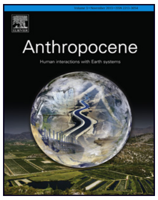
Anthropocene 34 (2021) 100295

Contents lists available at ScienceDirect

Anthropocene

journal homepage: www.elsevier.com/locate/ancene



Of marsh and mangrove: coupled biophysical and anthropogenic drivers of 20th century wetland conversion in Tampa Bay Estuary, Florida (USA)

Kendal Jackson[a,*], Gregg R. Brooks[b], Rebekka A. Larson[b]

[a] University of South Florida, Department of Anthropology, 4204 E. Fowler Ave., Tampa, FL, 33620, USA
[b] Galbraith Marine Science Laboratory, Eckerd College, 4200 54thAve. S., St. Petersburg, FL, 33711, USA

ARTICLE INFO

Article history:
Received 29 May 2020
Received in revised form 18 April 2021
Accepted 19 April 2021
Available online 22 April 2021

Keywords:
Coastal wetlands
Historical ecology
Gulf of Mexico
Anthropogenic seascapes
Sediment analysis

ABSTRACT

Dense mangrove swamps currently dominate tidal wetlands of the Tampa Bay Estuary System on the central peninsular Gulf Coast of Florida (USA). Late-19th century Coast and Geodetic Survey topographical charts and Government Land Office surveys, however, depict wetland systems dominated by salt marsh—therefore suggesting mangrove dominance as a product of 20th century encroachment. To clarify the primary drivers of ecosystem change, this study integrates sedimentological, paleobotanical, and radiometric analyses of sediment cores collected in 2018 with analyses of aerial photography taken between 1940 and 1997. Results empirically ground truth the wetland conversions inferred through analytical comparisons of historical and modern mapping and establish a high-resolution chronology for coastal environmental change. These results showed that salt marsh and salt prairie habitats persisted within Tampa Bay study areas until the mid-20th century. Mangrove forest rapidly encroached the study areas over a 20 year interval, between 1960 and 1980, immediately following intensive ditching for mosquito control. The findings demonstrate how coastal geoengineering, mangrove autoecology, and sea-level rise interacted across the late-20th century to accelerate the creation of novel seascapes.

1. Introduction

Across the neotropics, coastal wetlands are transforming in response to climate change and sea-level rise (Harley et al., 2006). At various global locales, mangroves are encroaching into salt marshes and transforming the ecology of tidal ecosystems (Saintilan et al., 2014). While oscillations between woody and herbaceous vegetation boundaries are common and well-documented in many ecosystems (Knapp et al., 2008; Van Auken, 2000), widespread marsh-to-mangrove conversion has accelerated in recent decades. Such conversion has caused alarm among ecologists and coastal resource managers (e.g., Armitage, 2015; Everitt et al. 2010; Perry and Mendelssohn, 2009; Williamson et al., 2011), who recognize that major shifts in foundation vegetation taxa will likely drive commensurate qualitative and quantitative changes in the ecosystem services provided by coastal habitats (see Engle, 2011). Further, salt marsh and salt prairie environments are widely recognized as critical habitats for terrestrial mammals, reptiles, amphibians, mollusks, migratory birds, and waterfowl not typically found in mangrove forest (Montague and Wiegert, 1990:508–512; Odum and McIvor, 1990:542–543).

Workers have identified various drivers of mangrove expansion and encroachment, including sea-level rise (Krauss et al., 2011; Lopez-Medellin et al., 2011; Rogers et al., 2006; Smith et al., 2013), warming winter temperatures (Duke et al., 1998; Osland et al., 2013), elevated atmospheric CO_2 (McKee et al., 2012), precipitation (Saintilan and Wilton, 2001), droughts (Rogers et al., 2006), and sedimentation (Woodroffe et al., 1985). At local- to meso-scales, however, patterns of mangrove expansion into adjacent habitats may be influenced substantively by interactions between climate change and direct anthropogenic impacts (He and Silliman, 2019; Patterson et al., 1997; Patterson and Mendelssohn, 1991). Such interactions will likely produce novel types of tidal wetlands (*sensu* Hobbs et al., 2013; Lugo et al., 2014), characterized by new species combinations and biophysical interactions that lack local historical analogs.

In 2012, Raabe et al. (2012) compared 19th century survey documents with contemporary aerial photography to infer a previously undocumented estuary-wide pattern of marsh to mangrove conversion within the tidal zone of Tampa Bay Estuary on the central peninsular Gulf Coast of Florida. The authors reviewed historical Coast and Geodetic Survey (CGS)

---

[*] Corresponding author.
E-mail address: Kendalj@usf.edu (K. Jackson).

http://dx.doi.org/10.1016/j.ancene.2021.100295
2213-3054/

Topographical Charts (T-Sheets) and General Land Office (GLO) survey records to generate models of late-19th century coastal vegetation cover within four areas of Tampa Bay Estuary. They compared these historical models to modern vegetation cover assessments based on 1999 aerial imagery (SWFWMD, 2005) and documented a dramatic trend of mangrove encroachment in each of the study areas (Raabe et al., 2012:1153). While latitudinal temperature gradients are known to condition the physiographic distributions of mangrove and salt marsh along the Florida peninsula and the Gulf of Mexico (Kangas and Lugo, 1990; Mongague and Wiegert 1990; Osland et al., 2013; Stevens et al. 2006), the latitudinal pattern of wetland conversion documented by Raabe et al. (2012:1155) within Tampa Bay Estuary does not implicate temperature change as a major factor. Instead, their interpretation broadly attributes mangrove encroachment to sea-level rise and suggests that impacts of coastal urbanization (e.g., drawdowns of local aquifers, damming of streams, and shoreline alteration) also functioned as important drivers (Raabe et al., 2012:1156).

While the comparative historical analyses by Raabe et al. (2012) is both rigorous and compelling, to date there are no published sediment records that ground-truth tidal wetland conversion in Tampa Bay Estuary. Further, while 19th century survey documents provide essential points of historical reference, they are not particularly suitable for investigating the chronology or process of mangrove encroachment across the 20th century. In their discussion, Raabe et al. (2012:1156) called for further comparative analyses of historical aerial photography to understand better the nature and timing of habitat transformation. As a complement to the research reviewed above, the present study integrates sedimentological and remote-sensing approaches to clarify 20th century marsh-to-mangrove conversion. We focus on two study sites located within Old Tampa Bay—the western basin of Tampa Bay Estuary. Building on Raabe et al. (2012), we provide new data from sediment core samples that ground-truth wetland conversion patterns observed in historical map comparisons. These new data enable refinement of the chronology of habitat transformation. To contextualize the sediment records and investigate the processes driving marsh-to-mangrove conversion we analyzed sequences of 20th century aerial photography at each study site and correlated sediment core records with changes apparent on the larger seascapes.

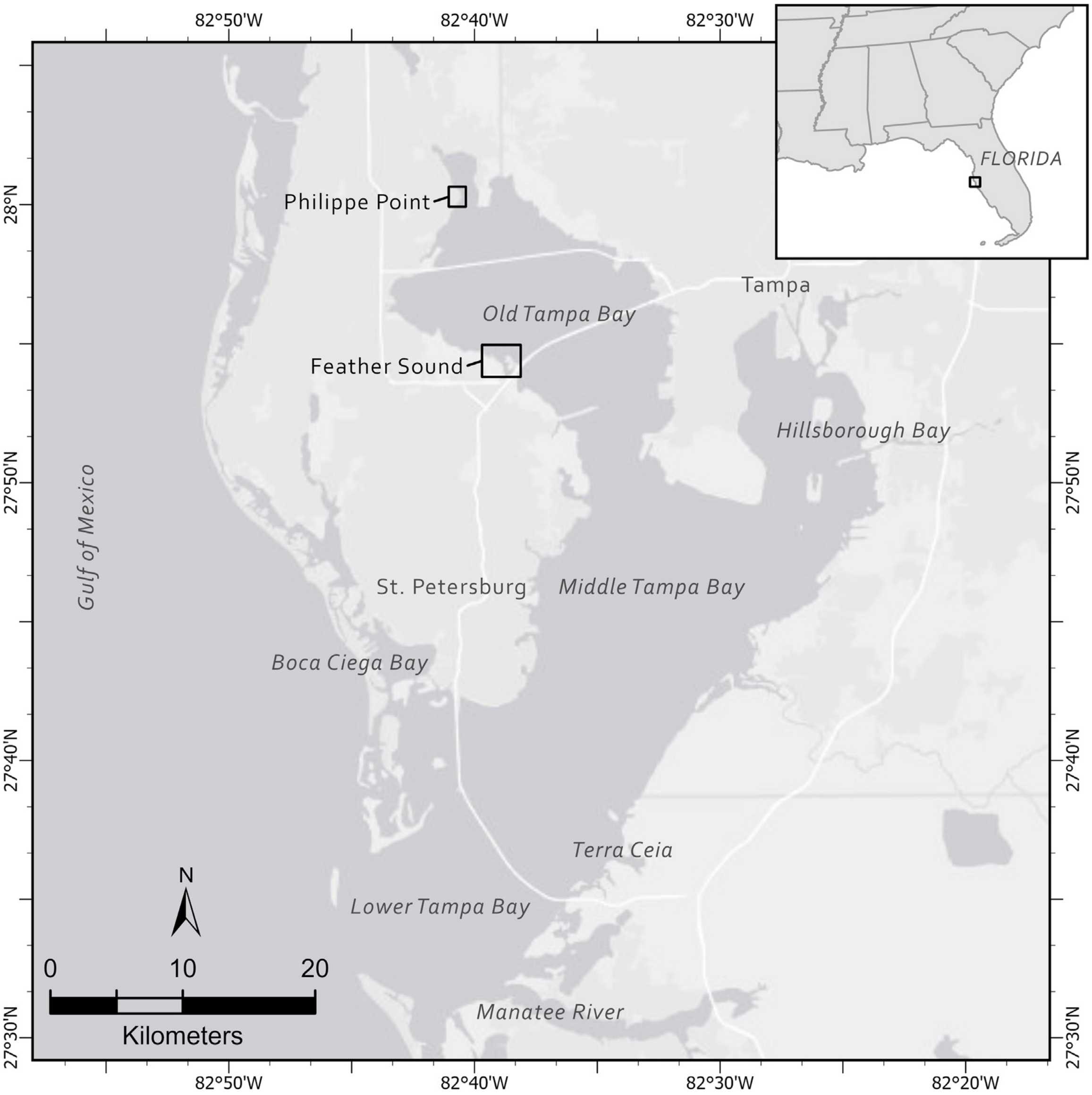


Fig. 1. Map of Tampa Bay Estuary and study site Locations.

This article thus addresses the following research questions. First, what near-historical (20th century) changes in habitat type do the tidal wetland sediments of Old Tampa Bay reveal, and how consistent are these records with current understanding of marsh-to-mangrove conversion? Second, what is the precise chronology of these transformations as recorded in the sedimentary record? In our discussion of these results, we evaluate the major factors that drove the process of wetland conversion at the study sites in Old Tampa Bay (sections 5.1 and 5.2). Additionally, we discuss the implications of marsh-to-mangrove conversion processes for the conservation and management of coastal wetlands (section 5.3).

2. Study area

2.1. Tampa Bay Estuary

Tampa Bay is a large (1000 km^2), shallow (4 m mean depth) microtidal estuary system on Florida's Central peninsular Gulf Coast.It comprises seven major sub-divisions, including: Old Tampa Bay, Hillsborough Bay, Boca Ciega Bay, Terra Ceia, Middle Tampa Bay, Lower Tampa Bay, and the tidal Manatee River mouth (Fig. 1) (Morrison and Yates, 2011). Tampa Bay is situated near the northern extent of the West Central Barrier Chain, a mixed-energy barrier island-inlet system that extends from the Anclote River to Cape Romano. It is bordered to the north and south by low-energy coasts dominated by vast expanses of low-gradient tidal wetlands (Davis, 2011). The region occupies a zone of climatic transition between temperate and sub-tropical zones (Chen and Gerber, 1990). Wet seasons (June through September) are typically warm and humid with average precipitation of 72 cm; Dry seasons (October through May) are typically cool and dry, with average precipitation of 49 cm. Mean annual temperature ranges from 33 °C (91 °F) to 17 °C (63 °F). Tropical storms and hurricanes originating in the Atlantic tropical cyclone basin impact Florida's Gulf Coast with moderate frequency, peaking from September through October (Brown et al., 1990:23–24; Jordan, 1984; Simpson and Riehl, 1981).

Unlike many states in the USA east of the Mississippi River, Florida remained an American frontier into the 20th century (Mormino, 2005:11–43). Urban development and population densities increased slowly until the boom years of the 1950s, when "explosions in population and built environments transformed Florida from the most rural of the southern states into an urban megastate characterized by sprawling suburbs, a car-dependent culture, and a powerful tourist industry" (Zacks, 2013:258). Following global trends, most of the urbanization in Florida has concentrated within the coastal zone. In 2010, some 14 million of Florida's 19 million total residents inhabited coastal areas (Mormino, 2005:11–43; Santamaria et al., 2010:10–11). The Tampa Bay region, surrounding Florida's largest open-water estuary, was no exception to this pattern. Spurred by lucrative mid-century real-estate markets, an expanding phosphate mining industry, and major public investments in tourism, the population of the three counties directly bordering Tampa Bay Estuary (Hillsborough, Pinellas, and Manatee counties) increased more than four-fold between the early 1950s and 2010 (Yates and Greening, 2011:5–6). Urban expansion and development of the Tampa Bay region proceeded through large-scale coastal engineering programs, including: the construction of port facilities, the dredging of deep-water channels, causeway and bridge construction, and a staggering number of dredge-and-fill operations that created new land from the sea floor (Janicki et al., 2001; Lewis and Estevez, 1988; Morrison and Yates, 2011:18–19). Fig. 2 displays two historical photographs depicting 1924 dredge and fill operations in Tampa Bay near Downtown Tampa and networks of artificial land dredged from the seabed for residential development along Boca Ciega Bay. By the time coastal environmental conservation and management efforts began in the late 1970s and early 1980s, the ecological state of the bay's once famed seagrass meadows, oyster reefs, and fisheries appeared almost irreparably destroyed, inspiring the popular conception among residents that Tampa Bay Estuary was ecologically dead (see Fehring, 1986).

Fig. 2. (a) Dredge operations in the construction of Davis Islands at the mouth of the Hillsborough River, Tampa Bay, ca. 1924. Photo by Bergert Brothers via USF Digital Collections. (b) Dredged landforms and residential construction in Boca Ceiga, Tampa Bay, ca. 1950s. Photo via Pinellas County Heritage Village archives.

The ecology of Tampa Bay Estuary, of course, did not die. Subsequent decades of integrated management and restoration work by ecological consultants, non-profit organizations, and government programs produced improvements in water quality and substantially increased the spatial extent of sea grass meadow habitat (Cicchetti and Greening, 2011; Janicki, 2000). Mitigation and restoration work in Tampa Bay followed the tenants of seminal coastal ecology work at study sites farther south along the peninsular Gulf Coast (Davis, 1940, 1943; Odum et al., 1982; Odum and Heald, 1972), aiding the legal protection of mangroves in 1984 and the promotion of mangrove forest as a climax habitat for the bay's intertidal wetlands (see Janicki et al., 1995; PBS&J 2010). This approach served ecologists and land managers alike, who understood that the tidal wetlands of Tampa Bay were naturally dominated by mangrove swamps, and that mangroves recruit and proliferate on restoration areas quickly and at relatively little financial expense.

Analyses of late-19th century survey documents for Tampa Bay's coastal zone showed that large areas of intertidal wetlands currently dominated by dense and homogenous mangrove forest were mosaics of salt marsh and salt prairie in the 1870s, with thin, fringing mangrove stands. In their robust GIS-based analysis, Raabe et al. (2012) quantified spatial conversion of non-mangrove coastal habitat to mangrove forest in four study areas within Tampa Bay over an approximately 125-year interval—from CGS T-Sheet publication (AD 1874–1876) to the creation of modern land use/land cover datasets (AD 1999–2000). Non-mangrove-to-mangrove conversion was documented across each study area, with spatial conversion varying between 52 and 95 percent and rates of conversion varying between 1.5 and 7.8 ha per year. Echoing previous research on mangrove encroachment along the Florida Gulf Coast (e.g., Krauss 2011), Raabe et al. (2012) implicated sea-level rise as the major factor in habitat transformation but stressed that urbanization also functioned as an important driver. Raabe et al. (2012) also hypothesized that marsh-dominated mosaics may have persisted well into the 20th century, and that rates of mangrove encroachment may have been much greater than reported at locales where shoreline alteration was most intense.

2.2. Study Sites

For the present study, we selected two tidal wetland sites within Old Tampa Bay, Tampa Bay Estuary (Fig. 1). Today, both wetland tracts support dense and homogenous mangrove forests, and both tracts were recorded and symbolized as salt marsh and/or salt prairie habitats during 1875 CGS surveys (Fig. 3). The Feather Sound study site is on the western shore of Old Tampa Bay within a stretch of coastal strand known as Feather Sound. Utilizing CGS maps, GLO documents, and 1999 land cover data, Raabe et al. (2012:1153) quantified the spatial extent of marsh-to-mangrove conversion at the Feather Sound study area and recorded 95 percent non-mangrove to mangrove conversion between CE 1875 and 2000—the most dramatic conversion documented within their study (Raabe et al., 2012:1153). The Philippe Point Tidal Creek study site is situated along a relic tidal creek east of Philippe Point—a recurved sand spit at the northwestern end of Old Tampa Bay. At Philippe Point, CGS surveyors symbolized areas bordering the tidal inlet as salt prairie habitat bounded by upland hammocks. While Philippe Point Tidal Creek fell outside the purview of previous comparative GIS-analyses, the history of this wetland tract contrasts markedly with that of nearby tidal wetlands in Upper Tampa Bay—which have not been intensively altered and exhibit the smallest extent of wetland conversion (52 %) reported for the Tampa Bay Estuary by Raabe et al. (2012:1153).

3. Methods

To empirically reconstruct 20th century habitat transformation and ground-truth the wetland conversion phenomenon inferred through remote sensing, we collected sediment core samples from each study site and conducted sedimentological and macrofossil analyses. Particle size distributions and total organic matter values,

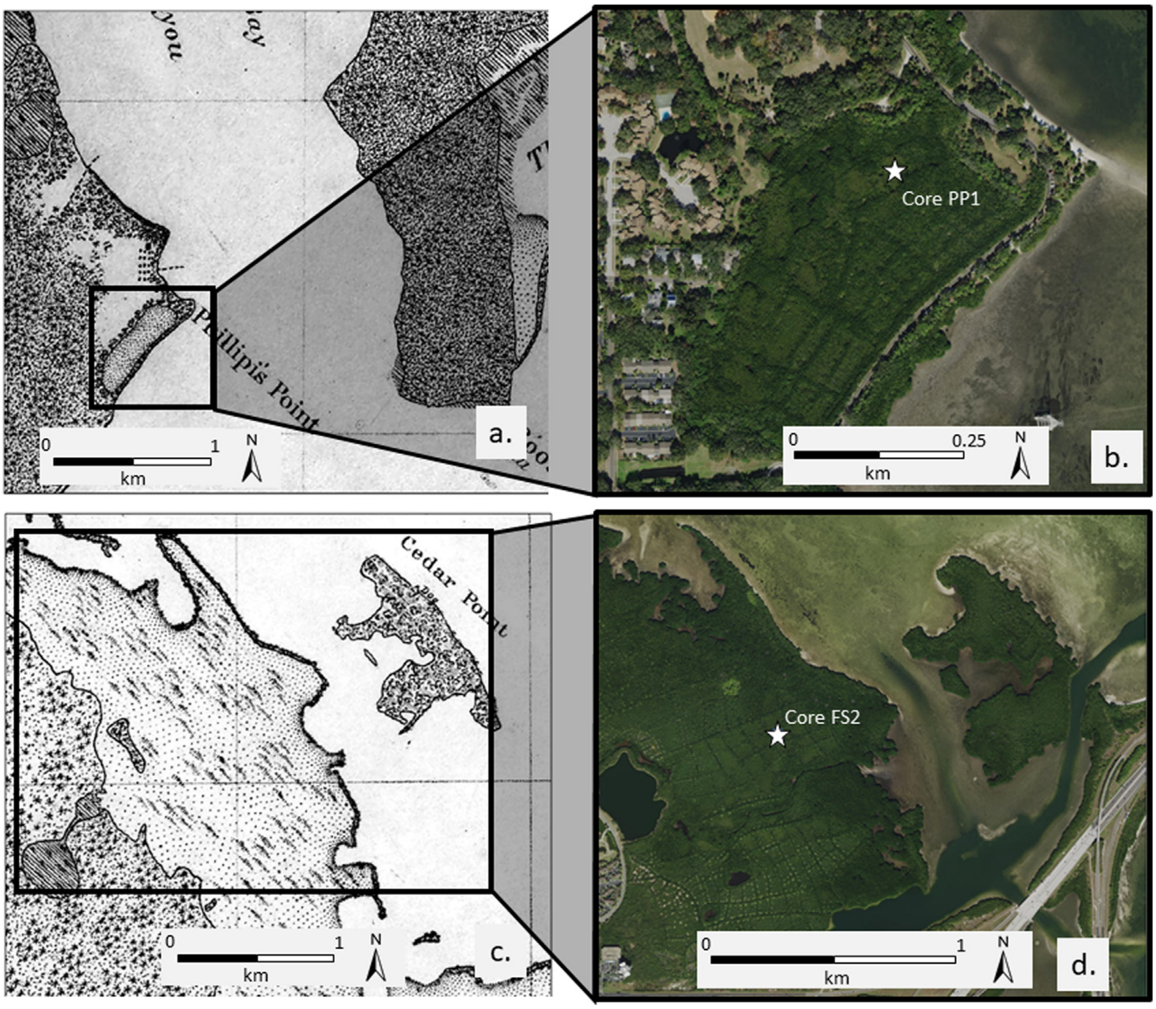


Fig. 3. (a) 1875 T-Sheet of Safety Harbor basin with Philippe Point Tidal Creek symbolized as shallow estuarine wetland habitat; (b) 2018 aerial photography of Philippe Point Tidal Creek study site showing homogenous mangrove forest and the location of core PP-1; (c) 1875 T-Sheet Map of Feather Sound with tidal wetlands symbolized as vast salt marsh fringed by shoreline mangroves; (d) 2018 aerial photography of Feather Sound showing homogenous mangrove forest and the location of core FS-2.

as well as botanical and molluscan macrofossil identifications, were utilized to demarcate sedimentary facies representing distinct depositional environments within the wetland sediment records. Stratigraphic interpretations were aided by qualitative and quantitative comparisons with a set of modern reference sediment samples that were collected within mangrove, salt marsh, and salt prairie habitats. Short-lived radio isotope (210-PB/137-CS) dating and Accelerator Mass Spectrometry (AMS) 14-C dating were integrated to establish a well-resolved, decade-scale stratigraphic chronology for habitat transformation at one of the two study sites. To contextualize the sediment records within larger seascape-scale processes and elucidate major factors that drove wetland conversion in our study areas, we analyzed sequences of aerial photography spanning the middle and late

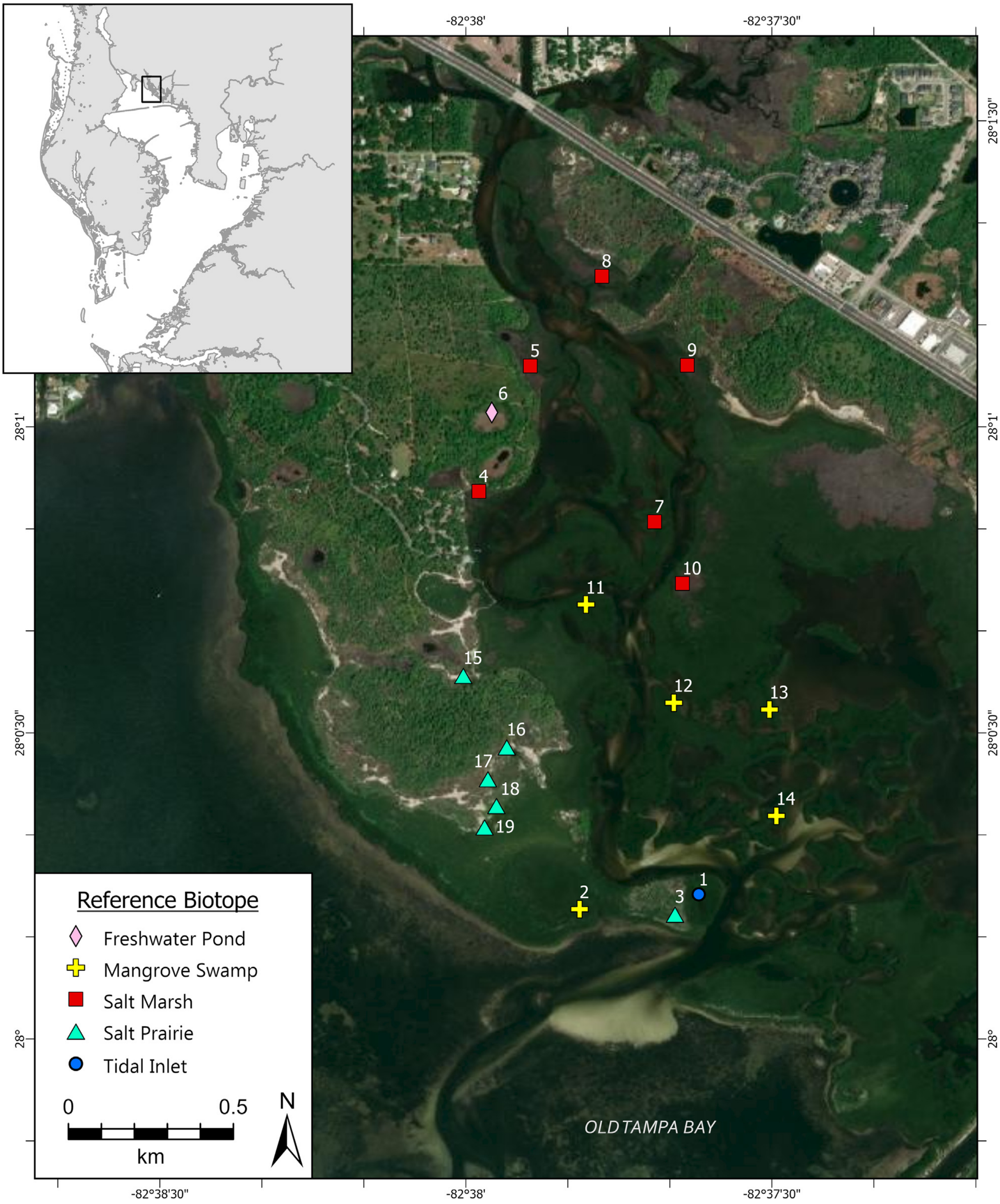


Fig. 4. Locations of reference surface sediment samples, Upper Tampa Bay Nature Preserve.

20th century (1940–1997). The synthesis of sedimentary records with aerial photography chronosequences enabled us to associate shifts in depositional environment preserved in the sub-surface with evidence of 20th century seascape alteration and patterned mangrove encroachment captured in aerial photographs.

3.1. Collection of Reference Sediments

We collected and analyzed surface sediment samples from distinct wetland habitats in Tampa Bay Estuary to develop a reference dataset of physical sediment characteristics and macrofossil constituents for regional salt marsh, salt prairie, and mangrove forest *biotopes* (habitats with consistent structure and biota). Each sample was collected from the uppermost centimeter of the surface sediment bed and processed to yield percent total organic matter values and particle size distribution data. These data were tabulated to produce descriptive statistics that were later compared with data from core samples to aid the interpretation of stratigraphic records. Reference specimens of dominant flora (particularly rhizome, root, and shoot tissues) were also collected to help facilitate the identification of diagnostic botanical macrofossils preserved within core records. Fig. 4 displays the locations of reference surface sediment samples collected for this study.

Salt marsh habitats within Tampa Bay Estuary are dominated by black needle rush (*Juncus roemerianus*) and generally occupy elevations inundated only during high spring tides and storm-driven flooding (see Clewell, 1997; Montague and Wiegert, 1990; Stout, 1984). Salt prairies (also called salturns or salt barrens) are shallow landward basins or flats where seawater pools and evaporates following high-water events; these haline zones are inhabited by a restricted association of flora, including *Salicornia* sp., *Batis* sp., *Monoanthochloe littoralis*, and *Limonium carolinianum* (see Choi et al., 2001; Leonard et al., 2012:720; Montague and Wiegert, 1990). Today, salt marsh and salt prairie biotopes in Tampa Bay are few, and the limited distribution of these habitats within Old Tampa Bay constrained reference sampling to sites within Upper Tampa Bay Nature Preserve—a 2.4 km^2 preserve managed by Hillsborough County encompassing Double Branch Creek, Double Branch Bay, and Cabbagehead Bayou. Mangrove forests on Florida's central peninsular Gulf Coast are principally composed of four taxa, including: *Rhizophora mangle*, *Avicennia germinans*, *Laguncularia racemosa*, and *Conocarpus erectus* (Pool et al., 1977). The dominant intertidal mangrove taxa in Tampa Bay—*R. mangle* and *A. germinans*—trap and accumulate organic detritus and mineral sediment within complexes of aerial roots and pneumatophores, which themselves are known to make up a substantial peat constituent (see Mckee et al., 2007). Today, mangrove forests are ubiquitous along the Tampa Bay shoreline. To maintain comparability among reference samples, however, we collected surface sediments from mangrove swamps within Upper Tampa Bay Nature Preserve.

3.2. Tidal Wetland Sediment Coring

We took sediment cores at Philippe Point Tidal Creek and Feather Sound in order to describe subsurface stratigraphy and collect physical evidence suitable for evaluating 20th century habitat transformations. We collected cores in overlapping 50 cm sections using a hand-operated Russian-style half-barrel coring device with a sample-chamber measuring 5 cm in diameter. This style of chamber corer was ideal because of its ability to collect core sections without introducing contaminants or vertically compacting the sediment profile (Jowsey, 1966; Skilbeck et al., 2017:109–110). Several cores were collected along transects within each study area to understand local stratigraphy; one particularly representative core sample from each study area was selected for detailed description, sub-sampling, and sedimentological analyses. Core sections were photographed and described in the field to record sediment coloration before the anaerobic soils oxidized; photography and facies descriptions were repeated in the lab under controlled lighting. Loss-on-ignition samples were collected at discrete 5 cm intervals throughout the length of each core. Bulk sediment samples were collected in 5 cm increments for particle size analysis. Macrofossils (shell and botanical remains larger than 500 μm) were sampled during facies description and during wet sieving to strengthen stratigraphic interpretations. Taxonomic identifications of macrofossils utilized a type collection of regional estuarine mollusks and reference macrobotanical material from surface samples.

3.3. Loss-on-Ignition and Particle Size Analyses

Total organic matter (TOM) values for surface samples and core samples were derived by loss-on-ignition following established methods (Davies, 1974; Dean, 1974) and more recent experimentation on marine and estuarine sediments by Wang et al. (2011). Crucibles were prepared, weighed, and loaded with 1.5 cm^3 of sediment before being weighed again and dried at 60 °C for 48 h. After measuring dry sediment weights, we combusted samples in a muffle furnace for 12 h at 550 °C, yielding TOM values. Particle size analysis involved measuring wet and dried sample weights, and then wet sieving samples through 63 μm (4 ϕ) mesh. Coarse fractions were dried, weighed, and processed through nested sieves to yield percent-composition values for sand-sized grains at whole phi intervals (-1, 0, 1, 2, 3, 4 ϕ) (Poppe et al., 2000). Fine fractions were processed via pipette extraction to yield silt/clay split percentages following Folk (1965) and Poppe et al. (2000). Mean grain size and sorting values were calculated by weighted averaging; mean size characterizations were made following Wentworth (1922) and sorting characterizations were made following Folk (1974) (Tables 1 and 2).

3.4. Radiometric Chronology for Philippe Point Core 1 (Core PP-1)

We established the chronology for the depositional sequence within core PP-1 with short-lived radioisotope dating using 210-Pb and 137-Cs, as well as a single AMS radiocarbon date on a basal wood macrofossil. Short-lived radioisotope geochronology was developed for excess 210-Pb and 137-Cs. Samples were run on a GWL Series HPGe (High-Purity Germanium) Coaxial Planar Photon Detectors for total 210-Pb (46.5 keV), 214-Pb (295 keV and 351 keV), 214-Bi (609 keV) and 137-Cs (661 keV) activities. Data were corrected for counting time, detector efficiency and geometry, as well as for the fraction of the total radioisotope measured yielding activity in DPM/g (disintegrations per minute per gram). The determination of detector efficiency utilized similar methods as Kitto (1991) with the IAEA 447 standard. A calibration template was produced relating the counts measured to the known activity of the standard for the range of sample weights.

Cesium-137 is a thermonuclear byproduct and represents the period of greatest atomic bomb testing in the early-mid 1960s

Table 1
Wentworth (1922) scale for particle size description.

Particle Size	ϕ	mm
Coarse Sand	(-1) – 1	2 – 0.5
Medium Sand	1 – 2	0.5 – 0.25
Fine Sand	2 – 3	0.25 – 0.125
Very Fine Sand	3 – 4	0.125 – 0.0625
Silt	4 – 8	0.0625 – 0.0039
Clay	> 8	<0.0039

Table 2
Folk (1974) scale for particle size sorting description.

Sorting	Stdev (ɸ)
Very Well	<0.35
Well	0.35 – 0.5
Moderately Well	0.5 – 0.71
Moderately	0.71 – 1
Poorly	1 – 2
Very Poorly	2 – 4
Extremely Poorly	>4

(Olsson, 1986). Excess 210-Pb (t1/2 22.3 years) is used for dating over the last c. 100 years. The activities of 214-Pb (295 keV), 214-Pb (351 keV), and 214-Bi (609 keV) were averaged as a proxy for the 226-Ra activity of the sample or background 210-Pb. Background 210-Pb was subtracted from total 210-Pb to determine excess 210-Pb (Holmes, 2001). Excess 210-Pb data were input into the constant rate of supply (CRS) model to provide dating of each sample analyzed within the last c. 100 years (Appleby and Oldfield, 1983; Binford, 1990). This was compared to 137-Cs data, an independent dating technique, to determine how well the CRS model was performing. Mean accumulation rates (MAR) (g/cm^2/yr) were calculated for each data point (i.e., date) produced by the CRS model. The use of MAR corrects for differential sediment compaction down core, thereby enabling a direct comparison of 210-Pb accumulation rates throughout the core (i.e., over the last c. 100 years). Short lived radio isotope dating model parameters, data, and graphs are presented within Supplementary Data Items 1 and 2.

Additionally, we collected a fragment of wood located at 113 cm depth within core PP-1 (total core depth =125 cm). The Center for Applied Isotope Studies at the University of Georgia conducted the AMS radiocarbon analysis.

3.5. Aerial Photography and Historical Maps

High-resolution aerial photographs of the study areas were available openly from the Florida Department of Transportation A-PLUS aerial photography archive We obtained digital image files from 1940 to 2019. Oblique aerial photography of the study areas was sourced from the Robert Graber aerial photography archive maintained by University of South Florida Digital Collections. Coast and Geodetic Survey topographic charts were available from the National Oceanic and Atmospheric Administration (NOAA) Geodetic Survey Information Center. We digitized and georeferenced all maps and aerial photographs in Esri ArcMap 10.6 using stable landmarks.

4. Results

4.1. Surface Sediments

Reference surface sediment data demonstrate that tidal wetland habitats within Old Tampa Bay may be differentiated from one another by physical sedimentological characteristics, which result from complex interactions between energy regimes, tidal inundation, vegetation community dynamics, and elevation. Mangrove forest surface sediments were highest in total organic matter (mean = 35.7 %) and fine particle content (mean = 28.4 %). Salt prairie habitats contained very low organic matter (mean = 1.7 %) and fine particle content (mean = 5.5 %). Salt marsh surface sediments exhibited intermediate values for total organic matter (mean = 10.5 %), and fine particle content (mean = 11.9 %). Tables 3 and 4 present particle-size distribution and TOM data for reference surface samples analyzed for this study.

Table 3
Provenience and sedimentological data for surface reference samples.

Habitat	Sample ID	Macrofossils (% Weight)	Sample Location (17 N)		∑ Sand	Silt	Clay	∑ Mud	x̄ Size (ɸ)	% LOI	σ Sorting (ɸ)	x̄ Grain Size	σ Sorting
			x	y	< 4ɸ	4–8ɸ	> 8ɸ	> 4ɸ					
Small Tidal Inlet	1	1.6	0243098	3098705	86.6	6.4	2.4	8.7	3.0	2.7	1.3	Medium Sand	Poor
Creek Channel	21	0.0	7123087	3087473	87.5	1.6	1.2	2.7	2.2	0.9	1.1		
Salt Marsh (J. rom.)	4	4.0	4513099	3099929	92.2	3.2	2.2	5.4	2.9	6.6	1.2	Medium Sand	Poor
	5	4.9	5943100	3100305	81.0	10.8	3.6	14.4	3.1	7.2	1.6		
	7	33.6	9213099	3099831	24.8	17.5	7.5	25.0	2.4	28.6	2.4	Fine Sand	Very Poor
	8	7.1	7893100	3100573	87.2	4.9	2.2	7.1	2.6	7.3	1.3	Medium Sand	Poor
	9	6.0	0143100	3100302	85.9	4.3	1.7	6.1	2.6	6.6	1.2		
	10	7.6	9933099	3099644	63.2	10.9	2.7	13.6	2.5	6.7	1.6		
Mangrove Swamp	2	16.2	7043098	3098664	58.3	23.0	8.6	31.6	3.7	17.8	2.1	Fine Sand	Very Poor
	11	42.5	7343099	3099584	38.0	14.0	5.1	19.1	2.4	32.5	2.1		
	12	36.6	9653099	3099284	44.2	19.4	5.9	25.3	3.0	49.4	2.0		
	13	52.8	2213099	3099260	33.6	15.0	7.2	22.2	2.5	28.0	2.2		
	PP 2	61.4	0053098	3098952	8.3	22.4	12.4	34.8	2.6	56.8	2.8		
	14	26.0	2353098	3098939	42.5	28.7	8.8	37.4	3.8	29.7	2.0		
Salt Prairie	3	0.9	9593098	3098648	92.7	5.3	1.6	6.9	2.8	2.3	1.2	Medium Sand	Poor
	15	0.1	4023099	3099376	93.7	3.4	2.0	5.4	2.7	1.9	1.2		
	16	0.3	5163099	3099158	95.5	0.2	1.5	1.7	2.5	0.6	1.1		
	17	0.3	4643099	3099063	91.5	2.6	1.3	3.9	2.5	1.8	1.0		
	18	0.7	4863098	3098982	92.6	3.6	1.3	4.9	2.6	1.8	1.0		
	19	0.2	4533098	3098918	91.8	6.6	3.7	10.2	3.0	1.7	1.5		
Seagrass Meadow	20	0.9	1253088	3088437	96.1	2.5	1.6	4.1	3.0	1.0	1.1	Medium Sand	Poor
22	0.1	1573087	3087441	95.2	3.8	1.9	5.7	2.8	1.2	1.2		
Freshwater Marsh	6	36.8	4893100	3100167	51.5	7.2	1.8	9.0	2.0	49.7	1.5	Medium Sand	Poor

Table 4
Descriptive statistics for surface reference samples by environment type.

Habitat	n=	TOM (%)		∑ Sand (%)		Silt (%)		Clay (%)	
		x̅	σ	x̅	σ	x̅	σ	x̅	σ
Mangrove Swamp	6	35.7	14.6	37.5	16.6	20.4	5.5	8.0	2.6
Salt Marsh	6	10.5	8.9	72.4	25.4	8.6	5.5	3.3	2.1
Salt Prairie	6	1.7	0.6	93.0	1.5	3.6	2.2	1.9	0.9
Seagrass Meadow	2	1.1	0.1	95.6	0.6	3.2	0.9	1.8	0.2

4.2. Core PP-1 Stratigraphy

Core PP-1 captured a stratified sedimentary record extending 125 cm below sediment surface before refusal in compact quartz sand (Fig. 5). Quartz sands at the base of the core sample (115–125 cm) are overlain by organic muddy sand containing oyster (*Crassostrea* sp.), hooked mussel (*Geukensia* sp.), and barnacle (*Balanus* sp.) shell (60–111 cm). Notably, the muddy sand unit lacks mangrove macrofossils and contains numerous monocot leaf fragments. Particle size distributions and TOM values for this stratum are consistent with reference *J. romerianus* marsh habitats (Tables 3 and 4). We interpret the lower sections of PP-1 to represent the establishment of high-intertidal salt marsh habitat atop a former salt pond or salt prairie environment. Salt marsh-associated sediments (60–111 cm) grade into an organic sandy mud stratum (30–60 cm) containing mangrove macrofossils; and from 30 cm to the sediment surface mangrove root and leaf tissues constitute much of the sedimentary matrix. Sedimentological data from this upper core section align with reference samples from mangrove forest soils and exhibit a fining-downward sequence (0–45 cm) that is characteristic of mangrove peat profiles. Supplementary Table 1 reports TOM and particle size distribution data for core PP-1.

A wood fragment collected at the transition between the quartz sand bed (salt pond/prairie deposit) and the organic muddy sand (salt marsh) stratum (113 cm) yielded an AMS radiocarbon age of 170 ± 20 years BP (UGAMS No. 41446). This age corresponds to a median date of calibrated CE 1746 at 95.4-percent probability (δ13C,‰= -23.55, pMC = 97.89 ± 0.27) (OxCal 4.3, IntCal 13) (Reimer et al., 2013). While pre-bomb radiocarbon assays from the past few hundred years should be considered with caution, this data point corresponds well with the 210-Pb/137-Cs chronology and suggests that this transition likely took place during the late-18th or early-19th century. The short-lived radioisotope chronology for Core PP-

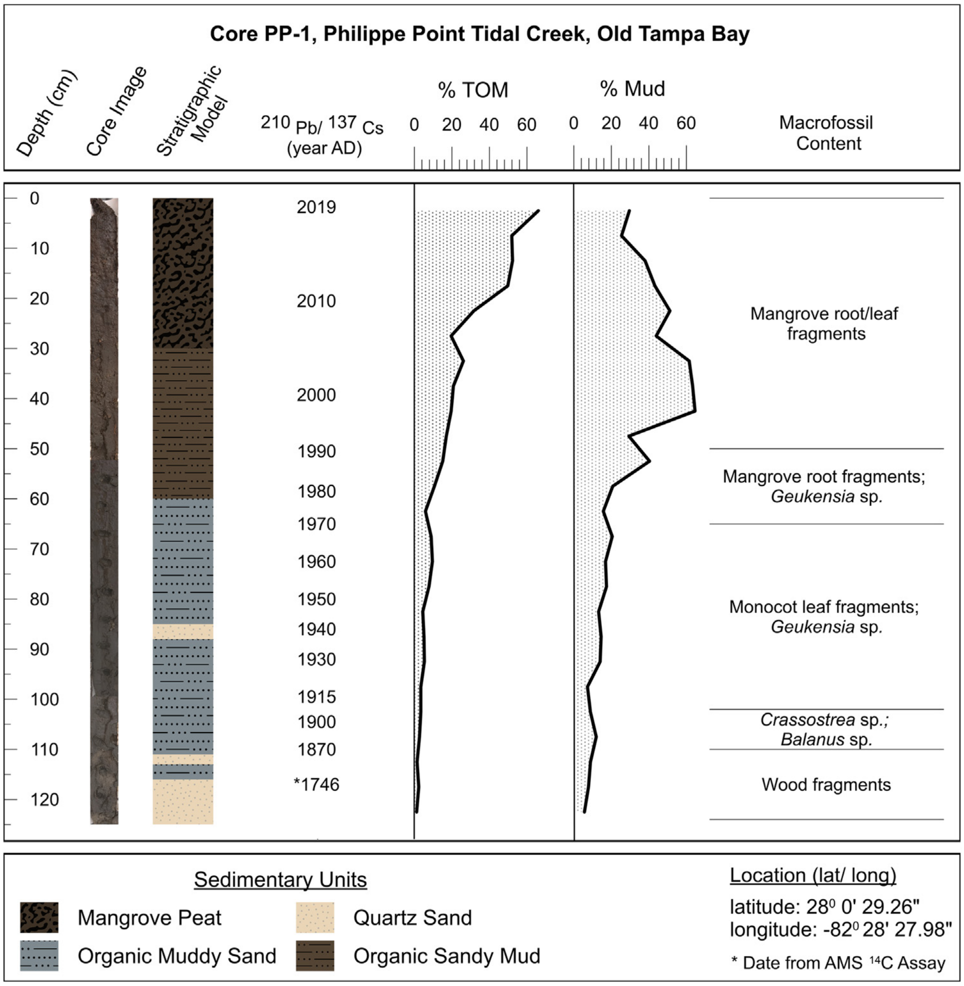


Fig. 5. Stratigraphic, chronometric, and sedimentological data from Core PP1.

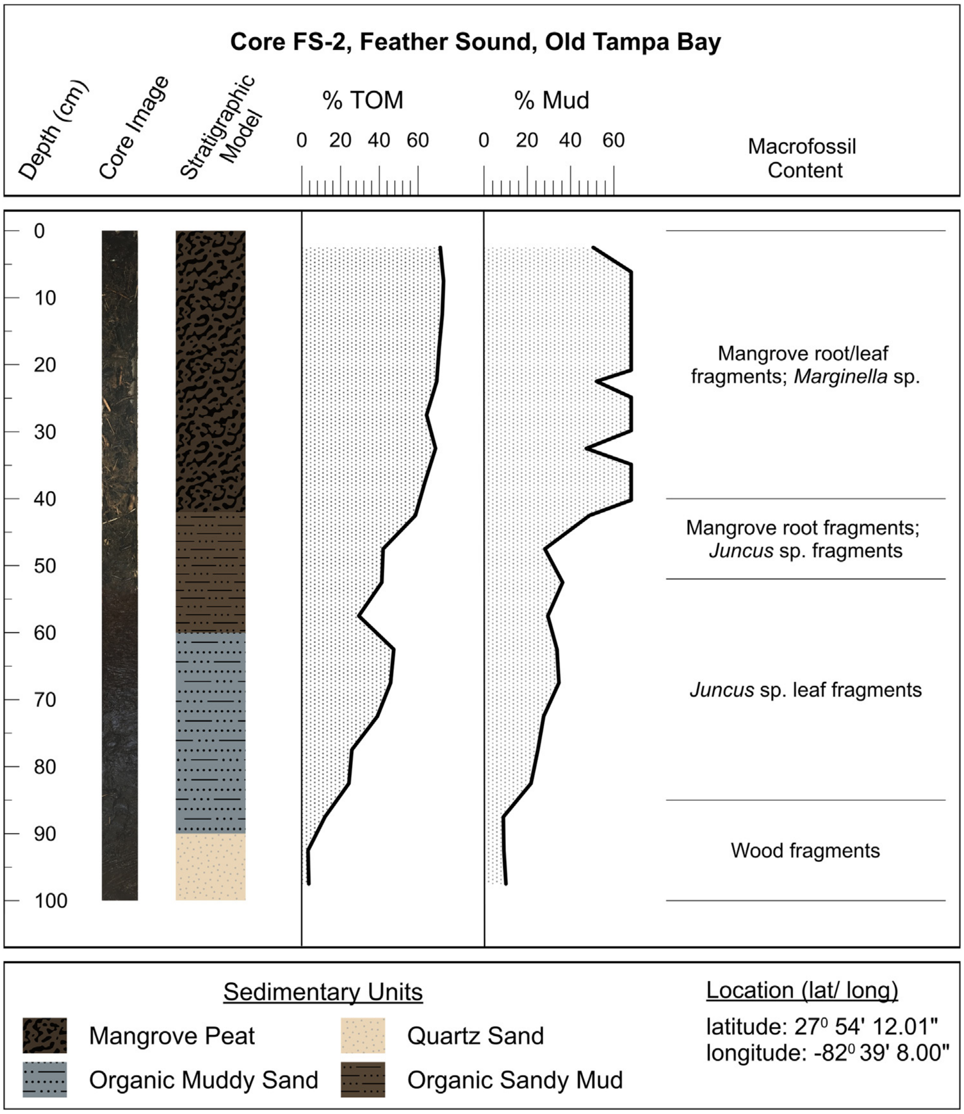


Fig. 6. Stratigraphic and sedimentological data from Core FS-2.

1 suggests that salt marsh-associated sediments accumulated at this locale from the early-19th century until the mid-1970s, after which mangrove-associated sediment (peat) becomes dominant.

4.3. Feather Sound Core 2 (Core FS-2) Stratigraphy

Core FS-2 was driven to 100 cm sediment depth before refusal and recovered a stratified record consistent with the record observed in core PP-1 (Fig. 6). Quartz sand at the base of the core (90–100 cm) is overlain by organic muddy sand containing *Juncus* sp. macrofossils (60–90 cm). As in PP-1, the physical characteristics of the gleyed organic muddy sand bed in core FS-2 are most consistent with reference salt marsh habitats. A transition to mangrove peat is apparent at 60 cm below surface in core FS-2. This stratum exhibits greater TOM, greater percent-composition of fine particles, and contains abundant mangrove macrofossils. Supplementary Table 2 reports TOM and particle size distribution data for core FS-2. Although we did not conduct radiometric analyses on core FS-2, the record is stratigraphically consistent with core PP-1 and encourages the tentative application of the chronosequence reported above.

4.4. Historical Maps and Aerial Photography

The earliest aerial photographs of Philippe Point from which the vegetation cover may be deciphered were from 1942 (Fig. 7a). In this image, the tidal creek west of the point refracts strikingly. A road is apparent along the shoreline, separated from the wetland by a barrow-ditch that closely parallels the roadbed. Additionally, a shallow linear (NW/SE) ditch is apparent through the wetland. The pale color and smooth-homogenous texture of the wetland's ground cover serve as clear indicators of high salt marsh/ salt prairie habitat surrounding the tidal creek channels. The same vegetation cover remains dominant in the 1957 aerial photograph (Fig. 7b), though mangroves (dark, textured areas) are apparent along the roadside barrow-ditch and sporadically along the banks of the tidal creek. The most conspicuous features of the Philippe Point Tidal Creek by 1962 (Fig. 7c) are numerous interconnected mosquito-control ditches (and associated spoil piles) that were excavated by the Pinellas County Mosquito Control Board with the intention to increase tidal flushing of high-marsh habitats and destroy breeding environments for biting insects. At Philippe Point tidal creek, ditches were most intensively dug across the high salt

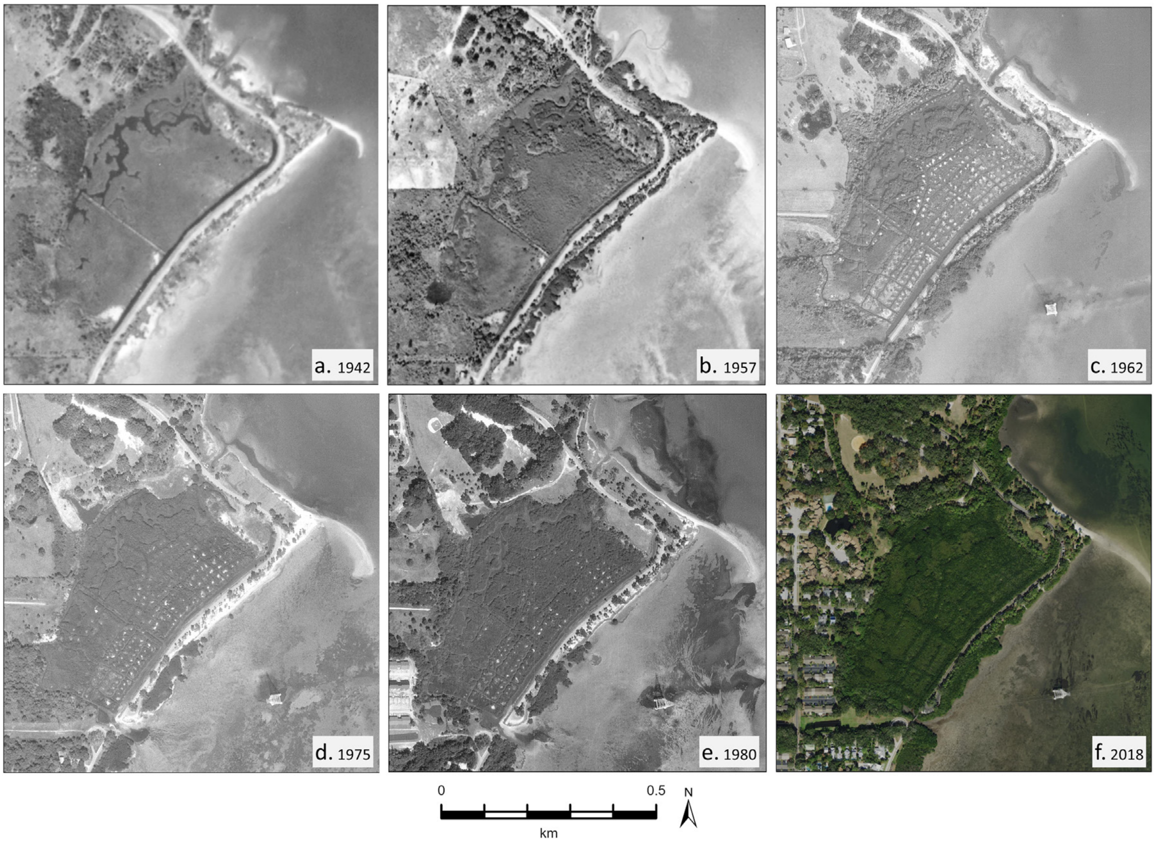


Fig. 7. Historical and modern aerial photography of Philippe Point Tidal Creek study site. (a) 1942 aerial imagery depicting tidal creek system, intact salt marsh (smooth, light grey), and recently excavated linear barrow ditch along roadway; (b) 1957 imagery showing limited mangrove fringe along the banks of the tidal creek system and barrow ditch; (c) 1962 photograph depicting recently excavated system of mosquito control ditches and associated spoil piles; (d) 1975 photograph depicting mangrove establishment (dark, textured areas) along mosquito ditches and small patches of salt marsh (smooth grey) remaining between ditches; (e) 1980 imagery showing continued mangrove encroachment across the wetland tract and the absence of salt marsh; (f) 2018 imagery depicting the modern homogenous mangrove forest at Phillipe Point Tidal Creek.

marsh/salt prairie habitat and connected the tidal creek meanders to the roadside barrow ditch—which was concurrently expanded roughly two-fold in width. Vegetation cover is more difficult to assess in the 1962 photograph due to the high spatial density of ditches and spoil piles. The interiors of rectangular inter-ditch areas, however, do not appear to support arboreal cover. In the 1975 aerial imagery (Fig. 7d), however, the ditched wetland is highly textured and appears to support relatively dense mangrove cover. Notably, this is also the case across the interfluvial lobes and creek shorelines lying north and west of the most intensive ditching. By 1980 (Fig. 7e) no visible evidence remains for stands of salt marsh or salt prairie, and the creek system appears dominated by homogenous mangrove forest. A modern aerial photograph of the study area (Fig. 7f) shows that mangrove proliferation has continued since ca. 1980, and *A. germinans—R. mangle* forest remains the dominant habitat across the tract.

A detailed 1943 aerial photograph of the Feather Sound tidal wetlands (Fig. 8a) readily conveys the character and distribution of vegetation cover. Much like the ca. 1875 CGS T-Sheet, mangrove forest (highly textured, dark grey) in 1943 was confined to the bay shoreline and the lower portions of several small tidal creeks. The texture and color value of the expansive interior wetland tract is consistent with salt marsh. West of the marshes lie scattered, sparsely wooded hammock areas (textured with individual trees visible) and broad salt barrens (white, smooth) with small ponds. The 1957 aerial imagery (Fig. 8b) shows very little change in wetland character or distribution since 1943. A road and barrow-ditch are apparent traversing the southern portion of the wetland tract, and some expansion of mangrove forest has occurred along low-energy portions of the bay shoreline. By 1962 (Fig. 8c), the Feather Sound tidal wetland tract had been intensively ditched for mosquito control, Cedar Point (or ‘Big Island’) has been dredged for construction fill, and the western causeway for the Howard Franklin bridge had been constructed on the southern end of the study area. A review of archived aerial photography by noted St. Petersburg photographer, Robert Graber, shows that Feather Sound wetlands were ditched in at least two stages between March (Fig. 9a) and November (Fig. 9b) of 1959. Similar to Philippe Point, and most other coastal wetlands around Tampa Bay, ditching was most extensive across high-marsh environments and extended through salt barrens to intersect small salt ponds. Blocks of inter-ditch wetland are larger at Feather Sound than at Philippe Point and in the 1962 aerial imagery intact swaths of salt marsh remain between the ditches. By 1980, however, dark and textured areas representing mangrove forest are apparent throughout the Feather Sound tidal wetland (Fig. 8d). Mangroves appear concentrated along ditches, but also have colonized many rectilinear inter-ditch wetland patches. An aerial photograph from 1997 (Fig. 8e) lacks evidence for intact salt marsh or salt prairie stands and contains only highly textured areas representing homogenous mangrove forest. A 2018 aerial image (Fig. 8f) demonstrates the continued expansion and proliferation of mangrove forest across the Feather Sound tidal wetlands into the 21 st century.

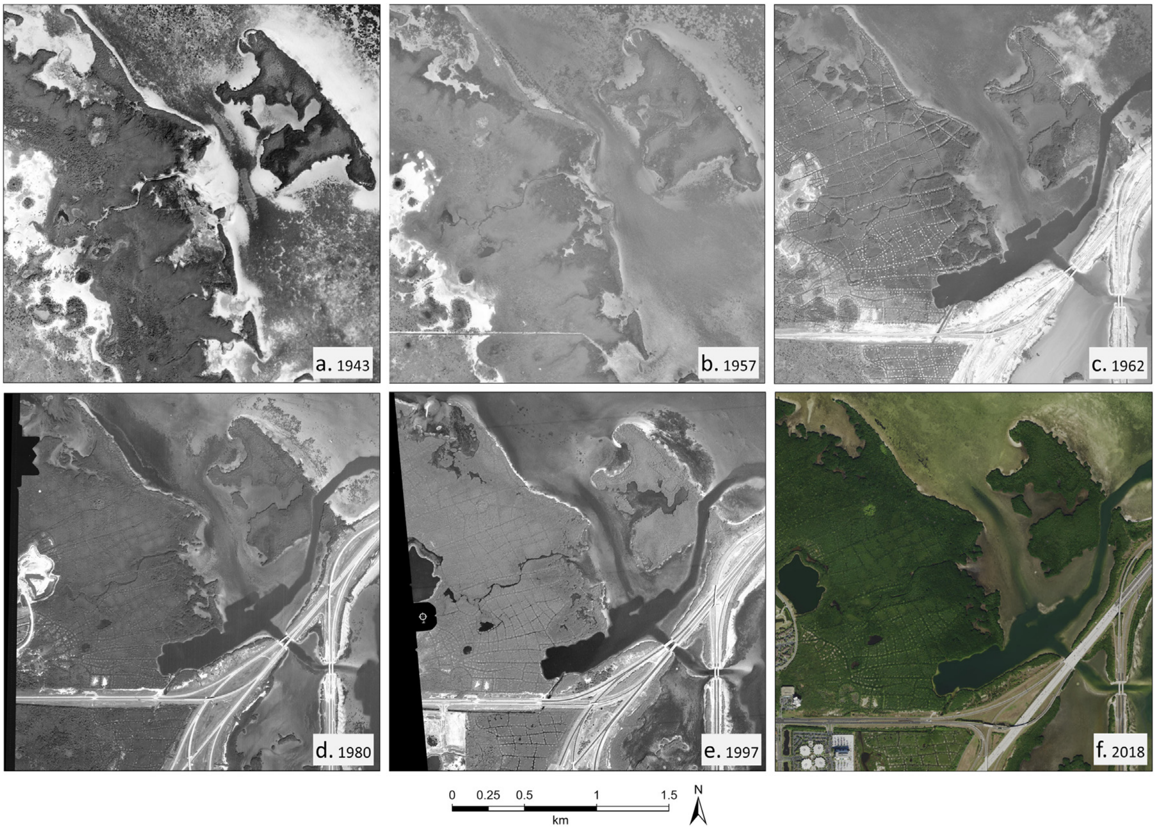


Fig. 8. Historical and modern aerial photography of Feather Sound. (a) 1943 photograph depicting large areas of salt marsh (smooth light grey) and salt prairie (landward, smooth white to light grey areas) and a shoreline fringe of mangrove forest (seaward, dark, textured areas); (b) 1957 imagery showing continued presence of wetland habitats documented in 1943 photograph; (c) 1962 photograph depicting the alteration of the wetland tract by mosquito control ditching, seaward dredging, and filling for causeway construction; (d) 1980 imagery showing mangrove encroachment throughout the wetland tract and most dramatic along ditches; (e) 1997 photograph depicting homogenous mangrove forest throughout the wetland tract; (f) 2018 imagery showing modern homogenous mangrove forest habitat extending from the bay shoreline to residential development to the west.

5. DISCUSSION

Ground-truthing previous work by Raabe et al. (2012), this study confirms the inference that Tampa Bay's coastal wetlands converted from salt marsh/salt prairie mosaics to mangrove forests over the 20th century. The PB-210/ Cs-137 chronology on core PP-1 sediments indicates that marsh-to-mangrove conversion began to substantially alter sedimentary conditions at Philippe Point Tidal Creek in the 1970s. Such alterations culminated in the establishment of homogenous mangrove forest by 1980. A nearly congruent stratigraphic sequence in core FS-2 suggests that a similar seascape-history played out within the tidal wetlands of Feather Sound. Environmental changes preserved within the sedimentary record at both study sites correlate closely with vegetation-community change apparent in sequences of aerial photography spanning the mid- to late-20th century. Below, we identify the likely primary drivers of wetland conversion in the study areas and discuss the implications of coupled anthropogenic and biophysical impacts for the conservation and management of coastal wetlands.

5.1. Twentieth century climate and sea-level change

Latitudinal climatic variation broadly regulates mangrove distribution on the Florida peninsula. Mangrove forest is the dominant intertidal habitat type in the sub-tropical zone of south Florida, whereas *Juncus* marsh is dominant in the more temperate climate along the northern peninsula and eastern panhandle (Kangas and Lugo, 1990). Tampa Bay Estuary is situated at the boundary between these climate zones and has supported both *Juncus* marsh and mangrove forest since at least the mid-late 19th century (Clewell, 1997; Kangas and Lugo, 1990; Montague and Weigert; Stevens et al., 2006), and likely throughout the late-Holocene (e.g., Gerlach et al., 2017). The well-demonstrated influence of climate over coastal biogeography (see Duke et al. 1988; Walsh 1974) suggests that shifting temperature over the 20th century—particularly warming winters—may have driven marsh-to-mangrove conversion on the central peninsular Gulf Coast (Osland et al., 2013). As Raabe et al. (2012) reported, however, while significant average annual warming and winter-season warming has been documented in the Gulf of Mexico region since 1975 (NASA, 2020), this trend has not been observed in the Tampa Bay area (Jones and Moberg, 2003). Winter (December - February) temperature data recorded at the NOAA weather station at Tampa International Airport from 1939 through 1997 showed that seasonal temperatures did not substantially change across the interval of tidal wetland conversion (Fig. 10). Further, days with freezing temperatures (less than or equal to 0 °C) did not decrease in frequency over the mid-late 20th century, but instead became slightly more frequent over the interval of mangrove expansion (Fig. 11).

The dominant mangrove taxa in Tampa Bay, *R. mangle* and *A. germinans,* are well-adapted to recruit and proliferate within frequently inundated low-intertidal positions (Kangas and Lugo, 1990; Odum and McIvor, 1990), while *Juncus* marsh and salt prairie vegetation prefer environments with infrequent inundation and freshwater sheet-flow (Clewell, 1997; Montague and Wiegert, 1990). Thus, marsh-to-mangrove conversion may be understood as a fundamentally transgressive process. Rising sea level across the

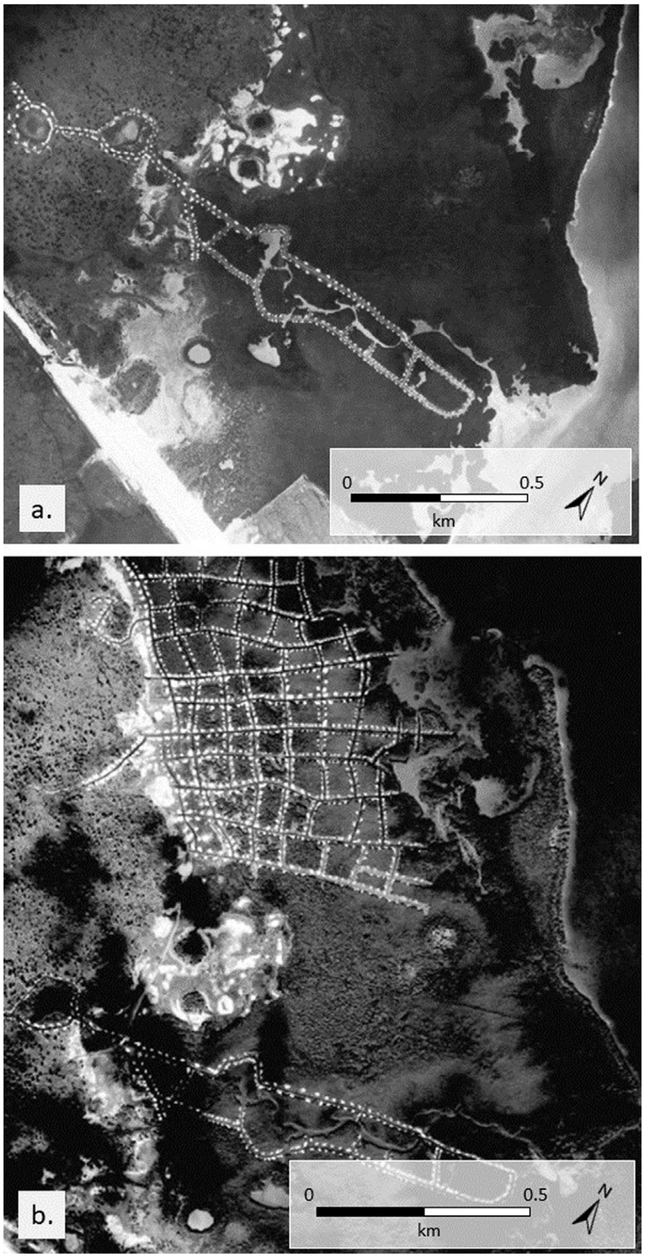


Fig. 9. Two phases of mosquito control ditching at Feather Sound; (a) April 5th, 1959; (b) October 27th, 1959. Photos via USF Digital Collections, Graber Collection.

20th century represents a suitable hypothetical driver of this ecosystem change. On the northern Gulf Coast, Williamson et al. (2011) and Comeaux et al. (2012) have shown relative sea-level rise to function as a major driver of mangrove expansion. Substantial mangrove encroachment into coastal Everglades marshes has also been attributed to contemporary sea-level rise (Krauss et al., 2011; Ross et al., 2000). Sea level in Tampa Bay has risen at an average rate of 2.4 mm/year over the 20th century (NOAA, 2001; Thompson, 2010; Zervas, 2001). Recent multi-proxy sedimentary analyses by Gerlach et al. (2017) at the mouth of the Little Manatee River in southeastern Tampa Bay reported similar results. The increase in sea-level from the most recent period of *Juncus* marsh dominance (CE 1957) to the earliest period of mangrove dominance in our study areas (CE 1980) likely measured approximately 5.5 cm. This trend does suggest some role for sea-level rise in 20th century wetland conversion, but the minor degree of change is insufficient to explain the expansion of mangrove forest hundreds of meters landward across high-marsh and salt prairie environments. As the scale of sea-level rise that has transpired from 1957 to present (2020) is considerably greater (approximately 15 cm), it may represent a primary driver of ongoing mangrove encroachment in areas without evidence of substantial shoreline alteration.

5.2. Mosquito Control Ditching

In the mid- to late-1950s, tidal wetlands in Tampa Bay were ditched as part of large-scale mosquito control efforts associated with infrastructural, commercial, and residential development (Haddad and Joyce, 1997; Smith 2007). Mosquito-control ditches have a deep history on the Atlantic coast, where across the 1930s, Works Progress Administration (WPA) crews excavated ditches to drain vast expanses of high-salt marsh (Corman et al., 2012; LeMap, 2007) (Fig. 12). WPA crews also excavated mosquito control trenches in the Tampa Bay region that connected high-intertidal and freshwater ponds to regular tidal flow. During Florida's 1950s development boom, marsh-breeding mosquitos and biting gnats presented serious obstacles for the expansion of real-estate and tourism markets in the coastal zone (Brothwell, 1956; Patterson, 2004). While entomological research at the time struggled to empirically show that marsh-breeding mosquitos functioned as vectors of infectious disease, and the environmental impacts of mosquito control were becoming widely known among natural scientists, weather-related mosquito proliferation and encephalitis outbreaks in the mid-1950s motivated government officials in the Tampa Bay region to adopt the intensive ditching of coastal marshes as an essential land-development practice (Philen and Carmichael, 1956; Pratt et al., 1963). There was strong political interest in using dynamite to excavate control ditches during the mid-1950s; and indeed, dynamite-lines were used in several experimental ditching operations on Pinellas County barrier islands (Fig. 13). County-owned draglines, however, carried out most of the ditching in Tampa Bay, in tandem with highway- and waterway-related infrastructure work (Fig. 14) (Mitchell, 1956).

The logic of mosquito control ditches was based on eliminating breeding and larval-development habitats for biting insects. Theory and method for reducing such environments varied considerably across the 20th century. In many cases, prime importance was placed on draining high marsh to prevent the accumulation of standing water (Boyd, 1949), whereas in other schemes, the role of small fish consuming larvae in flooded ditches was emphasized (Clement, 1965; Dale and Knight, 2006). Regardless of the specific intended mechanism of mosquito control, ditch networks in the Tampa Bay region have consistently faced serious infilling problems due to sedimentation. By 1956, before the most intensive dragline ditching program began, observers noted that the 1930s WPA ditches at Weedon Island and Mullet Key (Pinellas County) had been encroached and choked-off by mangroves (St. Petersburg Independent., 1956). This same process played out again in the decades following intensive 1950s ditching. By ca. 1980, many of Tampa Bay's mosquito control ditches had closed-off with mangroves and sediment infill and facilitated neither drainage nor access for foraging juvenile fishes (Albright, 1981).

The relationship between mosquito control ditching and mangrove encroachment is complex and involves ecological interactions between altered geomorphological/hydrological conditions, vegetation ecology, and rising sea level. The dominant tidal mangrove taxa in Tampa Bay, *R. mangle* and *A. germinans*, reproduce by dispersing large quantities of viviparous propagules that float on the tide and root themselves once settled onto shallow or exposed sediments (Hogarth, 2005). This dispersal strategy historically restricted mangrove forests to the fringes and seaward

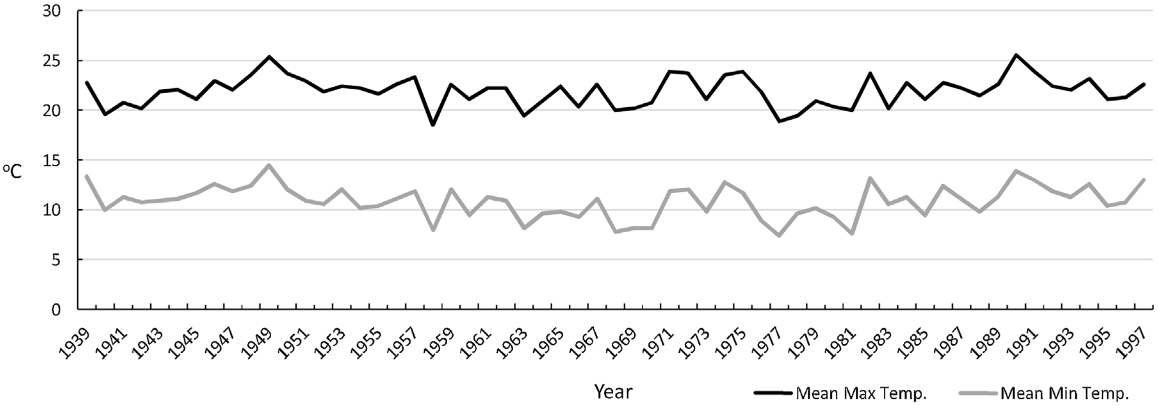


Fig. 10. Mean maximum and minimum winter (Dec.-Feb.) temperatures, 1939-1997 Recorded in Tampa (NOAA, 2020).

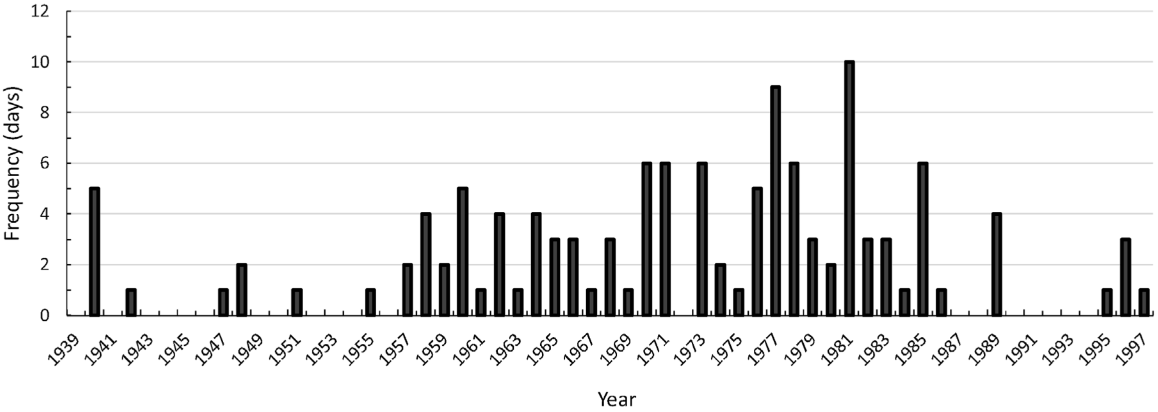


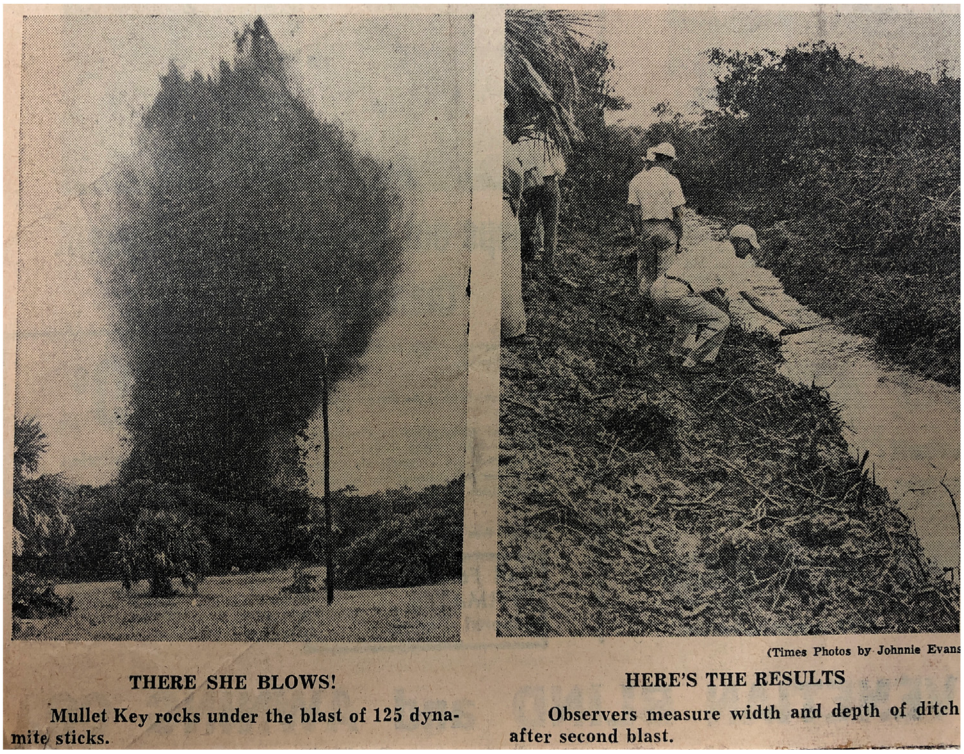
Fig. 11. Days with freezing temperatures (equal to or below 0 °C), 1939-1997 Recorded in Tampa (NOAA, 2020).

Fig. 12. A WPA crew excavates a mosquito-control ditch in Leon County, Florida. ca. 1940s. Photo via Florida Memory archives.

overwash zones of Tampa Bay's coastal wetlands (Raabe et al., 2012). Mosquito ditching created networks of artificial tidal channels cut into high marsh and salt prairie environments, through which propagules were free to disperse and settle. Experimental work by Breitfuss et al. (2003) documented this process on the southeast coast of Queensland by tracking the tidal transport of *Avicennia* sp. propagules into mosquito control ditches and well beyond their normal limits of dispersal. Once mangroves became established along ditch margins, they were well-positioned to rapidly proliferate across the remaining small patches of marsh and salt-prairie. In-between the ditch channels, marsh and salt prairie flora were segmented and unable to maintain networks of rhizomatic reproduction, which require broad continuous areas of marsh (Crain et al., 2008; Stout, 1984). Spoil piles created during the ditching process inhibited the freshwater sheet-flow required for *Juncus* marsh and facilitated the introduction of invasive flora—most notably the Brazilian Peppertree (*Shinus terebinthifolius*) (Smith et al., 2007). Ditches extended tidal shorelines landward, and likely predisposed formerly high-intertidal and supratidal landforms to higher salinities and hydroperiods known to stress marsh flora and facilitate mangrove encroachment (Comeaux et al., 2012; McKee et al., 2012; Pennings et al., 2005; Steinmuller et al., 2020; Williamson et al., 2011).

5.3. Implications for Coastal Management

Mosquito ditching in the mid-20th century led to the elimination of historical salinity and topographical gradients that organized ecological interactions within and across Tampa Bay's coastal wetlands. Once expansive mosaics of marsh and salt-prairie habitat are now highly restricted and occur exclusively in thin strands at the mangrove/terrestrial interface, at river mouths, or in areas that were not intensively ditched for mosquito control. The excavation of ditch networks opened vast stretches of the high-intertidal and supratidal zones to inundation and shifted the estuary shoreline hundreds of meters landward. Today, mosquito ditches directly abut urban zones and may present considerable added risk under storm-surge conditions. Further, the conversion of salt marsh-prairie mosaics to homogenous mangrove swamp wrought unknown alterations of seascape-scale biodiversity,



(Times Photos by Johnnie Evans

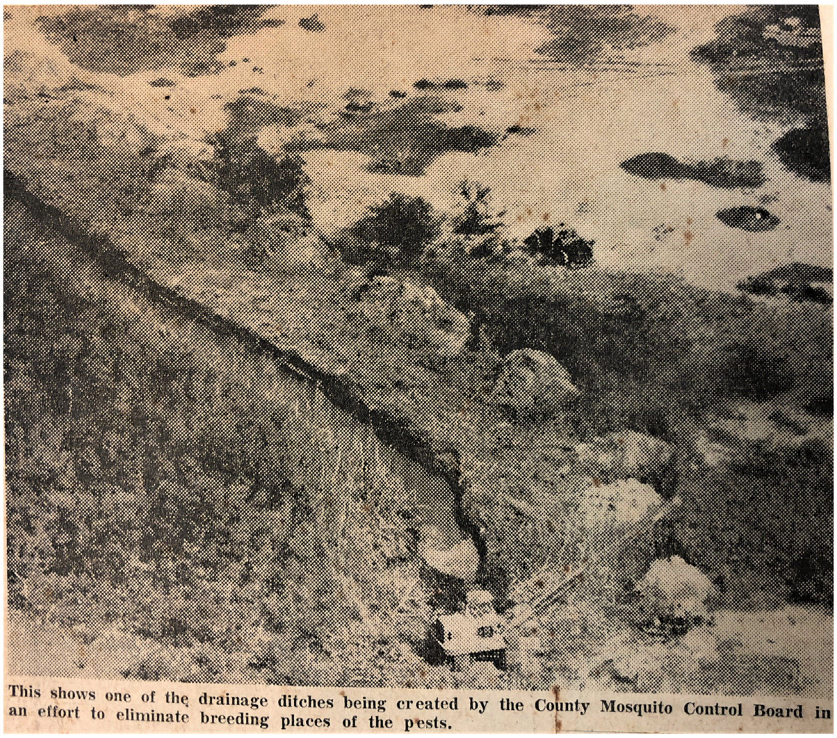
THERE SHE BLOWS!

Mullet Key rocks under the blast of 125 dynamite sticks.

HERE'S THE RESULTS

Observers measure width and depth of ditch after second blast.

Fig. 13. Dynamite-excavation of mosquito control ditches on Mullet Key, Tampa Bay, ca. 1956. Clip from St. Petersburg Times, June 8th, 1956.



This shows one of the drainage ditches being created by the County Mosquito Control Board in an effort to eliminate breeding places of the pests.

Fig. 14. Dragline operation excavating a linear mosquito-ditch in Pinellas County, Florida, ca. 1956. Clip from Tampa Sunday Tribune, February 2nd 1956.

trophic-web complexity, and resilience. Due to a paucity of historical ecological research in the region, little is known about the biodiversity or ecological interactions within the tidal wetland communities that were dominant in Tampa Bay before ditching. However, high-marsh and salt prairie environments on Florida's Gulf Coast provide critical habitats for terrestrial mammals, reptiles, migratory birds, and waterfowl that are not commonly found within homogenous mangrove forest. These fauna include several federally and state-listed species, such as the black skimmer (*Rynchops niger*), American oystercatcher (*Haematopus* *palliates*), wood stork (*Mycteria americana*), bald eagle (*Haliaeetus leucocephalus*), roseate tern (*Sterna dougallii d.*), and least tern (*Sterna antillarum*).

The recent and anthropogenic developmental history of Tampa Bay's mangrove forests does not discount the well-documented ecosystem services that mangrove habitats provide. Comeaux et al. (2012) documented increased mineral-sediment trapping in encroaching mangrove patches on the north peninsular Gulf Coast, perhaps representing improved buffering of coastal wetland inundation. Mangrove forests act as structural impediments to coastal erosion by attenuating energy from swell and surf (Doughty et al., 2016). They are also widely known to provide nursery habitat for various economically and ecologically important infaunal and nekton taxa (Koenig et al., 2007; Whitfield, 2017). Further, recent work in Tampa Bay Estuary by Radabaugh et al. (2018) has demonstrated that the region's mangrove forests sequester blue carbon at greater rates than extant salt marsh and salt-prairie habitats in the region (also see Bianchi et al. 2013; Breithaupt et al., 2012). While studies on the northern Gulf Coast have found that recently established stands of *Avicenna* sp. within *Spartina* sp. marshes are not associated with increased organic matter production or carbon assimilation (e.g., Perry and Mendelssohn, 2009), this study demonstrates that, within ~ 20 years of encroachment, mangrove forests may begin to produce peat soils with total organic content equivalent to native mangrove swamps. These findings agree with the experimental work by Osland et al. (2012). Attempting to revert now mangrove-dominated wetlands to historical marsh/salt prairie conditions is probably not advisable. Experimental plots converted from ditched mangrove swamp areas to salt prairie by leveling spoil piles and filling ditches (Smith et al., 2007) were re-encroached by mangroves within a decade. The prospects of restoration to historical conditions are particularly limited by increasing rates of sea-level rise, which are presently facilitating the landward encroachment of mangroves into high-intertidal habitats even in areas without ditches. In this way, 20th century wetland conversion in Tampa Bay represents a regional scale state-shift of tidal wetlands that cannot be reversed by the simple remediation of the original disturbance—what Lockwood and Lockwood (1993) term *hysteresis*.

As changes in seasonal temperature minima continue to expand the latitudinal range of mangroves into more temperate, marsh-dominated coastlines and estuaries (see Osland et al., 2013), land managers and coastal communities must prepare for ecosystem transfer driven by coupled biophysical and anthropogenic factors. Efforts to model future mangrove encroachment must not only account for sea-level rise and storm surge as mechanisms of mangrove propagule dispersal, but also the influence of artificial shoreline features over the pattern and pace of habitat transformation. Relic mosquito control ditches are ubiquitous features of temperate, marsh-dominated coasts. In these regions, restoration of pre-ditching topographic/ hydrological conditions before mangroves become established may protect large tracts of high intertidal marshes from rapid conversion and prevent losses in biodiversity, interaction, and resilience.

6. Conclusions

Analyses of sediment core samples and sequences of historical aerial photography at two study sites within Old Tampa Bay produced the following answers to the research questions posed in this paper. First, at both study sites (Philippe Point Tidal Creek and Feather Sound) sediment records document relatively dramatic transformations in habitat type. In both cores, gleyed estuarine muddy sand beds containing marsh-associated botanical macrofossils are overlain by mangrove associated organic sandy mud and mangrove-peat deposits. The sedimentary records described here, along with close matches in sedimentological values and macrofossil constituents between core strata and reference wetland surface sediments, confirms the marsh-to-mangrove conversion suggested by previous analyses of historical documents and aerial imagery.

Second, short-lived radio isotope dating of core PP-1 facilitated decade-scale chronological control for tracking changes in the depositional environment across the 20th century. In agreement with a single AMS radiocarbon date at the base of the core record, the 210-PB/ 137-CS chronology indicates that salt marsh habitat was established by the early 19th century. Salt marsh-associated sediments accumulated until the 1970s, when an abrupt transition to mangrove-associated sediment deposition occurred. These radiometric data are supported by analyses of aerial photography sequences that show dramatic mangrove encroachment into salt marsh habitats between the early-1960s and 1980—when mangrove forest appears as the dominant vegetation cover at both study sites.

Further, our analyses of aerial photography sequences suggest that industrial geoengineering was a primary driver of tidal wetland conversion. The aerial imagery indicates that salt marsh remained the dominant habitat type at both study sites until the early 1960s, when these tidal wetlands were intensively ditched for mosquito control. Imagery from subsequent decades shows mangroves encroaching landward along ditch banks before expanding to replace marsh stands and create dense, homogenous tracts of mangrove forest. Review of regional temperature data and near-historical sea-level research from Tampa Bay Estuary suggests that the role of sea-level rise and winter temperature minima over the period of 20th century wetland conversion was likely comparatively minor.

Finally, the marsh-to-mangrove conversion documented in this study carries several pressing implications for the management of coastal wetlands, and particularly those in areas near the shifting latitudinal range limits of mangrove forest habitat. In Old Tampa Bay, wetland conversion resulted not only in the elimination of salt marsh/ salt prairie mosaics, but also in the loss of topographical and salinity gradients that supported the functions and services of these historical ecosystems. As sea-level continues to rise over the 21st century and the latitudinal range of mangroves continues to expand, scientists and land managers working along more temperate, marsh-dominated coasts must plan for the exceptionally rapid pace at which mangroves may replace high-intertidal habitat. This planning is especially important given low topographical gradients and the presence of shoreline alteration features, such as mosquito ditches, canals, and other small channels by which mangrove propagules may disperse.

Funding

This research was made possible through funding provided by the 2018 Florida SeaGrant Student Scholar Program, and by the University of South Florida Anthropology Department.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper

Acknoledgments

We express utmost gratitude to University of South Florida St. Petersburg Conservation Biology and University of South Florida

Anthropology for access to facilities and instrumentation. We thank Dr. Thomas J. Pluckhahn, Heather Draskovich, and Amy Spurling Gatenbee for assistance with strenuous and muddy fieldwork. We thank Pinellas County Parks and Conservation Resources and Upper Tampa Bay Park for granting permission for sample collection and working tirelessly to protect and manage conservation lands within Tampa Bay Estuary. We also thank Editor in Chief Dr. Anne Chin, and three anonymous reviewers for substantive comments that improved the quality of this paper.

Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.ancene.2021.100295.

References

[NASA] National Aeronautics and Space Administration, 2020. GISS Surface Air Temperature Analysis. . http://data.giss.nasa.gov.

[NOAA] National Oceanic and Atmospheric Administration, 2001. Sea Levels Online. . http://tidesandcurrents.noaa.gov/sltrends/sltrends.html.

[NOAA] National Oceanic and Atmospheric Administration, 2020. Daily Summaries Station Details: Station GHCND:USW00012842. . http://ncdc.noaa.gov/cdoweb/datasets/GHCND/stations/GHCND:USW00012842/detail.

[SWFWMD] Southwest Florida Water Management District, 2005. Land use/land Cover 1999. FLUCCS. http://www.swfwmd.state.fl.us/data/gis/layer_library/.

Albright, M., 1981. Mosquito Miseries Multiplying. The Evening Independent, St. Petersburg May 19, 1981.

Appleby, P.G., Oldfield, F., 1983. The Assessment of 210PB Data from Sites with Varying Sediment Accumulation Rates. Hydrobiologia 103, 29–35.

Armitage, A.R., 2015. The contribution of mangrove expansion to salt marsh loss on the Texas Gulf Coast. PLoS One 10 (5) e0125404.

Binford, M.W., 1990. Calculation and uncertainty analysis of 210PB dates for PIRLA project lake sediment cores. J. Paleolimnol. 3, 253–267.

Boyd, M.F., 1949. Malariology, Vol. I. II. W.B. Saunders Co., Philadelphia.

Breitfuss, M.J., Connolly, R.M., Dale, P.E.R., 2003. Mangrove distribution and mosquito control: transport of *Avicennia marina* propagules by Mosquito-Control Runnels in Southeast Queensland Saltmarshes. Estuar. Coast. Shelf Sci. 56, 573–579.

Breithaupt, J.L., Smoak, J.M., Smith III, T.J., Sanders, C.J., Hoare, A., 2012. Organic carbon burial rates in mangrove sediments: strengthening the global budget. Global Biogeochem. Cycles 26, GB3011. doi:http://dx.doi.org/10.1029/2012GB004375.

Brothwell, D., 1956. Voracious Skeeters Cost City Residents, Visitors. St. Petersburg Times April 28, 1956.

Brown, R.B., Stone, E.L., Carlisle, V.W., 1990. Soils. In: Myers, Ronald L., Ewel, John J. (Eds.), ECosystems of Florida. University of Central Florida Press, Orlando, Florida, pp. 35–69.

Chen, E., Gerber, J.F., 1990. Climate. In: Myers, R.L., Ewel, J.J. (Eds.), ECosystems of Florida. University of Central Florida Press, Orlando, Florida, pp. 11–34.

Choi, Y., Wang, Y., Hsieh, Y.P., Robinson, L., 2001. Vegetation succession and carbon sequestration in a coastal wetland in Northwest Florida: evidence from carbon isotopes. Global Biogeochem. Cycles 15 (2), 311–319.

Cicchetti, G., Greening, H., 2011. Estuarine biotope mosaics and habitat management goals: an application in Tampa Bay, FL, USA. Estuaries Coasts 34, 1278–1292.

Clement, R.C., 1965. Mosquito control and the ecosystem. Proc. NJ Mosq. Exterm. Soc. 52, 55.

Clewell, A.F., 1997. Vegetation. In: Coultas, C.L., Hsieh, Y.P. (Eds.), Ecology and Management of Tidal Marshes: A Model from the Gulf of Mexico. St. Lucie Press, Delray Beach, pp. 77–110.

Comeaux, R.S., Allison, M.A., Bianchi, T.S., 2012. Mangrove expansion in the Gulf of Mexico with climate change: implications for wetland health and resistance to rising sea levels. Estuar. Coast. Shelf Sci. 96, 81–95.

Corman, S.S., Roman, C.T., King, J.W., Appleby, P.G., 2012. Salt marsh mosquito-control ditches: sedimentation, landscape change, and restoration implications. J. Coast. Res. 28 (4), 874–880.

Crain, C.M., Albertson, L.K., Bertness, M.D., 2008. Secondary succession dynamics in estuarine marshes across landscape-scale salinity gradients. Ecology 89, 2889–2899.

Dale, P.E.R., Knight, J.M., 2006. Managing salt marshes for mosquito control: impacts of runnelling, open water marsh management and grid-ditching in sub-tropical australia. Wetland Ecol. Manag. 14, 211–220.

Davies, B.E., 1974. Loss on ignition as an estimate of soil organic matter. Soil Sci. Soc. Am. Proc. 38, 150–151.

Davis Jr., J.H., 1940. Publication 517. Tortugas laboratory paper. The Ecology and Geologic Role of Mangroves in Florida, vol. 32. Carnegie Institute, Washington, D.C, pp. 303–412.

Davis Jr., J.H., 1943. The natural features of South Florida, Especially the vegetation and the Everglades. Florida Geological Survey Bulletin 25, 1–333.

Davis, R.A., 2011. Beaches, barrier Islands, and inlets of the Florida Gulf Coast. In: Buster, N.A., Holmes, C.W. (Eds.), Gulf of Mexico, Origin, Waters, and Biota, Vol. 3. Texas A&M Press, College Station, pp. 89–98.

Dean, W.E., 1974. Determination of carbonate and organic matter in calcareous sediments and sedimentary rocks by loss on ignition: comparison with other methods. J. Sediment. Petrol. 44, 242–248.

Doughty, C.L., Langley, J.A., Walker, W.S., Feller, I.C., Schaub, R., Chapman, S.K., 2016. Mangrove range expansion rapidly increases coastal wetland carbon storage. Estuaries Coasts 39, 385–396.

Duke, N.C., Ball, M.C., Ellison, J.C., 1998. Factors influencing biodiversity and distribtutional gradients in mangroves. Glob. Ecol. Biogeogr. Let. 7, 27–47.

Engle, V.D., 2011. Estimating the provision of wetland services by Gulf of Mexico coastal wetlands. Wetlands 31, 179–193.

Fehring, W.K., 1986. NWRC Open-file report. Data Bases for Use in Fish and Wildlife Mitigation Planning in Tampa Bay, Florida: project Summary. U.S. Fish and Wildlife Service, Tampa 86-6.

Folk, R.L., 1965. Petrology of Sedimentary Rocks. Hemphills, Austin. .

Gerlach, M.J., Engelhart, S.E., Kemp, A.C., Moyer, R.P., Smoak, J.M., Bernhardt, C.E., Cahill, N., 2017. Reconstructing common era relative sea-level change on the gulf coast of Florida. Mar. Geol. 390, 254–269.

Haddad, K.D., Joyce, E.A.J., 1997. Management. In: Coultas, C.L., Hsieh, Y.P. (Eds.), Ecology and Management of Tidal Marshes. Florida: St Lucie Press, Delray Beach, pp. 309–330.

Harley, C.D.G., Hughes, A.R., Hultgren, K.M., Miner, B.G., Sorte, C.J.B., Thomber, C.S., Rodriguez, L.S., Tomanek, L., Williams, S.L., 2006. The impacts of climate change in coastal marine systems. Ecol. Lett. 9, 228–241.

He, Q., Silliman, B.R., 2019. Climate change, human impacts, and coastal ecosystems in the anthropocene. Curr. Biol. 29, 1021–1035.

Hobbs, R.J., Higgs, E.S., Hall, C.M. (Eds.), 2013. Novel Ecosystems: Intervening in the New Ecological World Order. Wiley-Blackwell, Chichester, West Sussex, UK.

Hogarth, P.J., 2005. The Biology of Mangroves. Oxford University Press, New York.

Holmes, C.W., 2001. Short-Lived Radioisotopes in Sediments: A Tool for Accessing Sedimentary Dynamics. USGS Open File Report 01-xxx. 6pp. .

Janicki, A., 2000. A Summary of Emergent Vegetation Habitat Coverage Data for Tampa Bay. Tampa Bay Estuary Program Technical Report. 08-00. .

Janicki, A.J., Wade, D.L., Robison, D.E., 1995. Habitat Protection and Restoration Targets for Tampa Bay. National Estuary Program, Technical Publication, St. Petersburg, pp. 07–93.

Janicki, A., Pribble, R., Zarbock, H., Janicki, S., Winowitch, M., 2001. Model-Based Estimates of Total Nitrogen Loading to Tampa Bay—Current Conditions and Updated 2010 Conditions. Tampa Bay Estuary Program, Technical Publication 08-01.

Jones, P.D., Moberg, A., 2003. Hemispheric and large-scale surface air temperature variations: an extensive revision and an update to 2001. J. Clim. 16, 206–223.

Jordan, C.L., 1984. Florida's weather and climate: implications for Water. In: Fernald, E.A., Patton, D.J. (Eds.), Water Resources Atlas of Florida. Institute of Scientific Public Affairs, Florida State University, Tallahassee, pp. 18–35.

Jowsey, P.C., 1966. An improved peat sampler. New Phytol. 65, 245–248.

Kangas, P.C., Lugo, A.C., 1990. The distribution of mangroves and Saltmarsh in Florida. Trop. Ecol. 31, 32–39.

Kitto, M.E., 1991. Determination of photon self-absorption corrections for soil samples. Int. J. Appl. Instrum. 42, 835.

Knapp, A.K., Briggs, J.M., Collins, S.L., et al., 2008. Shrub encroachment in North American Grasslands: Shifts in growth form dominance rapidly alters control of ecosystem carbon inputs. Glob. Change Biol. 14, 615–623.

Koenig, C.C., Coleman, F.C., Eklund, A.M., Schull, J., Ueland, J., 2007. Mangroves as essential nursery habitat for goliath grouper (*Epinephelus itajara*). Bull. Mar. Sci. 80 (3), 567–586.

Krauss, K.W., From, A.S., Doyle, T.W., Doyle, T.J., Barry, M.J., 2011. Sea-level rise and landscape change influence mangrove encroachment onto marsh in the ten thousand islands region of Florida (USA). J. Coast. Conserv. 15, 629–638.

LeMap, L.E., 2007. The Impact of Drainage Ditches on Salt Marsh Flow Patterns, Sedimentation and Morphology: Rowley River, Massachusetts. MS Thesis. School of Marine Science, College of William and Mary, Virginia.

Leonard, R.I., Judd, F.W., Stalter, R., 2012. The biological flora of coastal dunes and wetlands: *salicornia biglovii* torrey. J. Coast. Res. 28 (3), 719–725.

Lewis III, R.R., Estevez, E.D., 1988. The Ecology of Tampa Bay. An estuarine profile. U. S. Fish and Wildlife Service. U.S. Dept. of the Interior, Florida BR-85:(7.18).

Lockwood, J.A., Lockwood, D.R., 1993. Catastrophe theory: a unified paradigm for rangeland ecosystem dynamics. J. Range Manag. 46 (4), 282–288.

Lopez-Medellin, X., Ezcurra, E., Gonzalez-Abraham, C., Hak, J., Santiago, L.S., Sickman, J.O., 2011. Oceanographic anomolies and sea-level rise drive mangroves inland in the Pacific coast of Mexico. J. Veg. Sci. 22, 143–151.

Lugo, A.E., Medina, E., McGinley, K., 2014. Issues and challenges of mangrove conservation in the anthropocene. Madera Y Bosques 20, 11–38.

McKee, K., Cahoon, D.R., Feller, I.C., 2007. Caribbean mangroves adjust to rising sea level through biotic controls on change in soil elevation. Glob. Ecol. Biogeogr. 16, 545–556.

McKee, K., Rogers, K., Saintilan, N., 2012. Response of Salt Marsh and mangrove wetlands to changes in atmospheric CO2, climate, and Sea level. In: Middleton, B.A. (Ed.), Global Change and the Function and Distribution of Wetlands. Springer, Dordrecht, pp. 63–96. doi:http://dx.doi.org/10.1007/978-94-007-4494-3_2.

Mitchell, P., 1956. Dynamite Gouges Mosquito Ditches. St. Petersburg Times August 19th 1956.

Montague, C.L., Wiegert, R.G., 1990. Salt marshes. In: Myers, R.L., Ewel, J.J. (Eds.), Ecosystems of Florida. University of Central Florida Press, Orlando, pp. 481–516.

Mormino, G.R., 2005. Land of Sunshine, State of Dreams: A Social History of Modern Florida. University Press of Florida, Gainesville.

Morrison, G., Yates, K., 2011. Chapter 3: origin and evolution of tampa Bay. In: Yates, Kimberly, Greening, Holly, Morrison, Gerold (Eds.), Integrating Science and Resource Management in Tampa Bay. U.S. Geological Survey, Circular, Florida, pp. 1348.

Odum, W.E., Heald, E.J., 1972. Trophic analyses of an estuarine mangrove community. Bull. Mar. Sci. 22, 671–738.

Odum, W.E., McIvor, C.C., 1990. Chapter 15: mangroves. In: Myers, R.L., Ewel, J.J. (Eds.), Ecosystems of Florida. University of Central Florida Press, Orlando, pp. 517–548.

Odum, W.E., McIvor, C.C., Smith, T.J.I.I.I., 1982. The Ecology of the Mangroves of South Florida: A Community Profile. U.S. Fish and Wildlife Service. Office of Biological Services, Washington, D.C FWS/OBS-81/24.

Olsson, I.U., 1986. Radiometric dating. In: Berglund, B.E. (Ed.), Handbook of Holocene Palaeoecology and Palaeohydrology. Wiley and Sons, Chichester, pp. 273–312.

Osland, M.J., Spivak, A.C., Nestlerode, J.A., Lessmann, J.M., Almario, A.E., Heitmuller, P.T., Russell, M.J., Krauss, K.W., Alvarez, F., Dantin, D.D., Harvey, J.E., From, A.S., Cormier, N., Stagg, C.L., 2012. Ecosystem development after mangrove wetland creation: plant-soil change across a 20-Year chronosequence. Ecosystems 15, 848–866.

Osland, M.J., Enwright, N., Day, R.H., Doyle, T.W., 2013. Winter climate change and coastal wetland foundation species: salt marshes vs. Mangrove forests in the Southeastern United States. Glob. Chang. Biol. 19, 1482–1494.

Patterson, G., 2004. The Mosquito Wars: A History of Mosquito Control in Florida. University Press of Florida, Gainesville.

Patterson, S., McKee, K.L., Mendelssohn, I.A., 1997. Effects of tidal inundation and predation on Avicennia germinans seedling establishment and survival in a sub-tropical mangal/salt marsh community. Mangroves Salt Marsh. 1, 103–111.

Patterson, C.S., Mendelssohn, I.A., 1991. A comparison of physicochemical variables across plant zones in a mangal/salt marsh community in Louisiana. Wetlands 11 (1), 139–161.

Pennings, S.C., Grant, B., Bertness, M.D., 2005. Plant zonation in low-latitude salt marshes: disentangling the roles of flooding, salinity and competition. Ecology 93, 159–167.

Perry, C.L., Mendelssohn, I.A., 2009. Ecosystem effects of expanding populations of *avicenna germinans* in a Louisiana salt marsh. Wetlands 29 (1), 396–406.

Philen, E.A., Carmichael, G.T., 1956. The management of water for mosquito control in the coastal marshes of Florida. Misquito News 16 (2) .

Pool, D.J., Snedaker, S.C., Lugo, A.E., 1977. Structure of mangrove forests in Florida, puerto Rico, Mexico, and Costa rica. Biotropica 9 (3), 195–212.

Poppe, L.J., Eliason, A.H., Fredericks, J.J., Rendings, R.R., Blackwood, D., Polloni, C.F., 2000. Grain Size Analysis of Marine Sediments: Methodology and Data Processing. USGS Open File Report 00-358 URL:. . http://pubs.usgs.gov/of/2000/of00-358/text/chapter1.htm.

Pratt, H.D., Barnes, R.C., Littig, K.S., 1963. Mosquitos of Public Health Importance and Their Control. Insect Control Series, Public Health Service Publication, Washington D.C No. 772.

Raabe, E.A., Roy, L.C., McIvor, C.C., 2012. Tampa Bay Coastal Wetlands: nineteenth to twentieth century tidal marsh-to-Mangrove conversion. Estuaries Coasts 35, 1145–1162.

Radabaugh, K.R., Moyer, R.P., Chappel, A.R., Powell, C.E., Bociu, I., Clark, B.C., Smoak, J. M., 2018. Coastal blue carbon assessment of mangroves, salt marshes, and salt barrens in Tampa Bay, Florida, usa. Estuaries Coasts 41, 1496–1510.

Reimer, P.J., Bard, E., Bayliss, A., Beck, J.W., Blackwell, P.G., Bronk-Ramsey, C., Grootes, P.M., Guilderson, T.P., Haflidason, H., Hajdas, I., Hatté, C., Heaton, T. J., Hoffmann, D.L., Hogg, A.G., Hughen, K.A., Kaiser, K., Felix, K., Bernd, M., Sturt, W., Niu, M., Reimer, R.W., Richards, D.A., Scott, E.M., Southon, J.R., Staff, R.A., Turney, C.S.M., van der Plicht, J., 2013. IntCal13 and Marine13 radiocarbon age calibration curves 0–50,000 years cal BP. Radiocarbon 55 (4), 1869–1887.

Rogers, K., Wilton, K., Saintilan, N., 2006. Vegetation change and surface elevation dynamics in estuarine wetlands of southeast Australia. Estuar. Coast. Shelf Sci. 66, 559–569.

Ross, M.S., Meeder, J.F., Sah, J.P., Ruiz, P.L., Telesnicki, G.L., 2000. The southeast saline Everglades revisited: 50 years of coastal vegetation change. J. Veg. Sci. 11, 101–112.

Saintilan, N., Wilson, N.C., Rogers, K., Rajkaran, A., Krauss, K.W., 2014. Mangrove expansion and salt marsh decline at mangrove poleward limits. Glob. Chang. Biol. 20, 147–157.

Saintilan, N., Wilton, K., 2001. Changes in the distribution of mangroves and saltmarshes in Jervis Bay, Australia. Wetl. Ecol. Manag. 9, 409–420.

Santamaria, M., Lazar, A., Kavanaugh, C., Adams, B., Dennis, J., Salinas, E., Voges, L., 2010. Florida Assessment of Coastal Trends. Florida Coastal management Program, Florida Department of Environmental Protection, Tallahassee.

Simpson, R.H., Riehl, H., 1981. The Hurricane and Its Impact. Louisiana State University Press, Baton Rouge.

Skilbeck, G.C., Trevathan-Tackett, S., Apichanangkool, P., Macreadie, P.I., 2017. Sediment sampling in estuaries: site selection and sampling techniques. In: Weckstrom, K., Saunders, K.M., Gell, P.A., Skilbeck, C.G. (Eds.), Applications of Paleoenvironmental Techniques in Estuarine Studies. Springer-Nature, Dordrecht, The Netherlands, pp. 89–120.

Smith III, T.J., Foster, A.M., Tilling-Range, G., Jones, J.W., 2013. Dynamics of mangrove-marsh ecotones in subtropical coastal wetlands: Fire, sea-level rise, and water levels. Fire Ecol. 9, 66–77.

Smith III, T.J., Tilling, G., Leasure, P.S., 2007. Restoring coastal wetlands that were ditched for mosquito control: a preliminary assessment of hydro-leveling as a restoration technique. J. Coast. Conserv. 11, 67–74.

St. Petersburg Independent, 1956. County approves move: $38,400 emergncy sketer funds drawn. XLIX 203 (June 27) 1956.

Steinmuller, H.E., Foster, T.E., Boudreau, P., Hinkle, C.R., Chambers, L.G., 2020. Tipping points in the mangrove March: characterization of biogeochemical cycling along the mangrove-salt marsh ecotone. Ecosystems 23, 417–434.

Stevens, P.W., Fox, S.L., Montague, C.L., 2006. The interplay between mangroves and Saltmarshes at the transition between temperate and subtropical climate in Florida. Wetl. Ecol. Manag. 14, 435–444.

Stout, J.P., 1984. The Ecology of Irregularly Flooded Salt Marshes of the Northeastern Gulf of Mexico: A Community Profile. Biological Report 85 (7.1). National Coastal Ecosystems Team, Fish and Wildlife Service, Washington, D.C.

Thompson, P., 2010. Climate Change in Our Own Backyard: Rising Seas Make Tampa Bay Coastline More Vulnerable. St. Petersburg Times.

Van Auken, O.W., 2000. Shrub invasions of North American semiarid grasslands. Annu. Rev. Ecol. Syst. 31, 197–215.

Wang, Q., Li, Y., Wang, Y., 2011. Optimizing the weight loss-on-Ignition methodology to quantify organic and carbonate carbon of sediments from diverse sources. Environ. Monit. Assess. 174, 241–257.

Whitfield, A.K., 2017. The role of seagrass meadows, mangrove forests, salt marsh and reed beds as nursery areas and food sources for fishes in estuaries. Rev Fish Biol Fisheries 27, 75–110.

Williamson, G.J., Boggs, G.S., Bowman, D.M.J.S., 2011. Late 20th century mangrove encroachment in the coastal australian monsoon tropics parallels the regional increase in woody biomass. Reg. Environ. Change 11, 19–27.

Woodroffe, C.D., Thom, B.G., Chappell, J., 1985. Development of widespread mangrove swamps in mid-Holocene times in northern Australia. Nature 317, 711–713.

Yates, K., Greening, H., 2011. An introduction to tampa Bay. In: Kimberly, K. (Ed.), Integrating Science and Resource Management in Tampa Bay, Florida. U.S. Geological Survey, Reston, pp. 1–16 Yates, Holly Greening, and Gerold Morrison, Circular 1348.

Zacks, M.H., 2013. From Table to Trash: The Rise and Fall of Mullet Fishing in Southwest Florida. Dissertation Manuscript, American Studies, University of Hawai'I, Manoa.

Zervas, C.E., 2001. Sea level variations Of the United States, 1854–1999. Silver Spring: U.S. Dept. Of commerce. NOAA Technical Report NOS CO-OPS. National Oceanic and Atmospheric Administration, pp. 36.