Geomorphology 412 (2022) 108329

Contents lists available at ScienceDirect

Geomorphology

journal homepage: www.journals.elsevier.com/geomorphology

Accelerating sea-level rise and the fate of mangrove plant communities in South Florida, U.S.A.

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ARTICLE INFO

Keywords:
Climate change
Accelerating sea-level rise
South Florida mangroves
Sediment accumulation rates

ABSTRACT

Analysis of four South Florida tide gauges, with records ranging from 27 to 116 yrs., indicate the average rate of sea-level rise has accelerated from 3.9 mm yr^{-1} (1900–2021) to 6.5 mm yr^{-1} (2000–2021), and 9.4 mm yr^{-1} over the past decade. Future rates are forecast to accelerate over the duration of this century. A predictive conceptual framework (model) was developed in which the resilience of South Florida mangrove plant communities is solely a function of the rate of sea-level rise and vertical sediment accumulation. The model was verified using historical (e.g., 1900–2021) and recent (e.g., 2000–2021) sediment accumulation rate data derived using three different methodological approaches. Results indicate by 2040–2050 South Florida mangrove plant communities, already subjected to the destabilizing effects of accelerating sea-level rise for decades, will begin a widespread conversion to estuarine conditions. This will initially trigger the formation and expansion of inundation ponds, as is already occurring in the study area. By the end of this century, most mangrove forested areas will be submerged. The loss of other coastal wetlands (e.g., brackish marsh) is also likely because their rates of sediment accumulation are lower than mangrove. The findings of this study are consistent with other resilience projections that have been conducted in South Florida and at the global scale. Several knowledge gaps were identified which must be filled to improve confidence in subsequent forecasts and the outcome of mitigation efforts undertaken to enhance resilience. These include the lack of (1) sediment accumulation data representative of transitional and freshwater wetlands, (2) a robust understanding of post-depositional processes (e.g., compaction) that can compromise resilience through shallow (~1 m) subsidence, and (3) recent sediment accumulation data necessary to determine how South Florida coastal wetlands have responded to a sustained acceleration in the rate of sea-level rise over the last decade.

1. Introduction

Coastal wetlands occur at the land-sea interface and are susceptible to changes in sea level (e.g., direction, magnitude, rate). Over the past several thousand years, sea level has been relatively stable (rate <1 mm yr^{-1}) in North America (Donoghue, 2011; Kopp et al., 2015) and globally (Nicholls and Cazenave, 2010). Under these conditions, trends in wetland distribution, ecology, and sedimentation were primarily constructive because the relatively slow rate of sea-level rise (SLR) was generally exceeded by the capacity of coastal wetland plant communities to keep pace through vertical sediment accumulation and lateral migration. This provided conditions for the formation, stabilization, and expansion of coastal wetlands (Farron et al., 2020; Goff and Chagué-Goff, 1999; Mackenzie et al., 2020; Parkinson, 1989). Over the past 120 yrs., the global rate of SLR has accelerated from 1.3 mm yr^{-1} (1901–1971) to 1.9 mm yr^{-1} (1971–2006), and further increasing to 3.7 (2006–2018) (Masson-Delmotte et al., 2021). Human influence was very likely the main driver since 1971. Faster rates (4.2–7.9 mm yr^{-1} from 2040 to 2060 and 4.3–15.9 mm yr^{-1} from 2080 to 2100) are forecasted towards the end of this century (Fox-Kemper et al., 2021).

Forecasts of accelerating rate of SLR in association with climate change have triggered myriad investigations designed to improve our understanding of how coastal wetlands will respond through adjustments in vertical sediment accumulation, lateral migration, and ultimately their fate or resilience over time (Chambers et al., 2021; Doughty et al., 2019; Holmquist et al., 2021; Lovelock et al., 2021; Meeder et al., 2021; Rogers, 2021; Saintilan et al., 2021). Of particular concern is the potential loss of the global, regional, and local ecosystem services they

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https://doi.org/10.1016/j.geomorph.2022.108329
Received 23 February 2022; Received in revised form 25 May 2022; Accepted 26 May 2022
Available online 30 May 2022

provide (Barbier et al., 2011; Craft et al., 2009). These concerns are especially acute in South Florida (Koch et al., 2015; Obeysekera et al., 2015) as saltwater encroachment has been shown to decrease root-biomass production, organic carbon storage, and rates of vertical sediment accumulation (Charles et al., 2019; Kominoski et al., 2021; Servais et al., 2018; Wilson et al., 2018). Prior to this investigation, South Florida historical rates of SLR were reported as 2.4 mm yr^{-1} at Key West (1932–2013) (Maul and Martin, 2015) and ~ 4.5 mm yr^{-1} at Virginia Key (pre-2006) (Wdowinski et al., 2016). More recently, the rate of SLR has been reported as 6.3 mm yr^{-1} at Key West (2003−2012) (Breithaupt et al., 2017) and between 5.9 mm yr^{-1} (2005–2015) and 9 mm yr^{-1} (post 2006) at Virginia Key by Park and Sweet (2015) and Wdowinski et al. (2016), respectively. The 6–9 mm yr^{-1} rates of sea-level rise detected in south Florida since 2003 are significantly higher than the 3.2–3.7 mm yr^{-1} global mean sea level rise rate calculated for the same time period (e.g., Masson-Delmotte et al., 2021). Similar high rates of sea-level rise were detected along most of the southern and middle US Atlantic coast and were attributed to ocean dynamic effects, including the slowing of the Gulf Stream (e.g., Ezer, 2013, 2019) and temperature increase of the Florida Current (Domingues et al., 2018).

Research on the capacity of coastal wetlands to keep pace with recent and future accelerations in the rate of SLR has yielded mixed results. Some have argued they have the capacity to accommodate future accelerations (Breithaupt et al., 2020; Kirwan et al., 2016; Schuerch et al., 2018). Others have provided evidence of a sediment accumulation rate (SAR) deficit, habitat loss, and/or conversion to open water (Hatton et al., 1983; Lagomasino et al., 2021; Valiela et al., 2018). These differences are largely attributed to the timescale over which estimates of SAR are measured and the availability of horizontal accommodation space into which they can migrate (Holmquist et al., 2021; Tornqvist et al., 2021). While there have been numerous studies suggesting accelerating SLR will ultimately result in submergence and retreat, most have not constrained the timing of this transition other than to say it is inevitable or likely by the end of this century (c.f., Breda et al., 2021). The only exception appears to be Saintilan et al. (2020) and Sklar et al. (2021), who suggested the global and regional tipping point for mangroves will occur within the next 30 to 50 yrs., respectively.

This project was designed to answer the question *how will South Florida coastal wetlands respond to future conditions of accelerating SLR?* To answer this question, a conceptual framework or model was constructed to forecast resilience based upon a comparison between historical (~1900 to present) and recent (2000-present) rates of SLR and sediment accumulation. The term resilience is used in the broadest sense and considers the capacity of a coastal wetland to sustain its ecosystem structure, function, and service under conditions of SLR and concomitant saltwater intrusion. Resilient wetlands can sustain their structure, function, and service by accumulating organic and inorganic sediment at rates equivalent to the commensurate rate of SLR. Those that cannot become unstable, degrade (Yando et al., 2021), and transition into a more salt-tolerant plant community, bare mudflats, open water ponds or a subtidal environment. This type of analysis has been previously conducted on a limited scale (e.g., mono-specific taxonomic focus, small geographic area). In contrast, the research described herein is based upon an aggregate of historical and recent published data obtained from numerous coastal wetland habitats (e.g., salt, brackish, fresh) and methodological approaches (e.g., radioisotope geochronologies, Sediment Elevation Table Marker Horizon or SETMH) located within a broad geographic area. The data were interrogated to (1) estimate *when* the threshold rate of survival (Saintilan et al., 2020) and subsequent loss of extant South Florida coastal wetlands is likely to occur and (2) advance our understanding of the mechanics and consequences of timescale bias (Breithaupt et al., 2018; Parkinson et al., 2017; Tornqvist et al., 2021) and differing methodological approaches on the outcome of resilience projections.

2. Material and methods

2.1. Study area

The South Florida study area (Fig. 1) consists of about 7200 km^2 of coastal wetlands located in the Southeast Saline Everglades (SESE), Northeast Florida Bay (NEFB), Lower Florida Keys (LFK), Southwest Everglades National Park (SWENP), the Ten Thousand Island National Wildlife Refuge (TTI) and Rookery Bay National Estuarine Research Reserve (RB). South Florida coastal wetlands historically existed in concentric bands or belts (Egler, 1952) roughly parallel to the coastline and extending inland as much as 20 km. The predominant habitats include mixed (e.g., red, black, white) mangrove (e.g., fringe, forest, basin), scrub mangrove, salt and brackish marsh, wet prairie, and freshwater marsh. Substantial dewatering of the Greater Everglades watershed began in the 1880s and these diversions continue to restrict the flow of freshwater in the coastal wetlands and disrupt ecosystem structure and function (Dessu et al., 2018; McVoy et al., 2011). In tandem with historical and recent increases in the rate of SLR, this has promoted saline surface and groundwater encroachment into freshwater environments, as has been locally documented (Gaiser et al., 2006; Meeder et al., 2017), and a landscape level migration of salt-tolerant wetlands inland (Howard et al., 2020; Krauss et al., 2011; Ross et al., 2000) to create a transgressive sediment succession consisting entirely of autochthonous material (e.g., organic matter, calcitic mud or marl) (Meeder et al., 2021).

The study area is located on a low latitude, gently sloping, tectonically stable carbonate platform. There are no substantial sources of allochthonous (i.e., inorganic, clastic) sediment (Chambers et al., 2015). As a result, sediment production and accumulation are solely the function of biological processes (Meeder et al., 2021). Bedrock is generally less than a few meters below grade (Egler, 1952; Hoffmeister et al., 1967). The combination of the study area's geologic, geomorphic, and ecologic attributes yields a stable landscape that responds quickly to subtle changes in sea level. These responses are ultimately preserved in the sedimentologic and stratigraphic record.

2.2. Methods

2.2.1. Trends in historical and recent sea level

Linear trends and periodic changes of sea level in South Florida were obtained by analyzing 27–116 year-long tide gauge records of four NOAA stations: Naples (NA), Key West (KW), Vaca Key (VC), and Virginia Key (VK). The analysis was based upon NOAA mean sea level (MSL) monthly data products sampled at 6-minute intervals (NOAA, 2021), including monthly mean, lowest, and highest values.

Trends of sea level change were characterized using a multi-parameter linear least-square estimator, which provides both best-fit parameters and their uncertainties. The best-fit analysis was conducted using Python's scipy.optimize.curve_fit function. The analyses included linear fit (two parameters: intercept and slope) and linear + periodic (sinusoidal) fit (four parameters: intercept, slope, amplitude, and phase). To simplify the periodic signal calculations, the sinusoidal fit was conducted as a best-fit of sine and cosine periodic signals with zero phase. Using trigonometric relations, the best-fit amplitudes of the sine and cosine can be converted to the amplitude and phase of the single periodic signal. The periodic signal was set a priori to 18.61 years, which is the lunar tide periodicity, and has a significant impact on mean and high sea level changes (Peng et al., 2019).

Both the linear and linear + periodic best-fit analyses were applied to three time series representing monthly mean, lowest, and highest sea level using four subsets of the monthly data products to detect linear trend changes during different time periods. The four subsets were: entire length of record (start point until 2021.5), start point until 2000, 2000–2021.5, and 2010–2021.5. The amplitude and phase of the periodic signals were determined using the best-fit estimation of the entire

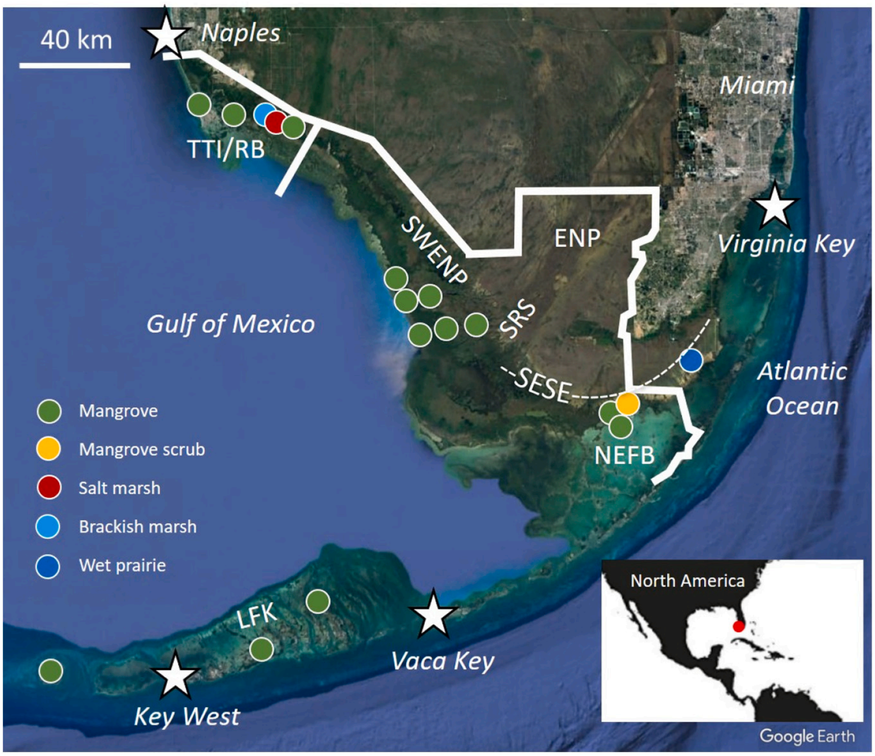

Fig. 1. Regional location map of South Florida peninsula. White stars mark the locations of the four tide gauges used in this study. Colored circles indicate approximate location and plant community type from which published sediment accumulation rate data were used in this study. SESE = Southeast Saline Everglades. NEFB = Northeast Florida Bay. LFK = Lower Florida Keys. ENP = Everglades National Park. SRS = Shark River Slough. SWENP = Southwest Everglades National Park. TTI = Ten Thousand Islands National Wildlife Refuge. RB = Rockery Bay National Estuarine Research Reserve002E.

dataset. When calculating linear + periodic fit with shorter span subsets, the amplitude and phase detected from the entire dataset was superimposed because the amplitude and phase of the 18.61 year long nodal tide periodicity can vary significantly when analyzing short subsets (<30 years). The values generated in this analysis are based on the linear + periodic calculations because these calculations provide better fit to the observations.

2.2.2. Sediment accumulation rates

Sediment accumulation rates representing the major coastal wetland plant communities in the region were aggregated from the published literature. This was accomplished by searching (e.g., Google, Google Scholar, FIU Libraries, authors' personal library) for relevant publications on the topic (e.g., sediment accumulation rate, sediment accretion rate, SETMH) and within the study area. The authors also sought data by directly contacting colleagues who have worked on this topic in South Florida. The sediment accumulation data were utilized in this study if it was presented in numerical format (e.g., 3.0 mm yr^{-1}) or in a graphical format wherein the values could be confidently converted to a numerical value (c.f., Smoak et al., 2013). The data were classified into one of three groups based on the method used to quantity SAR: core lump sum, core section intervals, and SETMH. Core lump sum SAR are based upon the analysis of a sediment core using radioisotope geochronologies (^{137}Cs, ^{210}Pb) (Delaune et al., 1978; Lynch et al., 1989). The results are reported as the historical average SAR for the entire core interval analyzed. Core section interval analysis emerged more recently, in which SAR data are reported as interannual values for specific sample intervals (typically 1–2 cm), corresponding depth and estimated age (c.f., Smoak et al., 2013). The interannual SAR data provides a basis for evaluating wetland resilience at historical, decadal, and recent timescales (Breithaupt et al., 2017). The SETMH method quantifies vertical accretion of above ground sediment deposition based upon measurements of sediment thickness above a marker horizon generally obtained at 6 months to 1 yr intervals. Changes in ground-level elevation are acquired simultaneously by measuring the displacement of a cluster of vertical pins relative to the station's elevation, which is precisely surveyed to a common datum (e.g., NAVD88). The difference between the two values is attributed to subsidence. The technique has evolved since it was first introduced (Boumans and Day, 1993; Callaway et al., 2013; Childers et al., 1993). The most recent configuration (Cahoon et al., 2002) is now widely utilized throughout the world (Lovelock et al., 2015; Webb et al., 2013). In this study, the vertical accretion values were used to calculate SAR. Consideration of the potential effects of below-ground, post-depositional processes on resilience were considered later.

2.2.3. Coastal wetland resilience

The resilience or fate of coastal wetlands was first *hindcast* by comparing various temporal combinations of historical and recent rates of SLR and published SAR data obtained from South Florida coastal wetlands. In contrast to other areas in the conterminous U.S. (e.g., Northern Gulf of Mexico, Hatton et al., 1983; Central and Northern Atlantic coast; Karegar et al., 2016), predictions of the response of coastal wetlands to SLR in South Florida do not have to consider vertical land motion caused by tectonic and isostatic adjustments or surface elevation changes caused by deep subsidence (aka., sediment consolidation) (Lynch et al., 2015; van Dobben et al., 2022). The projections are further simplified because there are no significant sources of inorganic allochthonous sediment that must be considered (c.f., Rogers, 2021). Therefore, resilience is simply modeled as the difference between the rate of SLR and SAR. As others have done (Osland et al., 2020; Sklar et al., 2021), a linear rate of change was assumed for both SLR and SAR over the duration of time being considered.

Forecasts of extant coastal wetland resilience were based upon a comparison between average historical and recent SAR and forecasted rates SLR to the end of this century. This is a common strategy, which has been locally used in Florida (Feher et al., 2020; Howard et al., 2020; Lynch et al., 1989; Parkinson et al., 1994; Sklar et al., 2021) and other locations throughout the world (Arias-Ortiz et al., 2018; Craft, 2012; Delaune et al., 1978; Lovelock et al., 2015; Saintilan et al., 2020; Webb et al., 2013). Predictions hinge on the magnitude and sign of the difference between the rate of SLR and SAR. If the rate of SLR exceeds the SAR (aka SAR deficit), instability (e.g., in situ degradation as defined by

Yando et al., 2021; habitat loss) is predicted. The larger the sediment deficit, the more rapid and widespread the destabilization. Stratigraphically, this generates a transgressive depositional succession consisting of a basal coastal wetland interval (e.g., freshwater peat, marl) that is overlain by sediments associated with more salt-tolerant plant communities (e.g., mangrove peat), mudflats, or open water (e.g., estuarine muds, organic floc, surface of non-deposition). If the rate of SLR equals SAR, a steady state is predicted. This produces a contiguous mono-specific sediment succession. If the rate of SLR is less than SAR wetland habitats are predicted to expand in both the vertical and lateral direction. Stratigraphically, this produces a regressive depositional succession of sediment types representing plant communities with ever shorter hydroperiods and/or an increasing tolerance to freshwater. Examples of all three sediment successions and their relationship to South Florida SLR can be found in Parkinson (1989). Three timescales were considered: historical (1900–2021), recent (2000–2021, 2010–2021), and future (present–2100).

Resilience was initially hindcast to determine whether the modeled outcomes were consistent with the published descriptions of coastal wetland evolution reported for the same area and timeframe. Similarity between the hindcasts and the outcome of previous studies provided a basis for validating the capacity of the conceptual model to accurately forecast the fate of coastal wetlands in South Florida under future conditions of accelerating SLR.

3. Results

3.1. Trends in historical and recent sea level

Historical SLR (1900–2021) (Fig. 2, Table 1) varied between the four stations, but on average was 3.8 mm yr^{-1}. Differences between the sea-level data recorded at each station are a consequence of variations in surface temperature, prevailing wind, semi-annual (e.g., sunny-day) and long-term (e.g., nodal) tidal cycles, the regional coastal sea level fingerprint (Mitrovica et al., 2009) and oceanic circulation (e.g., the Gulf Stream) (Ezer, 2020) unique to each location. During the 21st century, the average rate of SLR in South Florida was 6.5 mm yr^{-1} (2000–2021) and 9.4 mm yr^{-1} (2010−2021). Hindcasts of historical and recent trends in wetland resiliency were based upon sea-level records collected at the Key West station because: (1) it is the longest period of record, (2) is located in the same ocean basin as the vast majority of published accumulation data utilized in this study and (3) the Virginia Key site is

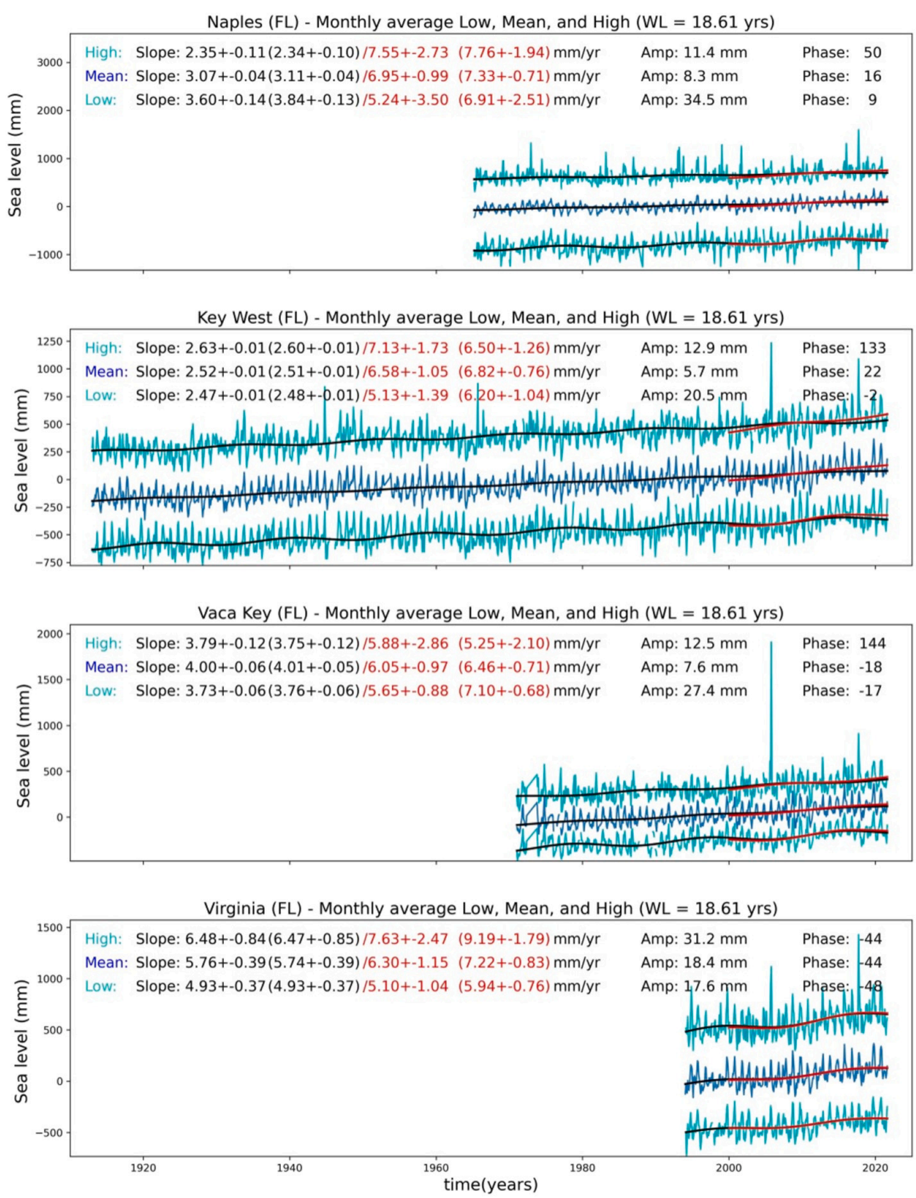

Fig. 2. Historical (all available data) and recent (2000–2021) mean high, mean, and mean low water level records of four South Florida NOAA tide stations. Best-fit parameter estimation of the historical data are shown in black fonts and recent (2000–2021) in red fonts. Values without parenthesis present the linear + periodic fit and those within the parenthesis present trends of the linear fit. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 1
Historical and recent average rates of sea level rise (mm yr^{-1}) in South Florida. Global value from Masson-Delmotte et al. (2021). Error expressed as uncertainty level.

Station	Historical	Recent	
	1900–2021	2000–2021	2010–2021
Naples	3.07 ± 0.1	6.95 ± 1.0	11.85 ± 4.3
Key West	2.52 ± 0.1	6.58 ± 1.1	9.72 ± 4.4
Vaca Key	4.06 ± 0.1	6.05 ± 1.0	7.34 ± 4.2
Virginia Key	5.76 ± 0.4	6.30 ± 1.2	8.73 ± 4.1
Average	3.85 ± 1.2	6.47 ± 0.3	9.41 ± 1.6
Global	1.3 ± 0.8	3.7 ± 0.5	3.7 ± 0.5

subjected to multidecadal variations in elevation associated with the Florida Current (e.g., Atlantic Overturning Circulation, see Ezer, 2013) to which all (n = 43) but the Southeast Saline Everglades (SAR locations = 4), Northeast Florida Bay (SAR locations = 3), and Lower Florida Keys (SAR locations = 5) coastal wetlands are not directly subjected.

3.2. Sediment accumulation rates

A total of 270 SAR values were obtained from 10 published South Florida studies (Fig. 1, Table 2). The complete dataset is available in Parkinson (2022). Ninety-five percent of the data were acquired in mangrove habitat. Seventeen *core lump sum* SAR values were obtained from the region. Eighty-seven percent (n = 26) of these data were obtained from mangrove habitat. The other values (n = 4) where acquired in the wet prairies of the Southwest Saline Everglades. The average lump sum SAR for South Florida mangroves was 2.4 mm yr^{-1} (S.D. 0.6) from samples obtained from Southwest Everglades National Park, 2.4 mm yr^{-1} (S.D. 2.4) from the Ten Thousand Islands, and 1.7 mm yr^{-1} (S.D. 0.5) in Rookery Bay. The average SAR for the wet prairies of the Southeast Saline Everglades was 1.4 mm yr^{-1} (S.D. 0.3).

Two hundred and twenty-five *core section interval* SAR values were obtained from previously published investigations. One hundred percent of these data were obtained from mangrove habitat. The core section interval data were organized into two groups based upon geographic location: Lower Florida Keys and Southwest Everglades National Park. Historical (e.g., 1900–2021) mangrove SAR in the Lower Florida Keys (n = 52) average 2.3 mm yr^{-1} (S.D. = 0.1) and range from 0.8 mm yr^{-1} (1905) to 5.6 mm yr^{-1} (2014). Recent (e.g., 2000–2021) mangrove core section interval SAR in the Lower Florida Keys (n = 9) average 3.7 mm yr^{-1} (S.D. 0.5) and range from 2.7 mm yr^{-1} to 5.6 mm yr^{-1}. Historical mangrove core section interval SAR in Southwest Everglades National Park (n = 173) average 3.7 mm yr^{-1} (S.D. = 1.9) and range from 0.4 mm yr^{-1} (1909) to 11.6 mm yr^{-1} (2006). Recent mangrove core section interval SAR in the Southwest Everglades National Park (n = 35) average 5.8 mm yr^{-1} (S.D. 2.1) and range between 2.7 mm yr^{-1} and 11.6 mm yr^{-1}. When the core section interval SAR data derived from mangrove habitat in the Lower Florida Keys and Southwest Everglades National Park were evaluated on an interannual timescale, SAR at both locations increased towards the present (Fig. 3), with the SAR between 1900 and 1960 lower than the Key West historical rate of SLR (2.5 mm yr^{-1}) in the Lower Florida Keys (mean = 1.4 mm yr^{-1}, S.D. = 0.4) and SWENP (mean = 1.8 mm yr^{-1}, S.D. = 0.6). Thereafter (1961–2021), SAR at both locations exceeded historical rates of SLR (Lower Florida Keys mean 2.9 mm yr^{-1}, S.D. = 0.8; Southwest Everglades National Park mean 4.4 mm yr^{-1}, S.D. = 1.7).

Fifteen SAR values were obtained from the *SETMH* studies conducted in South Florida (Fig. 4, Table 2). The period of observation for the data collected in Northeast Florida Bay and SWENP extended from 1998 to 2008 (Koch et al., 2015) and 2002–2018 (Feher et al., 2020), while the period of data collection in the Ten Thousand Islands National Wildlife Refuge began in 2012 and ended in 2017 (Howard et al., 2020). The Rookery Bay National Estuary Research Reserve SETMH data (Cahoon and Lynch, 1997) were acquired between 1994 and 1996. For this study, all SETMH data were considered representative of recent time. Seventy-four percent (n = 11) of the data were acquired in mangrove habitat. The average mangrove SAR (Table 2) ranged from 0.2 mm yr^{-1} (Lower Florida Keys) to 6.2 mm yr^{-1} (Ten Thousand Islands). The SAR of the

Table 2
Summary of sediment accumulation rate data used in this study organized by source of record, location, and habitat type.

Accumulation rate (mm yr^{-1})					Historical				Recent				Source
Source of record	Location	Habitat type	n	Method	Min.	Mean	Std. dev.	Max.	Min.	Mean	Std. dev.	Max.	
Lump sum	SESE	Wet prairie	4	Pb^{210}	1.0	1.4	0.3	1.6	–	–	–	–	Meeder et al., 2017
	SWENP	Mangrove	13	Pb^{210}	1.5	2.4	0.6	3.4	–	–	–	–	Breithaupt et al., 2017
	TTI	Mangrove	5	Pb^{210}	1.9	2.4	0.5	3.2	–	–	–	–	Breithaupt et al., 2017
	RB	Mangrove	8	Pb^{210}/ Cs^{137}	1.4	1.7	0.2	2.0	–	–	–	–	Lynch et al., 1989
Core section interval	LFK	Mangrove	52	Pb^{210}	0.8	2.3	0.1	5.6	–	–	–	–	Chappel, 2018
			9	Pb^{210}	–	–	–	–	2.7	3.7	0.5	5.6	Chappel, 2018
		1900–1960	20	Pb^{210}	–	1.4	0.4	–	–	–	–	–	Chappel, 2018
		1961–2021	32	Pb^{210}	–	2.9	0.8	–	–	–	–	–	Chappel, 2018
	SWENP	Mangrove	173	Pb^{210}	0.4	3.7	1.9	11.6	–	–	–	–	Breithaupt et al., 2014, 2017; Smoak et al., 2013
			35	Pb^{210}	–	–	–	–	2.7	5.8	2.1	11.6	Breithaupt et al., 2014, 2017; Smoak et al., 2013
		1900–1960	47	Pb^{210}	–	1.8	0.6	–	–	–	–	–	Breithaupt et al., 2014, 2017; Smoak et al., 2013
		1961–2021	92	Pb^{210}	–	4.4	1.7	–	–	–	–	–	Breithaupt et al., 2014, 2017; Smoak et al., 2013
SETMH	NEFLB	Mangrove	1	–	–	–	–	–	–	1.3	–	–	Koch et al., 2015
		Mangrove scrub	2	–	–	–	–	–	1.8	2.9	1.6	4.0	Koch et al., 2015
	SWENP	Mangrove	3	–	–	–	–	–	1.4	2.8	1.4	4.2	Feher et al., 2020; Koch et al., 2015
	LFK	Mangrove	2	–	–	–	–	–	−0.7	0.2	1.2	1.1	Chappel, 2018
	TTI	Mangrove	1	–	–	–	–	–	–	6.2	–	–	Howard et al., 2020
	TTI	Salt marsh	1	–	–	–	–	–	–	2.9	–	–	Howard et al., 2020
	TTI	Brackish	1	–	–	–	–	–	–	2.9	–	–	Howard et al., 2020
	RB	Mangrove	4	–	–	–	–	–	4.6	6.1	1.3	7.8	Cahoon and Lynch, 1997

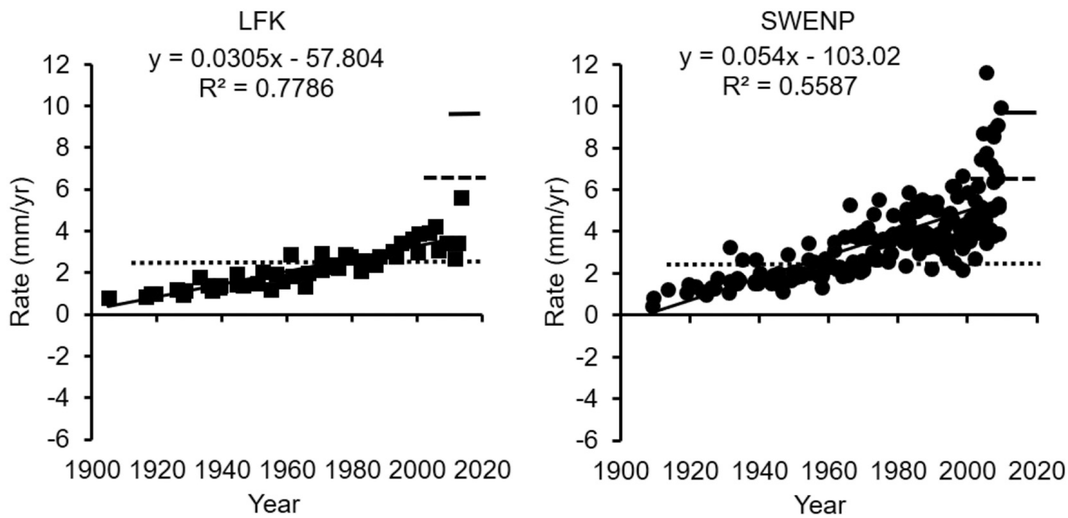

Fig. 3. Core section interval sediment accumulation rate data and trendlines obtained from mangrove plant communities located in the Lower Florida Keys and Southwest Everglades National Park. This type of data was not available for other study locations or plant community types. Horizontal dotted line = Key West historical (1913–2021) rate of SLR (2.5 mm yr^{-1}). Horizontal dashed line = Key West recent (2020−2021) rate of SLR (6.6 mm yr^{-1}). Horizontal line = Key West past decade (2010–2021) rate of SLR (9.7 mm yr^{-1}).
Source: Breithaupt et al. (2014, 2017), Chappel (2018), Smoak et al. (2013).

three other plant communities is based upon a very limited data set and reported as 2.9 mm yr^{-1} (S.D. = 1.6) for mangrove scrub (n = 2), 2.9 mm yr^{-1} for salt (n = 1) and brackish marsh (n = 1).

3.3. *Coastal wetland resilience*

The original intent of this investigation was to evaluate the response and resiliency of all South Florida coastal wetland plant community types to future conditions of accelerating SLR. However, as the investigation progressed and as has been described in the preceding sections, it is evident that the available SAR data sets for all coastal wetland plant communities except mangrove (e.g., mangrove scrub, salt and brackish marsh, wet prairie) are too limited in number and geographic location to warrant modeling their resiliency to SLR (Fig. 4). As a result, the modeling of resiliency was constrained to mangrove plant communities. The paucity of SAR values for South Florida coastal wetlands other than mangrove has not been previously reported and is a significant finding because it revealed a substantial knowledge gap. This gap will have to be closed before predictions of their resiliency under conditions of accelerating SLR can be attempted.

3.3.1. *Historical*

Based upon a comparison between the Key West historical (1900–2021) rate of SLR and lump sum mangrove SAR (Fig. 5a), instability was hindcast for Southwest Everglades National Park, the Ten Thousand Islands, and Rookery Bay. When the mean SAR of each of the mangrove core section interval datasets was used, historical instability was hindcast for the Lower Florida Keys and stability in Southwest Everglades National Park because the mean SAR was equal to or greater than the historical rate of SLR (Fig. 6a). A comparison between the interannual core section interval SAR data and the historical rate of SLR (Fig. 3) indicates a sediment deficit until about 1960 at both locations. Thereafter, rates of sedimentation steadily increased resulting in a hindcast of stability during the latter half of the 20th century. A comparison between the interannual core section interval SAR data and contemporaneous rates of historical SLR (e.g., 2.1 between 1900 and 1960, 3.7 between 1961 and 2021) (Fig. 6b) suggests mangrove habitats at both locations were unstable prior to mid-century. Thereafter, the Southwest Everglades National Park mangrove was hindcast to have stabilized as SAR was on average larger than the average rate of SLR during that interval of time. The mangrove SAR in the Lower Florida Keys did not exceed SLR between 1961–present, so instability is hindcast. The SETMH SAR data (Fig. 5b) were not compared to historical SLR because the sediment data were collected during recent time.

3.3.2. *Recent*

Rates of Key West SLR during the 21st century (6.6 mm yr^{-1}, 2000–2021; 9.7 mm yr^{-1}, 2010–2021) were much faster than all mangrove SAR data (Fig. 4), yielding a hindcast of instability in all South Florida (Table 3). Again, a transition to mud flats, inundation ponds or subtidal estuarine environments is hindcast.

3.3.3. *Future*

The recent rate of SLR during the 21st century is faster than *all* recent mangrove SAR data evaluated in this study, except for eleven core section interval values that range between 6.7 and 11.6 mm yr^{-1} obtained from the Southwest Everglades National Park (Fig. 3) and one SETMH SAR value from Rookery Bay mangrove (7.8 mm yr^{-1}) (Table 2). Only two Southwest Everglades National Park mangrove core section interval samples generated a SAR faster than the rate of recent SLR over the past decade (Fig. 3). Breithaupt et al. (2020) have argued these faster rates of SAR are indicative of an acceleration in organic sediment production that has occurred in response to the recent acceleration in the rate of SLR. Assuming this hypothesis is correct, the forecast of mangrove stability to SLR in all of South Florida was based upon the average rate of recent core section interval SAR data (5.8 mm yr^{-1}) (Table 2) obtained from Southwest Everglades National Park mangrove forests (n = 35). The outcome is therefore a best-case scenario. Future rates of SLR were based upon the intermediate global SLR scenario for the United States between 2000 and 2100 (Sweet et al., 2017) (Fig. 7). This SLR scenario was selected because its historical and recent rates of rise are the same as those observed in South Florida.

Results (Fig. 4, Table 3) indicate mangrove plant communities in South Florida will begin converting to intertidal and subtidal environments around 2040–2050 (Fig. 7). This will initially result in increase in the extent and rate of localized plant community destabilization and a transition to mudflats or the formation of inundation ponds, the latter of which is already occurring (Lagomasino et al., 2021; Meeder and Parkinson, 2018). By the end of this century, most mangrove forested areas will be submerged and replaced by an estuarine environment because at

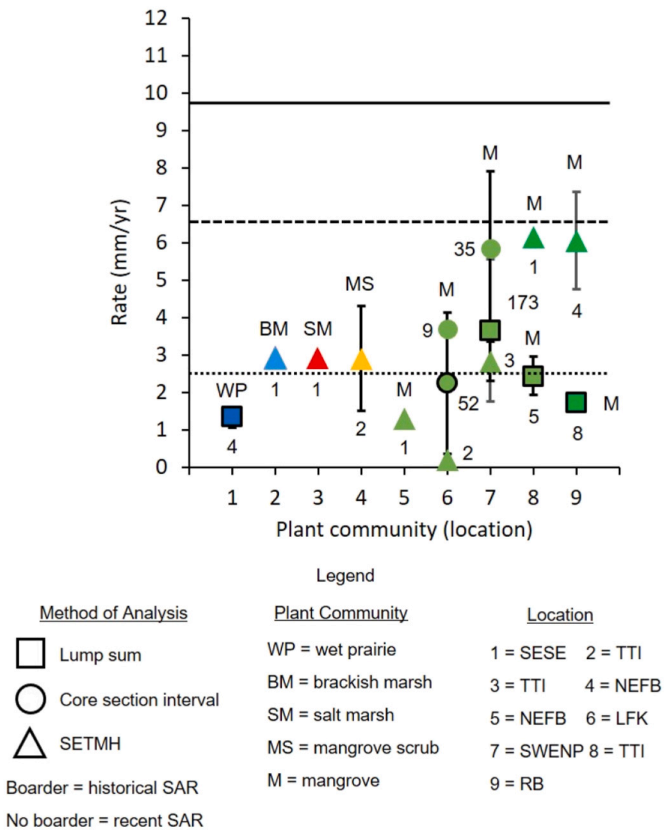

Fig. 4. Mean sediment accumulation rates plotted as a function of (1) coastal wetland plant community type, (2) method of analysis (e.g., lump sum and core section interval radioisotope geochronologies [Pb210, Cs137], Sediment Elevation Table Marker Horizon [SETMH]), and (3) timeframe (historical, 1900–2021; recent, 2000–2021; and past decade, 2010–2021). Horizontal dotted line = Key West historical (1913–2021) rate of SLR (2.5 mm yr^{-1}). Horizontal dashed line = Key West recent (2000–2021) rate of SLR (6.6 mm yr^{-1}). Horizontal solid line = Key West past decade (2010–2021) rate of SLR (9.7 mm yr^{-1}). Error bars = 1 S.D. N value associated with each series is number of sediment samples used to calculate mean sediment accumulation rate.

that time the elevation of mean-low water will have exceeded grade level (Fig. 7). Although the data is limited for the other four coastal wetland plant communities (Fig. 4, Table 3), their loss appears equally likely because the SAR in these areas is much lower than the more seaward mangroves.

4. Discussion

The results of this investigation clearly demonstrate predictions of coastal wetland response to historical, recent, and future rates of SLR are dependent upon the methodology utilized to enumerate rates of sediment accumulation and timescale considered. However, hindcasts of the instability of mangrove plant communities in South Florida in response to historical SLR were not always consistent with published studies (Howard et al., 2020; Krauss et al., 2011; Lagomasino et al., 2021; Meeder et al., 2017; Ross et al., 2021). The hindcasts of historical instability were generated from mangrove lump sum and core section interval SAR data. This contradiction appears to be an outcome of the assumption that SAR values are *always* representative of in situ conditions at the time of deposition. The core section interval data aggregated during this investigation indicate the contrary, an observation that has not been previously reported. The observed historical trends in the rate and dispersion of Southwest Everglades National Park and Lower Florida Keys mangrove SAR values (Fig. 3) are consistent with the (1) cumulative effects of post-depositional compaction and decomposition that reduce the thickness of the peat interval and corresponding estimates of SAR and (2) inverse relationship between SAR variance and time (c.f., Hatton et al., 1983). The effects of compaction on a variety of coastal wetland habitat types have been documented in several South Florida studies (Feher et al., 2020; Howard et al., 2020; Koch et al., 2015). They report an average subsidence in recent mangrove soils of ~2 mm yr^{-1}, although faster rates have been reported (Osland et al., 2020) (Table 4). This would account for the apparent *historical* sediment deficit reported in mangrove soils of South Florida during the first half of the 20th century (Fig. 3). If, as suggested by these data, there is a continuous reduction in the thickness of older sediments as a function of time, then an argument can be made that the recent increase in SAR values as reported by Breithaupt et al. (2020) is not a consequence of an acceleration in the rate of SLR, but rather the cumulative effects of shallow subsidence in older sediments (e.g., timescale bias). Resolution of this uncertainty will require the acquisition of additional and more recent (e.g., 2010 – present) SAR data to determine whether the observed increases in SAR have persisted in response to even faster rates of SLR over the last decade.

Forecasts of South Florida mangrove resilience to accelerating rates of SLR predict widespread instability and habitat loss by mid-century. This is consistent with the global projections of (Saintilan et al., 2020), but differs from Sklar et al. (2021), who projected a regional loss in South Florida mangroves by 2070, but only under conditions of extreme SLR (1.3 m, 22.6 mm yr^{-1}). This discrepancy is a consequence of their use of a faster SAR (11 mm yr^{-1}) than the 5.8 mm yr^{-1} SAR value used in this study. Sediment accumulation rates higher than 5.8 mm yr^{-1} have occurred (n = 11, Parkinson, 2022), but are outliers relative to the other 259 mangrove SAR values aggregated herein. By the end of this century, the widespread and rapid collapse of mangrove habitats in South Florida coastal and their conversion to estuarine conditions is expected. Because SAR in the more landward brackish and freshwater plant communities is lower than mangrove, they too will likely experience conversion to a more salt-tolerant taxa and/or submergence at an increasing rate over the duration of this century. The forecast of mangrove resiliency is considered to be a best-case scenario for several reasons. First, the current and projected emissions gap (United Nations Environmental Program, 2021) will exceed the conditions associated with the intermediate SLR scenario used in this study and will result in even faster rates of SLR than those used in this study, as is already being considered by others (Thompson et al., 2021). Second, the forecast assumes the recent and faster rates of mangrove SAR documented between 2000 and 2010 can be sustained over time through peat production and accumulation. There is at present no data to indicate this is occurring. Third, the forecast does not consider the cumulative effects of post-depositional process (e.g., compaction, decomposition) that yield a decrease in substrate elevation or shallow subsidence as has been reported in mangrove, scrub mangrove, salt marsh, and brackish marsh communities throughout South Florida (Feher et al., 2020; Howard et al., 2020; Koch et al., 2015). With time this will reduce resilience by lowering elevation and increasing the relative rate of SLR.

It could be argued that this model is flawed because it either over simplifies or ignores post-depositional above and below-ground processes that, if considered, would substantially alter the outcome (Table 4). For example, an increase in the frequency and duration of soil saturation may reduce the rate of organic matter decomposition (Chambers et al., 2021) and promote soil profile swelling (Feher et al., 2020; Whelan et al., 2005), both of which would enhance resiliency. However, these responses would likely be of limited duration and consequence because, for example, once the soils are permanently saturated with sea- or freshwater, swelling (and shrinking) will be of limited consequence to the elevation of evolving soil profile. And there are numerous other processes (e.g., mechanical compaction/shallow subsidence, elevated frequency of storm events and associated peat collapse and ponding) that will over time exert a substantially greater

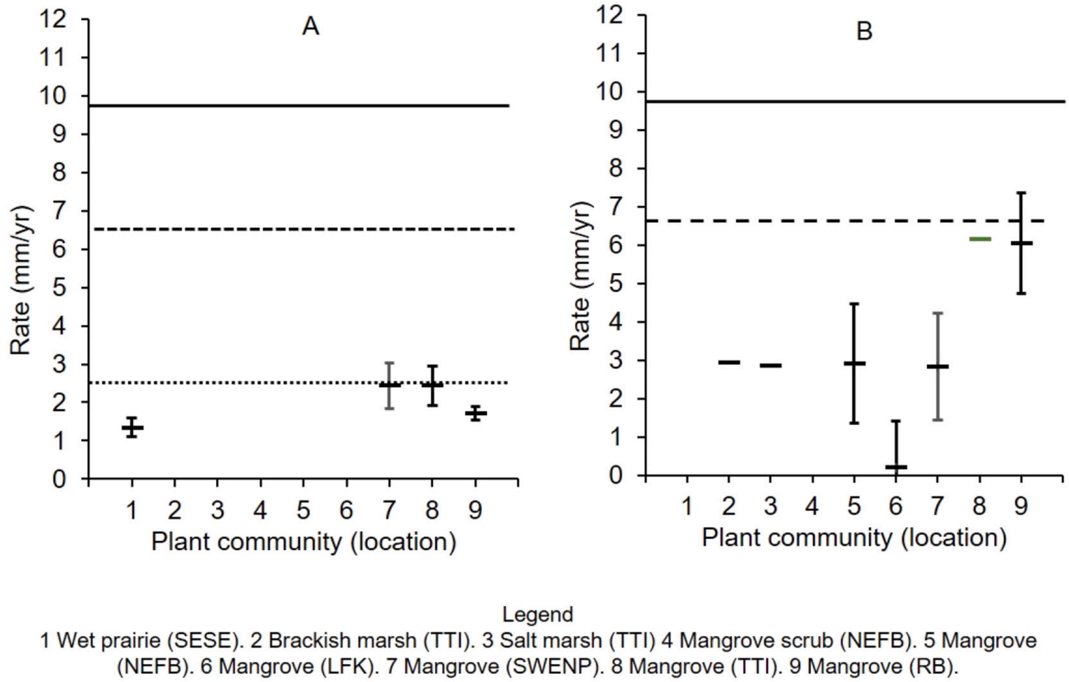

Fig. 5. (A) Mean sediment accumulation rates based upon core lump sum data grouped by location and habitat type. (B) Mean SETMH sediment accumulation data plotted as a function of location and habitat type. Horizontal dotted line = Key West historical (1913–2021) rate of SLR. Horizontal dotted line = Key West historical (1913–2021) rate of SLR (2.5 mm yr^{-1}). Horizontal dashed line = Key West recent (2020–2021) rate of SLR (6.6 mm yr^{-1}). Horizontal line = Key West past decade (2010–2021) rate of SLR (9.7 mm yr^{-1}). Error bars are ±1 S.D. if sample n > 1. Lump sum source: Breithaupt et al. (2014, 2017); Chappel (2018); Lynch et al. (1989); Meeder et al. (2017); Smoak et al. (2013).
SETMH source: Cahoon and Lynch (1997), Chappel (2018), Feher et al. (2020), Howard et al. (2020), Koch et al. (2015).

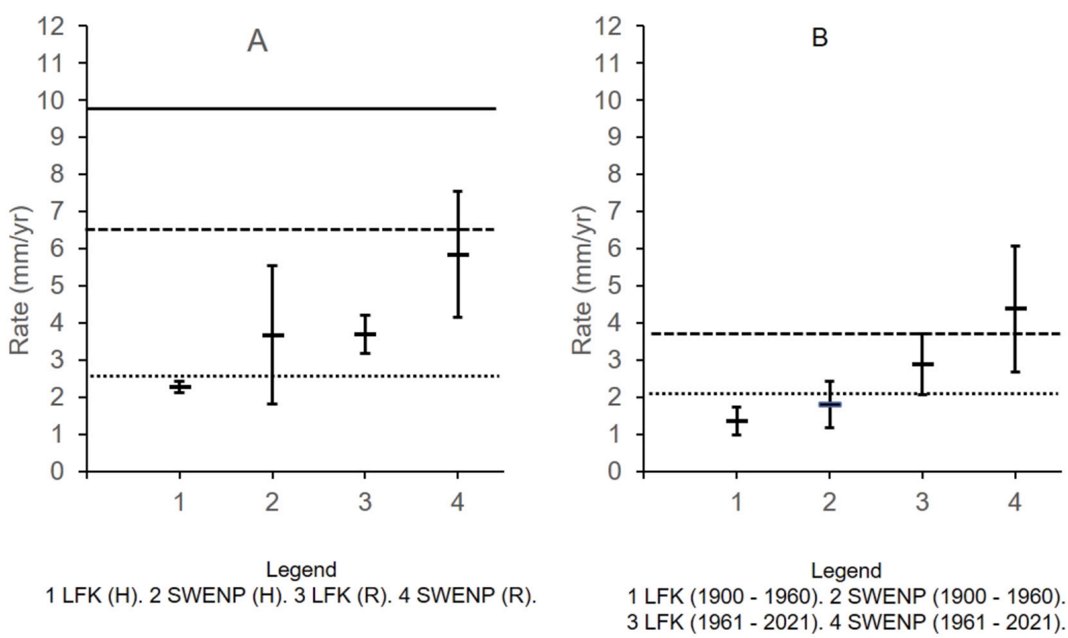

Fig. 6. Core section interval sediment accumulation rates for mangrove habitats in the LFK and SWENP. (A) grouped as historical (H; mean value of all data between 1900 and 2021) and recent (R; mean value of all data between 2000 and 2021). Horizontal dotted line = Key West historical (1913–2021) rate of SLR (2.5 mm yr^{-1}). Horizontal dashed line = Key West recent (2020–2021) rate of SLR (6.6 mm yr^{-1}). Horizontal line = Key West past decade (2010–2021) rate of SLR (9.7 mm yr^{-1}). (B) grouped into two timeframes based upon the intersection of the average historical rate of SLR and SAR as illustrated in Fig. 3. Horizontal dotted (1900–1960) and dashed (1961–2021) lines are contemporaneous rates of SLR at Key West (2.1 mm yr^{-1}, 3.7 mm yr^{-1}, respectively). Error bars are ±1 S.D. if sample n > 1. Sediment accumulation rate source: Breithaupt et al. (2014, 2017), Chappel (2018), Smoak et al. (2013).

deleterious effect on mangrove plant community elevation and resiliency. To the contrary, it has been suggested that hurricane landfall events promote resiliency by providing a significant influx of sediment and nutrients (Chambers et al., 2021; Feher et al., 2020; Smoak et al., 2013; Whelan et al., 2009). Yet others have reported that storms have little impact on long-term elevation dynamics (Rogers et al., 2013) or

Table 3
Summary of South Florida wetland response to historical, recent, and predicted rates of SLR. Cells with boarders indicate wetlands predicted to be stable (e.g., SAR ≥ SLR). Unstable conditions occur when SAR < SLR.

	Sea-Level Rise (mm yr^{-1})			
	Historical		Recent	Future
	Average	Contemporaneous		
	2.5	1900-1960 (2.1)	2000 - 2021 (6.6)	>6.6
Method of Analysis		1961-2021 (3.7)	2010 - 2021 (9.7)	
Lump Sum Historical				
SESE wet prairie	1.4	-	Unstable	Unstable
SWENP mangrove	2.2	-	Unstable	Unstable
TTI mangrove	2.4	-	Unstable	Unstable
RB mangrove	1.7	-	Unstable	Unstable
Core Section Interval				
Historical (mean)				
LFK mangrove	2.3	-	Unstable	Unstable
SWENP mangrove	3.7	-	Unstable	Unstable
Recent				
LFK mangrove	-	-	3.7	Unstable
SWENP mangrove	-	-	5.8	Unstable
Interannual				
1900-1960				
LFK mangrove	1.4	1.4	-	-
SWENP mangrove	1.8	1.8	-	-
1961-2021				
LFK mangrove	2.8	2.8	-	-
SWENP mangrove	4.4	4.4	-	-
SETMH				
NEFB mangrove	-	-	1.3	Unstable
NEFB scrub mangrove	-	-	2.9	Unstable
LFK mangrove	-	-	0.2	Unstable
SWENP mangrove	-	-	1.0	Unstable
TTI mangrove	-	-	6.2	Unstable
TTI salt marsh	-	-	2.9	Unstable
TTI brackish marsh	-	-	2.9	Unstable
RB mangrove	-	-	6.1	Unstable

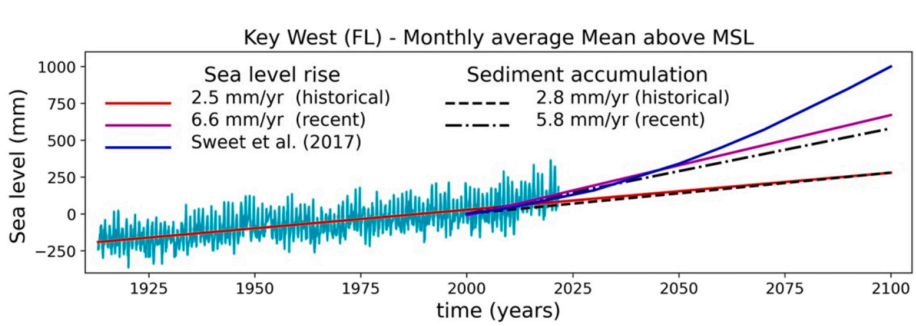

Fig. 7. Left side: Historical and recent trends in Key West sea level elevation. Right side: Predicted Intermediate sea-level rise scenario and mangrove sediment accumulation rates based upon historical average derived from all study sites (2.8 mm yr^{-1}) and recent average (5.8 mm yr^{-1}) generated from Southwest Everglades National Park recent core interval data. Cyan line marks the Key West mean monthly value (1913–2021). These projections assume ground level is at zero mean sea level in year 2000. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 4
Summary of commonly reported above and below ground post-depositional processes that contribute to substate elevation change in South Florida mangrove plant communities. Positive results contribute towards elevation change and/or an increase in plant productivity. Superscripts identify the source of the data or examples thereof. Magnitude of change is an estimate of the vertical or horizontal scale at which the process operates.

Process	Result			Comment	Reference
	Positive	Negative	Magnitude of change		
Decomposition	x[a]		?	Decreases with increasing duration of inundation	Chambers et al., 2021
		x[a]	?	Increases with time, decreases with depth	Romero et al., 2005
Hydrology	x[a]		mm to cm	Profile swells (e.g., expansion, dilation) when saturated (e.g., astronomical tides, wind, storm surge, groundwater)	Feher et al., 2020; Whelan et al., 2005, 2009
		x[a]	mm to cm	Profile shrinks (e.g., compaction, consolidation) with falling tides, water table (e.g., draw- or dry-down), and evapotranspiration	Feher et al., 2020; Whelan et al., 2005, 2009
Peat collapse and ponding		x[a,b]	Local to regional	Indicative of cumulative environmental stress (e.g., dry-down, increasing duration of inundation) or storm events	Chambers et al., 2019; Lagomasino et al., 2021
Storms	x[a]		cm^{-event}	Above ground sediment deposition, nutrient influx	Castaneda-Moya et al., 2020; Feher et al., 2020; Whelan et al., 2009
		x[a]	cm to dm^{-event}	Erosion, mass tree mortality, especially in poorly drained soils	Lagomasino et al., 2021; Osland et al., 2020
		x[a,c]	mm^{-event}	Surface compressed by water column loading, substrate erosion	Feher et al., 2020; Whelan et al., 2009
Mechanical or auto-compaction, contraction, shallow subsidence		x[a]	mm to cm yr^{-1}	Proportional to thickness of overburden	Cahoon and Lynch, 1997; Feher et al., 2020; Osland et al., 2020; Rogers and Saintilan, 2021

[a] Data generated by the author(s).
[b] Literature cited by the author(s).
[c] Observational statement of author(s).

documented their role as a process that results in a significant reduction in forest productivity and substrate elevation (Griffiths and Mitsch, 2021; Lagomasino et al., 2021; Osland et al., 2020). To date, however, no studies have quantified the spatial scale of storm layer deposition; rather the few values that have been reported are from long-term ecological research monitoring stations located no more than 10 km inland from the southwest Everglades National Park shoreline (e.g., Harney River, Lostmans River, Shark River) (Castaneda-Moya et al., 2020; Whelan et al., 2009). Nor has it been determined whether the constructive consequences of those nutrient and elevation gains are enough to offset the loss of productive habitat caused by substrate deflation, tree falls, peat collapse, and ponding. This seems unlikely because the few hurricane-derived sediment deposits that have been described occur within no more than a kilometer of the shoreline or river channel from which the material was derived. In contrast, the destructive effects (e.g., loss of canopy height, defoliation, mortality) have been shown to exert an impact measured in thousands of hectares (Lagomasino et al., 2021). Furthermore, the rate of forest recovery was slowest in poorly drained soils, the extent of which will increase under conditions of SLR. Secondly, the trend analysis of Feher et al. (2020, c.f. Fig. 4) indicates the elevation gains are short term and do not substantially alter their long term trajectory. The long-term legacy of South Florida storm deposits is doubtful because they are composed of 'marine carbonate sediment' (Feher et al., 2020) Given the pH of mangrove peats is generally low (Pazi et al., 2016; Reef et al., 2010), it is doubtful these carbonate intervals will persist over time. This supposition is supported by the fact that none of the published storm landfall investigations that utilized sediment cores (Breithaupt et al., 2019; Smoak et al., 2013) reported the presence of any storm layers other than those associated with the recent landfall events (e.g., Wilma, Irma) they were investigating. Core lengths ranged between 25 cm and 50 cm, so the duration of time represented by the recovered sediment succession is more than 100 yrs., during which numerous other landfall events have occurred in South Florida (e.g., Great Miami Hurricane, 1926; Labor Day Hurricane, 1935; Hurricane Donna, 1960, Betsy, 1965). An absence of older storm layers is consistent with carbonate dissolution, which then neutralizes the short-term elevation gains associated with event landfall.

Although the beneficial effects of an expanding horizontal accommodation space were not considered in this study, it would be of limited consequence because the continuity of the landscape has already been disrupted by the presence of coast parallel roads and flood control structures (Lagomasino et al., 2021; Meeder et al., 2021). The only exception is the Shark River Slough (Fig. 1), where there are no obstructions proximal (<40 km) to the coastline. But even if it is assumed the plant communities will find a way past these topographic barriers, the very low topographic gradient of the south Florida coastal zone (~5 cm km^{-1}) (c.f. Parkinson et al., 2015, Fig. 2) will promote the landward *translation* of the shoreline at a theoretical rate of about 200 m yr^{-1} coincident with a *sustained* SLR of 10 mm yr^{-1}. Rates of SLR nearly twice as fast are possible by the end of this century. This rate of translation is much faster than the rates of landward migration reported in South Florida mangrove plant communities (~30 m yr^{-1}) by Meeder and Parkinson (2018) in the SWSE and Krauss et al. (2011) in the TTI. Hence the space between the shoreline and advancing plant community will expand and deepen with time. The successful establishment of peat-producing mangrove plant communities in the newly created horizontal accommodation space is not therefore guaranteed. Moreover, the plant communities migrating into the newly created accommodation space will not be accumulating sediment at the rates measured in mature habitat (Sandi et al., 2021) and it is not currently known what duration of time is required to complete that transition in South Florida. In a southeastern Australian estuary, Rogers et al. (2013) observed relative water level within a mangrove encroachment zone increased while a decrease was evident in mature mangrove and salt marsh over a 10-yr period of observation. It follows that their resilience to SLR and saltwater encroachment will initially be lower than the mature South Florida coastal wetlands from which they are derived as has been described by Meeder and Parkinson (2018).

This analysis has revealed several knowledge gaps that must be filled to improve the spatial and temporal resolution of resiliency forecasts. The vast majority (97%) of SAR data collected in South Florida were obtained in mangrove habitat. However, other coastal wetland habitats occupy an equivalent amount of the landscape. It is recommended these transitional and freshwater wetlands be subjected to analyses that improve our understanding of the effects of SLR and saltwater encroachment on the production and accumulation of sediment.

Predictions could also be improved by undertaking additional studies designed to quantity post-depositional processes (e.g., compaction, decomposition) and their cumulative effect on shallow subsidence and SAR, as has been done by Feher et al. (2020) and Whelan et al. (2005, 2009) in the mangrove forests of the Shark River Slough (Fig. 1). The mangrove SAR data derived from recent Southwest Everglades National Park core section intervals appears favorable towards sustained resilience (see also Breithaupt et al., 2020) but are limited in geographic context and reported prior to the most recent decade of SLR acceleration. Hence it cannot be known whether the observed acceleration in SAR has persisted. Additional studies are therefore needed to document how coastal wetlands of all types respond to a sustained acceleration (e.g., decades) in the rate of SLR over the past decade. The potential benefits of horizontal accommodation space have repeatedly been cited a means by which coastal wetland resilience can be sustained despite accelerating SLR. However, it is not reasonable to assume the plant communities migrating into the newly created accommodation space are accumulating sediment at the same rates as their mature counterparts and we currently do not know the duration of time required to complete that transition. Additionally, in South Florida the utilization of this accommodation space by transgressing plant communities does not necessarily ensure ecosystem services are preserved because they are migrating into and replacing other systems that have historically been a significant element of the landscape. This may result in a loss of diversity, rare or endangered species, or an entire plant community (e.g., wet prairie) (Kelleway et al., 2017; Meeder, 2022; Osland et al., 2022). More research is needed in these transitional areas to quantify the processes and products of sedimentation, the rates at which different plant communities are capable of migrating landward, and the net effect on ecosystem services.

There are currently two mitigative strategies typically considered to offset the combined effects of SLR and saltwater encroachment on wetland resilience: (1) augmentation of SAR and (2) removal of topographic and hydrologic barriers to lateral migration and freshwater flow. Augmentation of SAR has been attempted through the addition of nutrients or thin-layer disposal. The addition of nutrients (e.g., N, P) is designed to overcome nitrogen and/or phosphorous limitations on plant growth (c.f., Feller et al., 2002). Experiments of nutrient enrichment have been limited in scale (e.g., plots) and duration (e.g., years), but shown to stimulate plant growth (Lovelock et al., 2004), above ground biomass, and surface elevation (Davis et al., 2017; Wilson et al., 2019). However, the benefits of increased growth may be offset by higher mortality rates during prevailing (Lovelock et al., 2009) and storm conditions (Feller et al., 2015), as well as changes in ecosystem structure and soil nutrients dynamics (Romero et al., 2012). Thin-layer placement describes the addition of a sediment-water slurry onto a deteriorating wetland to enhance resilience via elevation gain. It is now widely utilized by the U.S Army Corps of Engineers (USACE) in coastal areas throughout the conterminous U.S. (Berkowitz et al., 2017). To date, it has not been attempted in South Florida although the USACE has recently (2021) proposed a proof-of-concept physical model to be tested in the Southeast Saline Everglades (Ralph, 2021). Should the model be deemed effective, the next step will likely be the implementation of a thin-layer placement project on a larger scale. However, there are many challenges yet to be overcome including the permitting of a project of this type, locating a suitable source and volume of compatible sediment, determining how that material will be transferred and applied to the project area, and whether the project could be completed at the scale and timeframe necessary to make a significant difference.

So at present, the only practical mitigative strategy to enhance the resilience of South Florida coastal wetlands to accelerating SLR is the removal of barriers and restoration of surface water flow, which is at the core of the Comprehensive Everglades Restoration Program (CERP) authorized by the U.S. Congress in 2000 (Sheikh and Carter, 2005). The intended outcome of this effort is to ameliorate (1) saltwater encroachment and (2) historical losses of freshwater wetlands. The strategy is currently being implemented by the Southwest Florida Water Management District, which recently replaced 10 km of roadbed with an elevated roadway at the headwaters of Everglades National Park. Numerous other projects to restore freshwater flow are either under construction or are being planned (Conservancy of Southwest Florida, 2017). It remains uncertain whether these projects can be completed at the requisite scale and timeframe (c.f., Lee et al., 2021; Meeder et al., 2017) necessary to slow or reverse saltwater encroachment, stimulate the restoration of historical ecosystem function, and stabilize the remaining wetland landscape as intended.

5. Conclusion

Over the last 20 years, the *average* rate of SLR in South Florida has increased from 6.5 mm yr^{-1} (2000–2021) to 9.4 mm yr^{-1} (2010–2021), compared to the historical rate of 3.9 mm yr^{-1}. These rates of rise and the accompanying saltwater encroachment have degraded the resilience of existing South Florida mangrove plant communities. Intermediate SLR scenarios predict even faster rates towards the end of this century, which will accelerate instability, degradation, and habitat loss by mid-century. The total collapse of existing mangrove forests is probable by the end of this century. A similar response in the more interior coastal wetland habitats is likely, although confidence in these projections is limited due to the lack of SAR data generated from these plant communities. The removal of topographic barriers and restoration of freshwater flow may slow this transition, as recent modeling as suggested (Dessu et al., 2021). However, it is currently unclear whether these mitigative strategies can be completed at the scale and tempo necessary to significantly enhance the resilience of South Florida coastal wetlands to accelerating SLR.

This investigation revealed several knowledge gaps that, if closed, would significantly improve resilience forecasts in South Florida. These include undertaking investigations designed to quantify: (1) the effects of SLR and saltwater encroachment on the production and accumulation of sediment in transitional and freshwater wetlands, (2) post-depositional processes (e.g., compaction) and the cumulative effect of shallow (~1 m) subsidence on SAR, and (3) how coastal wetlands of all types have responded to a sustained acceleration in the rate of SLR over the last decade.

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.

Acknowledgments

The authors appreciate the comments on multiple drafts provided by John (Jack) F. Meeder and the initial draft by Michael Ross (FIU) that resulted in substantial improvements to the manuscript. The manuscript benefited significantly from the comments and suggestions of five anonymous journal reviewers. We are thankful to Kristen Vanderbilt who facilitated making the data available through the FCE LTER Environmental Data Initiative's archive repository. Mike Rugge (Florida Coastal Everglades LTER Program Manager), René M. Price (FIU), and Mark Zucker (USGS-CFWSC-Davie, Florida) are acknowledged for facilitating access to South Florida water level data. This study was supported by NSF grant #DEB-1832229 and NASA grant #80NSSC21K0982 to SW. This is contribution #1453 from the Institute of Environment at Florida international University.

References

Arias-Ortiz, A., Masqué, P., Garcia-Orellana, J., Serrano, O., Mazarrasa, I., Marbà, N., Lovelock, C.E., Lavery, P., Duarte, C.M., 2018. Reviews and syntheses: 210Pb-

derived sediment and carbon accumulation rates in vegetated coastal ecosystems: setting the record straight. Biogeosci. Discuss. 1–47 https://doi.org/10.5194/bg-2018-78.
Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C., Silliman, B.R., 2011. The value of estuarine and coastal ecosystem services. Ecol. Monogr. 81, 169–193. https://doi.org/10.1890/10-1510.1.
Berkowitz, J., Piercy, C., Welp, T., 2017. Thin Layer Placement: State of the Science. Presented at the RSM Interagency Workshop. USACE, Jacksonville, Florida.
Boumans, R.M.J., Day, J.W., 1993. High precision measurements of sediment elevation in shallow coastal areas using a sedimentation-erosion table. Estuaries 16, 375–380. https://doi.org/10.2307/1352509.
Breda, A., Saco, P.M., Sandi, S.G., Saintilan, N., Riccardi, G., Rodríguez, J.F., 2021. Accretion, retreat and transgression of coastal wetlands experiencing sea-level rise. Hydrol. Earth Syst. Sci. 25, 769–786. https://doi.org/10.5194/hess-25-769-2021.
Breithaupt, J.L., Smoak, J.M., Smith III, T.J., Sanders, C.J., 2014. Temporal variability of carbon and nutrient burial, sediment accretion, and mass accumulation over the past century in a carbonate platform mangrove forest of the Florida Everglades. J. Geophys. Res. Biogeosci. 119, 2032–2048. https://doi.org/10.1002/2014JG002715.
Breithaupt, J.L., Smoak, J.M., Rivera-Monroy, V.H., Castañeda-Moya, E., Moyer, R.P., Simard, M., Sanders, C.J., 2017. Partitioning the relative contributions of organic matter and mineral sediment to accretion rates in carbonate platform mangrove soils. Mar. Geol. 390, 170–180. https://doi.org/10.1016/j.margeo.2017.07.002.
Breithaupt, J.L., Smoak, J.M., Byrne, R.H., Waters, M.N., Moyer, R.P., Sanders, C.J., 2018. Avoiding timescale bias in assessments of coastal wetland vertical change: vertical change in coastal wetlands. Limnol. Oceanogr. https://doi.org/10.1002/lno.10783.
Breithaupt, J.L., Hurst, N., Steinmuller, H.E., Duga, E., Smoak, J.M., Kominoski, J.S., Chambers, L.G., 2019. Comparing the biogeochemistry of storm surge sediments and pre-storm soils in coastal wetlands: Hurricane Irma and the Florida Everglades. Estuar. Coasts. https://doi.org/10.1007/s12237-019-00607-0.
Breithaupt, J.L., Smoak, J.M., Bianchi, T.S., Vaughn, D.R., Sanders, C.J., Radabaugh, K. R., Osland, M.J., Feher, L.C., Lynch, J.C., Cahoon, D.R., Anderson, G.H., Whelan, K. R.T., Rosenheim, B.E., Moyer, R.P., Chambers, L.G., 2020. Increasing rates of carbon burial in Southwest Florida Coastal Wetlands. J. Geophys. Res. Biogeosci. 125 https://doi.org/10.1029/2019JG005349.
Cahoon, D.R., Lynch, J.C., 1997. Vertical accretion and shallow subsidence in a mangrove forest of southwestern Florida, U.S.A. Mangrove Salt Marshes 1, 173–186. https://doi.org/10.1023/A:1009904816246.
Cahoon, D.R., Lynch, J.C., Perez, B.C., Segura, B., Holland, R.D., Stelly, C., Stephenson, G., Hensel, P., 2002. High-precision measurements of wetland sediment elevation: II. The rod surface elevation table. J. Sediment. Res. 72, 734–739.
Callaway, J.C., Cahoon, D.R., Lynch, J.C., DeLaune, R.D., Reddy, K.R., Richardson, C.J., Megonigal, J.P., 2013. The Surface Elevation Table–Marker Horizon Method for Measuring Wetland Accretion and Elevation Dynamics. In: SSSA Book Series. Soil Science Society of America. https://doi.org/10.2136/sssabookser10.c46.
Castaneda-Moya, E., Rivera-Monroy, V.H., Chambers, R.M., Zhao, X., Lamb-Wotton, L., Gorsky, A., Gaiser, E.E., Troxler, T.G., Kominoski, J.S., Hiatt, M., 2020. Hurricanes fertilize mangrove forests in the Gulf of Mexico (Florida Everglades, USA). Proc. Natl. Acad. Sci. U. S. A. 201908597 https://doi.org/10.1073/pnas.1908597117.
Chambers, L.G., Davis, S.E., Troxler, T.G., Entry, J.A., 2015. Sea level rise in the Everglades: plant-soil-microbial feedbacks in response to changing physical conditions. In: Microbiology of the Everglades Ecosystem. CRC Press, Boca Raton, Florida, pp. 89–112.
Chambers, L.G., Steinmuller, H.E., Breithaupt, J.L., 2019. Toward a mechanistic understanding of "peat collapse" and its potential contribution to coastal wetland loss. Ecology, e02720. https://doi.org/10.1002/ecy.2720.
Chambers, R.M., Gorsky, A.L., Castañeda-Moya, E., Rivera-Monroy, V.H., 2021. Evaluating a steady-state model of soil accretion in Everglades Mangroves (Florida, USA). Estuar. Coasts. https://doi.org/10.1007/s12237-020-00883-1.
Chappel, A.R., 2018. Soil Accumulation, Accretion, and Organic Carbon Burial Rates in Mangrove Soils of the Lower Florida Keys: A Temporal and Spatial Analysis, p. 176. USFSP Master's Theses (Graduate). https://digital.usfsp.edu/masterstheses/176.
Charles, S.P., Kominoski, J.S., Troxler, T.G., Gaiser, E.E., Servais, S., Wilson, B.J., Davis, S.E., Sklar, F.H., Coronado-Molina, C., Madden, C.J., Kelly, S., Rudnick, D.T., 2019. Experimental saltwater intrusion drives rapid soil elevation and carbon loss in freshwater and Brackish Everglades marshes. Estuar. Coasts 42, 1868–1881. https://doi.org/10.1007/s12237-019-00620-3.
Childers, D.L., Sklar, F.H., Drake, B., Jordan, T., 1993. Seasonal measurements of sediment elevation in three mid-Atlantic estuaries. J. Coast. Res. 986–1003.
Conservancy of Southwest Florida, 2017. First Year Post-Restoration Aquatic Fauna Monitoring in the Picayune Strand Restoration Project Area (2016-2017). Naples, Florida.
Craft, C.B., 2012. Tidal freshwater forest accretion does not keep pace with sea level rise. Glob. Chang. Biol. 18, 3615–3623. https://doi.org/10.1111/gcb.12009.
Craft, C., Clough, J., Ehman, J., Joye, S., Park, R., Pennings, S., Guo, H., Machmuller, M., 2009. Forecasting the effects of accelerated sea-level rise on tidal marsh ecosystem services. Front. Ecol. Environ. 7, 73–78. https://doi.org/10.1890/070219.
Davis, J., Currin, C., Morris, J.T., 2017. Impacts of fertilization and tidal inundation on elevation change in microtidal, low relief salt marshes. Estuar. Coasts 40, 1677–1687. https://doi.org/10.1007/s12237-017-0251-0.
Delaune, R.D., Patrick, W.H., Buresh, R.J., 1978. Sedimentation rates determined by 137Cs dating in a rapidly accreting salt marsh. Nature 275, 532–533. https://doi.org/10.1038/275532a0.
Dessu, S.B., Price, R.M., Troxler, T.G., Kominoski, J.S., 2018. Effects of sea-level rise and freshwater management on long-term water levels and water quality in the Florida Coastal Everglades. J. Environ. Manag. 211, 164–176. https://doi.org/10.1016/j.jenvman.2018.01.025.
Dessu, S.B., Paudel, R., Price, R.M., Davis, S.E., 2021. Using empirical data and modeled scenarios of Everglades restoration to understand changes in coastal vulnerability to sea level rise. Clim. Chang. 168, 19. https://doi.org/10.1007/s10584-021-03231-9.
Domingues, R., Goni, G., Baringer, M., Volkov, D., 2018. What caused the accelerated sea level changes along the U.S. East Coast during 2010–2015? Geophys. Res. Lett. 45, 13367–13376. https://doi.org/10.1029/2018GL081183.
Donoghue, J.F., 2011. Sea level history of the northern Gulf of Mexico coast and sea level rise scenarios for the near future. Clim. Chang. 107, 17–33. https://doi.org/10.1007/s10584-011-0077-x.
Doughty, C.L., Cavanaugh, K.C., Ambrose, R.F., Stein, E.D., 2019. Evaluating regional resiliency of coastal wetlands to sea level rise through hypsometry-based modeling. Glob. Chang. Biol. 25, 78–92. https://doi.org/10.1111/gcb.14429.
Egler, F.E., 1952. Southeast saline Everglades vegetation, Florida and its management. Vegetatio 3, 213–265.
Ezer, T., 2013. Sea level rise, spatially uneven and temporally unsteady: why the U.S. East Coast, the global tide gauge record, and the global altimeter data show different trends: SPATIAL AND TEMPORAL SEA LEVEL RISE. Geophys. Res. Lett. 40, 5439–5444. https://doi.org/10.1002/2013GL057952.
Ezer, T., 2019. Regional differences in sea level rise between the Mid-Atlantic Bight and the South Atlantic Bight: is the gulf stream to blame? Earth's Future 7, 771–783. https://doi.org/10.1029/2019EF001174.
Ezer, T., 2020. Analysis of the changing patterns of seasonal flooding along the U.S. East Coast. Ocean Dyn. 70, 241–255. https://doi.org/10.1007/s10236-019-01326-7.
Farron, S.J., Hughes, Z.J., FitzGerald, D.M., 2020. Assessing the response of the Great Marsh to sea-level rise: migration, submersion or survival. Mar. Geol. 425, 106195 https://doi.org/10.1016/j.margeo.2020.106195.
Feher, L.C., Osland, M.J., Anderson, G.H., Vervaeke, W.C., Krauss, K.W., Whelan, K.R.T., Balentine, K.M., Tiling-Range, G., Smith, T.J., Cahoon, D.R., 2020. The long-term effects of Hurricanes Wilma and Irma on soil elevation change in Everglades mangrove forests. Ecosystems 23, 917–931. https://doi.org/10.1007/s10021-019-00446-x.
Feller, I.C., Mckee, K.L., Whigham, D.F., O'Neill, J.P., 2002. Nitrogen vs. phosphorus limitation across an ecotonal gradient in a mangrove forest. Biogeochemistry 62, 145–175.
Feller, I.C., Dangremond, E.M., Devlin, D.J., Lovelock, C.E., Proffitt, C.E., Rodriguez, W., 2015. Nutrient enrichment intensifies hurricane impact in scrub mangrove ecosystems in the Indian River Lagoon, Florida, USA. Ecology 96, 2960–2972. https://doi.org/10.1890/14-1853.1.
Fox-Kemper, B., Hewitt, H., Xiao, G., Aoalgeirsdottir, G., Drijfhout, S., Edwards, L., Golledge, N., Hemer, M., Kopp, R., Krinner, G., Notz, D., Nowicki, S., Nurhati, I., Ruiz, L., Sallee, J., Slangen, A., Yu, Y., 2021. Chapter 9: ocean, cryosphere and sea level change. In: Sixth Assessment Report. https://doi.org/10.5285/77B64C55-7166-4A06-9DEF-2E400398E452.
Gaiser, E.E., Zafiris, A., Ruiz, P.L., Tobias, F.A., Ross, M.S., 2006. Tracking rates of ecotone migration due to salt-water encroachment using fossil mollusks in coastal South Florida. Hydrobiologia 569, 237–257.
Goff, J.R., Chagué-Goff, C., 1999. A late Holocene record of environmental changes from coastal wetlands: Abel Tasman National Park, New Zealand. Quat. Int. 56, 39–51. https://doi.org/10.1016/S1040-6182(98)00016-0.
Griffiths, L.N., Mitsch, W.J., 2021. Estimating the effects of a hurricane on carbon storage in mangrove wetlands in Southwest Florida. Plants 10, 1749. https://doi.org/10.3390/plants10081749.
Hatton, R.S., DeLaune, R.D., Patrick, W.H., 1983. Sedimentation, accretion, and subsidence in marshes of Barataria Basin, Louisiana. Limnol. Oceanogr. 28, 494–502.
Hoffmeister, J.E., Stockman, K.W., Multer, H.G., 1967. Miami limestone of Florida and its recent Bahamian counterpart. Geol.Soc. Am.Bull. 78, 175. https://doi.org/10.1130/0016-7606(1967)78[175:MLOFAI]2.0.CO;2.
Holmquist, J.R., Brown, L.N., Mac Donald, G.M., 2021. Localized scenarios and latitudinal patterns of vertical and lateral resilience of tidal marshes to sea-level rise in the Contiguous United States. Earth'sFuture 9, e2020EF001804. https://doi.org/10.1029/2020EF001804.
Howard, R.J., From, A.S., Krauss, K.W., Andres, K.D., Cormier, N., Allain, L., Savarese, M., 2020. Soil surface elevation dynamics in a mangrove-to-marsh ecotone characterized by vegetation shifts. Hydrobiologia 847, 1087–1106. https://doi.org/10.1007/s10750-019-04170-4.
Karegar, M.A., Dixon, T.H., Engelhart, S.E., 2016. Subsidence along the Atlantic Coast of North America: insights from GPS and late Holocene relative sea level data: subsidence along the NA Atlantic Coast. Geophys. Res. Lett. 43, 3126–3133. https://doi.org/10.1002/2016GL068015.
Kelleway, J.J., Cavanaugh, K., Rogers, K., Feller, I.C., Ens, E., Doughty, C., Saintilan, N., 2017. Review of the ecosystem service implications of mangrove encroachment into salt marshes. Glob. Chang. Biol. 23, 3967–3983. https://doi.org/10.1111/gcb.13727.
Kirwan, M.L., Temmerman, S., Skeehan, E.E., Guntenspergen, G.R., Fagherazzi, S., 2016. Overestimation of marsh vulnerability to sea level rise. Nat. Clim. Chang. 6, 253–260. https://doi.org/10.1038/nclimate2909.
Koch, M.S., Coronado, C., Miller, M.W., Rudnick, D.T., Stabenau, E., Halley, R.B., Sklar, F.H., 2015. Climate change projected effects on coastal foundation communities of the Greater Everglades using a 2060 scenario: need for a new management paradigm. Environ. Manag. 55, 857–875. https://doi.org/10.1007/s00267-014-0375-y.
Kominoski, J., Neubauer, S., Bremen, R., Camacho, A., Camacho-Santamans, A., Charles, S., Cherry, J., Gaiser, E., Gedan, K., Helton, A., Herbert, E., Ishtiaq, K., Kirwan, M., Krauss, K., Lamb-Wotton, L., Morant, D., Noe, G., Osland, M.,

Solohin, E., Troxler, T., Tully, K., Wilson, B., 2021. Salinity Reduces Coastal Marsh Respiration More Than Photosynthesis (preprint) In Review. https://doi.org/10.21203/rs.3.rs-880205/v1.
Kopp, R.E., Horton, B.P., Kemp, A.C., Tebaldi, C., 2015. Past and future sea-level rise along the coast of North Carolina, USA. Clim. Chang. 132, 693–707. https://doi.org/10.1007/s10584-015-1451-x.
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. https://doi.org/10.1007/s11852-011-0153-4.
Lagomasino, D., Fatoyinbo, T., Castañeda-Moya, E., Cook, B.D., Montesano, P.M., Neigh, C.S.R., Corp, L.A., Ott, L.E., Chavez, S., Morton, D.C., 2021. Storm surge and ponding explain mangrove dieback in Southwest Florida following Hurricane Irma. Nat. Commun. 12, 4003. https://doi.org/10.1038/s41467-021-24253-y.
Lee, D.Y., Kominoski, J.S., Kline, M., Robinson, M., Roebling, S., 2021. Saltwater and nutrient legacies reduce net ecosystem carbon storage despite freshwater restoration: insights from experimental wetlands. Restor. Ecol. https://doi.org/10.1111/rec.13524.
Lovelock, C.E., Feller, I.C., Mckee, K.L., Engelbrecht, B.M.J., Ball, M.C., 2004. The effect of nutrient enrichment on growth, photosynthesis and hydraulic conductance of dwarf mangroves in Panamá. Funct. Ecol. 18, 25–33. https://doi.org/10.1046/j.0269-8463.2004.00805.x.
Lovelock, C.E., Ball, M.C., Martin, K.C., Feller, C.I., 2009. Nutrient enrichment increases mortality of mangroves. PLoS ONE 4, e5600. https://doi.org/10.1371/journal.pone.0005600.
Lovelock, C.E., Cahoon, D.R., Friess, D.A., Guntenspergen, G.R., Krauss, K.W., Reef, R., Rogers, K., Saunders, M.L., Sidik, F., Swales, A., Saintilan, N., Thuyen, L.X., Triet, T., 2015. The vulnerability of Indo-Pacific mangrove forests to sea-level rise. Nature 526, 559–563. https://doi.org/10.1038/nature15538.
Lovelock, C.E., Reef, R., Masqué, P., 2021. Vulnerability of an arid zone coastal wetland landscape to sea level rise and intense storms. Limnol. Oceanogr. 14 https://doi.org/10.1002/lno.11936.
Lynch, J.C., Meriwether, J.R., McKee, B.A., Vera-Herrera, F., Twilley, R.R., 1989. Recent accretion in mangrove ecosystems based on 137Cs and 210Pb. Estuaries 12, 284–299.
Lynch, J.C., Hensel, P., Cahoon, D.R., 2015. The Surface Elevation Table and Marker Horizon Technique - A Protocol for Monitoring Wetland Elevation Dynamics (Natural Resource Report No. NPS/NCBN/NRR-2015/1078). National Park Service, Fort Collins, Colorado.
Mackenzie, L., Moss, P., Ulm, S., 2020. A late-Holocene record of coastal wetland development and fire regimes in tropical northern Australia. The Holocene 30, 1379–1390. https://doi.org/10.1177/0959683620932970.
Masson-Delmotte, V., Zhai, P., Pirani, A., Connors, S., Pean, C., Berger, S., Caud, N., Chen, Y., Goldfarb, L., Gomis, M., Huang, M., Leitzell, K., Lonnoy, E., Matthews, J., Maycock, T., Waterfield, T., Yelekci, O., Yu, R., Zhou, B., 2021. Climate Change 2021 The Physical Science Basis - Summary for Policymakers, Work Group I Contribution to the Sixth Assessment Report. Intergovernmental Panel on Climate Change, Switzerland.
Maul, G.A., Martin, D.M., 2015. Merged Miami sea level. Mar. Geod. 38, 277–280. https://doi.org/10.1080/01490419.2015.1014584.
McVoy, C., Said, W.P., Obeysekera, J., Vanarman, J., Dreschel, T.W., 2011. Landscapes and Hydrology of the Predrainage Everglades. University Press of Florida, Gainsville, Florida.
Meeder, 2022 J.F. Meeder 2022. Personal communication.
Meeder, J.F., Parkinson, R.W., 2018. SE Saline Everglades transgressive sedimentation in response to historic acceleration in sea-level rise: a viable marker for the base of the Anthropocene? J. Coast. Res. 342, 490–497. https://doi.org/10.2112/JCOASTRES-D-17-00031.1.
Meeder, J.F., Parkinson, R.W., Ruiz, P.L., Ross, M.S., 2017. Saltwater encroachment and prediction of future ecosystem response to the Anthropocene Marine Transgression, Southeast Saline Everglades, Florida. Hydrobiologia 803, 29–48. https://doi.org/10.1007/s10750-017-3318-9.
Meeder, J.F., Parkinson, R.W., Ogurcak, D., Ross, M.S., Kominoski, J.S., 2021. Changes in sediment organic carbon accumulation under conditions of historical sea-level rise, Southeast Saline Everglades, Florida, USA. Wetlands 41, 12. https://doi.org/10.1007/s13157-021-01440-7.
Mitrovica, J.X., Gomez, N., Clark, P.U., 2009. The sea-level fingerprint of West Antarctic collapse. Science 323, 753. https://doi.org/10.1126/science.1166510.
Nicholls, R.J., Cazenave, A., 2010. Sea-level rise and its impact on coastal zones. Science 328, 1517–1520.
NOAA, 2021. CO-OPS Products - NOAA Tides & Currents [WWW Document]. URL. https://tidesandcurrents.noaa.gov/products.html (accessed 9.30.21).
Obeysekera, J., Barnes, J., Nungesser, M., 2015. Climate sensitivity runs and regional hydrologic modeling for predicting the response of the Greater Florida Everglades ecosystem to climate change. Environ. Manag. 55, 749–762. https://doi.org/10.1007/s00267-014-0315-x.
Osland, M.J., Feher, L.C., Anderson, G.H., Vervaeke, W.C., Krauss, K.W., Whelan, K.R.T., Balentine, K.M., Tiling-Range, G., Smith, T.J., Cahoon, D.R., 2020. A tropical cyclone-induced ecological regime shift: mangrove forest conversion to mudflat in Everglades National Park (Florida, USA). Wetlands 40, 1445–1458. https://doi.org/10.1007/s13157-020-01291-8.
Osland, M.J., Hughes, A.R., Armitage, A.R., Scyphers, S.B., Cebrian, J., Swinea, S.H., Shepard, C.C., Allen, M.S., Feher, L.C., Nelson, J.A., O'Brien, C.L., Sanspree, C.R., Smee, D.L., Snyder, C.M., Stetter, A.P., Stevens, P.W., Swanson, K.M., Williams, L.H., Brush, J.M., Marchionno, J., Bardou, R., 2022. The impacts of mangrove range expansion on wetland ecosystem services in the southeastern United States: current understanding, knowledge gaps, and emerging research needs. Glob.Chang.Biol.n/a. https://doi.org/10.1111/gcb.16111 n/a.
Park, J., Sweet, W., 2015. Accelerated sea level rise and Florida current transport. Ocean Sci. 11, 607–615. https://doi.org/10.5194/os-11-607-2015.
Parkinson, R., 1989. Decelerating Holocene sea-level rise and its influence on Southwest Florida coastal evolution: a transgressive/regressive stratigraphy. SEPM J.Sediment. Res. 59, 960–972. https://doi.org/10.1306/212F90C5-2B24-11D7-8648000102C1865D.
Parkinson, R.W., 2022. Sediment Accumulation Rate Data Aggregated From Publications Describing Research Conducted in South Florida on Coastal Wetlands Prior to 2022. https://doi.org/10.6073/PASTA/D61C8BB72F0B4C7C827BA15AA60BDA10.
Parkinson, R.W., DeLaune, R.D., White, J.R., 1994. Holocene sea-level rise and the fate of mangrove forests within the wider Caribbean region. J. Coast. Res. 10, 1077–1086.
Parkinson, R.W., Harlem, P.W., Meeder, J.F., 2015. Managing the Anthropocene marine transgression to the year 2100 and beyond in the State of Florida U.S.A. Clim. Chang. 128, 85–98. https://doi.org/10.1007/s10584-014-1301-2.
Parkinson, R.W., Craft, C., DeLaune, R.D., Donoghue, J.F., Kearney, M., Meeder, J.F., Morris, J., Turner, R.E., 2017. Marsh vulnerability to sea-level rise. Nat.Clim. Change 7, 756. https://doi.org/10.1038/nclimate3424.
Pazi, A.M.M., Rosli, N., Hanafi, A., Tindit, A.E., 2016. Spil pH and carbon at different depth in three zone of mangrove forest in Sarawak, Malaysia. Malays.For. 79, 164–173.
Peng, D., Hill, E.M., Meltzner, A.J., Switzer, A.D., 2019. Tide gauge records show that the 18.61-year nodal tidal cycle can change high water levels by up to 30 cm. J. Geophys. Res. Oceans 124, 736–749. https://doi.org/10.1029/2018JC014695.
Ralph, G., 2021. Building Elevation in Mangrove communities: Use of Regional Sediment Management to Increase Coastal Wetland Resilience to Sea Level Rise.
Reef, R., Feller, I.C., Lovelock, C.E., 2010. Nutrition of mangroves. Tree Physiol. 30, 1148–1160. https://doi.org/10.1093/treephys/tpq048.
Rogers, K., 2021. Accommodation space as a framework for assessing the response of mangroves to relative sea-level rise. Singap. J. Trop. Geogr. 42, 163–183. https://doi.org/10.1111/sjtg.12357.
Rogers, K., Saintilan, N., 2021. Processes influencing autocompaction modulate coastal wetland surface elevation adjustment with sea-level rise. Front. Mar. Sci. 8, 694039 https://doi.org/10.3389/fmars.2021.694039.
Rogers, K., Saintilan, N., Howe, A.J., Rodríguez, J.F., 2013. Sedimentation, elevation and marsh evolution in a southeastern Australian estuary during changing climatic conditions. Estuar. Coast. Shelf Sci. 133, 172–181. https://doi.org/10.1016/j.ecss.2013.08.025.
Romero, L.M., Smith, T.J., Fourqurean, J.W., 2005. Changes in mass and nutrient content of wood during decomposition in a south Florida mangrove forest. J. Ecol. 93, 618–631. https://doi.org/10.1111/j.1365-2745.2005.00970.x.
Romero, I.C., Jacobson, M., Fuhrman, J.A., Fogel, M., Capone, D.G., 2012. Long-term nitrogen and phosphorus fertilization effects on N2 fixation rates and nifH gene community patterns in mangrove sediments. Mar. Ecol. 33, 117–127. https://doi.org/10.1111/j.1439-0485.2011.00465.x.
Ross, M.S., Meeder, J.F., Sah, J.P., Ruiz, P.L., Telesnicki, G.J., 2000. The southeast saline Everglades revisited: 50 years of coastal vegetation change. J. Veg. Sci. 11, 101–112.
Ross, M.S., Stoffella, S.L., Vidales, R., Meeder, J.F., Kadko, D.C., Scinto, L.J., Subedi, S.C., Redwine, J.R., 2021. Sea-level rise and the persistence of tree islands in coastal landscapes. Ecosystems. https://doi.org/10.1007/s10021-021-00673-1.
Saintilan, N., Khan, N.S., Ashe, E., Kelleway, J.J., Rogers, K., Woodroffe, C.D., Horton, B.P., 2020. Thresholds of mangrove survival under rapid sea level rise. Science 368, 1118–1121. https://doi.org/10.1126/science.aba2656.
Saintilan, N., Kovalenko, K., Guntenspergen, G., Rogers, K., Lynch, J., Cahoon, D., Lovelock, C., Friess, D., Ashe, E., Krauss, K., Cormier, N., Spencer, T., Adams, J., Raw, J., Ibanez, C., Scarton, F., Temmerman, S., Meire, P., Maris, T., Thorne, K., Brazner, J., Chmura, G., Bowron, T., Gamage, V.P., 2021. Global Patterns and Drivers of Tidal Marsh Response to Accelerating Sea-level Rise (preprint) In Review. https://doi.org/10.21203/rs.3.rs-363398/v1.
Sandi, S.G., Rodriguez, J.F., Saco, P.M., Saintilan, N., Riccardi, G., 2021. Accelerated sea-level rise limits vegetation capacity to sequester soil carbon in coastal wetlands: a study case in southeastern Australia. Earth's Future 9, e2020EF001901. https://doi.org/10.1029/2020EF001901.
Schuerch, M., Spencer, T., Temmerman, S., Kirwan, M.L., Wolff, C., Lincke, D., McOwen, C.J., Pickering, M.D., Reef, R., Vafeidis, A.T., Hinkel, J., Nicholls, R.J., Brown, S., 2018. Future response of global coastal wetlands to sea-level rise. Nature 561, 231–234. https://doi.org/10.1038/s41586-018-0476-5.
Servais, S., Kominoski, J.S., Charles, S.P., Gaiser, E.E., Mazzei, V., Troxler, T.G., Wilson, B.J., 2018. Saltwater intrusion and soil carbon loss: testing effects of salinity and phosphorus loading on microbial functions in experimental freshwater wetlands. Geoderma. https://doi.org/10.1016/j.geoderma.2018.11.013.
Sheikh, P.A., Carter, N.T., 2005. South Florida Ecosystem Restoration and the Comprehensive Everglades Restoration Plan (No. RS20702). Congressional Research Service, Washington, D.C.
Sklar, F.H., Carlson, C., Coronado-Molina, C., Maran, A.C., 2021. Coastal ecosystem vulnerability and sea level rise (SLR) in South Florida: a mangrove transition projection. Front. Ecol. Evol. 9, 646083 https://doi.org/10.3389/fevo.2021.646083.
Smoak, J.M., Breithaupt, J.L., Smith, T.J., Sanders, C.J., 2013. Sediment accretion and organic carbon burial relative to sea-level rise and storm events in two mangrove forests in Everglades National Park. Catena 104, 58–66. https://doi.org/10.1016/j.catena.2012.10.009.
Sweet, W., Kopp, R., Weaver, C., Obeysekera, J., Horton, R.M., Thieler, E.R., Zervas, C.E., 2017. Global and Regional Sea Level Rise Scenarios for the United States (Technical Report No. NOS CO-OPS 083). National Oceanic and Atmospheric Administration, Silver Springs, Maryland.

Thompson, P.R., Widlansky, M.J., Hamlington, B.D., Merrifield, M.A., Marra, J.J., Mitchum, G.T., Sweet, W., 2021. Rapid increases and extreme months in projections of United States high-tide flooding. Nat. Clim. Chang. 11, 584–590. https://doi.org/10.1038/s41558-021-01077-8.

Tornqvist, T.E., Cahoon, D.R., Morris, J.T., Day, J.W., 2021. Coastal wetland resilience, accelerated sea-level rise, and the importance of timescale. AGU Adv. 2 https://doi.org/10.1029/2020AV000334.

United Nations Environmental Program, 2021. Emissions Gap Report 2021: The Heat Is On - A World of Climate Promises Not Yet Delivered. Nairobi.

Valiela, I., Lloret, J., Bowyer, T., Miner, S., Remsen, D., Elmstrom, E., Cogswell, C., Robert Thieler, E., 2018. Transient coastal landscapes: rising sea level threatens salt marshes. Sci. Total Environ. 640–641, 1148–1156. https://doi.org/10.1016/j.scitotenv.2018.05.235.

van Dobben, H.F., de Groot, A.V., Bakker, J.P., 2022. Salt marsh accretion with and without deep soil subsidence as a proxy for sea-level rise. Estuar. Coasts. https://doi.org/10.1007/s12237-021-01034-w.

Wdowinski, S., Bray, R., Kirtman, B.P., Wu, Z., 2016. Wdowinski et al. 2016 Increasing flooding hazard in coastal communities due to rising sea level - case study of Miami Beach, Florida. Ocean Coast. Manag. 126, 8. https://doi.org/10.1016/j.ocecoaman.2016.03.002.

Webb, E.L., Friess, D.A., Krauss, K.W., Cahoon, D.R., Guntenspergen, G.R., Phelps, J., 2013. A global standard for monitoring coastal wetland vulnerability to accelerated sea-level rise. Nat. Clim. Chang. 3, 458–465. https://doi.org/10.1038/nclimate1756.

Whelan, K.R.T., Smith, T.J., Cahoon, D.R., Lynch, J.C., Anderson, G.H., 2005. Groundwater control of mangrove surface elevation: shrink and swell varies with soil depth. Estuaries 28, 833–843. https://doi.org/10.1007/BF02696013.

Whelan, K.R.T., Smith, T.J., Anderson, G.H., Ouellette, M.L., 2009. Hurricane Wilma's impact on overall soil elevation and zones within the soil profile in a mangrove forest. Wetlands 29, 16–23. https://doi.org/10.1672/08-125.1.

Wilson, B.J., Servais, S., Charles, S.P., Davis, S.E., Gaiser, E.E., Kominoski, J.S., Richards, J.H., Troxler, T.G., 2018. Declines in plant productivity drive carbon loss from Brackish coastal wetland mesocosms exposed to saltwater intrusion. Estuar. Coasts. https://doi.org/10.1007/s12237-018-0438-z.

Wilson, B.J., Servais, S., Charles, S.P., Mazzei, V., Gaiser, E.E., Kominoski, J.S., Richards, J.H., Troxler, T.G., 2019. Phosphorus alleviation of salinity stress: effects of saltwater intrusion on an Everglades freshwater peat marsh. Ecology, e02672. https://doi.org/10.1002/ecy.2672.

Yando, E.S., Sloey, T.M., Dahdouh-Guebas, F., Rogers, K., Abuchahla, G.M.O., Cannicci, S., Canty, S.W.J., Jennerjahn, T.C., Ogurcak, D.E., Adams, J.B., Connolly, R.M., Diele, K., Lee, S.Y., Rowntree, J.K., Sharma, S., Cavanaugh, K.C., Cormier, N., Feller, I.C., Fratini, S., Ouyang, X., Wee, A.K.S., Friess, D.A., 2021. Conceptualizing ecosystem degradation using mangrove forests as a model system. Biol. Conserv. 263, 109355 https://doi.org/10.1016/j.biocon.2021.109355.