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Hot and Fresh: Evidence of Climate-Related Suboptimal Water Conditions for Seagrass in a Large Gulf Coast Estuary

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Abstract

Seagrasses have long been a focal point for management efforts aimed at restoring ecosystem health in estuaries worldwide. In Tampa Bay, Florida (USA), seagrass coverage has declined since 2016 by nearly a third (11,518 acres), despite sustained reductions of nitrogen loads supportive of light environments for growth. Changing physical water quality conditions related to climate change may be stressing seagrasses beyond their optimal growth ranges, requiring an assessment to determine if this decline can be linked to climate stress. Three ambient water quality datasets of varying sampling designs and coverage were evaluated to characterize physicochemical environments in Tampa Bay and the potential relationships with seagrass change. Tampa Bay has become hotter and fresher with water temperature increasing by 0.03–0.04 °C per year and salinity decreasing by 0.04–0.06 ppt per year, translating to an increase of 1.3 to 1.7 °C and a decrease of 1.6 to 2.6 ppt over the last 50 years. Additionally, the number of days when temperature was above 30 °C or salinity was below 25 ppt has increased on average across all bay segments by 48 and 37 days, respectively, since 1975. These changes varied spatially and seasonally, with the most dramatic changes observed in the upper bay. Generalized Additive Models provided a weight-of-evidence that recent seagrass declines are somewhat associated with hotter and fresher conditions. Trends in warming and increased precipitation in the region are likely to continue, further creating suboptimal conditions for seagrasses in Tampa Bay. These results should compel resource managers to consider the likelihood that reduced resilience of estuarine resources due to shifting ecological baselines driven by additional climate change drivers will complicate long-standing management paradigms. While conventional management approaches that focus on limiting nutrient loads should be continued, their future effectiveness may be confounded by climate change drivers and warrant additional, complementary interventions and continuous monitoring data to support ecosystem health into the future.

Keywords Climate change · Seagrass · Salinity · Temperature · Tampa Bay

Introduction

The monitoring and management of seagrasses in coastal environments has received substantial attention on a global scale. Seagrasses are fundamental indicators of coastal ecosystem health (Roca et al. 2016; Orth et al. 2017), while also serving as foundation species that provide numerous ecosystem services (Fourqurean et al. 2012; Orth et al. 2020; Orth and Heck 2023). Seagrasses have been in global decline with rapid development of coastal environments, particularly in the latter half of the twentieth century with accelerating losses estimated at a rate of 110 km^2 yr^{-1} since the 1980s (Waycott et al. 2009; Dunic et al. 2021). These losses are comparable to, if not more significant than, other interconnected critical coastal environments such as mangroves

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and salt marshes (Duarte et al. 2008). Losses have been attributed to numerous stressors including decline in light environments with nutrient enrichment, sedimentation, and physical disturbance primarily from human activities (Duarte 1995; Hall et al. 1999; Orth et al. 2006; Burkholder et al. 2007). Furthermore, natural disturbances such as storm events and disease have also been implicated (Robblee et al. 1991; Tomasko et al. 2020). Contemporary management actions aimed at mitigating loss and ultimately supporting restoration require adaptive approaches to address the effects of multiple stressors that have contributed to seagrass decline (Dunic and Côté 2023).

The sustained coverage or restoration of seagrasses in coastal environments requires environmental conditions that support vegetative growth, reproduction, and coverage expansion. A long-standing approach adopted by numerous management entities has been the control of external nutrient inputs in systems where excessive algal growth has created poor light environments for seagrasses (Boesch et al. 2001; Greening and Janicki 2006; Greening et al. 2014; Han and Liu 2014). There are limited examples of successful recovery of seagrass through control of nutrient inputs alone, primarily because of the difficulty in identifying and regulating both point and diffuse non-point sources. Notable exceptions include Tampa Bay on the west coast of Florida (Greening et al. 2014) and the much larger Chesapeake Bay on the east Atlantic US coast (Lefcheck et al. 2018), where both showed significant increase in seagrass areal coverage through sustained and long-term reductions in external nutrient loads. Cooperation among management, regulatory, public, and private sectors were critical aspects of both examples (Sherwood et al. 2016; Tango and Batiuk 2016). In other cases, the reversal of seagrass losses through nutrient reductions alone may not be possible because of system hysteresis, where the path to recovery is not the same as the path to decline (Maxwell et al. 2016). Top-down effects and associated trophic cascades can also complicate the sustained growth or recovery of seagrass, particularly where herbivores are abundant (e.g., sea turtles, manatees, Heck and Valentine 2007; Fourqurean et al. 2010). Complementary management actions, in addition to nutrient reductions, are needed in these situations.

Climate change has complicated the understanding of ecosystem response to conventional stressors, presenting new challenges and expectations for how ecological resources will respond to management actions (Statham 2012; Sherwood and Greening 2013). In addition to sea-level rise, the most anticipated effects of climate change in coastal environments are increased temperature and altered precipitation patterns. These changes will profoundly alter physicochemical habitats, creating suboptimal or uninhabitable conditions for many species (Madeira et al. 2012; Lefcheck et al. 2017; Hammer et al. 2018; Hense et al. 2023). Lefcheck et al. (2017) evaluated interactive effects of water clarity and rising temperatures on seagrasses with over 30 years of data in Chesapeake Bay. The environmental stress on seagrasses from acute warming related to climate change was compounded by stress from poor light environments in shallow waters, demonstrating a concerning synergy of stressors most likely to also affect Tampa Bay. Moreno–Marín et al. (2018) produced similar results using a multifactorial experiment that considered temperature, light, and nutrient (nitrogen) availability for North Sea eelgrass (*Zostera marina*). Species shifts are also expected to occur as changing physical conditions decrease the competitive advantages of historically abundant species. In Chesapeake Bay, the abundance of widgeongrass (*Ruppia maritima*) has responded positively to nutrient reduction and is replacing the formerly dominant but now heat-stressed *Z. marina* (Hense et al. 2023; also see Bartenfelder et al. 2022). Changing frequency and severity of precipitation patterns may further alter salinity regimes and with it the distribution and abundance of seagrasses throughout the estuary (Rasheed and Unsworth 2011; Webster et al. 2021). These changes may produce hypersaline conditions under periods of prolonged drought or more freshwater conditions with increased storm events. Seagrass species may respond differently under individual or multiple stressors and each watershed will respond differently to climate change (Hall et al. 2016; Lefcheck et al. 2017; Zhang et al. 2023), suggesting place-based empirical assessments will be needed to properly inform management decisions.

In Tampa Bay, Florida, seagrasses are a primary indicator of bay health and have been the focus of management efforts for the last 30 years (Sherwood et al. 2017). Through successful reduction of external nitrogen loads, seagrasses have recovered from a low, system-wide coverage in the 1980s to an all-time high in 2016 of 41,655 acres (16,857 ha, Greening and Janicki 2006; Greening et al. 2014). Seagrass distribution and abundance have been dynamic throughout this period, overall responding positively to increases in water clarity with nutrient load reductions, while also demonstrating more short-term variability in response to regional climate events (Greening and Janicki 2006). Since 2016, seagrass areal cover has decreased by nearly 1/3 despite relatively stable water quality (Janicki and Wade 1996; Beck 2020a). Factors that have influenced this recent decline are unknown, and the effects of climate change drivers on physical water quality conditions independent of light environments have been implicated as potential stressors. Following global trends, recent work has demonstrated a broad long-term trend of increasing water temperature in Tampa Bay, although at a relatively coarse scale (Nickerson et al. 2023). Changing salinity has not been well-described, nor have potential links of changing salinity and temperature with recent seagrass change. Tampa Bay is rich with historical data that can be

used to evaluate long-term trends. This information can fill a critical knowledge gap that can inform regional management activities, while also demonstrating the confounding effects of climate change with ecosystem response to conventional stressors. The improvement of bio-optical models that can describe light requirements for seagrasses may also result from this information.

This paper describes a comprehensive assessment of long-term trends in water temperature and salinity in the Tampa Bay estuary over the last 50 years. Three datasets of varying sampling designs and temporal coverage were used to assess the primary hypothesis that Tampa Bay is trending towards hotter and fresher conditions that are likely stressing seagrasses beyond their optimal tolerance ranges, particularly in upper bay segments that are more shallow and receive a majority of hydrologic inflow. This hypothesis was generated from preliminary assessments of datasets used herein and discussions with the regional scientific and management community. While other studies have suggested negative effects from climate change in the form of increased salinity (e.g., Costa et al. 2023), lower salinity may be a stressor for seagrass in Tampa Bay. The analysis was supported by 30-year seagrass datasets including aerial surveys of total seagrass distribution, annual transect monitoring describing species-specific percent cover, and synoptic seagrass data collected with routine biotic and water quality surveys.

Methods

Study Area

Tampa Bay is the largest open-water estuary in Florida covering 400 mi^2 (1036 km^2) and the second largest in the Gulf of Mexico. The watershed covers an additional 2200 mi^2 (5872 km^2) with the Hillsborough, Alafia, Manatee, and Little Manatee Rivers contributing the majority of freshwater inflow to the bay. The climate of Tampa Bay is subtropical with warm humid summers and cool, less humid winters (Garcia et al. 2023). Unique to Tampa Bay and the entire central Florida peninsula is that this region is within a transition zone from more temperate weather to the north, similar to the rest of the southeastern United States, and more tropical weather to the south (Morrison et al. 2006). El Niño weather events have also been associated with prolonged periods of heavy rainfall contributing to seagrass reductions through increased stormwater nutrient loads (Schmidt and Luther 2002; Greening and Janicki 2006; Morrison et al. 2006). The watershed is heavily developed and includes over 3 million people (Todd et al. 2023) with 42% of the land as urban or suburban contributing substantial inputs of wastewater and stormwater runoff that can stress bay resources (Beck et al. 2023). The geology of the watershed is rich in phosphates and mining activities have greatly altered the landscape, with notable spills and releases of wastewater that have affected water quality and biological resources (Garrett et al. 2011; Beck et al. 2022).

Tampa Bay is divided into distinct sub-segments defined by physical and natural boundaries to assist with water quality management activities (Lewis III et al. 1985): Old Tampa Bay (OTB) in the northwest; Hillsborough Bay (HB) in the northeast; Middle Tampa Bay (MTB); and Lower Tampa Bay (LTB) that connects to the Gulf of Mexico (Fig. 1a). Old Tampa Bay and Hillsborough Bay have historically had the most degraded water quality primarily from direct external nutrient inputs from wastewater and stormwater (Greening et al. 2014). Hydrologic conditions vary between the two, such that Hillsborough Bay receives a majority of direct surface water inflow from the Hillsborough and Alafia Rivers, whereas Old Tampa Bay receives much less inflow with a majority from multiple small, channelized tributaries and manmade flood control conveyances (Janicki Environmental, Inc. 2023). Notably, Old Tampa Bay has restricted circulation from multiple land bridges associated with causeways that traverse the bay, causing longer residence times and accumulation of legacy pollutant loads compared to the other bay segments (Sherwood et al. 2015; Luther and Meyer 2022). Recurring seasonal harmful algal blooms of the dinoflagellete *Pyrodinium bahamense* have contributed to exceedances of the chlorophyll-a regulatory standard in Old Tampa Bay (Lopez et al. 2023). By comparison, water quality conditions in Middle Tampa Bay and Lower Tampa Bay are generally better than the upper two bay segments primarily from more frequent water exchanges with the Gulf of Mexico and lower nutrient loading (Janicki Environmental, Inc. 2023). All bay segments are shallow, with a baywide mean depth of approximately 3 m. Light penetration typically reaches bottom habitats under current conditions (Fig. 2c), although seagrasses were historically limited by high phytoplankton production that affected light environments (Greening et al. 2014; Johansson and Janicki Environmental, Inc. 2015).

Seagrass Change in Tampa Bay

The long-term recovery of seagrass habitats in Tampa Bay since the 1980s is a nationally recognized success story that demonstrates application of a successful management paradigm through the EPA-administered National Estuary Program (Greening and Janicki 2006; Greening et al. 2014; Sherwood et al. 2017). From 1988 to 2016, seagrasses increased by 79% to 41,655 acres (16,857 ha), surpassing the regional management goal of restoring coverage to 95% of what was present in the 1950s. Though Tampa Bay was far from pristine at the time, aerial imagery was sufficient

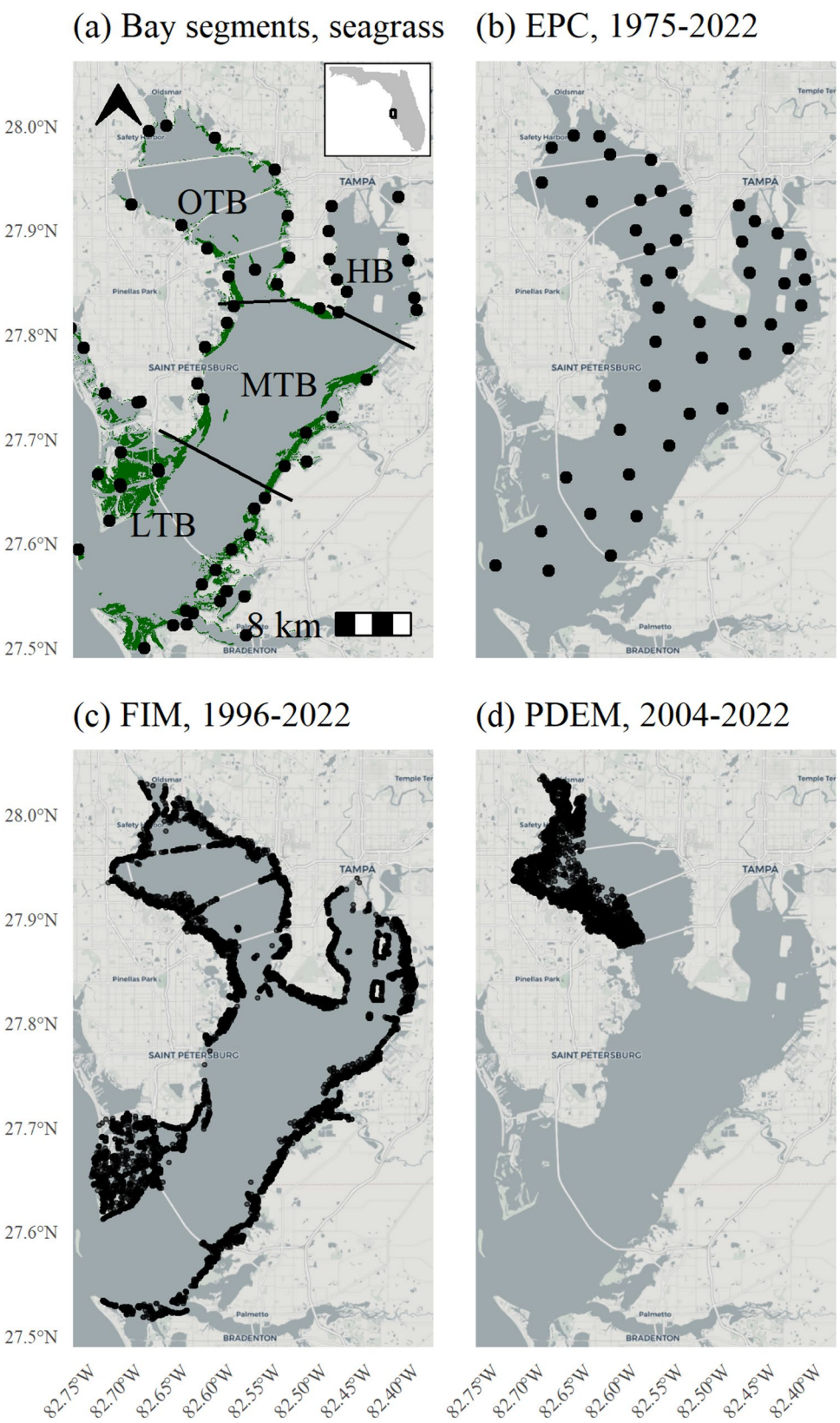


Fig. 1 Map of Tampa Bay and the three datasets used for trend analysis, including bay segments, 2022 seagrass coverage (green), and transect starting points (black) **a**; Environmental Protection Commission (EPC) long-term monitoring sites **b**; Fisheries Independent Monitoring (FIM) random sampling for seine hauls **c**; and OTB portion of Pinellas County Division of Environmental Management (PDEM) random sampling **d**. Date ranges for each dataset are shown in the title. OTB Old Tampa Bay, HB Hillsborough Bay, MTB Middle Tampa Bay, LTB Lower Tampa Bay

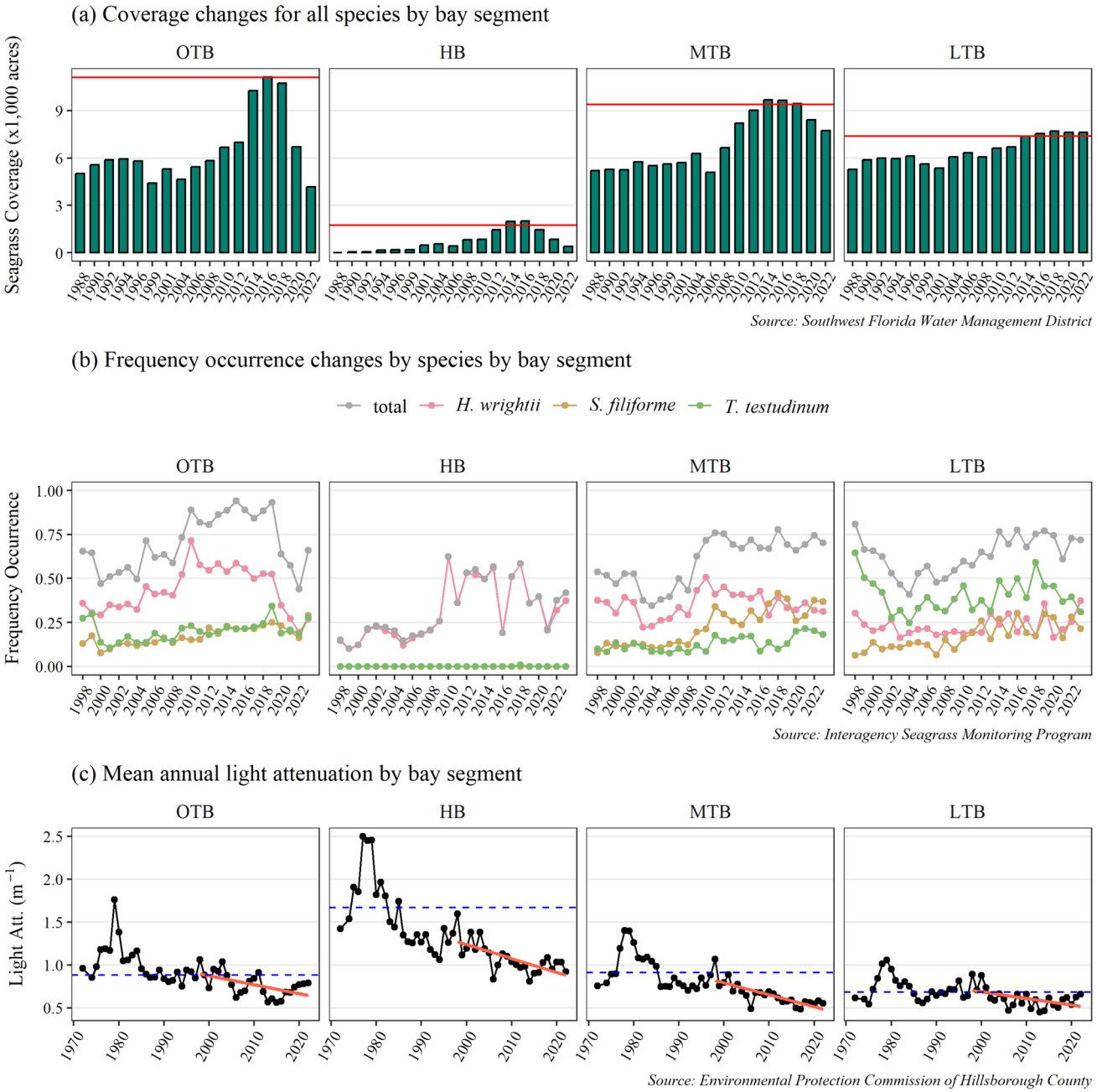


Fig. 2 Seagrass changes over time in Tampa Bay for areal coverage 1988–2022 from mapping **a**, frequency occurrence of major species 1998–2022 from annual transect monitoring **b**, and mean annual light attenuation from 1970 to 2022 **c**. Changes are shown for major bay segments. Red lines in **a** show the approximate capacity of seagrass coverage based on the baywide target of 40,000 acres and red lines in **c** show the approximate linear trend for the period of record covered by the transect data in **b**. The dashed line in **c** shows the light attenuation threshold to support seagrass growth in each bay segment. OTB Old Tampa Bay, HB Hillsborough Bay, MTB Middle Tampa Bay, LTB Lower Tampa Bay

to estimate a relatively unimpacted condition for seagrass coverage. Since then, the greatest areal coverage expansions were observed in the upper bay segments (OTB, HB, and MTB; Fig. 2a). Light environments have improved (Fig. 2c) through a 2/3 reduction of external nitrogen loadings from a peak 1970s estimate of 8.9×10^6 kg/year, largely from advanced wastewater treatment upgrades and in part from the cumulative effects of habitat restoration and additional stormwater control projects implemented in the watershed (Greening et al. 2014; Beck et al. 2019).

From 2016 to present, dramatic seagrass loss has been observed in Tampa Bay, despite light environments remaining supportive of growth (Fig. 2c) as defined by historical relationships between nitrogen, chlorophyll-a, and water

clarity (Janicki and Wade 1996; Greening et al. 2011). Total cover in Tampa Bay has decreased by 28% (11,518 acres/4661 ha) from the 2016 peak to a total baywide coverage of 30,137 acres (12,196 ha) in 2022. Losses have been most pronounced in Old Tampa Bay (62%; 6963 acres/2818 ha loss) and Hillsborough Bay (80%; 1599 acres/647 ha loss). The current estimate for Old Tampa Bay of 4183 acres (1693 ha) is the lowest ever recorded in that bay segment since mapping efforts began in the 1980s. Coverage in Middle Tampa Bay decreased by 20% (1926 acres/779 ha loss), whereas coverage in Lower Tampa Bay has remained stable, with only a 2% loss which is close to the mapping error. As such, trajectories of recovery and decline have varied by bay segment in magnitude and timing of the change, although a consistent decline, baywide has been observed since 2016. Change by species has been most notable for *Halodule wrightii* (shoal grass) in OTB and HB, whereas some losses have been observed for *Thalassia testudinum* (turtle grass) in LTB, likely related to the dominance of each species across the salinity gradient and proximity to hydrologic inputs (Lewis III et al. 1985).

Seagrass Data

Two primary sources of data have been used to track seagrass change in Tampa Bay (Table 1). The Southwest Florida Water Management District (SWFWMD) has estimated areal coverage of seagrasses approximately biennially since the late 1980s (Fig. 2a, available at https://data-swfwmd.opendata.arcgis.com/). These maps are created from aerial images collected specifically to map seagrass and are acquired during a flight window from December to February. The maps are created by photointerpretation of image signatures coupled with a robust field verification and accuracy assessment. The maps provide a spatial estimate of seagrass cover at the landscape scale, irrespective of species. The Tampa Bay Interagency Seagrass Monitoring Program is complementary to the SWFWMD seagrass maps (Fig. 2b, https://tampabay.wateratlas.usf.edu/seagrass-monitoring/). Annual transect surveys have been conducted since 1998 at 62 fixed locations in Tampa Bay, many of which were chosen to target seagrass beds of interest (Johansson 2016; Sherwood et al. 2017). This dataset provides species information on cover-abundance, frequency occurrence (number of sample points with seagrass divided by total points on a transect, as in Sherwood et al. 2017), and condition, collected at fixed meter marks along a transect extending from the shoreline to the deepwater edge of the seagrass bed. Although the areal maps provide the standard for assessment of restoration goals, the maps are produced every other year. The transect data are collected each year, allowing inter-annual comparison at greater temporal resolution, particularly for the recent period of interest when seagrasses have declined. As such, the transect data were used for comparison with temperature and salinity changes for the major bay segments. Additional sources of seagrass data are described in the next section.

Water Quality Data

Several datasets with distinct sample designs are available to assess long-term changes in water temperature and salinity in Tampa Bay (Table 1). These datasets were evaluated individually to assess trends and relationships with seagrass change to provide a weight-of-evidence approach for potential causal relationships driving the recent decline. First, the Environmental Protection Commission (EPC) of Hillsborough County has collected discrete water quality measurements monthly at fixed stations in the major bay segments since the early 1970s (Fig. 1b). The 45 stations with the longest and most complete temporal record from 1975 to present were used herein. Water quality samples are collected at each station from surface water grabs (e.g., nutrients, biological, and chemical constituents) or in situ measurements of physical parameters (e.g., salinity, temperature) collected at the surface, mid-depth, and bottom. Most analyses herein used only bottom water measurements given the shallow depth and mixed water column of most of Tampa Bay (Weisberg and Zheng 2006), although 1975 bottom salinity used middle water column sampling since the former was not available until the following year. Most samples are collected from mid-morning to early afternoon. Compared to the additional datasets described below, the monitoring stations are generally in deeper water beyond where seagrasses occur along the shallow margins of the bay. The data were obtained using the *tbeptools* R package that imports the data directly from a stable web address provided by the EPC (Beck et al. 2021).

The second dataset used to evaluate water quality trends was available from the Florida Fish and Wildlife Conservation Commission (FWC). The Fisheries Independent Monitoring (FIM) program administered by FWC provides monthly surveys of the entire nekton community in Tampa Bay, including species richness and abundance, using multiple gear types that target different habitats (Schrandt et al. 2021). A stratified sampling design is used to select sites for 21.3-m center-bag seines that target shallow habitats (< 1.5 m) where seagrasses are predominantly found in Tampa Bay and include the longest consistent sampling protocol (1996 to present, Fig. 1c). In addition to collecting fish and selected invertebrates, in situ physical measurements for water temperature and salinity are collected at the bag, and at the surface and at 1-m intervals to the bottom. Only measurements from the bottom were used. Seagrass data are also provided for each site, with information on species and cover. The total percent cover for all species at a site was used for comparison with temperature and salinity

Table 1 Summary of datasets used for trend analyses and comparisons with seagrass datasets. Temporal and spatial scales are those used for analysis. SWFWMD Southwest Florida Water Management District, EPC Environmental Protection Commission of Hillsborough County, FIM Florida Fish and Wildlife Commission, Fisheries Independent Monitoring, PDEM Pinellas County Division of Environmental Management

Dataset	Description	Temporal	Spatial	Analysis
SWFWMD aerial maps	Seagrass coverage in acres	1988–2022, ~ biennial	Whole bay	Biennial trends by bay segment, visual only
Transect data	Seagrass frequency occurrence by species	1998–2022, annual	Whole bay, 62 transects	Annual trends by bay segment and species, comparison with temperature, salinity, and light attenuation as stressor metrics or observed data at annual scale
EPC	Water quality monitoring samples	1975–2022, monthly	Whole bay, fixed sites	Trends in annual change and seasonal Kendall tests, estimate of stressor metrics as number of days above/below threshold
FIM	Nearshore temperature and salinity, seagrass species and cover	1996–2022, monthly	Whole bay shallow, stratified random sites	Trends in annual observed temperature, salinity, comparison to annual seagrass % cover
PDEM	Water quality and seagrass presence/absence	2004–2022, monthly	Old Tampa Bay, stratified random sites	Trends in annual observed temperature, salinity, comparison to annual seagrass frequency occurrence
Tampa International Airport	Air temperature	1975–2022, annual	27.979° N, 82.535° W	Annual trend
SWFWMD precipitation	Area-weighted precipitation for the wet season (June–September) for the Tampa Bay watershed	1975–2022, annual	Whole watershed	Annual trend

measurements. Sites exclusively with macroalgae were not included in the analysis. All FIM data were provided from FWC staff upon request.

The third and final dataset evaluated was from the Pinellas County Division of Environmental Management (PDEM). Data were obtained by request to PDEM staff for the western portion of Old Tampa Bay where sampling occurred from 2004 to present (Fig. 1d, also available at https://wateratlas.usf.edu/). We focused primarily on OTB for the analysis of the PDEM data given the length of record, consistency of sampling, and relative loss of seagrass compared to the other bay segments. Water quality samples at each site are similar to those collected by EPC but can occur in shallower locations. Only bottom temperature and salinity were used for analysis. Seagrass presence/absence is also recorded at each site and all sites were defined as "seagrass" if only seagrass species were identified (any with macroalgae were excluded) and "no seagrass" if bare sediment was observed.

Trend Analysis

The first goal of the analysis was to describe spatial and temporal trends in water temperature and salinity using the three water quality datasets described above. This assessment provided an indication of the extent of water quality change in Tampa Bay as context for understanding potential relationships with seagrass change. An assumption was that any changes in physical characteristics in Tampa Bay were driven by interannual changes in weather conditions related to long-term (multi-decadal) climate change drivers. For comparison to water quality conditions, daily air temperature (Tampa International Airport [TIA] National Weather Service site) and precipitation (SWFWMD area-weighted watershed summaries) were used to characterize regional conditions for the most consistent period of record covered by the water quality samples (i.e., 1975 to present for the EPC data, Table 1). The *rnoaa* R package (Chamberlain and Hocking 2023) was used to obtain the TIA temperature data. Regional precipitation summaries were obtained directly from the SWFWMD (https://www.swfwmd.state.fl.us/resources/data-maps/rainfall-summary-data-region). Only rainfall data for the wet season (June to September) were evaluated for trends, whereas the complete record was used to calculate the Standardized Precipitation Index (SPI, Beguería et al. 2013) to identify periods of time when rainfall deviated from the long-term average (using the *spei* R package, Beguería and Vicente-Serrano 2023). All climate data were evaluated annually with simple linear regression trends to assess change over time. Water temperature and salinity trends using the EPC, FIM, and PDEM data were similarly evaluated by averaging the monthly data each year for each bay segment.

Formal trend tests were used to assess station-level changes in water temperature and salinity in the EPC data. These analyses also provided a detailed spatial assessment of trends because the EPC data is the only dataset of the three where the same sites have been sampled over time. Seasonal Kendall trend tests were used to evaluate the monotonic change for temperature and salinity from 1975 to present at each water quality station (Hirsch et al. 1982; Millard 2013). Kendall tests were also used to evaluate changes over time for each month across years to determine when the trends were most pronounced seasonally (e.g., all January estimates across years, all February estimates, etc.).

Quantifying Potential Stress

The second goal of the analysis was to evaluate if seagrass changes were linked to long-term changes in water temperature and salinity. The conceptual model for evaluating these changes describes the fundamental niche space where seagrass growth and reproduction is hypothesized to be greatest within optimal ranges for forcing factors that are present in the environment (Hutchinson 1957; Vandermeer 1972). In the simplest form, this can be conceptualized as a bell curve with optimal conditions defined within a range of values for a single parameter, where reduced growth or mortality is observed outside of these ranges. Because both water temperature and salinity were evaluated, the same model can be conceptualized in two-dimensional space (Fig. 3). Seagrass growth can be limited when temperature is below or

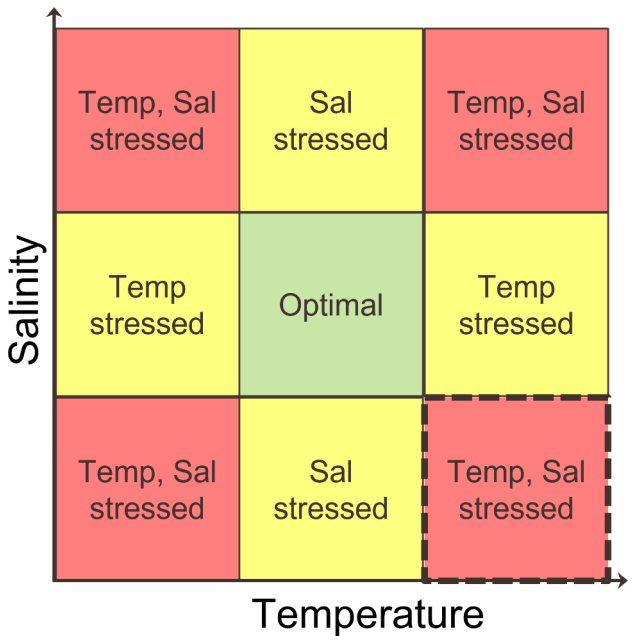


Fig. 3 Conceptual stressor diagram demonstrating a two-dimensional niche space for temperature and salinity. Tampa Bay is trending towards the bottom-right

above the optimum range, when salinity is below or above the optimum range, or when both temperature and salinity conditions are outside of the optimum range. Based on the results of the trend tests, we hypothesized that seagrasses are likely stressed by both high temperature and low salinity (bottom right, Fig. 3). Although the fundamental niche space can be defined in multiple dimensions for many parameters, we focus on water temperature and salinity given that other dominant forcing factors, i.e., light availability, have been sufficient for growth in recent years (Fig. 2c).

A fundamental challenge describing niche space is identifying the boundaries for optimal conditions. In Tampa Bay, three dominant seagrass species occur: *Halodule wrightii* (shoal grass), *Syringodium filiforme* (manatee grass), and *Thalassia testudinum* (turtle grass) (Lewis III et al. 1985; Phillips and Meñez 1988). Other less common species include *Ruppia maritima* (widgeon grass) and *Halophila engelmannii* (star grass), where the former is often mapped during wet years in the upper bay segments from the aerial surveys. These species co-occur often in mixed beds throughout the bay, although some differences in abundance are observed across salinity ranges. Shoal grass is tolerant of a wide range of salinity (Lirman and Cropper 2003) but is more abundant in oligo/mesohaline portions of Tampa Bay. Conversely, turtle grass is less tolerant of low salinity and is more abundant in more euryhaline conditions near the mouth of Tampa Bay. Reported salinity ranges for each of these species vary depending on location, season, and other co-occurring factors like temperature (Phillips 1960; McMillan and Moseley 1967; Zieman 1975; Lewis III et al. 1985), although most studies place lower limits of salinity in the range of 15–25 ppt. Optimal temperature ranges are similar between these temperate-tropical species, with reduced growth observed at temperatures above 30 °C (Zieman 1975; Lewis III et al. 1985).

Because of the uncertainty in defining in situ thresholds for optimal temperature and salinity ranges, multiple thresholds were evaluated to describe the potential for stress and how it may be related to changes in seagrass. Distinctions were not made between species, primarily due to lack of consensus between studies and likely site-specific ranges that affect seagrass growth in Tampa Bay, as well as challenges of modeling fundamental and realized niche spaces between competing species (Araújo and Guisan 2006). First, we developed metrics of potential temperature and salinity stress by quantifying the maximum number of continuous days each year when temperature was above or salinity was below a given threshold. This approach assumed that stress could be observed based on duration of exposure (i.e., maximum number of continuous days each year) relative to a threshold that may or may not be outside of the optimum range for seagrasses. These metrics were quantified from the monthly long-term observations in the EPC data. To quantify daily counts each year, a continuous prediction of temperature and salinity over time at each of the 45 stations was estimated using Generalized Additive Models (GAMs) fit to temperature or salinity with a single predictor for decimal year (Wood 2017). Model fit for each station was considered sufficient to calculate daily predictions to assess potential stressor metrics (Fig. S5, R^2 ranged from 0.85 to 0.95 for temperature models, 0.66 to 0.95 for salinity models, Tables S1, S2).

Counts of the maximum continuous number of days each year that temperature was above or salinity was below a threshold were obtained from the daily GAM predictions. This was done at each of the 45 stations in the EPC data using temperature thresholds of 29, 30, and 31 °C and salinity thresholds of 15, 20, and 25 ppt. The number of days when both temperature was above and salinity was below the thresholds was also estimated as a combined potential stress measure. Stressor metrics were further aggregated across stations in each bay segment using a mixed-effects regression model where the annual stressor counts for stations in a bay segment were fit against year (1975 to 2022) using a random intercept for station (Zuur et al. 2009; Bates et al. 2015). This produced an overall assessment of how the stressor metrics have changed over time by bay segment.

Links to Seagrass

For comparison to seagrass, the annual metrics calculated from the EPC data were referenced to approximate periods of time between the annual seagrass transect surveys, as opposed to the calendar year for describing trends above. Bay segment stressor metrics were calculated as the average counts in each "transect year" from all stations in each segment from 1998 to 2022. Preliminary analyses evaluated different lagged associations between the stressor metrics and seagrass change, although initial results suggested no additional insight could be gained using lagged assessments compared to the transect year summaries. As such, the stressor metrics were compared to frequency occurrence (all species) each year by bay segment; note that the seagrass transect data were collected at different locations and depths ($\bar{Z} = 0.76$ m) than the EPC water quality data ($\bar{Z} = 3.88$ m, excludes LTB not evaluated). GAMs were used to evaluate frequency occurrence in response to the independent variables, where the latter were the stressor metrics for temperature, salinity, or both. Additional predictors included year and light attenuation as estimated from Secchi depth (Janicki and Wade 1996). From 1998 to 2022, 17% of the Secchi observations were recorded on the bottom. A single smooth term was used for each predictor using a thin plate regression spline. Initial models also included a tensor product interaction term with a cubic regression spline that evaluated the potential interacting effects of each predictor with year. However, most of

the interaction terms were not significant and were excluded from the final models. Two models were evaluated, one with the bottom temperature and salinity metrics together and another with the "both" metric. All models excluded Lower Tampa Bay because of minimal seagrass change over time. Additionally, total seagrass frequency occurrence was used as a response variable as an aggregate measure of community change to potential stressors given the likelihood that species-specific changes may be more difficult to model and that the majority of seagrass loss in recent years was dominated by *H. wrightii* in the upper bay segments (Fig. 2b). All GAMs were fit using restricted maximum likelihood evaluation (Wood 2011) with the *mgcv* R package (Wood 2017).

Two other models were constructed to provide an additional weight-of-evidence for the FIM and PDEM temperature and salinity datasets relative to seagrass change. These models used direct measurements of bottom salinity and temperature as independent variables because the stressor metrics could not be calculated using the sampling designs from these monitoring programs (i.e., each sample was a distinct location). GAMs were used for the FIM data to evaluate annual average seagrass percent cover using year, temperature, and salinity as independent variables. Separate smoothed terms for each predictor and each bay segment were included as above. GAMs for the PDEM data were constructed similarly, except only Old Tampa Bay was evaluated due to spatial limitations of the data. Neither the FIM nor PDEM models used light attenuation as a predictor variable given that a large percentage of Secchi observations were measured on the bottom (92% and 45%, respectively), providing further support that light environments have not been limiting for seagrasses. Model input data were further subset to include only months from July to October to describe seagrasses during the growing season and to reduce potential seasonal effects. Lastly, all data were averaged annually for the monthly subsets for comparability of sample size (i.e., power) with the EPC models. For the PDEM models, presence/absence was converted to frequency occurrence as the number of sites with seagrass in a year divided by the total number of sites, and all data were subset to less than 2 m to better characterize locations where seagrass occurs.

Results

Temperature and Salinity Trends

Long-term meteorological data showed increasing trends for air temperature and precipitation (Fig. 4). The mean annual air temperature has increased by 0.04 °C per year ($n = 48$, $p < 0.001$, R^2 0.51). The mean annual air temperature in 1975 was 22.1 (+/− 0.17 st. err.) °C, whereas the current mean annual air temperature in 2022 was 24.1 (+/− 0.17 st. err.) °C, showing an overall increase in the period of record of 2 °C. Similarly, total precipitation during the rainy season has increased by 2.21 mm per year, although the trend was weak ($n = 47$, $p = 0.104$, $R^2 = 0.04$). Removing September from the rainy season showed a slightly larger trend of 2.4 mm per year and slightly more powerful trend model ($n = 48$, $p = 0.039$, $R^2 = 0.07$), suggesting precipitation increases were driven by the earlier months (June–August). Using this model, the mean precipitation for June to August in 1975 was 559.1 (+/− 30.72 st. err.) mm, whereas the current mean precipitation for June to August in 2022 was 669.3 (+/− 29.76 st. err.) mm, showing an overall increase in the period of record of 110.3 mm. Notably, rainfall during the dry season (October through May) has decreased slightly over time at 1.97 mm per year, although the trend model was weak ($n = 48$, $p = 0.261$, $R^2 = 0.01$). The SPI showed notable anomalies in precipitation, with pronounced rainy periods in the early 1980s, late 1990s, 2005, and 2015–2020 (third row, Fig. 4).

Increasing water temperature and decreasing salinity generally followed the meteorological trends for all three in situ datasets (EPC, FIM, and PDEM, Fig. 4, Tables 2, 3, Figs. S1, S2, S3). Note that for Tables 2 and 3, comparable time periods were evaluated between the datasets when possible given the different sample sizes, and therefore power, to detect trends (starting the year 1975, $n = 48$; 1996, $n = 27$; 2004, $n = 19$). The strongest trends were observed for the EPC dataset based on the standard errors for the slope estimates. The top and bottom water temperature or salinity changes were similar across bay segments (Fig. 4). Trends in water temperature were similar across bay segments with large increases for all four bay segments varying from 0.03 to 0.04 °C per year, with a total change from 1.3 (OTB) to 1.7 (HB) °C across the period of record from 1975 to 2022 (Table 2). Salinity trends were also similar between bay segments, although overall salinity was predictably higher for bay segments closer to the Gulf of Mexico. Only Old Tampa Bay and Lower Tampa Bay had notable decreasing trends, with decreases of 0.06 (OTB) and 0.04 (LTB) ppt per year, with a total decrease of 2.6 (OTB) and 1.6 (LTB) ppt from 1975 to present. Increasing temperature trends and decreasing salinity trends were also observed for the FIM and PDEM dataset, although confidence in the slope estimates (larger standard errors) generally decreased for shorter time periods of assessment.

The EPC dataset was also used to provide detailed information on station-level trends from 1975 to present (Fig. 5, see Fig. S4 for 1998 to 2022). All stations had increasing temperature and decreasing salinity from 1975 to present, although some stations in HB had weak salinity trends (Fig. 5a). Seasonally, most bay segments had more stronger trends in the early fall/winter periods for

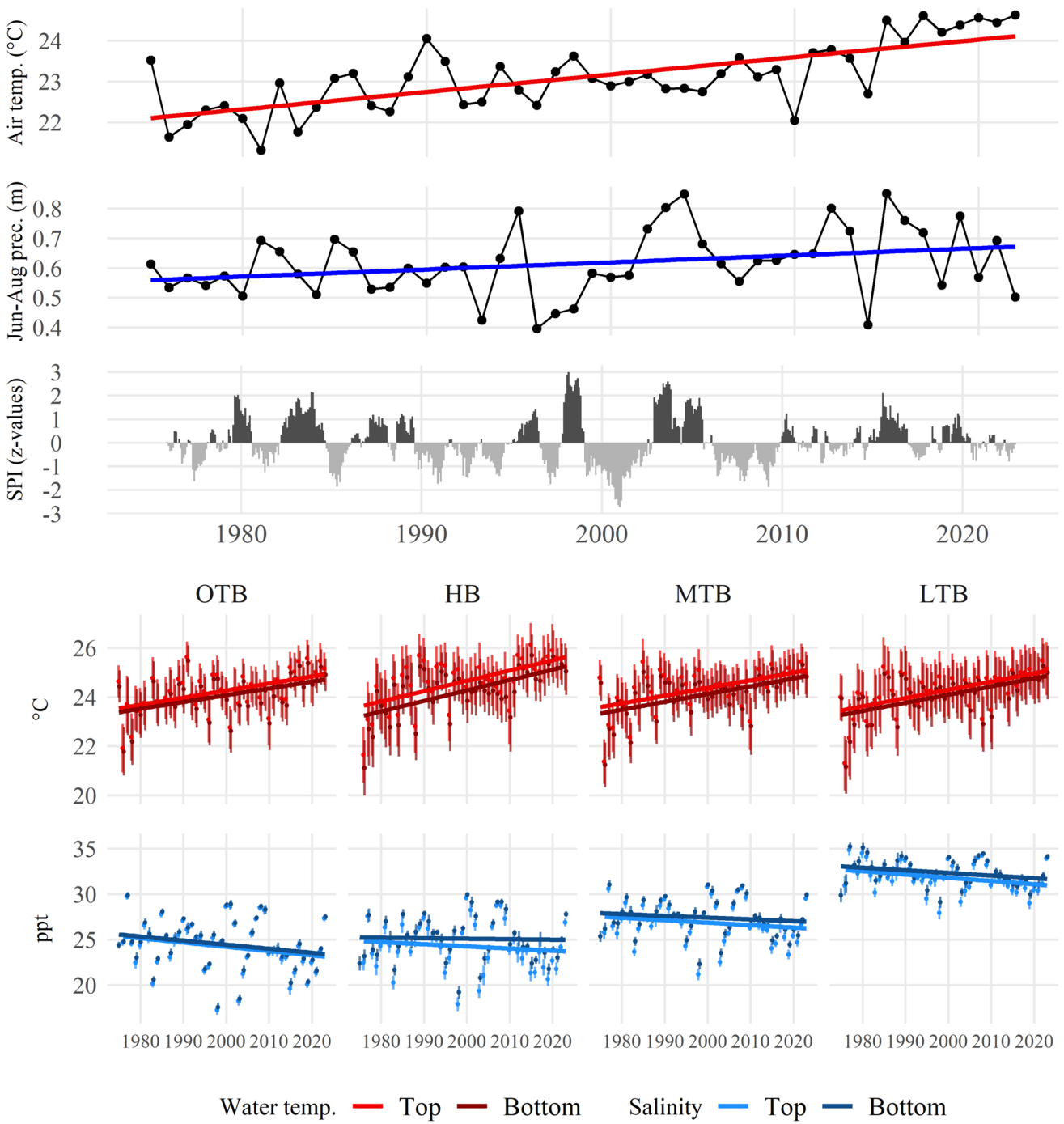


Fig. 4 Long-term air temperature, precipitation (Jun–Aug), Standard Precipitation Index (SPI), water temperature, and salinity trends from 1975 to 2022. The color shades for water temperature and salinity indicate the sampling location and the values shown are the averages (95% confidence interval) across all Environmental Protection Commission (EPC) stations in each bay segment and sampling months for each year. OTB Old Tampa Bay, HB Hillsborough Bay, MTB Middle Tampa Bay, LTB Lower Tampa Bay

both temperature and salinity (Fig. 5b), although some variation was observed throughout the bay. Temperature trends were generally stronger in the fall for OTB, whereas the remaining bay segments showed the strongest trends in the winter (February). Seasonal trends in salinity showed the largest decreases in the fall following the rainy season, with trends being especially strong in OTB. Small increases in salinity were observed in the spring for all but LTB.

Table 2 Bottom temperature trends (°C) by bay segments and datasets. Change per year is the model slope with standard error in parentheses. The start year describes the range of the trend test to 2022 (1975, $n=48$; 1996, $n=27$; 2004, $n=19$). The starting value is the estimated temperature at the start year and the end value is the estimated temperature in 2022. Total change is the difference between the two. The datasets evaluated were from the Environmental Protection Commission of Hillsborough County (EPC), Fisheries Independent Monitoring (FIM), and Pinellas County Division of Environmental Management (PDEM). Note that PDEM includes data only for the western portion of Old Tampa Bay from 2004 to present and EPC is the only dataset beginning prior to 1996. OTB Old Tampa Bay, HB Hillsborough Bay, MTB Middle Tampa Bay, LTB Lower Tampa Bay

Start year	Bay segment	Dataset	Change/year	Start value	End value	Total change
1975	OTB	EPC	0.03 (0.01)	23.4	24.7	1.3
	HB	EPC	0.04 (0.01)	23.4	25.1	1.7
	MTB	EPC	0.03 (0.01)	23.3	24.8	1.5
	LTB	EPC	0.03 (0.01)	23.3	24.8	1.5
1996	OTB	EPC	0.04 (0.02)	23.6	24.8	1.2
		FIM	0.05 (0.02)	23.5	24.8	1.3
	HB	EPC	0.06 (0.01)	23.8	25.3	1.5
		FIM	0.05 (0.02)	24.0	25.3	1.3
	MTB	EPC	0.04 (0.01)	23.8	24.8	1.0
		FIM	0.03 (0.02)	24.0	24.8	0.8
	LTB	EPC	0.03 (0.01)	23.9	24.8	0.9
		FIM	0.02 (0.02)	24.1	24.6	0.5
2004	OTB	EPC	0.06 (0.02)	23.8	25.0	1.2
		FIM	0.06 (0.03)	23.8	24.9	1.1
		PDEM	0.04 (0.03)	24.4	25.2	0.8
	HB	EPC	0.09 (0.02)	23.9	25.5	1.6
		FIM	0.1 (0.04)	23.9	25.7	1.8
	MTB	EPC	0.07 (0.02)	23.8	25.0	1.2
		FIM	0.07 (0.04)	23.8	25.1	1.3
	LTB	EPC	0.06 (0.02)	23.8	25.0	1.2
		FIM	0.05 (0.03)	23.9	24.8	0.9

Stressor Metrics

Linear mixed-effects models for the EPC data showed similar trends in each bay segment for the number of days when temperature was above different thresholds. All the temperature models for each of the three thresholds (29, 30, 31 °C) showed increasing trends for each bay segment, with the largest slope of 1.5 days per year in OTB when temperature was above 29 °C (Table S3). The estimated slopes for the number of days when temperature was above 30 °C varied from less than 1 day per year in LTB to 1.1 days per year in MTB. Likewise, the mean number of days when temperature was above 30 °C at the beginning and end of the period of record were similar between bay segments, with an average increase of 48 days across the bay segments for the period of record (Table 4, Table S5, see Table S6 for 1998 to 2022). The increase in the number of days each year when temperatures were above 29 or 31 °C was similar.

The salinity models were less similar between bay segments compared to the temperature models, primarily because of the natural salinity gradient along the bay's longitudinal axis (Tables S3, S4). None of the bay segments had strong trends in the number of days per year when salinity was below 15 ppt. Both OTB and HB had models showing an increasing number of days when salinity was below 20 or 25 ppt, whereas MTB only showed an increase in the number of days when salinity was below 25 ppt. Some salinity models for LTB had decreasing trends, although the total number of days at the beginning and end of the period of record were negligible (Table S5). Overall, OTB showed the largest increase in the number of days each year when salinity was below a threshold, particularly for 25 ppt, where the change was 86 days per year from 1975 to 2022 (130 to 216 days, Table 4, Table S5, see Table S6 for 1998 to 2022). Across all bay segments, the average increase in the number of days salinity was below 25 ppt from 1975 to 2022 was 36 days, although a distinct gradient towards the mouth of the bay was observed.

The number of days when both temperature was above and salinity was below a threshold also varied by bay segment (Table S3). The number of days when temperature was above 29 °C and salinity was below 25 ppt had the largest slopes of 1.4, 1.1, and 0.7 days per year for OTB, HB, and MTB, respectively. Likewise, the average number

Table 3 Bottom salinity trends (ppt) by bay segments and datasets. Change per year is the model slope with standard error in parentheses. The start year describes the range of the trend test to 2022 (1975, $n=48$; 1996, $n=27$; 2004, $n=19$). The starting value is the estimated salinity at the start year and the end value is the estimated salinity in 2022. Total change is the difference between the two. The datasets evaluated were from the Environmental Protection Commission of Hillsborough County (EPC), Fisheries Independent Monitoring (FIM), and Pinellas County Division of Environmental Management (PDEM). Note that PDEM includes data only for the western portion of Old Tampa Bay from 2004 to present and EPC is the only dataset beginning prior to 1996. OTB Old Tampa Bay, HB Hillsborough Bay, MTB Middle Tampa Bay, LTB Lower Tampa Bay

Start year	Bay segment	Dataset	Change/year	Start value	End value	Total change
1975	OTB	EPC	−0.06 (0.03)	25.7	23.1	−2.6
	HB	EPC	−0.01 (0.02)	25.4	24.7	−0.7
	MTB	EPC	−0.03 (0.02)	28.0	26.7	−1.3
	LTB	EPC	−0.04 (0.01)	33.1	31.5	−1.6
1996	OTB	EPC	−0.08 (0.07)	24.9	22.8	−2.1
		FIM	−0.05 (0.08)	24.1	22.7	−1.4
	HB	EPC	−0.08 (0.07)	26.1	23.9	−2.2
		FIM	0 (0.08)	24.0	23.9	−0.1
	MTB	EPC	−0.07 (0.05)	28.0	26.3	−1.7
		FIM	−0.03 (0.06)	26.6	25.8	−0.8
	LTB	EPC	−0.04 (0.03)	32.5	31.4	−1.1
		FIM	0.01 (0.04)	31.6	32.0	0.4
2004	OTB	EPC	−0.2 (0.09)	25.5	22.0	−3.5
		FIM	−0.15 (0.09)	24.6	22.0	−2.6
		PDEM	−0.19 (0.1)	24.3	20.9	−3.4
	HB	EPC	−0.19 (0.08)	26.5	23.1	−3.4
		FIM	−0.08 (0.1)	24.7	23.4	−1.3
	MTB	EPC	−0.17 (0.07)	28.6	25.6	−3.0
		FIM	−0.11 (0.08)	27.2	25.3	−1.9
	LTB	EPC	−0.12 (0.04)	33.0	30.9	−2.1
		FIM	−0.05 (0.05)	32.5	31.5	−1.0

of days when both temperature was above 29 °C and salinity was below 25 ppt from the beginning to the end of the period showed the greatest increase for OTB of 68 days (Table S5; 4 to 72 days per year from 1975 to 2022).

Figure 6 provides visual examples of the mixed-effects models for the estimated number of days over time for each bay segment from 1975 to present when temperature was above 30°, salinity was below 25 ppt, and when both occurred (see Fig. S6 for 1998 to 2022). Temperature trends were similar between segments, whereas the number of days when salinity was below the threshold decreased with proximity to the Gulf of Mexico (Table 4). The number of days when both temperature was above and salinity was below the threshold generally followed the trends for the number of days when salinity was below the threshold. These thresholds were used for comparison to seagrass changes described below, based primarily on the statistical strength of the trends and the variance of counts across stations within each bay segment (points in Fig. 6). That is, more restrictive thresholds did not provide sufficient counts of days per year to more rigorously develop seagrass response models and the chosen thresholds were based primarily on statistical considerations.

Seagrass Response

GAMs to assess the potential effects of year, light attenuation, temperature, and salinity on seagrass change provided some evidence that seagrass change was influenced by climate stressors, particularly for the FIM and PDEM datasets. The seagrass cover data ($n=75$, Adj. $R^2 = 0.90$, Deviance explained = 94%, EPC model) had notable associations with year for all bay segments (OTB, $p<0.001$; HB, $p=0.004$; MTB, $p<0.001$) that followed long-term changes in Fig. 2. Associations with light attenuation were also observed for the HB ($p<0.001$) and MTB ($p=0.004$) bay segments, although the relationship in HB was flat beyond 0.75 m^{-1} and opposite as expected for MTB (Fig. 7, Table S7). As noted above, light attenuation in HB and MTB in recent years is well within the range supportive of seagrass growth (Fig. 2c), such that these associations may not be describing meaningful biological relationships. Associations of

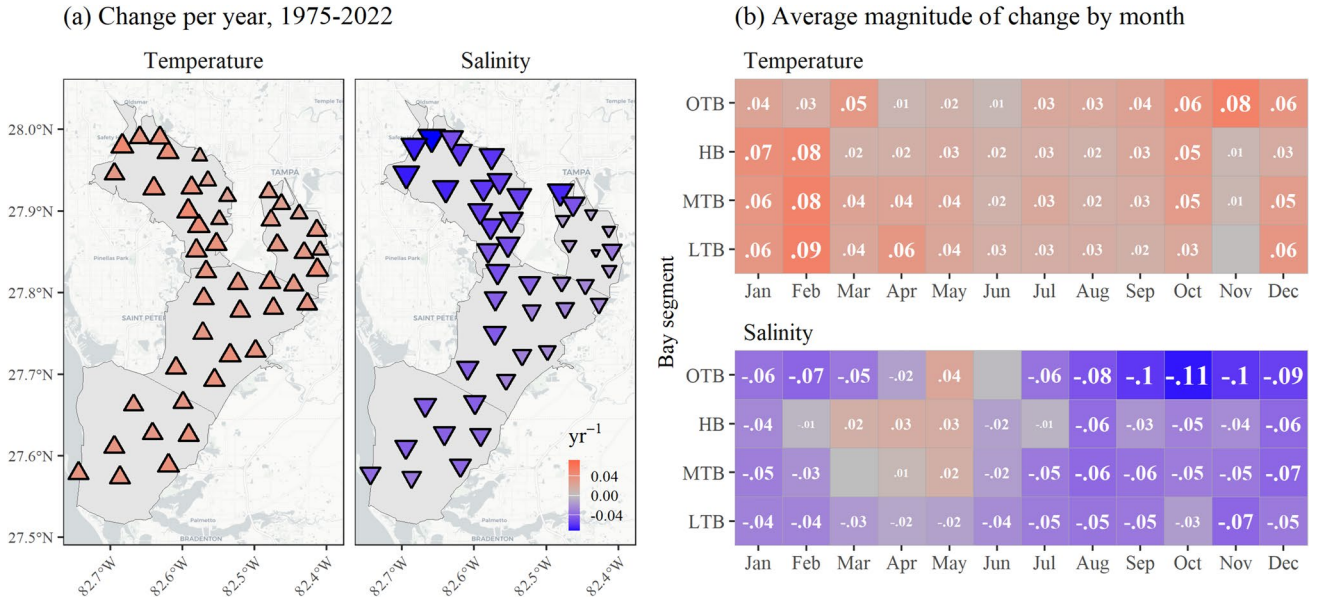


Fig. 5 Trends from 1975 to 2022 for bottom water temperature and salinity measurements at long-term monitoring stations in Tampa Bay. Results for seasonal Kendall tests by station are shown in **a** with color, size, and shape corresponding to the estimated annual slope as change per year (yr^{-1}). Summarized seasonal trends by month are shown as **b** the average magnitude of change (slope) for stations in each bay segment for temperature and salinity, indicated by color and text scaled by absolute magnitude. Bay segment outlines are shown in **a**; OTB Old Tampa Bay, HB Hillsborough Bay, MTB Middle Tampa Bay, LTB Lower Tampa Bay

seagrass change with the temperature (OTB, $p = 0.129$; HB, $p = 0.428$; MTB, $p = 0.346$) and salinity (OTB, $p = 0.040$; HB, $p = 0.959$; MTB, $p = 0.186$) metrics were not present or contrary to expectation (e.g., increase in seagrass with an increasing salinity metric in OTB, slight increase in seagrass cover with an increasing temperature metric in all bay segments). Similarly, the EPC model using the "both" stressor metric ($n = 75$, Adj. $R^2 = 0.90$, Deviance explained = 94%, Fig. S7, Table S8) had positive associations of the "both" metric with seagrass change in OTB ($p = 0.006$) and HB ($p = 0.004$).

Table 4 Summary of mixed-effects models evaluating increases in the number of days each year from 1975 to 2022 when the bottom temperature was above 30 °C, bottom salinity was below 25 ppt, or both temperature and salinity were above/below the thresholds. The start and end columns show the estimated number of days (mean, standard error) in 1975 and 2022 when temperature or salinity were above or below the thresholds. OTB Old Tampa Bay, HB Hillsborough Bay, MTB Middle Tampa Bay, LTB Lower Tampa Bay

Bay segment	Threshold	Slope	Start	End
OTB	Temperature > 30	1.05	7 (3.3)	57 (3.2)
	Salinity < 25	1.83	130 (13)	216 (13)
	Both	0.81	−1 (2.7)	37 (2.7)
HB	Temperature > 30	0.99	8 (4.9)	55 (4.8)
	Salinity < 25	0.8	134 (12.2)	171 (11.5)
	Both	0.76	−1 (2.9)	35 (2.7)
MTB	Temperature > 30	1.06	8 (3.1)	58 (3.1)
	Salinity < 25	0.53	56 (11.8)	81 (11.8)
	Both	0.37	−1 (1.7)	16 (1.7)
LTB	Temperature > 30	0.97	19 (3.4)	64 (3.4)
	Salinity < 25	−0.09	4 (1.2)	0 (1.2)
	Both	-	-	-

The GAMs for the FIM and PDEM datasets had notable associations with temperature and salinity stress, where an increase in temperature and a decrease in salinity were often associated with a decline in seagrass depending on bay segment. The FIM model ($n = 81$, Adj. $R^2 = 0.23$, Deviance explained = 35%, Fig. 8a, Table S9) showed a slight decrease in seagrass with increasing temperature in MTB ($p = 0.022$) and a similar change as salinity decreased in MTB ($p = 0.002$) and OTB ($p = 0.048$). The PDEM model ($n = 19$, Adj. $R^2 = 0.70$, Deviance explained = 81%, Fig. 8b, Table S10) that evaluated only OTB showed a slight decrease with seagrass as temperature increased ($p = 0.072$), but no association with salinity.

Discussion

Global increases in temperature and altered precipitation patterns related to climate change have had measurable effects on the structure and functioning of a wide range of natural environments (Osland et al. 2015; Oliver et al. 2018). For Tampa Bay, these changes have been demonstrated using long-term trends in water temperature and salinity, which

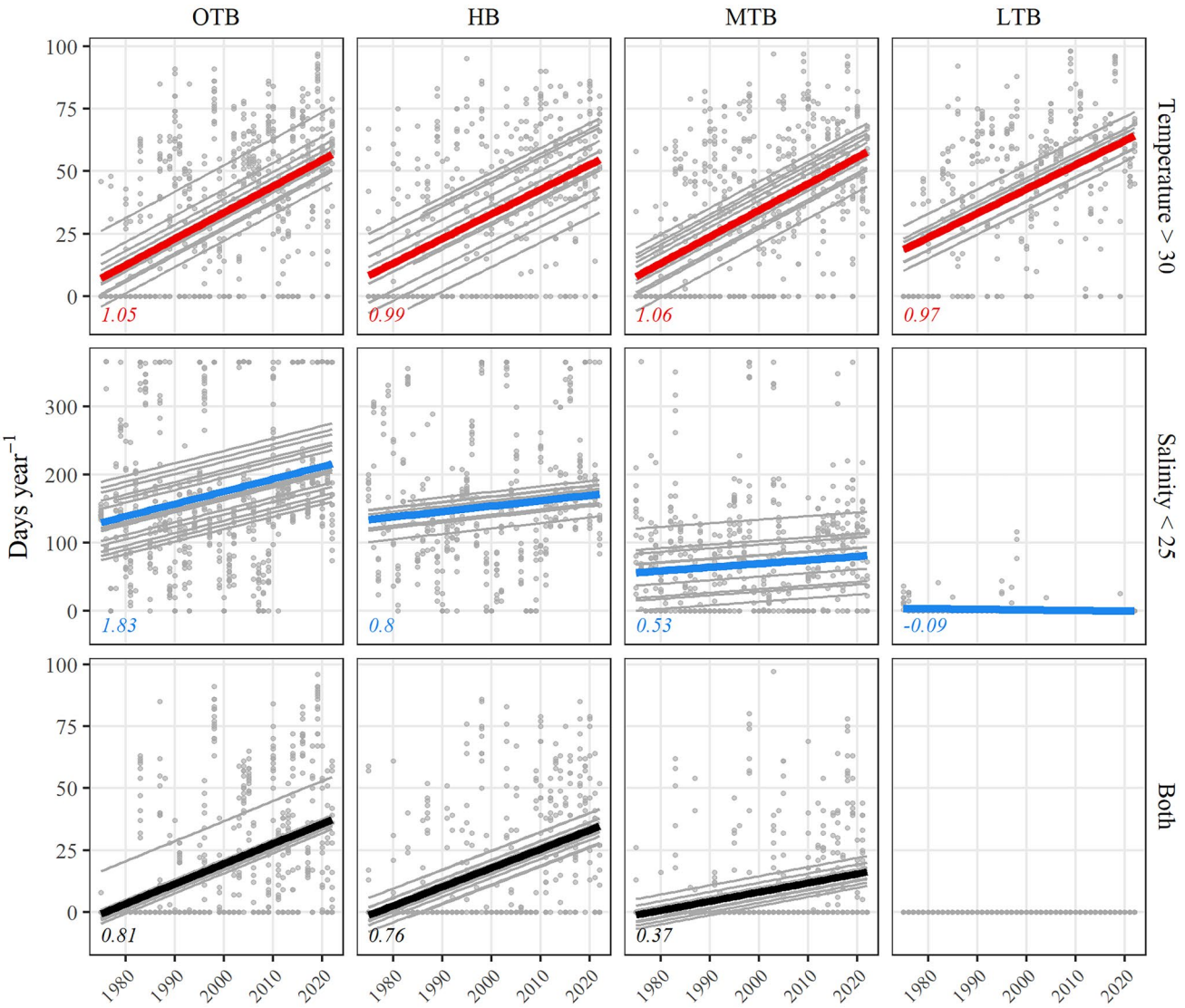


Fig. 6 Example of mixed effects models for the estimated number of days per year that bottom temperature (red) or salinity (blue) were above or below thresholds of 30 °C or 25 ppt, respectively, from the EPC data. The bottom row (black) shows the number of days when both temperature was above and salinity was below the thresholds. The models included station as a random effect for each bay segment, with gray lines indicating individual station trends, gray points as the actual number of days for each station, and thicker lines indicating the overall model fit. Slopes are shown in the bottom left of each facet. OTB Old Tampa Bay, HB Hillsborough Bay, MTB Middle Tampa Bay, LTB Lower Tampa Bay

mirrored long-term changes in air temperature and precipitation. Tampa Bay has gotten hotter and fresher; water temperature has increased by 0.03–0.04 °C per year and salinity has decreased by 0.04–0.06 ppt per year, translating to an increase of 1.3 to 1.7 °C and a decrease of 1.6 to 2.6 ppt over the past 50 years. The number of days each year when temperature was above 30 °C or salinity was below 25 ppt has also increased consistently, with an average increase baywide of 48 and 37 days, respectively, since 1975. These changes were demonstrated in three long-term datasets with different sampling methods and periods of record. Understandably, the trends were most clearly observed in the dataset with the longest period of record (EPC), covering nearly 50 years of monthly observations. These long-term changes manifested into consistent trends in known seagrass stressors; the continuous number of days increased each year when temperature, salinity, or both crossed thresholds.

Similar regional, long-term changes in coastal waters and estuaries have been observed by others (Carlson et al. 2018; Nickerson et al. 2023; Shi and Hu in review). Nickerson et al. (2023) evaluated sea surface trends at a larger spatial scale for Tampa Bay, the West Florida Continental Shelf, and the adjacent Gulf of Mexico. Temperature trends were similar to those herein for Tampa Bay (the EPC dataset was

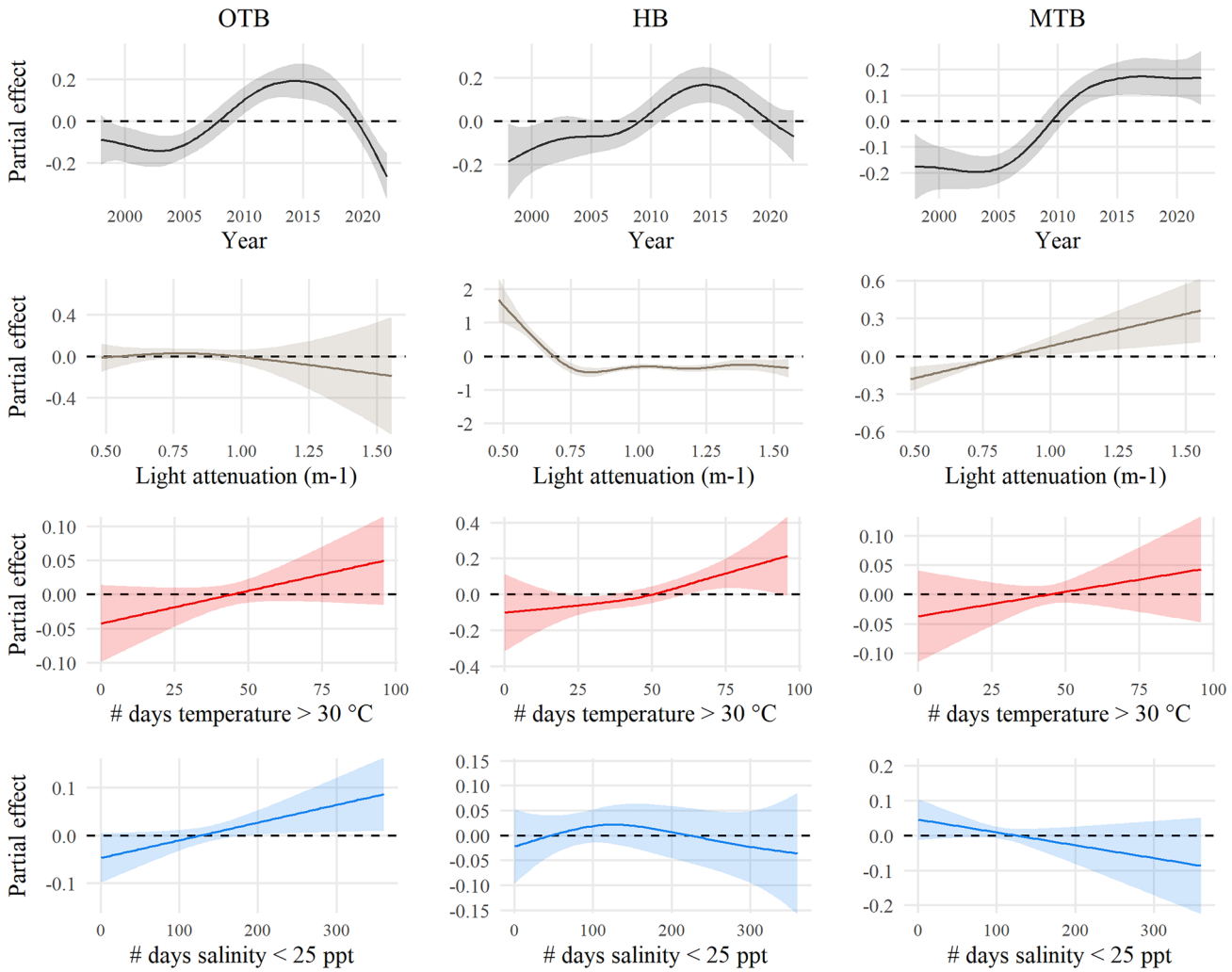


Fig. 7 Partial effects of smoothers (rows) by bay segment (columns) from a Generalized Additive Model used to describe seagrass change relative to year, light attenuation, the number of days each year when bottom temperature was above 30 °C, and the number of days each year when bottom salinity was below 25 ppt. The EPC data were used for the independent variables in the model. Partial effects describe the modeled association between each predictor and seagrass frequency occurrence after accounting for the effects of the other predictors. See Table S7 for additional model fit statistics. OTB Old Tampa Bay, HB Hillsborough Bay, MTB Middle Tampa Bay. $n = 75$, Adj. $R^2 = 0.90$, Deviance explained = 94%

also used). Nickerson et al. (2023) also noted that temperature increases in Tampa Bay were most pronounced in the winter, although they rightfully acknowledge the sensitivity of their results to conditions at the start and end of the time series. Our assessment evaluated non-parametric trends (i.e., Kendall tests, less sensitive to outliers) at individual EPC stations and bay segments, whereas Nickerson et al. (2023) evaluated the EPC data as an average for the entire bay for consistency of comparison to their larger spatial area and model domain. Our results showing increases in temperature and decreases in salinity in the fall, winter period provided a finer-scale comparison, where trends were most notable for OTB and northern stations of HB (Fig. 5)—likely related to hydrodynamic characteristics of these segments relative to MTB and LTB that flush more regularly with the Gulf of Mexico. These upper bay segments are more affected by hydrologic inflows (HB), lack of circulation (OTB), or thermal stress related to more rapid warming with shallower depths. The significant reduction in salinity for LTB is also of note, perhaps related to gravitational circulation patterns that export lower salinity water from upstream in the main shipping channels (Weisberg and Zheng 2006). Additionally, Shi and Hu (in review) provided a recent assessment of a 2023 heatwave in south Florida, supported by a 20-year trend assessment that suggested estuaries were warming at nearly double the rate of the Gulf of Mexico. The upper limit of our warming estimate for Tampa Bay is comparable. Notably, Carlson et al. (2018) suggest a link between

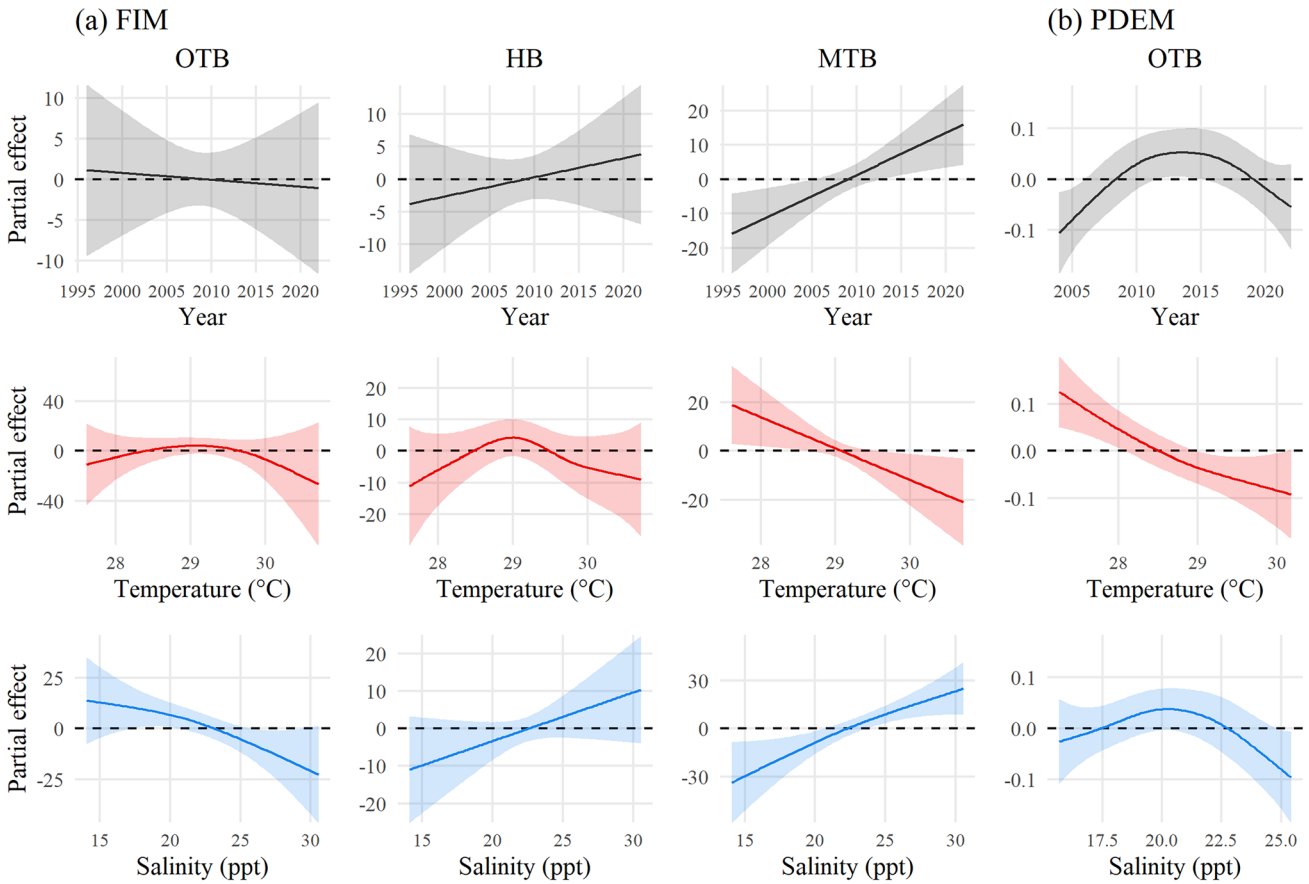


Fig. 8 Partial effects of smoothers (rows) by bay segment (columns) from Generalized Additive Models evaluating seagrass changes versus year, bottom temperature, and bottom salinity for the FIM (first three columns evaluating mean annual percent cover from 0 to 100, $n = 81$, Adj. $R^2 = 0.23$, Deviance explained = 35%) **a** and PDEM (right column evaluating annual frequency occurrence from 0 to 1 in OTB only, $n = 19$, Adj. $R^2 = 0.70$, Deviance explained = 81%) **b** datasets. Partial effects describe the modeled association between each predictor and seagrass change after accounting for the effects of the other predictors. See Tables S9 and S10 for additional model fit statistics. OTB Old Tampa Bay, HB Hillsborough Bay, MTB Middle Tampa Bay

historical seagrass losses in Florida Bay and rapid warming in shallow areas with low surface reflectance.

Our relatively simple modeling approach provided some evidence that climate-related stressors impart some effect on recent seagrass losses in Tampa Bay. The models did not provide a consistent explanation that increasing temperature and decreasing salinity were key (or the sole) drivers. However, evaluating all models together as weight-of-evidence suggests there is value in considering multiple datasets and models to interpret noisy patterns and compounding ecological processes. Model results for the FIM and PDEM datasets both suggested that increasing temperature and decreasing salinity were associated with potential seagrass loss in recent years (Fig. 8). For the EPC model, the salinity metric was positively associated with seagrass in OTB and all segments showed a slight increase in seagrass with increases in the temperature metric. The positive association of seagrass change with the temperature (or "both") metric could be a signal of increased vegetative growth with temperature anomalies (e.g., Wong and Dowd 2023), whereas the positive association with the salinity metric in OTB is not easily explained since extreme precipitation events have been linked to seagrass loss in Tampa Bay (Greening and Janicki 2006). An important distinction between the EPC and other models is that the former evaluated the number of days above/below thresholds each year to quantify annual temperature or salinity stress, whereas the latter evaluated observed temperature and salinity values at the time of seagrass sampling. Additionally, the EPC models evaluated water quality changes from data collected at relatively deeper locations than where seagrass typically grows (Fig. 1b), suggesting a potential disconnect in relating the two. As such, both models attempted to describe the role of these stressors on potential seagrass change but used different independent variables given the different sampling designs of each monitoring program. These differences highlight challenges describing autecological relationships in long-term datasets, while also demonstrating the utility of our weight-of-evidence approach to describe such relationships.

An additional caveat of our models was the use of "thresholds" to define potential stressor metrics for temperature and salinity on an annual time scale. Our choice to use 30 °C and 25 ppt for temperature and salinity was primarily a statistical consideration given a consistent increase over time in the number of days when these thresholds were crossed. That is, sufficient change and variation in the independent variables for the models of seagrass change were needed to statistically describe potential relationships. The reported threshold values in tropical and sub-tropical environments suggest that the limits of the ecological niche for seagrasses are higher for temperature and lower for salinity (Phillips 1960; McMillan and Moseley 1967; Zieman 1975; Lirman and Cropper 2003). Because we did not see a dramatic increase in the number of days each year when the thresholds were crossed at more stressful values, conditions in Tampa Bay in recent years are likely suboptimal but within the ecological niche for seagrasses. This is especially true for *H. wrightii* that had the greatest changes over the period of record and is tolerant of a wide range of salinity. However, this does not suggest that these factors are unimportant, both currently and in the future. Extreme temperature or precipitation events acting individually or in combination are likely captured by the trends in stressor metrics using these lower thresholds, i.e., an increase in a bay segment median number of days also suggests extremes are increasing given the variation around these summary metrics (Fig. 6). Without more continuous, diel observations of these metrics over the period of record, we were heavily reliant on these model outputs to determine relevant thresholds for Tampa Bay seagrass. This further highlights a long-standing data gap and the need for the estuary. Regardless, our thresholds seem to be indicative of the potential for chronic sublethal effects of stress on seagrasses, reducing their resilience to other stressors. Further, our models suggested that temperature and salinity changes are at least associated with seagrass loss and, if so, long-term trends in both are set to amplify their effect in the future.

Additional limitations of our models may relate to an incomplete description of factors influencing seagrass growth, such as the inclusion of additional drivers and an incomplete or overly simplified causal network. For the former, the primary management paradigm in Tampa Bay for the past three decades has relied on the role of external nitrogen inputs in affecting light environments for seagrass growth (Greening et al. 2014; Sherwood et al. 2017). Our inclusion of light attenuation in the EPC models was meant to account for how the light environment may be influencing seagrass growth, in addition to climate-related stressors. However, light attenuation has improved over the period of record and is currently within the limits estimated to be supportive of seagrass growth in Tampa Bay (Janicki and Wade 1996; Greening et al. 2011), particularly in OTB where the most loss occurred (Fig. 2c). Additional water quality parameters could be included in the models to provide further evidence that light-limitation is not the present driver for seagrass change (e.g., nitrogen loading, chlorophyll-a, color), although nutrient management for the benefit of seagrass growth will likely continue to be a dominant management paradigm for Tampa Bay. A final consideration for our models relates to how seagrasses may influence their environment, particularly for the PDEM and FIM datasets where temperature and salinity were measured at the same locations as seagrass. For example, temperature may simply be lower in locations where seagrasses are present and can absorb solar radiation, i.e., seagrasses may be influencing their environment rather than the environment influencing seagrasses (Carlson et al. 2018). This explanation cannot be ruled out with the existing datasets, although the trend analyses and models suggest that climate-related stressors are a more likely scenario. This is especially true for water temperature trends captured by the EPC dataset which includes deeper, fixed sites adjacent to shallow seagrass flats.

The seagrass loss in Tampa Bay since 2016 is a notable phenomenon that is not limited to our study area (Lizcano-Sandoval et al. 2022). Losses have been observed throughout southwest Florida during this time period, including Sarasota Bay directly south of Tampa Bay and Charlotte Harbor further south (Tomasko et al. 2020). These regional losses suggest that large-scale stressors are driving these changes, supporting our initial hypothesis that climate-related stressors could partially explain the change in Tampa Bay. Based on our results, the losses elsewhere may potentially be explained by temperature and salinity and are worth exploring in other southwest Florida coastal regions where long-term datasets exist (Tomasko et al. 2005). Additional factors that could explain these changes are also likely co-occurring with climate stress, some of which are unique to Tampa Bay and others that are more likely pervasive. For Tampa Bay, annual summer/fall blooms of the toxic dinoflagellate *Pyrodinium bahamense* have occurred in OTB since 2008 (Usup et al. 1994; Lopez et al. 2023) and the specific relationships of these blooms with seagrass change is unclear, although the expectation is that seagrass growth may be limited by the degradation of the light environment with algal growth. These blooms are exacerbated by the hydrologic conditions in OTB that contribute to relatively longer water residence times (Phlips et al. 2006; Lopez et al. 2021). The effect of warming temperature and decreasing salinity in OTB will further complicate the understanding of how these blooms manifest and persist each year (Koch et al. 2007; Stelling et al. 2023), and ultimately contribute to changes in the light environment affecting seagrass resources in this bay segment.

Additional biotic factors could be influencing regional patterns in seagrass growth. In Tampa Bay and elsewhere, enhanced macroalgal production has been a recent concern

(Hall et al. 2022; Janicki Environmental, Inc. 2022; Brewton and Lapointe 2023; Scolaro et al. 2023). Attached macroalgae abundance has increased over time and has been observed to colonize locations where seagrass was formerly present in Tampa Bay (Beck 2020b). Competitive differences between seagrasses and macroalgae are poorly understood in these systems (but see Bell and Hall 1997; Taplin et al. 2005; Brewton and Lapointe 2023), in addition to insufficient macroalgae data in Tampa Bay that cannot clearly describe seasonal growth, distribution patterns, and nutrient cycling. Discrete pollutant loading events in Tampa Bay have been documented to promote both phytoplankton and macroalgae growth (Beck et al. 2022; Scolaro et al. 2023; Tomasko 2023). The role that evolving nutrient loading and changing climatic conditions may have on Tampa Bay's primary producers—particularly algal and seagrass growth and interactions in recent years—is not well understood. Finally, additional research has focused on how diseases and pathogens can influence seagrass growth patterns in Florida (Robblee et al. 1991; Van Bogaert et al. 2018; Duffin et al. 2021). For example, the parasitic slime mold *Labryinthula* spp. that causes seagrass wasting disease has been known to infect *T. testudinum* in Tampa Bay (Blakesley et al. 2001), although it is unclear if these infections have had large-scale, population-level effects. Existing research has primarily focused on describing spatial patterns, past die-off events, or immunology of these pathogens (Robblee et al. 1991; Duffin et al. 2021). More research should be directed towards the influence of climate stressors on seagrass pathogen vulnerability.

Lastly, our result showing that salinity has decreased in Tampa Bay is contrary to expectations for how sea-level rise will affect coastal systems (Costa et al. 2023; Alarcon et al. 2024), as salinity increases with sea-level rise has already caused numerous alterations of subtidal and nearshore habitats (Brinson et al. 1995; White and Kaplan 2017). In southwest Florida, the most common ecological example is the upland expansion of mangroves in response to increased porewater salinity and water levels over the past few decades (Borchert et al. 2018). Alteration of salinity regimes for surface and groundwater resources have been well documented. In Florida Bay, for example, widespread decline of *T. testudinum* has been attributed to altered hydrology and drought-induced hypersaline conditions, and sea level rise is expected to further modify salinity dynamics in the region (Hall et al. 2016). However, elevated summertime temperatures were also implicated in the decline. Dessu et al. (2018) noted that sea level rise is expected to have the largest effect on salinity changes during periods of low freshwater outflow from the Florida Everglades, emphasizing that measured salinity represents the relative contributions of oceanic and freshwater inflows. In Tampa Bay, the long-term trends of decreasing salinity, especially in the upper bay segments, suggest that the hydrologic loading has had a greater influence on salinity regimes than the effects of sea-level rise. This hypothesis is supported by our assessment of precipitation patterns over time, where the long-term increase is inversely associated with the decrease in salinity.

Conclusions

This study provided a detailed assessment of long-term water temperature and salinity changes in Tampa Bay supported by datasets from three long-term monitoring programs of different lengths and sampling designs. An evaluation of each dataset showed a clear pattern of increasing temperature and decreasing salinity mirrored by long-term changes in air temperature and precipitation, suggesting that Tampa Bay has become hotter and fresher with the trends likely continuing in the future. GAMs provided partially supporting evidence that these changes can be linked to recent seagrass losses. Future analyses may show stronger associations between physicochemical habitat conditions and seagrass change as the trends are very likely to continue to push seagrasses further outside of their tolerance ranges. These analyses should be supported by additional data collection efforts, particularly high-resolution continuous monitoring data that provide a more precise assessment of diurnal stress across multiple time scales. Ongoing work in OTB using continuous data loggers in shallow areas where seagrass has been gained or lost will provide insights into short-term diurnal changes as potential acute temperature stress (see https://tbep-tech.github.io/otb-temp/tempeval). Morphological or physiological measurements at the individual level could also provide early indications of heat and osmotic stress (Congdon et al. 2023).

Natural resource managers should consider how these climate-related stressors may alter the effectiveness of intervention activities aimed at protecting ecological resources in Tampa Bay. Management actions that have historically been effective may become less so, resulting in diminished ecosystem resilience compounded by climate change. For example, nitrogen load reductions have historically been the most effective strategy to restore seagrass in Tampa Bay (Greening and Janicki 2006; Greening et al. 2014). As Tampa Bay becomes hotter and fresher, current nutrient load targets may no longer be effective, resulting in further shifts in algal and seagrass ecology dynamics. Strategies that mimic or restore pre-development hydrology or that further reduce allowable load inputs from regulated entities (e.g., additional stormwater controls, hydrological modifications) may be needed to confer additional resilience and adaptive capacity for seagrasses responding to climatic changes in Tampa Bay. These considerations are especially critical for upper Tampa Bay where a majority of seagrass loss has occurred and where temperature and salinity trends appear most pronounced. Reversal of recent

trends may be more likely to occur if aggressive actions and controls are pursued sooner rather than later, especially since the challenges of restoring these long-lived foundation species once lost will be exacerbated by ongoing development in the watershed and the current climate trajectory.

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Data Availability All data and analysis code for this manuscript is available on GitHub at https://github.com/tbep-tech/temp-manu. A preprint of an earlier version of this manuscript is available at https://doi.org/10.21203/rs.3.rs-3946855/v1.

Declarations

Competing Interests The authors declare no competing interests.

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References

Alarcon, V.J., A.C. Linhoss, C.R. Kelble, P.F. Mickle, A. Fine, and E. Montes. 2024. Potential challenges for the restoration of Biscayne Bay (Florida, USA) in the face of climate change effects revealed with predictive models. *Ocean & Coastal Management* 247: 106929. https://doi.org/10.1016/j.ocecoaman.2023.106929.

Araújo, M.B., and A. Guisan. 2006. Five (or so) challenges for species distribution modelling. *Journal of Biogeography* 33: 1677–1688. https://doi.org/10.1111/j.1365-2699.2006.01584.x.

Bartenfelder, A., W.J. Kenworthy, B. Puckett, C. Deaton, and J.C. Jarvis. 2022. The abundance and persistence of temperate and tropical seagrasses at their edge-of-range in the western Atlantic Ocean. *Frontiers in Marine Science* 9: 917237. https://doi.org/10.3389/fmars.2022.917237.

Bates, D., M. Mächler, B. Bolker, and S. Walker. 2015. Fitting linear mixed-effects models using lme4. *Journal of Statistical Software* 67: 1–48. https://doi.org/10.18637/jss.v067.i01.

Beck, M. W. 2020a. *tbep-tech/wq-dash: v1.0 (version v1.0)*. Zenodo. https://doi.org/10.5281/zenodo.3648664.

Beck, M. W. 2020b. *tbep-tech/seagrasstransect-dash: v1.0 (version v1.0)*. Zenodo. https://doi.org/10.5281/zenodo.4319936.

Beck, M.W., E.T. Sherwood, J.R. Henkel, K. Dorans, K. Ireland, and P. Varela. 2019. Assessment of the cumulative effects of restoration activities on water quality in Tampa Bay, Florida. *Estuaries and Coasts* 42: 1774–1791. https://doi.org/10.1007/s12237-019-00619-w.

Beck, M.W., M. Schrandt, M. Wessel, E.T. Sherwood, G.E. Raulerson, A. Prasad, and B. Best. 2021. tbeptools: An R package for synthesizing estuarine data for environmental research. *Journal of Open Source Software* 6: 3485. https://doi.org/10.21105/joss.03485.

Beck, M.W., A. Altieri, C. Angelini, M.C. Burke, J. Chen, D.W. Chin, J. Gardiner, et al. 2022. Initial estuarine response to inorganic nutrient inputs from a legacy mining facility adjacent to Tampa Bay. *Florida. Marine Pollution Bulletin* 178: 113598. https://doi.org/10.1016/j.marpolbul.2022.113598.

Beck, M.W., D.E. Robison, G.E. Raulerson, M.C. Burke, J. Saarinen, C. Sciarrino, E.T. Sherwood, and D.A. Tomasko. 2023. Addressing climate change and development pressures in an urban estuary through habitat restoration planning. *Frontiers in Ecology and Evolution* 11: 1070266. https://doi.org/10.3389/fevo.2023.1070266.

Beguería, S., and S. M. Vicente-Serrano. 2023. SPEI: Calculation of the Standardized Precipitation-Evapotranspiration Index (R package version 1.8.1).

Beguería, S., S.M. Vicente-Serrano, F. Reig, and B. Latorre. 2013. Standardized precipitation evapotranspiration index (SPEI) revisited: Parameter fitting, evapotranspiration models, tools, datasets and drought monitoring. *International Journal of Climatology* 34: 3001–3023. https://doi.org/10.1002/joc.3887.

Bell, S.S., and M.O. Hall. 1997. Drift macroalgal abundance in seagrass beds: Investigating large-scale associations with physical and biotic attributes. *Marine Ecology Progress Series* 147: 277–283. https://doi.org/10.3354/meps147277.

Blakesley, B., P. Hall, D. Berns, J. Hyniova, M. Merello, and R. Conroy. 2001. *Survey of the distribution of the marine slime mold Labyrinthula sp. in the seagrass Thalassia testudinum in the Tampa Bay area, fall 1999-fall 2000*. Technical Report 01–01. St. Petersburg, Florida: Tampa Bay Estuary Program.

Boesch, D.F., R.B. Brinsfield, and R.E. Magnien. 2001. Chesapeake bay eutrophication: Scientific understanding, ecosystem restoration, and challenges for agriculture. *Journal of Environmental Quality* 30: 303–320. https://doi.org/10.2134/jeq2001.302303x.

Borchert, S.M., M.J. Osland, N.M. Enwright, and K.T. Griffith. 2018. Coastal wetland adaptation to sea level rise: Quantifying potential for landward migration and coastal squeeze. *Journal of Applied Ecology* 55: 2876–2887. https://doi.org/10.1111/1365-2664.13169.

Brewton, R.A., and B.E. Lapointe. 2023. The green macroalga *Caulerpa prolifera* replaces seagrass in a nitrogen enriched, phosphorus limited, urbanized estuary. *Ecological Indicators* 156: 111035. https://doi.org/10.1016/j.ecolind.2023.111035.

Brinson, M.M., R.R. Christian, and L.K. Blum. 1995. Multiple states in the sea-level induced transition from terrestrial forest to estuary. *Estuaries* 18: 648–659. https://doi.org/10.2307/1352383.

Burkholder, J.M., D.A. Tomasko, and B.W. Touchette. 2007. Seagrasses and eutrophication. *Journal of Experimental Marine Biology and Ecology* 350: 46–72. https://doi.org/10.1016/j.jembe.2007.06.024.

Carlson, D.F., L.A. Yarbro, S. Scolaro, M. Poniatowski, V. McGee-Absten, and P.R. Carlson. 2018. Sea surface temperatures and seagrass mortality in Florida Bay: Spatial and temporal patterns discerned from MODIS and AVHRR data. *Remote Sensing of Environment* 208: 171–188. https://doi.org/10.1016/j.rse.2018.02.014.

Chamberlain, S., and D. Hocking. 2023. rnoaa: NOAA weather data from R (R package version 1.4.0).

Congdon, V.M., M.O. Hall, B.T. Furman, J.E. Campbell, M.J. Durako, K.L. Goodin, and K.H. Dunton. 2023. Common ecological indicators identify changes in seagrass condition following disturbances in the Gulf of Mexico. *Ecological Indicators* 156: 111090. https://doi.org/10.1016/j.ecolind.2023.111090.

Costa, Y., I. Martins, G.C. de Carvalho, and F. Barros. 2023. Trends of sea-level rise effects on estuaries and estimates of future saline intrusion. *Ocean & Coastal Management* 236: 106490. https://doi.org/10.1016/j.ocecoaman.2023.106490.

Dessu, S.B., R.M. Price, T.G. Troxler, and J.S. Kominoski. 2018. Effects of sea-level rise and freshwater management on long-term water levels and water quality in the Florida Coastal Everglades. *Journal of Environmental Management* 211: 164–176. https://doi.org/10.1016/j.jenvman.2018.01.025.

Duarte, C.M. 1995. Submerged aquatic vegetation in relation to different nutrient regimes. *Ophelia* 41: 87–112. https://doi.org/10.1080/00785236.1995.10422039.

Duarte, C.M., W.C. Dennison, R.J.W. Orth, and T.J.B. Carruthers. 2008. The charisma of coastal ecosystems: Addressing the imbalance. *Estuaries and Coasts* 31: 233–238. https://doi.org/10.1007/s12237-008-9038-7.

Duffin, P., D.L. Martin, B.T. Furman, and C. Ross. 2021. Spatial patterns of Thalassia testudinum immune status and Labyrinthula spp. load implicate environmental quality and history as modulators of defense strategies and wasting disease in Florida Bay, United States. *Frontiers in Plant Science* 12: 612947. https://doi.org/10.3389/fpls.2021.612947.

Dunic, J.C., and I.M. Côté. 2023. Management thresholds shift under the influence of multiple stressors: Eelgrass meadows as a case study. *Conservation Letters* 16: e12938. https://doi.org/10.1111/conl.12938.

Dunic, J.C., C.J. Brown, R.M. Connolly, M.P. Turschwell, and I.M. Côté. 2021. Long-term declines and recovery of meadow area across the world's seagrass bioregions. *Global Change Biology* 27: 4096–4109. https://doi.org/10.1111/gcb.15684.

Fourqurean, J.W., S. Manuel, K.A. Coates, W.J. Kenworthy, and S.R. Smith. 2010. Effects of excluding sea turtle herbivores from a seagrass bed: Overgrazing may have led to loss of seagrass meadows in Bermuda. *Marine Ecology Progress Series* 419: 223–232. https://doi.org/10.3354/meps08853.

Fourqurean, J.W., C.M. Duarte, H. Kennedy, N. Marbà, M. Holmer, M.A. Mateo, E.T. Apostolaki, et al. 2012. Seagrass ecosystems as a globally significant carbon stock. *Nature Geoscience* 5: 505–509. https://doi.org/10.1038/ngeo1477.

Garcia, L., C. J. Anastasiou, and D. Robison. 2023. *Tampa Bay Surface Water Improvement and Management (SWIM) Plan*. Brooksville, Florida: Southwest Florida Water Management District.

Garrett, M., J. Wolny, E. Truby, C. Heil, and C. Kovach. 2011. Harmful algal bloom species and phosphate-processing effluent: Field and laboratory studies. *Marine Pollution Bulletin* 62: 596–601. https://doi.org/10.1016/j.marpolbul.2010.11.017.

Greening, H.S., and A.J. Janicki. 2006. Toward reversal of eutrophic conditions in a subtropical estuary: Water quality and seagrass response to nitrogen loading reductions in Tampa Bay, Florida, USA. *Environmental Management* 38: 163–178. https://doi.org/10.1007/s00267-005-0079-4.

Greening, H.S., L.M. Cross, and E.T. Sherwood. 2011. A multiscale approach to seagrass recovery in Tampa Bay, Florida. *Ecological Restoration* 29: 82–93. https://doi.org/10.3368/er.29.1-2.82.

Greening, H.S., A.J. Janicki, E.T. Sherwood, R. Pribble, and J.O.R. Johansson. 2014. Ecosystem responses to long-term nutrient management in an urban estuary: Tampa Bay, Florida, USA. *Estuarine, Coastal and Shelf Science* 151: A1–A16. https://doi.org/10.1016/j.ecss.2014.10.003.

Hall, M.O., M.J. Durako, J.W. Fourqurean, and J.C. Zieman. 1999. adal changes in seagrass distribution and abundance in Florida Bay. *Estuaries* 22: 445. https://doi.org/10.2307/1353210.

Hall, M.O., B.T. Furman, M. Merello, and M.J. Durako. 2016. Recurrence of *Thalassia testudinum* seagrass die-off in Florida Bay, USA: Initial observations. *Marine Ecology Progress Series* 560: 243–249. https://doi.org/10.3354/meps11923.

Hall, L.M., L.J. Morris, R.H. Chamberlain, M.D. Hanisak, R.W. Virnstein, R. Paperno, B. Riegl, L.R. Ellis, A. Simpson, and C.A. Jacoby. 2022. Spatiotemporal patterns in the biomass of drift macroalgae in the Indian River Lagoon, Florida. *United States. Frontiers in Marine Science* 9: 767440. https://doi.org/10.3389/fmars.2022.767440.

Hammer, K., J. Borum, H. Hasler-Sheetal, E. Shields, K. Sand-Jensen, and K. Moore. 2018. High temperatures cause reduced growth, plant death and metabolic changes in eelgrass *Zostera marina*. *Marine Ecology Progress Series* 604: 121–132. https://doi.org/10.3354/meps12740.

Han, Q., and D. Liu. 2014. Macroalgae blooms and their effects on seagrass ecosystems. *Journal of Ocean University of China* 13: 791–798. https://doi.org/10.1007/s11802-014-2471-2.

Heck, K.L., and J.F. Valentine. 2007. The primacy of top-down effects in shallow benthic ecosystems. *Estuaries and Coasts* 30: 371–381. https://doi.org/10.1007/bf02819384.

Hense, M.J.S., C.J. Patrick, R.J. Orth, D.J. Wilcox, W.C. Dennison, C. Gurbisz, M.P. Hannam, et al. 2023. Rise of Ruppia in Chesapeake Bay: Climate change-driven turnover of foundation species creates new threats and management opportunities. *Proceedings of the National Academy of Sciences*. https://doi.org/10.1073/pnas.2220678120.

Hirsch, R.M., J.R. Slack, and R.A. Smith. 1982. Techniques of trend analysis for monthly water quality data. *Water Resources Research* 18: 107–121. https://doi.org/10.1029/wr018i001p00107.

Hutchinson, G.E. 1957. Concluding remarks. *Cold Spring Harbor Symposia on Quantitative Biology* 22: 415–427. https://doi.org/10.1101/sqb.1957.022.01.039.

Janicki, A. J., and D. L. Wade. 1996. *Estimating critical nitrogen loads for the Tampa Bay Estuary: An empirically based approach to setting management targets*. Technical Report 06–96. St. Petersburg, Florida: Tampa Bay Estuary Program.

Janicki Environmental, Inc. 2022. *Identifying potential drivers of change in seagrass and algal community composition in SWFL aquatic preserves*. Charlotte Harbor; Estero Bay Aquatic Preserves.

Janicki Environmental, Inc. 2023. *Estimates of total nitrogen, total phosphorus, total suspended solids, and biological oxygen demand loadings to Tampa Bay, Florida: 2017–2021*. Technical

Report 06–23. St. Petersburg, Florida: Tampa Bay Estuary Program.

Johansson, J. O. R. 2016. *Seagrass transect monitoring in Tampa Bay: A summary of findings from 1997 through 2015*. Technical Report 08–16. St. Petersburg, Florida: Tampa Bay Estuary Program.

Johansson, J. O. R., and Janicki Environmental, Inc. 2015. *Long-term underwater light climate variation and submerged seagrass trends in Tampa Bay, Florida: With a discussion of phytoplankton and CDOM interactions*. Technical Report 06–15. St. Petersburg, Florida: Tampa Bay Estuary Program.

Koch, M.S., S.A. Schopmeyer, O.I. Nielsen, C. Kyhn-Hansen, and C.J. Madden. 2007. Conceptual model of seagrass die-off in Florida Bay: Links to biogeochemical processes. *Journal of Experimental Marine Biology and Ecology* 350: 73–88. https://doi.org/10.1016/j.jembe.2007.05.031.

Lefcheck, J.S., D.J. Wilcox, R.R. Murphy, S.R. Marion, and R.J. Orth. 2017. Multiple stressors threaten the imperiled coastal foundation species eelgrass (*Zostera marina*) in Chesapeake Bay, USA. *Global Change Biology* 23: 3474–3483. https://doi.org/10.1111/gcb.13623.

Lefcheck, J.S., R.J. Orth, W.C. Dennison, D.J. Wilcox, R.R. Murphy, J. Keisman, C. Gurbisz, et al. 2018. Long-term nutrient reductions lead to the unprecedented recovery of a temperate coastal region. *Proceedings of the National Academy of Sciences* 115: 3658–3662. https://doi.org/10.1073/pnas.1715798115.

Lewis, R.R., III., M. Durako, M. Moffler, and R. Phillips. 1985. Seagrass meadows of Tampa Bay - a review. In *Proceedings, Tampa Bay Area Scientific Information Symposium, May 1982*, ed. S.F. Treat, J.L. Simon, R.R. Lewis, and R.L. Whitman, 210–246. Tampa, Florida: Bellweather Press.

Lirman, D., and W.P. Cropper. 2003. The influence of salinity on seagrass growth, survivorship, and distribution within Biscayne Bay, Florida: Field, experimental, and modeling studies. *Estuaries* 26: 131–141. https://doi.org/10.1007/bf02691700.

Lizcano-Sandoval, L., C. Anastasiou, E. Montes, G.E. Raulerson, E.T. Sherwood, and F.E. Muller-Karger. 2022. Seagrass distribution, areal cover, and changes (1990–2021) in coastal waters off West-Central Florida, USA. *Estuarine, Coastal and Shelf Science* 279: 108134. https://doi.org/10.1016/j.ecss.2022.108134.

Lopez, C. B., S. Shankar, S. G. Kaminski, M. Garrett, and K. A. Hubbard. 2021. *Linking Pyrodinium bahamense physiology and behavior to population growth and loss in nature and implications for management*. Technical Report 07a-21. St. Petersburg, Florida: Tampa Bay Estuary Program.

Lopez, C.B., S. Shankar, S.G. Kaminski, and K.A. Hubbard. 2023. *Pyrodinium bahamense* bloom dynamics in Old Tampa Bay, FL, with a focus on Feather Sound. *Florida Scientist* 86: 62–71.

Luther, M. E., and S. D. Meyers. 2022. *Designing a genetic algorithm for the selection of causeway cut-throughs in Old Tampa Bay: Planning and prototyping*. Technical Report 08–22. St. Petersburg, Florida: Tampa Bay Estuary Program.

Madeira, D., L. Narciso, H.N. Cabral, and C. Vinagre. 2012. Thermal tolerance and potential impacts of climate change on coastal and estuarine organisms. *Journal of Sea Research* 70: 32–41. https://doi.org/10.1016/j.seares.2012.03.002.

Maxwell, P.S., J.S. Eklöf, M.M. van Katwijk, K.R. O'Brien, M. de la Torre-Castro, C. Boström, T.J. Bouma, et al. 2016. The fundamental role of ecological feedback mechanisms for the adaptive management of seagrass ecosystems-A review. *Biological Reviews* 92: 1521–1538. https://doi.org/10.1111/brv.12294.

McMillan, C., and F.N. Moseley. 1967. Salinity tolerances of five marine spermatophytes of Redfish Bay, Texas. *Ecology* 48: 503–506. https://doi.org/10.2307/1932688.

Millard, S.P. 2013. *EnvStats: An R package for environmental statistics*. New York: Springer.

Moreno-Marín, F., F.G. Brun, and M.F. Pedersen. 2018. Additive response to multiple environmental stressors in the seagrass *Zostera marina* L. *Limnology and Oceanography* 63: 1528–1544. https://doi.org/10.1002/lno.10789.

Morrison, G., E.T. Sherwood, R. Boler, and J. Barron. 2006. Variations in water clarity and chlorophylla in Tampa Bay, Florida, in response to annual rainfall, 1985–2004. *Estuaries and Coasts* 29: 926–931. https://doi.org/10.1007/bf02798652.

Nickerson, A.K., R.H. Weisberg, L. Zheng, and Y. Liu. 2023. Sea surface temperature trends for Tampa Bay, West Florida Shelf and the deep Gulf of Mexico. *Deep Sea Research Part II: Topical Studies in Oceanography* 211: 105321. https://doi.org/10.1016/j.dsr2.2023.105321.

Oliver, E.C.J., M.G. Donat, M.T. Burrows, P.J. Moore, D.A. Smale, L.V. Alexander, J.A. Benthuysen, et al. 2018. Longer and more frequent marine heatwaves over the past century. *Nature Communications*. https://doi.org/10.1038/s41467-018-03732-9.

Orth, R.J., and K.L. Heck. 2023. The dynamics of seagrass ecosystems: History, past accomplishments, and future prospects. *Estuaries and Coasts* 46: 1653–1676. https://doi.org/10.1007/s12237-023-01252-4.

Orth, R.J., T.J.B. Carruthers, W.C. Dennison, C.M. Duarte, J.W. Fourqurean, K.L. Heck, A.R. Hughes, et al. 2006. A global crisis for seagrass ecosystems. *BioScience* 56: 987–996. https://doi.org/10.1641/0006-3568(2006)56[987:agcfse]2.0.co;2.

Orth, R.J., W.C. Dennison, J.S. Lefcheck, C. Gurbisz, M. Hannam, J. Keisman, J.B. Landry, et al. 2017. Submersed aquatic vegetation in Chesapeake Bay: Sentinel species in a changing world. *BioScience* 67: 698–712. https://doi.org/10.1093/biosci/bix058.

Orth, R.J., J.S. Lefcheck, K.S. McGlathery, L. Aoki, M.W. Luckenbach, K.A. Moore, M.P.J. Oreska, R. Snyder, D.J. Wilcox, and B. Lusk. 2020. Restoration of seagrass habitat leads to rapid recovery of coastal ecosystem services. *Science Advances* 6: 1–9. https://doi.org/10.1126/sciadv.abc6434.

Osland, M.J., N.M. Enwright, R.H. Day, C.A. Gabler, C.L. Stagg, and J.B. Grace. 2015. Beyond just sea-level rise: Considering macroclimatic drivers within coastal wetland vulnerability assessments to climate change. *Global Change Biology* 22: 1–11. https://doi.org/10.1111/gcb.13084.

Phillips, R. C. 1960. *Observations on the ecology and distribution of the Florida seagrasses*. 44. Florida State Board of Conservation, Marine Laboratory.

Phillips, R.C., and E.G. Meñez. 1988. *Seagrasses*. Washington, D.C.: Smithsonian Institution Press.

Phlips, J., S. Badylak, E. Bledsoe, and M. Cichra. 2006. Factors affecting the distribution of *Pyrodinium bahamense var. bahamense* in coastal waters of Florida. *Marine Ecology Progress Series* 322: 99–115. https://doi.org/10.3354/meps322099.

Rasheed, M., and R. Unsworth. 2011. Long-term climate-associated dynamics of a tropical seagrass meadow: Implications for the future. *Marine Ecology Progress Series* 422: 93–103. https://doi.org/10.3354/meps08925.

Robblee, M.B., T. Barber, P. Carlson Jr., M. Durako, J. Fourqurean, L. Muehlstein, D. Porter, L. Yarbro, R. Zieman, and J.C. Zieman. 1991. Mass mortality of the tropical seagrass *Thalassia testudinum* in Florida Bay (USA). *Marine Ecology Progress Series* 71: 297–299.

Roca, G., T. Alcoverro, D. Krause-Jensen, T.J.S. Balsby, M.M. van Katwijk, N. Marbà, R. Santos, et al. 2016. Response of seagrass indicators to shifts in environmental stressors: A global review and management synthesis. *Ecological Indicators* 63: 310–323. https://doi.org/10.1016/j.ecolind.2015.12.007.

Schmidt, N., and M.E. Luther. 2002. ENSO impacts on salinity in Tampa Bay, Florida. *Estuaries* 25: 976–984. https://doi.org/10.1007/bf02691345.

Schrandt, M.N., T.C. MacDonald, E.T. Sherwood, and M.W. Beck. 2021. A multimetric nekton index for monitoring, managing and

communicating ecosystem health status in an urbanized Gulf of Mexico estuary. *Ecological Indicators* 123: 107310. https://doi.org/10.1016/j.ecolind.2020.107310.

Scolaro, S., M.W. Beck, M.C. Burke, G.E. Raulerson, and E.T. Sherwood. 2023. Piney Point, seagrass, and macroalgae: Impact assessment and a case for enhanced macroalgae monitoring. *Florida Scientist* 86: 339–345.

Sherwood, E.T., and H.S. Greening. 2013. Potential impacts and management implications of climate change on Tampa Bay estuary critical coastal habitats. *Environmental Management* 53: 401–415. https://doi.org/10.1007/s00267-013-0179-5.

Sherwood, E. T., H. S. Greening, L. Garcia, K. Kaufman, A. J. Janicki, R. Pribble, B. Cunningham, et al. 2015. *Development of an integrated ecosystem model to determine effectiveness of potential watershed management projects on improving Old Tampa Bay*. Technical Report 10–15. St. Petersburg, Florida: Tampa Bay Estuary Program.

Sherwood, E.T., H.S. Greening, A.J. Janicki, and D.J. Karlen. 2016. Tampa Bay estuary: Monitoring long-term recovery through regional partnerships. *Regional Studies in Marine Science* 4: 1–11. https://doi.org/10.1016/j.rsma.2015.05.005.

Sherwood, E.T., H.S. Greening, J.O.R. Johansson, K. Kaufman, and G.E. Raulerson. 2017. Tampa Bay (Florida, USA): Documenting seagrass recovery since the 1980's and reviewing the benefits. *Southeastern Geographer* 57: 294–319. https://doi.org/10.1353/sgo.2017.0026.

Shi, J., and C. Hu. in review. Response of South Florida estuaries to the 2023 heatwave. *Estuaries and Coasts*. https://doi.org/10.21203/rs.3.rs-3528678/v1.

Statham, P.J. 2012. Nutrients in estuaries-An overview and the potential impacts of climate change. *Science of the Total Environment* 434: 213–227. https://doi.org/10.1016/j.scitotenv.2011.09.088.

Stelling, B., E. Phlips, S. Badylak, L. Landauer, M. Tate, and A. West-Valle. 2023. Seasonality of phytoplankton biomass and composition on the Cape Canaveral shelf of Florida: Role of shifts in climate and coastal watershed influences. *Frontiers in Ecology and Evolution* 11: 1134069. https://doi.org/10.3389/fevo.2023.1134069.

Tango, P.J., and R.A. Batiuk. 2016. Chesapeake Bay recovery and factors affecting trends: Long-term monitoring, indicators, and insights. *Regional Studies in Marine Science* 4: 12–20. https://doi.org/10.1016/j.rsma.2015.11.010.

Taplin, K.A., E.A. Irlandi, and R. Raves. 2005. Interference between the macroalga *Caulerpa prolifera* and the seagrass *Halodule wrightii*. *Aquatic Botany* 83: 175–186. https://doi.org/10.1016/j.aquabot.2005.06.003.

Todd, A., H. Walsh, and J. Neville. 2023. *2023 economic valuation of Tampa Bay*. Technical Report 09–23. St. Petersburg, Florida: Tampa Bay Estuary Program.

Tomasko, D.A. 2023. Ecological impacts to Sarasota Bay from Piney Point discharges-Examining the evidence. *Florida Scientist* 86: 301–313.

Tomasko, D.A., C.A. Corbett, H.S. Greening, and G.E. Raulerson. 2005. Spatial and temporal variation in seagrass coverage in Southwest Florida: Assessing the relative effects of anthropogenic nutrient load reductions and rainfall in four contiguous estuaries. *Marine Pollution Bulletin* 50: 797–805. https://doi.org/10.1016/j.marpolbul.2005.02.010.

Tomasko, D.A., M. Alderson, R. Burnes, J. Hecker, N. Iadevaia, J. Leverone, G. Raulerson, and E. Sherwood. 2020. The effects of Hurricane Irma on seagrass meadows in previously eutrophic estuaries in Southwest Florida (USA). *Marine Pollution Bulletin* 156: 111247. https://doi.org/10.1016/j.marpolbul.2020.111247.

Usup, G., D.M. Kulis, and D.M. Anderson. 1994. Growth and toxin production of the toxic dinoflagellate *Pyrodinium bahamense var.compressum* in laboratory cultures. *Natural Toxins* 2: 254–262. https://doi.org/10.1002/nt.2620020503.

Van Bogaert, N., K. Rosario, B.T. Furman, M.O. Hall, A.M. Greco, and M. Breitbart. 2018. Discovery of a novel potexvirus in the seagrass *Thalassia testudinum* from Tampa Bay, Florida. *Limnology and Oceanography Letters* 4: 1–8. https://doi.org/10.1002/lol2.10098.

Vandermeer, J.H. 1972. Niche theory. *Annual Review of Ecology and Systematics* 3: 107–132. https://doi.org/10.1146/annurev.es.03.110172.000543.

Waycott, M., C.M. Duarte, T.J.B. Carruthers, R.J. Orth, W.C. Dennison, S. Olyarnik, A. Calladine, et al. 2009. Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *Proceedings of the National Academy of Sciences* 106: 12377–12381. https://doi.org/10.1073/pnas.0905620106.

Webster, C.L., K.L. Kilminster, M. Sánchez Alarcón, K. Bennett, S. Strydom, S. McNamara, P.S. Lavery, and K.M. McMahon. 2021. Population-specific resilience of *Halophila ovalis* seagrass habitat to unseasonal rainfall, an extreme climate event in estuaries. *Journal of Ecology* 109: 3260–3279. https://doi.org/10.1111/1365-2745.13648.

Weisberg, R.H., and L. Zheng. 2006. Circulation of Tampa Bay driven by buoyancy, tides, and winds, as simulated using a finite volume coastal ocean model. *Journal of Geophysical Research: Oceans*. https://doi.org/10.1029/2005jc003067.

White, E., and D. Kaplan. 2017. Restore or retreat? Saltwater intrusion and water management in coastal wetlands. *Ecosystem Health and Sustainability* 3: e01258. https://doi.org/10.1002/ehs2.1258.

Wong, M.C., and M. Dowd. 2023. The role of short-term temperature variability and light in shaping the phenology and characteristics of seagrass beds. *Ecosphere* 14: e4698. https://doi.org/10.1002/ecs2.4698.

Wood, S.N. 2011. Fast stable restricted maximum likelihood and marginal likelihood estimation of semiparametric generalized linear models. *Journal of the Royal Statistical Society (b)* 73: 3–36. https://doi.org/10.1111/j.1467-9868.2010.00749.x.

Wood, S. N. 2017. *Generalized additive models: An introduction with R*. 2nd ed. Chapman; Hall/CRC.

Zhang, M., H. Li, L. Zhang, and J. Liu. 2023. Heat stress, especially when coupled with high light, accelerates the decline of tropical seagrass (*Enhalus acoroides*) meadows. *Marine Pollution Bulletin* 192: 115043. https://doi.org/10.1016/j.marpolbul.2023.115043.

Zieman, J.C. 1975. Seasonal variation of turtle grass, *Thalassia testudinum König*, with reference to temperature and salinity effects. *Aquatic Botany* 1: 107–123. https://doi.org/10.1016/0304-3770(75)90016-9.

Zuur, A.F., E.N. Ieno, N. Walker, A.A. Saveliev, and G.M. Smith. 2009. *Mixed effects models and extensions in ecology with R*. New York: Springer. https://doi.org/10.1007/978-0-387-87458-6.