



# Coastal Wetland Geomorphic and Vegetative Change: Effects of Sea-Level Rise and Water Management on Brackish Marshes

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## Abstract

Conservation and management of coastal ecosystems require an understanding of how accelerated sea-level rise (SLR) and altered hydrology impact community shifts over time. This study evaluates the response of tidal wetlands of the Ten Thousand Islands, Collier County, Florida, to SLR and water management, with a focus on the development and distribution of tidal ponds across a wetland landscape. Sediment cores collected from marshes, mangroves, and tidal ponds reveal a clear transgressive stratigraphy. Facies analyses demonstrate that ponds originate from the surface downward through the degradation of marsh peat. Analyses of 1953 and 2009 aerial imagery using ArcGIS® software clearly identified tidal pond initiation, growth, and merger over time. Wetlands west of the Faka Union Canal, which have limited freshwater sheet flow due to canalization, are experiencing a greater increase in pond count, pond density ( $p = 0.0038$ ), and mean pond area ( $p < 0.0001$ ). Qualitative observations also recognize a relatively larger influence of mangrove envelopment over time in western sites compared with those retaining near-natural flows. Future land management plans must account for the expected submergence of inland marsh ecosystems driven by SLR and accelerated by hydrologic alteration. Continued restoration of freshwater sheet flow is necessary for slowing the regional transition (and loss) of graminoid marshes to either mangrove or pond environments. Without such action, a complete loss of these biologically diverse marsh ecosystems as mangrove forests encroach and marsh surfaces submerge is probable in the short term.

**Keywords** Sea-level rise · Water management · Marsh · Mangrove · Tidal pond

## Introduction

Sea-level rise (SLR) is recognized as a current and long-term threat to the vegetation and geomorphology of coastal wetlands worldwide (Parkinson et al. 1994; IPCC 2014). In the Americas, SLR has been essentially continuous but expressed at varying rates throughout the Holocene (Parkinson et al. 1994; Toscano and Macintyre 2003). Satellite altimetry data estimate the present global SLR rate at 3.4 mm/y ( $\pm 0.4$  mm/y) (Nerem et al. 2010), an assessment supported by regional (Maul and

Martin 1993; Wanless et al. 1994; Savarese and Hoeflein 2012) and global studies (Church and White 2011; IPCC 2014). This rate vastly exceeds that experienced over the last 5000 years and is rapid enough to alter the current distribution and composition of wetland ecosystems (Gleason and Stone 1994; Wanless et al. 1994; Savarese and Hoeflein 2012). Conservation and management of the Ten Thousand Islands (TTI) complex of Southwest Florida, where this study was conducted (Fig. 1), will thus require a detailed understanding of how local coastal communities will respond to accelerated SLR. How regional altered hydrology could modify, and potentially exacerbate, identified changes should also be explored.

Accelerated SLR influences the arrangement of coastal communities in several ways. Transgression may result in a net loss of estuarine habitat and inland graminoid marshes, especially where species are limited in geographic expansion (Barry 2009; Kearney and Rogers 2010; Kearney and Riter 2011; Barry et al. 2013; IPCC 2014; Sandi et al. 2018). Such coastal marsh loss is prevalent worldwide as marsh landscapes are either replaced by mangroves migrating inland (Barry 2009; Gaiser et al. 2006; Krauss et al. 2011; Sandi et al.

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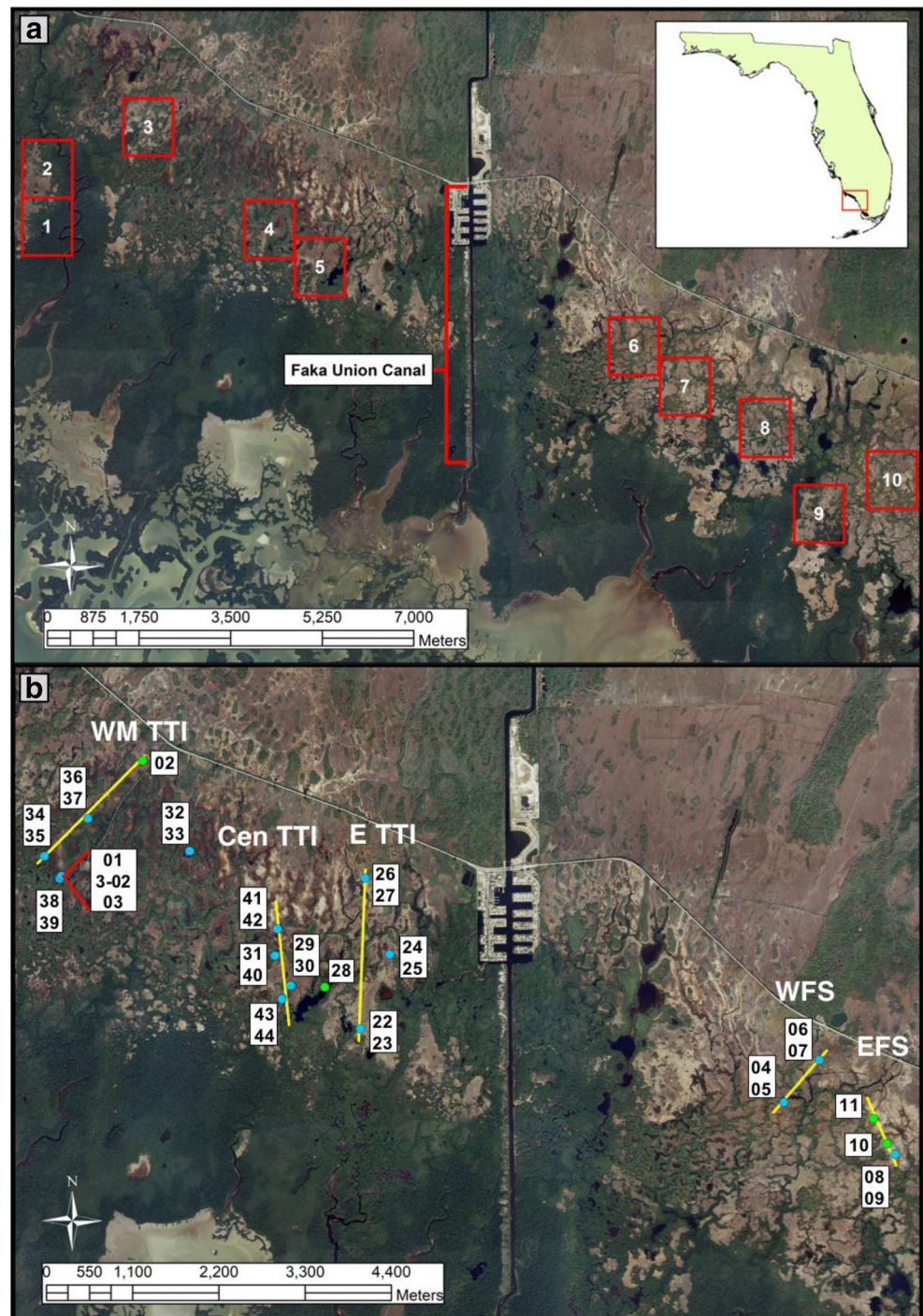
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**Fig. 1** **a** Aerial image of the Ten Thousand Islands complex, highlighting the placement of the ten 1.0 km<sup>2</sup> study sites (numbered 1 through 10) for tidal pond analysis. Location of the Faka Union Canal highlighted in red. **b** Location of the five study transects: a westernmost (WM TTI, .84 km), central (Cen TTI, .90 km), and eastern (E TTI, 1.88 km) transect west of the Faka Union Canal, and a western (WFS, .75 km) and eastern (EFS, .54 km) transect east of the canal. Cores are numbered by last two digits of the total core number (e.g., 0902-2 is listed as 02). Blue circles represent a marsh-pond pair of cores, and green circles represent individual cores. Cores 32 and 33 were extracted along an incomplete transect due to impenetrable terrain to the south. Images acquired from the Collier County Property Appraiser, 2009



2018) or go subtidal (Stevenson et al. 1985; DeLaune et al. 1994; Wilson et al. 2010; Kirwan and Megonigal 2013). Slower to migrate and accrete, marsh species' physiological responses to rising sea waters trail those of mangroves (Sandi et al. 2018). Regionally, extensive brackish and salt marsh conversion to mangrove shrub or forest have been recognized within the TTI (Barry 2009; Krauss et al. 2011) and adjacent Rookery Bay (Barry et al. 2013) within the last century.

Allochthonous sediment inputs via freshwater sheet flow, adjacent tidal channels, or seaward from the tides may be critical in combating wetland losses in communities disproportionately reliant upon in situ peat deposition (Kearney et al. 1988; DeLaune et al. 1994; Kearney and Riter 2011; Sandi et al. 2018). Compaction and subsidence (Cahoon and Lynch 1997; Rogers et al. 2006), plant physiological responses and productivity (Kearney and Riter 2011; Kirwan and Megonigal

2013; Howard et al. 2016), and regional geomorphology (Adamowicz and Roman 2005; Kearney and Rogers 2010) also play a role. In the absence of human intervention, ecogeomorphic feedbacks between site characteristics (flooding depth and duration, nutrient inputs, sediment delivery) and biota (accretion, biomass production) allow coastal wetlands to keep pace with high relative rates of SLR (Kirwan and Megonigal 2013; Rodriguez et al. 2017). Focused studies accounting for the interaction of local factors with regional vegetative environments are necessary to truly understand how a location and its flora will fare against existing and predicted SLR.

Anthropogenic control over hydrology and sediment delivery can disrupt these feedbacks and impede wetland vegetation's capacity to persist (Sklar and Browder 1998; Kirwan and Megonigal 2013; Michot et al. 2015; Rodriguez et al. 2017; Sandi et al. 2018). Ditches, canals, levees, and other structures modify hydroperiod, potentially shifting flooding depth and duration above (or below) wetland species' physiological thresholds (Rodriguez et al. 2017). Increased channelization can diminish sediment transport to marshes and other wetlands (Kirwan and Megonigal 2013). Control structures limiting sheet flow or reducing groundwater supply further increase subsidence (Turner 2004; Rogers et al. 2006), autocompaction (Kearney and Riter 2011), and impact local salinities (Popowski et al. 2004; Soderqvist and Patino 2010; Michot et al. 2015). The role in sediment transport in countering vegetation loss cannot be understated (Stevenson et al. 1985; Rodriguez et al. 2017).

Highly modified hydrologic flow via the Faka Union Canal (see below) is thus likely exacerbating the effects of SLR on the vegetation within the TTI. Mangrove encroachment into inland freshwater wetlands within the Ten Thousand Islands National Wildlife Refuge (TTINWR) and other parts of South Florida has increased disproportionately since the 1960s, correlating with wide-scale hydrologic alterations combined with accelerated SLR (Gaiser et al. 2006; Krauss et al. 2011; Michot et al. 2015). The northward shift of mangroves into previous marsh environments was conspicuously more severe within the marsh/mangrove ecotone of the TTINWR than within a less hydrologically impacted site of Everglades National Park, indicating hydrological changes might have an even greater impact than SLR on vegetative change (Barry 2009). If slowing mangrove encroachment to maintain ecologically valuable marsh habitat is a management goal for the TTI, then understanding how the changing dynamics of these landscapes are influenced by water management in light of SLR are critical (Michot et al. 2015).

The development of subtidal ponds—herein termed “pocks” for their appearance—has also been identified within marshes faced with accelerated SLR. Pocks have been recognized in brackish and salt marshes of Chesapeake Bay (Stevenson et al. 1985; Kearney et al. 1988; Kearney and Rogers 2010),

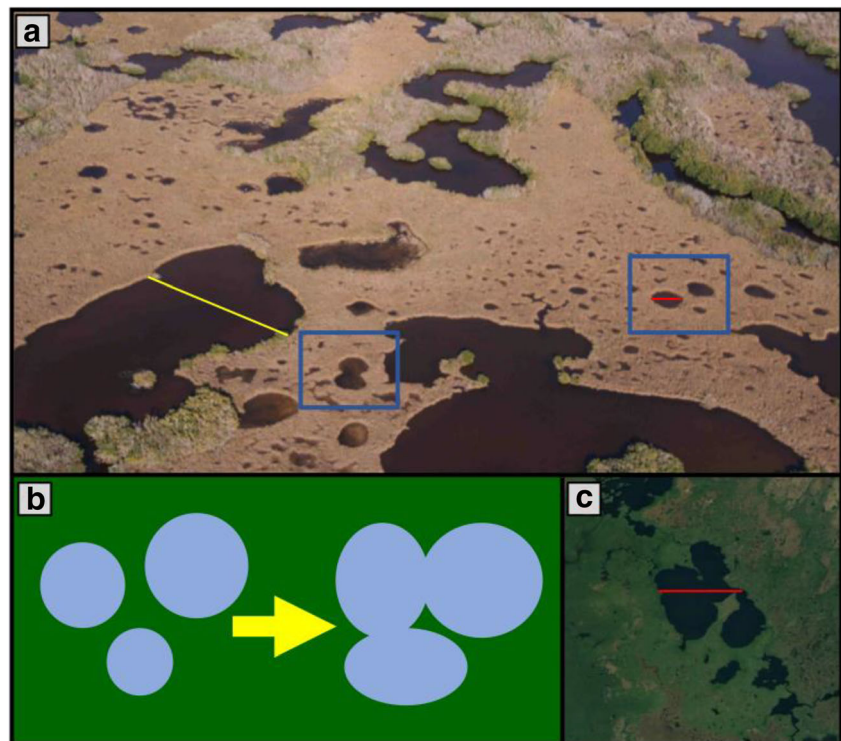
coastal Maine (Adamowicz and Roman 2005; Wilson et al. 2007; Wilson et al. 2010), Delaware Bay (Kearney and Riter 2011), Louisiana (Adamowicz and Roman 2005; DeLaune et al. 1994), South Florida (Hoye 2009), and southeast Australia (Rodriguez et al. 2017; Sandi et al. 2018). Recent decades have witnessed alarming rates of conversion of some Atlantic and Gulf Coast marshes to open water as ponds develop and enlarge over time (Stevenson et al. 1985; Kearney et al. 1988; DeLaune et al. 1994; Wilson et al. 2010).

It is believed that accelerated SLR may compound the initiation, growth, and merger of pocks at the expense of graminoid marsh environments (Kearney et al. 1988; DeLaune et al. 1994; Hoye 2009; Fig. 2). Research implicates disturbance as a primary driving force for pond inception. Wind and waves from storms (Harshberger 1916; Kearney et al. 1988; Kearney and Riter 2011), herbivory (Stevenson et al. 1985), and anthropogenic alterations to hydrology and sediment supply (Kearney et al. 1988; Sklar and Browder 1998; Kirwan and Megonigal 2013; Rodriguez et al. 2017; Sandi et al. 2018) can all result in pronounced changes in stability, biomass, and accretion (DeLaune et al. 1994; Kearney and Riter 2011). Shrinkage of root pore space with plant death promotes elevation loss (DeLaune et al. 1994). Over time, these physiological stressors to site disturbance can lead to marsh degradation and subsequent pond formation (Stevenson et al. 1985; DeLaune et al. 1994; Adamowicz and Roman 2005; Kearney and Rogers 2010).

Pocks influence sediment transport, soil chemistry, and hydrodynamics, often effectively inhibiting future reestablishment of vegetation and providing a conduit for further pocking (Stevenson et al. 1985; Wilson et al. 2007; Wilson et al. 2010; Sandi et al. 2018). Pocks in Chesapeake Bay and Maine were revealed to accumulate a high proportion of sediment available to the marsh surface (Stevenson et al. 1985; Wilson et al. 2010), indicating a deficit in sediment delivery to adjacent marshes. Sediment entrapment by pocks exacerbates marsh elevation loss in coastal regions with small tidal range and limited allochthonous sediment supply (Kearney et al. 1988; Kearney and Rogers 2010; Kirwan and Megonigal 2013).

Erosion of the pock margin provides a means for the enlargement and merger of ponds (Stevenson et al. 1985; Wilson et al. 2010). The formation of large bays, as seen in the northwestern coastal Everglades, is believed to result from the continued development of pocks over time (Hoye 2009; Fig. 2c); comprehensive quantitative data are limited. The distribution and mechanism of pock formation have not yet been described for subtropical South Florida coastal wetlands. Whether inception, growth, and merger follow similar patterns as recognized elsewhere, and whether these patterns will be apparent in the varied marsh and mangrove communities prevalent along the coast, is unknown. Understanding the role of water management on pock dynamics and wetland loss within a hydrologically altered wetland is also critical. Australian

**Fig. 2** **a** Examples of pocks within the marsh landscape of the TTI. The blue boxes highlight individual pocks and outline pond merger in progress. The diameter of the individual tidal pond (blue inset box, right, red line) is 12 m, and the diameter of the larger pond (yellow line) is 66 m; these are included for scale. **b** This schematic highlights how pond growth and ultimate merger could occur over time. **c** Large inland bays are currently evident in the coastal northwestern Everglades, to the south of the TTI (as per Hoye 2009). The diameter of Alligator Bay/Dads Bay (shown) is 2.5 km. A similar phenomenon is apparent within the inland marshes of the TTI (a), albeit on a strikingly different scale. Imagery in a was captured via helicopter survey, March 1, 2010



estuarine marshes showed faster rates of submersion in flood attenuated wetlands (Rodriguez et al. 2017; Sandi et al. 2018), but pock dynamics have yet to be quantified in this context. Exploring pocking across Florida's coastal landscape may better prepare land managers of the TTI, and managed coastal wetlands worldwide, in protecting these landscapes into the future.

This study provides an opportunity to identify evidence of a transgressive ecologic and geomorphic signature landward of the coastal fringe along Southwest Florida's shoreline. This work aims to expand current understanding of pock development within varied marsh systems under stress from SLR and human influence. The role of anthropogenic water management in exacerbating sea level's effect can be isolated by comparing wetland sites east (retaining near-natural water flows) and west (in highly water-managed wetlands) of the Faka Union Canal (see below). Though geographically focused, the results of this research have implications for the prolific and seemingly non-intuitive influences of SLR on a much broader scale. Additionally, the work demonstrates the value of freshwater flow restoration, a costly and often controversial investment, to mitigating the influence of SLR on natural systems.

Four hypotheses were tested as follows: (H1) Vegetative, and thus wetland community shifts, follow a predictable pattern over time in response to accelerated SLR (e.g., Parkinson et al. 1994; Wanless et al. 1994; Gaiser et al. 2006); (H2) specific precursor (here, *Spartina bakeri*-dominated) marshes are more likely than others to transform into tidal ponds; (H3)

a pattern of tidal pond initiation, growth, and ultimate merger into inner coastal bays in response to SLR can be identified (e.g., Kearney et al. 1988; DeLaune et al. 1994; Wilson et al. 2007; Kearney and Rogers 2010; Wilson et al. 2010) in brackish marsh environments; and (H4) the effects of wetland community change and tidal pond expansion due to SLR are more pronounced west of the Faka Union Canal; the nearly 40 years of altered sheet flow between canal construction and study implementation provides a significant time frame to view the differential effects of increased marine incursion between these sites.

## Methods

### Study Site

The TTI complex, comprising 110,000 contiguous shoreline acres, is composed of mangroves, estuaries, fresh and brackish water marshes, mudflats, and upland habitats in the Big Cypress Basin in Collier County, Southwest Florida (McPherson and Halley 1996; Shirley and Brandt-Williams 2001). The TTI ecosystems comprise two dominant geomorphological formations: the outer bays and oyster reef-founded islands for which the region is named, and a span of fresh and brackish water marshes and mangrove communities inland of the coastal fringe (Fig. 1a). Extending approximately 20 miles from County Road 92 in Naples eastward to Everglades City and bordered by U.S. 41 three miles to the north, portions of

the TTI are managed as protected lands by various entities, including the Cape Romano-Ten Thousand Islands (CR-TTI) Aquatic Preserve, Ten Thousand Islands National Wildlife Refuge (TTINWR), Fakahatchee Strand Preserve State Park (FSP), Everglades National Park (ENP), and Rookery Bay National Estuarine Research Reserve (RBNERR) (Gardner 1988; U.S. Army Corps 2004).

The TTI wetlands receive freshwater primarily via sheet flow from inland wetlands and saltwater tidally from the Gulf of Mexico (U.S. Army Corps 2004). Salinity within these inland coastal wetlands varies between the wet (June–November) and dry season (December–May) (Shirley and Brandt-Williams 2001; Popowski et al. 2004), and also differs across the regional landscape due to the gently sloping topography (Gleason and Stone 1994; Shrestha 2010; Michot et al. 2015). Elevation within the TTI shallows from roughly 1.5 m (NGVD29) south of U.S. 41 to 0 m at the water's edge, which constitutes approximately 5–7 km of landscape distance (Shrestha 2010). The regional climate is marked by strong seasonal variation in precipitation; rainfall averages 50–55 in. (127–140 cm) per year (Shirley and Brandt-Williams 2001) but is primarily concentrated in the wet season months (Gardner 1988).

Historically, freshwater sheet flow from Lake Okeechobee moved slowly across Southwest Florida and eventually reached the wetlands of the TTI. The timing, frequency, and abundance of freshwater flows have been drastically altered since pre-settlement times due to extensive region-wide water management (McPherson and Halley 1996; Shirley and Brandt-Williams 2001; U.S. Army Corps 2004). Much of the Picayune Strand to the north was drained during the late 1960s–mid-1970s in preparation for the Southern Golden Gate Estates (SGGE) community (Shirley and Brandt-Williams 2001; U.S. Army Corps 2004). This project has channelized freshwater flow from the Camp Keais watershed through a series of canals, and ultimately through the Faka Union Canal (Fig. 1a). This effectively concentrates flow to Faka Union Bay and significantly reduces sheet flow to surrounding estuaries (U.S. Army Corps 2004).

The Camp Keais watershed of the Picayune Strand historically encompassed a roughly 234-mile<sup>2</sup> area, including the entire SGGE east to just beyond the terminal Faka Union Canal and south to the estuaries of the TTI. The gentle slope of this watershed slowly moves freshwater sheet flow in a southwest direction toward the coast, and once provided sheet flow and groundwater recharge to the TTI wetlands now west of the canal (U.S. Army Corps 2004; Booth et al. 2014). Wetlands now east of the canal primarily receive sheet flow from the near-pristine Fakahatchee Strand watershed (U.S. Army Corps 2004; Booth et al. 2014). Construction of the four feeder canals within the SGGE and the terminal Faka Union Canal has effectively reduced sheet flow across the western watershed and simultaneously increased point

discharge directly into Faka Union Bay (U.S. Army Corps 2004; Soderqvist and Patino 2010). As culverts constructed in 2006 as part of the Tamiami Trail Culverts Project and additional bridges under U.S. 41 are generally uniformly placed across the region, diversion of freshwater flow to western wetlands via the canal system is considered the principal cause of this difference (Soderqvist and Patino 2010; Abteu and Cuica 2011).

Altered hydrology has had a notably larger impact on wetlands to the west since canal construction (U.S. Army Corps 2004; Booth et al. 2014). In fact, mean monthly flow out of the Faka Union Canal was nearly 10 times higher than the rate flowing from adjacent Little Wood River to the west during the 2008 wet season (Soderqvist and Patino 2010). The greater flow and rate of freshwater output into Faka Union Bay have resulted in increased local salinities to surrounding bays and estuaries (Popowski et al. 2004; U.S. Army Corps 2004; Soderqvist and Patino 2010). Altered salinities have affected both habitat quality and vegetative distribution, thus impacting biota, within the estuaries and bays in the TTI (Alexander and Crook 1974; Popowski et al. 2004; Krauss et al. 2011). Additionally, the canal system draws down the water table and reduces groundwater delivery to TTI wetlands, exacerbating drought conditions across the landscape (U.S. Army Corps 2004). Combined, these circumstances have resulted in shifts in vegetative communities, exposure and oxidation of once-submerged wetland soils, and increased marine incursion for downstream environments (U.S. Army Corps 2004; Barry 2009; Krauss et al. 2011).

## Experimental Design

The study was approached using two methodologies. First, a stratigraphic analysis of wetland and pock history was undertaken to document a pattern of environmental change in response to SLR, both prior to and during anthropogenic influence. Stratigraphy also provides insight into the mechanism of vegetative change and pock formation. Second, geographic information system (GIS) software was used to conduct a geospatial analysis of the landscape east and west of the Faka Union Canal. This assessment quantified the development and distribution of pocks between 1953 and 2009 aerial imagery.

## Stratigraphic Methodologies

Five north-to-south transects were laid out within the study region (Fig. 1b). The western (WFS) and eastern (EFS) Fakahatchee Strand transects were located east of the Faka Union Canal and thus represent the retention of near-natural water flows to coastal communities. The westernmost (WM TTI), central (Cen TTI), and eastern (E TTI) TTI transects were to the west of the canal in a region highly water-managed since canal construction (U.S. Army Corps 2004).

Each transect was oriented parallel to the tidal gradient. Transect length ranged from .54 km for the shortest (EFS) to 1.89 km for the longest (E TTI) transects; in some locations impassable terrain made further sampling difficult. Nonetheless, all transects were extended for a great enough distance to traverse multiple habitats and ecotones. Gathering cores along transects in this manner allowed a variety of wetland environments to be sampled, and, combined, provided ample data to elucidate any stratigraphic variation among sites.

Three-inch diameter aluminum irrigation pipe and standard hand coring techniques were employed in sediment core collection. A total of 35 cores were extracted for analysis. Twenty-eight of these sediment cores were collected in 14 pairs along or near the five transects, with one core of each pair taken within the dominant wetland vegetation, and the other within an adjacent tidal pond. Except where logistically difficult to do so, cores paired in this manner were extracted in close geographical proximity (< 15 m) to each other. Communities dominated by *Eleocharis cellulosa*, *Spartina bakeri*, *Distichlis spicata*, and *Rhizophora mangle* were sampled multiple times in this manner.

Cores ranged from 66 to 249 cm in length. Bedrock was never as shallow as 66 cm; however, as analyses were focused on environmental transition since the onset of wetland characteristics on the landscape, hand-coring to bedrock was deemed both impractical and unnecessary. After extraction, each core was cut into 1-m sections and opened longitudinally using a 7 1/4" blade circular saw. Both halves of each 1-m section were labeled and photographed. The history of wetland transition and tidal pond formation was interpreted through the description and analysis of facies contained within the 35 sampled cores. Facies were described based on sedimentologic characteristics and distinguished by Munsell color, texture, and compositional variation, and facies contacts were noted as being sharp or gradational. A minimum of ten samples were collected from each identified facies for grain size analysis. Grain size and sorting characteristics were determined with a Malvern Mastersizer 2000 laser particle size analyzer.

Collectively, these data were utilized to construct detailed stratigraphic columns for each core. The decompacted lengths of each facies were calculated using a superjacent decompaction model, lithostratigraphic correlations were made, and fence diagrams were produced. The decompaction model employs the assumption that organic- and mud-rich sediments are comparatively more susceptible to compaction than quartz sands during core extraction. Thus, the model computes decompacted heights for all facies contacts resting above basal quartz sands. The heights of the upper contact of quartz facies and all contacts below remain unchanged, while contacts above the quartz sands are decompacted upward. Quartz sand facies, and anything resting below them, thus

remain the same relative thicknesses (while facies above are effectively thickened), but their position relative to the surface is shifted downward as facies above are decompacted.

Five organic-rich samples composed of roots, plant fibers, or organic-rich muds were obtained from three cores (0911-30, Cen TTI; 1003-01, near WM TTI; 1005-05, WFS) and submitted to Beta Analytic in Miami, FL for radiocarbon dating via accelerator mass spectrometry (AMS). Of these, cores 0911-30 and 1005-05 were extracted from modern *Spartina bakeri* marshes and 1003-01 was sampled from a *Rhizophora mangle*-encircled tidal pond. Each sample was collected at the base, or initial onset, of a facies. Samples were chosen from these cores primarily because they span the study region geographically (Fig. 1b), allowing for comparison of environments across the landscape. Samples further represent facies transitions, and therefore times of key environmental change through wetland development.

Fence diagrams were crafted for the following: (1) the marsh cores along each of the five transects; (2) a representative marsh-pond pair for each dominant vegetative environment; and (3) a transect from the center to the edge of a *Rhizophora mangle*-lined tidal pond. A vertical datum could not be achieved through surveying methods. Consequently, all stratigraphic columns along the transects were hung using the sediment surface as a datum. Tidal pond cores were vertically calibrated against the height of the pond's water surface in relation to their specific marsh pair. As the literature describes a gentle regional decrease in elevation moving coastward (Gleason and Stone 1994; Shrestha 2010; Michot et al. 2015), interpretations of depositional trends along transects could still be inferred.

## GIS Methodologies

ESRI, Inc. ArcMap (10.2.1) geographic information system (GIS) software was employed to inventory and analyze the aerial distribution of pocks within marshes of the TTI over time. The analysis compared the 1953 landscape with that in 2009. This time frame incorporates aerial images both before and after construction of the SGGE development and the Faka Union Canal. Over the 57 years encompassed in the scope of this study, relative sea level has risen by approximately 17–23 cm, (following estimates by Maul and Martin 1993), an increase likely large enough to alter the mosaic of vegetative communities across the landscape (IPCC 2014).

The 1953 and 2009 aerial imagery was acquired from the University of Florida Digital Collections Library and the Collier County Property Appraiser, respectively. The 1953 historic images were spatially georeferenced manually in ArcMap against the 2009 imagery, and subsequently converted to the Transverse Mercator Projection, NAD1983 Florida State Plane East coordinate system. Image resolution varied by 1 survey foot between 1953 (3 ft) and 2009 (2 ft) aerials;

based on the size and scope of the analyses conducted here, the impact of this variation on overall results should be inconsequential.

All image tiles gathered for analysis were taken during dry season months. Specifically, 1953 aerial images were taken between January 1 and January 4, 1953, and 2009 images utilized were taken January 9, 2009. Marsh flooding should be minimized during this time, allowing for greater differentiation of pond margins from the surrounding vegetation. In a region where seasonal hydrologic fluctuations can drastically shift the appearance of a landscape, evaluation of imagery reflecting the same seasonal conditions from year to year is critical (Ozesmi and Bauer 2002). Interior marshes of the TTI experience minimal tidal influence (Shresta 2010; Michot et al. 2015) and the delimitation of pond margins within aerial imagery should only nominally be affected by tide stage. The 2011 imagery from the Collier County Property Appraiser was referenced to confirm pond edges and minimize error that may result from digitizing ponds enlarged by short-term processes.

Ten marsh sites of 1.0 km<sup>2</sup> each were randomly selected for tidal pond analysis (Fig. 1a). Five of these were located west, and five east, of the Faka Union Canal to consider the influence of the absence (west) or presence (east) of natural sheet flow on landscape change. Supervised reclassifications of the image tiles did not suitably spectrally isolate tidal ponds from heavily vegetated landscapes. Thus, each pond within the ten marsh sites was manually digitized in ArcMap for each time frame and its area computed in square meter.

Pocks were assessed based upon their surrounding vegetation, tidal connectivity, and interaction with other ponds on the landscape within both time frames. Each feature was coded within the GIS software to reflect these characteristics. Such classification allowed for separate analyses of independent marsh ponds, mangrove-lined ponds, and tidally connected ponds to discriminate any trends that may have occurred in pond change over time among and between these groups. Tidal channels were digitized but removed from the analyses described here. Distinct spectral variation between landscapes within aerial imagery rendered classifying pocks as marsh or mangrove-lined relatively straightforward. Again, more recent regional imagery was consulted to confirm vegetation where uncertainties presented. Pocks enclosed within a mangrove canopy were likely underestimated in count and area due to difficulty differentiating between mangrove foliage and open water within the raster datasets. The size and number of mangrove-lined ponds are likely low, and thus data acquired for such ponds can be assumed to be a conservative estimate of what was truly on the landscape.

Statistical analyses of pock area and the change in pock density over time were explored using Synergy JMP 12 statistical software. Mean tidal pond area was statistically compared between time frames within marshes over the entire

study region, and by location east or west of the Faka Union Canal. Wilcoxon signed-rank nonparametric tests were performed due to the non-normality of the data. Significance was tested at the  $\alpha = 0.05$  level. Tidal ponds with 50% or greater mangrove lining were analyzed separately from marsh ponds, so that results could focus on the progression of pond development within a marsh landscape or within a mangrove forest, respectively. Results were then compared between the two groups to examine whether trends in pond change over time varied between the two.

The density of ponds (the number of ponds per total site area in km<sup>2</sup>) for each of the 10 sites was calculated. The total number of ponds and the change in pond count over time were extracted from this density analysis. Regression analyses compared the change in tidal pond density over time in relation to the site's distance from the Faka Union Canal. Distance was measured in kilometer from the center of each site perpendicularly to the edge of the canal.

## Results

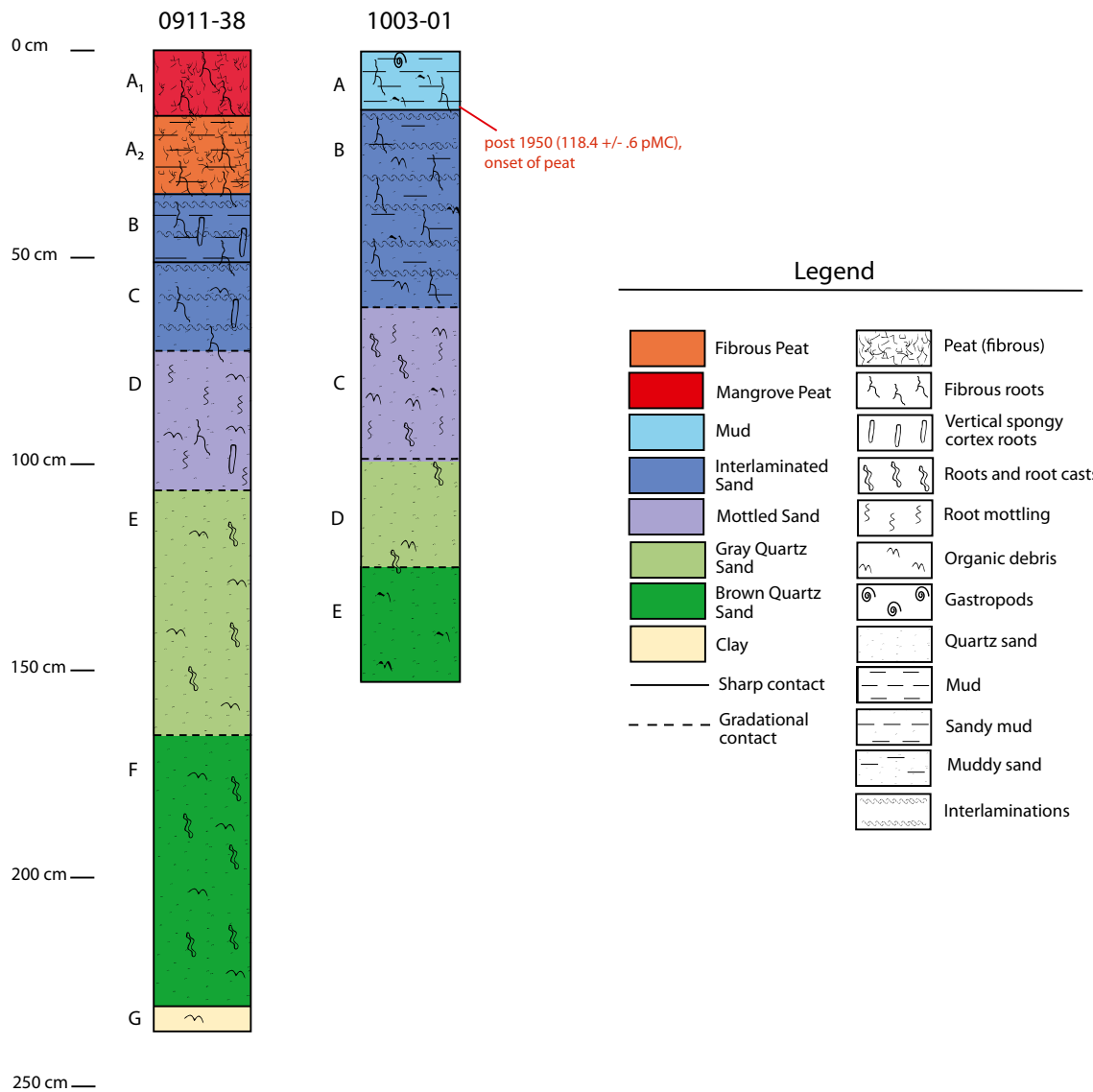
### Analysis of Vegetative Community Change

#### Stratigraphy and Sedimentology

Ten distinct facies were identified and described from analysis of the 35 cores: an incipient peat, fibrous peat, dense spongy cortex fibrous peat, mud, interlaminated sand, mottled sand, gray quartz sand, brown quartz sand, clay, and a carbonate-rich surface veneer (Fig. 3; Online Resource 1). Fence diagrams exhibit the paleoenvironmental interpretation of each facies and their lithocorrelation across localities (Figs. 4, 5, and 6; Fig. S1, Online Resource 2). The paleoenvironmental interpretations are justified in the discussion but used here to further the presentation of facies successions and patterns.

The most prevalent facies transition identified among all cores was from basal quartz sand facies, to interlaminated (or mottled, then interlaminated) sand facies, and ultimately to peat facies. This transition was maintained between marsh and tidal pond cores, except that most tidal pond cores (11 of 16) lacked peat and instead transitioned directly to surficial mud facies (Figs. 3 and 4). Two additional tidal pond cores contain both mud and peat facies above interlaminating sands. The surficial mud in 1003-01 (near WM TTI) and three other cores contain sparse roots and *Floridobia* sp. gastropod shells.

Comparable facies transitions occurred across transects. For example, the Cen TTI's 4 marsh and 1 mangrove core contain basal brown or gray quartz sand topped directly by interlaminated sands, or mottled then interlaminated sands (Fig. 5). In four of these cores, these facies are followed upsection by Fibrous or dense spongy cortex fibrous (DSCF) peat (see "Mangrove peat," Fig. 3; Online Resource



**Fig. 3** Stratigraphic columns for *Rhizophora mangle* community core 0912-38 of the WM TTI transect, and *Rhizophora mangle* tidal pond core 1003-01 of the transected tidal pond (near WM TTI) within the TTI. The

radiocarbon date for 1003-01 (red text) represents the onset of the mud facies and is reported in percent modern carbon (pMC). Core locations are noted in Fig. 1 b

1); core 0912-42 (Cen TTI) contains a surficial mud facies and lacks any peat. The four pond cores extracted along this transect also contain quartz sand below interlaminating sands, but contain surficial mud facies and lack peat. The tidal pond cores of the E TTI and the WFS transects also contain surficial mud and not peat above the interlaminated sands (Fig. 6; Fig. S1, Online Resource 2). Freshwater *Floridobia* sp. B gastropod shells and thin interlaminations are present within the surficial mud of 0911-27 (E TTI). The height of the lower facies contacts for interlaminated sands and surficial peats/muds generally moves upward within the marsh/mangrove cores traveling landward along each transect (Figs. 5 and 6; Fig. S1, Online Resource 2), although additional marsh cores spanning a greater transect distance would be useful to confirm this trend on a broader scale.

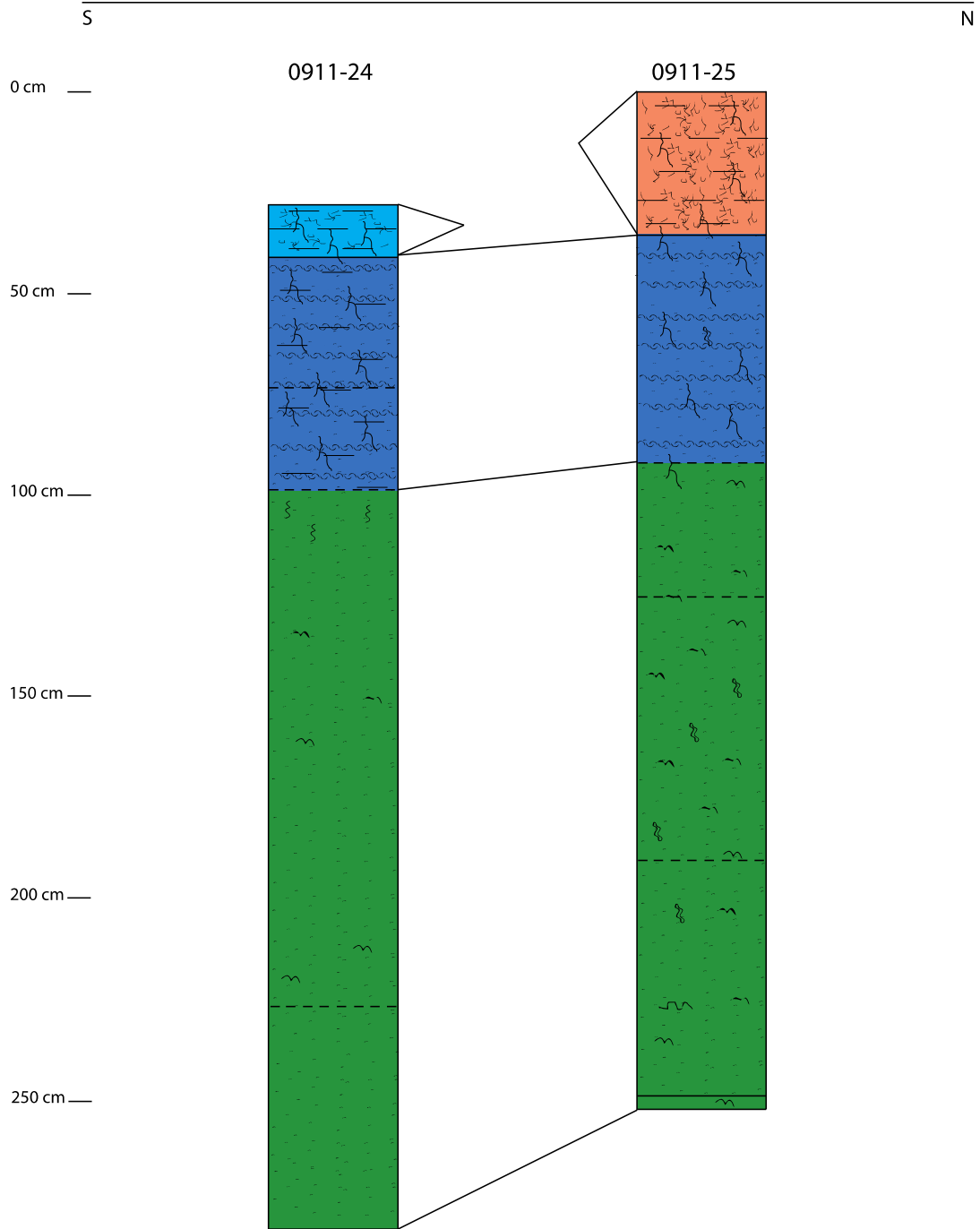
**Geochronology**

Radiocarbon samples have been named for the core number and its decompacted depth below the core surface (Table 1). Results for 0911-30 (Cen TTI; 71.7 cm), 1005-05 (WFS; 76.6 cm), and 1005-05 (104.2 cm) are listed in calibrated calendar years before present (Cal BP), where “present” refers to the year 1950. Due to occasional ambiguous results for the

**Fig. 4** Fence diagram of a *Distichlis spicata* marsh-pond pair from the E TTI transect, with colors depicting the interpreted depositional environments. Pond 0911-24 was vertically calibrated in relation to the adjacent marsh by measuring pond water depth, which provides a measure of the difference in height between the marsh surface and pond sediment surface

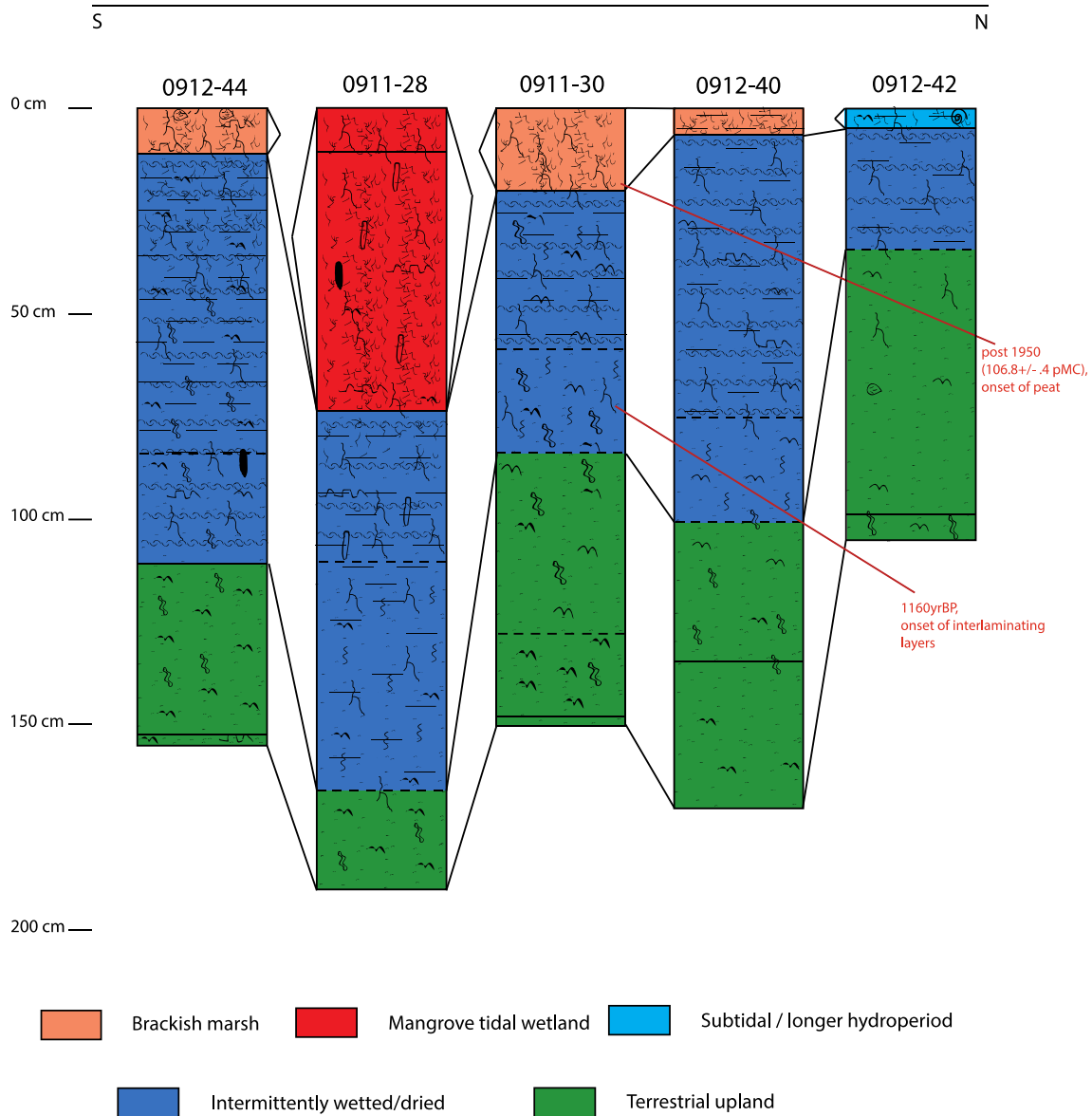


### Distichlis Marsh-Pond Fence Diagram



- Brackish marsh
- Subtidal / longer hydroperiod
- Intermittently wetted/dried
- Terrestrial upland

Central TTI Marshes Fence Diagram



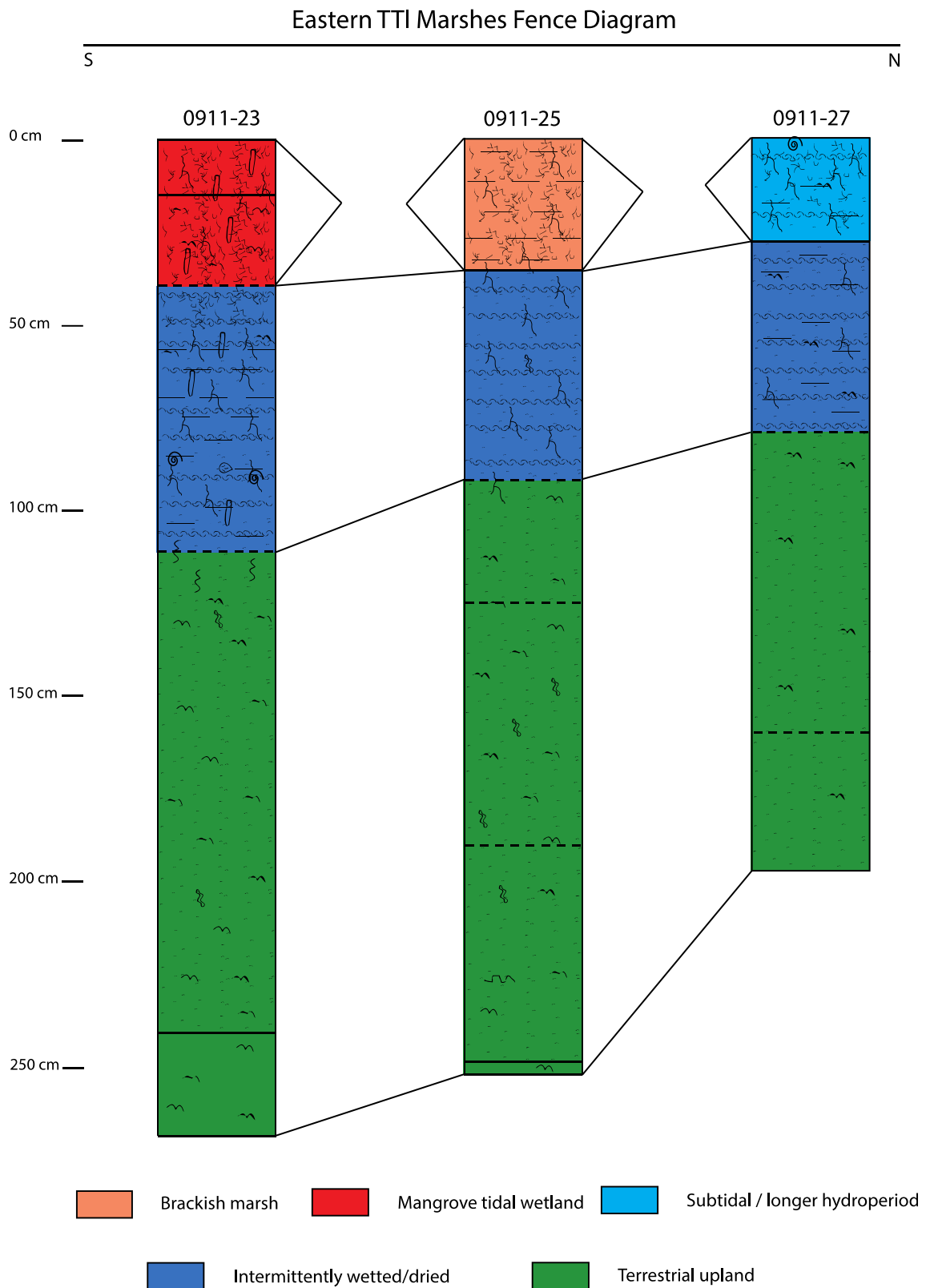
**Fig. 5** Fence diagram of the Cen TTI transect, with colors depicting the interpreted depositional environments. Lacking vertical reference data, all marsh cores have been placed at 0-cm elevation. Radiocarbon dates (red

text) are reported as the midpoint of the 2-sigma calibrated range (95% probability) in calendar years before present (yBP), or as percent modern carbon (pMC), where appropriate

2-sigma calibrated age range, dates are presented as the midpoint of the 2-sigma calibrated range (95% probability), where applicable. Levels of <sup>14</sup>C within samples 0911-30 (17.3–18.5 cm) and 1003-01 (near WM TTI; 13.3 cm) exceed that of the modern (1950) reference standard. Thus, dating results for these two samples are presented in percent modern carbon (pMC) rather than Cal BP (Table 1).

For all cores sampled, downsection facies are older than upsection facies; there is no stratigraphic disordering within a core. Additionally, the onset of the surficial mud and peat

facies of cores 1003-01 and 0911-30, respectively, returned modern (post-1950) radiocarbon dates. Onset of peat development within eastern marsh core 1005-05 occurred earlier in time (Cal BP 150, 2-sigma midpoint date) than in western marsh core 0911-30 (post-1950), although one 2-sigma calibrated age range for the 1005-05 sample includes recent times (Table 1). Onset of interlaminations within core 0911-30 (Cal BP 1160) occurred prior to the onset of the same facies in core 1005-05 (Cal BP 1040), although the 2-sigma calibrated ranges for these two dates overlap.



**Fig. 6** Fence diagram for the marsh and mangrove cores of the E TTI transect, with colors depicting the interpreted depositional environments. Lacking vertical reference data, all marsh cores have been placed at 0-cm elevation

**Table 1** Radiocarbon (RC) AMS ages and location by sample. One-sigma (68% probability) and 2-sigma (95% probability) calibrated age ranges are included, and ages are listed in calendar years before present (Cal BP), or, where applicable, in percent modern carbon (pMC). The calibrated age in Cal BP represents the midpoint of the 1-sigma range. The 2-sigma midpoint date represents the estimated median date of all combined 2-sigma age ranges. This midpoint date was used in interpretation

| Sample no.              | Facies                       | Modern Env't.                         | Lat           | Long          | Conventional RC age 14C | Calibrated age   | 1- $\Sigma$ calibrated range (68%)   | 2- $\Sigma$ calibrated range (95%)                        | 2- $\Sigma$ midpoint date |
|-------------------------|------------------------------|---------------------------------------|---------------|---------------|-------------------------|--|--|---|---------------------------|
| 0911-30<br>17.3–18.5 cm | Onset of peat                | <i>S. bakeri</i> marsh                | N 25° 56.758' | W 81° 32.243' | 106.8 ± .4 pMC          | Post-1950  | –  | –   | –                         |
| 0911-30<br>71.7 cm      | Onset of interlaminated sand | <i>S. bakeri</i> marsh                | N 25° 56.758' | W 81° 32.243' | 1230 ± 30 BP            | Cal BP 1170  | Cal BP 1230–1210 and Cal BP 1180–1170 and Cal BP                             | Cal BP 1260–1060  | Cal BP 1160               |
| 1003-01<br>13.3 cm      | Onset of mud                 | <i>R. mangrove</i> -rimmed tidal pond | N 25° 57.513' | W 81° 33.995' | 118.4 ± .6 pMC          | Post-1950  | –  | –   | –                         |
| 1005-05<br>76.6 cm      | Onset of peat                | <i>S. bakeri</i> marsh                | N 25° 55.963' | W 81° 28.471' | 200 ± 30 BP             | Cal BP 280 and Cal BP 170 and Cal BP 150 and Cal BP 0 and Cal BP 0 | Cal BP 290–270 and Cal BP 190–180 and Cal BP 170–150 and Cal BP 10-post-1950 | Cal BP 300–260 and Cal BP 220–140 and Cal BP 20-post-1950 | Cal BP 150                |
| 1005-05<br>104.2 cm     | Onset of interlaminated sand | <i>S. bakeri</i> marsh                | N 25° 55.963' | W 81° 28.471' | 1130 ± 30 BP            | Cal BP 1060 and Cal BP 1030 and Cal BP 1010                        | Cal BP 1120–1110 and Cal BP 1080–960   | Cal BP 1120–1110 and Cal BP 1080–960                      | Cal BP 1040               |

## Analysis of Tidal Pond Distribution and Dynamics Via GIS

### Pond Count

Pock count data were derived from the density analysis. Whether ponds encircled with 50% or greater mangrove perimeter (mangrove-lined ponds) are included or excluded (Table S1 and S2, respectively, Online Resource 2) the trend is the same. The number of pocks increased between 1953 and 2009 for each site except site 10, and there was an overall increase in pock count both west (of 1811 ponds) and east (of 646 ponds) of the Faka Union Canal (Fig. 7a). The greatest increase in total pock count (167%) and rate of change in pond count (~20 ponds/y) west of the canal occur in site 5, nearest the Faka Union Canal. The smallest increase in total pond count (2.7%) and lowest rate of change (<1 pond/y) occurred at site 1, the site farthest away from the canal (Fig. 7b; Tables S1, Online Resource 2).

East of the canal, the greatest increase in total pond count (71.7%) and rate of increase (6 ponds/y) occurred at site 7. Site 9 had a much higher rise in marsh ponds (98.5%) than other eastern sites, although the rate of pond count growth was still relatively low (2.3 ponds/y; Tables S1–S2, Online Resource 2). Site 10 was anomalous in that it was the only site to decrease in pond count; the decrease in rate is nearly constant when all ponds (11%) or only marsh ponds (10.5%) were counted (Fig. 7b).

### Pond Area

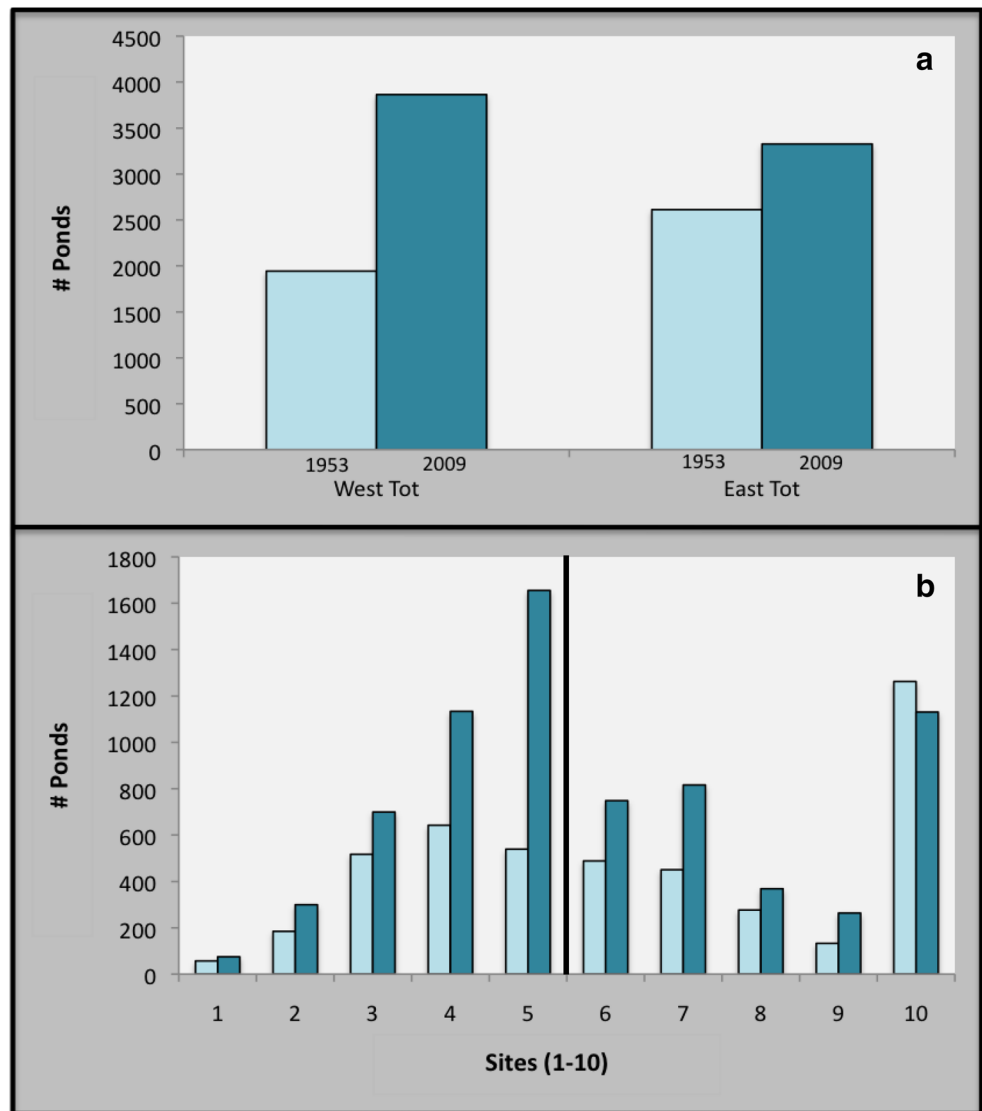
Overall mean marsh pond area significantly increased by 39% (or 44 m<sup>2</sup>) from 1953 to 2009 ( $p < 0.0001$ ). Wilcoxon Pairwise Tests also indicated a significant increase in mean pond area between east 1953 and east 2009 (of 21%, or 13 m<sup>2</sup>;  $p < 0.0001$ ) and between west 1953 and west 2009 (of 85%, or 31 m<sup>2</sup>;  $p = 0.03$ ). West 2009 had a significantly lower mean pond area than east 2009 ( $p < 0.0001$ ) (Table S3, Online Resource 2).

For mangrove-lined ponds, a statistically significant decrease in mean pond area was found both overall (8%, or 16 m<sup>2</sup>;  $p < 0.0011$ ) and by location ( $p < 0.0001$ ). Pairwise analyses indicated that east 2009 had a significantly lower mean pond area than east 1953 (36 m<sup>2</sup>;  $p = 0.0045$ ). Mean mangrove-lined area was not statistically different between west 1953 and west 2009 ( $p = 0.0591$ ) (Table S4, Online Resource 2). West 2009, however, had a significantly higher mean mangrove pond area than east 2009 (11 m<sup>2</sup>;  $p = 0.0179$ ).

### Pond Density

The change in pond density per site significantly increased as distance to the Faka Union Canal decreased. This result was apparent whether marsh ponds ( $p = 0.0038$ ) or mangrove-

**Fig. 7** Tidal pond count summary for marsh ponds **a** west and east of the Faka Union Canal for 1953 and 2009, and **b** for each of the 10 sites for 1953 and 2009. Light colored bars represent 1953 pond counts; darker bars represent counts in 2009. The location of the Faka Union Canal relative to the position of the sites is represented in **b** by the vertical black bar



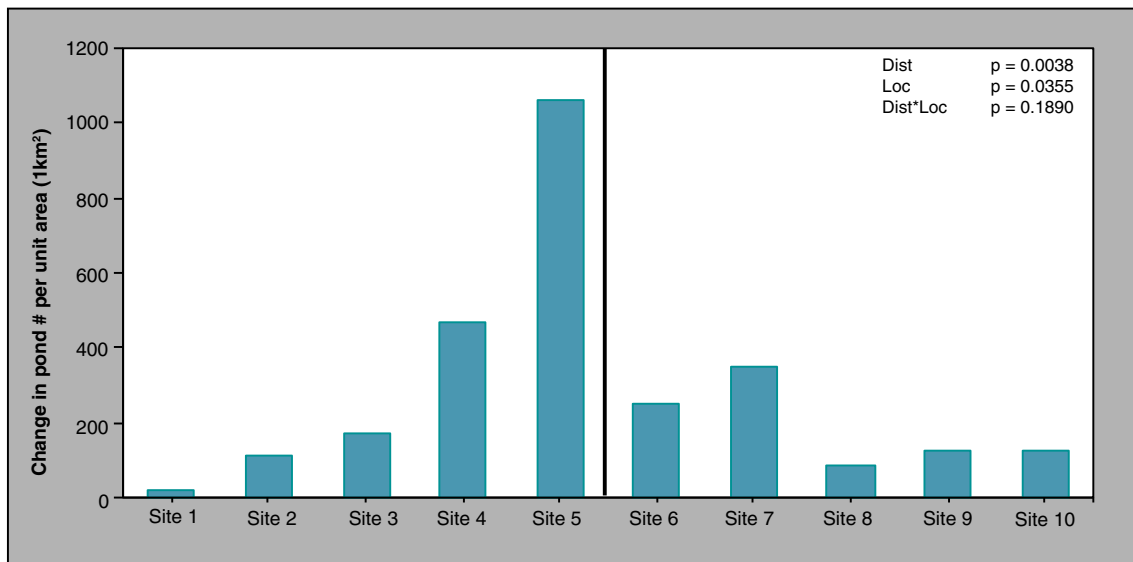
lined ponds ( $p = 0.0040$ ) were examined (Fig. 8). Regression analyses indicate that, in addition to distance, location east or west of the canal had a significant impact on the change in density over time ( $p = 0.0355$ ). Specifically, the rate of increase in pond density moving closer to the canal was greater in western than in eastern sites.

## Discussion

### Identifying Patterns of Vegetative and Wetland Community Change Over Time

Analyses of the ten identified stratigraphic facies reveal five depositional environments (Table 2; Online Resource 3). Commonalities in the transition of these depositional environments were identified among all cores (Figs. 4, 5, and 6). Moving upsection, a distinct transgressive shift

from pre-wetland terrestrial sands to an intermittently wetted and dried (likely seasonally or longer) wetland is present in all but one core (1005-04, WFS). These periodically wet environments are directly succeeded by marsh or mangrove communities in 14 of the 35 cores. Twelve additional cores display a trend from short hydroperiod wetlands to terminal modern longer hydroperiod environments. Five cores indicate a landscape shift from uplands, to intermittent wetlands, subsequent longer hydroperiod wetlands, and then ultimately modern marsh or mangrove environments. In general, sediments become muddier, more organic-rich, and exhibit greater interlamination character upsection, indicating greater aquatic influences. All cores clearly indicate former upland communities giving way to periodically more aquatic environments over time. Thus, wetland community transition clearly follows a predictable and transgressive pattern in response to regional SLR, confirming hypothesis H1.



**Fig. 8** Graphical representation of the change in tidal pond density per site between 1953 and 2009. Black line in the center separates western and eastern sites. Statistical *p* values are included

The coastal margin of the TTI has been relatively static for the last 3200 years (Parkinson 1989; Parkinson et al. 1994; Wanless et al. 1994). If this landscape is in fact transgressive, the depth of the base (representing the initial onset) of wetland facies should shift upward toward the modern surface traveling landward along each transect. It is important, however, to note the limitations of this analysis. Any error associated with the differential compaction of the cores would make the depths of facies transitions approximations, rather than exact measurements. Given the remoteness of the area and the impracticality of using RTK survey technology, it was difficult to calibrate marsh surfaces against a vertical datum. Regardless of these hindrances, the shallow gently sloping regional topography has been established (Shresta 2010; Michot et al. 2015), making the problems of these assumptions less tenuous.

The onset of intermittent wetland facies is generally deeper (allowing for some error due to the estimation of elevation

patterns), and presumably older, among marsh cores closest to the intracoastal margin, suggesting an earlier initiation and longer persistence among more coastal sites (Figs. 5 and 6; Fig. S1, Online Resource 2). Radiocarbon dates derived from cores 0911-30 (Cen TTI) and 1005-05 (WFS) indicate an onset of the intermittent wetland occurred approximately 1000 Cal BP (2-sigma midpoint date; see Table 1). This periodically inundated environment’s expansion landward into new environments likely accompanied the relatively slow regional rate of SLR (of 4 cm/100y) experienced during that time (Gleason and Stone 1994; Wanless et al. 1994; Savarese and Hoeflein 2012).

Dates derived from the onset of the long hydroperiod facies in core 1003-01 (near WM TTI) indicate this sediment appeared on the landscape in modern times (post-1950), after the appreciable increase in the rate of SLR described by the literature (Table 1; Maul and Martin 1993; Wanless et al.

**Table 2** Nine stratigraphic facies and their interpreted depositional environments. The CaCO<sub>3</sub> facies, derived from local periphyton communities, is not included in this interpretive grouping

| Facies  | Depositional environment         | Diagnostic features of depositional environment  |
|---|----------------------------------|--|
| Fibrous peat<br>Incipient peat                | Brackish marsh                   | Dense to loose peat; Fine vertical roots and randomly oriented in situ fibrous roots; Greater mud matrix-to-root ratio than mangrove peats (below).  |
| DSCF peat<br>Incipient peat                   | Mangrove tidal wetland           | Dense, well-developed peat, mat of roots in all orientations; Thick roots with protective sheath layers prominent; Reddish to dark brown in color, driven largely by vegetation; Greater root: matrix ration, coarse roots, and woody fragments than marsh peat (above). |
| Interlaminated sand<br>Mottled sand           | Intermittent wetland             | Alternating layers of organic-rich muds with quartz sand, or alternating layers of quartz sands; Indicative of a system where deposition occurs in cyclical shifts.  |
| Mud   | Long hydroperiod wetland or pond | High proportion of mud-rich matrix; Often scattered fine fibrous roots; Occasional presence of freshwater dominant hydrobiid shells.   |
| Gray quartz sand<br>Brown quartz sand<br>Clay | Terrestrial upland               | Predominantly medium-quartz sands; Varied presence of scattered organics, but overall low; Clay and clayey sands included, as a clay-rich dissolution residuum frequently resides atop bedrock regionally.   |

1994; Toscano and Macintyre 2003; Savarese and Hoefflein 2012). Using a higher resolution technique, such as Pb-210 dating, to acquire further wetland onset dates both east and west of the canal could confirm this result on a broader scale. Marsh peat was determined to be of modern origin (post-1950) in western core 0911-30 (Cen TTI) and Cal BP 150 in eastern core 1005-05 (WFS), with overlap in the age range for these estimates. Much like their pond counterparts, modern marshes are relatively recent, proliferating within the last 200 years, and likely since the onset of increased SLR.

Marsh cores east of the Faka Union Canal exhibit the largest thicknesses of the long hydroperiod facies among the cores in which it occurs. The greater facies thickness could indicate a longer duration for the long hydroperiod wetland on the landscape, or a higher sedimentation rate during the same time frame east of the canal. The comparatively less disturbed freshwater sheet flow in eastern sites would provide for higher quantity and duration of freshwater delivery, allowing more opportunity for accumulation of fine-grained sediments within shallow submerged environments than west of the canal.

### Identifying Tidal Pond Precursor Vegetative Habitats

This research suggests that ponds of the TTI are concentrated within graminoid marshes. Of the 16 tidal pond cores extracted, nine were surrounded by marsh vegetation, two were encircled with mixed marsh-mangrove stands, and five were bordered by mangroves (with two of these five cores from the same tidal pond). While the loss of specific marsh types was not quantified here, the results demonstrate that there are notably more ponds within *Spartina bakeri* marshes than other sampled community types. Whether this observation is more directly attributable to *S. bakeri*'s regional prominence, or to its unique physiological responses to stress is unknown (Barry 2009; Barry et al. 2013). More research is needed to substantiate the preferential wetland loss suggested in H2. Specifically, studies focused on the responses of regional dominant marsh and mangrove vegetation to variations in sediment dynamics, hydrologic regime, and tidal flushing brought about by SLR and altered hydrology will help elucidate where—and why—pocks are most likely to develop.

Results synthesized here indicate the long hydroperiod wetland facies represents the initiation of pock development on the landscape. These sediments are present in 14 of the 16 pocks sampled (and directly at the surface in 12 of these). Long hydroperiod facies thickness is greater in pock cores than marsh cores for 12 of the 14 pairs (Fig. 4), indicating greater mud accretion since pock inception and implying this layer is depositional within a pond environment. More in-depth sedimentological analyses of water content, percent organics, and nutrient ratios of the long hydroperiod wetland facies are needed to determine whether this pond facies is characteristically synonymous with the unique pond sediments described by Wilson et al. (2010) for Maine pocks.

The upsection peat facies depicting modern marsh and mangrove environments were noticeably truncated or absent within partner pock cores (Fig. 4). In fact, only five of the 16 pock cores contained any peat. This reduction in surficial peat in pock cores suggests degradation of modern peats from the surface down in pock inception, implicating a similar mechanism of development as described by Pethick (1974), Stevenson et al. (1985), and DeLaune et al. (1994) for marsh systems elsewhere. Peat degradation has been hypothesized in portions of the Everglades over the Fort Thompson Formation due to marine transgression, and in sediments beneath Lake Okeechobee resulting from increased lake submersion (Gleason and Stone 1994).

A comparable mechanism of pond development, growth, and merger has been described for marsh systems in non-karsted environments (Stevenson et al. 1985; Kearney et al. 1988; DeLaune et al. 1994; Wilson et al. 2007; Kearney and Rogers 2010; Wilson et al. 2010), and there is no evidence to corroborate that pock development is occurring as a result of local karst topography. Due to the composition of the Tamiami Formation limestone, karst features are not as prominent in subtropical South Florida as they are in other parts of the state (Denizman and Randazzo 2000). Areas within Collier County only infrequently develop sinkholes (Sinclair and Stewart 1985). The clayey sand residuum layer frequently recognized just above bedrock in Southwest Florida (Parkinson 1987; Klay 1989; Echols et al. 2009) can act as a layer of negligible permeability, largely preventing water from filtering through to the limestone and ultimately inhibiting further dissolution from occurring (Sinclair and Stewart 1985; Lane 1986). Even more, sediments just above the reaction site would be most impacted by the weathering of bedrock. Limestone dissolution, then, would likely result in the truncation of facies from the core bottom moving up, rather than the top down as is validated here.

As SLR continues at the current, or an even greater rate, the ecologically significant marshes of the TTI are expected to experience increased submersion through pock development in addition to shifts in range (Barry 2009; Krauss et al. 2011; IPCC 2014). The increase in salinity and tidal incursion into marsh environments concomitant with rising sea level could shift the regional distribution and extent of local marsh and mangrove species (Michot et al. 2015; Howard et al. 2016) and result in plant mortality and subsequent peat loss as described here (Pethick 1974; DeLaune et al. 1994; Kearney and Riter 2011; Sandi et al. 2018). The inland migration of marine conditions has already raised salinity levels within previous brackish marsh environments of the TTINWR (Krauss et al. 2011) and resulted in the growth of *S. bakeri*/*D. spicata* mixed-marshes, a community that essentially did not exist 70 years ago (Barry 2009). Higher localized salinities from marine incursion (which may be further exacerbated due to autocompaction and subsidence) likely provide ample stress

to marsh vegetation. As their adaptive capacity is compromised, individual mortality increases, providing a conduit for subsidence, submersion, and ultimate pond development (Stevenson et al. 1985; DeLaune et al. 1994).

Studies by Pethick (1974) and Adamowicz and Roman (2005) indicate pock size and prevalence increasing with marsh height. Pock frequency in *S. bakeri* stands in the TTI may thus be attributable to its incidence in susceptible mid- to high-marsh zones (Pethick 1974; Kearney et al. 1988; Adamowicz and Roman 2005). A notable and abrupt decrease in elevation has been observed at the mangrove-marsh interface within TTI wetlands (e.g., Krauss et al. 2011), a zone often dominated by *S. bakeri* or mixed *S. bakeri* stands (Barry 2009). Groundwater-induced peat shrinkage and peat oxidation with exposure can naturally occur in high marsh environments with seasonal drying (DeLaune et al. 1994; Rogers et al. 2006; Krauss et al. 2011), and result in a net elevation loss over time (Kearney and Riter 2011). These have been exacerbated by drought-like conditions that extend into the wet season due to anthropogenically reduced sheet flow (U.S. Army Corps 2004; Rogers et al. 2006).

### Identification of Tidal Pond Initiation, Growth, and Merger in Response to SLR

GIS analyses indicate a net loss of wetlands through the growth of subtidal pocks over time. Qualitative and quantitative evaluation of aerial imagery clearly confirms tidal pond initiation, growth, and merger are occurring in support of hypothesis H3. The total number of pocks on the landscape increased by 2457 over the 57 years analyzed. Trends in pond count and mean pond area varied among all 10 sites, corroborating evidence that pock formation is not uniform across wetland surfaces (Adamowicz and Roman 2005). Sites 1 through 9 experienced an increase in total pond count, with the lowest rate of positive change (0.05 ponds/y) occurring in site 1 and the highest rate (19.91 ponds/y) in site 5. A small increase (or decrease) in pond number and concomitant large increase in pond area was recognized in sites 3, 8, and 10. This indicates that either a) a few ponds are increasing in size disproportionately to the rest of the ponds in these sites, or b) that pond merger is occurring across the landscape (Fig. 2; Tables S1 and S3, Online Resource 2). Site 1 exhibits a small increase in pond number and a decrease in pond area between 1953 and 2009 (Fig. 7; Tables S1 and S3, Online Resource 2), signifying that aerial loss of extant ponds is greater than the rate of pond growth within this site.

Pocks alter the hydrodynamics and sediment transport along a marsh surface. Resulting site characteristics are often less suitable for marsh species reestablishment and encourage further pond formation (Stevenson et al. 1985; DeLaune et al. 1994). While localized stress often leads to the development and subsequent growth of ponds, dynamic patterns of

inception and revegetation are prominent within pock history (Harshberger 1916; Adamowicz and Roman 2005; Wilson et al. 2007). Within the TTI, recolonization of submerging wetlands could occur via marsh or mangrove vegetation. We posit that mangrove incursion onto marsh surfaces, and the envelopment of marsh pocks by mangroves, is a more likely vector for pond loss than marsh recolonization.

In southeast Australia, mangrove expansion increased in salt marshes where autocompaction and subsidence resulted in a net loss of marsh surface elevation, despite marsh vertical accretion (Rogers et al. 2006). The increase in salinity, tidal range, water depth, and hydroperiod concomitant with marine incursion may reduce the competitive capacity of marsh species while simultaneously promoting the dispersal of mangroves onto marsh surfaces (Krauss et al. 2011). Brackish marsh species may further facilitate mangrove incursion by providing structural support to, or by ameliorating the micro-environment around, the propagule during establishment (McKee et al. 2007; Sandi et al. 2018). When successful colonization results, the establishing mangroves become a competitive force for marsh dominants. The combined effects of submersion and competition via continued mangrove encroachment likely result in the eventual loss of marsh vegetation (e.g., Rogers et al. 2006).

Mean tidal pond area over the entire region significantly increased for marsh ponds ( $p < 0.0001$ ) and significantly decreased for mangrove-lined ponds ( $p < 0.0011$ ) from 1953 to 2009. Results clearly demonstrate that the net increase in marsh pond area over time (+ 275,670 m<sup>2</sup>) vastly exceeds the aerial loss in tidal ponds due to the lateral growth of mangroves (− 50,127 m<sup>2</sup>), implicating a net wetland loss within the marsh/mangrove complex of the TTI over the 57 years analyzed. The overall increase in pond count over time was driven by higher rates of pocking in marsh (rather than mangrove) wetlands, supporting the stratigraphic analyses. Mangrove incursion into previously pocked marshes, though not quantified, is easily identifiable within the aerial imagery. Data indicate a net loss in the number of mangrove-lined ponds in 9 of the 10 sites; site 5 exhibited only a marginal increase of 20 mangrove-lined ponds (Tables S1–S2, Online Resource 2). With slower accretion rates and lower productivity than mangroves (Sandi et al. 2018), marsh species' physiological thresholds may be reached before those of mangroves when faced with additional stressors driven by pock development. The physiological responses of mangrove species to pond growth are currently unknown.

Due to overlapping spectral signatures of mangrove and open water within the aerial imagery, it is possible that ponds considered nonexistent by 2009 are persisting under overhanging mangroves, or that new ponds within mangrove overgrowth cannot yet be identified. Evaluation of more recent aerial images coupled with field assessments, where feasible, could elucidate this assumption. Whether such ponds,



however, have the capacity to overcome the surrounding mangrove vegetation and enlarge is yet to be seen. At the very least, incipient mangrove ponds, and marsh ponds that become fully encompassed within dense mangrove vegetation, pose a low risk to further marsh degradation.

The incursion of mangroves is believed to be a later step in tidal pond evolution. Five of the seven *R. mangle* or mixed-mangrove ponds sampled were large, well-developed, deeper lakes in comparison with pocks farther inland from the coastal margin. The mean area of mangrove-encircled ponds was comparably larger than those of marsh ponds (Tables S3–S4, Online Resource 2), suggesting *R. mangle* ponds have been developing for longer relative periods of time than their marsh counterparts. These trends support the interpretation that mangrove establishment generally occurs after pond formation. The increase in overall pond area, and the potential for mangrove encroachment to decrease tidal pond expansion described here, corroborates the results of Hoyer (2009). Again, the mangrove encroachment and occasional complete consumption of tidal ponds within aerial imagery are interpreted as a permanent reduction, or even total loss, of such ponds (Table S4, Online Resource 2).

The extensive merger of individual ponds into indistinct pond networks evident across the study region indicates a potential mechanism leading to the subsequent formation of larger bays (Fig. 2c; Hoyer 2009). Sites 3, 4, and 5 experienced comparably larger increases in pond area than all other sites due to the expansive growth of pond networks from individual ponds. The total pond area of site 3, for example, increased by just over 77 km<sup>2</sup> between 1953 and 2009, equivalent to a rate of over 1.3 km<sup>2</sup> mean pond area per year. Although the geomorphologies of the TTI and northwestern Everglades, where such bays are common, are unique, both indicate pond expansion at the expense of graminoid-dominated habitats, and the patterns of growth evident within sites 3 through 5 illustrate how the degradation of wetland landscapes may promote bay formation.

### Effects of Wetland Change and Tidal Pond Expansion East and West of the Canal

GIS analyses confirm H4. Pond count increased at nearly three times the rate in the hydrologically altered western compared with eastern sites (Tables S1–S2, Online Resource 2). Mean marsh pock area rose at a significantly greater rate (by 85%) for sites west than those east (by 21%) of the canal ( $p < 0.0001$ ). Pond density also increased in all 10 sites between 1953 and 2009, and the magnitude of this increase lessened as the distance from the canal grew ( $p = 0.0038$ ). The rate of pond density change was greater moving toward the canal in the west (Fig. 8). It can thus confidently be concluded that tidal ponds are growing in size and number across the landscape, and that western sites experienced strikingly

greater pond inception and mean tidal pock growth from 1953 to 2009 than eastern sites. This signifies that ecosystem changes following canal construction are exerting some influence on pond growth.

Increased saltwater intrusion, the development of pocks, and mangrove encroachment are all expected both west and east of the canal (Gaiser et al. 2006; Krauss et al. 2011; Sandi et al. 2018). The interception of freshwater sheet flow and concomitant drawdown of groundwater reserves resulting from canal construction have clearly impacted TTI wetlands to the west much more so than wetlands to the east (e.g., Alexander and Crook 1974). Freshwater disruption has resulted in more pronounced and extended drought-like conditions, higher localized salinities, and marine conditions incurring farther inland west than east of the canal. Salinity maps of TTI wetlands indicate higher salinities within the western inner bays (Soderqvist and Patino 2010). Booth et al. (2014) report a mere 9% of all the freshwater sheet flow measured moving into TTI wetlands under U.S. 41 between 2008 and 2010 came from west of the canal, compared with 31% east of the canal.

Eastern mean mangrove-lined pond area decreased significantly from 1953 to 2009 ( $p = 0.0045$ ). Pairwise statistical analyses indicate the reduction in western mean mangrove-lined pond area is not significant ( $p = 0.0591$ ), though reveals a strong statistical trend. Yet, data indicate the west has a higher loss of net pond area, representing 26,322 m<sup>2</sup>, to mangrove envelopment than eastern sites. In the Everglades, as likely here, landward mangrove migration onto marsh surfaces was exacerbated by altered freshwater flow resulting from the Faka Union Canal (Hoyer 2009). Yet the extensive pock formation recorded in western sites is dwarfing the impact of, even as altered hydrology may be promoting the inward movement of, mangroves across the landscape (e.g., Krauss et al. 2011; Michot et al. 2015).

The retention of near-natural surface flow in the east is likely providing an opposing effect to seawater invasion and reduced hydrologic resistance resultant from SLR (Gaiser et al. 2006; Barry 2009; Krauss et al. 2011; Michot et al. 2015). When present, surficial flows provide marsh surfaces with allochthonous sediment supply and promote marsh preservation (Kearney et al. 1988; Adamowicz and Roman 2005; Rodriguez et al. 2017). In systems where the primary mechanism for accretion is organic deposition, such sediment transport may be critical for vegetation to combat submersion (Stevenson et al. 1985; Kearney et al. 1988; DeLaune et al. 1994; Kirwan and Megonigal 2013).

Anthropogenic alterations to hydrology impose sizable impacts to a wetland's ability to counter SLR (Rodriguez et al. 2017; Sandi et al. 2018). Channelization effectively starves local wetland surfaces of both water delivery and sediment supply, resulting in elevation loss, submersion, and often increased drainage density (Stevenson et al. 1985; Kearney et al.

1988; Adamowicz and Roman 2005; Kirwan and Megonigal 2013). Specifically, it is likely that the loss of surficial sheet flow is driving the greater relative increase in pond count and area in the west, and also why the west experiences a higher rate of change as proximity to the canal increases. Here, the more hydrologically altered western sites exhibited greater overall tidal pond expansion; the rate of this growth increased as the distance to the Faka Union Canal decreased. Western TTI sites with greater tidal connectivity exhibited the most overall pond growth, as evident by the higher rates of pond area increases and the merger of once-independent ponds into pond networks in sites 3 through 5. Should large bays comparable with those described by Hoye (2009) become a threat to the TTI, regions with greater tidal connectivity and impaired sheet flow would likely be the first lost.

### Conclusions and Management Implications

When sea levels are rising, a coast and its vegetative communities may respond in one of two ways: (1) to keep pace with, or even prograde in the face of SLR, or (2) to experience marine incursion and the translation or ultimate loss of coastal environments. This study identified a clear and universal transgressive ecosystem shift within the TTI marshes over time. As site conditions change, plant stress may result in mortality or migration, ultimately shifting the abundance and distribution of local environments irrespective of pond development (DeLaune et al. 1994; Barry 2009; Kearney and Rogers 2010; Krauss et al. 2011; Howard et al. 2016). Recent studies have already identified a transition to more salt-tolerant species in low marsh environments (Barry 2009; Barry et al. 2013), as well as the encroachment of mangroves into previous marsh sites (Gaiser et al. 2006; Barry 2009; Krauss et al. 2011; Barry et al. 2013). With slower rates of accretion and lower tolerance to salinity and flooding than *R. mangle*, local marsh dominants such as *D. spicata* and *S. bakeri* are being lost to mangrove encroachment.

Submersion of wetland surfaces via pocking over time is clearly threatening wetlands of the TTI, as marsh systems elsewhere (e.g., Kearney et al. 1988; DeLaune et al. 1994; Wilson et al. 2010). Wetlands on both sides of the Faka Union Canal experienced marked pond inception and enlargement within the 57 years analyzed. Yet, significantly higher loss to pocking was quantified, and the greater influence of mangrove encroachment inferred, in water-managed western wetlands. Thus, land management techniques for the TTI will be largely ineffective without addressing the water management, increased marsh submersion, and accelerated SLR impacting this system.

Freshwater diversion through the canal system has resulted in the near-complete attenuation of sheet flow and a reduction in the groundwater table, ultimately leading to an overall shift in flooding depth and duration within western sites (U.S Army

Corps 2004; Shrestha 2010; Soderqvist and Patino 2010; Booth et al. 2014). The confounding stress of these changes is likely altering ecogeomorphic feedbacks within the marsh and mangrove systems (Kirwan and Megonigal 2013; Rodriguez et al. 2017), and is clearly accelerating the inception, growth, and merger of western ponds. Reduction in freshwater flows has increased the range of tidal flushing, promoting the dispersal of mangrove propagules farther inland into these wetlands (Krauss et al. 2011). Since 1940, the bulk of inland marsh loss to mangrove advancement in the TTINWR has occurred since the completion of the Faka Union Canal (Barry 2009; Krauss et al. 2011).

The highly hydrologically altered TTI is currently part of the largest proposed watershed restoration project in the country (U.S. Army Corps 2004). The Picayune Strand Restoration Project, a component of the Comprehensive Everglades Restoration Project, seeks to reduce freshwater flow from the Faka Union Canal by up to 99%, and return natural sheet flow distribution to TTI's wetlands (Shirley and Brandt-Williams 2001; U.S. Army Corps 2004). This restoration project outlines a plan to fill nearly 44 miles of canals, create pump stations, and remove 227 miles of roads. Progress toward these goals is well underway.

Proposed effects include the re-establishment of more natural patterns of overland sheet flow and discharge to bays other than Faka Union Bay. Such effects could potentially reduce subsidence and ultimately pocking, and thus minimize deleterious effects from accelerated SLR, especially west of the Faka Union Canal (Turner 2004, U.S. Army Corps 2004). The appreciably lower growth in pond count and area east of the canal can largely be attributed to the opposing influence of, and the likely delivery of allochthonous sediments with, surface flow from the near-natural Fakahatchee Strand watershed. Modeling scenarios indicate significant increases in hydroperiod and concomitant decreases in salinity should be expected with the return of freshwater flow to TTI wetlands, and that these changes increase with greater distance from the coast (Shrestha 2010; Michot et al. 2015). Enhancing hydroperiod and reducing salinity in this manner may shift the competitive advantage to marsh over mangrove species, decreasing the loss of marsh habitat and slowing the inland encroachment of mangroves already documented (Krauss et al. 2011; Michot et al. 2015).

Species' unique physiological responses to SLR and water management may influence the pattern and rate of community shifts and submersion (Kirwan and Megonigal 2013; Rodriguez et al. 2017); further studies are needed to explore the roles local species play in the patterns described here. Yet as recent model analyses of Australian marshes indicate, flow attenuation raises the rates of coastal wetland submersion overall. Returning freshwater flows, then, has the capacity to slow the rate of submersion and loss of all community types (Rodriguez et al. 2017; Sandi et al. 2018). Thus, the impact

surface flow restoration may thus have on combating tidal submersion should not be understated. Restoring natural sheet flow may too shift the convergence of marsh habitat from subtidal ponds to proportionately more mangrove tidal wetlands and slow the rate of these transitions (Barry 2009; Michot et al. 2015). Consequently, hydrologic restoration projects are vital, and should continue, despite the complexities of SLR and its influence within these environments.

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