Addressing climate change and development pressures in an urban estuary through habitat restoration planning

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## Abstract

Native habitats in Florida face dual pressures at the land-sea interface from urban development and sea-level rise. To address these pressures, restoration practitioners require robust tools that identify reasonable goals given historical land use trends, current status of native habitats, and anticipated future impacts from coastal stressors. A target-setting approach for native habitats was created for the Tampa Bay watershed that identifies current restoration opportunities and establishes short-term (2030) targets and long-term (2050) goals. The approach was informed through a three-decade habitat change analysis and over forty years of habitat restoration projects in the region. Restoration goals were defined based on what is possible today and the projected needs for the future, rather than attempting to replicate past ecological conditions. The new approach also accounts for the expected impacts of sea-level rise, climate change, and watershed development - stressors which are pervasive in Florida. The resulting habitat goals are provided as maps that identify remaining restoration opportunities, while also providing an approach for the entire watershed that targets subtidal, intertidal, and uplands. This approach also supports coastal planning decisions that need to address competing interests and could be applied in other coastal settings where sustainable urbanization practices need to co-exist with natural environments. Methods for repeatable analyses are also available using an open source workflow to update progress over time and for adoption by others.

*Key words*: Florida, land use change, restoration planning, sea-level rise, Tampa Bay, urbanization

## 1 Introduction

The health of estuarine systems and coastal habitats is tightly linked to land use and management of the watershed (Yoskowitz and Russell, 2015). Coastal habitats provide multiple ecosystem services, including wildlife shelter and migratory corridors (Yoskowitz and Russell, 2015), fisheries production (Houde and Rutherford, 1993), water quality improvement (Kushlan, 1990; Sprandel et al., 2000; Ávila-García et al., 2020), erosion and flood attenuation (Calil et al., 2015; Menéndez et al., 2018), carbon sequestration (Dontis et al., 2020) and recreation (Chung et al., 2018). Pervasive stressors can negatively impact the services provided by these habitats. Anthropogenic land development can remove or substantially modify existing habitats and can alter hydrology of nearby streams and rivers (Theobald et al., 1997; Sim and Mesev, 2014). Habitat changes in response to climate change include landward migration of mangroves into salt marshes, upstream migration of salt marshes within tidal tributaries, and upland forest migration (Brinson et al., 1995; Vogelmann et al., 2012; Cavanaugh et al., 2019). Management of coastal habitats requires an understanding of how stressors influence these resources, while balancing competing societal uses.

The combined effects of land development and climate change are especially problematic for prioritizing habitat restoration activities in coastal environments. Landward migration of critical habitats in response to sea-level rise may not be possible due to anthropogenic barriers in the watershed. Sea-level rise can occur quicker than landward migration of salt marshes and the upland slope may already be lost to urban development and hardening (Titus et al., 2009). Coastal development and climate change diminish the available space for future restoration in urban estuaries, thereby impacting ecosystem services provided by these habitats and the wildlife they support (Enwright et al., 2015). Given projected habitat losses and the limited resources available, appropriate and realistic sites for restoration need to be identified that account for future stressors and past trends.

Past approaches for guiding restoration planning have been successfully used in other contexts, but they do not fully balance competing needs among public and private sectors, nor do they fully account for anticipated effects of multiple stressors. For example, an integrated watershed approach (Environmental Protection Agency, 1996) has been utilized since the early 1990s to diagnose and manage water quantity and quality problems by addressing issues within hydrologically-defined geographic areas. Additionally, the habitat mosaic approach (Henningsen, 2005) of including multiple habitat types within restoration projects has been recognized as an effective means of allowing ecosystem state changes in response to different environmental pressures (Duarte et al., 2009; Palmer, 2009). Adaptive management (Holling, 1978; Gregory et al., 2006) components have also been used to address challenges of sea-level rise, climate change, and development stressors, including monitoring to identify critical restoration decision points and needed intervention with contingency plans. Elements of each of these approaches could be combined to create a more holistic approach to guide restoration and conservation activities for coastal habitats in urban settings.

The Tampa Bay watershed (Florida, USA) is a valuable case study for developing a habitat restoration plan that addresses pervasive coastal stressors. Compared to other estuaries, the ratio of watershed to estuary area is small and the area is heavily developed with 42% of land use classified as urban and suburban residential (Southwest Florida Water Management District, 2018). A retrospective approach to setting habitat protection and restoration targets in Tampa Bay was previously used (Lewis and Robison, 1996; Robison, 2010; Cicchetti and Greening, 2011; Russell and Greening, 2015). Priority was given to restoration activities focused on habitat types that were important for a suite of estuarine faunal guilds disproportionately lost or degraded compared to a circa 1950 benchmark period considered as pre-development. Criticisms of this approach included lack of consideration for future sea-level rise and other climate change factors (Yoskowitz and Russell, 2015), use of expanded and different habitats outside the Tampa Bay watershed (Robison, 2010), lack of attention to upland or freshwater wetland habitats, and little recognition of land development trends or actual available space for restoration efforts. These challenges are shared by restoration practitioners in other coastal environments and an approach that accommodates these challenges for planning would be highly transferable.

In this paper, we describe an approach for habitat restoration and conservation planning that addresses the above challenges by considering the whole watershed, addressing historical changes, focusing on trajectories that have occurred during contemporary time periods, and considering both current and future stressors – particularly land development and sea-level rise. Current and historical data are available for most Tampa Bay habitats, representing a time period when federal, state and local regulations were in effect and regional impacts from climate change have been documented (Raabe et al., 2012; Cavanaugh et al., 2014). The approach establishes a framework that can guide both watershed-level habitat planning and site-level restoration activities and incorporates applicable elements of other habitat restoration paradigms discussed above (Palmer, 2009). The general approach includes 1) designation of habitat types by strata relative to the aquatic-terrestrial gradient, 2) quantification of historical trends by habitat types to identify appropriate future targets in coverage, and 3) identification of opportunity areas that could be used by practitioners to achieve restoration goals based on habitat type and past trajectories. These opportunity areas provide a first assessment of where restoration could occur, where on the ground assessments could be pursued to further quantify restoration potential. The outcomes of the approach are also spatially specific by providing maps to identity opportunity areas and reproducible using an open science workflow (Lowndes et al., 2017) that allows regular updates as new data become available.

## 2 Methods

### 2.1 Study area

Tampa Bay is a large open water estuary (surface area approximately 983 km2) on the west-central coast of Florida ([Figure 1](#fig-lulcmap)). The watershed covers approximately 5,872 km2, for a total combined area of approximately 6,855 km2. The climate is subtropical and within the 2020 ecotone for mangrove and salt marsh habitats. Native habitats in the watershed include pine flatwoods, forested freshwater wetlands and non-forested vegetated wetlands. The watershed is heavily developed with an estimated population of 3.3 million people in the four major counties (Rayer and Wang, 2020)*.* Numerous anthropogenic changes have altered the natural habits of Tampa Bay, including direct removal of habitat (e.g., dredge and fill of bay bottom, mining activities), alteration of hydrology, and destruction and fragmentation of habitat from development.

### 2.2 Habitats of Tampa Bay

The major habitat types of Tampa Bay were stratified by tidal influence and location in the watershed to define broad categories for restoration planning. Subtidal habitats included those that are submerged all or most of the time, intertidal habitats included emergent tidal wetlands that are submerged during high tides but exposed during low tides, and supratidal habitats included those that occur above the high tide line (i.e., the remainder of the watershed).

Subtidal habitats included hard bottom (Jaap and Hallock, 1990; Ash and Runnels, 2005; Kaufman, 2017; CSA Ocean Sciences, 2019), artificial reefs (Dupont, 2008), tidal flats (Moore et al., 1968; Eisma, 1998)*,* seagrasses (Heck et al., 2003; Sherwood et al., 2017), and oyster reefs (Coen et al., 2007; Ermgassen et al., 2013). Intertidal habitats (or emergent tidal wetlands) included mangroves (Odum and McIvor, 1990), salt marshes (Comeaux et al., 2012; Raabe et al., 2012), salt barrens (Bertness, 1985; Hsieh, 2004)*,* tidal tributaries (Sherwood, 2008; Wessel et al., 2022), and living shorelines (National Oceanic and Atmospheric Administration, 2015; Restore America’s Estuaries, 2015; Smith et al., 2018). Supratidal habitats included non-developed uplands (Meyers and Ewel, 1990), freshwater forested wetlands (Conner et al., 2007), and freshwater non-forested wetlands (Kushlan, 1990)*.* Uplands were further sub-divided into coastal and non-coastal uplands based on location relative to the 5-foot contour that covers an area of land from the local Mean Lower Low Water (MLLW) elevation landward to an elevation 5 feet above Mean Sea Level. This 5-foot contour or coastal stratum is an area of intense urban development and is expected to be affected by sea-level rise based on current estimates (Burke et al., 2019). Since 1946, the St. Petersburg tidal gauge (NOAA gauge 872650) has documented a nearly 20 cm increase in mean tidal height to present day. Projections from the year 2000 to 2100 suggest sea levels can increase between 58 and 259 cm in the region (Burke et al., 2019).

### 2.3 Approach

Coverage targets for habitat types and opportunity areas for restoration were identified by integrating multiple datasets available for the region. First, habitat status and historical trends were quantified using land use/land cover and subtidal datasets to understand relative changes that have occurred over time. Second, historic habitat restoration efforts conducted in the watershed were synthesized to inform on a practical and feasible level of effort that could be conducted by restoration practitioners in the future. The first two steps were used to identify short-term (2030) targets and long-term (2050) goals for native habitat coverage (hectares). The short-term targets provided an interim set of native habitat coverages to attain within a reasonable planning horizon, after which progress in attaining the long-term goals will be re-assessed. Finally, remaining restoration opportunities were spatially identified by combining current coverages with existing or proposed protected areas and areas anticipated to be affected by sea-level rise. As such, the approach identifies reasonable goals and targets based on past trends and provides spatially explicit information that identifies where restoration practitioners could prioritize projects based on opportunities within their respective jurisdictions.

### 2.4 Habitat status and trends

For the majority of subtidal, intertidal and supratidal habitats, coverages were quantified from two routine spatial assessment programs conducted by the Southwest Florida Water Management District (SWFWMD). For subtidal habitats, 2018 data were used to estimate current coverage of seagrasses, tidal flats, and oysters (Southwest Florida Water Management District, 2019). Historical datasets for subtidal habitats began in 1988 with updates occurring on an approximate biennial basis (Sherwood et al., 2017; Tomasko et al., 2020)*.* Oyster bed coverage has been routinely estimated in these data products beginning in 2014.

Current intertidal and supratidal habitat coverages were estimated using the 2017 SWFWMD Land Use Land Cover (LULC) maps (Southwest Florida Water Management District, 2018). Land use and cover types (natural and developed) are classified following the Florida Land Use Cover and Forms Classification System (FLUCCS, Florida Department of Transportation, 1999; Southwest Florida Water Management District, 2014; Kawula and Redner, 2018)*.* Mangroves, salt barrens, and salt marshes were reported individually. While the photointerpretation of specific freshwater wetland types is often difficult, forested wetlands and non-forested wetlands can be distinguished with these data. Therefore, all applicable FLUCCS codes representing natural freshwater wetlands were combined for these classifications. Native upland habitats were also combined in one classification. Historical estimates for all intertidal and supratidal habitats were also quantified starting with the earliest database in 1990 and occurring every two to three years until the current estimate in 2017.

To address data gaps for habitats not included in the routine SWFWMD datasets, results from special studies were compiled to obtain current estimates. These included hard bottom subtidal habitats, artificial reefs, tidal creeks, and living shorelines (Robison et al., 2020). No information on historical trends is available for these habitats.

Finally, a habitat coverage change analysis between the terminal years of data (1988 to 2018 for subtidal, 1990 to 2017 for intertidal and supratidal) was conducted to understand how habitats were changing between types. This required an intersection of the data layers to quantify if habitat types were unchanged or changed for any given location and identifying the type of change (e.g., seagrass to tidal flats). The results were summarized as Alluvial diagrams showing relative proportions of habitat change by type and between years (Allaire et al., 2017).

### 2.5 Restoration and enhancement projects

Restoration projects conducted over the past 40 years were quantified for each of the major habitat types to inform expectations for setting short-term targets and long-term goals. Restoration was defined as any activity that involved earthwork to reshape the land or the addition of structural elements (e.g., rock). Enhancement was defined as any activity not including earthwork that improved the environment (e.g., planting native vegetation, invasive species or debris removal, prescribed burns, etc.). Data were gathered from the SWFWMD Surface Water Improvement and Management Program, Federal Government Performance and Results Act reporting, the Tampa Bay Water Atlas (<https://www.tampabay.wateratlas.usf.edu/>), Tampa Bay Watch, and the Technical Advisory Committee of the Tampa Bay Estuary Program. The collected data included project name, year, description, size (area or length), and location (latitude and longitude). Data gaps were supplemented by archival research, site visits, contacting entities, and expert knowledge from local professionals.

### 2.6 Opportunity areas and restoration potential

Spatially explicit estimates of the opportunity areas and their restoration potential in the Tampa Bay watershed were obtained using a spatial analysis shown in [Figure 2](#fig-gisworkflow). The two main processes included 1) binning existing datasets into relevant categories and 2) overlaying multiple datasets to identify opportunities. Opportunity areas were defined as locations where habitat protection and restoration activities are possible and where they could occur to attain the targets and goals described above. Identifying opportunity areas is necessary to quantify the restoration potential for a particular habitat type, which is a measure of what is actually possible given underlying soil conditions, expected land use change, and sea-level rise. The identification of these areas on a broad spatial scale serves as a planning tool for restoration practitioners, where follow-up assessments are expected to more fully quantify restoration potential at selected sites.

The Land Use Land Cover 2017 dataset from SWFWMD was used for binning existing coverages into the relevant habitat types in the intertidal and subtidal strata. All FLUCCS classification codes were placed into one of three categories. First, native habitats were those that included the full range of natural plant communities and other habitats that are endemic to the watershed Second, restorable habitats included existing altered but non-hardened and pervious FLUCCS codes that could potentially support native habitats through restoration. Third, existing development included developed land FLUCCS codes that are hardened and impervious (e.g., structures and pavement) and not suitable for habitat restoration.

After binning, the native and restorable lands were overlaid with additional layers to identify 1) coastal reservation native and coastal reservation restorable areas, and 2) existing and proposed native and restorable areas. Collectively, each of these unique products are considered the opportunity areas in the Tampa Bay watershed (Table 1). The coastal reservation native and coastal reservation restorable areas are native and restorable habitats, respectively, that occur in the 5-foot contour or coastal stratum. Native habitats in this stratum were identified as those to be reserved, whereas restorable habitats were identified as those where tidal wetlands or coastal uplands could be restored.

The existing and proposed native and restorable areas were those that occurred in public lands that are currently acquired or proposed for acquisition. To identify these areas, native and restorable lands were intersected with data created from the Florida Natural Areas Inventory and permit databases of conservation and drainage easements. This workflow created the existing conservation and proposed conservation layers in [Figure 2](#fig-gisworkflow). Intersecting the native and restorable lands in these areas produced four unique opportunity areas: existing conservation native, proposed conservation native, existing conservation restorable, and proposed conservation restorable.

All opportunity areas identified as restorable included coastal reservation restorable, existing conservation restorable, and proposed conservation restorable. To identify discrete habitat types that could be the goal of future restoration projects, restorable lands in the coastal stratum and on existing conservation areas (coastal reservation native and existing conservation restorable) were further grouped into their restoration potential by underlying soil types. Proposed conservation areas were excluded from the analysis to provide a more confident assessment of restoration potential in areas that have already been acquired (i.e., existing conservation) or are immediately threatened by sea-level rise and/or coastal development (i.e., the coastal stratum).

Compared to vegetation communities, soil characteristics typically change slowly (e.g., decades to centuries) in response to hydrologic impacts, unless physically disturbed (Osland et al., 2012; Stockmann et al., 2014). Therefore, soil distributions can be used to estimate historical habitat distribution and restoration potential. A soils suitability layer was used for the Tampa Bay watershed (Ries and Scheda, 2014) that classified soils as xeric, mesic, or hydric. The mesic and hydric categories were combined to represent wetland restoration potential and the xeric category was used to represent upland restoration potential. A distinction was made between tidal and freshwater wetland restoration potential by intersecting the mesic and hydric soils with the coastal stratum. Mesic or hydric soils that occur below the 5-foot contour were assigned a restoration potential for tidal wetlands, whereas mesic or hydric soils above the 5-foot contour were assigned a restoration potential for freshwater wetlands. This distinction explicitly accounts for potential salinity changes to soil properties as a function of sea-level rise based on regional projections in the time period for establishing the targets and goals.

Two distinct mapping products were created from the above analysis. The first was an opportunities map that showed areas in the watershed identified as existing conservation native, existing conservation restorable, proposed conservation native, proposed conservation restorable, coastal reservation native, and coastal reservation restorable. The second was a map that identified the restorable lands (either existing or coastal reservation) based on their restoration potential as coastal uplands, freshwater wetlands, native uplands, or tidal wetlands. All spatial analyses described above and as outlined in [Figure 2](#fig-gisworkflow) were conducted using the R statistical programming language (R Core Team, 2022), specifically leveraging functions from the tidyverse package for data wrangling (Wickham et al., 2019) and the simple features (sf) package for geospatial analysis (e.g., the *st\_intersection* and *st\_union* functions for intersect and union operations, Pebesma, 2018). The workflows and data are provided in an open-access repository available on GitHub (<https://github.com/tbep-tech/hmpu-workflow>) (Beck et al., 2022).

## 3 Results

### 3.1 Habitat status and trends

Current estimates and trend information on subtidal habitats were available for seagrasses, tidal flats, and oyster bars (Table 2). Oyster bars were estimated at 67 ha in 2018 (Table 2), showing a 29% increase since mapping began in 2014. This increase may represent improved ground-truthing and photointerpretation, rather than a true increase in coverage. Tidal flats have generally increased from 1988 to the mid-2000s, followed by a decrease to present. The current estimate for tidal flats is 6,569 hectares, showing a 24% decline compared to the 1988 estimate of 8,700 hectares. Seagrasses have increased by 75% (6,986 ha) since 1988 to a current estimate of 16,293 ha. The change analysis comparing 1988 to 2018 for subtidal habitats ([Figure 3](#fig-subtchgdatalluv)) confirmed trends in Table 2 and showed that the seagrass increases were primarily associated with the colonization of non-vegetated areas of tidal flats, as well as unclassified areas of open water. Current estimates for subtidal habitats without historical trend information included 171 ha for hard bottom habitat and 67 hectares for artificial reefs.

Total intertidal habitat (mangroves, salt barrens, and salt marshes) increased by 12% to 8,340 ha from 1990 to 2017 (Table 3). Mangroves increased by 15% to 6,276 ha, salt barrens increased by 7% to 203 ha, and salt marshes increased by 3% to 1,861 ha. Despite a net increase in salt marsh habitat, the change analysis showed that 153 ha were replaced by mangroves ([Figure 4](#fig-chgdatalluv)). The current extent of tidal tributary length is 622 km (no trend information is available).

Trend assessments for supratidal habitats showed the effects of increasing land development and loss of restorable habitats in the Tampa Bay watershed (Table 3). Developed lands increased by 44% to 217,047 ha from 1990 to 2017. Coastal uplands decreased by 30% to 1,446 ha, native uplands decreased by 38% to 57,836 ha, and restorable lands decreased by 18% to 189,512 ha. Non-forested freshwater wetlands increased by 24% to 27,358 ha, whereas forested freshwater wetlands decreased by 5% to 61,667 ha. The change analysis ([Figure 4](#fig-chgdatalluv)) showed that a majority of conversion to developed lands came from restorable areas (21,292 ha) and native uplands (7,184 ha), with smaller proportions converted from forested freshwater wetlands (1,407 ha) and coastal uplands (193 ha). Habitats converted to restorable areas primarily included native uplands (8,304 ha), forested freshwater wetlands (1,700 ha), and developed lands (2,794 ha). The increase in non-forested freshwater wetlands was primarily from restorable lands (2,759 ha).

### 3.2 Habitat restoration and enhancement

A total of 460 restoration projects were documented in Tampa Bay and its watershed between 1971 and 2019. These projects were divided among habitat types that included estuarine (n = 228), freshwater (n = 53), uplands (n = 119), and a mix of all three (n = 60). A total of 1,978 ha have been restored, whereas 12,930 ha and 42.8 km (as shoreline or tributaries) were enhanced. Forty partners were responsible for these projects, although some were from departments within the same agency. Eighty-nine living shoreline projects, seawall enhancements, and oyster reef installations were documented, totaling 18.2 km. Although projects were documented for the whole period of record, few projects were completed prior to 1990. From 1990 to 2010 and from 2010 to 2019, an annual mean of 68 ha/yr and 81 ha/yr of habitat was restored, respectively. These means were used to define appropriate expectations for future restoration, described below.

### 3.3 Summary of opportunity areas and restoration potential

The current extent of each habitat type is shown in Table 4 as summaries for the opportunity areas and restoration potential. The extent of each habitat in existing conservation lands and proposed conservation lands is shown. Summaries of the restoration potential under existing and proposed conservation lands is also shown. Most restoration opportunities on existing conservation lands are for native uplands and freshwater wetlands. Less opportunities exist for intertidal wetlands (mangrove forests, salt barrens, and salt marshes). These summaries are also shown spatially in [Figure 5](#fig-oppmap) and [Figure 6](#fig-restmap).

The map of the remaining opportunity areas provided a spatial summary of where practitioners could target future restoration projects ([Figure 5](#fig-oppmap)). Native habitats currently protected (existing conservation native), proposed for protection (proposed conservation native), or in the coastal stratum (coastal reservation native) totaled 119,410 ha (20.3% of the watershed above MLLW). Similarly, restorable lands currently protected (existing conservation restorable), proposed for protection (proposed conservation restorable), or in the coastal stratum (coastal reservation native) totaled 83,423 ha (14.2% of the watershed). Understandably, most of the native and restorable lands occurred in undeveloped areas in northern and southeastern areas of the watershed ([Figure 6](#fig-restmap)). Existing conservation lands (existing conservation native, existing conservation restorable) totaled 79,396 ha (13.5% of the watershed) and proposed conservation lands (proposed conservation native, proposed conservation restorable) totaled 123,437 ha (21% of the watershed). Reservation areas in the coastal stratum (coastal reservation native, coastal reservation restorable) totaled 6,498 ha (1.1% of the watershed).

Combining the restorable lands on existing conservation areas and in the coastal stratum with soils data provided a spatial summary of the restoration potential grouped by habitat type ([Figure 6](#fig-restmap)). A total of 17,205 ha (2.9% of the watershed) of potentially restorable lands were identified, further partitioned as coastal uplands (128 ha, < 0.1% of the watershed), freshwater wetlands (11,034 ha, 1.9% of the watershed), native uplands (5,419 ha, 0.9% of the watershed), or tidal wetlands (624 ha, 0.1% of the watershed).

### 3.4 Establishment of targets and goals

Identifying short-term (2030) targets and long-term (2050) goals for the restoration extent of native habitats in Tampa Bay was informed by the assessment of current extents, past trends, and relative effort for past restoration and enhancement projects. Table 5 shows the targets and goals identified through this analysis and the associated rationale. For example, the targets and goals are established based on the current extent and the restoration potential. If restoration potential exists and coverage restored from past projects suggests a reasonable level of effort, the targets and goals reflect the current extent plus the restoration opportunity. Conversely, other habitats with no identified restoration opportunity, or with sufficient current extents (e.g., mangrove forests), were assigned targets and goals similar to the current extent, i.e., these habitats should be protected and further restoration will only increase resilience. Implicit in the targets and the goals is recurring re-assessment over time to evaluate progress and adjust expectations as appropriate.

## 4 Discussion

Priorities for comprehensive, watershed-wide habitat restoration should be informed by current assessments and what is possible to achieve in the future. These priorities are necessary given anticipated impacts of land development and climate change, while also considering competing societal interests for use of the environment and limited resources for land acquisition and restoration. Our approach balances these tradeoffs by identifying targets and goals that are informed by current extent, past trends, and realistic effort from past projects. Further, spatially explicit locations are identified where these targets and goals could be achieved based on existing opportunities for restorable habitats, including areas anticipated to be impacted by coastal stressors (i.e., sea-level rise and land development). This approach departs from previous restoration paradigms by identifying what is possible rather than attempting to recreate an ideal historical baseline. Methods are also provided using open source tools (Beck et al., 2022) that 1) allow for the most current datasets to be synthesized to assess progress, and 2) can be used in other locations with similar needs for identifying restoration priorities.

### 4.1 Habitat trends

Identifying appropriate targets and goals would not have been possible without a detailed assessment of current extent and past trends over thirty years of native habitats in Tampa Bay and its watershed. The most notable trends included 1) an increase of seagrasses by 75%, 2) an increase of emergent tidal wetlands (12%) and freshwater wetlands (24%), and 3) a loss of native uplands (38%).

Seagrass recovery in Tampa Bay is a well-known success story that demonstrated how public-private partnerships can effectively reduce total nitrogen loads into Tampa Bay (Greening et al., 2014; Sherwood et al., 2017). The nutrient reductions, primarily from point-source controls and advanced wastewater treatment, contributed to improvements in water quality and light environments that were favorable for seagrass growth. Reducing nitrogen inputs into Tampa Bay remains the primary strategy for maintaining water quality conditions. However, the most recent (2020) coverage estimate showed a seagrass loss of 18% baywide since peak coverages in 2016, falling below the target defined herein. These data were unavailable at the time this habitat restoration workflow was initially developed and trends informed by the new restoration paradigm have prompted bay managers to assess barriers in achieving the seagrass restoration goal. In particular, much of the seagrass losses have occurred in Old Tampa Bay (northwest segment of Tampa Bay), where recurring algal blooms of *Pyrodinium bahamense* have contributed to water quality decline (Lopez et al., 2019). The greatest percent loss of seagrass in 2020 was observed in Hillsborough Bay (northeast segment of Tampa Bay), which does not experience *P. bahamense* blooms. Ongoing research to understand mechanisms for mitigating blooms that negatively affect water quality, in addition to identifying potential regional stressors, will be critical for restoring seagrass in Tampa Bay.

Emergent tidal and freshwater wetlands in Tampa Bay have also experienced dramatic changes over the last three decades. Dual pressures from sea-level rise and changes in the length of the freeze-free season have affected tidal wetlands, such that mangrove forests are outcompeting salt marshes and salt barrens for available niche space. Mangrove expansion as a result of climate change has been observed throughout the Gulf of Mexico (Comeaux et al., 2012; Osland et al., 2022). Anthropogenic water withdrawals have also reduced freshwater flows reaching tidal marshes, contributing to reductions in coverage of key species (e.g., *Juncus roemerianus*) in place of mangroves (Raabe et al., 2012). As such, the identified targets and goals for mangroves indicate protection of these habitats, without the need for additional restoration. However, mangroves are expected to continue colonization of the intertidal zone, contributing to additional losses of salt marshes and salt barrens. The reservation areas identified in [Figure 5](#fig-oppmap) represent critical remaining areas in the intertidal zone that could be protected to prevent additional losses of tidal wetlands. Likewise, gains in non-forested freshwater wetlands are a reflection of 1) constructed stormwater ponds required by state and federal regulatory programs, and 2) the cumulative gains from publicly-funded habitat restoration projects. Creative restoration approaches (e.g., habitat acquisition and optimal management of freshwater flows) that address the likely expansion of mangroves at the expense of salt marshes and salt barrens will be required to meet the targets and goals for these habitats.

The decrease in native uplands is the result of continued development in the Tampa Bay watershed ([Figure 4](#fig-chgdatalluv)) and lack of regulatory protection of these habitats. Attaining the target and goal will require restoration of upland habitats on existing conservation lands (i.e., restoration potential in [Figure 6](#fig-restmap)) and new conservation lands to offset the continued loss of these habitats to development. The long-term conversion of restorable areas to developed lands ([Figure 4](#fig-chgdatalluv)) presents additional challenges for restoration of native uplands. Additional education about best practices in land development, market-based incentives, and amendments to existing planning, zoning, and land development regulations will be needed to address these issues. Although federal and state regulations for endangered species provide some protection to rare habitats, such as scrub jay (*Aphelocoma coerulescens*) habitat, common and historically abundant native habitats are largely unprotected (e.g., pine flatwoods). Voluntary approaches to low impact urban or suburban development may also gain traction among developers as more viable methods for land conversion that minimize impacts to natural resources while increasing quality of life (Jones et al., 2009). Education and outreach activities that target land developers to raise awareness of the benefits of alternative practices are critical in this effort. Market-based approaches to mitigating urban sprawl may also be practical (e.g., conservation subdvisions, Mohamed, 2006) given the estimated economic gains relative to conventional approaches. Regardless, reductions in native uplands will likely continue in the short-term unless local governments improve regulatory protections, such as strengthening language within comprehensive plans and development regulations to maintain a defined extent of these habitats within a rapidly urbanizing coastal watershed.

### 4.2 Achieving restoration targets and goals

Achieving the defined targets and goals will require diverse approaches for habitat restoration and management. Focusing efforts on publicly-owned conservation lands is expected to have long-term benefits and will be most cost-effective given the level of restoration effort compared to habitats that have already been impacted by anthropogenic activities. As such, public acquisition of remaining critical lands (e.g., coastal uplands) is a high priority given current development trends in the watershed. Other restoration targets (e.g., salt marshes) will not be obtained without additional public acquisition or initiating novel public-private partnerships as a mechanism for doing so (Holl and Howarth, 2001; Benson et al., 2018). Therefore, varied approaches to leverage resources for restoration are needed and could include pursuing traditional grants, matching funds from multiple partners, or voluntary initiatives that incentivize habitat restoration (e.g., Blue Carbon investments, Sheehan et al., 2019). Recent gains in tidal and freshwater wetlands are primarily due to publicly-funded habitat restoration projects, state and federal wetland regulatory programs, and to a lesser extent, regulatory mitigation. Restoration activities for habitats without similar regulatory frameworks should pursue the options above to achieve the defined targets and goals.

Other restoration activities could be pursued for the opportunity areas. Substantial opportunities exist for upland restoration on reclaimed mined lands within the watershed ([Figure 1](#fig-lulcmap), [Figure 5](#fig-oppmap)). For estuarine habitats, opportunity areas could include dredged holes or spoil disposal areas, either for enhancing existing subtidal habitats or creating areas that could be colonized by seagrasses. Some opportunities also exist on developed lands primarily through enhancement projects, although these have not been explicitly identified in the products herein. Examples include the construction of living shorelines in place of hardened seawalls, placement of submerged habitat modules along urban shorelines (e.g., artificial oyster reefs), and creation of backyard habitats. Tidal tributary restoration could also include removal of salinity barriers and filling of dredged channel sections. Overall, restoration practitioners must consider several options and choose those that are most feasible given the available resources and likelihood of success. Further, finer-scale land cover classification datasets are currently being investigated to refine identification of opportunity areas within the urbanized, developed landscape of the watershed.

Creative approaches may be required in areas affected by sea-level rise if land acquisition is not possible. These approaches are necessary to accommodate future landward migration of tidal wetlands or the protection of coastal uplands, while also reducing risks to built infrastructure that, when inappropriately sited, can inhibit landward habitat shifts. Coastal setbacks, buffers, or public easements are traditionally used to restrict development within a given distance from the shoreline. However, rolling easements may be an alternative approach whereby protected areas are allowed to “roll” landward with expected changes in sea-level rise. Rolling easements could disincentivize more intense urban development of low-lying coastal uplands in less developed agricultural or recreational land uses. Landowners could maintain current economic uses with a rolling easement, while reserving such lands to accommodate future landward habitat migration. These approaches also offer risk-reduction to built infrastructure by offering increased protection from potential affects of sea-level rise and other coastal stressors (e.g., storm surge).

Finally, wetland impacts and associated compensatory mitigation projects authorized under wetland regulatory programs could serve as more directed restoration mechanisms to help achieve watershed-wide goals. Mitigation activities have historically been conducted independent of watershed-level planning and monitoring processes. This disconnect has contributed to fragmented implementation, marginal habitat function, and inconsistent compliance monitoring of mitigation projects, including historically poor documentation of wetland losses and gains in the Tampa Bay watershed. However, if properly focused and coordinated, compensatory mitigation activities could significantly contribute to the attainment of restoration targets and goals for the region.

### 4.3 Limitations of the approach

Identifying restoration priorities was data intensive and would not have been possible without the resources available for the region. The workflow for identifying priorities required detailed and spatially explicit datasets specific to the Tampa Bay watershed. Long-term datasets describing land use and cover and the extent of subtidal habitats were necessary to categorize current extent and past trends. Similarly, supporting datasets included those that described existing and proposed conservation areas, soils, past restoration activities, and relevant spatial boundaries (i.e., watershed and coastline). Many of these datasets are available outside of Tampa Bay, although temporal and spatial resolutions may limit application to other areas. Despite the region’s data richness, limitations still exist in classifying restoration opportunities based on the spatial-scale of the LULC datasets. Additional refinements within the classified developed lands and coastal reservation space are currently being explored with 1-m scale national land cover datasets, which could expose additional opportunities in urban or suburban areas (e.g., improving stormwater infrastructure) or differential changes within native habitat classes (e.g., interspecific differences in mangrove colonization). Additionally, considerable effort was made in working with regional partners to identify and fill knowledge gaps for relevant habitat types. For example, inventories of hard bottom habitats, living shorelines, tidal tributaries, and artificial reefs were created through special studies or were available only as current estimates from regional entities. Tracking progress towards these habitat targets and goals is heavily reliant on regular updates to these datasets, as well as routine land use and cover map updates.

An additional assumption of the workflow, particularly for tracking progress, is that implemented restoration projects reported by partners will ultimately manifest into a classification within the map products. Specifically, restoration effort by regional partners is cataloged in the available restoration database, which not only depends on voluntary reporting, but also represents a source of information on restoration extent that is separate from land cover maps. An expectation is that the reported coverage restored by a partner will ultimately be shown as a change in land use and cover on regional maps. The temporal lag between an actual project and how it may be reflected in a GIS product is unknown, which may create a disconnect between the updates in achieving targets and goals as new data layers are released and the effort reported by partners becomes represented within the data layers. The spatial resolution of mapping products may also be insufficient to detect habitat changes as reported in the restoration database. For these reasons, projects reported by partners are currently summarized separately from the assessments above that depend on GIS layers. Additional work is needed to reconcile these datasets for more streamlined reporting.

### 4.4 Conclusion

The establishment of targets and goals that account for climate change, development trajectories, land availability, and past restoration effort expands the restoration opportunities to a more comprehensive list of habitats for the entire watershed. Land acquisition is critical for attaining the defined targets and goals and will also provide new opportunities for outdoor access to the broader community. Successful restoration is also contingent on engaging multiple partners, non-governmental organizations, and private citizens. The products created herein will guide these efforts for the next thirty years by providing a continuously updated assessment of where the opportunities exist and if targets and goals are expected to be met. The Tampa Bay region is not unique in the challenges resource managers face to protect and restore native habitats, and the approach described herein is readily transferable to other locations where restoration priorities are needed in response to pervasive coastal stressors.

## Acknowledgments

The Technical Advisory Committee of the Tampa Bay Estuary Program reviewed drafts of the original reports that are the basis of this paper and their comments are immensely appreciated. We also recognize the extensive efforts of regional partners, of which there are too many to mention, for past restoration efforts in the Tampa Bay region that have contributed to native habitats. These efforts will be critical for continued success of restoration in the future. Substantial efforts by partners in creating and curating routine datasets for the region, especially by the Southwest Florida Water Management District and the Florida Natural Areas Inventory, are also gratefully recognized.

## Figures

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| Figure 1: Land use, cover, and subtidal habitats for the Tampa Bay watershed, Florida, USA. The watershed includes the natural hydrologic boundary with minor modifications to include partners working with the Tampa Bay Estuary Program. Categories are based on the Florida Land Use Cover and Forms Classification System (Florida Department of Transportation, 1999; Southwest Florida Water Management District, 2014; Kawula and Redner, 2018) with relevant codes combined for presentation in the figure. |

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| Figure 2: Spatial analysis workflow used to identify opportunity areas (existing conservation native, proposed conservation native, existing conservation restorable, proposed conservation restorable, reservation native, reservation restorable) and restoration potential (coastal uplands, uplands, freshwater wetlands, tidal wetlands) in the Tampa Bay watershed. Workflows are divided into binning of land use/land cover categories into relevant habitat types and spatial overlay of datasets to identify the opportunity areas and restoration potential. The approach was applied to both the intertidal and supratidal strata of the watershed. |

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| Figure 3: Change analysis of habitat categories in the subtidal strata of the Tampa Bay watershed. The left column shows relative areas in 1988 and the right column shows relative areas in 2018 for each habitat category, where the bar heights are proportional to extents in each year. The grey lines show the proportional change in area of each habitat category between the years. |

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| Figure 4: Change analysis of habitat categories in the intertidal and supratidal strata of the Tampa Bay watershed. The left column shows relative areas in 1990 and the right column shows relative areas in 2017 for each habitat category, where the bar heights are proportional to extents in each year. The grey lines show the proportional change in area of each habitat category between the years. |

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| Figure 5: Opportunity areas for habitat restoration in the Tampa Bay watershed. Green indicates existing conservation, blue indicates proposed conservation, and pink indicates reservation opportunities. Each category is also grouped into native and restorable habitats. The outline is the Tampa Bay watershed. |

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| Figure 6: Habitat restoration potential in the Tampa Bay watershed. Areas are identified as those where habitat restoration could target the four identified categories as coastal uplands, freshwater wetlands, native uplands, or tidal wetlands. Categories are based on the opportunity areas, soil types, and coastal boundaries. The outline is the Tampa Bay watershed. |

## Tables

Table 1: Description of the opportunity areas in the Tampa Bay watershed identified through spatial analysis. Native habitats include those in the watershed not considered developed or restorable (e.g., freshwater wetlands, forested uplands, etc.). Restorable areas are altered but non-hardened and pervious lands that could potentially support native habitats through restoration. Existing and proposed conservation areas are those that are publicly owned or on conservation easements that currently exist or are proposed for acquisition, respectively, as identified primarily in the Florida Natural Areas Inventory. Coastal reservation areas occur within the coastal stratum identified from the bay shoreline to the 5-foot elevation contour.

| Opportunity area | Description |
| --- | --- |
| Existing Conservation Native | Native habitats currently within existing conservation lands |
| Existing Conservation Restorable | Restorable areas currently within existing conservation lanbs |
| Proposed Conservation Native | Native habits within proposed conservation lands |
| Proposed Conservation Restorable | Restorable areas within proposed conservations lands |
| Coastal Reservation Native | Native habitats within the coastal stratum |
| Coastal Reservation Restorable | Restorable areas within the coastal stratum |

Table 2: Change over time in hectares for subtidal habitats in Tampa Bay. Columns show years with available data and the final two columns show the change and percent change from 1988 to 2018. Oyster bars were not meaningfully quantified prior to 2014.

| Habitat Type | 1988 | 1990 | 1992 | 1994 | 1996 | 1999 | 2001 | 2004 | 2006 | 2008 | 2010 | 2012 | 2014 | 2016 | 2018 | 1988 to 2018 | % change |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
| Seagrasses | 9,307 | 10,086 | 10,299 | 10,609 | 10,758 | 9,920 | 10,417 | 10,795 | 11,309 | 11,862 | 13,171 | 13,874 | 16,153 | 16,701 | 16,293 | 6,986 | 75 |
| Tidal Flats | 8,700 | 8,207 | 8,272 | 8,117 | 8,199 | 10,878 | 10,300 | 11,601 | 11,387 | 10,878 | 9,617 | 8,714 | 5,976 | 5,557 | 6,569 | -2,130 | -24 |
| Oyster Bars | - | - | - | - | - | - | - | - | - | - | - | - | 52 | 65 | 67 | - | - |

Table 3: Change over time in hectares for intertidal and supratidal habitats in Tampa Bay. Columns show years with available data and the final two columns show the change and percent change from 1990 to 2017.

| Stratum | Habitat Type | 1990 | 1995 | 1999 | 2004 | 2007 | 2011 | 2014 | 2017 | 1990 to 2017 | % change |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
| Intertidal |  |  |  |  |  |  |  |  |  |  |  |
|  | Mangrove Forests | 5,472 | 5,808 | 5,793 | 6,318 | 6,300 | 6,299 | 6,266 | 6,276 | 804 | 15 |
|  | Salt Barrens | 189 | 194 | 199 | 197 | 185 | 203 | 199 | 203 | 14 | 7 |
|  | Salt Marshes | 1,814 | 1,795 | 1,798 | 1,877 | 1,874 | 1,863 | 1,939 | 1,861 | 47 | 3 |
| Supratidal |  |  |  |  |  |  |  |  |  |  |  |
|  | Coastal Uplands | 2,055 | 2,122 | 2,014 | 1,672 | 1,515 | 1,498 | 1,999 | 1,446 | -609 | -30 |
|  | Developed | 150,724 | 159,180 | 171,066 | 193,986 | 203,438 | 209,081 | 214,710 | 217,047 | 66,324 | 44 |
|  | Forested Freshwater Wetlands | 64,573 | 63,766 | 62,726 | 63,109 | 62,258 | 62,081 | 63,562 | 61,667 | -2,906 | -5 |
|  | Native Uplands | 93,076 | 83,850 | 75,313 | 64,482 | 61,277 | 60,319 | 62,794 | 57,836 | -35,239 | -38 |
|  | Non-Forested Freshwater Wetlands | 22,037 | 20,831 | 20,710 | 23,662 | 26,363 | 27,893 | 27,972 | 27,358 | 5,320 | 24 |
|  | Restorable | 231,288 | 232,195 | 228,531 | 212,549 | 201,609 | 195,529 | 184,342 | 189,512 | -41,777 | -18 |

Table 4: Summary of habitat restoration opportunities in the Tampa Bay watershed. Summaries are based on 2017 land use data, 2018 subtidal data, best estimates for habitat types not in existing GIS layers, and current extent of existing and proposed conservation lands. Proposed conservation lands are those identified for acquisition. Total restoration opportunity does not account for lands currently existing or proposed for conservation. N/A: not applicable, I/D: insufficient data.

|  | | Native Habitats | | | Restorable Habitats | | |
| --- | --- | --- | --- | --- | --- | --- | --- |
| Stratum | Habitat Type | Current Extent | Existing Conservation Lands | Proposed Conservation Lands | Total Restoration Opportunity | Existing Conservation Lands Restoration Opportunity | Proposed Conservation Lands Restoration Opportunity |
| Subtidal | | | | | | | |
|  | Hard Bottom | 171 ha | 171 ha | N/A | N/A | N/A | N/A |
|  | Artificial Reefs | 88 ha | 88 ha | N/A | N/A | N/A | N/A |
|  | Tidal Flats | 6,569 ha | 6,569 ha | N/A | I/D | I/D | N/A |
|  | Seagrasses | 16,293 ha | 16,293 ha | N/A | 5,719 ha | 5,719 ha | N/A |
|  | Oyster Bars | 67 ha | 67 ha | N/A | I/D | I/D | N/A |
| Intertidal | | | | | | | |
|  | Living Shorelines | 18 km | N/A | N/A | N/A | N/A | N/A |
|  | Mangrove Forests | 6,276 ha | 4,516 ha | 1,604 ha | 1,044 ha | 521 ha | 522 ha |
|  | Salt Barrens | 203 ha | 177 ha | 25 ha |
|  | Salt Marshes | 1,861 ha | 881 ha | 917 ha | 527 ha | 102 ha | 424 ha |
|  | Tidal Tributaries | 622 km | N/A | N/A | N/A | N/A | N/A |
| Supratidal | | | | | | | |
|  | Coastal Uplands | 1,446 ha | 722 ha | 664 ha | 513 ha | 128 ha | 385 ha |
|  | Non-Forested Freshwater Wetlands | 27,358 ha | 4,761 ha | 10,353 ha | 63,705 ha | 11,034 ha | 52,671 ha |
|  | Forested Freshwater Wetlands | 61,667 ha | 24,052 ha | 22,399 ha |
|  | Native Uplands | 57,836 ha | 27,083 ha | 21,256 ha | 17,637 ha | 5,419 ha | 12,217 ha |

Table 5: Recommended 2030 targets and 2050 goals for habitat restoration and protection in the Tampa Bay watershed. Targets and goals are based on 2017 land use data, 2018 subtidal data, best estimates for habitat types not in existing GIS layers, and current extent of existing and proposed conservation lands. Total restoration opportunity does not account for lands currently existing or proposed for conservation. N/A: not applicable, I/D: insufficient data.

| Stratum | Habitat Type | Current Extent | Total Restoration Opportunity | 2030 Target | 2050 Goal | Target Narrative and Restoration and Protection Rationale |
| --- | --- | --- | --- | --- | --- | --- |
| Subtidal | | | | | | |
|  | Hard Bottom | 171 ha | N/A | >171 ha | >171 ha | Protect existing hard bottom; continue to identify new hard bottom area through mapping |
|  | Artificial Reefs | 88 ha | N/A | >88 ha | >88 ha | Protect existing artificial reefs; enhance habitat complexity where feasible; expand reef area to promote fish and wildlife benefits |
|  | Tidal Flats | 6,569 ha | I/D | 6,564 ha | 6,564 ha | Identify and protect existing tidal flats; assess restoration potential of other non-vegetated subtidal areas |
|  | Seagrasses | 16,293 ha | 5,719 ha | >16,188 ha | >16,188 ha | Protect existing seagrasses; assess restoration potential of non-vegetated subtidal areas |
|  | Oyster Bars | 67 ha | I/D | 87 ha | 189 ha | 2030: Protect existing oysters and restore 20 hectares; increase target by 20 hectares each out-decade |
| Intertidal | | | | | | |
|  | Living Shorelines | 18 km | N/A | 34 km | 90 km | 2030: Construct 1.6 kilometers each year; better define opportunity areas; increase target to 2.4 and 3.2 kilometers per year for out decades |
|  | **Total Intertidal** | **8,340 ha** | **1,570 ha** | **8,745 ha** | **9,737 ha** | **2030: Protect existing intertidal mosaic and restore 405 hectares; increase target by 61 hectares each out-decade; includes the mosaic of mangrove, salt barren, and salt marsh habitats** |
|  | Mangrove Forests | 6,276 ha | 1,044 ha | >6,276 ha | >6,276 ha | Protect existing mangrove forests; restore opportunistically within the intertidal mosaic |
|  | Salt Barrens | 203 ha | 223 ha | 324 ha | 2030: Protect existing salt barrens and restore 20 hectares; increase target by 20 hectares per out decade |
|  | Salt Marshes | 1,861 ha | 527 ha | 1,962 ha | 2,225 ha | 2030: Protect existing low salinity salt marshes and restore 101 hectares; increase target by 20 hectares each out-decade |
|  | Tidal Tributaries | 622 km | I/D | 628 km | 651 km | Inventory mapped tidal tributaries and identify restoration potential; restore 6.4 kilometers of urban tidal creek habitat where feasible; increase target by 3.2 kilometers per out decade |
| Supratidal | | | | | | |
|  | Coastal Uplands | 1,446 ha | 513 ha | 1,507 ha | 1,689 ha | 2030: Protect existing coastal uplands and restore 61 hectares; increase target by 20 hectares each out decade |
|  | Non-Forested Freshwater Wetlands | 27,358 ha | 63,705 ha | 27,904 ha | 29,058 ha | 2030: Protect existing non-forested freshwater wetlands and restore 546 hectares; increase target by 20 hectares each out decade |
|  | Forested Freshwater Wetlands | 61,667 ha | 61,728 ha | 61,910 ha | 2030: Protect existing forested freshwater wetlands and restore 61 hectares; increase target by 20 hectares each out decade |
|  | Native Uplands | 57,836 ha | 17,637 ha | 58,018 ha | 58,443 ha | 2030: Protect existing native uplands and restore 182 hectares; increase target by 20 hectares each out decade; focus on pine flatwoods and protect current extent |

## References

Allaire, J. J., Gandrud, C., Russell, K., and Yetman, C. (2017). *networkD3: D3 JavaScript network graphs from R*. Available at: <https://CRAN.R-project.org/package=networkD3>.

Ash, T., and Runnels, R. (2005). Hard bottom habitats an overview of mapping and monitoring needs on epibenthic communities in Tampa Bay, Florida. in *Proceedings of the tampa bay area scientific information symposium (BASIS 4)*, ed. S. F. Treat (St. Petersburg, Florida: Tampa Bay Estuary Program), 179–182.

Ávila-García, D., Morató, J., Pérez-Maussán, A. I., Santillán-Carvantes, P., Alvarado, J., and FA., C. (2020). Impacts of alternative land-use policies on water ecosystem services in the rio grande de comitan-lagos de montebello watershed, mexico. *Ecosystem Services* 45. Available at: <https://doi.org/10.1016/j.ecoser.2020.101179>.

Beck, M. W., Raulerson, G. E., and Sherwood, E. T. (2022). *Tbep-tech/hmpu-workflow: v1.2.0*. Zenodo doi: [10.5281/zenodo.7032909](https://doi.org/10.5281/zenodo.7032909).

Benson, C. E., Carberry, B., and Langen, T. A. (2018). Public-private partnership wetland restoration programs benefit species of greatest conservation need and other wetland-associated wildlife. *Wetlands Ecology and Management* 26, 195–211. Available at: <https://doi.org/10.1007/s11273-017-9565-8>.

Bertness, M. D. (1985). Fiddler crab regulation of *spartina alterniflora* production on a New England salt marsh. *Ecology* 66, 1042–1055.

Brinson, M. M., Christian, R. R., and Blum, L. K. (1995). Multiple states in the sea-level induced transition from terrestrial forest to estuary. *Estuaries* 18, 648–659.

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

Calil, J., Beck, M. W., Gleason, M., Merrifield, M., Klausmeyer, K., and Newkirk, S. (2015). Aligning natural resource conservation and flood hazard mitigation in california. *PLoS One* 10, e0132651. Available at: <https://doi.org/10.1371/journal.pone.0132651>.

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>.

Cavanaugh, K. C., Kellner, J. R., Forde, A. J., Gruner, D. S., Parker, J. D., Rodriguez, W., et al. (2014). Poleward expansion of mangroves is a threshold response to decreased frequency of extreme cold events. *Proceedings of the National Academy of Sciences* 111, 723–727. Available at: <https://doi.org/10.1073/pnas.1315800111>.

Chung, M. G., Dietz, T., and Liu, J. (2018). Global relationships between biodiversity and nature-based tourism in protected areas. *Ecosystem Services* 218, 11–23. Available at: <https://doi.org/10.1016/j.ecoser.2018.09.004>.

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

Coen, L. D., Brumbaugh, R. D., Bushek, D., Grizzle, R., Luckenbach, M. W., Posey, M. H., et al. (2007). Ecosystem services related to oyster restoration. *Marine Ecology Progress Series* 341, 303–307.

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>.

Conner, W. H., Doyle, T. W., and Krauss, K. W. (2007). *Ecology of tidal freshwater forested wetlands of the southeastern United States*. Dordrecht, Netherlands: Springer Available at: <https://doi.org/10.1007/978-1-4020-5095-4>.

CSA Ocean Sciences (2019). Tampa Bay hard bottom mapping project. Tampa Bay Estuary Program, St. Petersburg, Florida.

Dontis, E. E., Radabaugh, K. R., Chappel, A. R., Russo, C. E., and Moyer, R. P. (2020). Carbon storage increases with site age as created salt marshes transition to mangrove forests in tampa bay, florida (USA). *Estuaries and Coasts* 43, 1470–1488. Available at: <https://doi.org/10.1007/s12237-020-00733-0>.

Duarte, C. M., Conley, D. J., Carstensen, J., and Sánchez-Camacho, M. (2009). Return to Neverland: Shifting baselines affect eutrophication restoration targets. *Estuaries and Coasts* 32, 29–36. Available at: <https://doi.org/10.1007/s12237-008-9111-2>.

Dupont, J. M. (2008). Artificial reefs as restoration tools: A case study on the West Florida shelf. *Coastal Management* 36, 495–507. Available at: <https://doi.org/10.1080/08920750802395558>.

Eisma, D. (1998). *Intertidal deposits: River mouths, tidal flats, and coastal lagoons*. London: CRC Press Available at: <https://doi.org/10.1201/9780138750308>.

Environmental Protection Agency (1996). Watershed approach framework. Office of Water (4501F).

Enwright, N. M., Griffith, K. T., and Osland, M. J. (2015). Incorporating future change into current conservation planning - evaluating tidal saline wetland migration along the U.S. Gulf of Mexico coast under alternative sea-level rise and urbanization scenarios. U.S. Geological Survey Available at: <https://dx.doi.org/10.3133/ds969>.

Ermgassen, P. S. E. zu, Spalding, M. D., and Grizzle, R. E. (2013). Quantifying the loss of a marine ecosystem service: Filtration by the eastern oyster in U.S. estuaries. *Estuaries and Coasts* 36, 36–43.

Florida Department of Transportation (1999). *Florida land use, cover and forms classification system*. Third. Tallahassee, Florida: FDOT Surveying; Mapping Office Geographic Mapping Section.

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

Gregory, R., Ohlson, D., and Arvai, J. (2006). Deconstructing adaptive management criteria for applications to environmental management. *Ecological Applications* 16, 2411–2425.

Heck, K., Hays, G., and Orth, R. (2003). Critical evaluation of the nursery role hypothesis for seagrass meadows. *Marine Ecology Progress Series* 253, 123–136.

Henningsen, B. (2005). The maturation and future of habitat restoration programs for the Tampa Bay estuarine ecosystem. in *Proceedings of the tampa bay area scientific information symposium (BASIS 4)*, ed. S. F. Treat (St. Petersburg, FL: Tampa Bay Estuary Program), 165–170.

Holl, K. D., and Howarth, R. B. (2001). Paying for restoration. *Restoration Ecology* 8, 260–267. Available at: <https://doi.org/10.1046/j.1526-100x.2000.80037.x>.

Holling, C. S. (1978). *Adaptive environmental assessment and management*. Chichester, UK: John Wiley & Sons.

Houde, E. D., and Rutherford, E. S. (1993). Recent trends in estuarine fisheries: Predictions of fish production and yield. *Estuaries* 16, 161–176. Available at: <https://doi.org/10.2307/1352488>.

Hsieh, Y. P. (2004). “Dynamics of tidal salt barren formation and the record of present-day sea level change,” in *The ecogeomorphology of tidal marshes*, eds. S. Fagherazzi, M. Marani, and L. Blum (Washington: American Geophysical Union), 231–245.

Jaap, W. C., and Hallock, P. (1990). “Coral reefs,” in *Ecosystems of florida*, eds. R. L. Myers and J. J. Ewel (Orlando, Florida, USA: University of Central Florida Press), 574–618.

Jones, P. H., Larson, B. C., and Clark, M. W. (2009). Reduced impact development practices at “restoration.” in *American institute of physics conference proceedings* (American Institute of Physics), 151–161.

Kaufman, K. (2017). Tampa Bay Environmental Restoration Fund final report: Hard bottom mapping and characterization for restoration planning in Tampa Bay. Tampa Bay Estuary Program, St. Petersburg, Florida.

Kawula, R., and Redner, J. (2018). Florida Land Cover Classification System. Center for Spatial Analysis, Fish; Widlife Research Institute, Florida Fish; Wildlife Conservation Commission, Tallahassee, Florida.

Kushlan, J. A. (1990). “Freshwater marshes,” in *Ecosystems of florida*, eds. R. L. Myers and J. J. Ewel (Orlando, Florida: University of Central Florida Press), 324–363.

Lewis, R. R., and Robison, D. E. (1996). Setting priorities for Tampa Bay habitat protection and restoration: Restoring the balance. Tampa Bay Estuary Program, St. Petersburg, Florida.

Lopez, C. B., Karim, A., Murasko, S., Marot, M., Smith, C. G., and Corcoran, A. A. (2019). Temperature mediates secondary dormancy in resting cysts of *Pyrodinium bahamense* (dinophyceae). *Journal of Phycology* 55, 924–935. doi: [10.1111/jpy.12883](https://doi.org/10.1111/jpy.12883).

Lowndes, J. S. S., Best, B. D., Scarborough, C., Afflerbach, J. C., Frazier, M. R., O’Hara, C. C., et al. (2017). Our path to better science in less time using open data science tools. *Nature Ecology & Evolution* 1, 1–7. doi: [10.1038/s41559-017-0160](https://doi.org/10.1038/s41559-017-0160).

Menéndez, P., Losada, I. J., Beck, M. W., Torres-Ortega, S., Espejo, A., Narayan, S., et al. (2018). Valuing the protection services of mangroves at national scale: The philippines. *Ecosystem Services* 34(A), 24–36. Available at: <https://doi.org/10.1016/j.ecoser.2018.09.005>.

Meyers, R. L., and Ewel, J. J. (1990). *Ecosystems of florida*. Orlando, Florida: University of Central Florida Press.

Mohamed, R. (2006). The economics of conservation subdivisions: Price premiums, improvement costs, and absorption rates. *Urban Affairs Review* 41, 376–399. doi: [10.1177/1078087405282183](https://doi.org/10.1177/1078087405282183).

Moore, H. B., Davies, L. T., Fraser, T. H., Gore, R. H., and López, N. R. (1968). Some biomass figures from a tidal flat in Biscayne Bay, Florida. *Bulletin of Marine Science* 18, 261–279.

National Oceanic and Atmospheric Administration (2015). *Guidance for considering the use of living shorelines, final guidance document prepared by the NOAA living shorelines workgroup*. Silver Spring, Maryland: National Oceanic; Atmospheric Administration.

Odum, W. E., and McIvor, C. C. (1990). “Mangroves,” in *Ecosystems of florida*, eds. R. L. Myers and J. J. Ewel (Orlando, Florida: University of Central Florida Press), 517–548.

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).

Osland, M. J., Spivak, A. C., Nestlerode, J. A., Lessmann, J. M., Almario, A. E., Heitmuller, P. T., et al. (2012). Ecosystem development after mangrove wetland creation: Plant-soil change across a 20-year chronosequence. *Ecosystems* 15, 848–866. Available at: <https://doi.org/10.1007/s10021-012-9551-1>.

Palmer, M. A. (2009). Reforming watershed restoration: Science in need of application and applications in need of science. *Estuaries and Coasts* 32, 1–17. Available at: <https://doi.org/10.1007/s12237-008-9129-5>.

Pebesma, E. (2018). Simple Features for R: Standardized support for spatial vector data. *The R Journal* 10, 439–446. doi: [10.32614/RJ-2018-009](https://doi.org/10.32614/RJ-2018-009).

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/>.

Raabe, E., Roy, L. C., and McIvor, C. (2012). Tampa Bay coastal wetlands: Nineteenth to twentieth century tidal marsh-to-mangrove conversion. *Estuaries and Coasts* 35, 1145–1162. Available at: <https://doi.org/10.1007/s12237-012-9503-1>.

Rayer, S., and Wang, Y. (2020). Projections of Florida population by county, 2020-2045, with estimates for 2019. *Florida Population Studies* 53, 186.

Restore America’s Estuaries (2015). *Living shorelines: From barriers to opportunities*. Arlington, Virginia: Restore America’s Estuaries.

Ries, T., and Scheda, S. (2014). Master plan for the protection and restoration of freshwater wetlands in the Tampa Bay watershed, Florida. Tampa Bay Estuary Program, St. Petersburg, Florida.

Robison, D. E. (2010). Tampa Bay Estuary Program Habitat Master Plan Update. Tampa Bay Estuary Program, Saint Petersburg, Florida.

Robison, D., Ries, T., Saarinen, J., Tomasko, D., and Sciarrino, C. (2020). Tampa bay estuary program: 2020 habitat master plan update. Tampa Bay Estuary Program, St. Petersburg, Florida Available at: <https://drive.google.com/file/d/1Hp0l_qtbxp1JxKJoGatdyuANSzQrpL0I/view?usp=drivesdk>.

Russell, M., and Greening, H. (2015). Estimating benefits in a recovering estuary: Tampa Bay, Florida. *Estuaries and Coasts* 38, 9–18. Available at: <https://doi.org/10.1007/s12237-013-9662-8>.

Sheehan, L., Sherwood, E. T., Moyer, R. P., Radabaugh, K. R., and Simpson, S. (2019). Blue Carbon: an Additional Driver for Restoring and Preserving Ecological Services of Coastal Wetlands in Tampa Bay (Florida, USA). *Wetlands* 39, 1317–1328. doi: [10.1007/s13157-019-01137-y](https://doi.org/10.1007/s13157-019-01137-y).

Sherwood, E. (2008). Tampa bay tidal tributary habitat initiative: Integrated summary document, Tampa Bay Estuary Program tidal tributaries project team. Tampa Bay Estuary Program, St. Petersburg, Florida.

Sherwood, E., Greening, H., Johansson, J. O. R., Kaufman, K., and Raulerson, G. (2017). Tampa Bay (Florida, USA): Documenting seagrass recovery since the 1980’s and reviewing the benefits. *Southeastern Geographer* 57, 294–319.

Sim, S., and Mesev, V. (2014). Measuring and modeling of urban growth and its impacts on vegetation and species habitats in greater orlando, florida. *International Journal of Geospatial and Environmental Research* 1, 1–5.

Smith, C. S., Puckett, B., Gittman, R. K., and Peterson, C. H. (2018). Living shorelines enhanced the resilience of saltmarshes to Hurricane Matthew (2016). *Ecological Applications* 28, 871–877. Available at: <https://doi.org/10.1002/eap.1722>.

Southwest Florida Water Management District (2014). Photo interpretation key for land use classification.

Southwest Florida Water Management District (2018). Land use land cover data. c1990-2017. Available at: <https://data-swfwmd.opendata.arcgis.com/>.

Southwest Florida Water Management District (2019). Seagrass in 2018. Available at: <https://data-swfwmd.opendata.arcgis.com/datasets/seagrass-in-2018>.

Sprandel, J. A., Gore, D., and Cobb, T. (2000). Distribution of wintering shorebirds in coastal florida. *Journal of Field Ornithology* 71, 708–720.

Stockmann, U., Minasny, B., and McBratney, A. B. (2014). How fast does soil grow? *Geoderma* 216, 48–61. Available at: <https://doi.org/10.1016/j.geoderma.2013.10.007>.

Theobald, D. M., Miller, J. R., and Hobbs, N. T. (1997). Estimating the cumulative effects of development on wildlife habitat. *Landscape and Urban Planning* 39, 25–36. Available at: <https://doi.org/10.1016/S0169-2046(97)00041-8>.

Titus, J. G., Hudgens, D. E., Trescott, D. L., Craghan, M., Nuckols, W. H., Hershner, C. H., et al. (2009). State and local governments plan for development of most land vulnerable to rising sea level along the US atlantic coast. *Environmental Research Letters* 4, 044008. Available at: <https://doi.org/10.1088/1748-9326/4/4/044008>.

Tomasko, D., Alderson, M., Burnes, R., Hecker, J., Iadevaia, N., Leverone, J., et al. (2020). The effects of Hurricane Irma on seagrass meadows in previously eutrophic estuaries in Southwest Florida (USA). *Marine Pollution Bulletin* 156, 111247. Available at: <https://doi.org/10.1016/j.marpolbul.2020.111247>.

Vogelmann, J. E., Xian, G., Homera, C., and Tolk, B. (2012). Monitoring gradual ecosystem change using Landsat time series analyses: Case studies in selected forest and rangeland ecosystems. *Remote Sensing of Environment* 122, 92–105. Available at: <https://doi.org/10.1016/j.rse.2011.06.027>.

Wessel, M. R., Leverone, J. R., Beck, M. W., Sherwood, E. T., Hecker, J., West, S., et al. (2022). Developing a water quality assessment framework for southwest Florida tidal creeks. *Estuaries and Coasts* 45, 17–37. Available at: <https://doi.org/10.1007/s12237-021-00974-7>.

Wickham, H., Averick, M., Bryan, J., Chang, W., McGowan, L. D., François, R., et al. (2019). Welcome to the tidyverse. *Journal of Open Source Software* 4, 1686. doi: [10.21105/joss.01686](https://doi.org/10.21105/joss.01686).

Yoskowitz, D., and Russell, M. (2015). Human dimensions of our estuaries and coasts. *Estuaries and Coasts* 38(S1), 1–8. Available at: <https://doi.org/10.1007/s12237-014-9926-y>.