Recommendations for Sampling Effort of Vegetation Communities in the Critical Coastal Habitat Assessment

Marcus W. Beck, Tampa Bay Estuary Program, St. Petersburg, Florida 33701 USA

Kara R. Radabaugh, Fish & Wildlife Research Institute, St. Petersburg, Florida 33701 USA

Kerry E. Flaherty-Walia, Fish & Wildlife Research Institute, St. Petersburg, Florida 33701 USA

November, 2022

Table of contents

## Acknowledgements

This document relies exclusively on data collected by field crews during the phase I (2015 and 2016 sampling) and phase II (2018 sampling) Critical Coastal Habitat Assessments. We acknowledge the efforts undertaken in gathering these important data to inform that state of coastal habitats in the Tampa Bay estuary. Specifically, we thank field staff from Atkins North America, Inc. (David Loy, Renee Price), Environmental Science Associates, Inc. (Doug Robison), and the Fish and Wildlife Research Institute of the Florida Fish and Wildlife Conservation Commission (Kara Radabaugh, Ryan Moyer). We also thank Pamela Latham for discussions that informed the methods described herein.

## Executive Summary

## 1 Introduction

Native coastal habitats provide multiple ecosystem services for natural communities and human-based uses of the resources. Climate change and sea-level rise are critical threats to the health and resilience of these communities. Habitat changes with projected estimates of sea-level rise are expected to manifest as substantial habitat loss or alteration and landward migration of intertidal species with changes in salinity. Mangroves, salt marshes, and salt barrens are collectively referred to as emergent tidal wetlands and are expected to be disproportionately altered by climate change (Sherwood and Greening, 2012; Cavanaugh et al., 2019; Osland et al., 2022). Alteration of these communities may contribute to reduced shoreline stability, increased vulnerability to flooding, changes in water quality, and decreases in fisheries production (Kennedy, 1990; Duarte et al., 2013; Gilby et al., 2020). These negative changes may translate to losses of recreational or economic opportunities of those that live in or near coastal environments (Spalding et al., 2014; Toimil et al., 2018).

The Tampa Bay region (Florida, USA) has been identified as one of the most vulnerable areas worldwide to the effects of climate change due to its shallow topography and dense urban population living near the coast (Sherwood and Greening, 2014; Fu et al., 2016). Projections of sea-level rise for Tampa Bay suggest an increase of two to 8.5 feet by the year 2100 (Burke et al., 2019), potentially inundating low-lying coastal areas with emergent tidal wetlands. Numerous efforts to track and plan for the projected effects of sea-level rise have been the focus of regional managers in recent decades. The Tampa Bay Estuary Program works to develop science-based solutions to managing the estuary by creating consensus among regional managers on approaches for maintaining a healthy bay and watershed. The Habitat Master Plan (HMP) (Robison et al., 2020) provides coverage targets and goals for all native habitats in the bay, with specific focus on intertidal habitats that are especially vulnerable to climate change and coastal development. These targets and goals are based on coverages that are likely to be achieved based on past restoration efforts, available remaining areas for acquisition, and trajectories of land use change for the prior decades.

The Critical Coastal Habitat Assessment (CCHA) complements the HMP by providing high-resolution surveys of emergent tidal wetlands (Moyer and Radabaugh, 2017; Price et al., 2017). Unlike the HMP that is based primarily on remote sensing products, the CCHA provides field-based estimates of coastal habitats at fixed sites for selected locations around Tampa Bay. Each of nine sites are sampled every 3-5 years using a transect design extending landward from the water to survey vegetation and tree communities, surface elevation, soil characteristics, interstitial porewater salinity, and faunal communities. The surveys also track discrete vegetation zones (e.g., mangrove fringe, coastal uplands, etc.) to assess potential landward shifts or reductions in habitats based on anticipated changes from sea-level rise. To date, two rounds of sampling have been completed. The first baseline surveys were conducted in 2015 and 2016 (Moyer and Radabaugh, 2017; Price et al., 2017) and a second set of surveys was conducted in 2018.

The CCHA surveys are expected to be conducted over the next several decades based on the timeline of likely changes to coastal habitats from climate change (Sherwood and Greening, 2012; Burke et al., 2019). The feasibility of continued surveys depends on the relative effort of field crews in collecting useful information to inform understanding of regional effects of sea-level rise and temperature change. The vegetation surveys are a core component of the field efforts that collect information on species richness and cover at each site and within each vegetation zone. Current effort is time-intensive and requires sampling vegetation within a 0.25 m quadrat every half meter along a transect, where the mean length of transects is approximately 150 meters. Understanding the quality of data provided by the vegetation surveys at reduced sampling effort will inform the level of effort required for future sampling.

This report evaluates the effect of reduced sample effort on key metrics for vegetation transects at all CCHA monitoring sites in Tampa Bay. A central hypothesis is that the current sampling effort using quadrats every half meter is not necessary to provide a robust assessment of the vegetation communities and a reduction of effort is possible without sacrificing the quality of information in the survey. A recommendation of reduced effort is expected to increase the likelihood that surveys will be completed in the future if less time and resources are required for field sampling. A desktop analysis was conducted to sub-sample the existing surveys from 1 meter (50% reduction in effort) to 10 meter (95% reduction in effort) sampling at half meter intervals. Scenarios evaluating more than 50% (1 meter sampling) or 66% (1.5 meter sampling) reduction in effort are unrealistic, but still useful to understand the effects on quantitative measures of habitat. Key metrics describing the vegetation communities were evaluated for each sub-sampled survey and compared to the original estimates at 0.5 meter sampling. Key metrics were evaluated based on information that was considered relevant to evaluate the long-term effects of climate change. These metrics included:

1. Total species richness at a site
2. Total species richness at a site by vegetation zone
3. Elevation at which 95% of key species occur (e.g., mangroves)
4. Vegetation zone distances by site

Recommendations are provided for the appropriate sampling interval of future surveys.

## 2 Methods

The current sampling design used at each CCHA site is shown in [Figure 1](#fig-transectdesign). The transects are perpendicular to the coastline and extend from the water to upland habitats using bench-marked beginning and end locations that are fixed at each site. The vegetation community is surveyrf at half-meter marks using a 0.25 m quadrat, where species richness and basal cover of each species is enumerated. Additional information collected at each transect includes tree surveys in 1 m quadrats (red boxes, [Figure 1](#fig-transectdesign)) using the Point-Center-Quarter method (Cottam and Curtis, 1956), elevation surveys along the length of the transect using Real Time Kinematic (RTK) satellite navigation via GPS, porewater interstitial salinity within each vegetation zone, feldspar horizons and soil samples analyzed for total percent organic content and sediment grain size, and faunal surveys at random plots evaluating species composition and burrow density of crabs (*Uca* spp.) and snails (*Littorina* spp.). Locations of the vegetation zones and the transitions between them are visually assessed and the meter marks are recorded where the transitions occur. Vegetation surveys represent a majority of the total sampling time at each transect.

|  |
| --- |
| Figure 1: Schematic of the sampling design at all CCHA sites (from Moyer and Radabaugh, 2017). Vegetation quadrats are placed every half meter along the main transect from the water to the landward end |

All existing CCHA vegetation surveys at each site and for both phases of sampling (baseline and 2018 surveys) were sub-sampled from the existing effort of sample plots every half meter. [Figure 2](#fig-subsampex) shows an example of the sub-sampling scheme, where effort was reduced in increments of a half meter, starting from the complete survey to an upper limit of sampling every ten meters. For simplicity, [Figure 2](#fig-subsampex) shows sub-sampling up to every three meters for a hypothetical 30 meter transect (mean transect length across all sites is approximately 150 meters). The top row represents the complete existing survey as the basis of comparison for the metrics estimated from the sub-sampled datasets. The existing surveys were sub-sampled at the specified meter interval for every unique subset (or replicate) that was possible. For example, two unique replicates can be created with sub-sampling every meter, three every 1.5 meters, etc. The large red points in [Figure 2](#fig-subsampex) show which of the existing survey points were sub-sampled for the specified sub-sample distance.

|  |
| --- |
| Figure 2: Schematic of the method for creating sub-samples of existing transect data. The full survey with sampling at 0.5 meter intervals is shown at the top, with increasing sampling intervals from top to bottom. Replicates for each sample interval show the unique points that were selected. |

The relative reduction in effort associated with each half-meter increase along a transect is shown with [Figure 3](#fig-releff). The percent reduction of the original effort with each half-meter reduction in sample effort decreases from one meter to ten meter sampling, with the largest reduction in the former and the smallest reduction in the latter.

|  |
| --- |
| Figure 3: Percent reductions in sample effort as a function of the sample interval. For example, sampling every one meter represent a 50% reduction in effort form the original sample at 0.5 meters. |

For each site and sample year, relevant vegetation metrics were estimated from the vegetation transect that was sub-sampled following the methods above. Estimates of species richness at each site and species richness in each vegetation zone were calculated for each sub-sample. The elevations at which key species occurred were also estimated for each level of sampling effort. Three mangrove species that commonly occur in tidal wetlands (red mangroves, *Rhizophore mangle*, white mangroves, *Laguncularia racemosa*, and black mangroves, *Avicennia germinans*) were assessed as key species expected to be affected by climate change (e.g., landward expansion to higher elevation, colonization of salt marshes and salt barrens, and changes in species distributions, Comeaux et al., 2012; Osland et al., 2022). Elevation estimates from the RTK surveys were combined with the vegetation data and cumulative distribution functions (CDF) were calculated for all points in a sub-sample where a key species was found. The 95th percentile of the elevation estimates for each CDF were used to identify an approximate elevation limit for each species and sub-sample. Lastly, the ability to identify each vegetation zone and the meter mark at which a zone began was also estimated from each sub-sample.

All replicates for each level of sampling effort were considered independent samples and results of each summary metric were averaged across replicates to provide a single estimate for the level of effort. The variance across the estimates was also estimated to describe how each metric varied depending on the level of effort and which meter mark was chosen as the start of the transect. For most metrics, additional characteristics of the data were also evaluated to identify potential factors that contributed to the greatest changes with reductions in effort. For example, estimates at sites with many rare species and potentially higher species richness may be disproportionately affected by reductions in sample effort. Metrics were evaluated as total change and the relative percent change to identify these potential effects. Metrics were also evaluated by comparing total change from 0.5 meter to 10 meter sampling as a function of site-level characteristics, e.g., total richness.

All analyses were conducted using the R Statistical Programming Language (R Core Team, 2022) using custom routines created by the authors. Analysis workflows, source data, and additional online content are available at <https://github.com/tbep-tech/ccha-sampling-effort>.

## 3 Results

### 3.1 Richness estimates

Total richness estimates at each site and for each year of sampling decreased as expected with reductions in effort ([Figure 4](#fig-richex)). The rate of reduction in the richness estimates decreased as the sampling intervals increased, in agreement with the level of effort as a percentage of the total shown in [Figure 3](#fig-releff). Each point in [Figure 4](#fig-richex) is the average estimate across replicates for the sampling distance shown on the x-axis. The lines are polynomial smooths to show the trend and the size of each point is in proportion to the variance of the species richness estimate across the replicates. The subplots are arranged left to right, top to bottom based on the greatest reduction in species richness as a percentage of the total.

|  |
| --- |
| Figure 4: Reductions in species richness estimates with sampling distance for all sites and each year of sampling. Point size shows the variance across the replicates for each level of sampling effort. |

[Figure 5](#fig-richperex) shows similar results as [Figure 4](#fig-richex), except richness is scaled as a percentage of the total at full sampling effort. This plot allows a comparison of a reduction in the estimate independent of the overall species richness because the sensitivity of the change may vary depending on the total. The greatest percent reductions in the richness estimates with decreased sampling were observed at the Little Manatee River and Hidden Harbor sites.

|  |
| --- |
| Figure 5: Relative percent reductions in species richness estimates with sampling distance for all sites and each year of sampling. Point size shows the variance across the replicates for each level of sampling effort. |

An assessment of percent loss in the richness estimates as a function of total species richness at 0.5 meter sampling is shown in [Figure 6](#fig-richloss). This plot tests the assumption that estimates at sites with higher species richness will be affected more strongly be reductions in effort. There is some evidence that a greater reduction in species richness (measured as the total percent loss from 0.5 meter to 10 meter sampling) is expected at sites with higher richness, although a linear regression fit through the points shows the model is insignificant.

|  |
| --- |
| Figure 6: Total percent loss in species richness estimates from 0.5 to 10 meter sampling as a function of actual species richness at 0.5 meter sampling. |

### 3.2 Richness estimates by zone

Similar to the richness estimates at each site, an assessment of changes with reduction in effort was also observed within the unique vegetation zones observed across all sites ([Figure 7](#fig-richzone)). A total of 28 unique zones were sampled across the nine sites. The lines show the estimated reduction in the richness estimate for each site and the thick line is the average trend for each zone (points are not shown to reduce clutter). As for the site-level assessments, the rate of reduction decreased with increasing sampling distance within each zone. The subplots are arranged left to right, top to bottom based on the greatest reduction in species richness as a percentage of the total. The highest overall richness was observed in the transitional marsh zone at Cockroach Bay (20 species in the first phase of sampling) and coastal upland zone at Weedon Island (19 species in the first phase of sampling). Consequently, the zones at these sites had large decreases in total richness with reduced effort. Conversely, aquatic zones (channel, freshwater pond, and tidal creek) had very little vegetation (*Juncus roemerianus* or *Blutaparon vermiculare*) and did not show any changes with reduction in effort. Changes in richness with reduced effort varied considerably for the coastal uplands, where total richness differed by site (highest at Weedon Island, lowest at Little Manatee River).

|  |
| --- |
| Figure 7: Reductions in species richness estimates with sampling distance for each unique vegetation zone across all sites. Each line is a unique site and sample year. The black line shows the average trend for the zone. Panels are arranged based on the greatest percent reduction in richness from full to minimum effort. |

[Figure 8](#fig-richzoneper) shows similar results as [Figure 7](#fig-richzone), except richness is scaled as a percentage of the total within a zone at full sampling effort. This plot allows a comparison of a reduction in the estimate independent of the overall species richness. The greatest percent reduction in the richness estimate with decreased sampling was observed in the coastal upland zone at Weedon Island (over 80% loss), which also had the second highest total species richness.

|  |
| --- |
| Figure 8: Relative percent reductions in species richness estimates with sampling distance for each unique vegetation zone across all sites. Each line is a unique site and the black line shows the average trend for the zone. Panels are arranged based on the greatest percent reduction in richness from full to minimum effort. |

[Figure 9](#fig-richzonesum) combines results from [Figure 8](#fig-richzoneper) into a single panel, showing only the average percent loss for all sites in each zone (i.e., the black lines in [Figure 8](#fig-richzoneper)). The line thickness is in proportion to the average total species richness for all sites in a zone at full sampling effort. Line thickness generally increases from top to bottom at the ten meter mark, providing evidence that percent loss in the estimate is a function of mean richness. An alternative depiction of this relation is shown in [Figure 10](#fig-richzoneloss), where a linear fit of total percent loss from 0.5 meter to 10 meter sampling versus average species richness by zone was significant (*p* < 0.05).

|  |
| --- |
| Figure 9: Relative percent reduction in species richness with sampling distance for each unique vegetation zone across all sites. Each line represents the average percent reduction for each occurrence of a zone. |

|  |
| --- |
| Figure 10: Total percent loss of richness from 0.5 meter to 10 meter sampling as a function of average species richness within a zone. |

### 3.3 Elevation of key species

Elevation estimates at which 95% of mangrove species were observed with reductions in sampling effort varied by site ([Figure 11](#fig-elevex)). Consistent reductions in the elevation estimates were observed for all species at some sites (e.g., Fort De Soto, Mosaic), whereas only some species showed reductions at some sites (e.g., only *Rhizophore mangle* at Big Bend - Teco). Additionally, the overall elevations at 0.5 meter sampling varied by species at each site, such that red mangroves which typically are found closer to the shoreline were observed at higher (Big Bend, Fort De Soto) or lower (Mosaic, Upper Tampa Bay Park) estimates than the other two species. The variance in the estimated elevation across all species generally increased with reductions in sampling effort.

|  |
| --- |
| Figure 11: Elevation estimates of three mangrove species with sampling distance at each site. Point size shows the variance across the replicates for each level of sampling effort. |

The total change in the elevation estimates as a percentage of the original was strongly influenced by overall frequency occurrence of each species at a site ([Figure 12](#fig-foperelevex), *p* < 0.05). Species with greater frequency occurrence at a site have smaler changes in the elevation estimates with reductions in sampling effort. Similarly, lower variance among the estimates was observed for species with higher frequency occurrences ([Figure 12](#fig-foperelevex), *p* < 0.05).

|  |
| --- |
| Figure 12: Total percent change in elevation estimates for three mangrove species from 0.5 meter to 10 meter sampling as a function of frequency occurrence of each species at 0.5 meter sampling. |

|  |
| --- |
| Figure 13: Variance of the elevation esimates of three mangrove species at 10 meter sampling as a function of actual frequency occurrence. |

### 3.4 Zone identification

The number of unique vegetation zones along each transect varied among the sites, with a maximum of twelve zones at Upper Tampa Bay Park and a minimum of four zones at Hidden Harbor. The minimum zone length observed was 1 meter for the tidal creek zone Hidden Harbor and the maximum zone length was 102 meters for the *Juncus* marsh zone at Little Manatee River. Mean zone length across all sites was 23 meters. An assessment of the ability to sample all of the zones at a site with reduced sampling effort showed that most but not all of the zones can be identified up to sampling every 10 meters ([Figure 14](#fig-zonecnt)). Each zone could be identified at any level of sample effort for Big Bend - TECO, Fort DeSoto, and Harbor Palms, whereas some zones were missed for the other sites depending on the sampling interval. For example, Hidden Harbor includes four zones, with the smallest as 1 meter in length. An average zone count of less than four occurs when the sampling interval is greater than 1 meter, i.e, 1.5 meters, which misses the smallest zone for some of the replicates. The length of the smallest zone for sites where the average count decreases with reduced effort can be seen at the point where the curve is no longer constant at the true zone count. As such, all zones will be identified if the sampling interval is less than the minimum length of all zones.

|  |
| --- |
| Figure 14: Number of unique zones identified at each site from 0.5 meter to 10 meter sampling. Point size shows the variance across the replicates for each level of sampling effort. |

Estimated lengths of each zone did not change systematically with increased sample effort, although the variance of the estimates increased with reduced sampling effort ([Figure 15](#fig-zonedst)). The change in variance was non-linear, such that it generally increased with reduced effort, but often was estimated at low or zero values depending on the sample interval. An example from Big Bend - TECO demonstrates how variance changes across each zone with the sampling effort. The total zone length is shown at the top of each subplot. The variance estimates in a zone fall to zero when the zone distance can be evenly divided by the sampling interval, i.e., the same distance estimate is obtained for each sub-sample at the specified sampling interval. Variance estimates peak between zero values when the remainder of the zone distance divided by the sample distance is large.

|  |
| --- |
| Figure 15: Total distance of each zone at each site from 0.5 meter to 10 meter sampling. Each line is a unique zone. Point size shows the variance across the replicates for each level of sampling effort. |

|  |
| --- |
| Figure 16: An example of the variance of the zone distances for each zone at the Big Bend - TECO site from 0.5 meter to 10 meter sampling. |

Detailed results for the zone lengths at each site are shown in Figures [17](#fig-zoneestbb), [18](#fig-zoneestcb), [19](#fig-zoneestfd), [20](#fig-zoneesthp), [21](#fig-zoneesthh), [22](#fig-zoneestlmr), [23](#fig-zoneestm), [24](#fig-zoneestutbp), and [25](#fig-zoneestwi). The shaded areas represent the true zones and the points show the estimates of the starting location for each zone at the specified sampling interval. The horizontal bars show the variance across the sub-samples for each level of sampling effort. The figures clearly show a reduction in precision for the zone lengths and an inability to identify some zones with decreasing sample effort.

|  |
| --- |
| Figure 17: Estimates and variance of the estimates in the zone starting locations at Big Bend - Teco for 1 meter to 10 meter sampling. The colored regions indicate the zone delineations at 0.5 meter sampling. |

|  |
| --- |
| Figure 18: Estimates and variance of the estimates in the zone starting locations at Cockroach Bay for 1 meter to 10 meter sampling. The colored regions indicate the zone delineations at 0.5 meter sampling. |

|  |
| --- |
| Figure 19: Estimates and variance of the estimates in the zone starting locations at Fort DeSoto for 1 meter to 10 meter sampling. The colored regions indicate the zone delineations at 0.5 meter sampling. |

|  |
| --- |
| Figure 20: Estimates and variance of the estimates in the zone starting locations at Harbor Palms for 1 meter to 10 meter sampling. The colored regions indicate the zone delineations at 0.5 meter sampling. |

|  |
| --- |
| Figure 21: Estimates and variance of the estimates in the zone starting locations at Hidden Harbor for 1 meter to 10 meter sampling. The colored regions indicate the zone delineations at 0.5 meter sampling. |

|  |
| --- |
| Figure 22: Estimates and variance of the estimates in the zone starting locations at LIttle Manatee River for 1 meter to 10 meter sampling. The colored regions indicate the zone delineations at 0.5 meter sampling. |

|  |
| --- |
| Figure 23: Estimates and variance of the estimates in the zone starting locations at Mosaic for 1 meter to 10 meter sampling. The colored regions indicate the zone delineations at 0.5 meter sampling. |

|  |
| --- |
| Figure 24: Estimates and variance of the estimates in the zone starting locations at Upper Tampa Bay Park for 1 meter to 10 meter sampling. The colored regions indicate the zone delineations at 0.5 meter sampling. |

|  |
| --- |
| Figure 25: Estimates and variance of the estimates in the zone starting locations at Weedon Island for 1 meter to 10 meter sampling. The colored regions indicate the zone delineations at 0.5 meter sampling. |

## 4 Conclusions and Recommendations

* Richness estimates for a site are reduced with lower sample effort
* The amount of reduction in the richness estimated was weakly associated with total species richness at a site
* Richness estimates for a zone are reduced with lower sample effort
* Richness estimates will be systematically lower with reduced effort for zones with overall higher richness. These include zones…
* Frequency occurrence estimates for individual species will not be systematically different at lower sample effort, but the estimates are imprecise
* Species with higher frequency occurrence will have less precise estimates with reduced effort compared to those with lower frequency occurrence
* Zone identification at each site is not affected by sample effort, so long as the sample interval is less than the length of the smallest zone at a site. Precision of the distance estimates for each zone decreases with reductions in sample effort, but the decrease is not linear and is affected by the actual zone length and sample interval. These issues are inconsequential for 1 meter sampling.

The estimated elevation at which mangrove species are observed did not show consistent changes across sites by species, although a general decrease was observed. This result was unexpected and it is unclear why reduced effort resulted in lower elevation estimates as elevation generally increases along a transect and the sub-sampling method was not expected to consistently sample lower elevations. An additional notable difference was differences among the species for the elevations at 0.5 meter sampling across the sites. In general, the three mangrove species are expected to follow an elevation gradient where red mangroves are more tolerant of higher salinities and inhabit lower elevations closer to the shoreline, white mangroves are intermediate, and black mangroves are less tolerant of higher salinities and inhabit higher elevations. Some sites showed red mangroves at higher elevations than black mangroves, which was not expected, although these differences could be related to site-specific characteristics of the elevation gradient and porewater salinities. Regardless, the elevation metrics showed considerable changes with relative sampling effort, although they were very minor with a 50% reduction in effort at 1 meter sampling. Any changes as a function of sample effort will also be notably less for species with high frequency occurrence at a site (Figures [12](#fig-foperelevex) and [13](#fig-fovarelevex)).

## References

Burke, M., Carnahan, L., Hammer-Levy, K., and Mitchum, G. (2019). Recommended projections of sea level rise for the Tampa Bay region (update). St. Petersburg, Florida: Tampa Bay Estuary Program Available at: <https://drive.google.com/file/d/1c_KTSJ4TgVX9IugnyDadr2Hc0gjAuQg2/view?usp=drivesdk>.

Cavanaugh, K. C., Dangremond, E. M., Doughty, C. L., Williams, A. P., Parker, J. D., Hayes, M. A., et al. (2019). Climate-driven regime shifts in a mangrove-salt marsh ecotone over the past 250 years. *Proceedings of the National Academy of Sciences* 116, 21602–21608. Available at: <https://doi.org/10.1073/pnas.1902181116>.

Comeaux, R. S., Allison, M. A., and Bianchi, T. S. (2012). Mangrove expansion in the Gulf of Mexico with climate change: Implications for wetland health and resistance to rising sea levels. *Estuarine, Coastal and Shelf Science* 96, 81–95. Available at: <https://doi.org/10.1016/j.ecss.2011.10.003>.

Cottam, G., and Curtis, J. T. (1956). The Use of Distance Measures in Phytosociological Sampling. *Ecology* 37, 451–460. doi: [10.2307/1930167](https://doi.org/10.2307/1930167).

Duarte, C. M., Losada, I. J., Hendriks, I. E., Mazarrasa, I., and Marbà, N. (2013). The role of coastal plant communities for climate change mitigation and adaptation. *Nature Climate Change* 3, 961–968. doi: [10.1038/nclimate1970](https://doi.org/10.1038/nclimate1970).

Fu, X., Song, J., Sun, B., and Peng, Z.-R. (2016). “Living on the edge”: Estimating the economic cost of sea level rise on coastal real estate in the Tampa Bay region, Florida. *Ocean & Coastal Management* 133, 11–17. doi: [10.1016/j.ocecoaman.2016.09.009](https://doi.org/10.1016/j.ocecoaman.2016.09.009).

Gilby, B. L., Weinstein, M. P., Baker, R., Cebrian, J., Alford, S. B., Chelsky, A., et al. (2020). Human Actions Alter Tidal Marsh Seascapes and the Provision of Ecosystem Services. *Estuaries and Coasts* 44, 1628–1636. doi: [10.1007/s12237-020-00830-0](https://doi.org/10.1007/s12237-020-00830-0).

Kennedy, V. S. (1990). Anticipated Effects of Climate Change on Estuarine and Coastal Fisheries. *Fisheries* 15, 16–24. doi: [10.1577/1548-8446(1990)015<0016:aeocco>2.0.co;2](https://doi.org/10.1577/1548-8446(1990)015<0016:aeocco>2.0.co;2).

Moyer, R., and Radabaugh, K. (2017). Phase 2: Critical Coastal Habitat Assessment Final Report. St. Petersburg, Florida: Tampa Bay Estuary Program Available at: <https://drive.google.com/file/d/1bDZ0JmuD2_1RM6VkSrsAR3g8bciQaf3F/view?usp=drivesdk>.

Osland, M. J., Hughes, A. R., Armitage, A. R., Scyphers, S. B., Cebrian, J., Swinea, S. H., et al. (2022). The impacts of mangrove range expansion on wetland ecosystem services in the southeastern United States: Current understanding, knowledge gaps, and emerging research needs. *Global Change Biology* 28, 3163–3187. doi: [10.1111/gcb.16111](https://doi.org/10.1111/gcb.16111).

Price, R., Loy, D., and Robison, D. (2017). Phase 1: Critical Coastal Habitat Assessment: Baseline monitoring report. St. Petersburg, Florida: Tampa Bay Estuary Program Available at: <https://drive.google.com/file/d/122AvajD3fxOVORHfO5HQUMdXkLNoQm48/view?usp=drivesdk>.

R Core Team (2022). *R: A language and environment for statistical computing*. Vienna, Austria: R Foundation for Statistical Computing Available at: <https://www.R-project.org/>.

Robison, D., Ries, T., Saarinen, J., Tomasko, D., and Sciarrino, C. (2020). Tampa Bay Estuary Program: 2020 Habitat Master Plan Update. St. Petersburg, Florida: Tampa Bay Estuary Program Available at: <https://drive.google.com/file/d/1Hp0l_qtbxp1JxKJoGatdyuANSzQrpL0I/view?usp=drivesdk>.

Sherwood, E. T., and Greening, H. S. (2012). Critical Coastal Habitat Vulnerability Assessment for the Tampa Bay Estuary: Projected Changes to Habitats due to Sea Level Rise and Climate Change. St. Petersburg, Florida: Tampa Bay Estuary Program Available at: <https://drive.google.com/file/d/11tu2-0y7wdAHsdvMYE8bNb68mgvcpWbK/view?usp=drivesdk>.

Sherwood, E. T., and Greening, H. S. (2014). Potential impacts and management implications of climate change on Tampa Bay Estuary critical coastal habitats. *Environmental Management* 53, 401–415. doi: [10.1007/s00267-013-0179-5](https://doi.org/10.1007/s00267-013-0179-5).

Spalding, M. D., Ruffo, S., Lacambra, C., Meliane, I., Hale, L. Z., Shepard, C. C., et al. (2014). The role of ecosystems in coastal protection: Adapting to climate change and coastal hazards. *Ocean & Coastal Management* 90, 50–57. doi: [10.1016/j.ocecoaman.2013.09.007](https://doi.org/10.1016/j.ocecoaman.2013.09.007).

Toimil, A., Díaz-Simal, P., Losada, I. J., and Camus, P. (2018). Estimating the risk of loss of beach recreation value under climate change. *Tourism Management* 68, 387–400. doi: [10.1016/j.tourman.2018.03.024](https://doi.org/10.1016/j.tourman.2018.03.024).