Oyster populations in large Florida estuaries that have collapsed are resistant to restoration using traditional methods – insights from ongoing efforts in multiple systems

*#Authorship to be determined (DEP, FWC, UF, NFWF, other ?)*

*#This document is written in manuscript form with the intent to publish. As an initial effort to assimilate and share information I have included some information that would need to be modified for publication such as references to reports that may or may not be easily accessible, a mix of metric and standard system units for consistency with agency convention, and longer textual discussion within figure legends.*

*Abstract*

*Introduction* - Eastern oyster populations in the northern Gulf of Mexico are depressed from historic levels for poorly understood reasons. Since 2010, the states of Florida, Alabama, Mississippi, Louisiana, and Texas have all declared state or federal level oyster fishery disasters citing reasons including prolonged drought, extreme rain events, or freshwater releases from water management structures (refs). Florida has engaged in litigation in the US Supreme Court over oyster population collapse in Apalachicola Bay (ref). Several of these states have implemented fishery closures in response to the depressed status of oyster stocks (i.e., Mobile Bay in Alabama, Apalachicola Bay in Florida) but only one of these stocks (Mobile Bay) has reopened to harvest with Apalachicola scheduled to re-open in 2025. Oyster populations in the Gulf of Mexico were damaged by the sinking of the *Deepwater Horizon* and subsequent oil spill (Deepwater Horizon Natural Resources Damage Assessment Trustees, 2016). This created substantial funding opportunities (more than $199M US) for oyster restoration in the Gulf of Mexico. The dollars allocated for restoration exceeded the annual value of oyster landings (Pine et al. 2022).

Many proposed, ongoing, and historical oyster restoration efforts focus on adding various materials for oyster spat (larvae) to settle and grow (Howie and Bishop 2021). Adding this material to the substrate is an effort to promote a positive oyster shell budget (harvest removes shell stock, Pine et al. 2015). These material additions provide material from outside of the system of management interest to replace natural oyster cultch, a complex matrix of living and dead material where oyster larvae settle and grow. These restoration efforts attempt to shift oyster reefs from an observed low but resilient state to a more desired productive state (Pine et al. 2022) through restoration.

We used opportunistic sampling from ongoing and recently completed efforts to shift oyster populations from undesired to desired states through restoration and fishery closure projects in estuaries in the northern Gulf of Mexico. Many of the large restoration programs currently funding these efforts are long-term (10-year) projects, but information as learning is needed over shorter time scales to inform other proposed restoration and management projects in similar systems. This is an issue of both temporal and spatial scale (Pine et al. 2022). To facilitate learning under an adaptive management framework as programmatically adopted by these funders, these restoration efforts should be assessed in-progress, and if necessary, corrective changes made to improve the likelihood of the restoration objective of shifting the oyster population from an undesired low-level, to a more desired level. This desired state can vary by location, and type of oyster bar (intertidal vs. subtidal), and management goals. But in general, the desired expectation motivating these restoration efforts are to provide and promote both ecosystem services and create opportunities for oyster harvest through fishery recovery.

Site description – We assessed trends in oyster populations in three estuaries in the Florida panhandle that currently have ongoing or recently completed oyster restoration projects. Pensacola Bay (Figure 1) in northwest Florida (Santa Rosa and Escambia counties) is the fourth largest estuary in Florida with a surface area of approximately 126,000 total acres. Reported oyster landings, trips, and CPUE for Pensacola Bay in recent decades have declined (Figure 2) since the current mandatory TRIP ticket program was implemented in 1985. The East Bay (Figure 1) arm of St. Andrew Bay, near Panama City, Florida (Okaloosa and Walton Counties) is one region of St. Andrew Bay which has a total surface area of approximately XYZ acres. Reported oyster landings and trips for East Bay are not available, but for the counties comprising St. Andrew Bay oyster trips and landings in recent decades have declined and harvest in recent years is near zero (Figure 2). Apalachicola Bay is a large estuary in Franklin County which historically supported the largest oyster fishery in Florida before collapsing in fall of 2012 (Pine et al. 2015) and was closed to commercial harvest in December 2020 through December 2025 by the Florida Fish and Wildlife Conservation Commission.

Management actions – Cultch material was deposited in each bay in phases by individual state management agencies (Florida Department of Environmental Protection, DEP; Florida Fish and Wildlife Conservation Commission, FWC; Florida Department of Agriculture and Consumer Services, FDACS) as part of three different projects funded to the State of Florida with funds made available following the *Deepwater Horizon* oil spill. In Pensacola Bay approximately 20,103 cubic yards of limerock aggregate were distributed at 17 different sites at an approximate density of 228 cubic yards per acre (FDACS 2016a) during September and October 2016. In St. Andrews Bay approximately 17,000 cubic yards of crushed granite was distributed on nine different oyster reefs at a density of about 200 cubic yards per acre (FDACS 2016b) in June 2016. In Apalachicola Bay four different restoration projects with similar objectives and methodologies occurred during this time. In the first (NRDA), approximately 24,840 cubic cards of fossil shell material was deployed on 16 different sites at an average cultch density of 200 cubic yards per acre. In the second project (FDEP), approximately 95,500 cubic yards of limerock aggregate was deployed as part of an FDEP project on fourteen different oyster reef sites. Average density of cultch material was 300 cubic yards per acre. The third project (FWC) deployed 9600 cubic yards of shell material in sites 2-acres in size at densities of 100, 200, 300, or 400 cubic yards per acre. The fourth project deployed XYZ (FWC NFWF 2) deployed XYZ cubic yards of limestone at a density of ABC at Z different stations. Across all studies the actual area and density of cultch material deployed varied due to construction challenges and storm events that occurred during the study.

Table 1. Summary of deployment date, location, and project description.

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| --- | --- | --- | --- | --- | --- | --- | --- |
| Year | Bay | Project name | Agency | Material | Amount (yds3 by convention) | Sites | Average material density (yds3 per acre by convention |
| Fall 2016 | Pensacola | NRDA 4044 | DEP | Limestone | 20,103 | 17 | 228 |
| June 2016 | St. Andrews | NRDA 4044 | DEP | Crushed granite | 17,000 | 9 | 200 |
| November 2016 ?? | Apalachicola | NRDA 4044 | DEP | Quarried shell | 24,840 | 16 | 200 |
| November 2017 | Apalachicola | GEBF 5007 | DEP | Limerock aggregate | 95,500 | 14 | 300 |
| August 2015 | Apalachicola | NFWF-1 | FWC | Quarried shell | 9,600 | 3 | 100,200,300,400 |
| July 2021 | Apalachicola | NFWF-2021 | FWC | Limerock aggregate |  |  | 300? |

*Methods*

Fisheries dependent data – We summarized commercial fisheries landings data for each of the three bays from Florida Fish and Wildlife Conservation Commission public database. For each bay we summed the landings and trips by county surrounding the bay, and the calculated catch-per-unit effort (CPUE) as annual landings/annual trips.

Reef construction – Reef construction methods across studies were similar and were designed to minimize costs and maximize amount of material deployed. Sites were selected for cultch placement based on local knowledge of historic or extant reef locations. Cultch material was deployed on site from barges by washing material from barge deck using high pressure hoses at a prescribed density. Reef materials were either quarried shell or a “Kentucky” limestone of graded size (often #4, 1.5-3 inches in size) transported on barges via inland and coastal waterway and then “planted” at specific locations.

Field collections – Similar methods were followed for all projects to estimate live oyster counts and mass of cultch material based on methods used in Florida since the 1980’s (FWC 2021 <https://myfwc.com/media/27745/oimmp-v2-ch11.pdf>) where divers haphazardly place ¼-m2 (0.5-m on each side) quadrats at selected sites and remove all oysters and cultch material to a “wrist deep” depth and place material in bags. Once bags are returned to the vessel, they are either processed on site or returned to the lab where counts of live and dead oysters, measurements of shell height, weight of cultch material, and other metrics depending on study were recorded.

Data Analyses – We followed methods for analyzing oyster count data described in Moore et al. (2020) modified based on how data were collected in the field. Briefly, we summed counts of live oysters at each restoration site and period into three size classes, spat (<26-mm shell height), sublegal (locally termed “seed” oyster; 26-75-mm shell height), and legal to harvest (>76-mm shell height). For some studies, counts were totaled in this way in the field and for other studies total counts (all sizes) were converted to counts per size class by calculating the proportion of oysters within each size class from concurrent oyster shell height samples to the sample of total oysters. We then assessed the distribution the count data follow by examining the ratio between the count mean and variance at each site (variance always exceeded mean). We used generalized linear models (GLMs; Bolker et al. 2009) with a negative binomial distribution to assess how oyster counts (dependent variable) vary over period (a time variable of equal length used to combine sampling months into winter [November-April] or summer [May-October]) and we used site as a random effect (to account for correlation among samples at each site). We assumed that the total oyster counts per site would be related to the number of quadrats collected at each site, so we included the number of quadrats as an offset of effort (log link function; Zuur et al. 2009, 2013). By using effort as an offset in this way we change the model from modeling counts, to modeling a rate measured as the count/quadrat as the response variable. Because the quadrats were the same size across study, the area sampled only changed as a function of the number of quadrats. Using counts and accounting for effort, as opposed to converting the counts to CPUE based on density sampled has two main advantages in our experience (1) maintains the response as an integer allowing the use of a negative binomial distribution (which we have observed oyster count data follow; Moore et al. 2020) and (2) fitted values and confidence intervals do not contain negative values (Zuur et al. 2009). We fit models to the data that included time (period), location (as a random effect), and then used the best fitting model (informed by AIC value and visual assessments of model fit to data) to predict oyster counts by period and location using the glmm.TMB (Brooks et al. 2017) and ggeffects packages (Lüdecke 2018) in R (R Core Team 2021). For Apalachicola Bay only, we assessed whether the number of days Apalachicola River discharge (the primary source of freshwater input to Apalachicola Bay) was below 12,000 CFS (by convention) measured at the Jim Woodruff gage (USGS 02358000) influenced counts of oyster spat. The 12,000 CFS convention has been identified as a reference point at different gage locations in Apalachicola River (Fisch and Pine 2016). This reference point is important because at discharge levels of about 12,000 CFS the adjacent floodplain becomes inundated (Light et al. 1998, Fisch and Pine 2016) although the exact point of inundation may have changed over time due to river bed degradation (S. Leitman, personal communication). Regardless, we use the number of days per Period Apalachicola River discharge was < 12,000 as an indicator of low freshwater inputs.

Because the oyster restoration projects had different starting points in time and cultch materials (Table 1), we summed the weight of cultch collected by divers by Bay, material, site, and period. We then used a similar generalized linear model framework as the live oyster count data to assess patterns in cultch material persistence across projects.

We examined river discharge for a small number of rivers entering each bay as a proxy for salinity and nutrient inputs by plotting the percent deviation in river discharge (CFS by convention) from the period of instrument record by month and year beginning in 2005. We began the time series about 10 years prior to restoration efforts to capture antecedent river discharge conditions prior to restoration beginning. Pensacola Bay has two rivers that enter the bay (Escambia and Blackwater rivers) and we used data from USGS gauge 02375500 from the Escambia River because this is the larger (by discharge) of the two river systems. St. Andrews Bay is unusual in that it has no major freshwater inputs (Crowe et al. 2008) thus no summary of freshwater inputs was made. For Apalachicola Bay we summarized river discharge information from USGS gauge 02358000 (Apalachicola at Chattahoochee). Data and all code used for analyses is available from the following Git repository ABCDEF.

*Results*

Trends in fisheries dependent data

Trends in fisheries dependent data from FWC since the implementation of the mandatory commercial fishery reporting requirements in 1985 show the Apalachicola Bay commercial fishery was several orders of magnitude larger (trips and landings) than Pensacola and St. Andrews bays combined (Figure 2). Apalachicola trips and landings increased sharply during the early 2000’s peaking in 2012 when the fishery collapsed (Figure 2). Apalachicola Bay was closed to oyster harvest by Florida Fish and Wildlife Conservation Commission in December 2020 with a scheduled reopening in 2025. All three bays show a common pattern of upticks in trips and landings in the mid-1980’s and again in the 2005-2010 period. Since 2010 trips and landings have declined in all three bays with extremely low levels of commercial fishing (trips and landings) since 2015 when regional oyster restoration programs began.

Trends in oyster counts across Apalachicola, Pensacola, and St. Andrews Bays across reefs following restoration

The dispersion parameter using the binom2 family formulation of a negative binomial distribution was <1 for all models suggesting extreme overdispersion (example for spat Figure 3). We hypothesized that trends in oyster counts may vary over similarly over time (Period), Bay (Pensacola, St. Andrews, and Apalachicola Bay) or trends in oyster counts may be different among Bays (Period\*Bay) over time and created mathematical models to represent these hypotheses (Table X). For each model we considered site within the Bay as a random effect and used the log of the number of quadrats as an offset to control for differences in sampling effort over time and in each bay. Because our interest is in how counts of oyster spat change over time (as a restoration effort to shift the system from an undesired to desired state) we were most interested in the Period \* Bay interactive model including Period as a continuous covariate. This is because this model provides insight into (1) whether restoration triggered a response in oyster counts over time and (2) if this response was consistent among the three bays. From an AIC perspective, Bay \* Period model had the lowest value (delta AIC between lowest AIC and model with second lowest AIC = 3.3; Table X). For this model, over time, Apalachicola Bay live spat counts declined (beta = -0.17, SE beta = 0.04, p<0.0001). Pensacola and St. Andrews bays both show different trends in live oyster spat counts compared to Apalachicola, but uncertainty in response was higher (Pensacola Bay, beta = 0.07, SE beta = 0.15, p=0.10; St. Andrews beta = 0.25, SE beta = 0.20, p=0.03). Even though the estimated slope is positive for Pensacola and St. Andrews bays, this trend is uncertain (high standard error on beta terms) and the value is low suggesting an increase of about 1 oyster spat per quadrat for each time period (example back transformation exp0.25=1.3). This contrasts with Apalachicola Bay which was declining at about 0.8 live oysters per quadrat per period (exp-0.18 = 0.8). Predicted mean live oyster spat counts (95% CI) for the last period of the time series (period 14) from a single ¼-m2 quadrat are Apalachicola = 6.7 (4.4 - 10.0), Pensacola = 15.3 (1.5 - 160.8), and St. Andrews = 570.5 (33.0 – 9864.0) with only St. Andrews having a predicted (and highly uncertain) increase since the beginning of the time series (Figure 4).

Fitting the same Bay \* Period model to counts of seed or legal sized oysters revealed a similar pattern as seen in live oyster spat – observed and predicted declines in seed oysters over time in Apalachicola Bay, relatively constant values in Pensacola Bay, and increasing, but highly uncertain trends in St. Andrews Bay. St. Andrews Bay was the only system to have at least one live oyster per quadrat predicted (1.6 live legal oysters [0.41 – 6.20 95% CI]) whereas the other bays were predicted to have less than one live legal oyster per quadrat (Apalachicola 0.65, [0.31 – 1.38]; Pensacola 0.14, [0.04 – 0.50]).

A detailed analysis Apalachicola Bay oyster spat response to restoration measured by counts from rock and shell cultch from multiple studies

Analyzing available data and understanding Apalachicola oyster response to restoration actions is complicated because of variability in the construction and monitoring programs used as part of ongoing restoration efforts. In Apalachicola Bay multiple restoration materials (limestone or quarried shell) have been used since 2015 and these materials have been placed in the bay at a variety of densities (Table 1). Because of construction challenges, some sites may have received both limestone and shell. Monitoring efforts to track oyster population response have been similar across studies, but the timing of monitoring post-construction has varied from monitoring beginning within weeks of cultch material being deposited, to monitoring not beginning for 1-2 years following cutch placement because of Covid-19 related delays in completing field efforts. Observed counts of oyster spat by research study highlight these challenges where the number of spat have ranged from 0 to more than 80,000 per 1/4-m2 depending on study and time (Figure 5) suggesting that these data are highly over dispersed, but over time oyster counts across study trend closer to zero (Figure 5).

We combined oyster count data from various surveys and standardized site names. We then fit GLM models assuming a negative binomial distribution to these data to describe the number of oysters of each size class over time (Period) with site as a random effect and the log of the number of quadrats as an offset to control for differences in sampling effort. Results from this model found Period was significant (beta = -0.17, SE = 0.04, p < 0.001) suggesting that over time for each period and across study and cultch material used, and density of cultch material deployed, counts of oyster spat did not respond positively to restoration action. Predicted number of oyster spat per ¼-m2 transect (95%) in Period 14 was 7.1 (4.8 – 10.6) much lower than in Period 1 (102.2, 58.6 – 178.3; Figure 6). We fit the same model as above but included an additional parameter describing the number of days river discharge was below 12,000 CFS in the model. Both Period (beta = -0.21, SE = 0.04, p < 0.001) and the low days term are important in the model with the terms for low days ((beta = -0.006, SE = 0.003, p = 0.07) suggesting that for each day increase in the number of days discharge is below 12,000 CFS the number of oyster spat declines slightly (exp-0.006) by about 1 oyster spat per ¼-m2 quadrat. The same model, but with a one Period lag on the number days discharge was below 12,000 CFS (as a measure of potential influence of antecedent flow conditions), suggested that the number of low days in the prior period did not influence the number of live spat in the current period (p = 0.27).

An examination of the different projects, which were deployed in different periods and monitoring begin in different periods, does not provide clear patterns into how counts of oyster spat change over time. We fit a GLM model assuming a negative binomial distribution that included Period and Project (four different projects, two using rock and two using shell) to the observed counts of oyster spat per quadrat. Comparisons of the performance of project in producing oyster spat are difficult because of variations in the timing of when the monitoring began on each project. As an example, for one project monitoring did not begin until nearly two years following construction, and if the response of cultch to restoration is different two years following restoration than immediately after restoration, then this would not be clear. However, the intent of the restoration is to provide substrate in a way that will allow colonization and accretion of material over many years, so if the restoration is successful, the count response should persist over multiple years. To create a comparative framework across studies with different materials and starting points, we predicted the number of oyster spat per ¼-m2 in period 14, the last period of monitoring. In this comparison three studies would have completed their construction efforts 3-5 years prior (NFWF-1, NRDA 4044, NRDA 5007) and FWC-2021 would be < 2 years since construction. If time since construction were a major influence, then the predicted values for each study in the common period should differ. For the NFWF-1 project which used quarried shell cultch, we predict in Period 14 about 26.2 (95% CI 8.6 - 79.4) live oyster spat per ¼-m2 quadrat. The NRDA 4044 project also used quarried shell cultch and the mean predicted number of live spat in Period 14 was lower at about 3.5 (95% CI 1.7 - 7.1). For the projects that used rock cultch the predicted number of live oyster spat per ¼-m2 quadrat vary. For project NRDA 5007 mean predicted live oyster spat count per ¼-m2 quadrat was about 15.4 (95% 8.3 – 28.3), and project FWC-2021 mean predicted = 7.0 (4.5 - 10.9). An interesting result is that the most recent (existing fewest number of years) constructed reef project FWC-2021 had predicted counts that was lower than the older rock cultch project NRDA 5007. Project NFWF-1, a shell cultch project, had very high initial (soon after restoration) observed live oyster spat counts that were more than 100x those any of the other projects (Figure 5). The extreme dispersion observed for this project (Figure 5, observed counts) resulted in poor model fit.

Total cultch weights for Apalachicola Bay were made integers by rounding to nearest whole kilogram. Data were then subset for each project and calculations of mean and variance by project suggested the data were over dispersed (variance > mean). We then fit similar GLM models assuming a negative binomial distribution as described for oyster count data to the observed cultch biomass. To create a comparative framework across substrates we predicted the amount of cultch per ¼-m2 in period 14, the last period of monitoring. Because Apalachicola was the only bay where both rock and shell were used, we focused analyses to compare substrates on this bay only. From an AIC perspective, models that included Period + substrate or models that examined the interaction between Period\*Substrate (both with log(number of transects) as an offset to control for effort) were not distinguishable (delta AIC between top models = 1.5). From a management perspective the interaction term is of interest to help understand how the biomass of either rock or shell changes over time. For rock substrate, the change in biomass over time was significant (beta = -0.08, SE = 0.03, p = 0.01) but the change was not significant for shell (beta = -0.05, SE = 0.04, p = 0.5). However, what is more important than the statistical significance is how the material persisted over time – the slope is negative for both substrates indicating declines over time. The predicted biomass of rock per ¼-m2 quadrat changed over time (Figure 7) from about 5.07 kg per ¼-m2 quadrat (95% CI 2.5 – 10.2) in Period 2 to about 2.0 kg per ¼-m2 quadrat (1.4 – 2.9) whereas the biomass of shell changed from about 1.7 kg per ¼-m2 quadrat (1.1 – 2.7) to about 0.93 (0.6 – 1.5). Because shell is less dense than rock, the differences in biomass per quadrat are not surprising - these results suggest a decline of about 60% biomass for the shell material by the end of period 13 and about 45% of the rock material. A critical point is that these are measures of biomass, not area, and oyster spat settle on area. However, if we assume a proportional loss in area to what is loss in biomass then shell biomass and likely areas degraded at a faster rate than rock material. Finally, we assessed the relationships among the biomass of cultch and number of live oyster spat from each quadrat. We graphically examined the relationship between the weight of cultch and number of spat per quadrat across projects in Apalachicola Bay and found no clear pattern (Figure 8). This is important because it suggests that for a given biomass of cultch, or at least across the range in cultch biomass observed in Apalachicola, the number of live spat observed can vary widely. How the biomass of cultch that persists on reefs at present relates to biomass of cultch when oyster populations were higher and supporting a commercial fishery is unknown.

River discharge as a proxy for salinity and nutrient patterns

Apalachicola River discharge deviated significantly (i.e., 50-100% below period of record) for three or more months in 2006, 2007, 2008 with extreme drought in 2011 (9 of 12 months), 2012 (12 of 12 months), 2016 (6 of 12 months), and 2017 (4 of 12 months). Escambia River discharge patterns were generally similar reflecting the regional effects of drought (Figure 9). Regional river discharge patterns in 2019-2021 generally been average to above average for most months (Figure 9).

*Discussion*

Our results suggest restoration efforts in Pensacola, St. Andrews, and Apalachicola bays have not had the intended response of shifting oyster populations from a resilient, low oyster state to a more desired, high oyster state. This conclusion is based on available data from three different bays in different watersheds, but with common restoration materials, techniques, and monitoring programs. Restoration efforts in all three bays are guided by previous efforts in Apalachicola Bay where irregular clutching has been part of oyster management efforts since at least 1949 (Whitfield and Beaumariage 1977). Hurricane Elena in 1985 is thought to have had major impact on oyster populations in Apalachicola Bay (Berrigan 1988, 1990), reducing oyster populations by as much as 95% (Livingston 2015). However, a very rapid population recovery was observed in Apalachicola Bay following Hurricane Elena (Berrigan 1990), for reasons that may or may not be singularly related to restoration (Fisch and Pine 2016). The observed changes both to the physical (Edmiston et al. 2008) and biological (Berrigan 1988; Edmiston et al. 2008; Livingston 2015) aspects of Apalachicola Bay post-hurricane Elena led to an intensive restoration effort for oyster populations including fishery closures, experimental clutching effort, and on-water check stations following fishery re-opening to monitor harvests (Berrigan 1990). Restoration efforts detailed in Berrigan (1990) identify that 156 ha of oyster reef received 472 m3 of *Rangia* clamshell per ha as part of the intensive restoration. Livingston et al. (1999) describe a major wild oyster spat set occurring in the fall of 1985 on remaining oyster reefs in Apalachicola Bay and subsequently within 18 months restored oyster bars (monitored as part of restoration; Berrigan 1990) were supporting 587 oysters/m2. Oyster population benchmarks to support harvest were met (Berrigan 1990), and the oyster fishery reopened, with reported landings and related excise fees recovering the cost of restoration (Berrigan 1990). If previous restoration efforts were successful, why is a similar response not observed from ongoing restoration efforts now?

*Cultch density, volume, area, and material*

The cultch density used by Berrigan (1990) of about 617 yds3 per acre is about twice the density used in the largest (rock cultch; NRDA 5007) and most recent (rock cultch; FWC 2021) restoration efforts and about 50% more than the highest treatment level of recent shell cultch projects (NFWF 1; Table 1) for Apalachicola Bay. Comparative data on historic clutching efforts for Pensacola and St. Andrews bays are not available. Pine et al. (2015) used a model fit to historic Apalachicola fisheries dependent and independent data to predict that the risk of an irreversible oyster fishery collapse in Apalachicola could be reduced through an intensive clutching program over an area of about 400 ac per year. This clutching area is about four times the average area cultched each year between restorations following Hurricane Elena (Berrigan 1990; Pine et al. 2015) and the beginning of regional restoration in 2015.

Importantly, these recommended or observed cultching levels are area estimates (400 ac recommend from simulation, 100 ac restored on average since the mid-1980s) which describe the surface area of cultch available for spat to settle. Pine et al. (2015) identified a key uncertainty to address of determining the volume to cultch to use in these efforts. Critically the volume of cultch material (as yd3) of material is what determines the vertical relief added to the extant reef for a given area of cultch. This vertical relief difference (the height from the bottom) may be important in terms of elevating the reef into suitable water quality or hydrodynamic conditions and has elevation recently been shown to be a critical factor in reef persistence or collapse (Colden et al. 2017). In an assessment of oyster reef restoration in the Chesapeake Bay region, Colden et al. (2017) found that oyster reefs higher than 0.3 m had higher oyster survival, density, and overall complexity than reefs < 0.3 m and these reefs higher reefs were less likely to go extinct. In 2017 the National Academy of Science (NAS) highlighted the NFWF 1 project assessed in this study as an example restoration project designed to experimentally assess oyster population responses to different cultch density treatments (NAS 2017). However, that experiment does not appear to have answered the questions as designed, perhaps because of construction challenges leading to limited contrast in elevation among the different cultch treatments.

Side-scan sonar mapping is routinely used as an assessment metric on a sub-set of restored reefs in Pensacola, St. Andrews, and Apalachicola bays and the restored reefs in these systems are highly variable in their observed elevation profiles. This is due to a combination of factors including uneven placement of material during construction and erosional and depositional shaping forces of shell and rock cultch materials from currents since construction. Because the material used for restoration efforts is either small and dense (#4 limestone 1.5-3” in size) or larger and less dense (quarried oyster shell X-Y” in size) it is likely susceptible to being transported away from the intended restoration site, buried in sediment, or sculpted by currents to a low-relief structure (2-4” of elevation, about 0.05-m) interrupted by subtle waves of higher-density of material (volumetrically) which results in higher vertical relief (?-?”, about Z-m). Regardless, cultch material in various forms at different biomass levels has persisted on these restored reefs (Figure 8), and critically oyster spat settlement on this material has been very low for reasons that are not known.

Smith et al. (2021) in a 15-year assessment of the performance of an oyster reef restoration project in the Chesapeake Bay found that restored reefs were like unrestored reference reefs based on a variety of metrics within six years following restoration. For some metrics such as shell height, the restored and reference reefs were similar within three years, and as the restored reefs aged, they were observed to become more stable and possibly more resilient. In Florida the restoration of Lone Cabbage Reef in Suwannee Sound demonstrated oyster spat settlement and persistence on the restored reef within six months following construction and oysters have persisted and successfully settled on the reef in each of the four years since construction and density on the restored and nearby reference reefs are now similar (W.E. Pine, *personal communication*; Pine et al. March 2022 quarterly report to NFWF). Relative restored reef elevation response (the increase in elevation) from the Smith et al. (2021) project in the Chesapeake Bay was about 0.14-m (see online supplemental information Smith et al. (2021)) and for the Lone Cabbage project in Florida was about 0.36-m (Pine et al. 2022). Combined with the results from Colden et al. (2017; 0.4-m) reported elevation changes associated with restored reefs which persisted over time are about 3-8x the elevation contrast than what has been constructed in Apalachicola or from the DEP studies in Pensacola or St. Andrews Bay. The Nature Conservancy is currently constructing a series of reefs in Pensacola Bay that use larger rock material, but the elevation change from this treatment is not yet known (Laura Geselbract, TNC, *personal communication*).

Materials used for reef construction and various oyster restoration efforts vary widely (Bersoza Hernandez 2018; Goelz et al. 2020) including in Florida where restoration materials include multiple types of limestone, quarried oyster shells, recycled clam shells, crushed granite. and various types of artificial materials (OIMMP documents?). While the State of Florida statutes are difficult to interpret, one section of the relevant statutes is 597.010

(23) OYSTER AND CLAM SHELLS PROPERTY OF DEPARTMENT.—

(a) Except for oysters used directly in the half-shell trade, 50 percent of all shells from oysters and clams shucked commercially in the state shall be and remain the property of the department when such shells are needed and required for rehabilitation projects and planting operations, in cooperation with the Fish and Wildlife Conservation Commission, when sufficient resources and facilities exist for handling and planting such shell, and when the collection and handling of such shell is practicable and useful, except that bona fide holders of leases and grants may retain 75 percent of such shell as they produce for aquacultural purposes.

From this statute, it appears that Florida requires 50% of the shell material removed from harvest to be retained for public use including planting (cultching). However, critically these statutes exempt shell material for oysters used in the half-shell market from this requirement, thus oysters sold as half-shell (a national market) are not required for re-use. Cultching efforts in Apalachicola Bay were at one time identified as a contributing factor to the long-term sustainability of harvest in the Bay (Zu Ermgassen et al. 2012) but the observed oyster fishery collapse in 2012 and a combination of modeling (Pine et al. 2015; Johnson et al. 2022) and empirical assessments (this work) suggest that may not be the case. Efforts are underway (J. Casteel, University of Florida grad student) to reconstruct shell biomass (volume) removed by harvest for the major oyster fisheries in Florida and her preliminary work suggests that shell material removals far exceed material placed by recent restoration efforts (Table 1) or historically (Pine et al. 2015, Figure 3). Understanding the magnitude of removals is likely important to gauge the scale of restoration required to shift degraded oyster populations from undesired to desired status (Johnson et al. 2022).

*Resistant to restoration or resistant to learning?*

There are at least five key takeaways from this analysis:

(1) Oyster populations in Pensacola, St. Andrews, and Apalachicola Bays do not appear to have responded to restoration efforts designed to promote spat settlement and accelerate population recovery. Critically this lack of response is occurring in different bays, within different watersheds, and at least within Apalachicola, with different restoration materials.

(2) The lack of measured population response has happened during a period when river discharge ranged from moderate drought to generally normal river discharge for the period of instrument record. Because river discharge is thought to be a major driver of salinity, and salinity a major driver of oyster survival (demonstrated by the State of Florida position in FL v GA) this would suggest that salinity (and other river-related ecosystem drivers such as nutrients) have been near normal in recent years for the period of instrument record. This lack of response has also happened while commercial fisheries have been closed by statute (Apalachicola) or appear to be extremely low based on commercial fisheries landings information (Pensacola and St. Andrews).

(3) The major projects described in Table 1 are primarily focused on subtidal oyster reefs – reefs that are not dewatered at some predictable tide level. The Lone Cabbage Reef restoration project has shown a positive response to restoration, and that project did focus on the restoration of intertidal oyster reefs that do dewater at a predictable tide level. Whether restoration practices used for Lone Cabbage can be extended to subtidal habitats is unknown.

(4) Careful assessment should be given to the types of restoration materials used. The Berrigan (1990) restoration project which is identified as a successful project in Apalachicola Bay used clam shells dredged from Lake Pontchartrain, LA as the cultch material. Smith et al. (2021) also used dredged clam shell in the Chesapeake Bay. Oyster shell material is cited as one of the most effective shell restoration materials (Frederick et al. 2016, summary figure across projects based on LaPeyre). The Lone Cabbage Reef project used a type of soft limestone that is of a geologic formation that is part of the exposed Florida platform within Suwannee Sound, and the limestone was quarried from within the Suwannee River basin (near Branford). This soft limestone is described as part of the “Ocala” formation, which is a term used to describe a variety of soft limestone types that are of relatively young age (thousands of years). Restoration projects in Pensacola, St. Andrews, and Apalachicola bays that have used limestone (Table 1) use material transported from Kentucky that upon examination appears to be denser (structure and mass), older (geologic age), and possibly comprised of less calcium carbonate based on a simple acid reactive test than the Ocala formation material used on Lone Cabbage (Jon Yeager, UF Geology, *personal communication*). A formal chemical and physical analysis of both types of limestone are ongoing now by the UF Geology Department.

(5) Pine et al. (2015) highlighted the risk of a catastrophic and persistent collapse in the Apalachicola oyster fishery if oyster recruitment levels remained below the average observed in the available fisheries independent monitoring data (1990-2013) used in their analyses. Johnson et al. (2022) further demonstrated the risk of a transition to a stable, resilient, low population state for oysters and the difficulty in reversing this to a more desired state in a generalized oyster population model. There is significant concern that the Pensacola, St. Andrews, and especially Apalachicola Bay oyster populations are in such a degraded state that no restoration or management action considered may be effective in altering current conditions.

*Conclusions*

Based on the empirical assessment of oyster recruits, seed, and legal-size oysters for Pensacola, St. Andrews, and Apalachicola Bay, low recruitment levels have persisted, and these systems may have transitioned to a low productivity state that has proven resistant to restoration at the scales undertaken (Table 1), management actions (Apalachicola fishery closure), or environmental conditions (generally normal river discharge levels). Unfortunately, a combination of experimental design deficiencies (e.g., absence of controls, lack of strong treatment contrasts) make it difficult to determine which factors that have been previously hypothesized to drive oyster population dynamics (i.e., river discharge, fishing effects) or are necessary components of successful restoration (reef material, area, or height) to learn going forward. In absence of an ability to evaluate these factors from the available data, we are left with a comparative assessment to restoration projects that have proven successful from the Chesapeake Bay region (e.g., Colden et al. 2017; Smith et al. 2022) or Florida (Pine et al. 2022) or an example wild oyster fishery that is highly regulated, carefully monitored, and adaptively managed that appears sustainable (Delaware Bay; https://hsrl.rutgers.edu/SAWreports/index.htm). Unfortunately, many of the same restoration and management uncertainties identified in this assessment have persisted for long periods of time in Florida (decades to centuries; Swift 1897, Pine et al. 2015, Camp et al. 2015). This resistance to learning to inform restoration is a widespread problem in ongoing restoration efforts in the Gulf of Mexico (NAS 2022) and has been a challenge to large-scale restoration and management efforts for decades (Walters 1986; Gunderson 1999; Walters 2007; Pine et al. 2022). Gunderson (1999) in a classic assessment of learning and barriers to learning in adaptive ecosystem assessment and management (AEAM) suggests

A central tenet of AEAM is learning, yet learning seems to be intertwined with cycles of policy success and failure (Westley 1995). If policies are working (or appear to be working), there is little or no emphasis on learning. It is when policy fails, either dramatically or chronically, that learning is deemed necessary and a priority. The challenge to develop a capacity for learning continues to be problematic among most resource institutions. Yet, when needed, that capacity seems to come by focusing on understanding (not efficiency) and by networking with those who practice learning.

Oyster restoration programs in the Florida panhandle appear to again be at a critical crossroads (Swift 1898; Pine et al. 2015; Camp et al. 2015; Pine et al. 2022) but are likely at their lowest point in the last 125 years (Swift 1897, 1898; this assessment). Stronger leadership and a commitment to learning are needed to guide these restoration and management efforts to achieve their stated goals of restoring oyster populations to support ecosystem services and viable fisheries for the benefit of the people of Florida and the Gulf of Mexico region.

*Acknowledgments*

*References*

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*Figures*

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###This will be a really great map of the panhandle with the FL inset

Figure 1. Pensacola, St. Andrews, and Apalachicola bays…

A picture containing scatter chart

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Figure 2. Publicly available fisheries dependent data from the Florida Fish and Wildlife Conservation commission (<https://myfwc.com/research/saltwater/fishstats/commercial-fisheries/landings-in-florida/>). Each row represents a different bay (Apalachicola top row, Pensacola middle row, St. Andrews bottom row) and each column represents a different metric with the commercial trips in the first column, middle column as CPUE (catch-per-unit-effort), and last column as the landings (by convention in pounds). Note the y-axis are different on most panels by row because of the large differences in observations for each Bay.

Chart, scatter chart

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Figure 3. Period of time (x-axis) and spat CPUE (y-axis) per quadrat in each of the three study systems (Apalachicola, Pensacola, St. Andrews bays). Even number Periods are winter (November-April) beginning in 2015 while odd number Periods are summer (May-October) beginning in 2016.

Chart, histogram

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Figure 4. Predicted count of live spat (y-axis) by period of time (x-axis) for a single ¼ m2 quadrat from each of the three study systems (Apalachicola, Pensacola, St. Andrews). The black line is the best predicted values for each period and the grey ribbon represent the 95% confidence intervals around this line of best fit. Even number Periods are winter (November-April) beginning in 2015 while odd number Periods are summer (May-October) beginning in 2016. Predictions are made for a single quadrat because of the large differences in the average number of quadrats completed in each Bay. Predicting for a single quadrat allows for comparisons of the predicted count, for a standardized unit of effort in each Bay, as a measure of abundance and population trajectory over time. Note the large differences in the y-axis for each plot.

Chart, scatter chart

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Figure 5 Version 1. Live oyster spat CPUE (y-axis, counts per ¼ m2 quadrat) from each study over time (Period, x-axis). Each panel is a different study completed by DEP or FWC. Even number Periods are winter (November-April) beginning in 2015 while odd number Periods are summer (May-October) beginning in 2016. The NFWF\_1 study uses shell cultch and the other studies use rock cultch.

Chart, scatter chart

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Figure 5 Version 2. Total live oyster spat (y-axis) from each study over time (Period, x-axis). This figure will probably go away because it isn’t standardized, but it is just an example of what the different projects are counting in Apalachicola. Even number Periods are winter (November-April) beginning in 2015 while odd number Periods are summer (May-October) beginning in 2016.

Chart, scatter chart

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Figure 6 Preamble. This is an example plot to demonstrate fit of the nbGLM from TMB. These data (dots on the plot) are the sum of the rounded weights of cultch from the NFWF\_1 study. The model in R is written as Roundwt ~ Period + offset(log(Num\_quads) and is fit to a subset of the data which is only the NFWF\_1 study. This is just a simple approach of sub-setting the data compared to fitting the interactive model, but both will fit and the values are nearly identical. I did both approaches to explore model performance. The predicted value (solid black line) is the predicted total (sum) rounded weight of cultch for an average number of quadrats (150) predicted for every period. The ribbon is the 95% confidence interval around the predicted value. The y-axis is large because this is the amount of material that would come from 150 quadrats. This plot is just inserted to demonstrate visually the performance of the nbGLM using TMB predicted values compared to the data. This same type of model will be used for live spat counts and cultch biomass.

Chart, histogram

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Figure 6 Preamble. This is an example plot to demonstrate fit of the nbGLM from TMB. The model in R is written as Sum\_spat ~ Period \* Project + offset(log(Num\_quads), which is an interactive model allowing for a unique slope for each Project across periods. These data (dots on the plot) are the total number of live spat for each period and site from the NFWF\_1 study. The predicted value (solid black line) is the predicted rounded weight of cultch for an average number of quadrats (150) predicted for every period. The ribbon is the 95% confidence interval around the predicted value. The y-axis is large because this is the amount of material that would come from 150 quadrats. This plot is just inserted to demonstrate visually the performance of the nbGLM using TMB predicted values compared to the data. This same type of model will be used for live spat counts and cultch biomass.

Graphical user interface, chart, histogram

Description automatically generated

Figure 6. These are the predicted live oyster count for a single ¼ m2 quadrat predicted using a nbGLM model in R generally written as Sum\_spat ~ Period \* Project + offset(log(Num\_quads), which is an interactive model allowing for a unique slope for each Project across periods. The predicted value (solid black line) is the predicted number of live spat for a single quadrat for every period. The ribbon is the 95% confidence interval around the predicted value. All studies had more than one quadrat sampled, and no study sampled in all periods. I have predicted over all periods and for a single quadrat to demonstrate the difference in predicted number of live oyster spat for a common level of sampling effort (a single quadrat) to demonstrate both the variability in predicted counts and population trajectory over time as a representation of live oyster spat trends for each study. This utility of this plot is up for discussion.

Graphical user interface

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Figure 7. Predicted change in cultch biomass from the four different studies in Apalachicola. The model in R is written as Roundwt ~ Period + offset(log(Num\_quads) and is fit individually to subsets of the data which represent the different studies. The predicted value (solid black line) is the predicted total (sum) rounded weight of cultch for a single quadrat for every period summed across sites. The ribbon is the 95% confidence interval around the predicted value. All studies had more than one quadrat sampled, and no study sampled in all periods. Predictions are only made for the periods that were sampled. The utility of this plot is up for discussion.

Chart, histogram

Description automatically generated

Alternate Figure 7. Predicted change in cultch biomass from a single study (NFWF 2021) in Apalachicola. The model in R is written as Roundwt ~ Period + offset(log(Num\_quads) and is fit individually to data from a single study. The predicted value (solid black line) is the predicted total (sum) rounded weight of cultch for a single quadrat for every period summed across sites. The ribbon is the 95% confidence interval around the predicted value. I can force the prediction and plotting for periods that were not sampled (as above, no sampling for FWC 2021 in Periods 2-11. But I don’t like predicting over a period of time when there are no data. The utility of this plot is up for discussion.

Chart, scatter chart

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Figure 8 Alternate. Live oyster spat (y-axis) and weight of cultch (x-axis, kg) for each quadrat across Period in Apalachicola Bay by study. The y-axis limited to a value of 1000 because of the high values of live counts observed in Period 2. For Apalachicola Bay the NFWF\_1 and NRDA\_4044 studies are shell cultch and the NRDA\_5007 and FWC\_2021 are limestone cultch.

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Figure 9. Deviations in river discharge from the instrument period of record from the Escambia and Apalachicola rivers. Darker colors equate to larger deviations from period of record with colors in the blue spectrum representing positive deviations (higher river discharge) and colors in the red spectrum representing negative deviations (lower river discharge). White, or near white colors represent values equal to the period of record or within +/- 10%.

Chart, scatter chart

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Figure 10. Live oyster spat CPUE across study and site from Apalachicola Bay (y-axis) and number of days Apalachicola River discharge (Chattahoochee gauge) is below 12,000 CFS (x-axis). The 12,000 CFS threshold is generally considered the point in which the river begins to inundate the floodplain. The more days the river is below 12,000 are periods of time discharge is low.