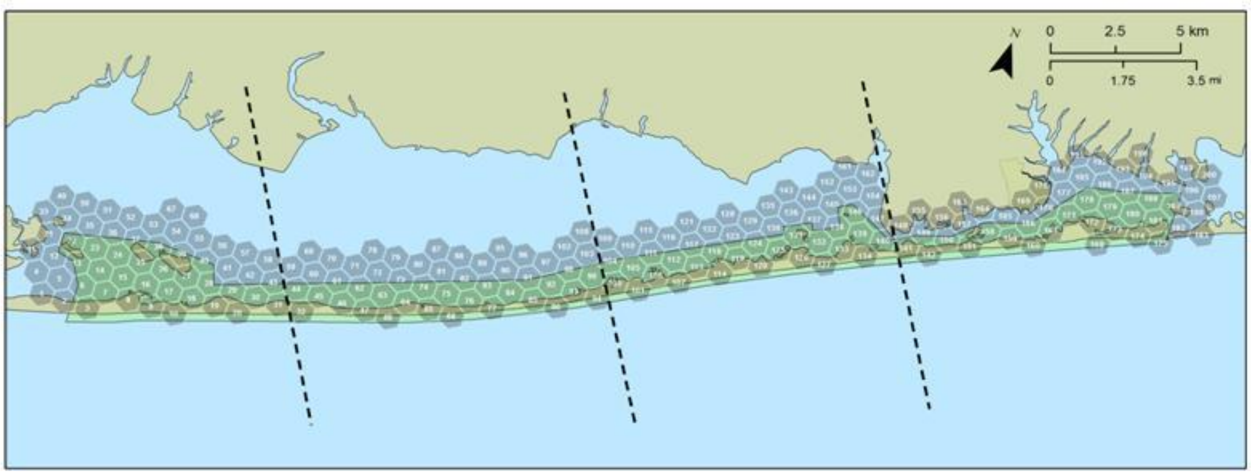

Figure 48. Submerged aquatic vegetation monitoring grid for systematic surveys conducted by Peterson 2015a; Heck and Peterson 2016. Dashed lines delineate sampling zones.

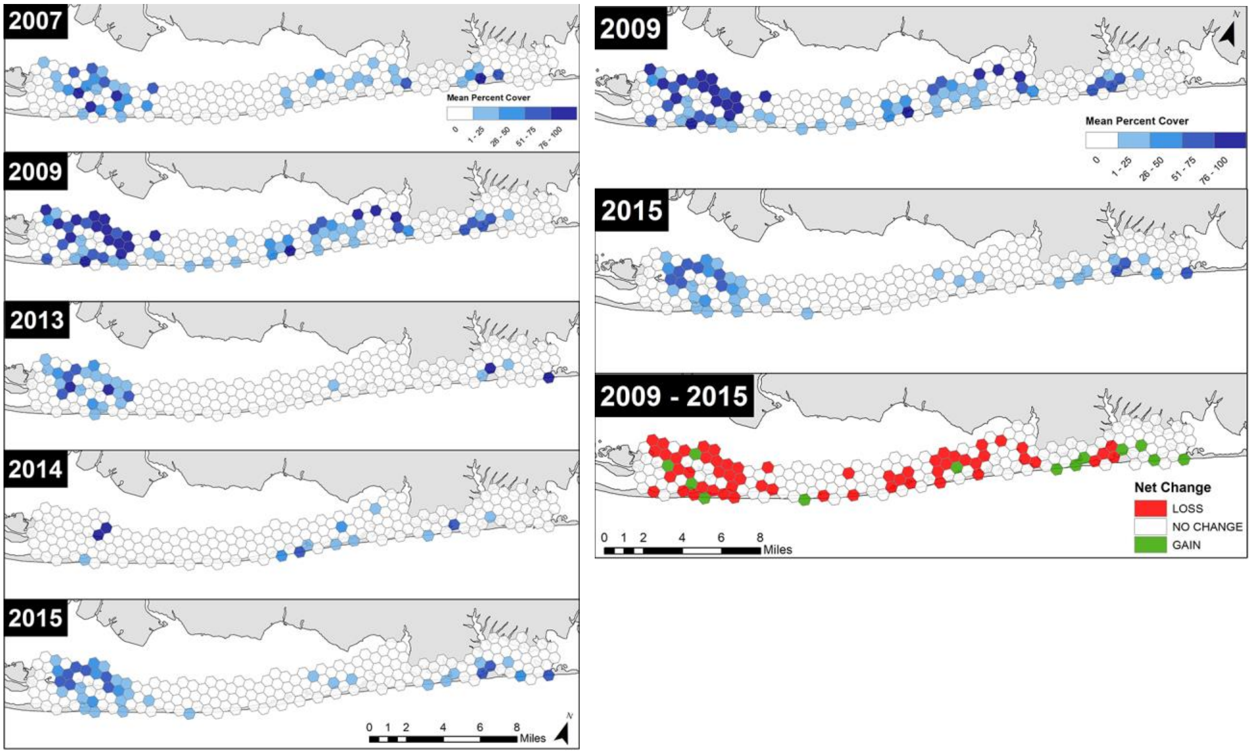

Figure 49. Eelgrass distribution 2007–2015 (left) and percent change 2009–2015 (right) (Peterson 2015a, 2015b).

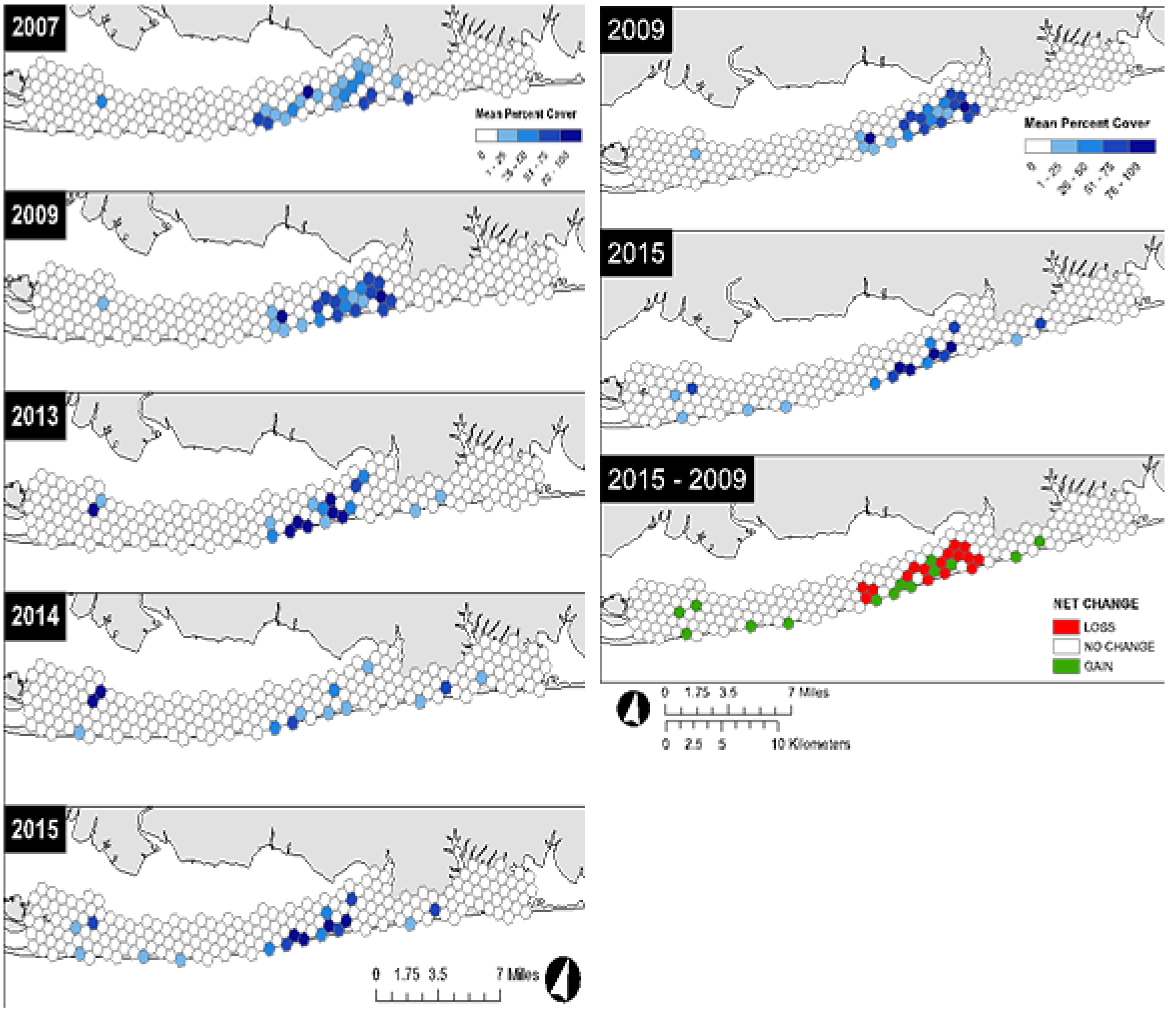

Figure 50. Widgeongrass distribution 2007–2015 (left), and change in percent cover 2009–2015 (Peterson 2015a, 2015b).

Data Gaps

There has been no long-term study of SAV distribution, species composition, and plant density in Great South Bay or in the immediate vicinity of the breach. The small amount of time that has passed since the formation of the breach limits our understanding of the dynamic long-term effects of the breach on SAV.

Summary of Changes since the Formation of the Breach

The formation of the breach has led to greater salinity, moderated summer water temperatures, and greater light penetration through the water column as a consequence of the mixing of water from Great South Bay with the ocean. This change in environmental conditions has favored the establishment of eelgrass communities just east of the breach. New platforms for SAV colonization were established following the breach in association with the flood-tidal delta.

Benthic Community

Benthic communities considered within this report include animals living in or on the sediment surface in subtidal and intertidal areas of the estuary. The wilderness breach has modified the benthic environment, and therefore may have had an impact on benthic communities. There are no pre- or post-breach benthic community data in the immediate vicinity of the breach. Therefore no field survey-based comparisons can be made to evaluate the effects of the breach. However, past studies of the benthic communities in Great South Bay do inform the discussion on the nature of benthic communities in this region.

Synthesis of Information on Benthic Communities

The proximity of a benthic community to an inlet has a direct effect on the composition of the community (Cerrato 2001). Benthic communities near Fire Island Inlet and Moriches Inlet have been described as "characteristic of a high salinity, high flow habitat" (Cerrato 2001). Abundant species in these near-inlet areas included the bivalves *Mytilus edulis* (blue mussel) and *Tellina agilis* (a bivalve mollusk), polychaetes (*Nephtys picta* and *Nereis arenaceodonta*), hermit crabs (*Pagurus longicarpus*), lady crabs (*Ovalipes ocellatus*), and the sea star (*Asterias forbesi*). Similarly, a survey conducted by the USACE (2004b) found that sampling stations nearest to inlets had the highest overall crab abundance, with green crabs dominant (88% of total catch) and other species much less common (blue crab, lady crab, rock crab, and portly spider crab each accounted for approximately 2% of overall catch). In contrast, areas further from the inlets had benthic communities that were more estuarine and less salt-tolerant in character. Abundant species included the polychaetes (*Sabellaria vulgaris* and *Trichobranchus glacilis*), snails (*Rictaxis punctostriatus* and *Acteocina canaliculata*), bivalves (*Mercenaria mercenaria, Mulinia lateralis*, and *Gemma gemma*), sand shrimp *Crangon septemspinosa*), and blue crab (*Callinectes sapidus*).

Large-scale spatial gradients have also been described in Great South Bay (Cerrato 2001). These gradients are thought to be related to differences in water column properties, which in turn are coupled to the existing Fire Island Inlet. For example the bivalve, *Tellina agilis*, and the lady crab (*Ovalipes ocellatus*), which prefer saltier water, were widely distributed in Islip waters but absent from Brookhaven waters in eastern Great South Bay; in contrast the razor clam (*Ensis directus*) which is less salt-tolerant, was abundant in Brookhaven waters but totally absent from western Great South Bay (Cerrato 2001).

Sediment type is also a strong driver of benthic community composition. In benthic samples collected in Great South Bay, macrofaunal abundance decreased with increased sediment grain-size, from 41,707 individuals per square meter in mud substrates to 19,418 per square meter and 26,096 per square meter in sand and shell substrates, respectively (Cerrato 2001, citing Larson 2000). A total of 148 distinct taxa were collected in benthic samples, with the number of taxa present increasing with sediment grain-size, from 91 in mud substrates to 112 in shell substrates. The pre-breach benthic subtidal community in unvegetated areas of Great South Bay, as described by Cerrato (2001), was diverse, highly affected by proximity to inlets, and strongly associated with sediment type.

Epibenthic communities are often associated with vegetation. Vegetated subtidal areas located on the bay side of Fire Island provide habitat for a number of epibenthic species including crabs. In a study by USACE (2004b) epibenthic communities were evaluated along with SAV in Great South Bay, Moriches Bay, and Shinnecock Bay in 2003. Common epibenthic species observed included the green crab (*Carcinus maenas*), Atlantic mud crab (*Panopeus herbstii*), eastern mudsnail (*Ilyanassa obsoleta*), grass shrimp (*Palaemonetes vulgaris*), golden star tunicate (*Botryllus schlosseri*), and red beard sponge (*Microciona prolifera*). Beds of SAV exhibited a diverse epibenthic community with 50 different species collected during the study.

The occurrence of intertidal benthic communities on the bay side of Fire Island is determined by frequent wetting and drying of intertidal areas. Sediment core samples completed in 2004 (USACE 2004a) collected a total of 13,218 organisms representing 68 different taxa using sandy sediments of the bay side intertidal habitats. The dominant taxa collected in the samples included Oligochaeta, Nematoda, Nematomorpha, *Corophium* sp. (a burrowing amphipod), and the amethyst gem clam (*Gemma gemma*). Pitfall trap sampling in the intertidal zone revealed 1,462 individuals from 83 different taxa, dominated by the Ephydridae (shore flies or brine flies) and Muscidae (house flies or stable flies) families, and the ant species (*Lasius neoniger*). The amphipods *Talorchestia longicornis, Talorchestia megalopthalma* (sand hoppers), and *Orchestia grillus* (a detritovore that feeds on *Spartina* and other marsh grass detritus) were also abundant in pitfall samples. Wrackline sight sampling revealed 1,268 individuals from 29 distinct taxa; dominants included the insects *Anurida maritima* (a small wingless insect) and *Acarina* spp. (mites). Other common groups in sight samples included bivalves, annelids, and amphipods.

Post-breach formation, the potential for changes in the benthic community prompted a workshop in January 2016, where SMEs shared information and discussed the potential breach-related changes in the benthic community. In general, SMEs asserted that it is likely that the benthic communities closest in proximity to the breach have changed, in response to increases in salinity, water flow, sediment grain size and cooler summer water temperatures, to more closely resemble benthic communities that occur in the vicinity of existing South Shore inlets. The shift in the community is likely to have occurred rapidly for populations of mobile, short-lived species, while populations of long-lived species including hard clams are expected to show slower changes[2]. The breach resulted in the burial of intertidal and subtidal communities located where flood-tidal deltas have formed. Formation of new habitat in response to the breach, and may have led to a shift in epibenthic species composition in the vicinity of the breach. For example, according to Cerrato and Frisk, it is possible

[2] Benthic communities respond on a scale of months to years following a disturbance (Wilber and Clarke 2007; Van Colen et al. 2010; Kotta, Kotta, and Kotta 2009; Keay and Mickelson 2000; Dernie, Kaiser, and Warwick 2003). Less stable habitats such as coarse, clean sands in high energy zones are thought to recover more quickly than stable, muddy sediments after a disturbance, although empirical tests of this paradigm are lacking (Dernie, Kaiser, and Warwick 2003; Newell et al. 1998); recovery rates are mediated by a combination of physical (e.g., hydrodynamics), chemical (salinity, dissolved oxygen) and biological (larval transport, spawning success of brood stock) factors that differ in their relative importance in different habitats. Although decolonization of disturbed benthic habitats is generally rapid, Van Colen et al. (2010) note that divergence in benthic community development can occur late (2 years +) after a disturbance, which can cause patchiness in benthic communities.

that changes in salinity could lead to a shift from blue crab to lady crab communities in the affected areas (Cerrato and Frisk pers. comm. 2016). The breach also created an opportunity for blue mussel (*Mytilus edulis*) populations to develop in this area due to preference for high salinity and cooler temperatures. Blue mussels were common in the area of the Old Inlet during the early 1800s when the inlet was open (Cerrato, Locicero, and Goodbred 2013). After the formation of the breach in 2012, a dense community of blue mussels was observed but they failed to establish a long-term population, possibly due to predation or other factors (Cerrato and Frisk pers. comm. 2016). Changes in epibenthic communities may have also occurred in response to the breach formation. Peterson (2015a, 2015b) found low densities of shrimp in SAV beds near the breach in 2014. The low shrimp numbers were thought to be associated with high predation rates attributed to greater densities of foraging fish, which likely entered the area from marine waters through the newly formed breach.

Data Gaps

No pre- or post-breach benthic community data exists for the area in the vicinity of the breach, where water quality is most heavily influenced by the breach (Bellport Bay, Narrow Bay, western Moriches Bay). The short time that has elapsed since the formation of the breach limits our understanding of the dynamic long-term effects of the breach on benthic communities. Like most biological communities, benthic communities can be highly dynamic, making it difficult to distinguish between natural variation and changes that occur as part of a recovery or transition to a different type of community. Additional monitoring of the benthic community performed over the coming years would improve the post-breach monitoring dataset, resulting in a greater potential for identifying the long-term, post-breach trends in the benthic community. A detailed discussion of benthic community recovery is provided in Appendix A.

Summary of Changes since the Formation of the Breach

The breach formation resulted in conditions that favor high-flow, high-salinity adapted benthic communities comparable to those Cerrato (2001) observed in the vicinity of the Fire Island Inlet. Overwash and development of flood shoals (consisting predominantly of sand) have likely increased the grain size in the area. The areas near the breach, including southern Bellport Bay, Narrow Bay, and western Moriches Bay are experiencing increases in salinity and dissolved oxygen, and more moderate water column temperatures in summer and winter. These post-breach conditions favor the development of a marine benthic community, as opposed to the pre-breach conditions that favored an estuarine benthic community.

Hard Clams

Hard clam (*Mercanaria mercenaria*) populations in Great South Bay fluctuated throughout the 1900s, peaked in the 1960s and 1970s, and have since declined (Frisk et al. 2015; Cashin Technical Services, Inc. 2011). Poor environmental conditions and fisheries removals have been the primary drivers of population declines (Starke and LoBue 2016; Frisk et al. 2015; Cashin Technical Services, Inc. 2011; Bricelj 2009). Densities are currently at a historic low, with average adult density ranging from 0.22–2.50 clams per square meter (average 1.1 clam per square meter) in 2008–2009 (Table 5) (as reported in Cashin Technical Services, Inc. 2011, based on data from field surveys conducted by the Towns of Babylon, Islip, and Brookhaven, and The Nature Conservancy (TNC)). Depressed clam density has contributed to lower rates of successful spawning and reproduction (Starke and LoBue 2016, Cashin Technical Services, Inc. 2011). Because hard clams are broadcast spawners that require proximity to spawning partners for successful reproduction, such low densities of adult spawning stock greatly minimize the potential for successful reproduction and population recovery. Sharp declines in fisheries harvest have mirrored the decline of hard clam populations in Great South Bay since the 1970s, with hard clam landings decreasing from nearly 600,000 bushels in 1970 to less than 10,000 bushels in 2015 (reported in Gobler 2014; Barnes pers. comm. 2016a).

Table 5. Clam density (clams per square meter) and standing stock (in millions of clams) bay-wide 2008–2009, by area (from Cashin Technical Services, Inc. 2011, Table 1A).

Area	Total Clam Density	Seed Density	Adult Density	Standing Stock Seed	Standing Stock Adult	Standing Stock Total
Babylon	2.95	0.45	2.50	17.7 mil	98.0 mil	115.7 mil
Islip	0.77	0.20	0.57	15.2 mil	42.9 mil	58.1 mil
The Nature Conservancy	0.60	0.38	0.22	20.5 mil	11.9 mil	32.4 mil
Brookhaven	1.83	0.67	1.16	49.1 mil	84.3 mil	133.4 mil
Total Bay	1.40	0.42	0.98	102.4 mil	237.1 mil	339.6 mil

The loss of hard clams from Great South Bay has also meant a loss of the crucial ecosystem function of water filtration and water quality improvement that hard clams once provided through suspension feeding, their mechanism for obtaining food resources. Cashin Technical Services, Inc. (2011) estimated that filtration rates declined 65% for Brookhaven, 84% for Islip, and 40% for Babylon between 1978 and 2009. Given this, they estimated that in 2009, the existing hard clam populations in Brookhaven, Islip, and Babylon took more than 3 times, 6 times, and 1.6 times longer, respectively, to filter these areas than they did in 1978. The section that follows synthesizes data on hard clams from before and after the breach.

Synthesis of Hard Clam Information and Comparison of Pre- versus Post-Breach

The breach has increased the exchange of water and organisms with the open ocean and increased flushing rates in eastern Great South Bay. Flagg and others (2015) and Hinrichs (2016) demonstrated that new water circulation patterns in the bay are decreasing water residence time for eastern Great South Bay while increasing water residence time for central Great South Bay (refer to the "Physical Resources" section). The influx of ocean water and reduced residence time of water in the eastern portion of the bay has the potential to moderate winter and summer temperatures, alleviate the effects of brown tide blooms by exporting small form algae to the ocean, and improve food availability for hard clams by importing large form algae from the ocean (Gobler pers. comm. 2016). Evidence for both positive and negative impacts of these changes in environmental conditions on measures of hard clam success is summarized below. Hard clam success is determined using various metrics, including increased tissue growth rates, increased shell growth rates, higher lipid content, greater condition index (CI), greater specimen densities, and decreased juvenile mortality rates.

Several factors that affect hard clam growth and reproduction have been affected by the breach. Chief among these is the availability of high quality food resources (i.e., large cell phytoplankton ≥ 5 µm) and potentially water temperature (optimal range for growth between 20 and 23°C; Stanley 1983). Sufficient food resources are essential for clam growth and reproduction. Severe food limitation can be caused by brown tide algal blooms which drive down the clam CI, a commonly used index of clam health and spawning potential (Starke and LoBue 2016). Brown tides are dominated by the small form algae *Aureococcus anophagefferens*, which is a poor food source for suspension feeding bivalves like hard clams. Algal blooms are driven by high nutrient concentrations in the bay and are compounded by high residence time of water in the bay. Two blooms in the same year can be devastating for hard clams, driving the CI so low that more than one spawning season can be affected (Starke and LoBue 2016). Although the breach may import higher quality phytoplankton into the bay, food limitation is possible right at the breach where chlorophyll-*a* levels are less than 5 µg/L (Gobler pers. comm. 2015). Increased exchange of water through the breach may also have decreased summer water temperatures in Bellport Bay, Narrow Bay, and western Moriches Bay (Gobler 2014) which has the potential to moderate summer and winter temperatures (refer to the "Physical Resources" section). In areas where the temperature reaches above or below the optimal range for hard clams, the impact on hard clams would be negative. However, it should be noted that data analyses conducted by SoMAS indicate that the impact of the breach on water temperature is inconclusive (Flagg pers. comm. 2015; refer to the "Physical Resources" section).

The important role of food limitation and temperature for hard clam growth was demonstrated in a pre-breach study in Great South Bay (USACE 2004c). For this study, four cages were deployed in 2001 and six cages were deployed in 2003 at three stations located in Moriches Inlet, Forge River and Smith Point (Figure 51). Forge River, where clam growth was the highest in both years, also had the highest concentration of total chlorophyll in both years (5.9 µg/L in 2001 and 28.59 µg/L in 2003), likely a result of increased nutrient inputs from nearby sources, including historical duck farms in that location (Figures 52 through 55). At Moriches Inlet, growth rates were significantly lower, coinciding with the lowest chlorophyll measurement reported (1.5 µg/L in 2001 and 1.06 µg/L in 2003). Growth of shell peaked from June–August, whereas tissue growth rate peaked somewhat later in the August–October timeframe. The relatively lower growth rates at Smith Point were attributed to temperatures spiking to 27°C during the summer which above the optimal range for growth. These results provided strong field-based evidence for the effect of food availability and temperature on hard clam growth rates in Great South Bay.

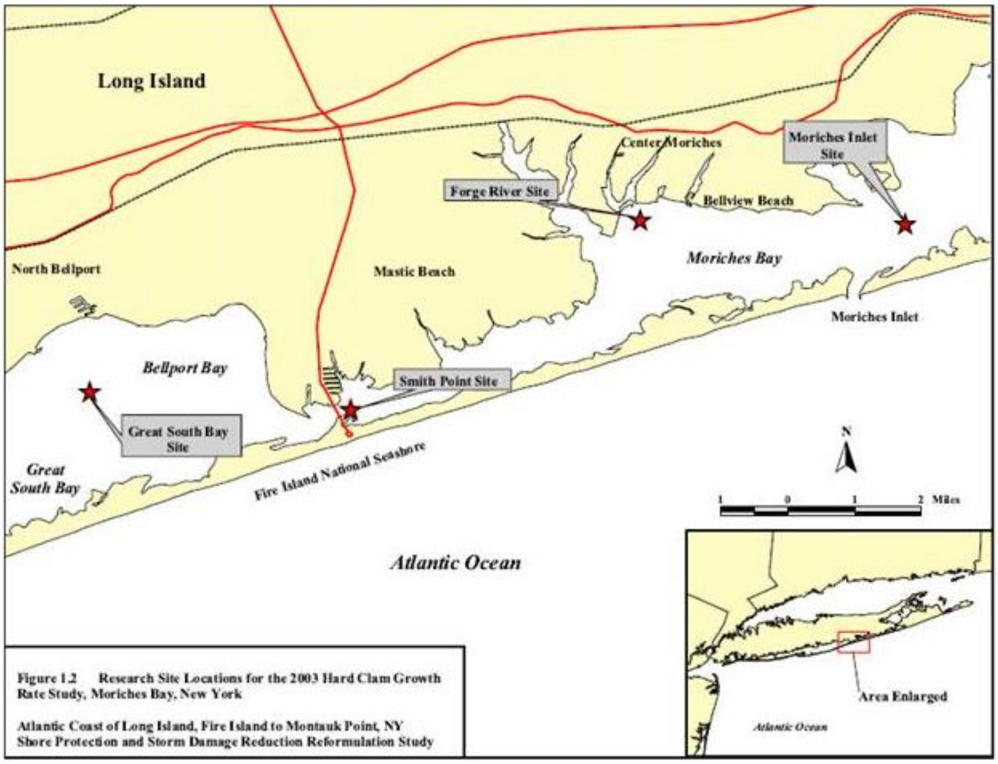

Figure 51. Location of sites of caging study performed by the US Army Corps of Engineers in 2004 (from USACE 2004c, Figure 1.2).

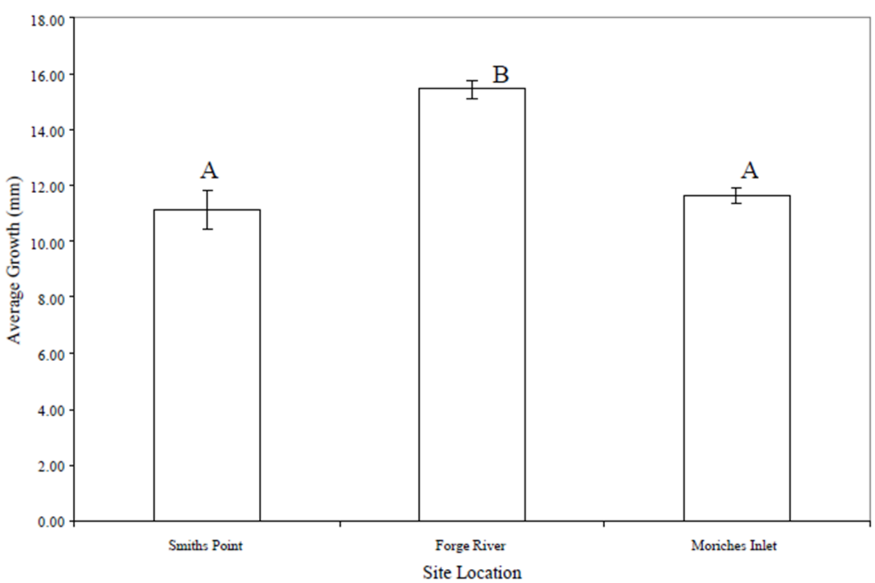

Figure 52. Average shell length growth of clams from the experimental sites in Moriches Bay, June 15 through October 4, 2001. Unlike letters indicate sites that had significant differences in Tukey-Kramer *post hoc* comparisons. Each site had an n = 4 and error bars represent ± one standard error (Figure 4.1 from USACE 2004c).

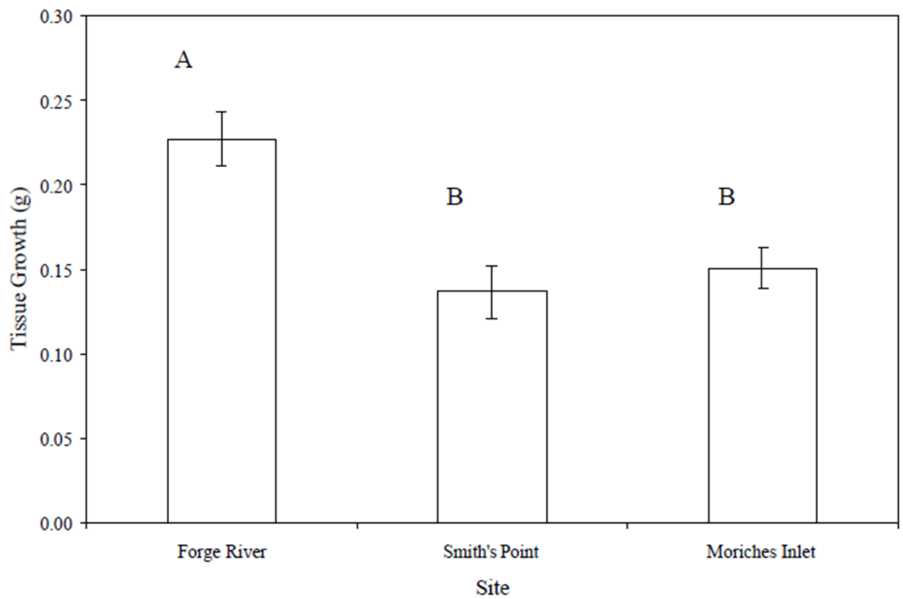

Figure 53. Average dry weight tissue grown of clams from the experimental sites in Moriches Bay, June 15 through October 4, 2001. Unlike letters indicate sites that had significant differences in Tukey-Kramer *post hoc* comparisons. Each site had an n = 4 and error bars represent ± one standard error (Figure 4.3 from USACE 2004c).

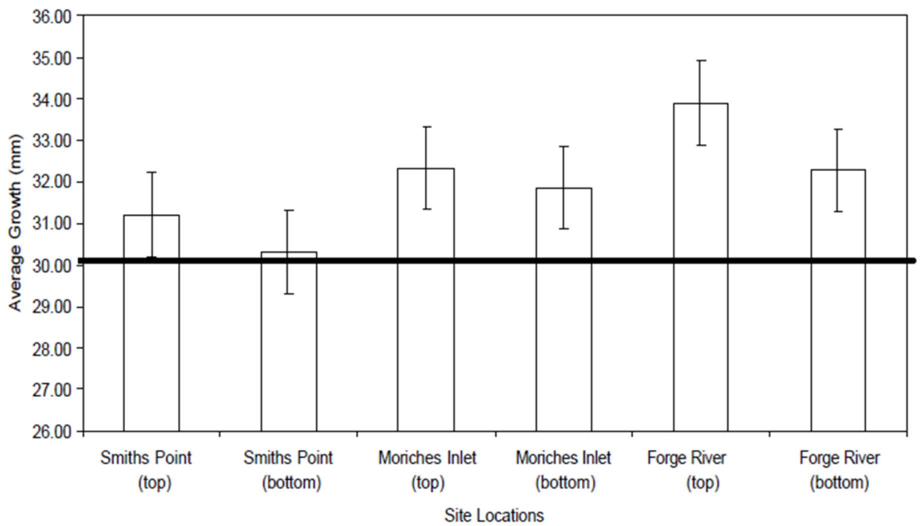

Figure 54. Average shell length growth of clams from the experimental sites in Moriches Bay, June 18, through October 20, 2003. Each site had an n = 4 and error bars represent ± one standard error (Figure 4.5 from USACE 2004c).

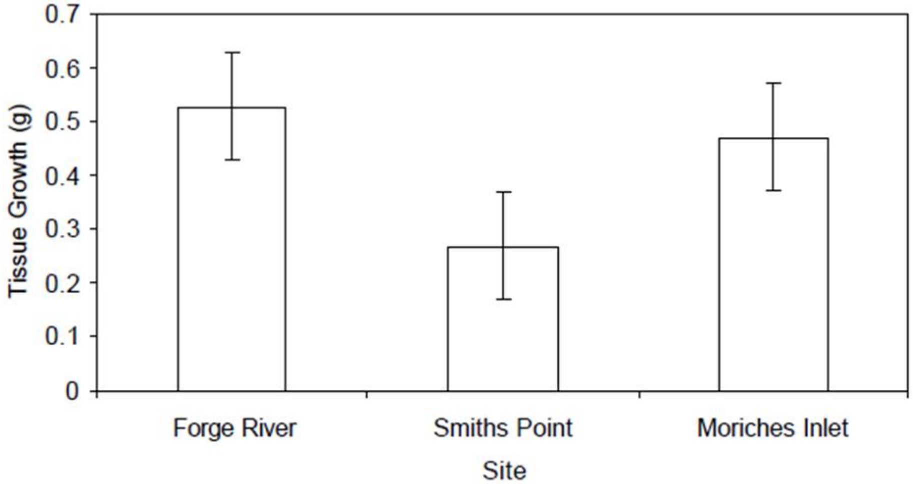

Figure 55. Average clam dry weight for three sites at Moriches Bay, June 18, August 21, and October 20, 2003 (Figure 4.7 from USACE 2004c).

Predation can exert a strong top down control on clam populations (Virstein 1997; Kraeuter and Castagna 1980). Predation on invertebrates can increase near inlets where environmental conditions allow for marine predators as well as high salinity tolerant estuarine predators to occur (USACE 2006b; EEA Inc. 2002). High juvenile predation by crabs has been observed near Fire Island Inlet (Cashin Technical Services, Inc. 2011). Predation by ctenophores and other grazers on clam larvae can also have a negative impact on the clam population in Great South Bay (Cerrato pers. comm. 2016; McNamara, Lonsdale, and Cerrato 2010). Although blue crabs are a potential predator on hard clams, the impact of blue crab predation is not well understood due to limited field data on blue crabs (Bricelj 2009). Post-breach blue crab densities appear to have decreased, however lady crab populations, another potential predator on hard clams, have increased significantly (Frisk et al. 2015). Given that the breach has created a new gateway through which ocean predators can enter Great South Bay, increased predation on hard clams may be expected within areas of Great South Bay that are affected by the marine influence.

Salinity may also play a role in hard clam distribution patterns (Baker et al. 2005). The optimal salinity for growth is 24–28 psu (Chanley 1958) although clams can be found in salinities ranging from 10–35 ppt. Salinity appears to be less of a factor for growth rates. Increased salinity in Great South Bay caused by the influx of ocean water through the breach could have adverse effects on hard clam populations if the range of optimal salinity for survival is exceeded (Barnes pers. comm. 2016b) although there are no recorded incidences of this. Additionally, high salinity water favors the growth of QPX (Quahog Parasite Unknown), a hard clam parasite that could have negative effects on the hard clam population (Perrigault et al. 2012). Taken together, this information indicates that the change in salinity as a result of the breach has the potential to create unfavorable conditions for hard clams.

Prior to the breach, researchers examined the hypothesis that exchange of water with the open ocean would increase the success of hard clams in the Long Island South Shore Estuary by improving the availability of high quality food resources (Gobler 2014; Weiss et al. 2007). Cage studies were performed pre-breach in 2005 and 2014 under post-breach conditions by deploying and monitoring cages of locally sourced clams at three stations in Shinnecock Bay and at three stations in Great South Bay (Figure 56). From April to November, water quality, physical attributes, and biological attributes were sampled. Central Great South Bay sites had significantly faster tissue and shell growth rates, lower mortality rates, higher lipid content, greater CI, and greater densities compared to sites near an inlet (Figures 57 and 58). Lower measures of success observed adjacent to the inlets were attributed to cooler temperatures, limited phytoplankton production, and suspended organic matter as food sources. In central Great South Bay locations, clams were more successful due to optimal temperatures and the presence of abundant high quality food, including cells greater than 5 µm. This is consistent with the findings of Newell et al. (2009) who found that reduced reproductive output appeared to be associated with abundant small phytoplankton cells which clams are unable to use as food. Gobler (2014) and Weiss et al. (2007) additionally found that prior to the breach, clams at the Bellport Bay site had particularly low rates of tissue growth compared to other sites, which was attributed to the presence of dinoflagellates and higher than optimal summer temperatures.

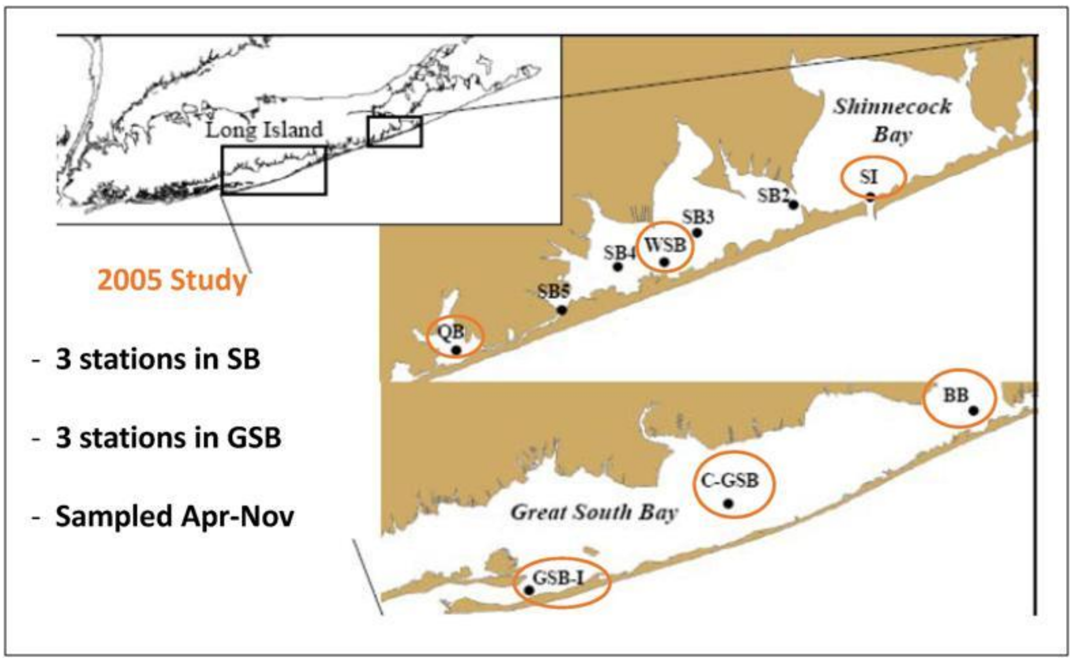

Figure 56. Location of study sites (from Gobler 2014). BB=Bellport Bay, C-GSB=Central Great South Bay, GSB=Great South Bay, SB=Shinnecock Bay, SI=Shinnecock Inlet, WSB=Western Great South Bay.

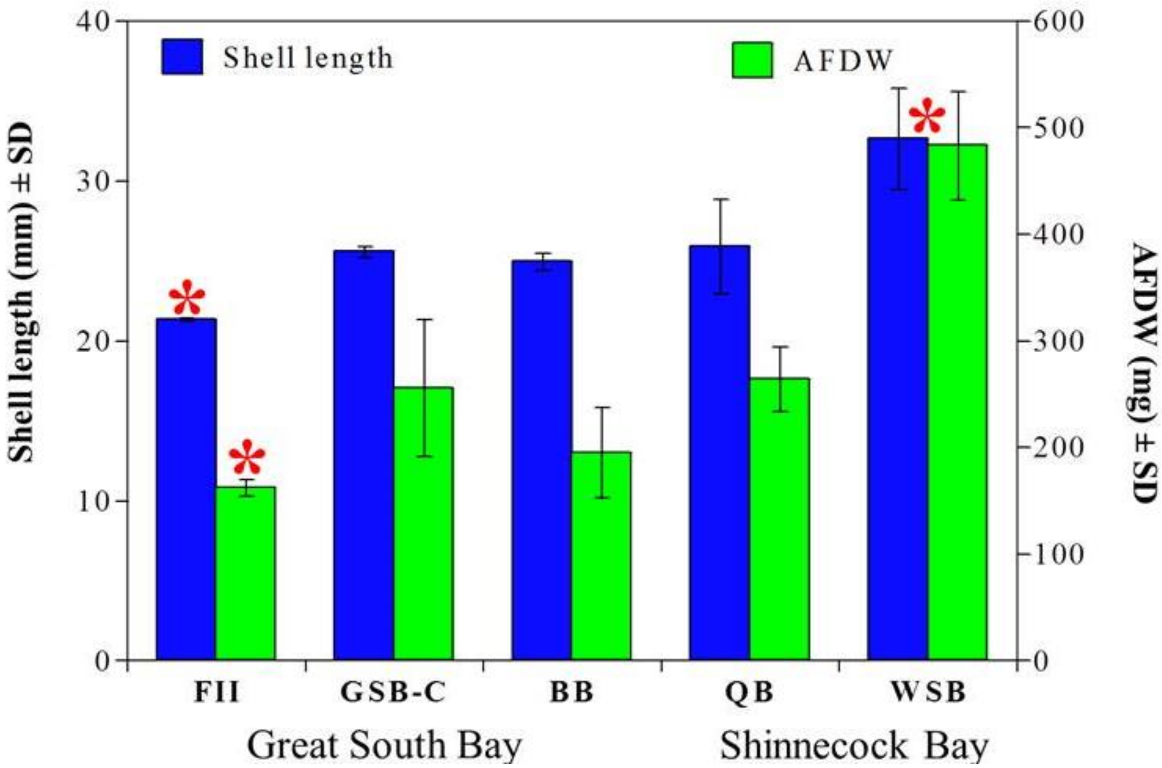

Figure 57. Shell length and biomass (ash-free dry weight (AFDW)) at each site (from Gobler 2014). West Shinnecock Bay (SB) had significantly longer shell lengths than Quantak Bay (QB), Fire Island Inlet (FII), Central Great South Bay (GSB-C), and Bellport Bay (BB). All clams in non-inlet locations were significantly longer than clams in Fire Island Inlet.

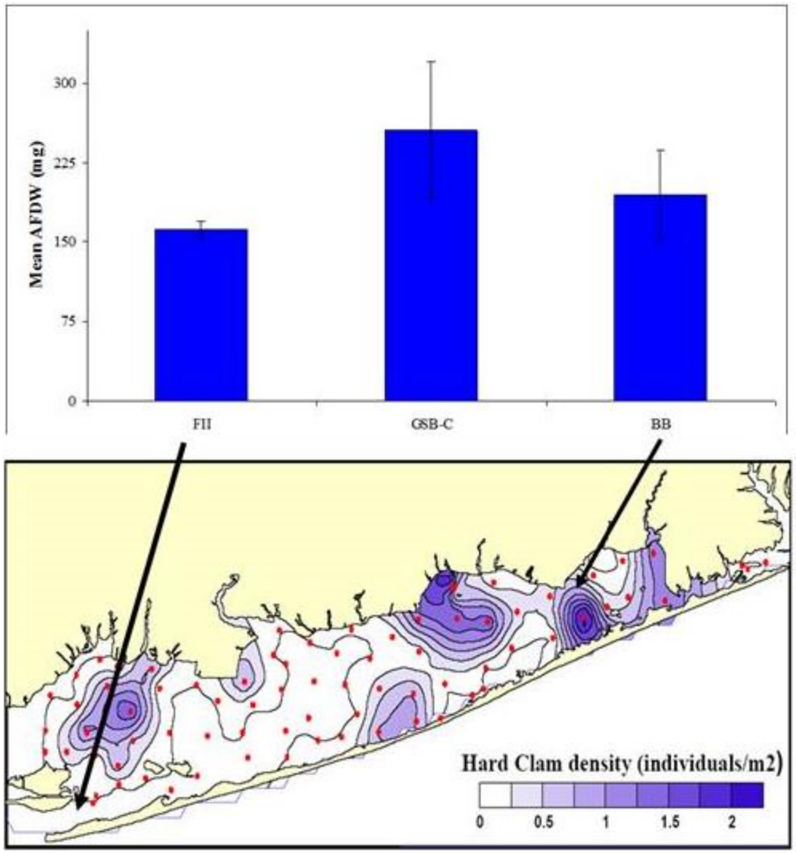

Figure 58. Mean ash-free dry weight (AFDW) at each site (from Gobler 2014). Red dots indicate randomly selected sampling stations. BB=Bellport Bay, FII-Fire Island Inlet, and GSB-C=Central Great South Bay.

Since the breach formed, measures of clam success have greatly improved for Bellport Bay. Gobler (2014) repeated his hard clam caging experiment during 2014 after the breach formed. Sites were studied from April to November and were a subset of those from the 2005 study located in Fire Island Inlet, central Great South Bay, and Bellport Bay (Figure 56). Results indicated that, in contrast to the 2005 results, growth rates were greatest for juveniles in Bellport Bay, Narrow Bay, and at Fire Island Inlet and lowest in central Great South Bay (Figure 59). However, direct comparisons between the 2005 and 2014 studies were hindered due to a high temperature anomaly and a brown tide occurrence in 2014. Nevertheless, the study highlights an emerging geographic pattern: clam growth rates have improved in Bellport Bay from 2005 to 2014 but have worsened in central Great South Bay in that same period. This suggests that conditions in Bellport Bay have improved for hard clams since the breach formed, while conditions in areas located west of the breach in central Great South Bay have continued to decline along with hard clam populations.

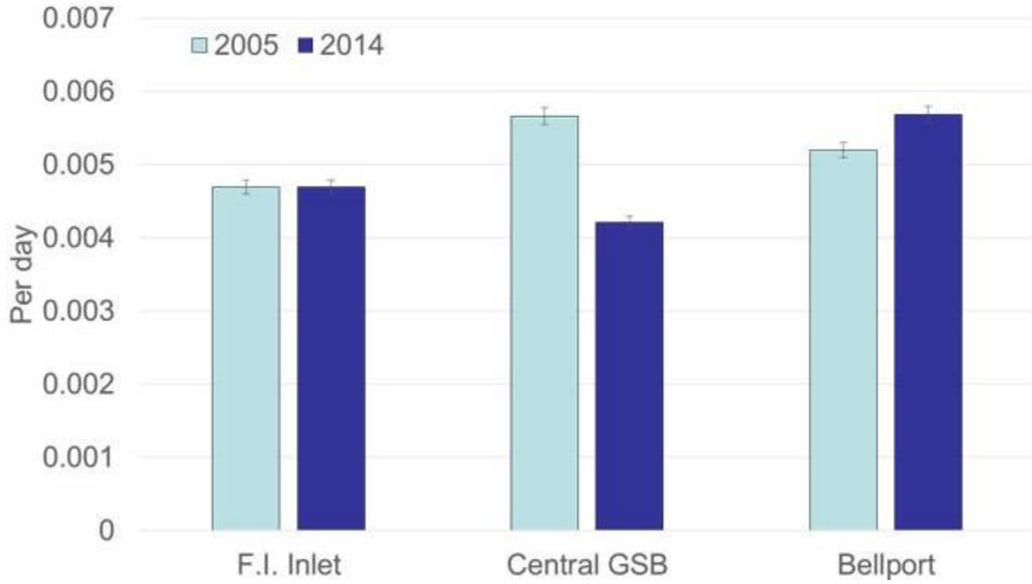

Figure 59. Instantaneous growth rates of hard clams, 2005 versus 2014, most similar cohorts (from Gobler 2014).

Improved measures of clam success in Bellport Bay after the breach have also been noted by Starke and LoBue (2016). Clams collected before and after the breach in Bellport Bay show marked differences in growth rates (Greene 1978; Starke and LoBue 2016). Whereas prior to Hurricane Sandy, clams in this location had slow growth rates and chalky white shells due to the acidic sediment in which they lived. Clams collected in Bellport Bay after Hurricane Sandy have secreted large and healthier-appearing growth rings in their shells, indicating improved growth rate and sediment habitat condition (Figure 60). Starke and LoBue (2016) also noted improvements in CI values for clams in Bellport Bay. Researchers sampled 20 individuals of naturally occurring clams from the same locations on the same date. The results show a striking improvement in CI from before the breach in 2009 compared to after the breach in 2015 at Bellport Bay, whereas CI for central Great South Bay was relatively unchanged over the same time period (Figure 61) (Starke and LoBue 2016). These improvements in clam growth and CI are attributed to improvements in water quality, as increased rates of flushing are able to locally suppress blooms of brown tide algae and improve food quality in Bellport Bay (Starke and LoBue 2016).

Figure 60. Hard clams collected in Bellport Bay after Hurricane Sandy (from Starke and LoBue 2016).

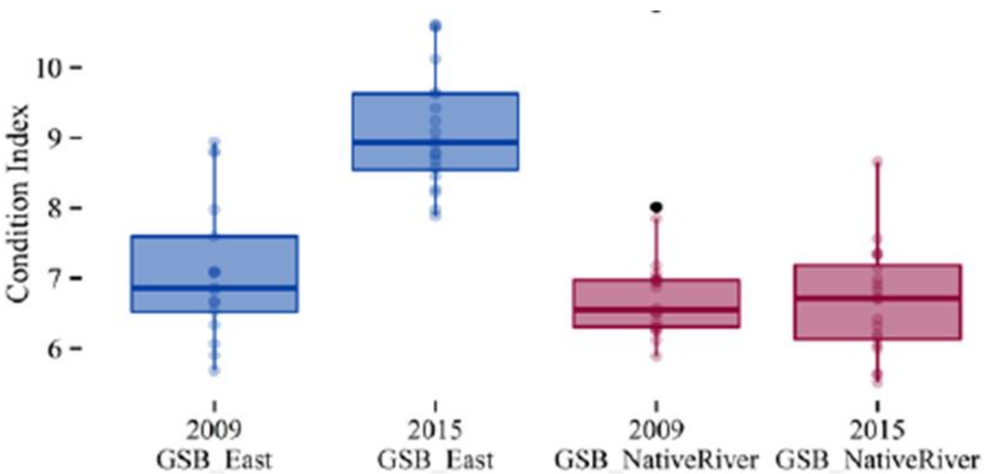

Figure 61. Boxplots displaying mid-November condition index measures at eastern Great South Bay / Great South Bay east and central Great South Bay / Great South Bay native river sites (from Stark and LoBue 2016).

Despite improved measures of clam success, there has been no reported change in the size of the hard clam population in Great South Bay since the breach formed. Landings data from before and after the breach formed indicate no major change in the number of clams harvested from Great South Bay (Barnes pers. comm. 2016a). However, given that hard clams require at least 4 years to attain harvestable size after settlement, any recovery in hard clam populations brought about by the breach would not yet be reflected by harvest statistics. There are no fisheries-independent bay-wide surveys of clam population size in Great South Bay; therefore, the response of the hard clam population standing stock to the change in environmental conditions resulting from the breach remains unknown. Although environmental conditions that favor hard clam success have occurred since the breach, it is not well understood whether these improvements will be able to overcome the low spawning and reproductive success that has resulted from extremely low clam densities throughout the bay.

Considerable effort has been focused on restoring hard clams to Great South Bay. TNC has created "spawner sanctuaries" since 2004 by stocking local stock from Long Island Sound and other nearby estuaries (TNC 2013). Given what is known about the distribution and size classes of clams in Great South Bay, and what is known about the physical forcing factors that influence the distribution of juvenile hard clams such as wind, currents, and dispersal, evidence suggested that stocking large numbers of adult clams in the spawner sanctuaries could increase recruitment in Great South Bay (TNC 2013). In spawner sanctuaries, clams are stocked at high density (>10 per square meter) on natural bottom in areas protected from harvest (GSBHCWG 2011). There are 108 restocking sites covering 81 acres of TNC property, and in 2012, 5 sites were added west of TNC stocking areas in the Town of Islip (Figure 62). Surveys of naturally occurring hard clam populations in Brookhaven, Islip, and Babylon (2004–2012) suggest that 2011 abundance of juvenile clams in Great South Bay had returned to near pre-project levels. Declines were attributed to an extensive brown tide that affected the area from 2007 through 2009 (TNC 2013). However, a strong 2011 cohort sampled in 2012 provided some optimism for a rebound (TNC 2013) (Figure 63). Strong year-classes of juvenile clams can move into larger size classes as time progresses and eventually into adulthood. The map in Figure 63 shows that summer abundance of hard clams in central Great South Bay was high in 2008 and declined following the brown tide event. A resurgence of juveniles appeared in 2012 prior to Hurricane Sandy and the occurrence of the breach, however no post-breach surveys have been performed to reassess trends in this cohort.

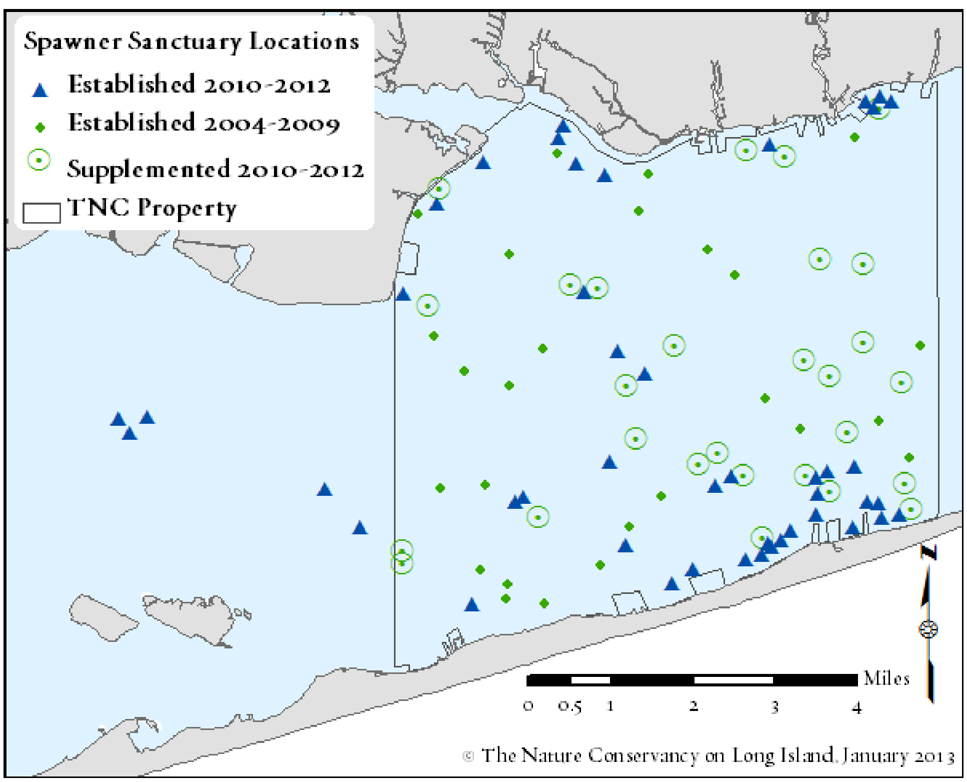

Figure 62. Approximate locations of The Nature Conservancy's previously established, new, and recently supplemented spawner sanctuaries as of October 2013 (from TNC 2013).

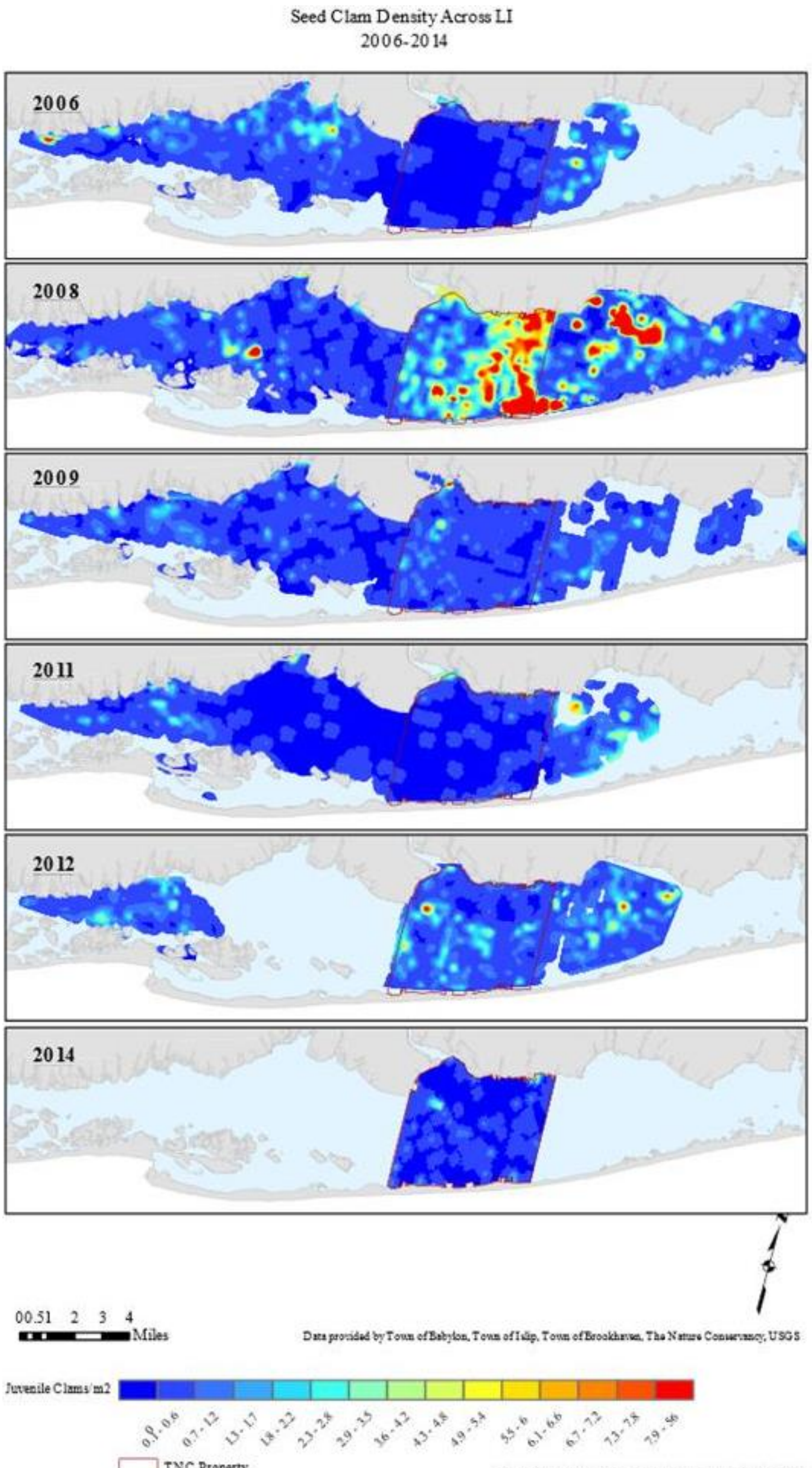

Figure 63. Abundance and distribution of juvenile clams (<2.54 centimeters [1 inch] shell length) in Great South Bay in 2006, 2008, 2009, 2011, and 2012 (from TNC 2013).

Post-breach changes in water quality have not affected the classification of seasonally certified[3] or uncertified shellfish lands. NYSDEC conducts intensive sampling of fecal coliform levels in seasonally certified shellfish lands in Great South Bay during its triennial evaluations to ensure that they meet the sanitary criteria for certification of shellfish lands during the period when they are certified. The most recent triennial report includes post-breach survey data and indicated that certified and seasonally certified shellfish lands are correctly classified as such; therefore no changes in classification are necessary at this time (NYSDEC 2015a). There may, however, be potential for changing the status in the southeastern area near Narrow Bay (currently closed year round uncertified) in response to post-breach declines in fecal coliform concentrations (Barnes pers. comm. 2016b). There will be no decision on a potential status change until 2018, when the next triennial evaluation will be conducted. Note that the NYSDEC surveys are only conducted during the open period for the shellfish lands and not during seasonal closures; Therefore, these surveys are not designed to collect data during periods when shellfish lands are closed to fishing and cannot determine whether waters may have become healthy for clams since the breach during periods of the year that are closed to harvest (Barnes pers. comm. 2016b). Also, NYSDEC (2015a) reported that the Sampling Station at Old Inlet (adjacent to the breach) was deemed inactive because it was unnavigable at the time the report was written.

Data Gaps

Limited fishery-independent data are available to describe the bay-wide population status of hard clams. Surveys for this species require specialized equipment, significant time and manpower, and financial resources. The lack of such data has prevented a spatially synoptic, temporally resolved understanding of hard clam population dynamics in Great South Bay. The short time period that has passed since the formation of the breach limits our ability to draw conclusions regarding the dynamic long-term effects of the breach on hard clams.

Larval distribution patterns are affected by hydrodynamics. Given that the hydrodynamics of the bay have changed since the formation of the breach (refer to the “Physical Resources” section), hard clam dispersal patterns may have also changed; however, there are no data available to address this. Understanding the dispersal patterns of hard clam larvae is essential for understanding the population dynamics for this species. A paucity of wind data has prevented a robust integration of hydrodynamic models with larval dispersal models. Wind influences current patterns, which in turn have a direct effect on the distribution of hard clam larval settlers. Larval clams transition through multiple phases prior to settling on the ocean floor. During the veliger larval phase, larvae are carried by currents throughout the water body until the clam metamorphoses to the pediveliger (pre-settlement) and eventually juvenile (settlement) stage. This veliger phase can last 6–14 days (Loosanoff and Davis 1949) and has the potential to carry larvae over large distances.

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[3] Seasonally certified and uncertified are designations assigned to shellfish lands by the NYSDEC. Hard clams may be taken only from areas designated as certified (or open) for the harvest of hard clams.

Summary of Changes since the Formation of the Breach

The formation of the wilderness breach has had positive and negative effects on hard clams depending on the region of the bay where they are located. In Bellport Bay, Narrow Bay, and Western Moriches, water quality has improved. In these areas, the export of water to the open ocean has ameliorated the effects of brown tide, moderated summer temperatures, and improved the quality and quantity of food resources for hard clams. These changes in Bellport Bay since the formation of the breach have coincided with improvements in indicators of clam success, namely growth rate and CI. In Central Great South Bay, circulation patterns have changed since the breach and may now create conditions that further favor brown tide blooms and negatively affect clams. In the immediate location of the breach, current data suggest that food resources are less abundant and of lower quality, and that predation is greater.

Finfish and Decapod Crustaceans

Great South Bay is a shallow, well-mixed lagoonal ecosystem that supports numerous finfish and decapod crustacean species (Briggs and O'Conner 1971). Changes in the abundance and distribution of salt water species in Great South Bay have occurred since the breach formed, particularly in the areas of the bay affected by the influx of ocean water. These changes are evident from comparisons made between faunal surveys conducted in the decade prior to the breach and surveys conducted after the breach formed. This section provides a synthesis of available pre- and post-breach data on the finfish and decapod crustacean communities in Great South Bay.

Synthesis of Finfish and Decapod Crustacean Information and Comparison of Pre- versus Post-Breach

Great South Bay has experienced an increase in biodiversity (i.e., number of species) and in marine species since the breach formed. These ecological changes are suspected in namely Bellport Bay, Narrow Bay, and western Moriches Bay (Gobler, Collier, and Lonsdale 2014) where water temperature has reportedly decreased and salinity has increased due to an influx of ocean water (Figure 64). (It should be noted that data analyses conducted by SoMAS indicate that the impact of the breach on water temperature is inconclusive (Flagg pers. comm. 2015; refer to the "Physical Resources" section).

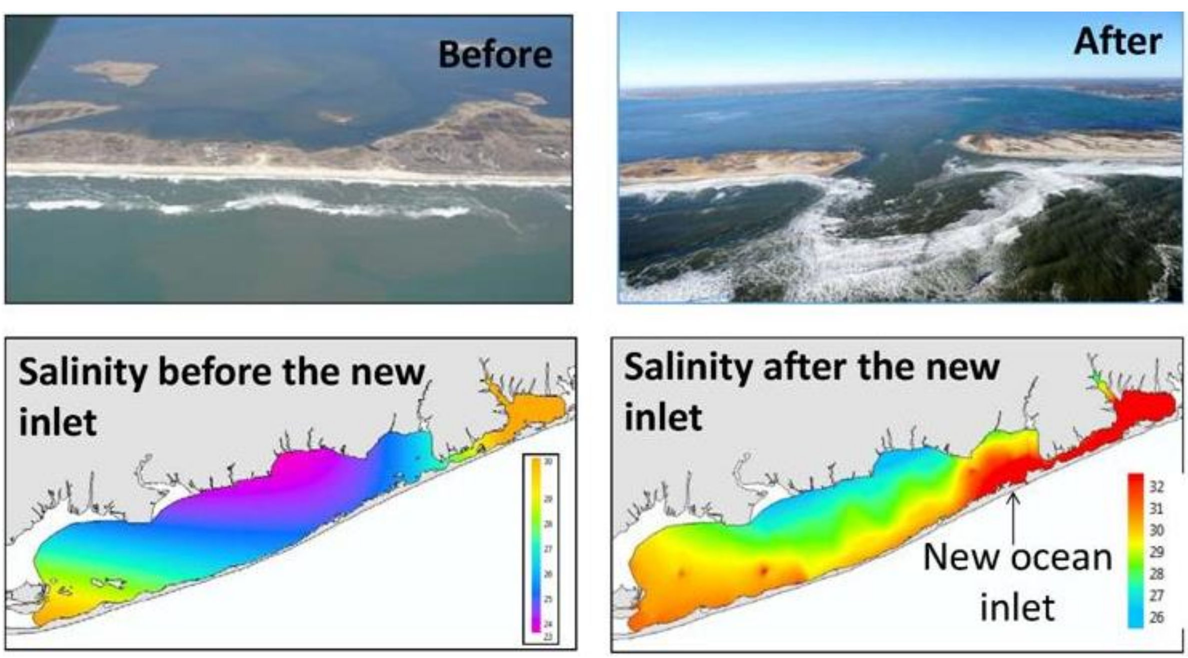

Figure 64. Map of salinity before the breach (left panel) and after the breach (right panel) (from Gobler, Collier, and Lonsdale 2014).

Frisk et al. (2015) compared 2007 (pre-breach) with 2013–2015 (post-breach) fish trawling data from 45 random stations in Great South Bay (Figure 65). Overall, there was an increase in species richness from 35 to 60 species and a change in species composition after the breach formed (Figure 66). The study also documented an 80% decline in blue crab populations, an estuarine species, and a 500%

increase in lady crab populations, a species adapted to marine environmental conditions (Figure 67). Similarly, squid catch per unit effort increased more than 300% after the breach, and butterfish populations increased more than 100%. Bay anchovy (*Anchoa mitchilli*) also increased in relative abundance (percentage of total catch per unit effort over time) from 18% in 2007 to 79% (2013), to 62% (2014), and to 72% in 2015. Although these studies are informative, it is not known whether these changes in catch size are associated with the breach.

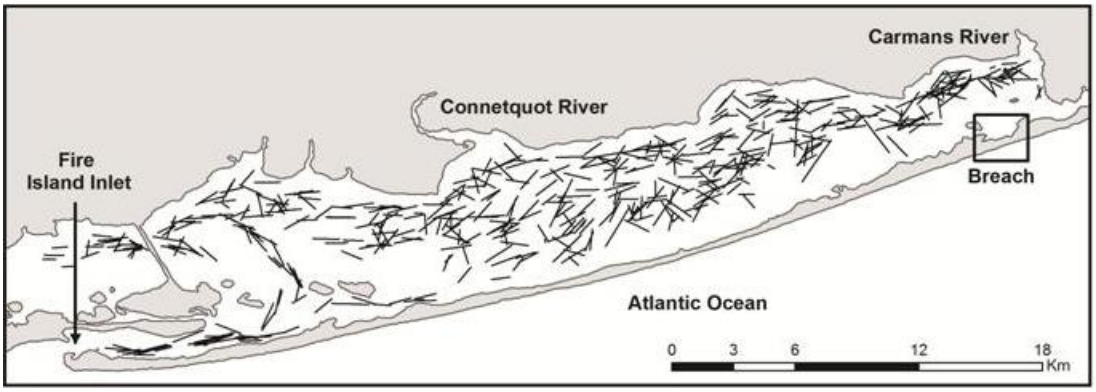

Figure 65. Location of trawl samples. Forty-five random samples were collected before and after the breach (from Frisk et al. 2015).

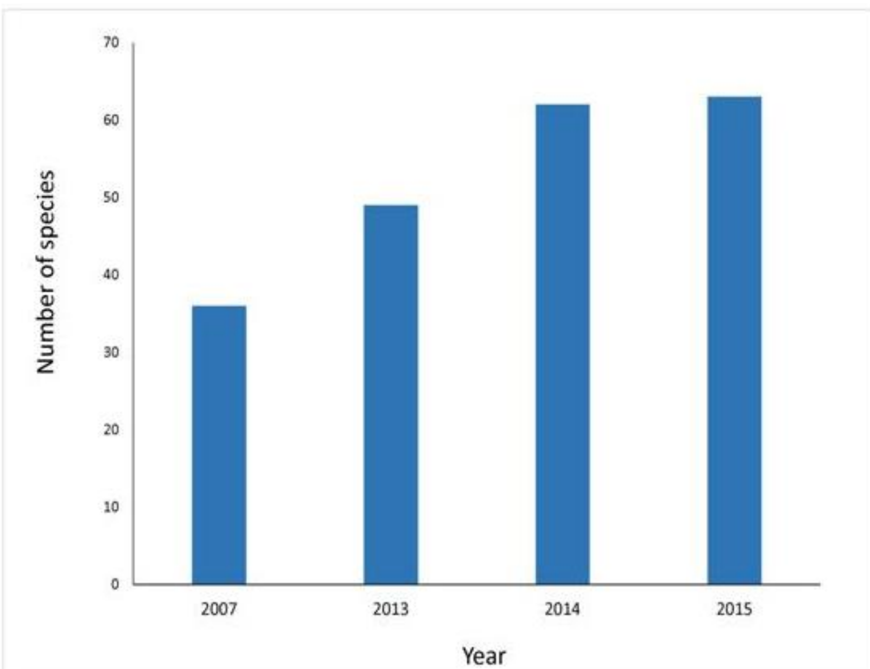

Figure 66. Number of species per year collected in a Great South Bay trawl survey (from Frisk et al. 2015).

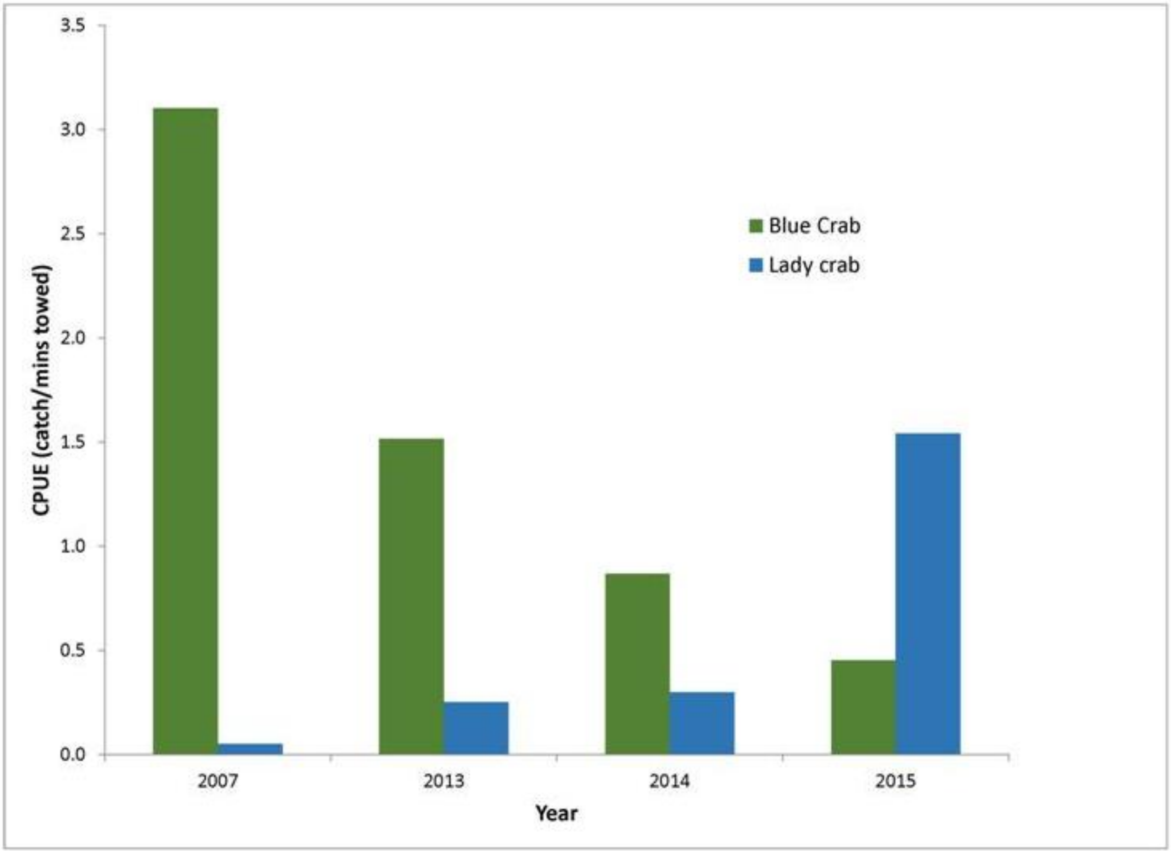

Figure 67. Change in catch per unit effort over time for blue crab and lady crab collected in Great South Bay trawl survey (from Frisk et al. 2015).

Finfish abundance prior to the breach was recorded at sites in Great South Bay, Moriches Bay, and Shinnecock Bay (Figures 68 and 69). These faunal surveys indicated that both finfish abundance and diversity were lowest in Great South Bay compared to Moriches Bay and Shinnecock Bay (USACE 2004b, 2006b; EEA Inc. 2002). USACE (2004b, 2006b) performed surveys of fish and invertebrate populations at 6 sites (selected specifically to assess fish populations in SAV beds) distributed evenly among Great South Bay, Moriches Bay, and Shinnecock Bay in 2003 and again in 2005. The study collected water quality data, characterized SAV habitat, and collected bimonthly samples from June to October 2003, and monthly samples from June to November 2005. Similar seining methods were used in each year, and the 2003 survey additionally employed visual observations by a snorkeler to identify any species that were not collected in the seine net. Results from the survey showed similar faunal distribution patterns in both years of the study, with both finfish abundance and species richness being lowest in Great South Bay, and increasing from west to east. The largest values reported at two sites near Shinnecock Inlet (Figures 70 through 73). The low finfish abundance and species richness observed in Great South Bay was attributed by the authors to greater habitat degradation in that location. In the 2003 samples, the numerically dominant fish species included fourspine stickleback (*Apeltes quadracus*) (32%), Atlantic silverside (*Menidia menidia*) (16%), and blackfish (*Tautoga onitis*) (15%). In 2005, Atlantic silversides (*M. menidia*) (26%), bay anchovy (*A. mitchilli*) (16.5%), and Atlantic tomcod (*Microgadus tomcod*) (13.9%) dominated the catch. Seasonal trends for abundance and species richness followed expected patterns for both years, with lower values in the early spring and a peak in the late summer early fall. This reflected an influx of fish into the bay as rising temperatures warm bay waters.

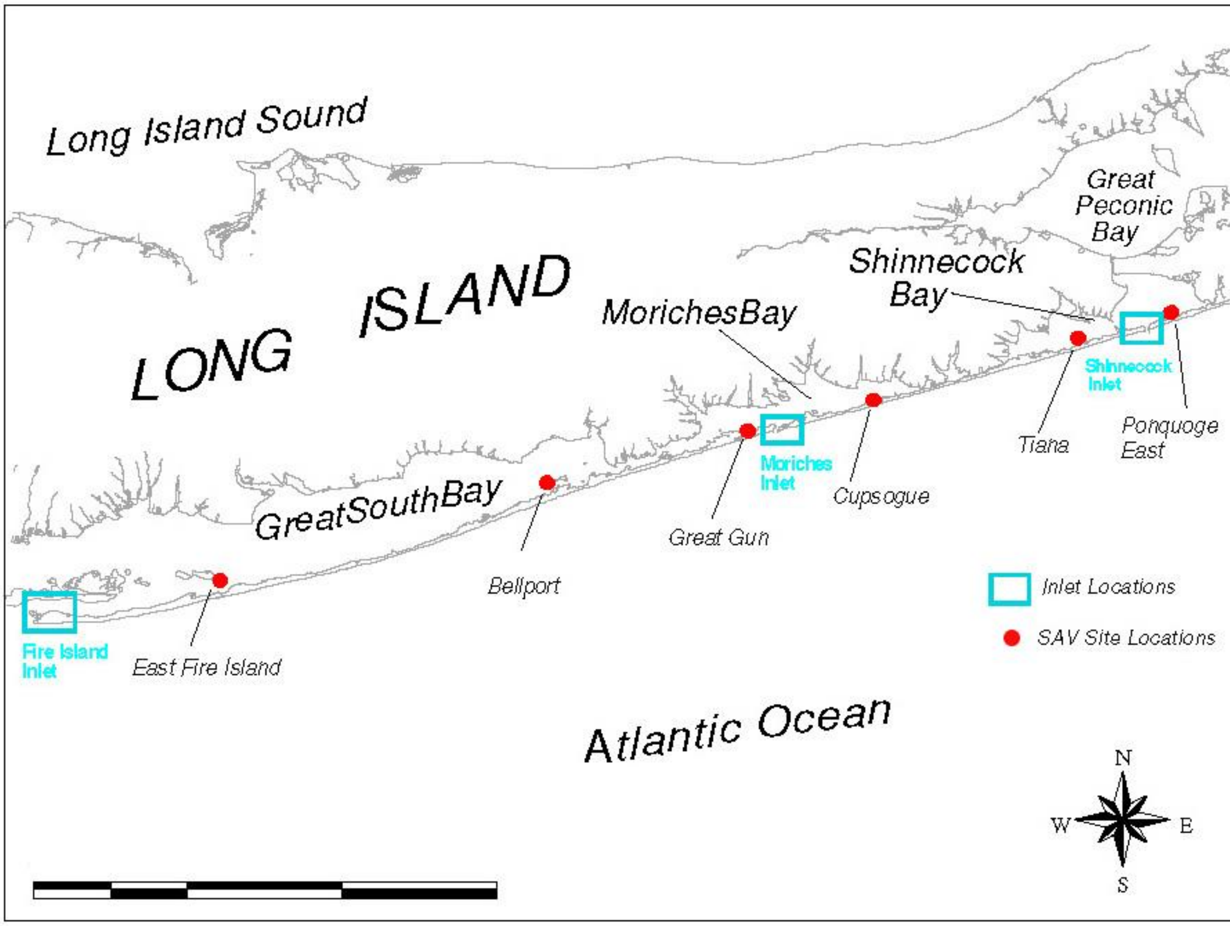

Figure 68. Site sampled during fish and invertebrate seine survey in 2003 (from USACE 2004b).

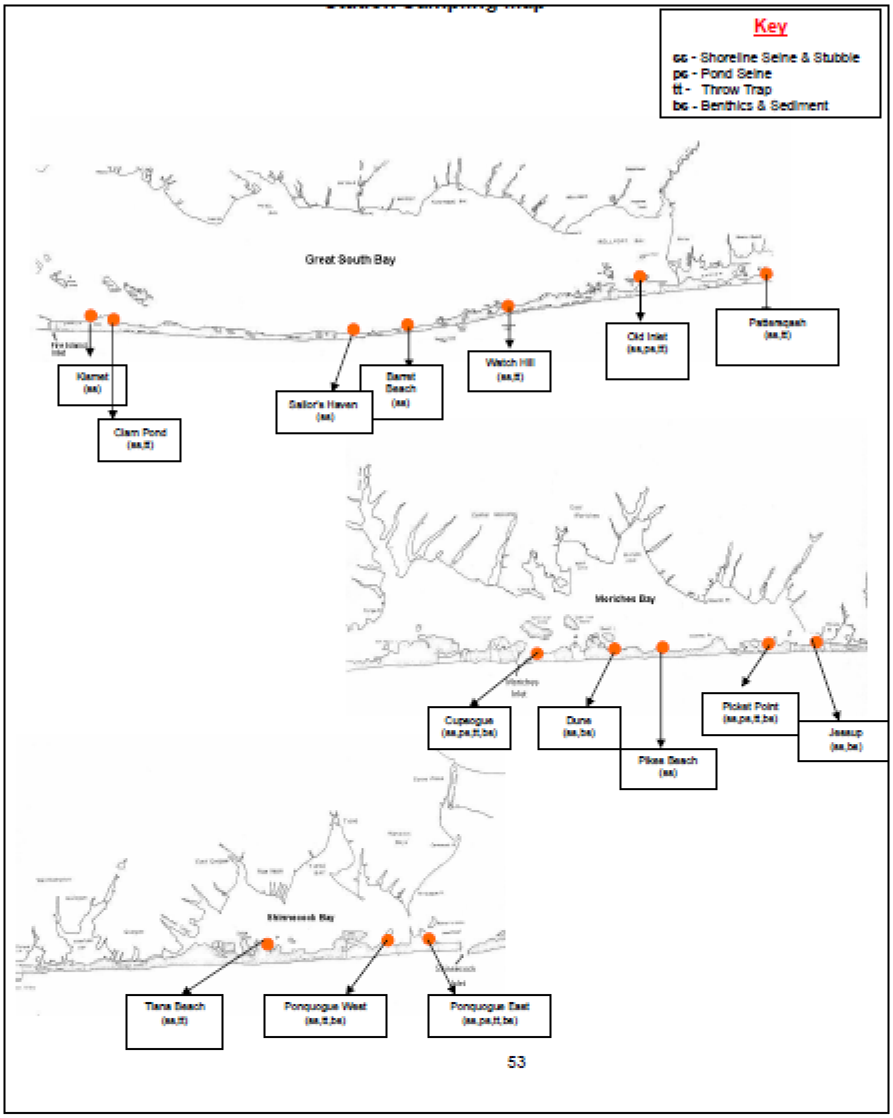

Figure 69. Sampling locations for seine and throw trap survey conducted in 2000–2001 (from EEA Inc. 2002).

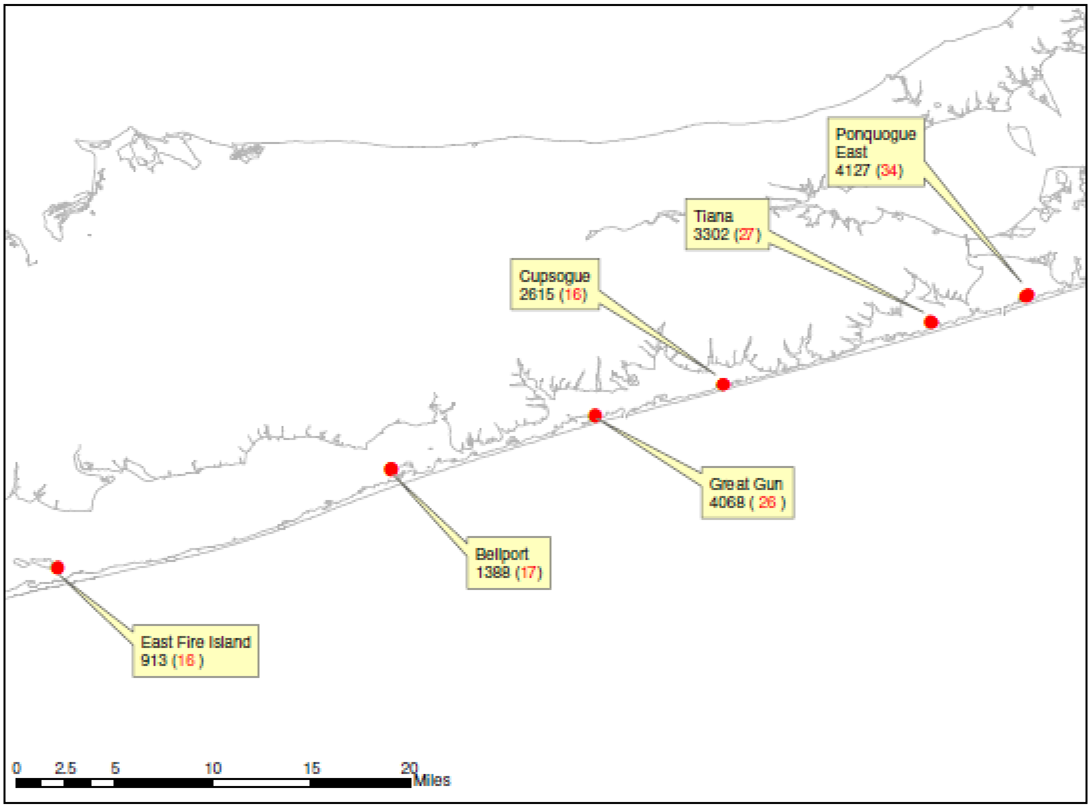

Figure 70. Finfish abundance (black) and species richness (red) per site in seine samples taken in submerged aquatic vegetation beds in 2003 (from USACE 2004b).

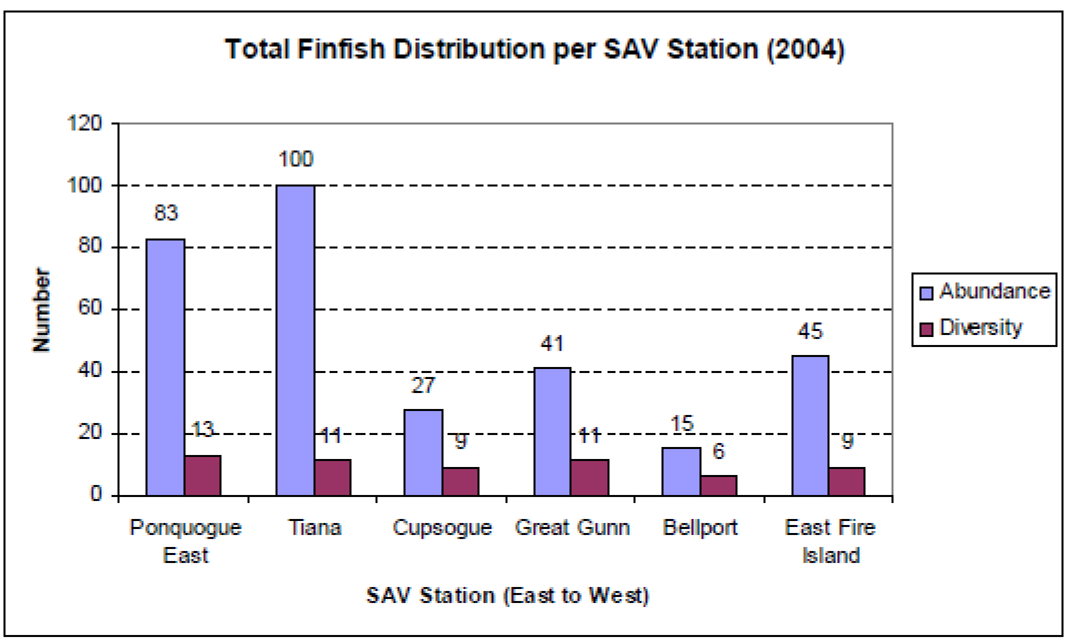

Figure 71. Finfish abundance and species richness per site in seine samples collected in 2004 (from USACE 2006b).

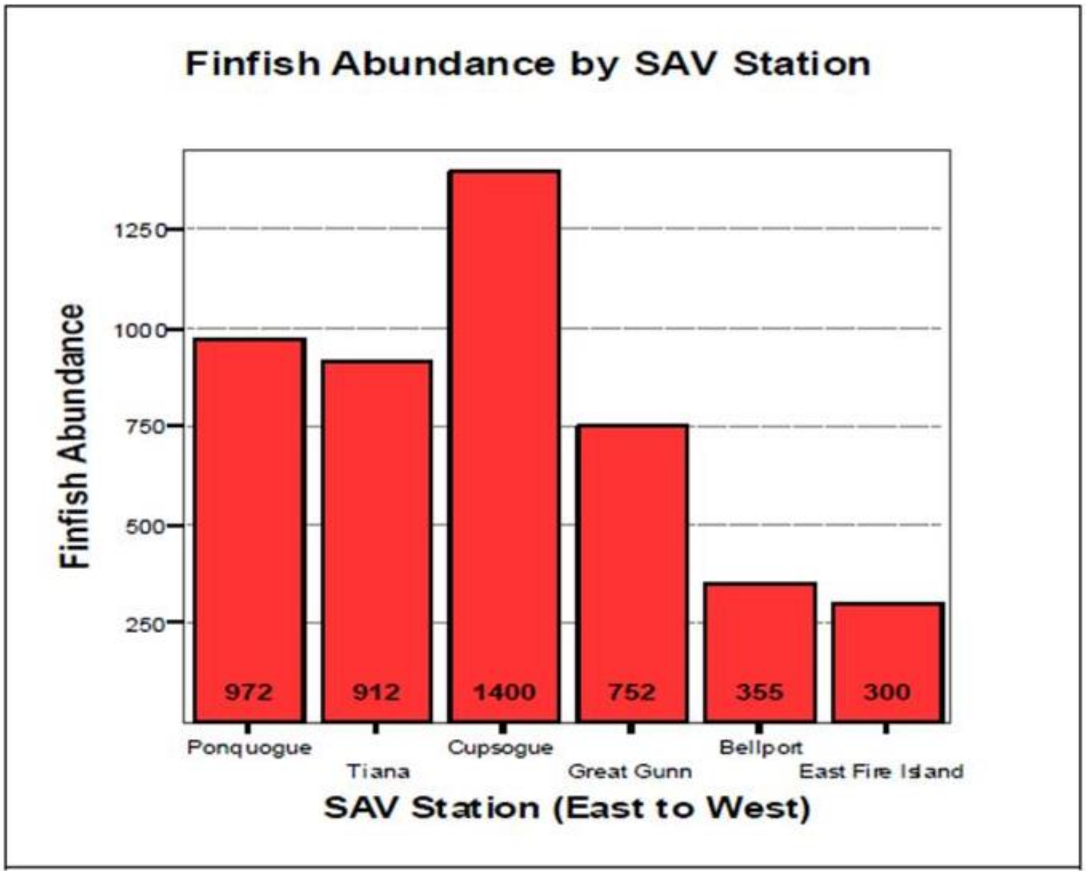

Figure 72. Finfish abundance per site in seine samples collected in submerged aquatic vegetation beds in 2005 (from USACE 2006b).

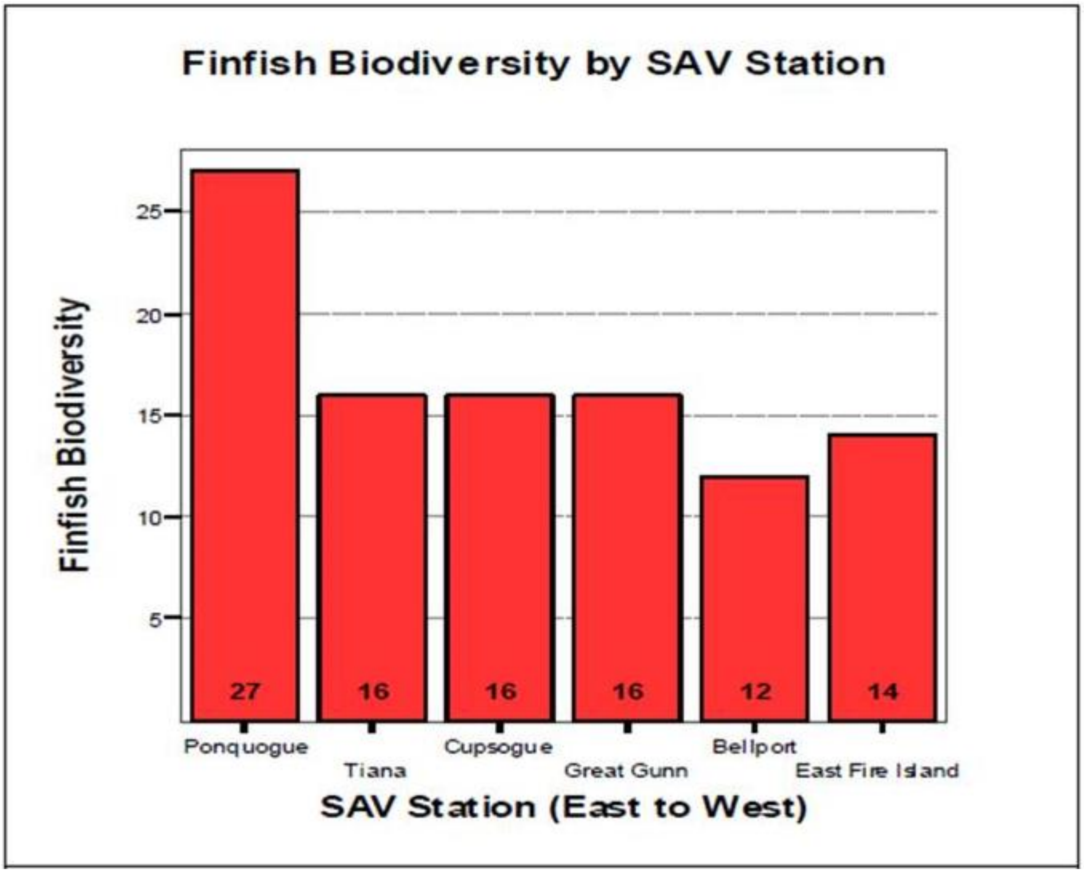

Figure 73. Finfish diversity per site from the 2005 seine survey in submerged aquatic vegetation beds (from USACE 2006b).

EEA Inc. (2002) studied fish and invertebrate populations in the region from June to October 2000 and April to May 2001. Shoreline seine surveys were conducted biweekly at 15 sites from Fire Island Inlet in the west to Shinnecock Inlet in the east (Figure 69). Of these, 7 sites were located in Great South Bay, 5 sites were located in Moriches Bay, and 3 sites were located in Shinnecock Bay. Additionally, a throw trap survey was conducted at a subset of 9 of these sites in September 2000, in which fish were sampled with a dip net during the evaluation of the SAV in the throw trap. Additionally, 4 tidal ponds were surveyed with a seine in September 2000. Similar to the geographic trends reported by the USACE studies (2004b, 2006b), the EEA Inc. (2002) shoreline seine survey found that Great South Bay had the lowest abundances of fish overall, and that in general, finfish abundance increased from west (Great South Bay) to the east (Shinnecock Bay) (Figure 74). Similar geographic patterns were observed in the tidal pond survey. Abundance and diversity of finfish in the shoreline seine peaked in the late summer and early fall with numerical dominants including Atlantic silversides (*M. menidia*), sand lance (*Ammodytes*), striped killifish (*Fundulus majalis*), bay anchovy (*A. mitchilli*), and spotfin killifish (*Fundulus luciae*). Fish densities in the throw traps were generally low and of similar composition to the shoreline seines. Mummichog (*Fundulus heteroclitus*) and sheepshead minnow (*Cyprinodon variegatus*) dominated catch in the four tidal ponds that were sampled.

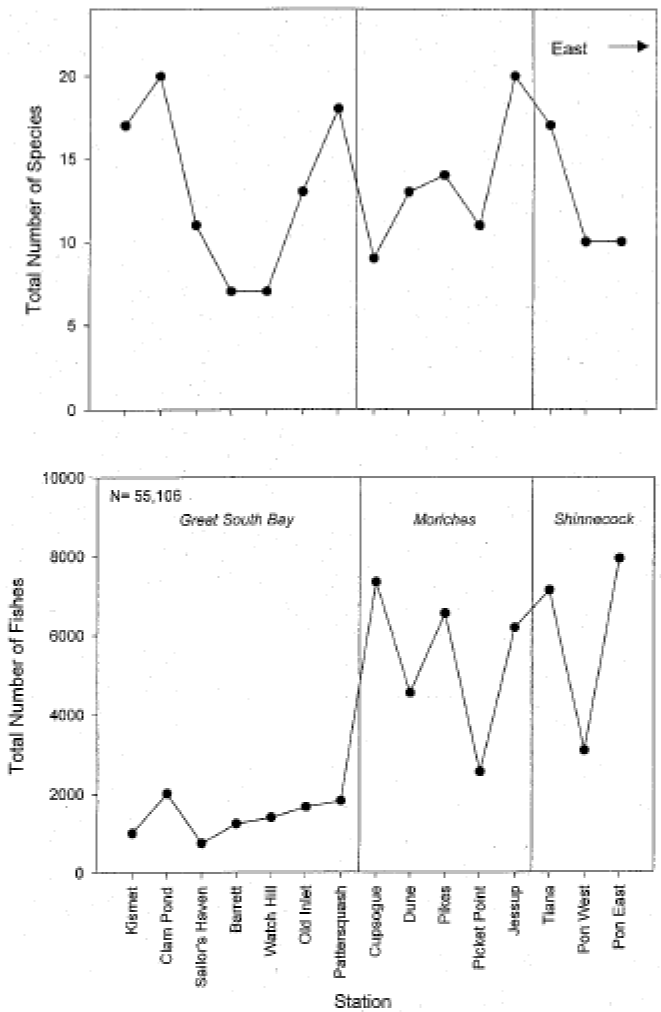

Figure 74. Finfish species richness (top panel) and finfish abundance (bottom panel) at each site during seine survey conducted in 2000–2001 (from EEA Inc. 2002).

After the breach formed, the relative abundance of fish near and east of the breach increased compared to other sites in the survey (Peterson 2015a, 2015b). Peterson (2015a, 2015b) found that sites near Old Inlet and at Pattersquash in Moriches Bay had higher finfish abundance compared to three other sites ranging geographically from Great South Bay to Moriches Bay (Figure 75). The most common species in the survey included bay anchovy, Atlantic silverside, and three-spine stickleback. In seines, the dominant species were bay anchovy, Atlantic silverside, and killifish; and Atlantic silversides and pipefish (*Syngnathinae*) dominated throw trap samples. Atlantic silverside density contributed to the higher abundance values observed at both Old Inlet and Pattersquash, and the decline from 2013 to 2014 in seine abundance noted at Old Inlet reflected a decrease in Atlantic silverside abundance at that site.

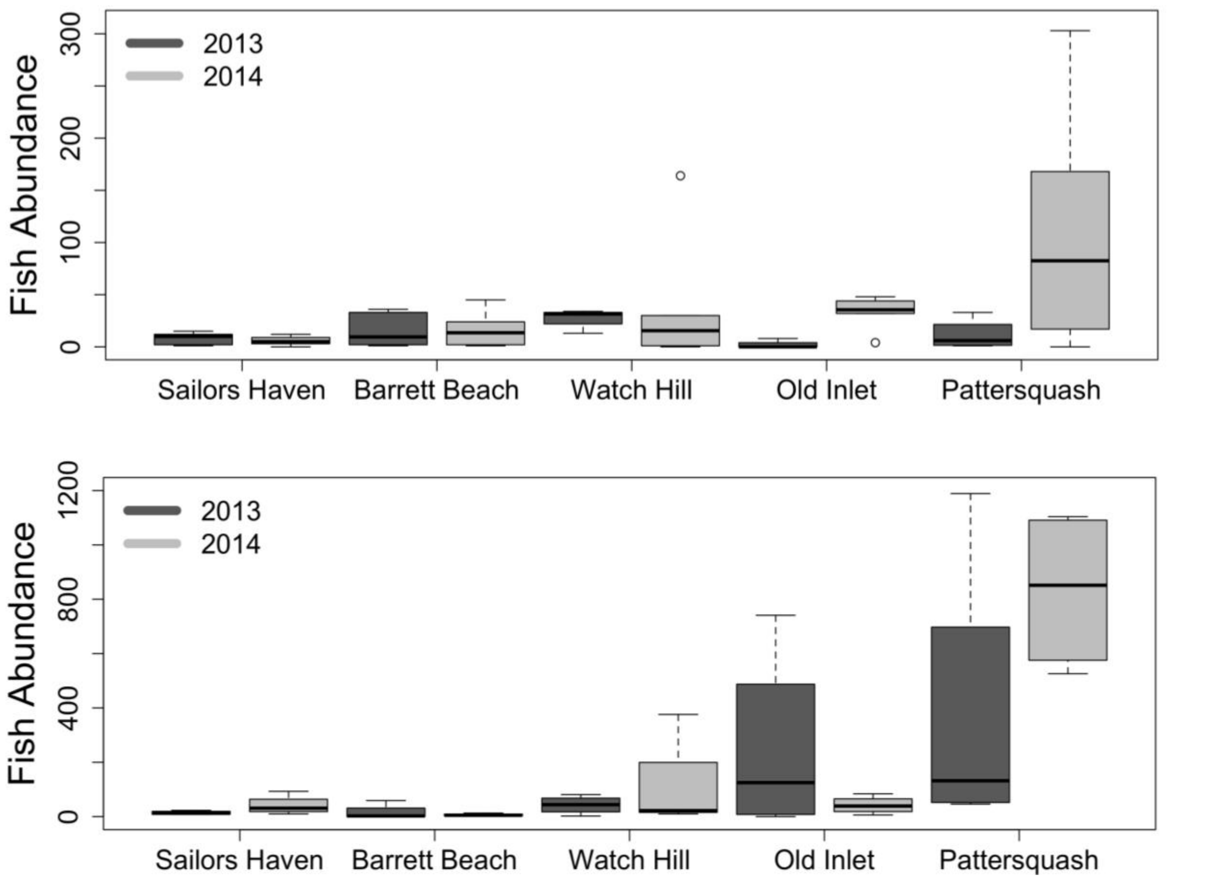

Figure 75. Fish abundance from seine surveys (top panel) and trawl surveys (bottom panel) conducted in 2013 and 2014 (from Peterson 2015a).

A comparison of pre-breach versus post-breach abundance for migratory finfish was made between data from long term surveys performed before and after the 2012 breach. The studies were performed downstream of the dam in the tidal portion of the Carmans River, which empties into Bellport Bay, and which may provide habitat for some life stages of migratory finfish. Long-term monitoring data suggests that the anadromous alewife (*Alosa pseudoharengus*) migration run returns to the Carmans River have increased since the breach formed, a possible indicator that alewife are entering Great South Bay through the breach (Frisk and Nye pers. comm. 2016). Since 2000, the NYSDEC has been conducting an annual survey of American eel (*Anguilla rostrata*) abundance in the Carmans River (NYSDEC 2015b), by deploying a fyke net for a 6-week period in early spring. Glass eel abundance, the early juvenile phase of the catadromous American eel, was notably higher during 2012 and 2013 when compared to abundance data from the previous nine years. However, glass eel abundance subsequently declined in 2014 and 2015 survey samples. Although increased numbers of glass eels during the 2013 survey could be associated with conditions caused by the breach, similarly high numbers were recorded in the spring 2012 survey which pre-dated the breach. It is possible that some other factor contributed to the observed patterns in glass eel abundance for the American eel species. Therefore, it is unclear whether the breach affected glass eel abundance at Carmans River.

Habitat for freshwater and brackish water species has declined since the breach formed. This is demonstrated by pre- and post-breach trawl surveys conducted by Frisk et al. (2015) which showed a sharp post-breach decline (80%) in blue crab abundance after the breach formed (Figure 67). As an estuarine species, blue crab are adapted to the reduced salinity levels that occur within estuary habitats and are less tolerant of ocean water salinity levels; therefore, it is possible that blue crab populations have retreated closer inland to habitat with lower salinity levels (such as brackish water found in tributaries), but no data are yet available to support this idea (McKown pers. comm. 2016).

Return of SAV is providing habitat for finfish and invertebrates since the breach formed. Prior to the breach, *Ruppia* had dominated seagrass beds in Great South Bay, and eelgrass which provides high quality nursery habitat and refugia from predators for fish (Raposa and Oviatt 2000), had undergone a significant decline (Peterson 2015a, 2015b; see the “Submerged Aquatic Vegetation” section). *Ruppia*, while providing some fish habitat, is a shorter grass with less physical complexity, and offers lower habitat value overall (Peterson 2015a, 2015b). Since the breach formed, the clearer, cooler water resulting from the open exchange of ocean water into Great South Bay has promoted the rapid recovery of eelgrass beds (see the “Submerged Aquatic Vegetation” section) particularly in areas adjacent to the breach and to the east towards Moriches Inlet (Peterson 2015a, 2015b). These new beds of SAV are providing habitat for fish. Peterson (2015a, 2015b) monitored fish abundance in beds of eelgrass and *Ruppia* from 2013 to 2014. Abundance increased between survey years and higher densities of juvenile summer flounder and tropical species (with higher salinity level tolerance) were observed in eelgrass beds adjacent to the breach. Previous studies have similarly found strong associations with SAV characteristics such as shoot height or density and finfish abundance (Raposa and Oviatt 2000; Briggs and O’Conner 1971; and USFWS 1981, but see USACE 2006b).

Spartina beds also provide habitat for fish and invertebrates. Prior to the breach, EEA Inc. (2002) documented increasing invertebrate abundance coinciding with *Spartina* biomass in all bays, and with finfish abundance in Great South Bay; however, these trends were not statistically significant.

Invertebrates may experience high predation by finfish near the breach. Pre-breach studies demonstrated an inverse relationship between finfish and invertebrate abundance at certain locations. For example, invertebrates were found to have greatest abundance where fish abundance was lowest in a 2004 survey reported by USACE (2006b) (marsh grass shrimp [*Palaemonetes vulgaris*] and green crab [*Carcinus maenas*] in Cupsogue, Moriches Bay) and in a 2000–2001 survey conducted by EEA Inc. (2002) (marsh grass shrimp [*P. vulgaris*]; sand shrimp [*Crangon septemspinosa*]; and blue crab [*Callinectes sapidus*] in Great South Bay) (Figures 71, 74, 76, 77). This is likely a result of reduced predation due to lower fish abundance. After the breach formed, Peterson (2015a, 2015b) observed lower grass shrimp densities near the wilderness breach where he also observed higher fish densities, which he hypothesized could be driving down shrimp abundance. Predation also affects seasonal trends in invertebrate abundance. For example, EEA Inc. (2002) found that invertebrates were most abundant in the summer, with the exception of an August low, which the authors attributed to predation by fishes which peaked in abundance during this month.

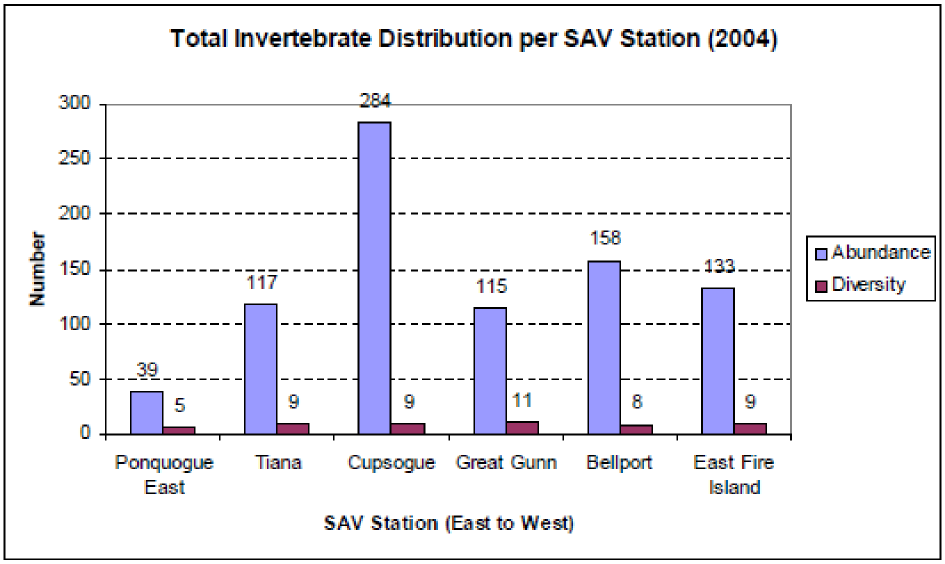

Figure 76. Invertebrate abundance and diversity in seine samples collected in submerged aquatic vegetation beds during 2004 survey (from USACE 2006b).

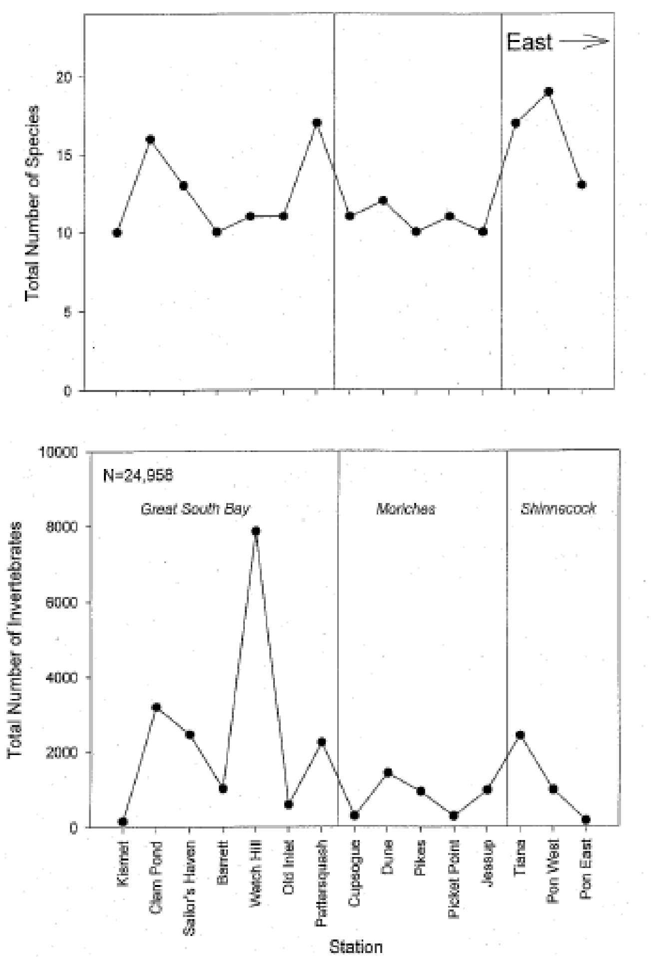

Figure 77. Invertebrate species richness (top panel) and abundance (bottom panel) from seine survey conducted in 2000–2001 (from EEA Inc. 2002).

Data Gaps

Patterns of change in the finfish and decapod crustacean community since the breach formed are just beginning to emerge. There has been little elaboration on how these changes may affect ecosystem function or how burgeoning populations of species such as lady crab or squid may affect the overall ecology of Great South Bay, although such efforts are planned (Frisk et al. 2015). Increased energy exchange with the open ocean could have important implications for finfish and decapod communities. Since the breach occurred in 2012, considerable work has been done to evaluate the potential implications of the breach. However, additional data and studies are needed to continue to evaluate and address questions related to response of finfish and decapods to the breach, and to develop and publish results of the research data in scientific journals. As such, it is too early to determine or predict potential long-term effects of the breach on the Great South Bay finfish and decapod communities.

Summary of Changes since the Formation of the Breach

The formation of the breach resulted in increased exchange of saltwater, organisms, and energy between the open ocean and Great South Bay, resulting in increased species richness. In addition, abundance of some species has decreased, while other species have shown an increase in abundance, a trend that has been attributed to improvements in water quality, moderated water temperatures, and greater habitat availability (i.e., increased SAV density and eelgrass). Decreased abundance for some species is likely a result of increased predation from increasing populations of associated predators or to changes in their environment (e.g., temperature and salinity).

Ecosystem Structure and Processes

An ecosystem is an integration of an ecological community with its environment, and the health of an ecosystem can be described as "a comprehensive, multiscale, dynamic, hierarchical measure of system resilience, organization, and vigor" (Costanza 1992). This technical synthesis report examined major habitat types and major faunal groups in separate sections; however, the biological reality is that the Great South Bay ecosystem is a dynamic integration of all of these groups and habitats interacting simultaneously. These interactions include trophic (i.e., feeding) relationships among the bay's diverse array of fauna and flora, the habitats where these interactions occur, and a suite of abiotic processes that operate in the system such as nutrient cycling and decomposition. The exchange of energy and nutrients among organisms that occurs through trophic relationships can be conceptualized as a food web (Figure 78). A change in population size of one species, taxa, or functional group, will have direct effects on groups to which it is directly linked in the food web, and indirect effects on potentially many more groups through diffuse food web linkages (Paine 1966). Food web-based metrics, such as the number of trophic links, amount of biomass at upper trophic levels, or the level of biodiversity, can provide some insight into the maturity (Odum 1969), stability, and resilience of the ecosystem to disturbance (Rooney and McCann 2012; Nuttall et al. 2011). This section synthesizes the existing information on community and ecosystem level patterns and considers how these patterns might be altered in response to the breach that occurred as a result of Hurricane Sandy in 2012.

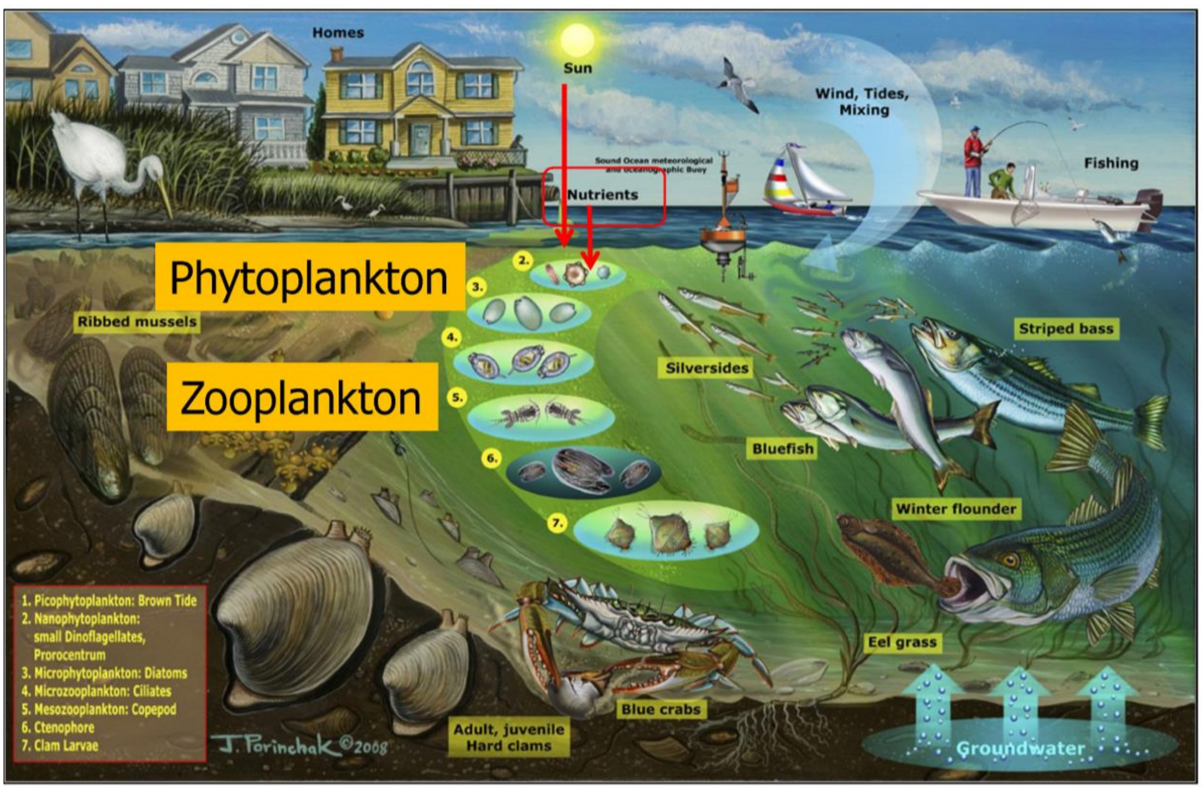

Figure 78. Food web structure of Great South Bay (from Gobler, Collier, and Lonsdale 2014).

Synthesis of Information: Comparison of Pre- versus Post-Breach

Enclosed or semi-enclosed lagoons like the Great South Bay are often sensitive to natural and anthropogenic stressors, which can directly affect ecological communities (Nichols and Boon 1994; Spaulding 1994). Ecological patterns can be disrupted or altered by human intervention (e.g., through fishery harvest or modification of natural inlets), the opening of inlets via stochastic weather events, or large-scale changes in environmental regimes driven by global climate events (Cerrato, Locicero, and Goodbred 2013). A combination of these factors led to dramatic ecological shifts in Great South Bay even before the breach formed (Nuttall et al. 2011). The opening of the wilderness breach by Hurricane Sandy created conditions in which seawater is freely exchanged between the bay and the ocean, resulting in increased salinity (Figures 27 and 28), higher flow, more moderate temperatures (cooler in summer, warmer in winter; Figures 29 and 30), and an increased exchange of organisms (Figures 66 and 67) with the ocean (Gobler, Collier, and Lonsdale 2014; Cerrato and Frisk pers. comm. 2016). All of these changes have taken place in the context of existing human disturbance and long-term historical dynamics.

Ecosystem maturity in Great South Bay has declined over the last 120 years (Nuttall et al. 2011). Lower maturity is associated with lower total biomass, fewer upper trophic level predators, lower connectivity to the ocean due to fewer migratory species, and a less complex food web exhibiting fewer trophic linkages and dominance by lower trophic levels. The shift in ecosystem maturity of Great South Bay has been attributed to the reduced role of rapid “state” changes such as the opening of inlets by storms, which has been replaced with human practices such as engineering of permanent inlets and closure of breaches (Nuttall et al. 2011). An increase in one or more of the ecosystem maturity metrics could provide evidence for a recovery of ecosystem maturity in Great South Bay. The formation of the breach has affected some of these metrics, with the resulting effects summarized below.

Prior to the breach, a shift away from the functional dominance of upper trophic level predators such as sand tiger shark, weakfish, and gadoids was reported for Great South Bay, resulting in an ecosystem dominated by mid-level predators (Nuttall et al. 2011) (Figure 79). The breach opening provides a portal through which these species, and the smaller fish that are their primary food resource, could enter the bay. Although a sand tiger shark nursery has recently been discovered in Great South Bay (New York State Aquarium 2016), it is unknown whether it is associated with the breach. There is insufficient information to determine or predict the potential long-term effects to the trophic structure of Great South Bay resulting from the breach opening.

Species richness has increased in the bay since the formation of the breach (Figure 66). Frisk et al. (2015) demonstrated that there are more species, more total biomass, and likely more trophic linkages and a more complex food web in the bay since the breach compared to before the breach formed. The authors suggest this as further evidence for a recovery of ecosystem maturity and increased resilience to disturbance (Cerrato and Frisk pers. comm. 2016; Frisk et al. 2015).

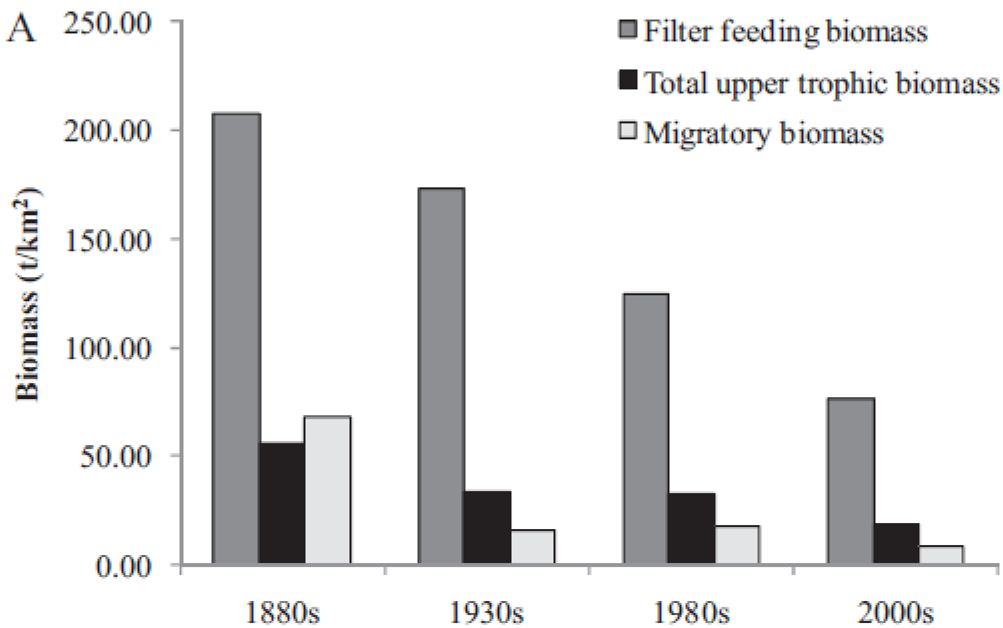

Figure 79. Biomass of filter feeders (i.e., suspension feeders), upper trophic levels, and migratory species has declined since the 1880s. Filter feeders included hard clams, oysters, other suspension feeders, sand shrimp, other shrimp, menhaden, and zooplankton. Migratory species included black sea bass, bluefish, gadids, menhaden, scup, sharks, skates, squid, striped bass, summer flounder, weakfish, winter flounder, and tropical fish (from Nuttall et al. 2011).

Suspension feeding by shellfish has been in decline since the 1880s and has been missing from the Great South Bay ecosystem for the last 30 years (Gobler, Lonsdale, and Boyer 2005; Kraeuter et al. 2008) (Figure 80). Hard clams, oysters, and menhaden once performed the crucial ecosystem function of suspension feeding, which removes phytoplankton and other suspended organic matter from the water column, thereby clarifying the bay water. Active commercial fishing, changes to the environment, and predation pressure brought about by the formation of Moriches Inlet in 1931 led to the collapse of the oyster population in the early 1950s (McHugh 1972). Similarly fisheries harvest, increased juvenile predation rates, and increased frequency of brown tide have severely depleted the bay's stock of hard clams (Gobler, Lonsdale, and Boyer 2005; Bricelj 2009; McNamara, Lonsdale, and Cerrato 2010). Since the wilderness breach formed in 2012, hard clam populations in Great South Bay have continued to exhibit low densities (as indicated by fishery landings provided by Barnes pers. comm. 2016a) and severely low spawning and reproductive success (Figure 63). There are insufficient data to determine whether a natural recovery in hard clam populations and the ecosystem function of suspension feeding will result even if conditions favoring recovery occur in any part of Great South Bay, either with the breach open or with the breach closed.

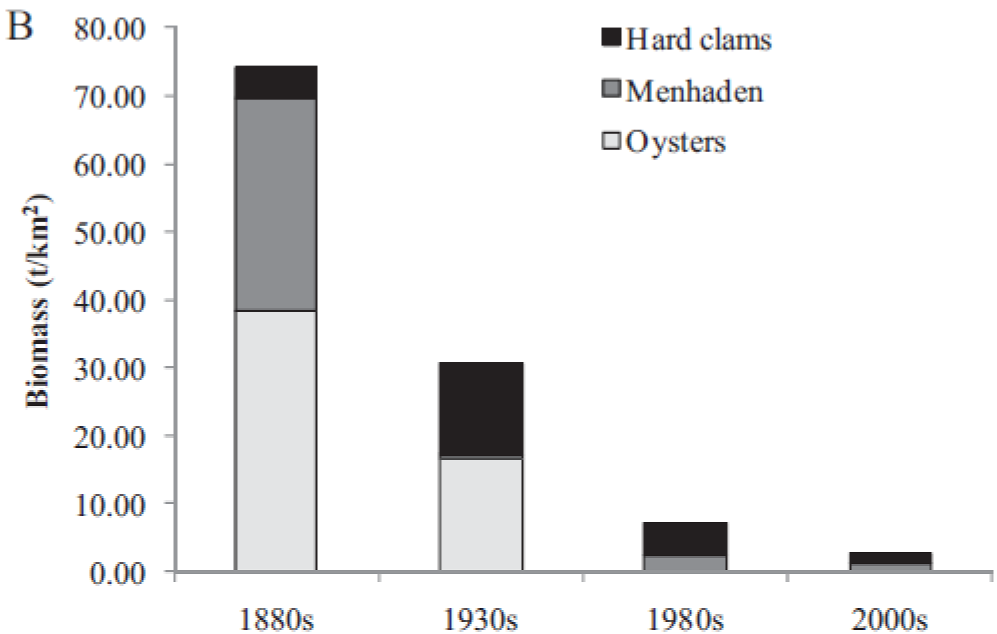

Figure 80. Biomass of three major suspension feeding groups has declined since the 1880s (from Nuttall et al. 2011).

Atlantic menhaden, a mobile suspension feeder, also once performed the vital function of suspension feeding for Great South Bay, removing detritus, phytoplankton, and zooplankton from the water column to use as food (Deegan, Peterson, and Portier 1990; Durbin and Durbin 1998; Nuttall et al. 2011). Intense fisheries for this species led to its collapse in 1966 (McHugh 1972). However, with the breach open there is a new physical entryway through which menhaden could enter Great South Bay. Survey data for menhanden suggest that this species is present but patchily distributed in Great South Bay, but that they are not nearly as abundant as they once were historically (Frisk et al. 2015). Currently, there are insufficient data to determine whether an increase in menhaden and the ecosystem function of suspension feeding that they provide will result with the breach open.

The pre-breach loss of suspension feeders, namely oysters, hard clams, and menhaden, has likely contributed to greater levels of phytoplankton in Great South Bay, a shift from benthic to pelagic primary production, and potentially a shift in the types of phytoplankton in the ecosystem (Figures 34 through 40). Low water clarity and light attenuation at the bay bottom prevents the growth and expansion of seagrass beds that can provide habitat for finfish and invertebrates (e.g., Peterson 2015a, 2015b). However, despite the absence of a dominant suspension feeder, post-breach water clarity has improved in the Bellport Bay, Narrow Bay, and western Moriches Bay areas (Gobler pers. comm. 2015) due to the export of suspended matter out to the ocean through the breach (Figure 41). This, together with the reportedly cool, saline water that the breach is bringing into the bay, has been linked to improved conditions for eelgrass (Figures 49 and 50) (Peterson 2015a; Heck and Peterson 2016).

Shifts in species composition in the areas affected by the breach have been reported; however the long term impact on these shifts on ecosystem function is not yet understood. For example, there has been a notable decline in blue crab and a subsequent increase in lady crab since the breach formed (Figure 67) (Frisk et al. 2015). It is not known whether this shift in species composition is related to the breach. However, both species are scavengers commonly feeding on mollusks and crustaceans. Total predation by crabs may increase if the post-breach density of lady crab exceeds the pre-breach density of blue crabs. Moreover, McKown (pers. comm. 2016) suggested the potential for a spatial shift in blue crab distribution, with blue crabs moving into the lower salinity tributaries to escape the higher salinities in the areas affected by the breach.

Habitats such as eelgrass, marshland, and the sediments on the bay bottom provide unique spatial resources for particular taxonomic groups. For example, eelgrass beds and other SAV provide nursery and refugia habitat for crustaceans and juvenile fishes (McElroy et al. 2009 and citations therein). Marshes provide habitat for shellfish and foraging habitat for shorebirds, and sand and mud sediments support unique benthic communities (Cerrato, Locicero, and Goodbred 2013). In addition to the potential for expansion of eelgrass, there is also potential for new marsh habitat to develop in newly formed flood-tidal deltas (Figure 42) and overwash areas that provide platforms on which marsh vegetation is likely to become established, given appropriate elevation and available propagules. These new flood-tidal deltas have the potential to support new marshes, which has occurred historically in other overwash areas throughout Great South Bay.

Paleo-ecological data for central Great South Bay has demonstrated a shift in the last ~650 years from coarse sediments to organically rich muddy sediments, and appears to favor a shift toward assemblages dominated by the dwarf surf clam (*Mulinia lateralis*) (Cerrato, Locicero, and Goodbreed 2013). The authors attribute this shift to global climate shifts affecting storm frequency as well as human land use, both of which affected sediment deposition in the central Great South Bay (Cerrato, Locicero, and Goodbreed 2013). The dwarf surf clam assemblage is associated with environmental disturbances; therefore, the increasing dominance of dwarf surf clam dominated assemblages over the last 300 years suggests that the Great South Bay has experienced increasing levels of environmental disturbance in that period (Cerrato, Locicero, and Goodbreed 2013). The authors suggest that engineering projects to stabilize inlets or nourish beaches would make it more likely that the deposition and expansion of muddy sediments will continue, which could make the ecosystem more susceptible to disturbance and limit the recovery of recent species that depend upon coarser sand sediments for habitat (Cerrato, Locicero, and Goodbreed 2013). This work suggests that engineering or dredging to stabilize the breach could further increase the susceptibility of Great South Bay to disturbance.

Data Gaps

The Great South Bay ecosystem has historically experienced many forms of directional, human-induced impacts as well as stochastic weather and climate events. The effects of the wilderness breach that formed during Hurricane Sandy in 2012 is just beginning to be quantified and understood. Little information is available, from either before or after the breach, to describe some of the ecosystem level processes such as nutrient cycling, decomposition, and biomass turnover rates.

Summary of Changes since the Formation of the Breach

Ecosystem maturity of Great South Bay has increased since the formation of the breach. This includes increased biomass and species diversity, greater connectivity to the ocean, and development of a more complex food web with more trophic links. More mature ecosystems are healthier, more stable, and more resilient to disturbance. Although post-breach improvement in some metrics of ecosystem maturity and ecosystem health are expected or have already occurred, it is not known whether other remaining ecosystem functions will recover, such as consumption by upper trophic levels or suspension feeding and its impact on water clarity. The formation of the breach has contributed to the expansion of eelgrass beds, which in turn have been associated with increased fish and invertebrate production. There is potential for marsh habitat expansion on the developing flood-tidal deltas as well, which could likewise provide new habitats for floral and faunal species.

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Appendix A: Benthic Community Recovery

Benthic communities are generally thought to recover quickly following a disturbance. However a number of factors influence the rate of recovery, including definitions of "recovery," and methods used to measure it. The purpose is to provide information on benthic recovery from the literature that may inform the general discussion of potential benthic changes in the vicinity of the wilderness breach. The following subjects are covered: (1) general principles of succession and benthic recovery; (2) factors affecting benthic recovery rates; (3) study methods for determining benthic change or recovery; (4) recovery rate estimates from the literature.

1. Benthic Community Recovery: General Principles

Benthic community recovery can be described using general benthic succession paradigm (Pearson and Rosenberg 1978; Rhoads and Germano 1986) that describes changes to the community following disturbance as a three-stage process. Pioneering (Stage I) organisms, such as tube-dwelling polychaetes and small bivalves quickly colonize the surficial sediments. These opportunistic taxa usually occur in high abundance and low species diversity. Over time they are replaced by larger, longer-lived and deeper burrowing (Stage II) organisms. With more time, Stage III benthic assemblage develops, and is characterized by a more diverse but less abundant group of larger taxa with more physical structure and functional groups.

While recovery may follow this general trend, and the early literature provides a basis on which to evaluate specific benthic community studies, a number of factors affect the nature of progression and rate of change in the benthos in any given situation.

2. Factors Affecting the Nature and Timing of Recovery

Variables known to affect recovery rates include sediment grain size and organic matter, spatial scale of disturbance, timing and frequency of disturbance, latitude, physical parameters such as slope and stability of affected sediments, and the life history of resident species (Kotta, Kotta, and Kotta 2009; Wilber and Clarke 2007; Newell et al. 1998), and the availability and transport of larval sources (Todd 1998). In addition, divergent reassembling can occur late in the benthic recovery process, making the prediction of community structure difficult (Van Colen et al. 2010).

3. Evaluating Change in Benthic Communities

Measuring changes in benthic communities can be difficult because of the natural variability inherent in most benthic systems. Looking for these changes is generally done by comparing a disturbed or new benthic community to a pre-disturbance baseline condition, a reference community, or both. Comparison to both is done using a "before-after-control-impact" study design. With any one of these approaches, data analysis methods matter, and these methodological factors influence study conclusions just as much as the physical factors that influence recovery (Wilber and Clarke 2007). Benthic communities are often compared using univariate methods such as total abundance, taxa richness, and total biomass. These parameters are useful descriptors but do not provide a complete analysis of whether a recovery has occurred. Diversity indices are sometimes used to reduce multivariate data (e.g., abundance of multiple species) into a single index such as the Shannon-Weiner diversity index (H'), or Pielou's evenness index (J'). Changes between sites or over time are plotted as means and confidence intervals for each site and time, and 'recovery' can be identified when values for an impact area are no longer significantly different from a reference or baseline condition. Similarly, if pre- and post-breach data were available, a change from a pre-breach to post-breach condition could be identified when values for the new condition are significantly different from those for the baseline. Multivariate analyses are also useful. Cluster analysis, which provides dendograms that group samples so that samples within a group are amore similar to each other than samples from different groups. By defining these different groups for a set of samples, distinct differences between the communities from a set of samples or stations can be described. Comparison of breach area benthos to those in areas outside the influence of the breach could be done using this method. Another method includes non-parametric multi-dimensional scaling ordination plots using the Bray-Curtis similarity measure to identify groups of samples having similar (or different) faunal assemblages. One last method for determining change in benthic communities is to define a baseline or expected range for some parameter such as diversity, determine a threshold value beyond which it is not expected to change as a result of 'normal' variation, and evaluate whether the measure approaches that threshold (Keay and Mickelson 2000). All of these methods require a large number of samples because of the natural patchiness and variability in the benthic community. In short, measuring change or recovery in benthic communities is complex, and teasing out the effects of the breach from natural variability will be challenging particularly given the lack of pre-breach data.

4. Recovery Times following Disturbance

Benthic communities appear to recover (or return to a condition indistinguishable from baseline or reference conditions) within months to a few years. Newell et al. (1998) characterized typical recovery times at 6–9 months for mud habitats and 2–3 years for sand and gravel. Kotta and others (2009) found recovery to be complete after one year following a major dredging project.

Wilber and Clarke (2007) summarized the results of more than 60 studies on benthic recovery time following dredging, filling, or capping, and showed that most systems recovered within a year, although a few of the studies reported recovery times as long as 3–4 years. The authors concluded that, although recovery rates are difficult to predict, suites of factors can be associated with either recovery measured in months (such as shallow, naturally disturbed habitats, unconsolidated, fine grain sediments, analyzed with univariate analytical approaches), or years (deep, stable habitats, sand and gravel sediments, and multivariate or functional group analytic techniques). Further, the most consistent physical parameter influencing recovery rates is the prior disturbance history of the site in question. The breach area can be characterized as sandy and shallow, previously stable but now changing to a more dynamic environment with higher flow. Benthic community recolonization, and development of a new, higher-salt community on new overwash and intertidal areas, is likely to have occurred since the breach. Additional field study would be useful in comparing the near-breach benthic system to a reference area.

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