



An Economic Evaluation of Panhandle Living Shorelines

– Final Report

**Prepared for:
FSU PC SASJBEP
CSTARS Project**

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Executive Summary

During the time period from years 2024 to 2026, the FSU Panama City SASJBEP and the Florida State University Center for Economic Forecasting and Analysis (CEFA) CSTARS team collaborated to quantify the ecosystem service benefits provided by living shorelines across three bays in the Florida panhandle: St. Andrew Bay, Choctawhatchee Bay, and Pensacola Bay. The purpose of this study is to provide an economic valuation of the ecosystem services generated by these living shorelines, which encompass a combined total of just over 26 acres of restored coastal habitat. The results demonstrate the substantial and measurable economic value these shorelines provide to the surrounding communities and environment across five ecosystem service categories. A brief summary of the findings is provided below.

I. Nitrogen release and storage. Living shorelines provide water quality benefits by removing and storing nitrogen through oyster filtration, denitrification, and sediment storage. Across the three bays, the living shorelines generated a combined annual nitrogen storage value of \$405,976 and a combined annual nitrogen removal value of \$322,335, for a total nitrogen-related economic benefit of \$728,311 per year. Economic values were estimated using the North Carolina Department of Mitigation Services nutrient offset credit rate of \$18.70 per pound of nitrogen. Pensacola Bay contributed the largest share of this value, at \$602,461 annually.

II. Property protection through erosion prevention. Living shorelines reduce coastal erosion, protecting the economic value of surrounding properties from land loss. Using property tax appraisal data from the 2025 Name, Address, and Legal Description (NAL) database and FEMA Special Flood Hazard Area designations, the total estimated value of property protection across the three bays was estimated with a lower and upper bound. The lower bound estimate of property protection value across all bays was \$2,187,058 and the upper bound estimate was \$7,105,475. The lower bound estimate reflects just value estimates at the parcel level, and the upper bound estimate reflects just value estimates at the 0.1 mi distance threshold level.

III. Recreational fisheries supported by living shoreline habitat. Living shorelines provide critical nursery habitat for recreationally and commercially significant fish and

shellfish species. Using population modeling and ex-vessel pricing from Florida Fish and Wildlife Conservation Commission commercial landings data, the total annual recreational fisheries value supported by the living shorelines was estimated at \$318,012 across the three bays.

IV. Carbon sequestration. Living shorelines sequester carbon in their sediments and biomass, providing climate-related benefits by removing carbon dioxide from the atmosphere. The annual economic value of carbon sequestration in 2024 is estimated to fall between \$44,667 and \$301,502 across the three bays, depending on whether carbon is priced using the Regional Greenhouse Gas Initiative (RGGI) cap-and-trade market rate of \$0.04 per pound of carbon or the EPA's Social Cost of Carbon estimate of \$0.27 per pound of carbon. These figures represent lower and upper bound estimates of value respectively.

V. Water clarity. Living shorelines reduce shoreline erosion and prevent sediment from entering adjacent coastal waters, improving water clarity. Applying hedonic property value estimates from Walsh et al. (2011) to estimated Secchi depth improvements across the three bays, annual water clarity benefits are estimated at \$1,236 for St. Andrew Bay, \$67,073 for Choctawhatchee Bay, and \$50,722 for Pensacola Bay, for a combined central estimate of \$119,031 with a sensitivity range of \$71,418 to \$392,801. These estimates carry notable uncertainty and should be treated as illustrative lower bounds.

Nitrogen Release and Storage

Methodology

I. Oysters

From 2024 to 2026, biologists quarterly monitored thirty living shorelines for live oyster sizes, counts, and densities at reef structures. At each site, the team monitored three reefs for oyster data, with three pseudoreplicates per reef structure. For reefs composed of bags of oyster shells or small limestones, the team carefully analyzed the bag and inspected the exposed surface for live oysters. The team then counted all live and dead oysters on the visible surface and measured sizes (from hinge to shell edge in mm) of up to ten live oysters. Biologists also captured images of these bags with ruler measurements included. These

images were processed for surface area estimation. For reefs composed of heavy limestones or concrete structures unable to be lifted, or bags of shells and limestones that were disintegrated, the team overlaid 0.5 x 0.5 m quadrat on the portion of the reef being monitored to quantify all live oyster sizes (up to ten oysters) and live oyster counts. For each year, quarter, and site, the team quantified each sampled replicate's live oyster densities by dividing the total number of live oysters by the surface area obtained using processed imagery, or the area of the quadrat = 0.25 m² as appropriate. Average density per year, quarter, and site (henceforth YQS) were then obtained by averaging across these replicates' live oyster densities.

The team then averaged live oyster sizes (mm) at each YQS, and used this average oyster size to calculate individual oyster filtration rate using the model developed by Pollack et al. (2013):

$$FR = \frac{(L^{0.96}T^{0.95})}{2.95}$$

where FR = filtration rate (ml/oyster/min), L = shell length (mm), and T = water temperature (°C).

According to Pollack et al. (2013), filtration rate depends on salinity and concentration of suspended solids. As a function of salinity, the original filtration rate was modified as follows:

$$\begin{aligned} \text{If: } S \geq 7.5 \text{ psu, Then: } FR_S &= FR \\ \text{If: } 3.5 \text{ psu} < S < 7.5 \text{ psu, Then: } FR_S &= \frac{FR \cdot (S - 3.5)}{4} \end{aligned}$$

$$\text{If: } S \leq 3.5 \text{ psu, Then: } FR_S = 0$$

where S = salinity (PPT or PSU).

As a function of total suspended solids, the filtration rate was further modified as follows:

$$FR_T = FR_S \cdot \left[1 - 0.01 \left(\frac{\log_{10}(TSS) + 3.8}{0.418} \right) \right]$$

where TSS = total suspended solids (g/lit).

The research team used the monitored average oyster size, salinity, and TSS to calculate the filtration rate at each YQS. While the team did not record TSS directly using YSI during monitoring efforts, TSS was estimated based on turbidity (NTU) readings by compiling all available TSS data throughout the service region from water quality stations databases at Florida Department of Environmental Protection Watershed Information Network (FDEP WIN) from 2024 to 2026. After gathering all FDEP WIN's available TSS data, linear regression was conducted using the turbidity readings that were recorded from the same water quality stations on the same date as TSS to generate a linear model of turbidity and TSS relationship. The model to convert field monitoring turbidities to TSS was as follows:

$$TSS = 0.6890447 \cdot Turb + 2.8967766$$

Where TSS = mg/L, and Turb = turbidity (NTU). Some of the service sites during third quarter of 2025 did not properly record turbidity due to equipment errors; the team replaced those values from nearest water quality stations in FDEP WIN database from the appropriate year and quarter.

According to Pollack et al. (2013), oysters' nitrogen removal rates depend on filtration rate and food concentration. Food concentration was calculated as follows:

$$Food = 0.085 \cdot Chl$$

Where Chl = concentration of chlorophyll-a (µg/lit).

All available Chl-a from FDEP WIN throughout the three bays from 2024 to 2026 was then gathered. The team used the numeric nutrient criteria (NNC) regions (or sub-bay regions) determined by FDEP to determine which water quality stations' Chl-a data will be applied to each YQS average-sized oyster. Chl-a values were averaged at each YQS per NNC region and applied to the respective sites within each NNC. Where no Chl-a data were available for a

specific quarter within an NNC, the bay-level average for that quarter was applied to sites in that NNC region.

Oyster nitrogen removal rates (or clearance rates) were calculated using the following formula by Pollack et al. (2013):

$$CR = FR_T \cdot Food \cdot 0.001 \frac{lit}{ml}$$

Where FR_T = filtration rate as a function of size, temperature, salinity, and TSS (ml/oyster/min), $Food$ = concentration of food based on Chl-*a* (µg/lit), and CR = clearance rate, or Nitrogen removal rate (µg N/oyster/min).

At each YQS, oyster clearance rate was converted to grams of Nitrogen per oyster per year, and the total number of live oysters was estimated by multiplying the average density (oysters/m²) by reef area. Reef areas were estimated using satellite imagery on Google Earth and measurement tools. Dimensions provided by partners were used if vegetation was impeding satellite imagery. For each year from 2024 to 2026, annual oyster nitrogen clearance rates were calculated by applying the per-oyster clearance rate to the estimated total live oyster population at each site, yielding total nitrogen removal expressed in both grams and pounds per year. An average across the three service years was used to obtain the average annual oyster nitrogen removal rate at each site.

II. Denitrification

In the year 2024, during the third quarter (July to September) at each site, three replicates of sediment samples (each replicate = 120 cm³) at the top 5-cm layer in marsh zones were collected. In 2025, this methodology was replicated. However, sampling in St. Andrew and Choctawhatchee Bays was done in the second quarter and sampling in Pensacola Bay was done in the third quarter. Sediment samples were shipped overnight to Rowan University on the same day as monitoring for immediate biogeochemistry processing to obtain potential maximum denitrification rates. Using the sediment biogeochemical analyses results, mass denitrification rate and sediment density, the team produced areal potential maximum denitrification rates as follows:

$$\text{Areal Rate} = \text{Mass rate} \cdot \text{Areal conversion} \cdot \text{Density}$$

Where Areal Rate = nmol N/m²/day, Mass rate = nmol N/gdw/day, Areal conversion = 50,000 cm³/m² (reflecting the 5 cm sediment sampling depth), and Density = gdw/cm³.

These areal rates were converted to mg Nitrogen/m²/day using the nitrogen atomic molar mass (14.01 g/mol); at each YQS, average areal rates were obtained by averaging these areal rates across all three replicates. Because these are potential maximum denitrification rates and not reflective of actual rates in nature, the team gathered available nitrate data from FDEP WIN and used the Michaelis-Menten equation to convert these rates to actual denitrification rates (Schutte et al. 2021, Kouanda & Hua 2021).

After gathering all available nitrate (mg N/lit) data from FDEP WIN at each of the service bays for the respective quarters, average nitrate values per NNC region were calculated to determine which nitrate values to apply to estimate actual denitrification. The following Michaelis-Menten equation was used to estimate actual denitrification rates:

$$V_i = \frac{(V_{max} \cdot [\text{Nitrate}])}{(K_m + [\text{Nitrate}])}$$

Where V_{max} = potential maximum denitrification rate (mg N/m²/day), [Nitrate] = nitrate concentration (mg N/lit), K_m = half-saturation constant, and V_i = actual denitrification rate (mg N/m²/day).

To determine half-saturation constant, the team reviewed literature with existing K_m values in salt marsh habitats from around the world. Values reported by Thompson et al. (1995) from salt marshes of Newport River Estuary in North Carolina were most appropriate for the denitrification conversions. Thompson et al. (1995) reported K_m ranges from 38 to 54.8; CEFA determined the mean K_m to be 46.4 after listing all K_m values from 38 to 54.8 at 0.1 increments.

After obtaining actual denitrification rates (mg N/m²/day) at each YQS, these values were converted to g N/m²/year and calculated annual nitrogen removal rate by multiplying these by total salt marsh areas per site (g N/year and lbs/year). This was averaged across the three years of monitoring to obtain the annual denitrification rate in monitored salt marshes.

III. Nitrogen Storage

At each YQS, the team collected nine replicates (each replicate = 120 cm³) of sediments at top 5-cm depth for carbon-nitrogen ratio (C/N) analyses. For sites without reefs, six sediment replicates were collected. These sediment samples were analyzed using an elemental analyzer (Marton & Roberts 2014) at the University of Florida's Stable Isotope Mass Spectrometry Facility. For each sediment sample, the following data was received (all units in mg): sediment mass, nitrogen content, and carbon content.

Using the nitrogen mass, nitrogen density was calculated by dividing nitrogen mass by 120 cm³ (mg/cm³). After converting this to g/m³, these densities were multiplied by total site sediment volume (area of the site * top 5-cm sediment depth = m³) to obtain each site's total nitrogen storage per YQS. The team then averaged this total nitrogen storage across all four quarters to obtain average total nitrogen storage (grams and pounds) for each of the monitoring years (2024-2026). This total nitrogen storage was also averaged across all three years.

Economic Results for Nitrogen Storage and Removal

Table 1. Complete Nitrogen Evaluation Results in 2026 Dollars

Bays	St. Andrew	Choctawhatchee	Pensacola	Total
Nitrogen Storage	\$12,021	\$100,056	\$293,898	\$405,976
Nitrogen Removal	\$1,080	\$12,692	\$308,563	\$322,335
Combined Value	\$13,102	\$112,748	\$602,461	\$728,311

To estimate the value of nitrogen, the North Carolina nutrient offset credit system was used. To reduce outlier bias, the median market rate for economic valuation was calculated via five North Carolina service areas: Jordan Lake Watershed, Neuse Program, Neuse Outside the Falls Lake Watershed, Neuse- Falls Lake Watershed, and the Tar-Pamlico Basin. The median market rate was \$18.70 per pound of nitrogen.

Applying the market rate to the estimated nitrogen stored at each site, the living shorelines at St. Andrew Bay, Choctawhatchee Bay, and Pensacola Bay generated annual economic benefits of \$12,021, \$100,056, and \$293,898, respectively, for a combined total of \$405,976 in nitrogen storage value across the three bays. The living shorelines at St. Andrew Bay, Choctawhatchee Bay, and Pensacola Bay generated estimated annual nitrogen removal benefits of \$1,080, \$12,692, and \$308,563, respectively, through the combined processes of filtration and denitrification, for a combined total of \$322,335 across the three bays. The total valuation attributable to nitrogen was \$728,311.

Property Protection

Methodology

Parcel boundary data for the three bays were drawn from the 2025 Name, Address, and Legal Description (NAL) files, which represent the official master list of real property parcels within a county, compiled from Florida county property appraisers and submitted to the Florida Department of Revenue. Shape files from this dataset were consolidated into a single polygon layer, providing parcel level information including land value, just value, square footage, and recent sales history. Site boundaries for the three CSTARS locations were incorporated as an additional spatial layer to delineate the study areas relative to surrounding parcels. All layers were reprojected into a common coordinate system (NAD 1983 HARN StatePlane Florida North FIPS 0903, US Feet) to ensure spatial consistency across datasets.

Flood risk was determined using FEMA's National Flood Hazard Layer for the State of Florida, from which Special Flood Hazard Areas (SFHAs) were isolated. These zones represent areas

with a 1% or greater annual probability of flooding. Parcel data were then spatially filtered to retain only those parcels intersecting a designated SFHA.

Sample sites were assigned to their nearest parcels using a spatial join with a closest feature criterion, relying on minimum straight-line distance. Because the geocoordinates of several sites were located offshore and therefore outside the interior of any parcel boundary, the closest parcel approach was used to identify the parcel most proximate to each site. In all cases, the closest parcel to each sample site also fell within a SFHA, meaning no additional filtering was necessary to distinguish flood risk status at that level of analysis. A distance threshold was applied to capture parcels within the broader vicinity of each site: 528 feet (0.1 miles). Parcels falling outside the cutoff were excluded from the analysis. A one-to-many spatial join was used to account for parcels that fell within range of multiple sample sites, which was relevant in cases where adjacent sites such as LOP2 and LOP3 had overlapping search radii. While individual parcels may therefore appear across more than one site, this does not affect the site level averages.

Parcel data were exported and aggregated by site, summing total land values, just values, and square footage. Average land value and just value per square foot were calculated for each site and converted to a per acre basis. These per acre values were then applied to the biology team's estimates of land loss avoided at each site to derive the total economic benefits of erosion prevention attributable to the living shorelines.

The property protection valuation rests on several key assumptions. Property tax appraisal assessments of land and just value are treated as accurate and economically sound proxies for the real value of land, and by extension, as reasonable measures of the avoided monetary damages associated with erosion-related land loss. FEMA's Special Flood Hazard Areas, defined by 1% or greater annual flood probability, are assumed to provide a reasonable basis for identifying parcels most susceptible to erosion risk, though it is acknowledged that this classification introduces a degree of risk homogeneity, as parcels within the SFHA may vary considerably in their actual flood exposure above the 1% threshold. The use of a distance threshold of 528 feet to filter parcels is assumed to represent a spatially and economically defensible means of identifying properties most likely to benefit from the living shorelines.

Finally, land values are the more conservative estimate of economic benefit, as shoreline erosion incrementally reduces physical land area over time rather than wholly impairing improvements such as structures and infrastructure. Just values reflect the broader economic benefits that erosion prevention may confer on parcel improvements in addition to the land itself and are used as the primary estimates.

Economic Results for Property Protection

Table 2. Property Protection Land Value Summary Table in 2026 Dollars

Bay	Acres	Closest Parcel	All Parcels within 0.1 mi	Flood-Risk Parcels within 0.1 mi
Choctawhatchee	8.07	\$281,337	\$319,949	\$300,804
Pensacola	17.47	\$1,571,666	\$2,135,442	\$1,705,429
St. Andrew	0.76	\$28,727	\$69,120	\$70,099
Total	26.3	\$1,881,730	\$2,524,511	\$2,076,332

Land values can be treated as a lower bound estimate, whereas just values can be treated as an upper bound and primary estimate. As a result of living shorelines constructed in Choctawhatchee Bay, the total estimated land value of economic damage avoided was \$281,337. In all bays, the closest parcel to each site was also always a parcel in a FEMA Special Flood Hazard Area. When expanding the range to 0.1 miles (528 feet) of each site to calculate land value, the total estimated value of economic damage was \$319,949. When calculating average land value using only parcels that were both in a FEMA Special Flood Hazard Area and within 0.1 miles of a site, the total estimated avoided economic damage was \$300,804.

In Pensacola Bay, the total estimated avoided economic damage was \$1,571,666 when using the average land value of the closest parcel to each site. When expanding the range of all parcels to 0.1 miles (528 ft) of each site to calculate the average land value, the total estimated avoided economic damage was \$2,135,442. When calculating the average land value using only parcels that were both in a FEMA Special Flood Hazard Area and within 0.1 miles of a site, the total estimated avoided economic damage was \$1,705,429.

Lastly, for St Andrew Bay, the total estimated avoided economic damage was \$28,727 when using the average land value of the closest parcel to each site. When expanding the range of all parcels to 0.1 miles (528 ft) of each site to calculate the average land value, the total estimated avoided economic damage was \$69,120. When calculating the average just value using only parcels that were both in a FEMA Special Flood Hazard Area and within 0.1 miles of a site, the total estimated avoided economic damage was \$70,099.

Table 3. Property Protection Just Value Summary Table in 2026 Dollars

Bay	Acres	Closest Parcel	All Parcels within 0.1 mi	Flood-Risk Parcels within 0.1 mi
Choctawhatchee	8.07	\$344,878	\$768,906	\$700,399
Pensacola	17.47	\$1,792,632	\$5,841,142	\$4,743,874
St. Andrew	0.76	\$49,549	\$495,427	\$300,430
Total	26.3	\$2,187,058	\$7,105,475	\$5,744,702

As a result of living shorelines constructed in Choctawhatchee Bay, the total estimated avoided economic damage was \$344,878 when using the average just value of the closest parcel to each site. When expanding the range of all parcels to 0.1 miles (528 ft) of each site to calculate the average just value, the total estimated avoided economic damage was \$768,906. When calculating the average just value using only parcels that were both in a FEMA Special Flood Hazard Area and within 0.1 miles of a site, the total estimated avoided economic damage was \$700,399.

Relating to Pensacola Bay, the total estimated avoided economic damage was \$1,792,632 when using the average just value of the closest parcel to each site. When expanding the range of all parcels to 0.1 miles (528 ft) of each site to calculate the average just value, the total estimated avoided economic damage was \$5,841,142. When calculating the average just value using only parcels that were both in a FEMA Special Flood Hazard Area and within 0.1 miles of a site, the total estimated avoided economic damage was \$4,743,874.

Lastly, for St Andrew Bay, the total estimated avoided economic damage was \$49,549 when using the average just value of the closest parcel to each site. When expanding the range of all parcels to 0.1 miles (528 ft) of each site to calculate the average just value, the total estimated avoided economic damage was \$495,427. When calculating the average just value using only parcels that were both in a FEMA Special Flood Hazard Area and within 0.1 miles of a site, the total estimated avoided economic damage was \$300,430.

Recreational Fisheries

Methodology

Recreational fisheries values were estimated using a two-stage biological and economic modeling approach. In the first stage, the biology team conducted quarterly field sampling from 2024 to 2026 across the CSTARS living shoreline sites using seine nets and minnow traps to monitor fish counts by species. From the species available in The Nature Conservancy's Fish Production Calculator (TNCFPC), eleven species with recreational significance in Florida's Gulf coastal waters and active harvest regulations under the Florida Fish and Wildlife Conservation Commission (FWC) were identified as candidates for economic valuation. Of these eleven, nine species were included in the final economic analysis: Sheepshead (*Archosargus probatocephalus*), Blue Crab (*Callinectes sapidus*), Spotted Seatrout (*Cynoscion nebulosus*), Brown Shrimp (*Farfantepenaeus aztecus*), Pink Shrimp (*Farfantepenaeus duorarum*), White Shrimp (*Litopenaeus setiferus*), Stone Crab (*Menippe* spp.), Striped Mullet (*Mugil cephalus*), and Red Drum (*Sciaenops ocellatus*).

For each of the nine species, young-of-year (YOY) recruitment estimates were generated using the TNCFPC based on the area of three habitat types present at each site: saltmarsh edge, oyster reef, and seagrass. Saltmarsh edge area was estimated by multiplying site length by three meters. Oyster reef area was estimated from satellite imagery and reef dimensions provided by the biology team's partners. Seagrass area was estimated from the percentage of submerged aquatic vegetation cover applied to the total site area minus reef and marsh area. All habitat areas were converted from square meters to hectares prior to entry into the TNCFPC. The Gulf of Mexico ecoregion was selected for all YOY estimations. It is important to note that YOY rates in the TNCFPC are habitat-specific, and not all nine species are

necessarily enhanced by all three habitat types. Where a species did not appear as a selectable option for a given habitat, that habitat was treated as not contributing to YOY recruitment for that species. YOY estimates from each applicable habitat type were then summed at the site level to produce a total annual YOY estimate per species per site, which was carried forward into the population models.

Because saltmarsh requires approximately three years to reach functional equivalency with natural marsh in terms of shoot densities, and restored seagrass beds require approximately two years to reach equivalency in faunal diversity, YOY contributions from each habitat type were not counted until those thresholds were met following the restoration year.

The TNCFPC-generated YOY estimates were then carried forward using population models to estimate the number of individuals surviving to legally harvestable size. For finfish species, the Von Bertalanffy Growth Function (VBGF) was used to estimate the age in years at which each species reaches the minimum legal harvest size established by FWC. The VBGF is expressed as:

Equation 1:

$$L_t = L_{\infty} \left(1 - e^{-K(t-t_0)}\right)$$

Where L_t is the harvest length in centimeters, L_{∞} is the asymptotic length in centimeters, K is the species-specific growth coefficient, t is the age of the fish in years at length L_t , and t_0 is the species-specific hypothetical age when fish length is zero. To solve for harvest age, Equation 1 was rearranged as follows:

$$t = t_0 - \frac{1}{K} \ln \left(1 - \frac{L_t}{L_{\infty}}\right)$$

For species with both minimum and maximum legal harvest sizes, specifically Spotted Seatrout ($L_t = 38.1$ to 48.26 cm) and Red Drum ($L_t = 45.72$ to 68.58 cm), the VBGF was solved at both bounds to establish a harvestable age range. Only YOY cohorts falling within that age window were counted as harvestable in a given year. Species-specific growth parameters

were sourced from FishBase, with values selected from the Gulf of Mexico region where available.

For Striped Mullet, FWC imposes no recreational size restriction, so the species maturity length of 27.94 cm was used as the harvestable threshold rather than a regulatory minimum. This length is also consistent with FWC's minimum commercial harvest length for the species. Because the instantaneous mortality values available in FishBase for Striped Mullet were collected from geographically distant regions including Taiwan, Tunisia, Iceland, and India, those values were considered inappropriate for the Gulf of Mexico context. The Gislason et al. (2010) model was therefore applied to estimate a regionally appropriate mortality rate for this species, as described further below.

For shrimp species, the continuous growth assumptions underlying the VBGF are less appropriate given their rapid growth rates and short lifespans, and the estimation approach differed from finfish in that target harvest size was derived from a target weight rather than a target length in most cases. For Brown Shrimp, the desired harvest weight of 4.1 grams was taken from the 2020 Southeast Data, Assessment, and Review (SEDAR) report as the average head-on commercial fishing weight. This weight was then converted to length using sex-specific length-weight models from Parrack (1979), yielding lengths of 86.2 mm for males and 87.5 mm for females. Ages corresponding to those lengths were then estimated in months using the sex-specific VBGF models from Parrack (1979) and converted to years, yielding 0.18 years for males and 0.15 years for females, for an average of 0.165 years across sexes. The estuarine instantaneous natural mortality rate for Brown Shrimp of $M = 0.8$ was sourced from Gracia (1997).

For Pink Shrimp, an initial attempt to derive harvest length from the SEDAR commercial weight of 25.6 grams using the Ramírez-Rodríguez et al. (2000) length-weight model produced an estimated length of 312 mm, which was recognized as implausibly large relative to commonly reported harvest sizes. The 203.2 mm harvest length was therefore substituted based on NOAA reporting of common minimum harvest size, and age at that length was estimated using VBGF parameters from Ramírez-Rodríguez and Arreguín-Sánchez (2003). For White Shrimp, the desired harvest weight of 20.0 grams from the 2020 SEDAR report was

converted to a length of 147.4 mm using the length-weight model from Hutchins et al. (1979), and age at that length was estimated using VBGF parameters from Arreguín-Sánchez (1999).

For Blue Crab, FWC imposes no recreational size restriction in Florida state waters. A commonly harvested carapace width of 127 mm corresponding to approximately one year of age was therefore used as the harvestable threshold, consistent with FWC life cycle documentation. Rather than applying the VBGF, the instantaneous natural mortality rate was estimated using the Lorenzen (1996) weight-based mortality model as applied by Hewitt et al. (2007):

$$M = 3.00 \cdot W^{-0.288}$$

Where W is the body weight in grams at the desired harvestable size, estimated using the Hewitt et al. (2007) allometric model relating carapace width to weight:

$$W = 0.000842 \cdot CW^{2.422}$$

Where CW is the carapace width in millimeters. This approach was used in place of FishBase or Gislason-derived mortality estimates given the availability of a species-specific and regionally validated model for this species.

For Stone Crab, two species co-occur in the study region, *Menippe adina* and *Menippe mercenaria*. Because Florida's Stone Crab fisheries are predominantly composed of *M. mercenaria*, all growth and mortality parameters were drawn from literature specific to that species. The VBGF was not applicable for Stone Crab given that crustacean growth occurs through discrete molts rather than continuous growth. Instead, time to harvestable claw size was estimated using the molt-based growth schedule described by Gerhart and Bert (2008), which tracks carapace width increments across successive molts for males and females separately. The minimum legal claw length of 73.025 mm was converted to carapace width using the following sex-specific claw length to carapace width allometric relationships from Crowley (2017), where CL is crusher claw length in millimeters, PL is pincer claw length in millimeters, and CW is carapace width in millimeters:

Male Crusher: $CL = 1.05 \cdot CW - 12.5$

Male Pincer: $PL = 0.86 \cdot CW - 8.8$

Female Crusher: $CL = 0.89 \cdot CW + 7.61$

Female Pincer: $PL = 0.72 \cdot CW + 3.5$

Based on the resulting carapace width targets and the molt schedule, male Stone Crabs were expected to reach harvestable claw size by year four and females by year five, yielding a conservative average of $t = 4.5$ years to harvestable size used in the survival calculations. The natural mortality rate of $M = 0.35$ was taken from Gerhart and Bert (2008). Rather than estimating Stone Crab weight through the carapace width to weight allometric models described above, the economic valuation for Stone Crab was based directly on measured claw poundage at each site, consistent with the fact that Stone Crab is legally harvested and sold by claw rather than by whole body weight. This approach more accurately reflects market conditions, as the claw is the unit that is harvested, retained, and sold.

The Ricker (1975) population model was applied across all nine species to estimate the number of individuals from each annual YOY cohort surviving to harvestable age:

Equation 2:

$$N_t = N_0 e^{-Mt}$$

Where N_t is the number of surviving individuals at time t , N_0 is the number of age-0 individuals (YOY), M is the species-specific instantaneous natural mortality rate, and t is the age in years. Because new YOY recruits are generated annually from the restoration year forward, the total number of harvestable individuals at a given site in a given year was calculated by summing surviving cohorts across all eligible years since restoration. Where direct instantaneous mortality estimates were unavailable in FishBase, or where available values were drawn from regions too geographically distant to be applicable, the Gislason et al. (2010) model was used to estimate M for finfish species:

Equation 3:

$$\ln M = 0.55 - 1.61 \ln L_t + 1.44 \ln L_\infty + \ln K$$

In the second stage, the estimated counts of harvestable individuals were converted to weight in pounds using species-specific length-weight allometric relationships. For finfish, the general form of the relationship is $W = aL^b$, where W is weight in grams, L is length in centimeters, and a and b are species-specific parameters sourced from FishBase from the Gulf of Mexico region where available. For Blue Crab and Stone Crab, sex-specific carapace width to weight allometric models were applied from Hewitt et al. (2007) and Crowley (2017) respectively, with male and female weights averaged to produce a single per-individual weight estimate. For shrimp species, length-weight models from the primary literature specific to each species were used, with sex-specific models averaged where applicable. All weight estimates were converted from grams to pounds prior to applying ex-vessel prices.

Ex-vessel prices, representing the price paid to commercial fishers at the first point of sale, were used as the economic proxy for the market value of each recreationally harvested species. This approach follows Jordan et al. (2012) and reflects the direct economic value of the aquatic species without incorporating downstream costs such as transportation, processing, or marketing. The CEFA research team downloaded two commercial fisheries landings summary datasets for 2025 from FWC, both using the Food and Bait species set. The first dataset provided area-level pricing delineated by location descriptions corresponding to the cities nearest each bay, specifically Destin for Choctawhatchee Bay, Pensacola for Pensacola Bay, and Panama City for St. Andrew Bay, allowing for locally specific price assignment. The second dataset provided coast-level pricing and served as a fallback when area-level data were unavailable for a given species or bay.

Because the naming conventions used in the CSTARS biological dataset did not match those used in the FWC commercial landings data, a species crosswalk was constructed to align the two datasets. Most assignments were straightforward, but several required explicit assumptions. Blue Crab pricing used hard shell values rather than soft shell, as hard shell blue crab was the more commonly reported variation with data available across all three

bays. Stone Crab pricing used the medium claw category as a conservative midpoint, given that the CSTARs dataset did not provide claw size information that would have allowed assignment of small, large, or jumbo claw pricing. Striped Mullet was matched to the FWC category of black mullet, which is an alternative common name for the same species. The following table summarizes the final species crosswalk and the level of pricing precision achieved for each species.

Table 4. Species Crosswalk and Level of Precision per Species

CSTARs Species	FWC/NOAA Species	Pricing Level
Blue Crab	Crab, Blue (Hard)	Bay-Level
Brown Shrimp	Shrimp, Brown	Bay-Level
Pink Shrimp	Shrimp, Pink	Coast-Level
Red Drum	Drum, Red	Mississippi (Inflation-Adjusted)
Sheepshead	Sheepshead	Bay-Level
Spotted Seatrout	Seatrout, Spotted	Bay/Coast-Level
Stone Crab	Crab, Stone, Medium Claws	Coast-Level
Striped Mullet	Mullet, Black	Bay-Level
White Shrimp	Shrimp, White	Bay/Coast-Level

Red Drum required a separate data source because Florida and most other Gulf states prohibit commercial harvest of this species, making FWC landings data unavailable. Mississippi is the only Gulf state that permits commercial Red Drum harvest and was therefore used as the nearest available economic proxy for Florida ex-vessel pricing. Red Drum prices were sourced from the 2024 NOAA commercial fisheries landings database for Mississippi and adjusted to 2026 dollars using the Consumer Price Index. All other species prices were drawn directly from the 2025 FWC datasets and were appropriately adjusted to 2026 dollars as well.

Total economic value for each species at each site and bay was calculated by multiplying the estimated weight in pounds by the corresponding average ex-vessel price per pound. Site-level estimates were aggregated to produce bay-level totals, and bay-level totals were summed to produce the aggregate recreational fisheries value across all three bays.

Economic Results for Recreational Fisheries

The total annual recreational fisheries value across the three bays is composed of estimated economic values by species at each site. Choctawhatchee Bay generated a total annual recreational fisheries value of \$36,868. Pensacola Bay produced the highest value among the three bays at \$266,616. St. Andrew Bay generated a total annual recreational fisheries value of \$14,528. In aggregate, the recreational fisheries value across the three bays totaled \$318,012.¹

Carbon Sequestration

Methodology

The economic value of carbon sequestration was estimated using two independent pricing approaches, which together provide a lower and upper bound on the economic value of carbon sequestered by the living shorelines. The first approach uses carbon prices derived from cap-and-trade auction results from the Regional Greenhouse Gas Initiative (RGGI), and the second uses the Social Cost of Carbon (SCC) as estimated by the U.S. Environmental Protection Agency. Because Florida does not operate a cap-and-trade system for carbon emissions, the RGGI represents the closest available U.S.-based market proxy for carbon pricing. The RGGI price reflects regulatory scarcity in carbon markets and is therefore treated as a conservative lower-bound estimate. The SCC, which captures a broader set of social damages associated with carbon emissions including impacts on agricultural productivity, human health, flood risk, and ecosystem services, is treated as the upper-bound estimate.

¹ It is noteworthy that the sharp disparity between Pensacola and the other bays is attributable to particularly large price per pound for stone crab, and a much higher volume of stone crab in the Pensacola Bay.

I. RGGI Approach

The RGGI is a sector-specific cap-and-trade program covering carbon dioxide emissions from electric power plants in ten northeastern states. Quarterly auction results, including the clearing price per U.S. short ton of CO₂, were sourced directly from the RGGI's official auction results database. Because the carbon sequestration data pertains to the 2024 measurement year, the average clearing price across all four quarterly auctions conducted in 2024 was calculated and used as the representative annual price. This average was then adjusted from 2024 dollars to 2026 dollars using the Consumer Price Index.

Because RGGI auction prices are denominated in U.S. short tons of CO₂ and the collected sequestration data reports sequestered carbon in pounds of carbon rather than pounds of CO₂, two unit conversions were applied. First, the price per ton of CO₂ was converted to a price per ton of carbon using a factor of $\frac{44}{12}$, which reflects the ratio of the molecular weight of CO₂ to the atomic weight of carbon. Second, the price per ton was converted to a price per pound using a conversion factor of 2,000 pounds per U.S. short ton. After these adjustments, the resulting price per pound of carbon in 2025 was \$0.04, which was multiplied by the CSTARS site-level carbon sequestration values in pounds to produce economic estimates by site and by bay.

II. Social Cost of Carbon Approach

The EPA's 2023 Social Cost of Greenhouse Gases report provides estimates of the SCC in 2020 dollars per metric ton of CO₂ by emission year and discount rate. The emission year 2024 was selected to correspond with the CSTARS measurement period. The SCC value associated with the 2.5% discount rate was used as it represents the most conservative of the available discount rate options, which also include 2% and 1.5% rates. The selected SCC value was adjusted from 2020 dollars to 2026 dollars using the Consumer Price Index.

As with the RGGI approach, two unit conversions were required to align the SCC pricing with the CSTARS data. The price per metric ton of CO₂ was first converted to a price per metric ton of carbon using the same molecular weight ratio of $\frac{44}{12}$. It was then converted to a price per pound of carbon using a conversion factor of 2,204 pounds per metric ton. After these

adjustments, the resulting price-per pound of carbon in 2026 dollars was \$0.27, which was multiplied by the CSTARS site-level carbon sequestration values in pounds to produce economic estimates by site and by bay.

Economic Results for Carbon Sequestration

The RGGI-based estimate of \$0.04 per pound of carbon and the SCC-based estimate of \$0.27 per pound of carbon together define a range of plausible economic values for carbon sequestration at each site. The estimated annual value (in 2026 dollars) of carbon sequestration in 2024 in St. Andrew Bay is between the RGGI estimate of \$4,606 and the SCC estimate of \$31,088. The estimated annual value of carbon sequestration in 2024 in Choctawhatchee Bay was between \$17,294 and \$116,732 and in Pensacola Bay was between \$22,768 and \$153,682. The total annual estimated value across all bays is between \$44,667 and \$301,502.

Water Clarity

Methodology

The economic value of water clarity improvements attributable to the living shorelines was estimated using a counterfactual benefit transfer approach, drawing on hedonic property value estimates from Walsh, Milon, and Scrogin (2011), supplemented by the Guignet et al. (2022) meta-analysis of hedonic water quality studies. Walsh et al. (2011) estimated implicit prices for water clarity improvements using a large-scale hedonic analysis of residential property sales around 146 monitored lakes in Orange County, Florida, finding that a one foot increase in Secchi depth was associated with an increase of approximately \$5,594.80 in the price of the mean lakefront property and \$717.50 for the mean non-lakefront property, expressed in 2000 dollars. These values were adjusted to 2026 dollars using the Consumer Price Index, yielding adjusted implicit prices of \$10,724.97 per foot for lakefront properties and \$1,375.38 per foot for non-lakefront properties. Because lakefront and non-lakefront parcels were not separately identified in the spatial analysis, the non-lakefront implicit price of \$1,375.38 per foot was applied to all parcels within 1,000 meters of each living shoreline site, producing a conservative lower bound estimate. Guignet et al.

(2022) provide additional support for the transferability of these implicit price estimates through a nationwide meta-analysis of 36 hedonic water quality studies including estuary-specific estimates, which are more closely analogous to the panhandle bay context than the freshwater lake setting of Walsh et al. (2011).

The valuation proceeded through a sequential series of conversions linking the biology team's site-level sediment erosion prevention estimates to an estimated improvement in water clarity. Sediment erosion prevention data were provided at the site level in units of cubic meters of sediment volume prevented. These site-level values were aggregated to the bay level for the economic analysis, as bay-level aggregation is more physically realistic than site-level analysis given that sediment disperses across the broader water body through tidal exchange and mixing over time.

A local turbidity-to-Secchi depth regression was developed using paired water quality observations from the Florida Department of Environmental Protection Watershed Information Network (FDEP WIN). Secchi depth and turbidity readings were extracted from 57 monitoring stations distributed across the East Bay, North Bay, St. Andrew Bay proper, and West Bay sub-areas, yielding 793 paired station-date observations after removing missing values, censored observations where Secchi depth equaled or approximated total water depth, and a small number of observations identified as recording convention artifacts at a single station. A linear regression of Secchi depth in meters on turbidity in NTU was estimated as follows:

$$\text{Secchi (m)} = 2.9476 + (-0.2683) \times \text{Turbidity (NTU)}$$

The regression was statistically significant with a relatively low R-squared of 0.1356. The relatively low R-squared likely reflects findings from Florida-specific literature suggesting that chlorophyll-a, rather than inorganic suspended sediment, is the dominant driver of Secchi depth variability in Florida's nearshore coastal waters. The regression is used here as a conservative approximation of the turbidity-clarity relationship in the panhandle bay environment, with the low explanatory power acknowledged as a meaningful limitation of the analysis.

The economic valuation employed a counterfactual approach. A baseline turbidity of 2.2793 NTU was established as the mean observed turbidity across all 793 paired station observations from St. Andrew Bay. For each bay, the total prevented sediment volume was converted to a sediment mass using an assumed bulk density of 0.5 g/cm^3 , representing the soft organic-rich surface sediments typical of living shoreline marsh environments. This value falls within the low-density class of 0.032 to 0.752 g/cm^3 identified by Salehi Hikouei et al. (2021) in a machine learning classification of salt marsh soil bulk density along the Georgia coast, with a sensitivity range of 0.4 to 1.1 g/cm^3 applied to capture the full range of plausible values across low and high bulk density marsh soils. The sediment mass was then divided by the full bay water mixing volume, calculated as the bay surface area multiplied by an assumed mixing depth of 1.5 meters, consistent with the shallow site depth studied by Schoellhamer (1995) in Old Tampa Bay and corroborated by Schoellhamer (1996) in Hillsborough Bay, both shallow microtidal subtropical Florida estuaries analogous to the panhandle bays. A sensitivity range of 1.0 to 2.0 meters was applied to reflect the range of shallow to moderate depth conditions across the three study bays. Bay surface areas were sourced from Keppner and Keppner (2001) for St. Andrew Bay (approximately 280 km^2), the Northwest Florida Water Management District (2017) for Choctawhatchee Bay (approximately 334 km^2), and the Pensacola and Perdido Bays Estuary Program for Pensacola Bay (approximately 144 square miles).

The estimated TSS concentration was converted to a turbidity increment by inverting the existing TSS-to-turbidity regression developed for the nitrogen analysis. Adding this turbidity increment to the baseline produced a counterfactual turbidity value representing conditions without the living shoreline. The Secchi depth regression was then applied to both the baseline turbidity and the counterfactual turbidity, and the difference between the resulting Secchi depth estimates was taken as the water clarity improvement attributable to the living shorelines.

The economic value of the water clarity improvement was calculated by multiplying the estimated Secchi depth improvement in feet by the CPI-adjusted non-lakefront implicit price from Walsh et al. (2011) and by the total number of deduplicated residential parcels within 1,000 meters of any living shoreline site within each bay. Parcel counts were derived from

a spatial join of the 2025 NAL parcel dataset to the CSTARS site locations at a 1,000 meter search radius, with a one-to-one join operation applied to ensure each parcel was counted only once regardless of proximity to multiple sites. The 1,000 meter distance threshold is consistent with the spatial boundary used in Walsh et al. (2011), who found that estimated implicit prices of water quality declined continuously over this distance and retained all residential property sales within 1,000 meters of a lake for estimation.

Several assumptions underlie this analysis and should be noted. Bulk density of 0.5 g/cm^3 is assumed based on southeastern United States salt marsh soil literature and should be updated with site-specific measurements where they become available. Mixing depth of 1.5 meters is assumed based on Schoellhamer (1995) and Schoellhamer (1996). The baseline turbidity derived from St. Andrew Bay stations is applied uniformly to all three bays, which is a limitation given that no comparable station data were available for Choctawhatchee and Pensacola Bays at the time of this analysis. The geographic transfer of the turbidity-to-Secchi regression from St. Andrew Bay to the other two bays introduces additional uncertainty. Finally, the non-lakefront implicit price from Walsh et al. (2011) was developed from a freshwater lake context in Orange County, Florida, and its transfer to coastal estuarine environments in the panhandle introduces uncertainty that the Guignet et al. (2022) meta-analysis partially addresses but does not fully resolve.

Economic Results for Water Clarity

The living shorelines at St. Andrew Bay, Choctawhatchee Bay, and Pensacola Bay generated estimated annual water clarity benefits of \$1,236, \$67,073, and \$50,722, respectively, for a combined central estimate total of \$119,031.² These estimates reflect Secchi depth improvements of less than 0.01 feet at all three bays, representing clarity improvements of less than 0.1% relative to baseline conditions, which is consistent with the modest scale of sediment erosion prevention at these living shoreline sites relative to the total water volume of the panhandle bays. It is noted that some individual sites within each bay showed net negative sediment values, indicating that those sites experienced net erosion rather than

² The comparatively low water clarity estimate for St. Andrew Bay reflects a substantially smaller volume of sediment erosion prevented relative to Choctawhatchee and Pensacola Bays.

accretion during the study period. These negative contributions partially offset the positive values from other sites within each bay and are appropriately retained in the bay-level totals as they represent genuine conditions at those locations.

Given the number of assumptions required to produce these estimates, a sensitivity analysis was conducted across low, central, and high values for the two primary assumptions. The bulk density assumption drives the lower and upper bounds through its effect on the estimated sediment mass entering the water column: a lower bulk density of 0.4 g/cm^3 reduces the estimated sediment mass and therefore the turbidity increment, while a higher bulk density of 1.1 g/cm^3 increases it. The mixing depth assumption affects results in the opposite direction: a shallower mixing depth of 1.0 meter concentrates the sediment into a smaller water volume, producing a larger TSS concentration change and therefore a larger estimated Secchi depth improvement, while a deeper mixing depth of 2.0 meters dilutes the sediment across a larger volume, reducing the estimated improvement. The combination of low bulk density and high mixing depth produces the most conservative estimate of approximately \$71,418 across all three bays, while the combination of high bulk density and low mixing depth produces the highest estimate of approximately \$392,801. The central estimate of \$119,031 represents the mid-range of these assumptions and is the recommended figure for reporting, with the sensitivity range noted as a reflection of the substantial uncertainty inherent in this analysis.

Economic Results Summary Table

Table 5. Final Summary Table of All Economic Value Across Ecosystem Service Type and Bay

Ecosystem Service	St. Andrew Bay	Choctawhatchee Bay	Pensacola Bay	Total Benefit Transfer (\$/yr)	Lower Bound Benefit Transfer	Upper Bound Benefit Transfer
Nitrogen*	\$13,102	\$112,748	\$602,461	\$728,311		
Property Protection*	\$272,488	\$556,892	\$3,816,887	\$4,646,267	\$2,187,058	\$7,105,475
Recreational Fisheries	\$14,528	\$36,868	\$266,616	\$318,012		
Carbon Sequestration	\$17,847	\$67,013	\$88,225	\$173,084	\$44,667	\$301,502
Water Clarity	\$1,236	\$67,073	\$50,722	\$119,031	\$71,418	\$392,801
Total	\$319,201	\$840,594	\$4,824,911	\$5,984,705	\$3,349,466	\$8,846,101

*All values are expressed in 2026 dollars. The reported Nitrogen value is the total of storage and removal estimates. The lower bound property protection total values were calculated using the closest parcel just value estimates, and the upper bound total values were calculated using 0.1 mi just value estimates. Main results for property protection value are an average of the lower and upper bound just values.

References

- Arreguín-Sánchez, F. (1999). Age and growth estimation for the white shrimp, *Penaeus setiferus*, from the offshore fishery of the northwestern Gulf of Mexico. *Proceedings of the 45th Gulf and Caribbean Fisheries Institute*, 146–159. <https://aquadocs.org/items/813c4bb4-ebec-4a35-b035-cc9f7269aa7b>
- Crowley, C. E. (2017). *Florida fishery-wide reproductive indices of stone crab, Menippe mercenaria, and their application to stock assessment and management* [Doctoral dissertation, University of South Florida]. ProQuest. <https://search.proquest.com/openview/0d09c61c795e3746f83d1b1f36d3acc7/1>
- Crowley, C., Chagaris, D., Gandy, R., & Daly, K. (2019). Evaluation of Florida stone crab life history and management scenarios using spawning potential ratios. *Fisheries Research*, 209, 196–207. <https://doi.org/10.1016/j.fishres.2018.09.017>
- Currin, C. A., Delano, P. C., & Valdes-Weaver, L. M. (2008). Utilization of a citizen monitoring protocol to assess the structure and function of natural and stabilized fringing salt marshes in North Carolina. *Wetlands Ecology and Management*, 16(2), 97–118. <https://doi.org/10.1007/s11273-007-9059-1>
- Currin, C. A., Davis, J., & Malhotra, A. (2017). Response of salt marshes to wave energy provides guidance for successful living shoreline implementation. In *Living Shorelines* (pp. 211–234). Taylor & Francis. <https://www.taylorfrancis.com/chapters/edit/10.1201/9781315151465-14/>
- Currin, C. A. (2019). Living shorelines for coastal resilience. In *Coastal Wetlands* (pp. 1023–1053). Elsevier. <https://doi.org/10.1016/B978-0-444-63893-9.00030-7>
- Fonseca, M. S., Meyer, D. L., & Hall, M. O. (1996). Development of planted seagrass beds in Tampa Bay, Florida, USA. II. Faunal components. *Marine Ecology Progress Series*, 132, 141–156. <https://doi.org/10.3354/meps132141>
- Gerhart, S. D., & Bert, T. M. (2008). Life-history aspects of stone crabs (genus *Menippe*): size at maturity, growth, and age. *Journal of Crustacean Biology*, 28(2), 252–261. <https://doi.org/10.1163/20021975-99990372>
- Guignet, D., Heberling, M. T., Papenfus, M., & Griot, O. (2022). Property values, water quality, and benefit transfer: A nationwide meta-analysis. *Land Economics*, 98(2), 191–218. <https://doi.org/10.3368/le.98.2.050120-0062r1>
- Gislason, H., Daan, N., Rice, J. C., & Pope, J. G. (2010). Size, growth, temperature and the natural mortality of marine fish. *Fish and Fisheries*, 11(2), 149–158. <https://doi.org/10.1111/j.1467-2979.2009.00350.x>

- Gracia, A. (1997). Simulated and actual effects of the brown shrimp, *Penaeus aztecus*, closure in Mexico. *Marine Fisheries Review*, 59(2), 18–24. <https://spo.nmfs.noaa.gov/sites/default/files/pdf-content/MFR/mfr592/mfr5922.pdf>
- Hewitt, D. A., Lambert, D. M., Hoenig, J. M., Lipcius, R. N., Bunnell, D. B., & Miller, T. J. (2007). Direct and indirect estimates of natural mortality for Chesapeake Bay blue crab. *Transactions of the American Fisheries Society*, 136(4), 1030–1040. <https://doi.org/10.1577/T06-078.1>
- Hutchins, D. L., Chamberlain, G. W., & Parker, J. C. (1979). Length-weight relations for several species of penaeid shrimp cultured in ponds near Corpus Christi, Texas. *Proceedings of the World Mariculture Society*, 10(1–4), 565–570. <https://doi.org/10.1111/j.1749-7345.1979.tb00054.x>
- Jordan, S. J., O'Higgins, T., & Dittmar, J. A. (2012). Ecosystem services of coastal habitats and fisheries: multiscale ecological and economic models in support of ecosystem-based management. *Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science*, 4(1), 573–586. <https://doi.org/10.1080/19425120.2012.703162>
- Keppner, E., & Keppner, L. (2001). Bay Environmental Study Team (BEST) Reports BEST Report 004: The St. Andrew Bay Ecosystem: A Look to the Future. Retrieved from <http://www.ecoastlife.com/keppner/BESTReports.html>
- Kouanda, A., & Hua, G. (2021). Determination of nitrate removal kinetics model parameters in woodchip bioreactors. *Water Research*, 195, 116974. <https://doi.org/10.1016/j.watres.2021.116974>
- Lorenzen, K. (1996). The relationship between body weight and natural mortality in juvenile and adult fish: a comparison of natural ecosystems and aquaculture. *Journal of Fish Biology*, 49(4), 627–642. <https://doi.org/10.1111/j.1095-8649.1996.tb00060.x>
- Marton, J. M., & Roberts, B. J. (2014). Spatial variability of phosphorus sorption dynamics in Louisiana salt marshes. *Journal of Geophysical Research: Biogeosciences*, 119(3), 451–465. <https://doi.org/10.1002/2013JG002486>
- Merrill, N., Mazzotta, M., Mulvaney, K., Sawyer, J., Twichell, J., Atkinson, S., Keith, D., & Erban, L. (2024). The value of water quality for coastal recreation in New England, USA. *Agricultural and Resource Economics Review*, 53(3), 492–514. <https://doi.org/10.1017/age.2024.12>
- Moksnes, P.-O., Röhr, M. E., Holmer, M., Eklöf, J. S., Eriander, L., Infantes, E., & Boström, C. (2021). Major impacts and societal costs of seagrass loss on sediment carbon and nitrogen stocks. *Ecosphere*, 12(7), e03658. <https://doi.org/10.1002/ecs2.3658>

- Northwest Florida Water Management District. (2017). Choctawhatchee River and Bay Surface Water Improvement and Management Plan. Northwest Florida Water Management District.
- Parrack, M. L. (1979). Aspects of brown shrimp, *Penaeus aztecus*, growth in the northern Gulf of Mexico. *Fishery Bulletin*, 76(4), 827–836. <https://spo.nmfs.noaa.gov/sites/default/files/pdf-content/fish-bull/parrack.pdf>
- Pensacola and Perdido Bays Estuary Program. (n.d.). Watersheds 101. Retrieved from <https://www.ppbeo.org/the-watersheds/watersheds-101>
- Pollack, J. B., Yoskowitz, D., Kim, H. C., & Montagna, P. A. (2013). Role and value of nitrogen regulation provided by oysters (*Crassostrea virginica*) in the Mission-Aransas Estuary, Texas, USA. *PLoS ONE*, 8(6), e65314. <https://doi.org/10.1371/journal.pone.0065314>
- Ramírez-Rodríguez, M., & Arreguín-Sánchez, F. (2000). Growth of pink shrimp *Farfantepenaeus duorarum* (Burkenroad, 1939) in Campeche sound, Mexico. *Crustaceana*, 1263–1272. <https://www.jstor.org/stable/20106398>
- Ramírez-Rodríguez, M., & Arreguín-Sánchez, F. (2003). Life history stage duration and natural mortality for the pink shrimp *Farfantepenaeus duorarum* (Burkenroad, 1939) in the southern Gulf of Mexico, using the gnomonic model for time division. *Fisheries Research*, 60(1), 45–51. [https://doi.org/10.1016/S0165-7836\(02\)00064-4](https://doi.org/10.1016/S0165-7836(02)00064-4)
- Ricker, W. E. (1975). Computation and interpretation of biological statistics of fish populations. *Fisheries Research Board of Canada Bulletin*, 191, 1–382. <https://waves-vagues.dfo-mpo.gc.ca/Library/1485.pdf>
- Salehi Hikouei, I., Kim, S. S., & Mishra, D. R. (2021). Machine-learning classification of soil bulk density in salt marsh environments. *Sensors*, 21(13), 4408. <https://doi.org/10.3390/s21134408>
- Schoellhamer, D. H. (1995). Sediment resuspension mechanisms in Old Tampa Bay, Florida. *Estuarine, Coastal and Shelf Science*, 40(6), 603–620. <https://doi.org/10.1006/ecss.1995.0041>
- Schoellhamer, D. H. (1996). Anthropogenic sediment resuspension mechanisms in a shallow microtidal estuary. *Estuarine, Coastal and Shelf Science*, 43(5), 533–548.
- Schutte, C. A., Huanca-Valenzuela, P., Lavik, G., Marchant, H. K., & de Beer, D. (2021). Advection drives nitrate past the microphytobenthos in intertidal sands, fueling deeper denitrification. *Frontiers in Microbiology*, 12, 556268. <https://doi.org/10.3389/fmicb.2021.556268>
- Southeast Data, Assessment, and Review (SEDAR). (2020). *SEDAR 87 research document 09: Final report — U.S. Gulf of Mexico commercial shrimp conversion factors validation*.

<https://sedarweb.org/documents/sedar-87-rd09-final-report-u-s-gulf-of-mexico-commercial-shrimp-conversion-factors-validation-2020/>

Thompson, S. P., Paerl, H. W., & Go, M. C. (1995). Seasonal patterns of nitrification and denitrification in a natural and a restored salt marsh. *Estuaries*, 18(2), 399–408. <https://doi.org/10.2307/1352322>

U.S. Environmental Protection Agency. (2023). *Report on the social cost of greenhouse gases: Estimates incorporating recent scientific advances*. https://www.epa.gov/system/files/documents/2023-12/epa_scghg_2023_report_final.pdf

Walsh, P. J., Milon, J. W., & Scrogin, D. O. (2011). The spatial extent of water quality benefits in urban housing markets. *Land Economics*, 87(4), 628–644. <https://doi.org/10.3368/le.87.4.628>