GEOMORPHOLOGY OF TIDAL WETLANDS: IMPACTS OF EXTREME AND ANNUAL FLOOD EVENTS TO SALT MARSH AND MANGROVE SYSTEMS

Frances R. Griswold
University of Massachusetts Amherst

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GEOMORPHOLOGY OF TIDAL WETLANDS: IMPACTS OF EXTREME AND ANNUAL FLOOD EVENTS TO SALT MARSH AND MANGROVE SYSTEMS

A Dissertation Presented

by

FRANCES R. GRISWOLD

Submitted to the Graduate School of the University of Massachusetts Amherst in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

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Department of Earth, Geographic, and Climate Sciences
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Approved as to style and content by:

_________________________________________________
Dr. Jonathan D. Woodruff, Chair

_________________________________________________
Dr. Brian C. Yellen, Member

_________________________________________________
Dr. Timothy L. Cook, Member

_________________________________________________
Dr. Don J. DeGroot, Member

Piper R. Gaubatz, Department Head
Department of Geosciences
DEDICATION

To friends and family who walked with me through the lows and celebrated alongside me during the highs.

I couldn’t have done this without you.
ACKNOWLEDGMENTS

I would like to thank my advisor, Jonathan D. Woodruff, for all his support, encouragement, and advice through this process. His kind and steady mentorship has grown me into the scientist that I am. Special thanks to the rest of my committee, Brian Yellen, Tim Cook, and Don DeGroot for input on my work and guidance on the direction of the projects. I am especially thankful for the mentorship and friendship of Brian Yellen who has been such an amazing team player, always willing to answer any questions, and makes lab and fieldwork extra fun. Also, for Tim Cook, who has been such a great example of critical thinking, careful work, and efficiency in the field and lab. Your guidance has made me a better scientist and mentor.

I would like to thank the members of the Sediment and Coastal Dynamics lab (aka “The Sed Squad”) Jon Woodruff, Brian Yellen, Tim Cook, Lucy Hansen, Kelly McKeon, Pedro Mato-Llavona, Hannah Baranes, Caroline Ladlow, Alycia DiTroia, Doug Beach, Molly Autery, Bonnie Turek, and the many undergraduates who passed through this lab. Thank you for teaching me, getting muddy with me, and adding so much fun to this process. I have learned so much from you and it has been an honor to work alongside you all these years. In addition, I want to thank other members of the UMass Geosciences community who I have had the privilege to work with on both coursework and DEI work. You have become such wonderful friends, and I am so happy we go the opportunity to teach, learn, grow, and make a difference in this place together. In particular, I want to thank Bill Clement who was always there to give advice, have lunch, or lighten a dark day.
Finally, I would like to thank my friends and family, both near and far. These past years have been some of the hardest moments, between loss and a pandemic, but also some of the most joyous celebrations. We have supported and developed into better versions of ourselves through these challenges and joys over the past few years. The phone calls, check-ins, long walks, and conversations have helped keep me balanced and taught me to heal and grow. I am so thankful for each of you, and I hope I can be a steady rock for you to lean on like you have been for me. Eric, thank you for all your support, encouragement, and love, especially in these last few months. Thank you, Mom, for always answering the phone, and Dad, for taking me on hikes and allowing the stresses of the world to stop for a bit. I am so thankful for each one of you, and I couldn’t have made it this far without you.
Tidal wetlands are vital for buffering coastal settings from the threats of accelerated sea level rise and storms. Understanding the factors that are most influential for the maintenance and recovery of tidal wetlands after extreme events compounded by future accelerated sea level rise is of the utmost importance, yet this knowledge is not well established. Two tidal wetland schemas investigated in this dissertation are mangrove systems in Vieques, Puerto Rico (including robust lagoonal-mangrove forest systems and fringing mangrove forests), and salt marshes in New England. While the climatic forcings, vegetation type, and locations are vastly different for these two tidal wetlands, there is considerable overlap in the stressors and factors needed for resilience. Chapter 1 provides background information on tidal wetlands, and the stressors impacting them.
Chapters 2 and 3 examine hurricane impacts and recovery for mangrove forests in Puerto Rico. The recovery potential of mangroves influences the long-term buffering capacity of these unique coastal forest systems in terms of wave attenuation, wind breaking, and sediment overwash deposition. Different mangroves, however, respond differently to hurricanes based on the direction and intensity of impact during the storm, and the indirect impacts that occur after the storm has passed. Fringing mangroves on exposed coasts, such as those examined in chapter 2 at Laguna Playa Grande on the south side of Vieques, are more impacted by direct damages. In more protected regions that have a more robust forest with a variety of mangrove species, impacts are more indirect as a result of a delayed mortality from damage occurred during the event and reduced resilience to environmental stressors following the event. In addition to the damages from hurricanes, human disturbance to mangrove forests plays a major role in the system’s ability to recover, leaving coastlines more vulnerable to damages and storm-induced deposition during the recovery phase following the event.

Chapter 4 presents a method for assessing the sourcing, timing, and mechanism of clastic sediment to New England marshes using a new cost-effective sediment trap design. In addition to the sediment trap methodology, we also present a regional assessment for the sourcing and timing of delivery for representative salt marsh in the region through analysis of instrumental data from turbidity sensors and water level loggers, as well as accumulation of clastic sediments on the marsh surface through sediment traps to assess the seasonal delivery of inorganic sediment to salt marshes. Three representative marshes were chosen to cover the wide variety of salt marsh systems in the region from the microtidal marshes south of Cape Cod, MA (Westport) to the
mesotidal marshes in Maine (Sprague and Wells). Due to oceanographic, geologic, and environmental differences in marsh systems throughout the region, a heterogeneity in sourcing and timing of sediment supply is observed. This study provides valuable regional context for the spatial and temporal differences of inorganic sediment sourcing and delivery to New England salt marshes.
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CHAPTER 1

INTRODUCTION TO COASTAL DYNAMICS WITHIN TIDAL WETLANDS

1.1 What are Tidal Wetlands

Tidal wetlands are low-lying areas along the coast, or along tidally influenced rivers, which are periodically flooded by the tide (Goodwin et al., 2001). Tidal wetland is the broad term for coastal features that can include salt marshes, mangrove forests, and freshwater or brackish wetlands. Regardless of the type of wetland, they are vital for buffering the coast from extreme events and flooding by attenuating wind and waves (Vernberg, 1993; Baldwin et al., 2001; Costanza et al., 2008; Das and Vincent, 2009; Branoff, 2019a). Currently, more than 40% of the human population lives in coastal areas, and as such, understanding the mechanisms that influence the survival or demise of these coastal buffers is of the utmost importance (UNEP, 2006).

Beyond their intrinsic value and role in coastal buffering, tidal wetlands provide many other coastal ecosystem functions including habitats for many juvenile fish and nesting birds (Lugo and Snedaker, 1974, Ewel et al., 1998; Tomlinson, 2016; Hapsari et al., 2020). They also are important filters of nutrients and pollutants that enter the estuary and coastal regions from terrestrial runoff, protecting offshore communities such as coral reefs and preventing toxic algal blooms (Gedan et al., 2009; Hapsari et al., 2020). Further, tidal wetlands are a sequester of carbon, making them vital for combatting the ongoing impacts of climate change (Burden et al., 2013; Beaumont et al., 2014).

Research by Burden et al., 2013 has shown that restoration of marshes can provide a valuable sink for atmospheric CO2.
Historically, tidal wetlands have been used by humans for a variety of purposes leading to changes to the system and in some cases marsh loss. These include ditching, waste disposal, livestock grazing, fishing, hunting, transportation on navigable waterways, harvesting of marsh hay, filling of wetlands to create dry ground for development, and clear cutting for timber in the case of mangroves (Ayala, 2001; Bromberg and Bertness, 2005; Gedan et al., 2009). Distal to marshes and mangroves, humans still have a major impact on the resiliency of these tidal wetlands. Damming of rivers, land clearance, coastal armoring, and redirecting of water all have an impact on the sediment delivery to wetlands (Kirwan et al., 2011; Weston, 2014; Yellen et al., 2018; Cook et al., 2020; Ralston et al., 2021). Some of these anthropogenic interactions lead to increased sediment flux (e.g. land clearance), while others result in sediment starved systems (e.g. damming and coastal armoring). Not all human activities, however, are detrimental to tidal wetlands. In some instances, increased erosion from poor agricultural practives provided needed inorganic sediment for salt marsh growth, and the creation of man made embayments provided protected areas where marshes could grow (Tiner, 2013; Yellen et al., 2021).

1.2 Assessing Tidal Wetland Responses to Extreme Events

Hurricanes are one of the major natural disturbances impacting tidal wetlands leading to a loss of coastal protection, property damage, and reconfiguration of coastlines (Boose et al., 1994; Rodríguez and Bush, 1994). Additional environmental stressors reduce the recovery potential after hurricanes. These environmental stressors include accelerated sea level rise, changing temperatures and weather patterns due to climate change, and human impacts (Gilman et al., 2008). Mangroves, living in the tropics,
typically experience more intense and frequent hurricane impacts than the more temperate salt marshes. Further, since mangrove forests are trees with higher growth elevation than salt marshes, they are more susceptible to direct impacts leading to mortality through windthrow (uprooting), defoliation, and breaking of the main stem (Jimenez and Cintron, 1985; Patrick et al., 2020). This dissertation investigates the effects of hurricane-induced mangrove damage and the future implications for flooding along tropical coastlines and paleo-hurricane interpretations.

Salt marshes are capable of attenuating storm waves and resultant protection of coastal communities and infrastructure. In some instances, however, these extreme events can provide much needed sediment, particularly to sediment starved back-barrier marshes (Ganju et al., 2019). A major limiter to salt marshes in New England, apart from sediment delivery, is a lack of landward accommodation space. With this lack of horizontal accommodation space, routine and extreme flooding are vital for the delivery of inorganic sediment so that marshes can gain the elevation capital to keep up with sea level rise.

1.3 Resiliency of Tidal Wetlands

Chapters 2 and 3 of this dissertation highlight mangroves’ capability to rebound after extreme events, withholding any anthropogenic influences. With time, mangrove seeds can be distributed, take root, and the system can regrow. Past and present disruptions to mangroves have a first order control on the level of protection they provide, and subsequent hurricanes can exacerbate damages and delay recovery. The location and extent of the forest (e.g., fringing mangroves in direct paths of hurricanes or
robust forests in protected lagoonal systems) play a large role in the capacity for mangroves to recover after an event.

In salt marsh systems, it has long been recognized that sediment supply is important to maintain elevation capital with respect to sea level rise (e.g. Redfield, 1972; Ganju et al., 2019; Baranes et al., 2022). Recent work by Ganju et al., (2019) have expanded on the predominant sourcing of inorganic sediment supply delivery including: (1) external to the marsh from marine sources brought in during storms and the flood tide, (2) external to the marsh from fluvial sources during high discharge events, and (3) internal sourcing of sediment from erosion of the marsh edge and resuspension of sediments. Results from Chapter 4 on the timing, mechanism, and sourcing of inorganic sediments to salt marshes provides one of the first assessments for the region. I envision this serving as a significant resource to coastal managers and their restoration initiatives as understanding the source of sediment will impact decisions concerning infrastructure in both the uplands and the coast.
CHAPTER 2

HUMAN AND HURRICANE IMPACTS TO MANGROVES MODULATE
OVERWASH DEPOSITION TO A BACKBARRIER LAGOON

2.1 Abstract

Laguna Playa Grande (LPG) is a backbarrier lagoon located on the southern coast of Vieques, a 135 km$^2$ island to the southeast of mainland Puerto Rico. Previous work identified hurricane deposits in the lagoon dating back over 5000 years, with periods of increased storm-induced overwash activity attributed to variability in regional hurricane climatology. In 2017 Hurricane Maria made direct landfall on the island of Vieques just below category 5 strength, providing the opportunity to revisit the site to improve upon interpretations of storm-induced deposition. The Maria event resulted in widespread wave-induced overwash of the barrier beach and extensive mangrove mortality. Ripped up and broken mangrove material was pushed to the backside of the barrier, forming an extensive wrack line. Significant sediment trapping and overwash deposition occurred within this mangrove debris which prevented sediments from being carried further landward into the lagoon. Thus, despite the high intensity of wind and waves associated with Hurricane Maria, no measurable overwash deposition was observed in the larger, western portion of the lagoon where previous hurricane reconstructions are derived. Significant overwash deposition was observed in the smaller, eastern portion of the lagoon where human cut paths through the mangroves allowed for unobstructed flow. Early historical photos reveal reduced vegetation in the early 1900s likely reflecting
deforestation of the barrier to support extensive sugarcane operations on the island. This early mangrove deforestation was followed by revegetation towards present day, which explains the discrepancy between previously observed overwash deposition by historical hurricanes compared to restricted deposition solely to locations with modern footpaths for the more recent Hurricane Maria event. Extensive mangrove mortality following Hurricane Maria resulted in roughly a 40% reduction in vegetative cover along the barrier beach at the site. Hurricane events occurring during the recovery-revegetation phase of such storms likely result in greater overwash deposition in backbarrier lagoons due to less barrier vegetation. Results highlight the important role of fringing mangrove forests in flood mitigation, as well as the vulnerability of back-barrier environments to enhanced flooding following both anthropogenic and event-driven habitat disturbance.

2.2 Introduction

Barrier beaches often serve as the first and primary line of flood defense for coastal populations and ecosystems that reside behind them. The integrity of these barrier systems as a form of flood protection is impacted by a wide variety of factors including sea level rise (Harris et al., 2010; Wdowinski, 2016; Meeder and Parkinson, 2018; Hapsari et al., 2020), storm activity (Harris et al., 2010; Johnstone et al., 2016; Sippo, 2018), sediment supply (Muto and Steel, 1997; Beets and van der Spek, 2000; Gilman et al., 2008; Allison et al., 2017; Hudson and Baily, 2018, Woodruff et al., 2021), inlet dynamics (Cahoon et al., 2003; Lewis et al., 2016, Yellen et al., in review), and changes to the morphology of the beach-dune system (Castañeda-Moya et al., 2010; Wdowinski, 2016; Meeder and Parkinson, 2018; Radabaugh et al., 2019). The distribution and density of barrier beach vegetation are also widely recognized as a key control on erosion.
(Mendelssohn et al., 1991; Harvey, 2006; Feagin et al., 2015), as well as the degree to which storm waves and surge attenuate before impacting back-barrier locations (Lugo and Snedaker, 1974; Ewel et al., 1998; Baldwin et al., 2001; Costanza et al., 2008; Das and Vincent, 2009; Tomlinson, 2016; Branoff, 2019; Hapsari et al., 2020). For tropical and many subtropical locations, mangroves represent the predominant form of tidal wetland vegetation, serving to buffer more inland locations from the damaging effects of storm induced wind and waves (Baldwin et al., 2001; Costanza et al., 2008; Das and Vincent, 2009; Branoff, 2019a), as well as filling many additional roles for coastal ecosystems including habitats for juvenile fish (Ewel et al., 1998; Tomlinson, 2016; Hapsari et al., 2020), nesting grounds for coastal birds (Lugo and Snedaker, 1974, Ewel et al., 1998; Tomlinson, 2016), and filtering pollutants and nutrients from land runoff to protect coral reefs (Hapsari et al., 2020). While mangroves are important for buffering against extreme wind and waves during hurricanes, they are also susceptible to damage during these events (Ellison, 1999, 1999; Baldwin et al., 2001; Costanza et al., 2008; Radabaugh et al., 2019; Patrick et al., 2020). Direct impacts during the storm itself include windthrow, breaking of the main stem, extensive defoliation, or shedding of bark (Jimenez and Cintron, 1985; Patrick et al., 2020). Oftentimes wracklines, or lines of debris will catch on mangroves, resulting in severe tree damage and mortality (Jimenez and Cintron, 1985). Indirect impacts from hurricanes, including siltation, erosion, flooding, and stagnant or hypersaline water can further stress mangroves weeks to months after a storm (Cintron, G. et al., 1978; Jimenez and Cintron, 1985; Medina, 1999). The additional stress from these environmental changes hinders mangroves’ ability to perform necessary functions or compete with other flora.
Sediments collected from back-barrier wetlands archive how storm-induced overwash activity has changed over time (e.g. Wallace et al., 2014; Muller et al., 2017). However, the dynamic nature of coastal environments makes it difficult to deconvolve the conditions that control the predominant characteristics of individual overwash deposits and trends in overwash activity over time. Observations on the impact of extreme coastal flooding events on barrier beaches shortly after their occurrence provide the unique opportunity to evaluate controls on resultant overwash deposition firsthand. Such assessments are valuable not only for interpreting the potential causes for the variability observed within sediment-derived paleo-overwash reconstructions from a particular site but also for evaluating the key controls that impact the resilience of such systems in response to future storm events. Towards this end, I provide here a detailed assessment of the impacts following the direct landfall of a major hurricane event (Hurricane Maria, 2017) on a representative barrier beach and mangrove fringed back-barrier system (Laguna Playa Grande, Vieques, Puerto Rico) through sediment core analysis, turbidity and water level data, and aerial imagery analysis. I then utilize these data to refine and update interpretations on historical and paleo-hurricane reconstructions previously derived from the site (Donnelly and Woodruff, 2007).

2.3 Field Site

2.3.1 Geographic Setting

Vieques is a semi-arid island located 12 km east of mainland Puerto Rico. The southern shore of Vieques is the most exposed to direct hurricane impacts, and has steep shores characterized by sandy beach berms underlain by a beach-rock platform (Woodruff et al., 2008). The region is micro-tidal with a mean tidal range of 0.2 m.
Laguna Playa Grande (LPG) is a 1.5 km long, southeast-facing back-barrier lagoon complex along the southwest shore of Vieques (Figure 2.1). The backside of the LPG barrier beach is fringed with predominantly red mangroves (*Rhizophora mangle*). A 5 m wide inlet lined with riprap, concrete and mangroves connects the lagoon to the ocean. We refer to the lagoon area west of the inlet as the western basin and east of the inlet as the eastern basin (Figure 2.1). The backbarrier of both basins are densely colonized with *R. mangle*. A mudflat commonly covered with microbial mats extends out into the lagoon from these fringing mangroves gradually increasing to a maximum depth of roughly 50 cm during times of high terrestrial runoff or large tides.

LPG’s location on the exposed southern shore of Vieques makes the fringing *R. mangle* forest along Playa Grande particularly susceptible to direct impacts from extreme wind and waves from hurricanes, as storm tracks generally progress from southeast to northwest across the region. *R. mangle* lacks the ability to develop sprouts from older axes or old wood directly after a hurricane due to a lack of dormant buds, which further reduces its resilience to damage from high winds (Tomlinson, 1980; Imbert et al., 1996; Baldwin et al., 2001; Imbert, 2018).

### 2.3.2 Land-use history

Sugar cane plantations and use by the US Navy have caused land cover clearance in LPG’s approximately 9 km² watershed since the mid 1800s (Wadsworth, 1950; Birdsey and Weaver, 1987). The onset of this deforestation began in the mid 1800s as sugarcane farming became the primary economic industry on Vieques, with one of the largest sugar mill located in the vicinity of LPG (Ayala, 2001). The landscape was cleared for cropland and to harvest firewood to support agricultural operations (Ayala,
2001). Estimates of mangrove coverage at Vieques was 446 ha in 1936 and then declined further by 20% to 355.4 in 1979 (Lewis et al., 1981). Furthermore, specific studies commissioned by the US Navy in 1981 identified several problems that caused mangrove forest decay including cattle grazing and trampling, excess erosion and sedimentation from upland deforestation, blockage of tidal inlets, and beach destabilization from military usage leading to excess sedimentation (Lewis et al., 1981).

In LPG sediment cores, Woodruff et al., (2008) observed a ~1840 CE increase in Titanium, a common proxy for terrestrial runoff (Croudace et al., 2006; Wallace et al., 2014), that they associated with this land clearance. In 1943, the US Navy occupied the island and ended the sugarcane industry there. Following this naval occupation, roughly 75% of Vieques was used for land training and ammunition drills with a focus on the eastern portions of the island (Gómez-Gómez et al., 2014) but with continued maintenance of a dirt road to the west along Playa Grade (personal communication, Mike Barandiarian, May 2018). In 1998, construction began for a radar station 300 m upslope from LPG’s eastern basin (Holaday and Langevin, 1998). In 2004, the US Navy ceased most Vieques operations, except for the radar station. Land previously used for military purposes was given to the US Fish and Wildlife Services and the Municipality of Vieques (Gómez-Gómez et al., 2014). Calculated percent vegetative cover indicates reforestation of Vieques has continued to present day, with 83% of the LPG watershed currently forested.

2.3.3 Historical Hurricane Activity

Hurricanes frequently impact Vieques due to its location within one of the most common hurricane tracks in the Caribbean (Xie et al., 2005; Kossin et al., 2010). Since
1900, there have been eight hurricanes that have passed within 50 km of Vieques at Category 2 or greater intensity. Of these only Hurricane San Felipe (1928, Category 5), Hurricane Betsy (1956, Category 2), Hurricane Hugo (1989, Category 4), and Hurricane Maria (2017, Category 4) left deposits at LPG (Brennan, 1991; Rodríguez and Bush, 1994; Donnelly and Woodruff, 2007; Woodruff et al., 2008; Pasch et al., 2019; Radabaugh et al., 2019). Hurricane Irma (2017, Category 5; tropical storm on Saffir-Simpson hurricane wind scale at Vieques) passed north of Vieques twelve days prior to Hurricane Maria, just outside the 50 km radius (Cangialosi et al., 2018).

Donnelly and Woodruff (2007) utilized LPG sediments to reconstruct a 5,000-year record of hurricane deposits and identify centennial to millennial scale periods of more frequent hurricane-induced overwash activity separated by periods of less activity. Using $^{14}$C and $^{137}$Cs dating methods, they associated coarse grained event deposits associated with Hurricanes Hugo, Betsy, and San Felipe. A comparison with additional paleo-climate records suggested that the variability seen in LPG was due to changes in regional hurricane activity associated with variations in the El Niño/Southern Oscillation and West African monsoon. Two step function increases in relative abundance of terrigenously sourced titanium were also observed previously in cores from LPG by Woodruff et al. (2008) and dated to 1350 yrs BP and 1840 CE. The 1350 yr BP increase in titanium was linked to an increase in regional precipitation in Puerto Rico (Nyberg et al., 2001) while the later 1840 CE increase in titanium was found to be associated with historical land clearance for sugar cane production.

The 2017 hurricane season was particularly impactful in Puerto Rico due to the dual impacts of hurricanes Irma and Maria, on September 8th and September 20th,
respectively (Feng et al., 2018; Keellings and Ayala, 2019; Radabaugh et al., 2019), however Hurricane Irma was not a major wind event in southern Vieques as it passed ~90 km to the north of Puerto Rico (Cangialosi et al., 2018). Hurricane Irma resulted in 89 km/h winds (119 km/h gusts), 0.44 m surge in Esperanza, Vieques and 0.55 m surge in Isabel Segunda, Vieques, and 25.4 cm of precipitation over high elevations on the mainland of Puerto Rico (Cangialosi et al., 2018). Hurricane Maria was a stronger storm and made a direct landfall on the south side of Vieques. Damage to meteorological equipment made direct observations sparse, but modeling of wind speeds indicate maximum wind speeds of 257 km/h on the southern coast of Vieques (Rivera-Torres, 2018). Incomplete observations from the more sheltered northern side of Vieques indicated that Hurricane Maria sustained wind speeds reached a maximum of at least 81 km/h, with gusts of 165 km/h on the northern (Pasch et al., 2018).

2.4 Field methods for assessing impact of Hurricane Maria in Laguna Playa Grande

2.4.1 Aerial Imagery and Lidar: Pre and post hurricane

We used high resolution, georeferenced, aerial imagery of Vieques, Puerto Rico from 1930 to 2019 to identify the landward migration and changes in the distribution of vegetative distributions of the LPG barrier beach over time (López Marrero et al., 2017). To track more recent changes in the morphology of the barrier beach, we compared beach elevations as recorded in bare earth lidar Digital Elevation Models (DEMs) from 2015 and shortly after Hurricane Maria in 2018 (OCM Partners, 2022). The 2015 DEM values were subtracted from the 2018 values such that positive (negative) values identify areas of aggradation (erosion). The raster result was clipped to the barrier beach area with the seaward edge defined by the limit of lidar values in the 2015 DEMs. We summarized the
geomorphic changes along 14 beach perpendicular transects to illustrate patterns of erosion and deposition. Transect 8 and 9 are aligned with where the overwash fans broke through the mangroves via footpaths.

2.4.2 Field observations and core collection

Field campaigns were conducted in May of 2018, September of 2019, and November of 2019, with operations that included sediment coring, water level and turbidity sensor deployments, and the gathering of first-hand accounts of storm impacts from local fishermen. Core transects were collected in the larger, western basin where previous work by Donnelly and Woodruff (2007) was conducted, and along overwash deposits in the smaller, eastern basin (See Figure 2.1 for locations). We collected six vibracores to obtain continuous sediment profiles down to ~550 cm. Two of the vibracore sites in the western basin (VC1-B and VC3-B) revisited locations originally cored by Donnelly and Woodruff, (2007) and Woodruff et al., (2008) (LG 12 and LG 4). To trace the extent of the surficial deposits from Hurricane Maria we collected 21 shore-normal and cross-parallel transects of 60 mm diameter push cores ranging from 10-20 cm in depth. Transect A, in the eastern basin, was measured using a transect tape from the start of the fan and all core locations were recorded with a handheld GPS unit. These shallow push cores were manually inserted into surficial sediments and dug out by hand to avoid disturbance of the sediment-water interface. At the location of vibracore sites additional 1 m gouge cores were also collected to provide uncompacted samples of surface sediments (Yellen et al., 2021).
2.4.3 Instrumentation Deployment

Water level loggers were deployed in the inlet and lagoon for 49 days between September 30, 2019, and November 17, 2019, to assess the response of the system to tides, background wave conditions and passing storm/precipitation events (Figure 2.1, Transect A). An additional turbidity sensor was also deployed at the inlet to assess timing of sediment transport within the inlet relative to tidal conditions. All sensors were mounted on 1-m rebar posts that were sunk into the substrate until the sensors with water logger windows set at 1 cm above the sediment water interface at time of deployment. All exposed surfaces around sensor windows were covered in copper tape to minimize biofouling (Gzara et al., 2016; Bushnell, 2018). At both stations Onset HOBO U20L-04 (0.4 cm accuracy) pressure sensors collected data every ten minutes with a third sensor deployed above ground at the inlet bridge to measure local barometric pressure. Pressure measurements were converted to water level by subtracting barometric pressure and dividing by an assumed ocean water density of 1023 kg/m³ and the acceleration due to gravity (9.81 m/s²). To assess variability in suspended sediment concentration an additional RBR Solo3 Tu sensor measured turbidity every 8 minutes at the inlet site (Baranes et al., 2022). This turbidity sensor was equipped with an automated wiper that cleaned the sensor window every 30 minutes to minimize biofouling with the turbidity window mounts ~ 30 cm above the bed.

2.5 Lab Methods for assessing presence of Hurricane Maria

2.5.1 XRF

Collected cores were transported to the University of Massachusetts at Amherst for processing and further analysis. All cores were stored under 4°C refrigeration both
before and after splitting. Split cores were first analyzed on a non-destructive Cox Analytical Systems Itrax X-ray fluorescence (XRF) core scanner to obtain high-definition optical and X-radiograph density images as well as down core relative elemental abundances. The XRF core scanner utilizes a molybdenum tube running at 30kV and 55mA (Croudace et al., 2006), and was run with a ten-second exposure times at measurement intervals of 200 μm. Our study builds off of previous work from the LPG site and other similar coastal lagoon settings where strontium (Sr) has been found to correlate with marine-sourced materials, and titanium (Ti) with terrestrial-derived sediments as it is associated with detrital mineral input (e.g. Croudace et al., 2006; Woodruff, JD, 2009; Cuven et al., 2013; Wallace et al., 2014; Baranes et al., 2016, Ladlow et al., 2019).

2.5.2 LOI and Grainsize Analysis

Cores were subsampled at 1 cm intervals and analyzed for organic content via loss on ignition (LOI). LOI was conducted by subsampling a ~1 cm³ of sediment, which was weighed following drying at 100 °C and then combusted at 550 °C for four hours. The percent mass organic of the sample was calculated by taking the combusted weight divided by the bulk dry weight (Dean, 1974). Combusted sediment was then gently disaggregated with a mortar and pestle for grainsize analysis with a Beckman Coulter LS 13 320 laser diffraction particle size analyzer (e.g. Yellen et al., 2021, Yellen et al., 2016). Each sample was sonicated for 15 seconds to prevent flocculation prior to a 60 second measurement window.
2.5.3 Cesium-137 dating and correlating to previous work in LPG

Previous work by Woodruff et al., (2008) and Donnelly and Woodruff (2007) obtained radiocarbon and \(^{137}\)Cs derived ages from a core in the western basin of LPG, near our VC1-B that they named LG12 (Figure 2.1). We sampled VC1-B for grainsize to compare to LG12 grainsize and age chronology. To correlate the depths of our cores with this previous work across the lagoon, in the eastern basin we sampled the first 20 cm of the 1 m long 60-mm wide GC4-A gouge core for \(^{137}\)Cs (See Figure 2.1 for locations). The gouge core provided uncompacted surface sediment at the location of the VC4-A vibracore, allowing for more accurate depth-to-age derived chronologies. Dried and powdered samples were run using a Canberra GL2020R Low Energy Germanium Detector with Cs-137 activity identified using the 662 keV gamma peak (McKeon et al., 2022).

2.5.4 Percent Vegetative Cover

Aerial imagery from 1936 (period of sugar cane production and post-San Felipe Hurricane of 1928), 1989 (pre-Hurricane Hugo in 1989), 1994 (post-Hugo), 2004 (end of naval operations), 2006, 2010, 2014, 2017 (pre-Hurricane Maria), 2017 (post- Hurricane Maria), and 2022 were analyzed for percent vegetative cover values. In QGIS, areas of vegetation were traced on georeferenced images for each aerial image within a total area in which all images were compared. After total area of vegetation was obtained, the vegetated area was divided by the total area to calculate the percent vegetative cover.
2.6 Results

2.6.1 Pre- and Post- Hurricane Maria Lidar & Field Observations

Figure 2.2 presents a comparison of pre- and post-Maria elevations of the LPG barrier beach obtained via 2015 and 2018 lidar flights within a polygon of study and summarized along our 14 shore-normal transects. Within the evaluated beach area (102,600 m$^2$), 27% eroded and 73% aggraded. In terms of volume, approximately 12,800 m$^3$ of sand was eroded from the front shoreface during the time between lidar missions, compared to roughly 27,600 m$^3$ of sand that aggraded towards the back of the barrier beach. The discrepancy of 50% greater deposition towards the backbarrier compared to erosion along the beach shoreface suggests that additional material was eroded from seaward of shoreline where lidar is available. Field observations in May 2018, provided further confirmation of landward sand transport, including the burial of the road along the berm. Additionally, field observations and aerial imagery identified the formation of two overwash fans within the lagoon along the smaller, eastern section of LPG (Figure 2.3). Pre-Maria aerial imagery and conversations with local fisherman helped to identify the location of fishing access paths where mangrove vegetation had been removed. These paths into the eastern basin corresponded to the locations of post-Maria overwash fan development. No access paths were present along the western portion of the lagoon in 2017, just prior to hurricane Maria (Figure 2.3). In this western portion of the lagoon, several overwash fans were also observed within a wrack line that terminated at the landward extent of the mangrove forest but did not reach the lagoon. Field observations and our pre- vs. post- Maria lidar comparison both indicate that sediment deposition was focused to within this wrackline, likely resulting from its buffering of flows and wave
dampening. Figure 2.4 shows the Maria-induced damages to these mangroves. While some vegetative regrowth had occurred along the barrier by the time of our 2018 field survey, this new vegetation remained mainly vines and other herbaceous, non-woody flora.

2.6.2 Historical Aerial Imagery

Aerial imagery for LPG is available back to 1936, when much of the landscape was devoted to sugar cane production (Figure 2.4). Much of the area was reforested by 1989 when the next aerial imagery is available, followed by more complete reforestation by 2017 (except for the actively maintained Naval radar system 300 m northeast of the site).

A comparison of percent vegetative cover long the LPG barrier support revegetation of the barrier after the end of sugar cane production in 1943, but with additional disturbances to mangrove vegetation following hurricane events (Figure 2.5, Table 2.1). Calculated percent vegetative cover from imagery in 1936 and 1989 (taken just prior to Hurricane Hugo) are 53% and 74%, respectively (Table 2.1). Percent vegetative cover obtained from imagery taken five years after Hurricane Hugo in 1994 decreased to 67%. The percent vegetative cover then increased to 76% in 2010 imagery and to 81% in imagery obtained just prior to Hurricane Maria in 2017. Percent vegetative cover then decreased to 42% in images obtained just after Hurricane Maria (2019), followed by a subsequent increase to 49% by 2021 showing some recovery to the system four years after the event.
Table 2.1 Percent vegetative cover values for the mangrove forest along LPG and mangrove cover estimates (hectares) (* Martinuzzi et al., 2009).

<table>
<thead>
<tr>
<th>Date</th>
<th>Context</th>
<th>Vegetative Cover (%)</th>
<th>Mangrove cover estimates (ha)*</th>
</tr>
</thead>
<tbody>
<tr>
<td>1936</td>
<td>Agricultural Period/Post- Hurricane San Felipe</td>
<td>53</td>
<td>466.2*</td>
</tr>
<tr>
<td>1976</td>
<td>-</td>
<td>-</td>
<td>355.4*</td>
</tr>
<tr>
<td>1981</td>
<td>-</td>
<td>-</td>
<td>367</td>
</tr>
<tr>
<td>1989</td>
<td>Pre- Hurricane Hugo</td>
<td>74</td>
<td>266</td>
</tr>
<tr>
<td>1994</td>
<td>Post- Hurricane Hugo</td>
<td>67</td>
<td>-</td>
</tr>
<tr>
<td>2002</td>
<td>-</td>
<td>-</td>
<td>446</td>
</tr>
<tr>
<td>2004</td>
<td>-</td>
<td>69</td>
<td>-</td>
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<tr>
<td>2019</td>
<td>Post- Hurricane Maria</td>
<td>42</td>
<td>-</td>
</tr>
<tr>
<td>2021</td>
<td>Present</td>
<td>49</td>
<td>-</td>
</tr>
</tbody>
</table>

2.6.3 Turbidity and Water Level

Water level observations from LPG indicate that tides attenuate only slightly when progressing from the inlet sensor to the sensor internal to the lagoon (Figure 2.6A). Higher frequency variability likely associated with wave impacts is also observed on the 5-minute sampling interval for the inlet sensor. Our sensors indicate that the lagoonal system is highly sensitive to along shore directed wind events (Figure 2.6D), with the peak wind event on October 8, 2019 (significant wave height of ~2m), raising water levels to some of the highest recorded during the 49-day deployment. A similar response
is observed for the 2nd highest onshore wind event beginning on Nov. 4th when water levels at LPG are elevated relative to tidal observations at the NOAA Esperanza gauge (Figure 2.6).

A step function drop in turbidity was typically observed at the onset of flood tide at the LPG inlet followed by a gradual rise in turbidity during the subsequent ebb (Figure 2.6G). This tidal trend supports a net export of sediment from the inlet during background conditions. An elevated sequence of turbidity occurs between October 14 – November 4, which could indicate reworking and local trapping of a mobile pool of sediment/debris near the inlet sensor following the Oct 8th wave event. After this elevated turbidity event, turbidity levels dropped to near what was observed prior Oct. 14th suggesting a purging of this material during a subsequent late Oct. wave event also associated with the greatest peak in precipitation over the deployment.

2.6.4 Surface Cores

For this study, we analyzed sediment from cores along two representative transects. Transect A was in the smaller, eastern portion of the lagoon, where isolated overwash was observed following Hurricane Maria. Transect B was in the larger, western basin (Figure 2.1). Photos and depth profiles of grain size and relative Sr abundance are provided in Figure 2.7 for the series of short push cores collected along these two transects.

The Hurricane Maria deposit is evident within surficial sediment from Transect A and is found to thin and fine landward. PC1-A is the Transect A core closest to the back barrier beach and the starting point for our transect. The short core was collected on the exposed part of the overwash fan and did not extend deep enough to capture the entire
Hurricane Maria deposit. Median grain sizes at this location ranged from 0.75-to-1.75 mm and were the largest measured for Hurricane Maria deposition at LPG. Following Transect A landward, PC2-A, located 8 m from the start of the transect, has a sandy deposit of 9 cm thickness, with a sharp lower contact. The median grain size was 1mm, slightly finer than PC1-A.

Sr counts, interpreted as a marine-sourced proxy are clearly elevated within the surficial Hurricane Maria deposit relative to the underlying fine-grained sediments. PC3-A, located 13 m from the start of Transect A, has a 5 cm thick Hurricane Maria deposit, with a very sharp lower contact between the sandy overwash material and the underlying fine-grained sediments, and a median grain size of 0.8 mm. Elevated Sr in this core also closely follows the sandy overwash sediments. GC4-A is located at the site of VC4-A, between PC3-A and PC5-A at 27 m from the start of transect A, and has a median grainsize of 0.23 mm. Cs-137 analysis was done on GC4A, which in combination with grainsize identifies four major hurricanes during the past century (Hurricanes San Felipe, Betsy, Hugo, and Maria) (Figure 2.8). The most landward core in Transect A is PC5-A, located at 32 m from the start of the transect, near a stand of dead mangroves. Surficial sediments exhibit a subtle rise in median grain size relative to underlying material of 0.04 mm and is accompanied by elevated Sr abundances.

Transect B is a revisiting of the previous core transect in the western basin of LPG by Woodruff et al., (2008) and Donnelly and Woodruff (2007). Unlike Transect A, no observable coarse-grained deposit was observed within surficial sediments along Transect B, (Figure 2.7). Further, grain sizes from surficial push cores along Transect B exhibited no detectible coarsening towards the surface. However, Sr abundances exhibit
elevated values in surficial sediments relative to sediments at the 18 cm base of Transect B push cores, particularly for cores PC1-B and P2-B located closest to the barrier where a clear step function Sr rise is observed at 12 cm and 6 cm respectively.

2.6.5 Deposit Dates Based on Cs-137 chronology and LPG Stratigraphy

Cs-137 chronology, in combination with a comparison of sediment records from post-Hurricane Maria sediment cores (this study) and cores collected by Woodruff and Donnelly (2007) are presented in Figure 2.8. Results from Transect B suggest very little deposition has occurred in the western basin in the years since the initial Donnelly and Woodruff (2007) study was published, which is consistent with the net transport of sediment shown with our turbidity measurements (Figure 2.6). The with primary deposition from Hurricane Maria occurred along Transect A in the eastern basin. Figure 2.8B provides a plot of the grain size for the upper 30 cm of push core GC4 located 27 m from start of Transect A, along with its Cs-137 based chronology. The deposit associated with Hurricane Maria is evident at the surface. A deposit at 6 cm located between the Hurricane Maria deposit and the 1963 Cs-137 peak at 11 cm dates to the 1989 Hurricane Hugo. The 1954 onset for Cs-137 is observed at 17 cm and occurs at roughly the same depth as a deposit associated with Hurricane Betsy in 1956. This 1954 onset for Cs-137 at 17 cm provides a sedimentation rate of 0.27 cm/yr which when applied to a deeper grain size peak at 24 cm provides and age consistent with the 1928 San Felipe Hurricane. In the western basin, there is no sandy deposit from Hurricane Maria in any cores collected from Transect B. Peaks in grain size obtained within the top 20 cm of the original LG 12 core along Transect B from Woodruff and Donnelly, (2007) generally match to the depth of grain size peaks from VC1-B core collected in 2018 along the same
transect (Figure 2.9B). The same depth for deposits collected over a decade apart supports limited accumulation over this period.

The two deeper step function increases in titanium previously dated to 1350 yrs BP and 1840 CE by Woodruff et al. (2008) in core LG12 were also observed in sediments from VC1-B in the western basin (Figure 2.9A) and VC4-A in the eastern basin (Figure 2.9C). Patterns in the distribution of coarse grained overwash deposits are also similar between LG 12, VC1-B, and VC4-A including a marked transition from high to low overwash activity at the 1350 yrs BP increase in titanium (red triangles, Figure 2.9), as well as a transition back to increased overwash activity following the later 1840 CE increase in titanium previously linked to the onset of sugar cane agriculture and associated land clearance (Woodruff et al., 2008).

2.6.6 Advective Settling Model

Landward fining and the uniformity of grain size distributions vertically at each sampling location within the Hurricane Maria deposit along Transect A are consistent with sediment predominantly being transported and settling out of suspension during barrier inundation (Wallace and Woodruff, 2014, Wallace and Woodruff, 2020). Based on this assumption we employed the advective settling model previously applied by Woodruff et al., 2008 at Transect A, where a fisherman path allowed for overwash deposition, to assess the model’s effectiveness in back calculating the storm and wave conditions during Hurricane Maria. Flood heights over the barrier are assumed to be the combination of storm surge (the rise in water associated with the storm that does not include waves) and wave run-up (the rise in water elevation produced by waves). The nearest tide gauge at Esperanza, Vieques went offline prior to peak flooding by Hurricane
Maria, however, storm surge from the Yubucoa Harbor station, 34 km to the east, was measured at 1.66 m (Pasch et al., 2019). Maximum wave height from the nearest NOAA buoy (Vieques Buoy 41056) was measured at 6.4 m and with an associated wave period of 14.8 s. From these wave observations and empirical relationships presented by Stockdon et al. (2006) we calculated a maximum wave runup ($R_{\text{max}}$) of 2.01 m. The barrier height along Transect A is approximately 2 m (Figure 2.2). Assuming an approximate elevation of the barrier along Transect A of 2 m (Figure 2.2) and a wave runup and storm surge of of 2.01 m and 1.66 m, respectively, we estimate an inundation depth above the LPG barrier of approximately 1.7 m during the Hurricane Maria event.

The following relationship by Woodruff et al., (2008) predicts the distribution of maximum particle settling velocities ($w_s$) capable of being advected landward a distance of $x_l$ as a function of inundation depth over the barrier ($h_b$) and gravity ($g$):

$$w_s = \frac{(h_b^3 g)^{1/2}}{x_l}$$

Figure 2.10 provides a comparison for: (1) the modeled settling velocity with distance from the barrier based on the above advective settling model and our assumed inundation depth during Hurricane Maria at LPG; and (2) the observed settling velocities at each sampling site along Transect A derived from the $d_{90}$ grain size and the grain settling velocity relationship described in Ferguson and Church (2004).

2.7 Discussion

2.7.1 Preservation of Hurricane Maria deposits

Pre- and post- Hurricane Maria lidar comparisons confirm our field observations that Hurricane Maria transported a substantial amount of sand from offshore onto the
landward side of the LPG barrier beach. Despite observable beach transgression, there was a notable lack of overwash deposits associated with this event beyond the barrier beach and in the backbarrier lagoon. This was particularly true for the western basin where our cores and past work by Donnelly and Woodruff (2007) displayed coarse-grained deposits dating to earlier historical hurricanes. Only two lagoonal overwash deposits were observed at LPG from Hurricane Maria, both restricted to the smaller, eastern basin. Several overwash fans, shorter in extent, can be seen in aerial imagery to the west of the inlet (Figure 2.1, Transect B inset) and further east from the two main overwash fans in the eastern basin, however, these fans did not extend landward beyond the mangrove fringing the lagoon’s barrier (Figure 2.1, 2.2). Lidar differencing showed that the largest elevation changes during Maria occurred in the eastern portion of the barrier beach, while there were also several areas in the west basin where large amounts of sand accumulated within the mangrove forest (Figure 2.2). For example, along transect 7 in Figure 2.2, aggradation within the mangrove-covered section of the beach averaged 0.55 m.

As mentioned previously, the overwash fans that were able to break through the fringing mangrove forest east of the inlet are both located at the end of footpaths cleared through the mangroves prior to the Hurricane Maria event (Figure 2.3E, 2.4A). The fit of the applied advective-settling models to observations extending landward from this overwash fan are reasonable (Figure 2.10) yet excludes the impacts of vegetation (e.g. Castagno et al., 2021). Thus, our results support the current unvegetated foot path at Transect A providing an effective transport conduit through barrier mangroves at LPG during Hurricane Maria. A similarly good fit was observed previously by Woodruff et al.
Percent vegetative cover assessments of the LPG barrier also highlight significant changes to the size and density of its mangrove forest overtime. Just prior to Hurricane Maria the fringing mangrove forest was the most robust it has been in available imagery from the past century (Table 2.1, Figure 2.5). The beach in 1936 is wider than the modern-day configuration, and there is a relatively low amount of vegetation occupying it (percent vegetative cover 53%). The steep beach face and 0.2 m tidal range mean that tidal timing cannot explain changes in beach width. We attribute this low vegetative cover along the LPG barrier to high sugarcane production activity at the time until 1943 and linked mangrove deforestation along the LPG barrier for road clearance, trails, footpaths, and firewood (Martinuzzi et al., 2009; Holdridge, L.R., 1940). The 1840 CE increase in titanium associated with land clearance is also associated with the onset of increased modern overwash activity at all core sites (Figure 2.9), supporting associated mangrove disruption along the LPG barrier resulting in enhanced wave-induced flooding of the lagoon. Overwash fans from the 1928 San Felipe Hurricane, confirmed in the grain size analysis and Cs-137, can clearly be seen extending into the western basin in 1936 imagery (red arrows in Figure 2.3A), despite this earlier 1928 hurricane event being of similar intensity to Hurricane Maria, in which overwash deposition was restricted to
active footpaths to the eastern basin along Transect A (personal communication, Kay, May 2018).

Hurricane Betsy (1956) was a less intense hurricane than Hurricane San Felipe, Hurricane Hugo, and Hurricane Maria, and as such, had less impact on LPG. The observed overwash deposits in the western basin are potentially due to the barrier still being in the recovery stage of land clearance as well as the road more actively being maintained for military operations. The barrier system appears to have been adequately revegetated, however, with a much higher percent vegetative cover value by March 1989, just prior to Hurricane Hugo. The post-Hurricane Hugo imagery from 1994, five years after the event, has a lower percent vegetative cover, but still more than that of the 1936 imagery.

The mangrove forest at LPG pre-Hurricane Maria was likely the most robust it has been in the historical era, which is supported by the 2017 (pre-Hurricane Maria) percent vegetative cover values being the highest observed in the past century. Hurricane Maria caused wide-spread mangrove mortality due to defoliation by high winds and breaking of the mainstems. This resulted in a mass loss of vegetation after the event highlighted by percent vegetative cover values being the lowest in our observed record. This is likely in part because imagery was available just prior and shortly after the Hurricane Maria event compared to the 1928, 1956, and 1989 hurricanes where post-event imagery is either not available or delayed 5-10 years after. The extreme mangrove mortality after events such as Hurricane Maria may lead to more overwash deposits in the western portion of the lagoon in the decades following these events.
Sediment analysis confirms the eastern portion of the lagoon was the only area with sand deposition associated with Hurricane Maria. In the eastern basin, the strontium closely follows the sandy overwash deposit and is a clear indicator of marine inundation associated with the event (Wallace et al., 2014). In the western basin, there is no sediment overwash deposit, however, the strontium signal was elevated in the top 6-8 cm proximal to the barrier, but we do not attribute this to an overwash signal (Figure 2.7B). We believe that the increased strontium is due to the presence of subaerial microbial mats that can trap heavy metals and the presence of nesting sandpipers whose largely marine diets may account for these elevated strontium levels (NOAA and Ridolfi., 2006; Mora et al., 2011; Van Aswegen et al., 2019).

2.7.2 Geomorphic responses to hurricanes and resultant berm alterations

The period of more intense overwash breaching into the western basin whose onset dates to 1840 CE likely provided the additional sediment, and potentially elevation capital, necessary for an observed mangrove expansion into the lagoon (Figure 2.4). The relatively low mangrove expansion and only slight growth from 1994 to 2010 suggests that the system has settled into an equilibrium in terms mangrove distributions along the lagoon shoreline. This left LPG in likely the most stable condition it had been in over the last century prior to Hurricane Maria.

Several studies have investigated the amount of time it takes for mangroves to recover post-hurricane. Recovery times are vary dependent on hurricane intensity, the extent of damage to the trees themselves, the frequency of subsequent events, anthropogenic factors impacting mangrove health in general, and the ability of seedlings to disperse and initiate regrowth, however we postulate a decadal scale of recovery
We observed a gradual monotonic increase in percent vegetative cover at LPG from 1994 to 2017, which suggests the barrier had reached a relative equilibrium just prior to Hurricane Maria. The presence of overwash fans in the 1936 imagery after the 1928 San Felipe Hurricane indicates that most likely the intensity of the damages after the event in combination with the ongoing stressors to the mangroves from the sugarcane industry, overgrazing, and poor land management inhibited recovery after this event (Lewis et al., 1981). For paleo-hurricanes, the cyclical presence of hurricane deposits identified by Donnelly and Woodruff (2008) might reflect an initial large event with subsequent events at a higher frequency due to mangrove mortality and subsequent greater sensitivity of the depositional environment to storm overwash. This would result in groupings of overwash deposits followed by periods of quiescence representing when the system had recovered/revegetated enough to fortify the back barrier from direct, less frequent storm impacts during a more inactive hurricane interval.

This study emphasizes the importance of mangroves in tropical regions for mitigating the effects of hurricanes to coastal ecosystems and communities. When in good standing, a mangrove forest can prevent catastrophic overstepping of the barrier while still allowing for more incremental landward migration. This is evident as we see sand being transported from the forefront of the beach to the landward side after the storm. The important natural buffer that mangroves provide can also have an impact on the sedimentological formation and preservation of hurricane overwash deposits, which has commonly been used to identify the history of hurricanes and the potential future implications for a region. Mitigating human and natural encroachments on mangrove
forests, particularly in coastal areas susceptible to hurricane impact, will allow for the forest to maintain its structure and integrity to better trap sediment and debris, decrease wind and wave energy, and protect coastal communities and ecosystems.

2.8 Conclusions

Hurricane Maria in 2017 was among the most intense historical hurricanes to make landfall near Vieques, Puerto Rico. The event resulted significant erosion along the front beach face of Laguna Playa Grande and deposition along the backside of the barrier. Overwash into the lagoon proper only occurred through preexisting footpaths that cut through mangroves. Early aerial photos document deforestation in the early 1900s associated with sugar cane production, followed by reforestation after the end of agriculture and ceasing of naval activities on Vieques in 2004. Just prior to Hurricane Maria the LPG barrier was more densely vegetated than it had been at any time in the last century due to reforestation. This dense mangrove vegetation resulted in Maria being the first historical hurricane in the last century not to result in extensive overwash into the primary wester basin of the lagoon. Extensive mangrove mortality as a result of the Maria event reduced vegetative cover on the LPG barrier by 40% making the location vulnerable to hurricane induced overwash during the recovery/regrown phase of the barrier. Our results emphasize the vulnerability of back-barrier environments to enhanced flooding following both anthropogenic and event-driven disruptions to this vegetation and highlight the important role of a fringing mangrove forests in flood mitigation along coastlines.
Figure 2.1 (top) Location of cores in Laguna Playa Grande (LPG) with location of LPG and Vieques indicated in the subset map with a yellow star. Red dots indicate cores analyzed in detail along transects A and B. Grey dots are all cores collected. The green dot is the location of LG12 collected previously by Donnelly and Woodruff (2007). The blue lines are the locations of the transects calculated from Lidar (see Figure 2.2) (bottom) Red circles are short push cores and yellow hexagons are our vibracores while the green hexagon is LG12. Green diamonds indicate location of sensors. Lagoon sensor is an Onset HOBO U20L-04 (13 ft) pressure sensor and the inlet sensors are Onset HOBO U20L-04 (13 ft) pressure sensor and a RBR Solo3 Tu sensor.
Figure 2.2 Differencing of lidar from 2015 and 2018, pre- and post- Hurricane Maria. (A) Top image shows the areas of erosion (red) and deposition (blue), with sediment transport direction lagoon-ward. Middle graphs show elevation (top) and elevation change (bottom) for the 14 transects noted in top plot. The yellow star indicates where the overwash fan is located along Transect A. (B) More detailed look at sand movement along eastern (top) and western (bottom) berms with transect locations and overwash fan along Transect A noted (star).
Figure 2.3 Aerial imagery of LPG from 1936, 1989, 1994, 2010, 2017, and 2019 with the percent of the barrier vegetated indicated on each image. (A) red arrows indicate areas of overwash from the 1936 San Felipe Hurricane. (F) yellow arrows indicate areas of overwash from Hurricane Maria.
Figure 2.4 (A) Location of two overwash fans in the eastern basin of LPG along primary footpaths from drone imagery in 2019. Overwash fan A, in the foreground, is our primary study area. The red star indicates the location of the footpath through the mangroves where overwash fan A was able to break into the lagoon. The dashed line is the approximate wrackline trace. (B) UAV image of the LPG barrier. The dashed line traces the wrackline to the western basin of LPG. No overwash fans are observed in the western basin, however, the algal mat near the seaward edge of the lagoon is evident. (C) Broken coconut trees along the berm following Hurricane Maria in May of 2018. The eastern basin of the lagoon is just behind the stand of trees. (D) A wrackline along the mangroves near the entrance of overwash fan A (location indicated with red star in A).
Figure 2.5 Percent of the barrier vegetated for LPG with major hurricanes and land use changes indicated.
Figure 2.6 (A) Water levels for the inlet (blue) and inner, eastern lagoon (orange) covering several tidal cycles. (B) water level for Esperanza, to the east of LPG. (C) Turbidity for the inlet. (D) offshore NOAA buoy wave data (E) precipitation data from the USGS station in Vieques, PR (F) wind speed and wind direction. Red arrows indicate direction of wind. (G) Inlet water levels and turbidity from October 1-October 6 during normal conditions (H) Inlet water levels and turbidity from October 20-October 23 during a high turbidity event.
Figure 2.7 Photos, Sr count (blue), and median grain size (red) for cores along Transect A (Top) and Transect B (Bottom). Each core transects are orientated with the most seaward to the left and most landward to the right. The distance from the start of the transect is indicated next to each core name.
Figure 2.8  (A)Cs-137 and grain size from LG12 (collected previously by Donnelly and Woodruff (2007)) compared to our grain size from VC1-B (blue). The correlations between historical overwash events are noted and indicate no deposition from the Maria event. (B) Cs-137 and grain size for GC4-A, a gouge core near VC4-A in the eastern basin with historical overwash events indicated, as well as the Maria deposit.
Figure 2.9 Down core comparison of Ti (blue) and grain size (black) between (A) vibracore LG12 (collected previously by Donnelly and Woodruff (2007)) and our vibracores from the (B) western basin (VC4) and (C) eastern basin (VC2). 1840 CE and 1350 BP environmental transitions are indicated with grey bars. Arrows indicate correlations between cores.
Figure 2.10 Advective settling model results (dashed line) vs. D90 settling velocities obtained from the Hurricane Maria overwash deposit (black dots) along Transect A (eastern basin of LPG).
3.1 Abstract

Mangroves’ ability to absorb wave energy and trap sediment make them a vital resource for buffering coastal communities from extreme flooding. While mangroves aid in the protection of coastal communities during hurricanes, they are not immune to the damaging effects of extreme events. Here we provide assessments of mangrove changes on the island of Vieques, Puerto Rico following Hurricane Maria in 2017. Initial mangrove damage on the northern side of the island shortly following the Maria event was primarily defoliation, however, in the ensuing months significant die-off was observed. Month long deployments of turbidity sensors and water level loggers following mangrove damage reveal significant tidal attenuation in the tidal channels connecting lagoonal systems to the ocean. We infer this tidal attenuation to reflect frictional effects significant enough to reduce hydraulic conductivity during extreme precipitation. Increased drag in the tidal channel likely increased time required for freshwater export resulting in some back up of elevated water levels in the system following Hurricane Maria. However, the role of this freshening on mangrove health was likely secondary relative to defoliation which is assumed to have served as the primary factor for observed mangrove mortality. Furthermore, ebb-dominated sediment transport suggests enhanced erosion following vegetative die-off. Recovery from such mass die-off events will likely
take decades, during which time the coastline is prone to enhanced flooding and erosion due to devegetation. Results highlight the natural cycle of mangrove die-off, lagoonal opening, and subsequent mangrove regrowth. Such information may be used to aid restoration efforts of mangrove forests and understand the limited protection such systems can provide in the years following initial hurricane-induced disruption.

3.2 Introduction

The ability of mangroves to withstand a wide range of salinities allows mangroves to be one of the most widespread coastal flora in the tropics (Cerón-Souza et al., 2010). While mangroves are generally very hardy trees, they experience stressors from both anthropogenic and natural impacts that lead to a cyclic pattern of establishment, growth, and die-off (Jimenez and Cintron, 1985). High wave energy, shoreline erosion and exposure to high winds all can cause substantial damage to mangroves during hurricane events (Wadsworth and Englerth, 1959; Cintron, G.; Lugo, A.; Pool, D.; Morris, G., 1978; Jimenez and Cintron, 1985; Cohen et al., 2016; Woodroffe et al., 2016). Human-induced stressors include pollution, removal of mangrove trees for shellfish farming, diverting freshwater sources, and land clearance for sugar cane or access paths (Ellison and Farnsworth, 1996; Griswold et al., in review; Paling et al., 2003; Linton and Warner, 2003; Zamprogno et al., 2016). Natural stressors include significant changes in sea level (Ellison and Farnsworth, 1997; Harris et al., 2010; Johnstone et al., 2016; Sippo et al., 2018), invasive fauna, flora, and diseases (Sherman et al., 2001; Biswas et al., 2012), and extreme coastal inundation events such as hurricanes (Smith et al., 2020; Lagomasino et al., 2021) or tsunamis (Dam Roy and Krishnan, 2022). In addition to coastal inundation, hurricanes often bring additional stressors to mangrove systems in the form of freshwater
flooding, enhanced erosion, sedimentation, soil subsidence, and abrupt changes in water level and salinity (Jimenez and Cintron, 1985; Medina, 1999; Biswas et al., 2012; Patrick et al., 2020). In some instances, hurricanes can also reverse hypersaline conditions through flushing of the systems by opening of inlets and thus serve to alleviate certain forms of mangrove stress (Cintron, G.; Lugo, A.; Pool, D.; Morris, G., 1978). While increased flushing in tidal systems can increase mangrove resilience in some cases, in most instances hurricanes reduce mangrove extent. Direct impacts of hurricanes (damage occurring during the high wind or wave event) are usually mechanical in nature and include defoliation, breakage of the mainstem, shedding of bark, or windthrow. Indirect impacts (damage occurring after the storm has passed) include siltation, erosion, flooding, reduction in flushing or stagnation of water, and salinity changes (Wadsworth and Englerth, 1959; Jimenez and Cintron, 1985). Damaging hurricanes can cause mass mortality of mangroves, leaving coastlines and communities more susceptible to erosion and coastal flooding (Cahoon et al., 2003; Loder et al., 2009; Woodruff et al., 2013; Atwater et al., 2014; Branoff, 2019; Radabaugh et al., 2019). Griswold et al. (in review) provides documentation for the impacts of hurricanes on fringing barrier mangroves and resultant overwash sedimentation. This study compliments this work by seeking to explain the lasting impacts of the 2017 hurricane season. We focus on the expansive and densely vegetated mangrove forests around Lagunas El Pobre and Kiani, in the northwestern portion of Vieques, Puerto Rico through both paleo and modern evidence.

3.2.1 Location

Vieques is a 135 km\(^2\) island off the southeastern coast of Puerto Rico composed primarily of sub-tropical dry to moist forest, with fringing mangroves to the south and a
denser mangrove forest in the north (Ewel and Whitmore, 1973). The northern side of the island has more robust mangrove forests, likely due to the lower energy environment and more gradual topography which allows mangroves to thrive and expand, than in the more wave-exposed southern side of the island. Small rocky headlands, with sandy beaches in between, characterize the northern coast of Vieques. The Northeast Trade Winds lead to dominant longshore transport from east to west, resulting in a ~3 km^2 beach ridge and lagoon complex built close to modern sea level at the island’s northwestern corner. We focus our studies on the lagoonal network this area, which is largely at suitable elevation for mangrove recruitment. The two of the lagoons of interest are Lagunas El Pobre and Kiani. This lagoonal network is characterized by an inlet with a main tidal channel that connects the two lagoons. Kiani is the larger inland lagoon and has a greater intertidal area beyond its banks, in which black mangroves persist. An ephemeral freshwater stream also feeds into Kiani from the south. Laguna El Pobre, the primary study area, is the smaller of the two systems. It is also located further seaward than Kiani and connected to the primary inlet via a side channel (Figure 3.1).

Caribbean Islands are susceptible to a variety of natural hazards (Wadsworth and Englerth, 1959; Donnelly, 1996; Ten Brink et al., 2004, 2006; Ten Brink, 2005; Woodruff et al., 2008; Bessette-Kirton, Erin K. et al., 2019). The island of Puerto Rico is situated within one of the most common Atlantic hurricane tracks and is susceptible to multiple hazards including coastal inundation (Palm and Hodgson, 1993; Bush et al., 1996; Donnelly, 2005; Donnelly and Woodruff, 2007), precipitation-induced flooding (Baum and Godt, 2010; Bessette-Kirton et al., 2019), and mass wasting in the form of landslides and cut-bank failures (Colón-Pagán, 2009; Baum and Godt, 2010; Atwater et
al., 2012, 2014; Bessette-Kirton et al., 2019). Hurricanes in this region primarily approach from the south and southeast (Boose et al., 1994, 2004; Feng et al., 2020). This directional approach makes the southern side of Vieques more susceptible to direct coastal impacts from hurricanes. The hurricane approach, along with the steeper coastal shelf results in the southern coast predominately populated by a thin, fringing mangrove forest (Griswold et al. 2022, in review). Separating the north and south sides of Vieques is steep terrain and higher topography that reaches a maximum elevation of 960 ft on the western side. These uplands shield the northwest end of Vieques from intense hurricane-induced winds. This shielding along with generally flatter terrain have allowed for the development of the more robust and extensive mangrove forests on the northwest side of Vieques that are the focus of this study (Cintron, G.; Lugo, A.; Pool, D.; Morris, G., 1978; Proisy et al., 2009; Swales et al., 2015; Woodroffe et al., 2016).

The three main species of mangroves that grow in Vieques, Puerto Rico are the red mangrove (*Rhizophora mangle*), the black mangrove (*Avicennia germinans*), and the white mangrove (*Lancunaria racemosa*) (Pool et al., 1977; Cintron, G.; Lugo, A.; Pool, D.; Morris, G., 1978; Lugo et al., 2007; Tomlinson, 2016). *R. mangle* can filter out salt and provide oxygen to underground roots via prop roots, whereas *A. germinans* removes salt through its leaves, with roots growing vertically out of the ground to breathe. *L. racemosa* does not grow aerial roots unless required due to severe soil anoxia (Tomlinson, 2016). The salinity tolerances of these trees dictate their zonation, with *R. mangle* most abundant along the fringes of lagoons and the coast and tolerating salinities of 60-65 ppt, *A. germinans* further inland and tolerating salinities >90 ppt, and *L.*
*racemosa* found furthest inland, at the highest elevation and tolerating salinities of >90 ppt (Bester et al., 2018, Jaramillo et al., 2018).

### 3.2.2 Hurricane Maria

Hurricane Maria made landfall on September 20, 2017, as a Category 5 hurricane over Vieques, later downgrading to a Category 4 as it reached mainland Puerto Rico (Figure 3.1). The Maria event was the largest hurricane since the 1928 San Felipe Hurricane, and the third costliest hurricane in US history with an estimated total of 90 billion dollars in damage (Hu & Smith, 2018; Keellings & Ayala, 2019; Pasch et al., 2019). Given its rarity, the extreme Maria event provides a unique opportunity to assess the impacts of a major hurricane on mangrove vegetated coastlines in the tropics. While not an abnormally wet hurricane, with 28.10 cm of rain on mainland Puerto Rico and 17.78 cm on Vieques, antecedent soil moisture and resultant runoff were exceptionally high due to the passage of Hurricane Irma on September 8, 2017 (Table 3.1). Prior to both Hurricanes Irma and Maria, the islands soil moisture from the previous two years was 10% higher than the island’s average, and the antecedent soil moisture was 11-13% higher in areas that experiences abundant landslides during Hurricane Maria (Bessette-Kirton, Erin K. et al., 2019; Bessette-Kirton et al., 2020). The passing of Hurricane Irma further increased soil moisture and enhanced landslide risk, leaving the landscape especially susceptible to precipitation related damages when Hurricane Maria hit just a few weeks later.

Table 3.1 outlines the conditions associated with all five major hurricanes that have impacted Vieques in the last century (Mitchell, 1928; Sugg, 1966; Brennan et al., 1990; Brennan, 1991; Cangialosi et al., 2018; Pasch et al., 2019), (Figure 3.1A). These
storms are Hurricane San Felipe (1928, Category 5), Hurricane Betsy (1956, Category 2), Hurricane Hugo (1989, Category 4), and Hurricane Irma (2017, Category 5) and Hurricane Maria (2017, Category 5) (Brennan, 1991; Rodríguez and Bush, 1994; Donnelly and Woodruff, 2007; Woodruff et al., 2008; Miller et al., 2019; Radabaugh et al., 2019).

Table 3.1 Hurricanes within the last century that have impacted Puerto Rico. The individual paths for these storms are provided in Figure 3.1A.

<table>
<thead>
<tr>
<th>Hurricane</th>
<th>Year</th>
<th>Category (1-5)</th>
<th>Sustained Wind (San Juan)</th>
<th>Precipitation (cm) (Vieques)</th>
<th>Precipitation (cm) (mainland PR)</th>
<th>Distance from El Pobre (km)</th>
</tr>
</thead>
<tbody>
<tr>
<td>San Felipe</td>
<td>1928</td>
<td>5</td>
<td>140-160 mph</td>
<td>-</td>
<td>63 (El Yunque)</td>
<td>30 (S)</td>
</tr>
<tr>
<td>Betsy</td>
<td>1956</td>
<td>2</td>
<td>92 mph</td>
<td>-</td>
<td>9.04</td>
<td>45 (S)</td>
</tr>
<tr>
<td>Hugo</td>
<td>1989</td>
<td>4</td>
<td>104 mph</td>
<td>-</td>
<td>7.62</td>
<td>0</td>
</tr>
<tr>
<td>Irma</td>
<td>2017</td>
<td>5</td>
<td>45 mph</td>
<td>7.59</td>
<td>25.40 (El Yunque)</td>
<td>92.6 (N)</td>
</tr>
<tr>
<td>Maria</td>
<td>2017</td>
<td>5</td>
<td>44 mph</td>
<td>17.78</td>
<td>38.10</td>
<td>30 (S)</td>
</tr>
</tbody>
</table>

3.3 Methods

3.3.1 Historical Imagery and GPR

To assess how the tidal channel morphology, inlet location, open lagoonal surfaces, and mangrove forest density have changed within the past century, we utilized a combination of historical maps and imagery dating back to provide a timeline for death, peat collapse, and regrowth of the mangrove forest over the past 80 years (Figure 3.2). To investigate how the location of the tidal inlet has changed, two ground penetrating radar (GPR) surveys were conducted using a MALA X3M GPR with a 500 MHz antenna. The east-west GPR transect followed a path parallel to the northern shore, east of the modern inlet along the northern shore of Laguna El Pobre (Figure 3.1C, 3.3). GPR data was post-
processed at University of Massachusetts at Amherst using GPR-Slice (Theime et al., 2013) to identify paleo-inlet locations. The historical imagery provided a timeline for the growth and death cycle of mangrove forests and better interpret the potential causes of mass die-off that can be applied to the modern mass die-off associated with Hurricane Maria.

### 3.3.2 Percent Vegetative Cover

Aerial imagery from 1936 (post-Hurricane San Felipe, 1989 (pre-Hurricane Hugo), 1994, 2014, 2017 (post-Hurricane Maria), and 2021 were analyzed for percent vegetative cover. In QGIS, areas of vegetation were traced on georeferenced images from each of the years within a total area in which all images were compared. After total area of vegetation was obtained, I calculated the percent vegetative cover by dividing the area of vegetation by the total area.

Several images were not in color, creating an additional challenge for delineating vegetative area. The images from 1936 and 1994 are gray-scale and the image from 1989 is infrared. For these images, green color could not be used to identify locations of living vegetation, however, texture and shading could be employed to identify these locations. While some additional error is included with these images, the error would potentially be an overestimation of percent vegetation.

### 3.3.3 Sediment Collection and Analysis

A 6 cm-wide, 1 m length gouge core was used to collect mangrove peat cores along two shore-perpendicular transects on the eastern and western sides of Laguna El Pobre (Figure 3.1C). The western transect was near a hypothesized abandoned inlet (as indicated by the GPR), while the eastern transects was west of the earliest known inlet.
At the distal end of the eastern transect we also collected a 5 cm-wide, 125 cm long piston push core. Gouge coring was done in September and November 2019 coincident with water level logger and turbidity sensor deployment and recovery. Piston push coring within the lagoon was conducted in May 2018.

Sediment cores were shipped to the University of Massachusetts at Amherst for processing. All cores were stored under 4°C refrigeration both before and after splitting. Split cores were first analyzed using a non-destructive Cox Analytical Systems Itrax X-ray fluorescence (XRF) core scanner at 200 µm resolution to obtain relative abundance of strontium (Sr) and titanium (Ti) and high-definition optical and and X-radiograph density imaging (Croudace et al., 2006). The XRF core scanner was operated with a molybdenum tube running at 30kV and 55mA, with a 10 second exposure time. Our work builds off Woodruff et al., (2008) and Woodruff et al., (2009) where Sr and Ti records derived from XRF scanning were identified as effective proxies for marine and terrigenous sourced material, respectively (Woodruff et al., 2008; Wallace et al., 2014). Their work, and others, indicate that Sr correlates with marine-sourced materials (Woodruff et al., 2008, 2009, 2015; Wallace et al., 2014; Baranes et al., 2016) and Ti correlates with terrestrial sources of sediment and associated detrital mineral input (Haug et al., 2001, Croudace et al., 2006; Cuven et al., 2013; Wallace et al., 2014).

Cores were subsampled at 1 cm intervals and analyzed for organic content via loss on ignition (LOI). LOI was conducted by subsampling an ~1 cubic cm of sediment which was dried at 100 °C and combusted at 550 °C for four hours, weighing the sample wet, dry, and post-combustion, with the change in mass calculated between each step. The percent mass organic (LOI) of the sample was then calculated by taking the
combusted weight divided by the bulk dry weight (Dean, 1974). Combusted sediment was disaggregated gently by hand and with a mortar and pestle for grain size analysis with a Beckman Coulter LS 13 320 laser diffraction particle size analyzer. Each sample was sonicated for 15 seconds prior to a 60 second measurement window to prevent flocculation.

3.3.4 Water Level and Turbidity

To assess tidal attenuation and ebb-flood asymmetry in sediment transport, instrumentation was deployed at the mouth of the El Pobre/Kiani inlet, as well as the entrance to the El Pobre side channel. An Onset HOBO U20L-04 (0.4 cm accuracy) water level logger was installed at both stations (Figure 3.1C). The junction site in the tidal channel (referred to as y-branch for the duration of this paper), had both a water level logger and an RBR Solo3 Tu turbidity sensor equipped with an automated wiper and copper tape to reduce biofouling (Bruel and Sabatier, 2011). The sensors were mounted on a rebar stake driven into the channel sediment to prevent uprooting during tidal movements, storms, or disruption from macro-organisms. Turbidity sensors and water level loggers were set to collect data every eight and ten minutes respectively. Installation of the stations occurred on September 29, 2019, and the sensors were retrieved on November 18, 2019, for a total deployment duration of fifty days, capturing an entire spring-neap tidal cycle.

Precise elevations of each sensor were surveyed using a Trimble RTK GPS, and later used to locate their elevations within the tidal datum. The base station was set up on a known benchmark and referenced back to a known datum at Esperanza, Vieques maintained by the National Geodetic Survey. Points were then collected with an R10
Rover while the base station was running. These points were then corrected using Trimble Business Center software (Maggie, 2015) to identify accurate latitude, longitude, and elevation.

3.4 Results

3.4.1 Inlet Migration and Mangrove Forest Density

Severe defoliation was observed during the initial post-Maria field campaign in May 2018, with some regrowth of *R. mangle* leaves along the edge of the lagoons, however, there was little to no recovery of *A. germinans* leaves. No wracklines, broken trunks, or windthrow in the mangrove systems was observed. During the 2018 field season, while much of the mangrove trees were defoliated, there was still an abundance of fauna including crabs, fish, and iguanas, and mangroves located in the channel between El Pobre and Kiani were alive. However, during subsequent field work in November 2019 a striking increase in dead mangroves and a lack of fauna was observed. The channels, which previously had had a mangrove canopy, had widened and become clogged with large woody debris in some sections.

From satellite images, the percent vegetative cover was calculated from before and after major disruptive events to quantify the change in vegetation (Figure 3.2A, Table 3.2). Ongoing research at Laguna Playa Grande (LPG), a lagoon on the southern side of Vieques, shows that in 1936 there was considerably less vegetation due to the ongoing agricultural activities from the sugar cane industry (Ayala et al., 2001, Griswold et al., 2022, in Review).

The same impacts from the sugar cane industry were not observed in the northwestern portion of Vieques, however, there was a greater impact from the US
Navy’s operations on the island. Starting in the 1940’s the western part of Vieques was used as a munitions depot and military training location (Ayala, 2001). The hashed area outlined in red in Figure 3.1 shows where a portion of the mangrove forest was used as a disposal site for empty containers of lubricants, oil, solvents, paints, rubble, inert concrete-filled practice bombs, and empty shell casings by the Navy in the 1960’s and 1970’s (Naval Facilities Engineering Command, 2018). This material, as well as the general solid waste and contaminated soil, was removed during a removal action in 2009. The military interference in the region could account for the loss of vegetation surrounding the lagoon observed between 1936 (Figure 3.2B) and 1989 (Figure 3.2D) aerial imagery. Post-Hurricane Hugo, and after the mangrove disposal site was no longer in use, the aerial photos show a continuous recovery of the mangrove forest up through 2014 (Figure 3.2F). Percent vegetative cover obtained from imagery taken just following Hurricane Maria showed the lowest percentage vegetated observed from historical imagery and is a reflection of mass mortality caused by the event (Figure 3.2G). By 2021 the percent vegetative cover increased to slightly more than what was observed post-Maria supporting some recovery of the system during the intervening four years. This appears to largely be due to *R. mangle* repopulating the edges of the lagoons, as little *A. germinans* regrowth had been observed as of 2021 (Figure 3.2H).
Table 3.2 Percent vegetative cover for Lagunas Arenas, El Pobre, and Kiani and a comparison with percent vegetative cover from Laguna Playa Grande, Vieques, PR (Griswold et al., 2022, in Review).

<table>
<thead>
<tr>
<th>Date</th>
<th>Vegetative Cover (%) Northwestern Mangrove Basin</th>
<th>Vegetative Cover (%) Playa Grande Barrier</th>
</tr>
</thead>
<tbody>
<tr>
<td>1936</td>
<td>76</td>
<td>53</td>
</tr>
<tr>
<td>1989 (Navy training, pre-Hurricane Hugo)</td>
<td>44</td>
<td>74</td>
</tr>
<tr>
<td>1994</td>
<td>67</td>
<td>67</td>
</tr>
<tr>
<td>2010</td>
<td>67</td>
<td>76</td>
</tr>
<tr>
<td>2014</td>
<td>73</td>
<td>77</td>
</tr>
<tr>
<td>2018 (post-Hurricane Maria)</td>
<td>32</td>
<td>42</td>
</tr>
<tr>
<td>2021</td>
<td>45</td>
<td>49</td>
</tr>
</tbody>
</table>

Comparison of aerial imagery from 1936, 1989, 1994, 2014, 2017, and 2021 (Figure 3.2A-G) also reveal changes in tidal channel configuration over the past 85 years. A USGS topographical map from 1951, while lacking the specificity of an aerial image, shows a westward movement of Kiani and El Pobre’s shared outlet to the sea (Figure 3.2C). The GPR transect collected parallel to the coast in September 2019 is consistent with westward inlet migration, with at least three paleo inlet locations indicated (Figure 3.3). The cause of the westward stepping of the inlet is not known, however, there is considerable westward longshore drift in this region, which can be seen through changes to the shoreline along Laguna El Pobre and the Escollo de Arenas offshore sandbar. Tidal channels connecting Lagunas El Pobre and Kiani, however, appear to have remained more stable over the period of available aerial imagery, with only the inlet connecting El Pobre to the open ocean changing location.

### 3.4.2 Water Levels, Turbidity, and Sediment Cores

Water level loggers indicate there is tidal attenuation between the inlet and the y-branch, as shown by a significant landward decrease in the predominant M2 harmonic
from 0.0801 m to 0.057 m (Figure 3.4D, 3.4E, Table 3.3). This attenuation supports tidal channel restrictions, channel curvature, and mangrove growth on the channel edges imposing significant drag for water moving through the system.

Table 3.3 Tidal harmonics for the Inlet and Y-branch of the El Pobre tidal system in meters.

<table>
<thead>
<tr>
<th></th>
<th>M2</th>
<th>M4</th>
<th>S2</th>
</tr>
</thead>
<tbody>
<tr>
<td>Inlet</td>
<td>0.0801</td>
<td>0.00503</td>
<td>0.0163</td>
</tr>
<tr>
<td>Y-branch</td>
<td>0.057</td>
<td>0.00342</td>
<td>0.0101</td>
</tr>
</tbody>
</table>

During typical hydrologic conditions, turbidity tended to increase during ebb tide, when water levels in the tidal creek were declining. Conversely, turbidity decreased during the flood tide when water levels were increasing (Figure 3.4D). This interplay during water levels and turbidity dynamics is consistent with sediment export from the lagoons to the ocean during these typical hydrologic conditions. A high energy wave event during Oct 28 – November 4, 2019, provided the opportunity to observe how tidal inlet and turbidity dynamics changed when marine turbidity and water levels increased (Figure 3.4C, 3.4E). Sediment cores collected along all transects did not show a sandy deposit or enrichment in marine derived Sr at the surface, which would be indicative of marine source material. Rather, the predominant sediment composition of this surficial material was muddy, mangrove peat (Figure 3.5). Thus, while some marine sourcing of sediment was documented during the small wave event in late October 2019, no overwash deposits or significant marine sourced sedimentation associated with Hurricane Maria was observed in cores collected from El Pobre. However, a thick sandy deposit enriched in Sr is observed at ~20 cm sediment depth (Figure 3.5).
3.5 Discussion

3.5.1 Mangroves

Neither subaerial gouge cores from the mangrove forest on the north seaward-facing side of the Laguna El Pobre nor piston push cores collected from the lagoon contain evidence of a sandy deposit at the surface that would correspond to Hurricane Maria overwash. An older sandy deposit at ~15-30cm in depth throughout the cores exhibits elevated Sr and larger grainsize indicative of marine sourcing during a high energy event (Figure 3.5). This deposit does not correspond with any known hurricanes or tsunamis, however, historical imagery and nautical maps, in combination with GPR tracks, indicate that this sandy deposit could correspond to a former inlet location (Figure 3.3).

Historical maps and imagery suggest that the inlet in this tidal system does not open and close, as is common with inlets in other systems with more exposed coastlines that experience higher wave activity. The inlet has exhibited a steady western migration during 1936 to present associated with the westward local longshore drift. Historical imagery shows stepwise shifts westward of the inlet from 1936 to 2020 within the bounds of Laguna El Pobre (Figure 3.3). In addition to inlet migration, percent vegetative cover analyses show the mangrove forest has undergone anthropogenic induced changes over the past 85 years, primarily through Naval activities in the 1960’s and 1970’s (Ayala, 2001) (Figure 3.2). In 1936 the percent vegetative cover value was relatively high (76%), despite having more open lagoons, primarily because the forest density was greater (Table 3.2). This was also before the Navy began its operations on Vieques in 1945, so the impacts from training and waste disposal in the 1960’s and 1970’s had not yet begun.
The 1989 percent vegetative cover value of 44% highlights the impacts of the disposal site located at Laguna Arenas (Table 3.2). By 1994, however, the system had begun to recover because the Navy operations began to taper off in the area and were finally abandoned by 2004 (Ayala, 2001). From 1994 until Hurricane Maria, a steady recovery of the system was observed. Following Hurricane Maria the mangrove cover in the region experienced wide-spread mortality, as evident by the percent vegetative cover decreasing from 73% (pre-Hurricane Maria) to 32% (post-Hurricane Maria) (Table 3.2). Some recovery has occurred along the edges of the lagoons and near Laguna Arenas in the five years post-Hurricane Maria, however, most of this has been within the area of *R. mangle* and not *A. germinans*, which makes up most of the forest around Lagunas El Pobre and Kiani (Figure 3.2).

Aside from the anthropogenic influences on the system, the length of the inlet appears to have an impact on the trapping of sediment and mangrove seeds that can lead to the replenishing of the forest. As the inlet moved further west, the inlet channel system became more complex and sinuous, leading to greater tidal attenuation, potential trapping of sediment, and a better environment for mangrove seeds to germinate and take root (Furukawa et al., 1997). This created a system that is connected to a more diverse hydrologic area consisting of multiple lagoons and channels, which allows for more sediment flux and subsequent sediment availability for mangroves to expand (McLachlan et al., 2020).

Hurricanes can have major impacts on mangrove forests, both directly and indirectly. The first aerial image in 1936 indicates the lagoonal system was more open than it was in 2018, and the mangrove forest was only fringing and not as robust as it was
prior to Hurricane Maria. The 1928 San Felipe Hurricane, while south of the island, could have still been damaging enough to cause a Hurricane Maria-type destruction to the mangroves forest in the El Pobre and Kiani tidal system. The San Felipe Hurricane was the largest hurricane to directly impact Puerto Rico prior to Hurricane Maria and caused catastrophic damage to most of Puerto Rico with sustained winds of 144 mph and rainfalls of 63 cm on the mainland in the nearby El Yunque mountains in Northeast Puerto Rico (Table 3.1) (Mitchell, 1928; Woodruff et al., 2008). If this hurricane was the catalyst for mass mangrove mortality, similar to Hurricane Maria, then the start of the mangrove death cycle was in 1928, or within a year or two after the event depending on the scope and intensity of the indirect damages. Current studies in Florida show that recovery of mangroves is variable, however, if there is not a peat collapse or transition into a mudflat due to the mortality (Cahoon et al., 2003; Smith et al., 2009), then recovery after hurricanes could be in the timescale of years to decades (Imbert et al., 2000; Ward et al., 2006).

The El Pobre and Kiani system, however, was potentially not allowed to recover naturally due to anthropogenic impacts including the mangrove disposal site and subsequent remediation by the navy, which included the removal of contaminants and debris (Ayala, 2001; Naval Facilities Engineering Command, 2018). The system was therefore excluded from the ideal recovery process, thus prolonging the recovery time. With subsequent hurricanes (Hurricane Betsy, 1956 and Hurricane Hugo, 1989) the mangrove forest couldn’t begin its recovery until the early 1990’s when Naval activity diminished, and the system was allowed to regrow with fewer anthropogenic influences. Table 3.2 and Figure 3.2 outline the steady recovery in vegetation from the decrease in
Naval activity through 2017 when Hurricane Maria occurred, with percent vegetative cover values going from 44% in 1989 to 73% in 2014. After hurricanes that result in mass mortality events, recovery times are highly variable based on the extent of indirect impacts after the storm, frequency and intensity of subsequent events, the forest structure, and climatic events (Feller et al., 2015; Sippo et al., 2018). While rates of recovery are relatively unknown, particularly at site-scales, the evidence put forth here indicates a recovery interval after catastrophic death to be between 2-3 decades if there are not additional negative stressors on the forest.

Defoliation can cause severe stress and damage to mangrove trees, with different species experiencing independent and variable impacts from different stressors. The two primary species of mangroves present in the El Pobre and Kiani lagoonal system are *R. mangle* (red mangrove) and *A. germinans* (black mangrove). *R. mangle* are the most resilient to defoliation as their leaves are not integral to the expulsion of salt or oxygen exchange, as is the case with *A. germinans* (Wadsworth and Englerth, 1959; Jimenez and Cintron, 1985). As a result, in May 2018, we saw some regrowth of *R. mangle* leaves along the edge of the lagoons, however, there was little to no recovery of *A. germinans* leaves.

Breakage of the mainstem, shedding of bark, and windthrow or uprooting would all cause irreversible damage to mangroves, regardless of the species, as trees would either be physically destroyed or not have the available pathways for nutrient and water transport. Higher magnitude wind and wave events can cause debris to be trapped in mangroves, leading to additional force on the branches and trunk by wrack lines which
can cause breakage and death, as observed at Laguna Playa Grande on the southern side of the island (Griswold et al., 2022, in review).  

### 3.5.2 Flooding and Stagnation

All mangroves are susceptible to stress from flooding, and ultimately death if flooding persists, regardless of salinity. However, not all species of mangroves are as susceptible to flooding damage as others. *Avicennia germinans* (i.e., black mangroves) are more susceptible to long-term damage from flooding than *Rhizophora mangle* (i.e., red mangroves), which if persistent enough can lead to mortality in *A. germinans*. The vertical prop roots that serve as breathing tubes, if submerged or buried by sediment, prevent *Avicennia* from respirating, which leads to anoxia. The area below the pink dashed outline in Figure 3.1C indicates the zone of *A. germinans* where a majority of the mangrove death occurred in the vicinity of Lagunas El Pobre and Kiani.

Table 3.1 indicates the amount of precipitation associated with the major hurricanes in the past century. Hurricane Hugo in 1989, the last major hurricane prior to the 2017 hurricane season, was considered a “dry hurricane” with ~7.62 cm of rain accumulation in San Juan, PR (Brennan et al., 1990; Brennan, 1991; Rodríguez and Bush, 1994; Pasch et al., 2019; Keellings and Hernández Ayala, 2019). In comparison, during Hurricane Maria, there was a maximum of 89-96 cm of rain on mainland PR and 13-18 cm of rain on Vieques during Hurricane Maria, that fell on soil already saturated due the passage of Hurricane Irma just a few weeks prior (Cangialosi et al., 2018; Pasch et al., 2019).

Based on data from the USGS Guayanes (50085100) stream gauge, the closest gauge on the mainland and with similar precipitation, similar physiography, and
antecedent conditions to Vieques, we assume ~80% runoff ratio during Hurricane Maria (U.S. Geological Survey, 2016). The 80% runoff was calculated by multiplying the total discharge at the Guayanes stream gauge during the event (8.09 E 06 m$^3$) by the total volume of precipitation in the Guayanes watershed (1.03 x 10$^7$ m$^3$). This high runoff ratio was most likely due to soil saturation from Hurricane Irma, and the preceding higher than normal antecedent soil moisture in the years prior (Bessette-Kirton, Erin K. et al., 2019; Bessette-Kirton et al., 2020). Hydrographs from Ceiba on the mainland of Puerto Rico show that peak storm surge, the timing of high tide, and peak precipitation were all nearly contemporaneous. Additionally, the National Hurricane Center reported an increase in the ocean level after Hurricane Maria of 0.61 m offshore of northern Vieques (Pasch et al., 2019). This temporary increase in sea stand, which lasted approximately 12 hours, could have provided a backwater effect preventing Laguna Kiani from efficiently draining.

Considering the amount of precipitation multiplied by the watershed area and a derived runoff ratio of 0.8 we estimate a total freshwater runoff volume of roughly 42,000 m$^3$. Dividing this volume of water inputted from runoff in the watershed by the lagoon area we get a ~1m increase in the water level within Lagunas El Pobre and Kiani. The extent of A. germinans death is at ~0.2 m PRVD02, which is below the ~1 m PRVD02 level of flooding, potentially indicating an additional hydrologic stressor that contributed to observed mass mortality (Figure 3.1A)(OCM Partners, 2022). The observed tidal attenuation within the system, from both the channel shape and frictional forces due to the mangrove roots, combined with the elevated offshore water levels, could have led to a backup of water within the lagoons. Increased precipitation from Hurricane Maria, and increased runoff from the saturated soil leading up to the event,
would have caused increased flooding, causing increased oxygen stress for *A. germinans* who respirate through vertical prop roots. The hydrodynamic conditions associated with flooding would have led to reduced flushing, producing stagnation within the mangrove forest that would have provided an additional stressor to mangroves already stressed by defoliation, even if this flooding was short-lived.

### 3.5.3 Changes in Salinity

The factors that impact salinity in mangroves can vary depending on climate and vegetation type (Robert et al., 2009). Most mangrove research has focused on the impacts of hyper salinity due to arid conditions, lack of runoff, evaporation, changes in precipitation patterns, and stagnation of water (Cintron, G.; Lugo, A.; Pool, D.; Morris, G., 1978; Medina, 1999; Robert et al., 2009; Jaramillo et al., 2018). While mangroves can tolerate high salinities (*R. mangle* can tolerate salinity of 60-65 ppt whereas *A. germinans* and *L. racemosa* can tolerate salinities of >90 ppt), excessive salination for extended periods of time without adequate flushing can cause stress and ultimately death. Rapid evaporation can cause hyper salinity of the soils, which also inhibits growth and can cause excess stress. Typically, the addition of freshwater into the system is beneficial for mangroves, and in the case of increased freshwater counteracting the effects of potential hyper salinity, hurricanes may benefit mangrove forests.

Jimenez and Cintron (1985) address that mangrove systems, while cyclic in nature and resilient, are susceptible to tree mortality after significant changes in environmental conditions. An ephemeral stream located in the southeastern corner of Laguna Kiani is the primary source of freshwater to the system. During high precipitation events, such as Hurricane Maria, this stream inundates the system with freshwater. While freshwater
input is, in general, beneficial to mangroves, particularly if they were under hyper saline conditions prior, any sudden change to the system can cause stress (Jimenez and Cintron, 1985). In conjunction with the defoliation and flooding, this rapid drop in salinity could potentially have been an additional stress leading to observed mangrove mortality.

3.6 Conclusions

Mass mortality of the mangrove forest in Lagunas El Pobre and Kiani was observed following Hurricane Maria in 2017. While there were several large overwash deposits associated with the event on the southern side of the island at Laguna Playa Grande, the topography and southern approach of Hurricane Maria shielded the northwestern corner of Vieques resulting in no overwash deposit. While no overwash was detected, severe defoliation, in conjunction with flooding and stagnation, caused the mortality of the mangroves in Laguna El Pobre after Hurricane Maria. The modern inlet, being more sinuous than historical inlets, restricts flow of water both in and out of the system, which inhibits the ability of the system to drain following high discharge and therefore trapping water after the large precipitation events. The percent vegetative cover analysis and historical observations indicate this system is capable of rebounding from this mortality within several decades, barring any anthropogenic interference with the system. Mangrove death in this part of Vieques, is delayed due to more indirect impacts to the mangrove forest. During the time period of recovery, the system is more susceptible to damage from intense hurricanes, which would further delay the recovery period.
Figure 3.1 Location of Lagunas El Pobre and Kiani in Vieques. (A) Lidar and watershed delineation of the northwestern mangrove basin. Red dashed line is the polygon for the percent vegetation calculations. White solid line is the watershed for Laguna Kiani. Orange hashed zone indicates the location of mangroves in the area (B) Hurricane tracks that impacted Puerto Rico in the past century. Yellow star indicates location of NOASS buoy (C) Locations of cores, water level logger and turbidity stations, GPR transect (white dashed line), and zone of flooding (area below the pink line). Areas of mangrove death around the lagoons are where the vegetation is brown.
Figure 3.2 Aerial imagery from 1936 (B), 1951 (topographic map) (C), 1989 (D), 1994 (E), 2014 (F), 2017 (G), and 2021 (H) with the percent vegetative cover indicated on each relevant image. Modern inlet location is indicated with a blue arrow and the 1936 inlet location is indicated with a red arrow. Zones of prolonged mangrove death after Hurricane Maria can be seen as the brown areas in the 2021 image. (A) shows the percent vegetative cover overtime for the lagoons. Major hurricanes are noted with vertical dashed lines and the shaded area between 1960 and 1970 indicates when the mangrove disposal site was in use by the Navy. Remediation for this area was in 2009. The location of the disposal site is indicated in Figure 3.1.
Figure 3.3 Ground Penetrating Radar (GPR) profile acquired from path along Laguna El Pobre (refer to Figure 3.1 for location), where two paleo inlets are identified.
Figure 3.4  (A) Water level data for the inlet (blue) and y-branch (green). The black box (D) shows the location of panel D. Grey bars indicate the flood tide (B) Turbidity data for the y-branch showing predominantly ebb dominated turbidity. (C) Significant wave heights from the NOAA buoy north of Vieques. The black box (E) shows the location of panel E. There are two wave events – one in early October and another in late October-early November. (D) Water level data (top) for the inlet and y-branch during typical hydrologic conditions. Higher turbidity is during the ebb tide (bottom). (E) Water level data (top) for the inlet and y-branch during an offshore wave event. Higher turbidity is during the flood tide (bottom).
Figure 3.5 Western transect of cores for most landward to most seaward (locations shown in Figure 3.1). Cores were taken at the prior inlet location abandoned in 1936. GC8 is just within the mangrove forest on the lagoon edge and GC12, the most seaward core, is just outside the mangrove extent. For each core location a photo of the core, XRF-derived elemental abundances of Sr (red) and Ti (blue) along median grain size (i.e., D50) is provided.
Figure 3.6 (A) R. mangle, (B) A. germinans, with vertical pneumatophores, (C) L. racemosa, (D) mangrove death looking from the road between Laguna’s Kiani and El Pobre towards Laguna El Pobre. Some regrowth of the R. mangle can be seen, with most of the death being the A. germinans. (E) Mangrove death looking southeast across Laguna Kiani. Monta Pirata is in the distance.
4.1 Abstract

Coastal salt marshes are one of the most important buffers of coastal settings from the threats of accelerated sea level rise and storms. It is vital for advising policy and future climate research to understand what factors are most important for salt marshes to keep pace with sea level rise. One of the major factors for ensuring salt marsh resilience to sea level rise is the supply of clastic sediment, however, there is limited data on the sourcing, timing, and mechanisms of sediment delivery. Here we present a regional assessment sediment delivery to marshes through analysis of instrumental data from turbidity sensors and water level loggers, as well as accumulation of clastic sediments on the marsh surface through sediment traps. The instrumentation and sediment traps were deployed synchronously in the spring, summer, and fall from 2020-2021. Three representative marshes were chosen to cover the wide variety of salt marsh systems in the region from the microtidal marshes south of Cape Cod, MA to the mesotidal marshes in northern Maine: Sprague (ME), Wells (ME), and Westport (MA). Due to differences in marsh systems throughout the region, a heterogeneity in sourcing and timing of sediment supply is observed. The Sprague site that is located proximal to the Kennebec River exhibited the highest sediment delivery during the spring freshet, indicating fluvial sourcing of sediment. Wells exhibited highest sediment delivery during the Halloween
Storm of 2021 with complimentary wave and river discharge data supporting sediment contributions via a combination of both fluvial and marine sources. Westport exhibited the highest turbidity levels during the summer, coinciding with increased crab activity supporting internal sourcing of sediment via biological remobilization of the marsh platform.

4.2 Introduction

Salt marshes are vital for numerous ecosystem services (UNEP, 2006) including natural buffers of the coast from storms by reducing impacts of erosion and flooding from increased wave action (Möller et al., 2014; Temmerman et al., 2023), filtering nutrients from upland runoff (Sousa et al., 2010), habitats for juvenile fish, crustaceans, and birds (Shenker and Dean, 1979; Boesch and Turner, 1984; Hughes, 2004; Whitfield, 2017), and the sequestration of carbon (Chmura et al., 2003; Kayranli et al., 2010; Mitsch et al., 2013). For salt marshes to keep up with sea level rise and continue to provide these ecosystem services, biophysical feedbacks between inorganic sediment supply (Redfield, 1972; Baranes et al., 2022), plant growth (Patterson and Mendelssohn, 1991; Johnson et al., 2016), flooding (Morris et al., 2002), and nutrients (Deegan et al., 2012, 2012) need to be maintained. However, little is known about the sourcing of external sediment, including the availability, timing of delivery, and distribution of inorganic sediment to marshes. For northeastern marshes, the complex post-glacial setting, and human-induced alternations to the landscape leads to uncertainty on the sources and mechanisms of sediment delivery (National Academies Report, 2018; Ganju, 2019; Wasson et al., 2019).

While some studies suggest increase in sediment supply due to early European land clearance followed by a reduction due to damming and reforestation (e.g. Kirwan et
al., 2011; Weston, 2014), other more recent work indicates less detectible increases to no
substantial variance in sediment yields from early deforestation (Cook et al., 2020;
LeNoir et al., 2022) and minimal reductions in sediment supply due to damming
particularly for local rivers associated with smaller watersheds (Yellen et al., 2018;
Ralston et al., 2021). Recent work by Baranes et al., 2022 has highlighted the importance
of not just upland sources of sediment, but also of marine contributions of inorganic
sediment from glacial features such as drumlins along the barrier and potentially finer
grained glacial lake or marine bed deposits offshore that are being reworked. There is a
growing appreciation for the role of sediment in allowing marshes to keep pace with
future sea level rise (Kirwan and Megonigal, 2013), however, heterogeneity in the
sediment sourcing and mechanisms of delivery in the Northeast limit future resiliency
assessments for marsh systems in the region.

There are three main sources of inorganic sediment delivery to coastal wetlands
(1) fluvial sourcing from upland runoff and river transport (external to the marsh)
(Meade, 1982; Syvitski et al., 2005), (2) offshore sediment delivery during flood tides
and storms from offshore deposits or from eroded coastal deposits (external to the marsh)
(Baranes et al., 2022), and (3) internal sources from the erosion of mudflats and
cannibalization of marsh platform via mechanisms that include wave erosion and
bioturbation (Holdredge et al., 2009; Hopkinson et al., 2018; Ganju, 2019). With respect
to timing for larger river systems, fluvial sediments in the Northeast are commonly
delivered during high discharge associated rain and snow melt events in the spring, i.e.
spring freshets (e.g. Woodruff et al., 2001), as well as more episodically during extreme
precipitation associated with tropically derived disturbances that impact the area in the
fall (e.g. Yellen et al., 2014). Recent work has shown that in smaller estuaries marine sourced sediment is a greater influence, however, the extent to which this applies broadly throughout New England needs to be investigated further (e.g., Baranes et al., 2022; Hopkinson et al., 2018). In this chapter I will provide a regional assessment of sediment delivery to marshes through analysis of instrumental data from turbidity sensors and water level loggers, as well as accumulation of clastic sediments on the marsh surface through sediment traps.

4.3 Field Sites

The location of the three field sites for this study are provided in Figure 4.1. With our field sites we sought to capture both meso- and micro-tidal systems, as well as marshes associated with both large and small river systems. Our most northern site, Sprague River Marsh, in Phippsburg, ME, is proximal to the Lower Kennebec and Androscoggin Rivers which have a watershed area of ~24,000 km². Marsh sites in Wells, ME, a back barrier marsh is split between two bar-built, fluvial estuaries -the Little River to the north, and the Webhannet River to the south, with significantly smaller watersheds of 81 km² and 36 km², respectively. The watershed for the Little River cuts through the Presumpscot formation, a marine clay unit from the last glacial transgression (Borns and Hagar, 1965; Dickson and Johnston, 2015) (Figure 4.4). The Wells marshes extend between the NOAA Wells National Estuarine Reserve (NERRS) and US Fish and Wildlife Service Rachel Carson National Wildlife Refuge. NERRS has additional instrumentation deployed at the mouth of the Little River which includes water depth (m), salinity (psu), and turbidity (ntu) from the channel at the mouth of the Little River. Both Sprague and Wells are meso-tidal and located within the reaches of the Gulf of
Maine. At the NOAA Portland Tide Gauge (8418150), located between Sprague and Wells, the average tidal range is 2.78 m, and the spring tide range is 5.45 m (NOAA).

The marsh at Westport, our most southern field site, is a micro-tidal salt marsh located at the end of Buzzards Bay, south of Cape Cod. Our sample sites are located along the East Branch of the Westport River, which has a watershed area of 260 km².

Nor’easters and extra-tropical storms impact all three field sites and serve as the predominant driver of local wave activity in the Gulf of Maine (Xie et al., 2017; Staudinger et al., 2019) and Buzzards Bay (MA) (Zhang et al., 2001, 2020; Chen et al., 2021). Westport, located just south of Cape Cod experiences more severe impacts from tropical cyclones, however, these are less common than nor’easters (Boldt et al., 2010).

The vegetation across all marshes is similar, with the low marshes dominated by tall form *Spartina alterniflora*, and high marshes is primarily *Spartina patens* and short *S. alterniflora*.

**4.4 Methods**

**4.4.1 Field Measurements**

At each site a series of channel normal transects with sediment traps and sensors were deployed during the spring, summer, and fall of either 2020 for Westport and 2021 for Wells and Sprague. Sensors and traps were not deployed during the winter due to limited access and disturbances from ice rafts, snow, and ice. At these deployment locations we also collected gouge cores and bulk density samples. The general spacing of sediment traps along each transect relative to the channel were 3 m, 10 m, 40 m, and 100 m. For Westport the first sediment trap along transect 2 (WP-EB 002) and transect 4
(WP-EB 004) were 7 m and 10 m, respectively. Table 4.1 outlines the distribution of sample locations on each transect at our three field sites.

At all sites sensors and sediment traps were measured with a Real Time Kinematic (RTK) GPS to obtain lat/lon location and elevation at 8 mm and 15 mm accuracy, respectively (Trimble, 2019). Four RTK points were taken at each sediment trap location and averaged. RTK measurements were taken at the middle of the sensor window for the water level logger, and at the marsh surface for each of the sediment traps.

Table 4.1 Sample locations across all field sites. Grey boxes indicate where sample locations are on each transect and grey boxes with black boarders indicate turbidity and pressure transducer locations for each site. X indicates where crab traps were deployed during the 2021 field season for SPG and WEL and 2020 field season for WP-EB.

<table>
<thead>
<tr>
<th>Distance from channel/creek (m)</th>
<th>SPG 001</th>
<th>SPG 002</th>
<th>SPG 003</th>
<th>WEL 001</th>
<th>WEL 002</th>
<th>WEL 003</th>
<th>WEL 004</th>
<th>WEL 005</th>
<th>WP-EB 001</th>
<th>WP-EB 002</th>
<th>WP-EB 003</th>
<th>WP-EB 004</th>
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<tr>
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4.4.2 Instrumentation Deployment

Onset HOBO U20L-04 pressure transducers and RBR Solo3 Tu sensors deployed at the 3 m sampling location in the spring, summer, and fall during the 2020 field season for Westport and the 2021 field season for Sprague and Wells (Table 4.1, Figure 4.1). An
additional pressure transducer was deployed nearby to collect atmospheric pressure measurements. Both sensors were mounted on 1-m rebar posts that were placed on a 1 ft\(^2\) tile on the marsh surface and grasses were trimmed in the immediate vicinity to reduce interference from vegetation. Rebar was sunk into the substrate until the bottom of the turbidity sensor window reached the surface of the tile with the middle of the sensor window located 1 cm from the bottom. The pressure transducers collected data every ten minutes. Pressure measurements were converted to water level by subtracting barometric pressure and dividing by an assumed ocean water density of 1023 kg/m\(^3\) and the acceleration due to gravity (9.81 m/s\(^2\)).

4.4.3 Turbidity Calibrations and Processing

Turbidity sensors were calibrated with a representative sample of marsh sediment collected from sediment traps at WEL 001. Each sensor was placed in an opaque basin with deionized water, and dried fine sediment from the sediment trap was added incrementally until ntu levels reached the desired amount. Readings were taken at 0, 10, 20, 30, 40, 50, 75, 100, 150, 200, and 300 ntu using the Ruskin software. Coarser (sandy) sediment, sourced from sediment cores at WEL 001, was then used to calibrate sensors for larger sediment sizes and readings were taken at 0, 100, 200, and 300 ntu. After each reading, sediment was pipetted out of the basin from directly in front of the sensor window and put through a filter attached to a vacuum pump. The filter was then dried and the mass of sediment from the concentration of sediment at the sensor window was calculated for each ntu reading. This was then used to calculate a calibration curve which was applied to all data to ensure quality of the measurements and provide a metric for suspended sediment concentrations (SSC).
Field turbidity measurements were obtained at a 5 min sampling interval. All turbidity measurements obtained while water levels were below the sensor were removed based on accompanying water level measurements. A 5-point median box filter was then applied to remove anomalous outliers (Baranes et al., 2022). An approximation for sediment trapping during seasonal deployments of instrumentation was obtained by taking the cumulative sum of the difference between maximum SSC during each flood tide and the minimum SSC during subsequent ebb multiplied by the respective tides flooding depths.

### 4.4.4 Sediment Trap Deployment

As a compliment to the turbidity sensor and pressure transducer data at each site, we obtained independent measurements of net sediment accumulation through sediment traps at each sampling location along the transects following methods similar to Baranes et al. (2022). Sediment traps were constructed with open 2.7-cm diameter 50 mL plastic centrifuge tubes covered with ¼-inch plastic mesh netting to exclude macrofauna such as crabs and shrimp residing on the marsh surface (Figure 4.2). Sediment traps were inserted into the marsh until the lower thread of the centrifuge tubes were flush with the marsh substrate such that the opening of the tube was ~1 cm above the marsh surface. Four tubes were placed in a row at each sampling location identified in Table 4.1. Seasonal traps recovered and capped at the end of the spring, summer, and fall deployments, and replaced with new traps in the case of spring and summer to capture the following season’s resultant deposition.
4.4.5 Sediment Trap Processing

After recovery, sediment traps were kept in sets of 4 in accordance with their location and stored under 4°C refrigeration until processing after field collection. Tubes were stored upright for 1-2 days to settle prior to decanting excess water. Once decanted, DI water was added until the tube was nearly full to remove excess salt, and then the tube was capped and shaken vigorously for 3-5 seconds. The tubes were then left to settle for 1-2 hours. Outlier tubes with volumes of sediment greater or less than 30% of the other samples in the same set were processed separately. Sediment trap material was then dried at 100 °C, weighed, and then combusted at 550 °C for four hours. The percent mass organic of the sample was calculated by taking the combusted weight divided by the bulk dry weight (Dean, 1974).

4.4.6 Crab Traps

Crab traps were deployed at each marsh 3 m from the channel edge in two-week intervals from June to October 2022 (see Table 4.1 for locations, Figure 4.1). Three short pitfall traps consisted of a yoghurt container (top diameter: 11.43 cm; depth: 13.34 cm; volume: 0.95 L) with a funnel attached to the top which were deployed in Westport beginning in late March to late October of 2022 with recovery and redeployment roughly every two weeks. Additionally, pitfall traps utilizing commonly available plastic tennis ball containers (top diameter: 6.86 cm; depth: 20.57 cm; volume: 0.35 L) were also deployed during peak crab activity in mid-August at Westport, Wells and Sprague (Figure 4.2). Table 4.2 outlines the timing of trap deployment as well as the type of trap or traps used.
Measurements for each trap were conducted in three steps (1) the number of crabs present in each trap were counted, (2) all crabs were weighed and (3) the minimum and maximum diameter of the crabs were recorded. This was done for all three traps in each transect, and the weights were averaged for each type of trap.

Table 4.2 Timing of crab trap deployment in Sprague, Wells, and Westport for each deployment period. The type of crab trap used is indicated as: short trap (S) and tennis container trap (T).

<table>
<thead>
<tr>
<th></th>
<th>05/27 - 06/08</th>
<th>06/08 – 07/01</th>
<th>07/01 – 07/26</th>
<th>07/26 – 08/12</th>
<th>08/12 – 08/22</th>
<th>09/08 – 10/09</th>
<th>10/09 – 10/25</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sprague</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>T</td>
<td>-</td>
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<tr>
<td>Wells</td>
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<td>-</td>
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<td>S</td>
<td>S</td>
<td>S</td>
<td>S/T</td>
<td>S/T</td>
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</tr>
</tbody>
</table>

4.5 Wave Height, River Stage, and Marsh Turbidity Comparison Results

4.5.1 Sprague

Turbidity sensors and water level loggers were deployed at Sprague River marsh at SPG001 3m from mid-March to mid-November 2021 (Figure 4.1). During this deployment period both the spring freshet and fall nor’easters were recorded, capturing a range of the potential sediment delivery mechanisms present at Sprague marsh. During this deployment peak turbidity was observed in May and April 2021. To compare to our turbidity data, we obtained river stage from the Kennebec River, the river proximal to Sprague River marsh, from the USGS station on the Kennebec River near the head of the tides (Sidney, ME, USGS gauge 01049265) and offshore wave data from the NOAA Central Maine Shelf Buoy (gauge 44032). For a majority of the sensor deployment period, the water level for the Kennebec River at the head of the tides is tidal, with the noted exception of two high river discharge events in April and May, which are
associated with the spring freshet (Figure 4.3, red and blue triangles, respectively). During the spring freshet the high discharge events overrode the tidal signal.

Peak turbidity for the Sprague River marsh aligns with these two highest discharge events for the year. During the April peak discharge and high turbidity there was concurrent increased offshore wave activity, however, during the May peak discharge event and high turbidity there was not increase offshore wave activity. The second high wave activity event occurs prior to both the increased discharge and peak turbidity at Sprague River marsh. The largest offshore wave event in terms of both duration and peak wave heights occurred in early November, associated with the Halloween Storm. This offshore wave event is not associated with an increase in turbidity at Sprague River marsh.

4.5.2 Wells

At the Wells marshes, the inorganic sediment input to the system differs from the primary sediment sourcing at Sprague. We see no significant increase in stage height or subsequent turbidity from of the spring freshet that was observed at Sprague marsh in April and May. On October 31, 2021, there was a significant storm event (referred to as the Halloween Storm for the duration of this paper). The Halloween Storm resulted in high offshore waves and increased discharge on Branch Brook, the primary tributary to the Little River (Figure 4.4). The red triangle in Figure 4.4 indicates the initial onset of high turbidity on the marsh in late fall, which occurs during the first peak wave heights from the NOAA Portland wave Buoy but before the rise in river stage on the Branch Brook. A second wave event associated with the Halloween storm has a stronger fluvial signal and is accompanied by a second rise in turbidity at WEL 001 (Figure 4.5).
Wells, which extends between the NOAA Wells National Estuarine Reserve (NERRS) and FWS Rachel Carson National Wildlife Refuge, has additional instrumentation deployed in the channel at the mouth of the Little River, near WEL 001. Figure 4.5 shows offshore wave heights from the NOAA Portland buoy, stage height on the Branch Brook, a tributary to the Littler River, and the NERRS data, in comparison with our turbidity and water level data at WEL 001 (river mouth) and WEL 003 (upriver) on the marsh platform. The NERRS salinity data shows a strong correlation with the increased stage height in the Branch Brook (USGS), with the salinity dropping to 0 psu. The NERRS turbidity increases slightly during the first major wave event, just prior to October 31, however the higher turbidity peak is with the second wave event. The NERRS sensor on the Little River appears to be bio-fouled after large discharge events, with drastic drops in turbidity occurring when the sensor is changed and cleaned. We see two such drops, one just prior to the Halloween storm (10/20/21), which provides confidence in the increased turbidity data from river discharge during the storm, and one after the Halloween storm (11/15/21) which corrects the post-storm heightened turbidity. WEL 003, upriver on the marsh platform, shows a turbidity increase on October 31, overlapping with the discharge event, however, this sensor does not get flooded regularly by waves, and as such is not affected by the initial wave event. WEL 001, at the channel mouth on the platform, is located such that it experiences both wave activity and river flooding. This sensor records a higher response to the initial wave event and a lesser response during the high discharge event.

4.5.3 Westport
Turbidity measurements for Westport were conducted in 2020, a year prior to Sprague and Wells. During this time the highest turbidity was observed in July and August, with no synchronous significant wave or discharge events recorded (Figure 4.6). Additionally, the periods of time with higher discharge on the Paskamanset River near Westport, MA shows increased discharge in from late March through May, with no notable increases in river discharge during the summer months, and a slight increase during mid-October and early November associated with nor’easters. The NOAA Block Island buoy likewise shows increased offshore significant wave heights during the spring and fall, with a notable peak mid-April, however, there is no subsequent rise in turbidity associated with this event.

During the 2020 fieldwork significantly greater crab activity was observed in the summer months than in the spring and fall. These observations were quantified using pit fall crab traps in 2022 at the site. The highest crab activity occurs in August when turbidity was the greatest during the earlier 2020 instrumental deployments. For a spatial comparison across our sites, tennis ball container pitfall traps were deployed in two-week deployments during the period of peak crab activity at Westport marsh (8/17/22). The crab activity at Westport (4.4 g/trap/day) is nearly double that of the activity Wells (2.6 g/trap/day) (Figure 4.7). Further, Sprague during the mid-August deployment shows activity between that of Wells and Westport with 3.6 g/trap/day (Figure 4.7). A complimentary crab trap deployment at Hampton/Seabrook marsh, located south of Wells at the New Hampshire and Massachusetts border, supports lower crab activity in Southern Maine/New Hampshire region with 2.0 g/trap/day.

4.6 Sediment Trap Results
Figure 4.10 (right panel) provides bar plots for the derived inorganic sedimentation rates from sediment traps during spring (blue), summer (green), and fall (red) deployments for 2021 at Sprague and Wells and 2020 for Westport (see Table 4.1 for locations that comprise distributions at each site). At Sprague the highest inorganic sediment deposition rate is in the spring, with summer having the lowest. The median sedimentation rates are 0.29 g/cm$^2$/yr, 0.049 g/cm$^2$/yr, and 0.10 g/cm$^2$/yr for the spring, summer, and fall, respectively. This agrees with the higher turbidity levels observed at Sprague during the spring, associated with the spring freshet in April and May. At Wells, the highest inorganic sediment rate was observed during the fall (0.095 g/cm$^2$/yr) followed closely by spring (and 0.07 g/cm$^2$/yr) and summer (0.04 g/cm$^2$/yr), (Figure 4.8). The higher inorganic sedimentation rate coincides with the turbidity measurements, indicating a combination of storm induced, marine sourcing as well as secondary fluvial supply of inorganic sediment to the system. Westport is the only marsh that has higher inorganic sedimentation rates during the summer (Figure 4.8). The median sedimentation rates are 0.036 g/cm$^2$/yr, 0.12 g/cm$^2$/yr, and 0.058 g/cm$^2$/yr for the spring, summer, and fall, respectively.

There is a close correlation between the sediment trap values and SSC derived from turbidity time-series. Figure 4.10 and Table 4.3 show the comparison for each site seasonally between the cumulative trapping calculated from turbidity sensor data and the mass accumulation from our sediment traps. The close correlation between the sensors and sediment traps highlights the reproducibility of this data and the reliability of the sediment traps for assessing inorganic sedimentation on marshes across a variety of environments, tidal conditions, and seasons.
Table 4.3 Comparison of mass accumulation between sensor data and sediment traps for each marsh in \((g/m^3)\). Bolded numbers represent the highest accumulation for each marsh. Sprague has the highest accumulation in the spring, Wells has the highest accumulation in the fall, and Westport has the highest accumulation in the summer. Agreements between the sensor mass accumulation and sediment trap mass accumulation agree within an acceptable margin of error.

<table>
<thead>
<tr>
<th></th>
<th>Sensor data (g/m³)</th>
<th>Sed trap data – clastic (g/m³)</th>
<th>Sed trap data – clastic + organic (g/m³)</th>
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### 4.7 Discussion

#### 4.7.1 Sediment Trap Method Assessment

All three marsh sites in this study vary in terms of timing, sourcing, and depositional rate for inorganic sediment. Due to the high trapping capacity of marsh grasses, we assume that the difference between flood and ebb turbidity represents the amount of sediment that is deposited on the marsh platform. The sediment trap design and installation process shown in this study prove to be a low-cost and efficient mechanism for quantifying sediment deposited on the marsh platform. We see a strong correlation between the cumulative sediment trapping estimated from the suspended
sediment concentrations (SSC) (calculated from turbidity measurements) and the mass accumulation of inorganic sediment from the sediment traps during the same time periods across all marshes (Figure 4.10). This supports the study’s sediment trap design as a viable method for determining sediment accumulation, and subsequently vertical accretion, across the marsh platform and in a variety of environmental conditions. The sediment trap design put forth in this study excels in trapping sediment on a season scale, with little loss due to disturbance or natural conditions, as evidenced by the comparison with our turbidity sensors. The estimates of this study can be compared to existing sediment trap techniques including sediment tiles or turf mats (Steiger et al., 2003; Nolte et al., 2019), floating lid (Schulze et al., 2022), SSC bottles (Honjo and Doherty, 1988; Eadie, 1997), artificial marker horizons (French and Spencer, 1993), SET stations (Rooth and Stevenson, 2000), and Cs$^{137}$ and Pb$^{210}$ dating (Naughton et al., 2010; Lane et al., 2011). Tiles, mats, and floating lid sediment traps are prone to greater uncertainty than our sediment traps due to the greater uncertainty due to sampling error, disturbance, or wash-out by rain. SSC bottles have a lower associated uncertainty; however, they only capture a single tide, rather than a season of sediment delivery. Artificial marker horizons, SET stations, and Cs$^{137}$ and Pb$^{210}$ dating deliver a lower resolution timescale that is decadal. These methods have a high associated cost and labor, making it difficult the match the temporal and spatial resolution that is possible with our sediment traps. Our sediment traps tendency for oversampling, however, the confirmation of the sediment traps with our time-series observations of suspended sediment allows for remote processes to be applied to marshes (e.g., Lidar), reducing the need for extensive field sampling.
Sprague Marsh is proximal to the Kennebec River, where there is abundant fine-grained material from the Presumpscot formation, a distinctive blue-grey clay deposit from the last glacial marine transgression, in the lowland regions of the watershed (Fenster and FitzGerald, 1996). The Presumpscot formation, however, is primarily along the coast, and as such other material from further up in the watershed is also introduced into the system through fluvial processes. The erodibility of the glacial deposits from the Presumpscot formation leads to an abundance of inorganic sediment being transported by the river to the estuary. During the spring freshet, increased discharge from melting snow can override the tidal signal, allowing for fluvial transport to dominate and provide terrestrial-derived clastic sediment to Sprague. Figure 4.3 shows the impact of the spring freshet of 2021 at Sprague. Two main freshet events occurred, in April and May, both overriding the tidal signal and increasing the turbidity within the marsh. There are offshore wave events in the same time period, however, the first wave event, while in the same time period as the increase in turbidity and the April freshet, arrives prior to both the freshet and the rise in turbidity. The first turbidity rise does not begin until the freshet signal reaches the head of the tides for the Kennebec River, despite the tides being high enough to flood the marsh. In locations where waves dominate the turbidity signal, such as Wells, the increase in both wave activity and turbidity has been observed to be contemporaneous. Baranes et al., (2022) also observed this phenomenon in the North/South River systems. The second offshore wave event in mid-April is associated with a relatively minor rise in turbidity, supporting the hypothesis that the fluvial signal is stronger than the marine signal at Sprague. The Kennebec River is an ebb dominated
system, exporting both fine and coarse grained material to the coast during high discharge events, such as the spring freshet (Fenster and FitzGerald, 1996; FitzGerald et al., 2000; Fenster et al., 2001). FitzGerald et al., 2000 discusses the formation of a sandy offshore gyre, driven by offshore waves and tidal currents, which provides ample sediment to the Sprague River Marsh estuary.

In the fall, the offshore buoy records the Halloween storm, but we see no subsequent increase in turbidity at Sprague. This supports our interpretation that the earlier rise in turbidity between the two freshet events during wave event in mid-April, was most likely due to a mobile pool of sediment, exported by estuary during the initial freshet, and later remobilized offshore and into the marsh during the mid-April wave event. The remobilization of offshore sediment appears to be a feature associated with larger river systems. Baranes et al. 2022 at the North and South Rivers, considerably smaller rivers compared to the Kennebec River, did not observe a remobilization of offshore sediments, but rather an erosional of glacial drumlins offshore. In systems such as the Mississippi River, remobilization of offshore sediments are the primary source of inorganic sediment delivery to marshes (Meade, 1982; Kennish, 2002; Turner et al., 2007; Chant et al., 2020). This same phenomenon, supports similar responses for smaller ebb dominate systems such as the Kennebec and Connecticut (Patton and Horne, 1992; Yellen et al., 2014).

4.7.3 Wells

The Little River marsh in Wells is a back barrier system, characterized by a relatively small watershed. The watershed for Wells is 117 km², with 36 km² and 81 km² being from the Little River and Webhannet River, respectively. The Kennebec River
adjacent to Sprague marsh has a watershed of 24,000 km², 205 times the size of the Wells watersheds combined. It is located behind a sandy barrier beach that is separated by bedrock headlands. The Webhannet/Littler River intertidal zone in this system is very well mixed and exhibits primarily flood dominated sediment transport which leads to the development of large tidal deltas (FitzGerald et al., 2000). We observe a similar process as observed by Baranes et al., 2022, where two eroding drumlin headlands (Third and Fourth Cliff), located on the north and south sides of the North/South River inlet, serve as a marine source of sediment to the systems backbarrier marshes during high wave activity (Kelley et al., 1995). The inlet at Wells faces to the southeast, so most waves in the region are approaching within 20 degrees of this direction, allowing for waves, particularly from coastal storms, to impact the site and rework offshore sediments (Jacobson, 1988). The Wells marsh exhibits a combination of both marine and fluvial sediment sourcing, however, the offshore glacial deposits in the intertidal and subtidal zones, in conjunction with the directionality of the inlet, make marine sediment sourcing slightly more influential. The Little River, while a much smaller watershed than the Kennebec River, is dominated by the Presumpscot formation (Kelley et al., 1996). The Little River incises into the Presumpscot formation, allowing for sediment to be transported to Wells marsh during high discharge events, particularly when landslides occur in the watershed mobilizing sediment (Dickson and Johnston, 2015).

Turbidity measured at WEL 001 is relatively stable throughout the spring and summer, with little variation or major peaks in turbidity. There were four higher precipitation events in the summer, however, we do not see an increase in turbidity as a result. The main instance of increased turbidity at WEL 001 is during the Halloween
storm. This storm is characterized by two large offshore wave events and one increase in stage height on the Branch Brook (the main tributary to the Little River) (Figure 4.4). The turbidity has two main increases the first occurs after the initial wave event and prior to the higher stage height. The second increase in turbidity is congruent with the rise in stage height on the Little River (Figure 4.4, 4.5). This indicates that the sediment delivery at Wells is a mixture of marine and fluvial sourced sediment that is driven by coastal storms in the fall.

The mixture of offshore and fluvial sourcing of sediments to Wells highlights the complexity of sediment sourcing to salt marshes. Some marshes exhibit primarily offshore sediment sourcing due to abundant availability of erodible offshore deposits (Baranes et al., 2022), yet larger systems like the Hudson or the Kennebec Rivers, have significant increases in sediment delivery to the estuary from the spring freshet (Geyer et al., 2001). Geyer and Ralston, 2018 highlight the availability of a mobile pool of sediment offshore from the Penobscot formation, which indicates a potential source of inorganic, marine sediment to the Wells marsh system. This offshore sediment can travel down-estuary during high discharge events, and back up-estuary during low discharge and higher wave events (Yellen et al., 2017). The dynamic movement of sediment, and heterogeneity in sourcing, is evident in the Wells system.

4.7.4 Westport

Westport is the only marsh where we observed the highest sediment delivery during the summer, when there is comparably fewer high discharge events and coastal storms, which typically observed in the spring and fall (Figure 4.6). The spring and fall, which have higher offshore wave and river discharge activity, do not result in increased
turbidity within the Westport marsh system (Figure 4.6). Our sediment traps likewise do not show an increase in trapping during the spring or fall. The lowest sediment trap values are during the spring, when there is the highest offshore wave activity and discharge on the Paskamanset River. The fall has relatively low discharge, but higher wave activity, also does not result in higher turbidity levels or sediment trapping (Figure 4.6). The summer high turbidity has little correlation with high discharge of larger offshore wave events; however, peak crab abundances overlap with both the high turbidity and increased sedimentation rate. This potentially highlights the influence of crab burrowing on resuspension and subsequent redeposition of sediment within the marsh system.

While we observed no major event that would lead to increased deposition during summer, this time period is the high season for crab activity on salt marshes (Reinsel, 2004). We postulate that crab burrowing and herbivory results in internal resuspension of sediment that is recorded both in turbidity time-series and independent observations from seasonal sediment traps. Both metrics, however, only record turbidity or deposition, not erosion. It is likely that the increased turbidity and deposition observed in Westport, a result of resuspension of existing sediment, does not represent an increase in elevation capital, but rather an off-set of prior erosion from the crab activity. Marshes, however, need this increased elevation capital to keep up with sea level rise, which could lead to an issue for Westport marsh. The lower sediment delivery in the spring and fall, as shown by lower turbidity and sediment trapping, indicates the Westport marsh is relatively sediment starved likely due to the micro-tidal conditions and location within erosive resistant till (Woodruff et al., 2021). This lack of sediment supply, in conjunction with
the increased stress from crab activity and resultant resuspension of internal sediment, could explain the observed losses in marsh area in the Westport and Long Island Sound regions (Schultz et al., 2016; Weiner et al., 2017). In a study south of Cape Cod, Holdredge et al., (2009) showed that high crab densities can lead to marsh die off. Crab populations are able to increase most likely because there is a lack of predators due to recessional fishing influences (Altieri et al., 2012).

While crab activity leading to increased erosion from burrowing and other herbivory is not unique to Westport, the density of the crab populations appears to be higher further south in New England. Contemporaneous crab counts at Sprague, Wells, and Westport during the highest observed crab density at Westport, with Sprague being the second highest. We see a decreasing northward trend in the crab population. With less crab influences on the sediment resuspension, other forcings can be the primary driver of sediment delivery to the system. One noted exception is a turbidity peak in late August in Sprague marsh that is not associated with any major offshore wave events or higher discharge on the Kennebec River, however, it does correspond to increased crab activity (Figure 4.3, 4.7). While this increased crab activity in late August led to a peak in turbidity, sediment traps and water column turbidity observations suggest this is not the primary driver of sediment delivery at Sprague (Fig 4.10).

4.8 Conclusions

Our sediment trap design provides a straightforward and cost-effective means of assessing sediment deposition with validation provided by independent instrumental assessments via water level observations and flood-eb differences in SSC. The inclusion of our sediment traps to restoration and remediation plans for marshes, will improve the
monitoring of marshes and expand the understanding of what is required to have a healthy marsh in the face of human influences and climatic stressors. Through the deployment of turbidity sensors, water lever loggers, and sediment traps, we were able to identify regional patterns of sediment delivery for post-glacial shorelines of the New England region. For marshes located adjacent to major rivers with large watersheds (>10,00 km²), sediment delivery typically occurs in the spring from fluvial sources associated with the spring freshet. In back-barrier marshes with relatively small watersheds, there is a mixture of marine and fluvial sourcing of clastic sediment associated with freshets in the spring and coastal storms in the fall. Westport is the most sediment starved of all the marshes in this study, with minimal sources of external sedimentation from either fluvial or marine derived sources. Turbidity within Westport marsh reflects cannibalization and remobilization of sediment within the marsh during the summer months due to increased crab activity (e.g., burrowing and herbivory). Independently derived estimates of sedimentation are consistent with the seasons of primary deposition with the greatest accumulation from sensor data occurring in spring for Sprague, fall for Wells and summer for Westport. The vast differences between the timing of delivery and the source of delivery for each of these marshes highlights the importance of identifying these trends prior to enacting any management or restoration efforts.
Figure 4.1 Location of transects at all salt marsh sites. At each red dot there is a transect of 3-4 sediment traps at 3 m, 10 m, 40 m, and 100 m (if possible). The blue dots in B, C, and D are where the transect of sediment traps also has a RBR Solo3 Tu sensor and an ONSET Hobo U20L-40 pressure sensor at 3 m (or the closest site in each transect to the marsh edge). (A) Regional map of New England indicating salt marsh systems in this study: Sprague, Wells, and Westport. (B) Locations of transects at Sprague. SPG 001 is where sensors were deployed in addition to sediment traps. (C) Locations of transects at Wells. WEL 001 is where sensors were deployed in addition to sediment traps. (D) Locations of transects at Westport. WP-EB 001 is where sensors were deployed in addition to sediment traps.
Figure 4.2 Sediment and crab traps deployed in each marsh. (A) Sediment traps. At each site sets of four sediment traps were deployed at regular intervals in transects across the marsh. Traps were deployed in spring, summer, and fall to identify spatial and temporal trends of sediment deposition. (B) Sediment trap made from a centrifuge tube with netting secured to the top. (C) Crab traps deployed during the summer of 2022. Top row is the tennis ball traps (E). Bottom row is the yogurt container traps (D). (D) Yogurt container traps constructed with a funnel secured to the top. (E) Tennis ball container pit trap.
Figure 4.3 Comparison of offshore wave heights, stage height on the Kennebec River, and our turbidity and water level measurements from Sprague (SPG 001) for 2021. The grey boxes indicate the spring freshet in April and May. (A) Offshore wave heights from the NOAA Central Maine Shelf buoy. (B) Stage height (m) for the mouth of the Kennebec River at the head of the tides (USGS). During the spring freshet of 2021 (black box) the stage height increased above the tidal range (C) turbidity (ntu) and water level (m) for SPG 001. There is an increase in turbidity associated with the spring freshet. An increase in turbidity is also associated with an increased stage height on the Kennebec River in May where the river water level overrides the tidal signal.
Figure 4.4 Comparison of offshore wave heights, stage height on Branch Brook (a tributary to the Little River), and our turbidity and water level measurements from Wells (WEL 001) for 2021. The box indicates a storm on October 31, 2021 (Halloween Storm of 2021). (A) Offshore wave heights (m) from the NOAA Portland Maine buoy. During the Halloween Storm of 2021 the offshore wave heights increase (B) Stage height (m) for Branch Brook, a tributary to the Little River that flows into the marsh at Wells (USGS). During the Halloween Storm in late October the stage height increases after the initial offshore wave heights increase. (C) turbidity (ntu) and water level (m) for WEL 001. There is an increase in turbidity associated with the Halloween Storm in time with the offshore waves. The orange triangles across all three plots shows the increase in turbidity has a stronger correlation with the offshore wave height increase than the increased, delayed stage height increase on the Branch Brook.
Figure 4.5 Halloween Storm of 2021 with data from October 15 – November 22, 2021. (A) NOAA Portland buoy offshore wave heights (red) for the Halloween Storm. There is an initial increase between October 24-31. (B) stage height (m) for the USGS Branch Brook. There is a rise in stage height (blue) on October 31, 2021. (C) NERRS Little River mouth water depth measurements (m) in the channel showing tidal cycles. October 31, 2021, the depth of the Little River mouth increases, but still has a tidal signal. (D) NERRS Little River mouth (channel) showing salinity (psu). Typical condition shows higher salinity during the flood tide. During higher stage height, the salinity drops to zero before returning to normal conditions. (E) NERRS Little River mouth turbidity measurements (ntu). There is an increase in turbidity during the increased stage height on the Branch Brook and Little River, during the second offshore wave event, however, there is no increase during the first offshore wave event. (F) Marsh platform turbidity measurements (ntu) from WEL 001 (orange) with water level (m) (grey). The largest increase in turbidity is during the first wave event with a smaller increase coinciding with the second wave event, increased stage height and higher turbidity at the NERRS station. (G) Turbidity further river on the marsh platform at our WEL 003. This shows less increase of turbidity, however, it is often dry, so it does not record the wave signals as well.
Figure 4.6 Comparison of offshore wave heights, stage height on the Paskamanset River, and our turbidity and water level measurements from Westport (WP-EBL 003) for 2020. (A) Offshore wave heights (m) from the NOAA Block Island buoy (B) Discharge (m$^3$/s) for The Paskamanset River. There is higher discharge during the spring, with very low discharge rates in the summer, and a slight increase in the fall (C) turbidity (ntu) and water level (m) for WP-EB 003 20 m. The peak turbidity is in the summer with very little response to the spring or fall high discharge and offshore wave events.
Figure 4.7 Distribution of crab populations for Sprague, Wells, and Westport marshes. (Left) Plot showing the distribution of crabs in grams per trap per day in Westport, MA from June to October 2022. Crabs from each trap were weighed and then the averages of each trap were averaged to normalize across traps on a given deployment. The highest record of crabs was during our August 17, 2022 deployment recovery. The crab traps type for this plot are the funnel traps, as this was the trap that was consistent across the entire deployment (Right) A comparison of the distribution of grams of crabs per trap per day for Sprague, Wells, Hampton Seabrook, and Westport for the August 17, 2022 deployment recovery when the greatest crab occurrence was recorded in Westport. The crab trap type for this plot is the tennis ball container pitfall traps.
**Figure 4.8** Box and whisker plots showing the inorganic sediment rates (g/cm²/yr) for Sprague, Wells, and Westport during Spring (blue), Summer (green), and Fall (red). At Sprague, the highest inorganic sediment rates are during the Spring, associated with the spring freshet. At Wells the highest inorganic sediment rate is during the fall, associated with storms. At Westport, the highest inorganic sediment rate is during the summer, associated with increased crab activity promoting resuspension of sediment within the system.
Figure 4.9 Lidar shaded relief map of the watersheds feeding into Wells. The southern portion of the marsh is within the Webhannet River watershed (WEL 002, WEL 004, and WEL 005) and the northern portion of the marsh is within the Little River watershed (WEL 001 and WEL 003). The Little River water shed is incising into the clay rich Presumpscot formation (indicated with the red arrow). The red dots indicate where sediment trap transects are located (see location figure for transect information). The blue dot indicates the transect where both sediment traps and sensors were deployed.
Figure 4.10 Comparison of cumulative sediment trapping, estimated from SSC (from turbidity measurements, orange) and water level (grey), with inorganic sediment collected in sediment traps for each site. Cumulative trapping shown on both the right and left plots are in the same units with the identical scaling. (left) Cumulative trapping (g/m²) from seasonal turbidity data (left) in each marsh. The grey bars are water dept in mm and the orange bars are SSC from turbidity sensor data in mg/L. The blue, green, and red lines represent cumulative trapping in the spring, summer, and fall, respectively. (right) Mass accumulation (g/m²) from seasonal sediment traps (right) in each marsh. Green, blue, and red correspond to spring, summer, and fall, respectively. The left column is the clastic mass accumulation, and the right column is the total mass accumulation of both clastic and organic matter (top) Sprague (middle) Wells (bottom) Westport.
CHAPTER 5

CONCLUSIONS

5.1 Conclusions for Chapter 2

1. Mangroves are an important buffer for coastlines to protect inland areas from overwash deposition.
2. Historical deforestation of barrier mangroves results in a clear onset for enhanced overwash deposition to backbarrier lagoon, followed by reduced deposition following reforestation.
3. Similar mangrove deforestation is observed following Hurricane Maria resulting in the barrier being more vulnerable to enhanced storm-induced deposition during recovery phase following the event.

5.2 Conclusions for Chapter 3

1. Even in relatively shielded regions, hurricanes can still cause severe defoliation in mangrove forests.
2. Indirect impacts can cause a delayed death in mangrove forests. Sinuous inlets can restrict the flow of water through the system, leading to flooding and stagnation after the event.
3. The percent vegetative cover analysis and historical observations indicate the northwestern mangrove system on Vieques is capable of rebounding from a mortality event, like the one from Hurricane Maria, within several decades, barring any anthropogenic interference with the system.

5.3 Conclusions for Chapter 4

1. For marshes located adjacent to major rivers with large watersheds (>1,000 km²) (e.g. Sprague marsh), sediment delivery typically occurs in the spring from fluvial sources associated with the spring freshet.
2. In back-barrier marshes (e.g., Wells marsh) with relatively small watersheds, there is a mixture of marine and fluvial sourcing of clastic sediment associated with freshets in the spring and coastal storms in the fall.

3. Marshes that are sediment starved with minimal sources of external sedimentation from either fluvial or marine derived sources (e.g., Westport marsh), internal sedimentation from cannibalization and remobilization of sediment within the marsh during the summer months from increased crab activity (e.g. burrowing and herbivory) can provide needed inorganic sediment.

4. Our sediment trap design provides a straightforward and cost-effective means of assessing sediment deposition with validation provided by independent instrumental assessments via water level observations and flood-eb differences in SSC.
APPENDIX A

CHAPTER 2 SUPPORTING INFORMATION

A.1 Percent vegetative cover for each year from 1936-2019.

Calculations for the percent vegetative cover were done by hand due to the nature of the available imagery. All images were georeferenced with the same benchmarks which were consistent throughout all images. The benchmarks were road junctions, building edges, and bridges that were constant from 1936 through to the present. The highest resolution images available were first georeferenced into QGIS, an open-source geographic imaging program. Then the total area (Figure A.1.1) was identified in order to have a constant area for calculations and comparison. The living vegetation within that area for the image from each year was carefully outlined. For black and white images there is slightly more error, however the texture of vegetation could be identified, and this was used to delineate between living vegetation and dead vegetation or debris. As fine of detail was carried out for each image. The difference between the calculated vegetation and the total area yielded the percent vegetation for each year.
A.1.1 Total area used for each percent vegetation calculation in QGIS.
Figure A.1.2 Percent vegetative cover for 1936 with aerial imagery for 1936 in the background. Vegetative cover was outlined by hand in QGIS.
Figure A.1.3 Percent vegetative cover for 1989 with satellite imagery for 1989 in the background. Vegetative cover was outlined by hand in QGIS.
Figure A.1.4 Percent vegetative cover for 1994 with satellite imagery for 1994 in the background. Vegetative cover was outlined by hand in QGIS.
Figure A.1.5 Percent vegetative cover for 2004 with satellite imagery for 2004 in the background. Vegetative cover was outlined by hand in QGIS.
Figure A.1.6 Percent vegetative cover for 2006 with satellite imagery for 2006 in the background. Vegetative cover was outlined by hand in QGIS.
Figure A.1.7 Percent vegetative cover for 2010 with satellite imagery for 2010 in the background. Vegetative cover was outlined by hand in QGIS.
Figure A.1.8 Percent vegetative cover for 2014 with satellite imagery for 2014 in the background. Vegetative cover was outlined by hand in QGIS.
Figure A.1.9 Percent vegetative cover for 2017, pre-Hurricane Maria with satellite imagery for 2017, pre-Hurricane Maria in the background. Vegetative cover was outlined by hand in QGIS.
Figure A.1.10  Percent vegetative cover for 2019 with satellite imagery for 2019 in the background. Vegetative cover was outlined by hand in QGIS.
A.2 Watershed for Laguna Playa Grande

Calculated percent vegetative cover was done in order to show the amount of forest cover within the watershed. Previous human interactions with the watershed had led to severe deforestation for sugar cane and military purposes. Our calculations indicate that the reforestation of Vieques has continued to present day, with 83% of the LPG watershed currently forested.

Figure A.2.1 Percent forested calculations for the watershed of Laguna Playa Grande done by hand in QGIS. Blue line is the delineation of the watershed. The green shading is the vegetated area of the watershed.
Figure B.1.1 Total area used for each percent vegetation calculation in QGIS.
Figure B.1.2 Percent vegetative cover for 1936 with aerial imagery for 1936 in the background. Vegetative cover was outlined by hand in QGIS.
Figure B.1.2.3 Percent vegetative cover for 1989 with satellite imagery for 1989 in the background. Vegetative cover was outlined by hand in QGIS.
Figure B.1.4 Percent vegetative cover for 1994 with satellite imagery for 1994 in the background. Vegetative cover was outlined by hand in QGIS.
Figure B.1.5  Percent vegetative cover for 2010 with satellite imagery for 2010 in the background. Vegetative cover was outlined by hand in QGIS.
Figure B.1.6 Percent vegetative cover for 2014 with satellite imagery for 2014 in the background. Vegetative cover was outlined by hand in QGIS.
Figure B.1.7 Percent vegetative cover for 2018, post-Hurricane Maria, with satellite imagery for 2018, post-Hurricane Maria, in the background. Vegetative cover was outlined by hand in QGIS.
Figure B.1.8  Percent vegetative cover for 2021 with satellite imagery for 2021 in the background. Vegetative cover was outlined by hand in QGIS.
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