San Francisco Bay Living Shorelines Project at Giant Marsh, Point Pinole Regional Shoreline: Final Summary Report December 2025

Katharyn Boyer[1], Marilyn Latta[2], Chela Zabin[3], Damien Kunz[4], Susan De La Cruz[5], Jeanne Hammond[6], Margot Buchbinder[1], Jeff Blumenthal[3], Michelle Orr[4], Edwin Grosholz[7], Laurie Hall[5], and Tanya Graham[5]

[1] Estuary & Ocean Science Center, San Francisco State University
[2] California State Coastal Conservancy
[3] Smithsonian Environmental Research Center
[4] Environmental Science Associates
[5] U.S. Geological Survey
[6] Olofson Environmental
[7] University of California Davis

Photo C. Zabin

This report may be cited as:
Boyer, K., M. Latta, C. Zabin, D. Kunz, S. De La Cruz, J. Hammond, M. Buchbinder, J. Blumenthal, M. Orr, E. Grosholz, L. Hall, and T. Graham. 2025. San Francisco Bay Living Shorelines Project at Giant Marsh: Final Summary Report.

Executive Summary
The San Francisco Bay Living Shorelines Project (LSP) at Giant Marsh was developed and managed by the State Coastal Conservancy in collaboration with a broad range of partners to continue demonstrating and studying living shoreline techniques in San Francisco Bay. The Conservancy's role as lead agency included technical involvement and a primary role in coordination of all project and design planning, grantee and contractor identification and management, environmental compliance and permitting, fabrication and construction coordination and implementation, post-construction monitoring, and raising and managing funding. Project partners included San Francisco State University's Estuary & Ocean Science Center, Smithsonian Environmental Research Center, Environmental Science Associates, US Geological Survey Western Ecological Research Center, and Olofson Environmental, Inc. Landowning partners included East Bay Regional Park District and the State Lands Commission (who owns a small portion of the project area). Construction was managed by Ducks Unlimited and included Triton Marine, Reef Innovations, Dixon Marine Services, and two local Conservation Corps: CiviCorps and Conservation Corps North Bay. The project was funded by the State Coastal Conservancy and with federal funds from the US Fish and Wildlife Service and the National Oceanographic and Atmospheric Administration.

The Giant Marsh LSP aimed to advance understanding of nature-based methodologies in adaptation to sea level rise in San Francisco Bay and build on lessons from the first living shorelines project in the Bay in San Rafael conducted by many of the same project partners. The Giant Marsh LSP included shoreline restoration and enhancements from the shallow intertidal up to the marsh/terrestrial transition zone, to promote wildlife habitat and test the effects of these modifications on physical processes that influence shoreline erosion. The project partners aimed to create a linked gradient of enhancements to promote a functional estuarine system that supports key species at various life stages and allows for connectivity between multiple habitat types.

The project focused on enhancement of key native foundational species: Olympia oysters (*Ostrea lurida*), eelgrass (*Zostera marina*), Pacific cordgrass (*Spartina foliosa*), California sea-blite (*Suaeda californica*), perennial pickleweed (*Salicornia pacifica*), and several estuarine-terrestrial transition zone plant species. Historically, native Olympia oysters were an abundant and ecologically important part of the fauna in West Coast estuaries. Studies of oysters in eastern US estuaries have shown that native oyster reefs act as a foundation species by creating a refuge from predators and physical stress as well as a food source resulting in increased local diversity of fishes and invertebrates. In the largely unstructured, soft sediment habitats of West Coast estuaries, aggregations of native oysters were likely to have provided similar functions and have been shown to increase invertebrate species richness. Seagrasses, such as native eelgrass (*Zostera marina*), support diverse communities of sediment infauna, epibenthic invertebrates, fishes, and waterfowl, provide attachment locations for algae and encrusting invertebrates, and stabilize sediments via extensive rhizomatous root systems. As the only vascular plant species that grows in the low native salt marsh zone and as a co-dominant in the mid-marsh plain in San Francisco Bay, Pacific cordgrass provides critical habitat cover and food resources for a variety of species. California sea-blite is a federally endangered high marsh shrub that is a focus of recovery efforts within San Francisco Bay, and which may be beneficial in providing refuge to birds and mammals during high tides. Perennial pickleweed is the dominant plant throughout California's salt marshes and provides vertical structure needed by a number of birds and mammals. Estuarine-terrestrial transition zone species provide structure habitat cover and high tide refuge for endangered salt harvest mouse and California Ridgway's rail during winter storms and high tide events.

Our interdisciplinary and multi-institution project team designed a multi-habitat and multi-objective living shorelines approach that included seven treatments:

1: Offshore eelgrass and oyster/habitat reefs
2: Oyster/habitat reefs at a range of elevations (with test of rockweed shading)
3: Nearshore oyster/habitat reef near the marsh edge
4: Pacific cordgrass revegetation inshore of nearshore reef
5: Pacific cordgrass revegetation adjacent to existing Pacific cordgrass
6: Marsh scarp revegetation and arboring of California sea-blite and pickleweed
7: Marsh-upland transition zone enhancement

The Project area comprises 368 acres of existing tidal marsh, intertidal oyster reefs, mudflats and subtidal aquatic beds. The seven treatments were placed within a roughly two net acre (0.8 hectares) area of net beneficial fill to build the project. The treatments were carefully designed and monitored to assess performance and outcomes at a pilot scale, with iterative stepwise goals of scaling up successful components towards larger outcomes in future larger-scale projects at a variety of sites across the bay.

Brief descriptions of objectives and methods are included in this summary report; details can be found in the Project Description (ESA, SCC 2017) and in the project team's detailed final progress reports (Buchbinder et al. 2024, 2025a; Graham et al. 2024; ESA 2024; Zabin and Blumenthal 2024; Hammond 2025). Most treatments were implemented in 2019; however, native cordgrass plantings in Treatments 4 and 5 were delayed until early 2022 when absence of invasive hybrid cordgrass was confirmed, and Treatment 7 plantings at the terrestrial transition were also delayed to early 2022 to allow for further planning and coordination with the East Bay Regional Parks District. In this final report, we summarize findings primarily through the fifth project year, 2023, with treatments 4, 5, and 7 monitored through fall 2024. Highlights of these findings include the following, by treatment:

<u>Offshore Eelgrass Bed and Oyster Reef (Treatment 1)</u>
This treatment included three plots of 60 reef balls placed at a depth of -1.6 to -1.7' NAVD88, with each reef ball topped with clean (cured to avoid unintentional spread of pathogens or invasive species; Cohen and Zabin 2009), recycled Pacific oyster half-shell in bags, for a total of 180 combination reef ball/shell bag elements. Eelgrass was planted adjacent to the reefs (bayward or shoreward) or in three additional, separate plots. This treatment was intended to test the effectiveness of combination reef ball / shell bag structures in supporting native Olympia oysters, as well as habitat for other species including macroalgae, sessile and mobile invertebrates, fish, and waterbirds. Further, we sought to understand how eelgrass planting position relative to the reefs (bayward, shoreward) or planted alone, as well as at a range of densities of plantings, would influence both eelgrass growth characteristics and habitat values. We expected that the offshore reefs would reduce flow to the benefit of eelgrass on the shore side, and could increase organic carbon accumulation in the sediments.

The recruitment of oysters to the reefs was minimal in the first summer, 2019, likely relating to the previous extended wet winter (resulting from a series of atmospheric river events) and prolonged low salinity below levels known to be lethal to oysters, and possibly also due to high temperatures that impacted intertidal species in the Bay Area that year. Oysters were first recorded in Treatment 1 in 2020 and the overall population at the site increased each year through 2022, when we estimated 1.3 million oysters. In the project's 5th year, another wet winter again reduced salinity in the project area, resulting in a loss of oysters across the project and in nearby natural oyster populations. In 2023, we estimated

320,000 oysters on the combined Reef Balls and shell bags; though this was a substantial decline, densities were slightly higher than those at similar elevations at a nearby reference site.

Eelgrass (*Zostera marina*) planted in June 2019 at the south reef area suffered losses in a closely following, unseasonable heatwave, and this area was replanted in May 2021; otherwise, eelgrass established well without further intervention. Overall, we showed that higher density plantings increased total numbers of shoots present, though not in proportion to the initial planting density (i.e., 4x greater planting density did not result in 4x greater shoot numbers). We found evidence of greater eelgrass abundance on the shore side of oyster/habitat reefs, compared to the bay side of reefs or no reefs. In contrast, the worst outcomes in both patch sizes and total shoots came from low density plantings with no association with reefs, suggesting that eelgrass planted on its own may benefit from higher density plantings. Invertebrate communities, including both epifaunal invertebrates and large mobile invertebrates, benefitted from oyster reefs as well, with species richness, diversity, and abundance often higher in reef-associated eelgrass compared with eelgrass growing alone.

We included multiple methods of fish sampling to maximize understanding of fish usage of restored eelgrass, with and without reefs present and relative to bare mudflats and natural eelgrass at a similar elevation. We found hoop nets captured larger fish on average than seines, which represented fish of a broader number of sizes, while eDNA picked up additional species not detected by other methods. No strong patterns were detected by treatment, but there was a trend of greater catch per unit effort (CPUE) in restored eelgrass with hoop nets and in natural eelgrass with seines. ARIS sonar captured more fish than physical sampling methods, and showed a trend of greater CPUE in eelgrass associated with reefs compared to restored eelgrass alone. However, neither had a greater CPUE than in bare mudflat. In general, we struggled to find a fish monitoring method that adequately represented abundances, sizes, and species richness of this complex reef/eelgrass treatment.

There was no change overall in waterbird abundance as a result of the installation of Treatment 1, with eelgrass and mixed eelgrass and oyster/habitat reef areas taken together. However, we did detect a difference early in the project (2020) when a statistically significant interaction between treatment and treatment period indicated that eelgrass and mixed eelgrass and oyster/habitat areas had a positive effect on waterbird abundance at low tides. Behavioral differences were not detected between control and treatment grid cells, nor were there differences in waterbird community composition. We also did not detect a significant effect of any treatment type (eelgrass, eelgrass/reef, reef) on infaunal invertebrate density or biomass compared to controls (areas with eelgrass that had recruited circa 2017).

The offshore reefs reduced relative water motion in winter, and there was a trend of organic carbon accumulation at sediment core depths of 4-12 cm (1.6-4.7 inches) near to the reefs on the shore side relative to the eelgrass-restored areas without adjacent reefs. However, organic carbon was not found to have accumulated in the surface sediments, suggesting that the reefs may have increased turbulence and led to inconsistent settlement of fine particles in close proximity to the reef structures. A small loss of elevation in sediments around reefs compared to the reefs themselves (subsidence of 0.18 m [0.6 feet] versus 0.12 m [0.4 feet], respectively), also suggests a minor amount of erosion occurred near the reef structures.

Reef Elements at Different Elevations, with and without shading from Rockweed (Treatment 2)

Closer to shore, Treatment 2 consisted of three rows of 10 oyster blocks per row set at three tidal elevations: a low elevation row at -0.10’ (NAVD88), a mid-elevation row at 0.93’, and a high-elevation row at ~1.86’. This treatment was intended to compare oyster recruitment on oyster blocks at a range of elevations, and to determine if transplants of a brown alga, Pacific Rockweed (*Fucus distichus)*, could counter the desiccation stress likely to affect oysters at higher elevations.

Oyster densities generally decreased with increasing tidal elevation, but oysters at the deepest Treatment 2 reef were more abundant than those on the Reef Balls (Treatment 1) for the first four years of the project. In 2023, oysters on Reef Balls fared better than those higher in the intertidal zone. There was a trend of higher oyster densities on the north sides of Treatment 2 oyster blocks in 2020, which became statistically significant in 2021-2022. In 2023, likely due to low numbers of oysters, there was no difference by aspect. These patterns suggest that oyster outcomes are affected by a combination of factors, including space competition and heat stress, which may become more significant if it follows low salinity stress.

Transplants of the brown alga *Fucus distichus* (on cobble) to the oyster blocks (intended to test for benefits of shade and moisture for oyster recruits) resulted in low survival. However, surviving algae had lower temperatures and higher relative humidity under the canopy, which corresponded with higher cover of both oysters and barnacles.

Focal observations of waterbirds showed waterbird abundance at low tide increased significantly after the installation of Treatment 2 and 3 oyster/habitat reefs, but did not change at high tide. Ruddy ducks spent significantly more time foraging, and less time engaged in other behaviors (alert, comfort, drinking, sleeping, preening, resting, and social) during the post-treatment as compared to the pre-treatment period. However, time spent on foraging and other behaviors did not differ significantly between the pre- and post-treatment periods for greater and lesser scaup and willets.

<u>Nearshore Oyster Reef Elements at Existing Marsh Edge (Treatment 3)</u>
This treatment consisted of 140 oyster blocks parallel to the shoreline, set in two staggered rows separate by a corridor, at an elevation of 1.5’ NAVD88. We specifically designed the 140 oyster block elements placed in two rows at a higher intertidal elevation in close proximity to the eroding tidal marsh edge in order to test shoreline protection benefits and oyster recruitment benefits. The design for this set of reefs is based on the intention that it would provide best near-term oyster habitat at the lower elevations of the reef elements, and that this may increase to longer term benefits at the mid and upper elevations of the reef elements over time as sea levels rise.

Using both sonic and image-based wave monitoring equipment, data suggest that this nearshore reef had the largest effect on waves at mid-tide elevations of 3.6-5.5’ NAVD88. Waves smaller than 3.6’ NAVD88 tended to dissipate over the mudflat, and there was little attenuation of waves larger than 5.5’ NAVD88. However, the nearshore oyster treatment’s effects on wave dissipation appeared to have a beneficial effect on mudflat sedimentation rates in its lee, which could further attenuate wave energy at the marsh edge and thus potentially reduce shoreline erosion. Mudflat elevation change is episodic and can be inconclusive at five-year timescales. Data were collected through the five-year monitoring period to produce digital elevation maps and to estimate erosion over recent years; additional funding will be needed to repeat this effort if we are to determine whether the measured reduction in wave energy at mid-tide levels translates to a reduction in the rate of shoreline erosion.

We were also interested in how well native oysters might recruit on this nearshore and relatively shallow reef. Not surprisingly given the intentionally high experimental elevation of Treatment 3, which is about midway between that of the middle and high rows in Treatment 2, we found few live oysters in

2023. We estimate that perhaps 588 oysters (SE ± 302) were present across all of Treatment 3, about 25% of what we estimated in the high oyster abundance year of 2022. In 2022 and in 2020, most oysters on the Treatment 3 nearshore reef were found on the bottom tier where the density was more than an order of magnitude higher than on the center tier. No oysters were recorded from the top tier. There were also far more oysters on the north faces compared with those on the south. Consistent with this pattern, the oysters recorded in 2023 were on the bottom tier and north sides of blocks.

Although not quantified with our survey methods, we noted that water was retained in the lower tier of some oyster blocks, which created tide pool-like habitat in which oysters settled. In some cases, oysters settled in small clusters of two and three individuals unattached to the substrate. And despite difficulty in quantifying oyster recruitment on the undersides of oyster block surfaces, we qualitatively noted that these surfaces were favorable for oysters, likely due to protection from heat and desiccation stress and possibly reduced predation.

Using a remote camera, we saw a significant increase in waterbird abundance during low tide at the Treatment 3 Nearshore Reef. Dabbling ducks were most abundant when the reef was partially exposed, whereas diving ducks were most abundant when the reef was nearly inundated. Large and small shorebirds were both most abundant when the tideline ebbed or flowed across the reef, with large shorebirds displaying species-specific differences in use of the reef area, particularly during winter.

Pacific Cordgrass Inshore from Nearshore Reef (Treatment 4)

Pacific cordgrass (*Spartina foliosa*) was planted in January 2022 in conjunction with the Treatment 3 nearshore reef to evaluate whether wave attenuation expected from the reef could improve cordgrass establishment relative to plantings outside the reef's influence just to the northeast. In three blocks at each of the protected or more exposed locations, three cordgrass planting methods were tested: bare-root plugs (~5-7 stems with rhizomes) planted in clusters, 0.1-m^2 sods placed in groups of four into shallow trenches, and bare-root plugs planted into burlap bags half-filled with nursery soil and placed into shallow trenches.

Sods were far more successful at establishing cordgrass than the other methods. As of the final monitoring in November 2024, sods had survived in all planted blocks, while all burlap bags had failed and bare root plugs had survived in only one of the blocks (one behind the reef). Minor expansion via rhizomes beyond the original sod footprint was observed in all surviving planted blocks. In all but one of the blocks the area covered by cordgrass originating from sods had more than doubled, with a mean 128% increase in area covered across all six blocks.

Counter to expectations, the Treatment 3 nearshore reef did not appear to influence cordgrass establishment in Treatment 4. This may have been in part because the mudflat area northeast of the reef experienced considerable sediment accretion since the project inception in 2019, presumably as a result of the reef's presence and to the benefit of establishing cordgrass. This area was also nearer to existing cordgrass, which accreted a substantial amount of sediment during the project period and may have provided some additional protection to Treatment 4 plantings outside of the expected protection of the nearshore reef.

Pacific Cordgrass Adjacent to Existing Pacific Cordgrass (Treatment 5)

For Treatment 5, Pacific cordgrass was planted adjacent to existing cordgrass to test if this could help to facilitate expansion of existing patches into unvegetated areas. In January 2022, cordgrass was planted in bands adjacent to each of three areas of existing cordgrass, with each band running roughly perpendicular to the shore. Within each of the three bands, two planting methods were compared: sods and bare-root plugs, as in Treatment 4.

Plugs survived in only one band (the northern-most), with 2 out of 24 plots (8%) of plugs surviving as of November 2024. Sod survivorship was much better than plugs (as with Treatment 4) but still only low to moderate, ranging from 11-54% across the three bands. We cannot say definitively that the planted bands aided in spread of existing cordgrass into unvegetated areas; however, in two of the three bands existing and planted cordgrass coalesced, filling in ~10m^2 such that they became indistinguishable. This filling in along the existing cordgrass as well as lateral spread of ~35m^2 in those two planted bands indicate that the sod plantings enhanced the cordgrass at the site overall.

<u>Marsh Scarp Revegetation and Arboring for Sea-blite and Pickleweed (Treatment 6)</u>
This treatment was intended to test whether plantings of the endangered California sea-blite (*Suaeda californica*) can be enhanced with additions of high tide wrack in planting holes as well as whether arboring with *Eucalyptus* branches can encourage greater height and size of sea-blite as well as existing pickleweed (*Salicornia pacifica*). Larger or taller plants along the eroding bay front could create refuge for birds and mammals during extreme high tides and storm flooding.

Storms led to sediment movement and repeated planting was necessary to maintain sea-blite at the site. However, monitoring of surviving plants showed a strong positive effect on plant size from both adding algal wrack deposits to planting holes and arboring with *Eucalyptus* branches inserted into the sediment. Later plantings suggested that protection from fringing marsh was beneficial to sea-blite persistence.

In addition, arboring of existing pickleweed (*Salicornia pacifica*) in spring 2019 led to observably taller plants as early as September, a trend that became statistically significant by January 2020 and continued through the fifth project year.

<u>Marsh-Upland Transition Zone Enhancement (Treatment 7)</u>
This treatment included planting multiple native species from the high marsh to upland transition to increase species diversity and habitat complexity that could be used by wildlife as refuge cover from predators. These native perennials were also intended to provide flower nectar and seed sources for foraging wildlife during more of the year than the non-native annual grasses present in the upper transition zone, which die off in late spring. The planting design included 1000 native plants of five species including marsh Baccharis (*Baccharis glutinosa*), western goldenrod (*Euthamia occidentalis*), creeping wildrye (*Elymus triticoides*), western ragweed (*Ambrosia psilostachya*), and Pacific aster (*Symphyotrichum chilense*). These species were planted in groups across the transition zone at the high marsh edge (9-11' NAVD88), steep slope/scarp (9-12'), and upland meadow (14-20').

The planting area was changed from the original location due to challenges with existing high densities of non-natives, within a mix of native species including purple needlegrass and others. There were also regulatory challenges with permitting the experimental site preparation that included a small amount of mechanical week whacking to remove invasive weeds prior to planting, which would have then required conservation measures such as exclusion fencing for salt marsh harvest mice. The exclusion fencing is installed within a 1.5-m (5 feet) deep trench, which would have been far more damaging to excavate and install than the weed whacking impacts. A new site was chosen to the northwest, which had similar issues with non-natives and ruderal impacted existing conditions with compacted soils and eroding shoreline slopes. We changed course and received permission to opportunistically and experimentally plant into these fairly common but degraded existing conditions with high marsh, eroding shoreline scarp, and an adjacent meadow area.

Survivorship varied by zone and species. High marsh edge plots had no surviving western goldenrod but supported relatively high survivorship of marsh Baccharis (49%) and creeping wildrye (42.5%). Steep slope/scarp creeping wildrye survivorship declined from 37% in December 2023 to 0% by October 2024. The upland meadow had high survivorship for all three species planted (western ragweed = 100%, Pacific aster, = 99%, and creeping wildrye = 77.5%). Unlike the upland meadow plots, high marsh edge and steep slope/scarp plots were not mulched or fenced and the lack of protection from weed competition (Fuller's teasel in/near the high marsh edge and steep slope/scarp, and perennial pepperweed at the high marsh edge) and trampling may have contributed to the lower survivorship observed. Overall, the creeping wildrye planted in steep slope/scarp plots likely experienced the worst growing conditions with disturbed, eroding soils, heavy weed competition, and close proximity to an unofficial trail (and no fencing/signage).

Our **major findings** from the Giant Marsh Living Shorelines Project are:

- In Treatment 1, offshore reef balls topped with oyster shell bags were successful at recruiting abundant native Olympia oysters, although low recruitment in the first year, 2019, and the final year, 2023, point to the importance of interannual variation in salinity and temperature on oyster densities. This variability in oyster abundance suggests that installing reefs in multiple places in the bay may provide insurance against spatial variability in conditions, and that natural recruitment might be supplemented in future projects with laboratory-reared propagules.
- Numbers of oysters rose steadily from 2020 to 2022, reaching ~1.3 million individuals in Treatment 1. Even in years of fewer oysters, the project greatly increased the population of Olympia oysters, which we expect benefitted recruitment to other rocky substrate in the region.
- The average size of oysters was similar across years, ~22 mm, with both new recruits and oysters in larger size classes (41-50 mm or larger) present on the Treatment 1 reefs, indicating oysters were thriving and growing as well as providing propagules to newly colonize.
- Providing suitable substrate quickly led to valuable habitat aside from Olympia oysters, as the offshore reefs supported 28 mobile and sessile taxa within two years.
- Planting eelgrass in the cooler spring months is advisable but does not always avoid heat waves that come sporadically at other times, often coinciding with low tides (early June 2019).
- High-density eelgrass plantings produced more shoots but not proportionately more shoots, indicating that the benefits of planting densely may not balance favorably against the increased effort and resources needed to do so.
- Planting eelgrass adjacent to oyster reefs is beneficial, as there was evidence of greater eelgrass abundance on the shore side of oyster reefs and greater numbers of mobile invertebrates in eelgrass when adjacent to reefs.
- Nearer to shore (Treatment 2), the highest oyster densities (and largest oysters) were found on the deepest oyster blocks (-0.10' NAVD88) and on the deepest part of those blocks, probably due to less exposure to high air temperatures during low tides.
- Oyster numbers were higher on the north sides of the oyster blocks in most years, suggesting benefits of cooler temperatures.
- Transplanting the brown alga *Fucus distichus* to the oyster blocks resulted in low survival, but proved favorable to both Olympia oysters and barnacles while the alga was present, likely due to increased relative humidity and cooler temperatures under the canopy.

- At the Treatment 3 nearshore reef at 1.5’ NAVD88, native oysters recruited to the lower portion of oyster blocks at numbers intermediate to the Treatment 2 high (1.9’) and mid-elevation (0.93’) blocks, as expected.
- The Treatment 3 nearshore reef was most effective at reducing wave energy at mid-tide elevations of 3.6-5.5’ NAVD88.
- There was no overall change in waterbird abundance with the offshore reef/eelgrass treatments, despite early evidence (2020) of increased abundance with these treatments.
- The shallower reefs (Treatments 2 and 3) showed increased waterbird use at low tides, and dabbling ducks, diving ducks, and both large and small shorebirds frequented the nearshore reef (Treatment 3).
- Waterbird species and guild use of the Treatment 3 nearshore reef varied by tide height and season, highlighting the importance of considering these factors when designing and evaluating avian response to living shoreline enhancements.
- Planting cordgrass using sods worked best to establish this native species, including along existing areas of cordgrass that may be encouraged to spread and coalesce with the plantings.
- Storm-driven dynamic sediment movement led to repeated losses of planted CA sea-blite; however, additional plantings in more protected locations (behind fringing pickleweed marsh) showed that survival is possible at this site, and self-seeding may reinforce population persistence.
- Surviving sea-blite plants grew substantially larger where wrack was added to planting holes, presumably due to moisture retention and nutrient availability as the wrack decomposed.
- Arboring of both planted sea-blite and existing pickleweed along eroding marsh edges produced larger plants, suggesting this simple method can be used to increase habitat value for birds and mammals during extreme high tides and storm-generated flooding.
- Transition zone plantings should be pursued from the high marsh edge to upland meadow zones to enhance wildlife refuge and foraging, but weed control will be needed, and fencing and signage should be added near locations where people and dogs are common.
- Steep scarp areas may require sediment augmentation (e.g., coarse beach addition to support wave-built berms) or other erosion control interventions to slow erosion, as indicated by poor performance of planted creeping wildrye along scarps in Treatment 7 and loss of two naturally-occurring pickleweed plots in Treatment 6.
- Across a wide array of treatments, we did not find that implementation of the Giant Marsh Living Shorelines Project increased non-native species beyond background levels or nearby reference sites.

We met our overarching objective to conduct scientific experimentation on best techniques for living shoreline approaches from the shallow subtidal to the estuarine-terrestrial transition zone.

We also met all the specific project objectives, to:
1) Create or enhance a variety of habitats ranging from the shallow subtidal to the tidal marsh to the estuarine-terrestrial transition zone,
2) Experimentally evaluate techniques to advance restoration practice for each of these habitat types,
3) Assist in recovery of particular species of concern, including Pacific cordgrass, eelgrass, Olympia oysters, and the endangered California sea-blite,
4) Evaluate the use of restored habitats for wildlife, including invertebrates, fish, and birds, and
5) Evaluate the efficacy of nearshore reefs in attenuating wave energy and reducing shoreline erosion.

In addition, the project met the overall goal of one or more of its specific success criteria being met over the five-year period following construction. In fact, the project met all the specific criteria in one or more years (See Section 3 below), with the exception of fish visitation to offshore reefs and eelgrass increasing by 50% relative to control areas; despite utilizing multiple methods and documenting substantial fish abundance and diversity, we continue (as in the San Rafael living shorelines project) to find the complex three-dimensional structure of living shorelines reefs and eelgrass very difficult to monitor for fish usage.

A number of **key recommendations** resulted from this project: 1) Considering the variation in abundances of important species such as the Olympia oyster over the years of the project, restoration of living shorelines should be conducted over a range of sites to buffer across conditions that may become detrimental at any one site in a particular year. Within-site heterogeneity in restoration design can also help practitioners to diversity and maximize components that can help to provide insurance against physical and biological stressors. For example, we sited reefs at a range of elevations across the site, and planted CA sea-blite in a variety of locations that differed in degree of protection from wave energy and erosion by fringing marshes. 2) Restoration projects and their monitoring should be conducted with an eye to understanding the source populations that are generating propagules, as well as adult populations, and connectivity between sites; such understanding can put restoration results in context of the surrounding setting and the siting and timing of restorations may be maximized to increase dispersal and connectivity benefits between sites. 3) We recommend supporting efforts to assess both the short- and long-term survivability of these types of restored habitats in the context of variation in environmental conditions and especially extreme events. Project goals and timeframes should be flexible enough to account for temporary fluctuations in responses and restoration projects should not be expected to perform better than existing conditions in relation to these dynamic and stochastic stressors. 4) We recommend continuing to test ways to maximize facilitative interactions between species and co-deployed restoration methods in future projects, as we saw such benefits with seaweeds on oysters, and with reefs on eelgrass establishment and spread. 5) Five years of robust and frequent post construction monitoring as part of this voluntary, experimental restoration project were just sufficient to document patterns and trends. We recommend longer-term assessments of this and other restoration projects where feasible.

Please see the full summary report below for a more complete discussion and information, including the closing sections with summary information and recommendations for future living shorelines efforts:

- **Findings Relative to Objectives and Success Criteria**
- **Project Considerations in Ephemeral Habitats with Ongoing Climate Impacts**
- **Enhancement of Native Species and No Increase in Non-native Species Habitat**
- **Key Recommendations for Future Living Shorelines Projects**
- **Capacity-Building Challenges - Pace, Scale, and Participation**
- **Next Steps to Regionally Advance Living Shorelines in San Francisco Bay**
- **Thank You to Our Funders and Partners**

This project could not have occurred without the tenacious and forward-thinking hard work by staff and organizations involved in this first-of-its-kind work in San Francisco Bay. We are grateful for the collaboration, contributions, and support by all partners, funders, landowners, practitioners, and others involved.

1. Introduction

The Giant Marsh Living Shorelines Project is a multi-institution collaboration initiated and managed by the State Coastal Conservancy with a wide range of engineers, ecologists, land-owning partners, and local, regional, and national marine restoration and construction firms, to advance and demonstrate living shorelines techniques: habitat restoration methodologies that also enhance shoreline protection. Located in San Francisco Bay within the subembayment San Pablo Bay along the north Richmond Shoreline (Fig. 1), this project builds on lessons from the first living shorelines project in the region, also led by the State Coastal Conservancy and many of the same partners, installed along the San Rafael shoreline in 2012.

The Giant Marsh Living Shorelines Project is intended to increase the scale of implementation and the range of shoreline enhancements employed, including the shallow subtidal with the addition of oyster/habitat reefs (intended to support the native oyster, *Ostrea lurida* as well as other species) and eelgrass (*Zostera marina*); the tidal marsh edge with Pacific cordgrass (*Spartina foliosa*) plantings and additional reefs installed at a range of depths; wave-built sand berms where the endangered plant California sea-blite (*Suaeda californica*) can be restored and experimentally enhanced for wildlife high tide refuge (with similar treatments to increase height growth of existing pickleweed [*Salicornia pacifica*]); and terrestrial transition zone management to support greater native plant species diversity and habitat function. By enhancing endangered species habitat, piloting climate adaptation techniques, and integrating habitat connectivity and functions, this project helps directly implement regional habitat recommendations in the San Francisco Bay Subtidal Habitat Goals Report (2010), Tidal Marsh Recovery Plan (USFWS 2013), Baylands Habitat Goals Science Update (2015), and the San Francisco Estuary Blueprint (2022).

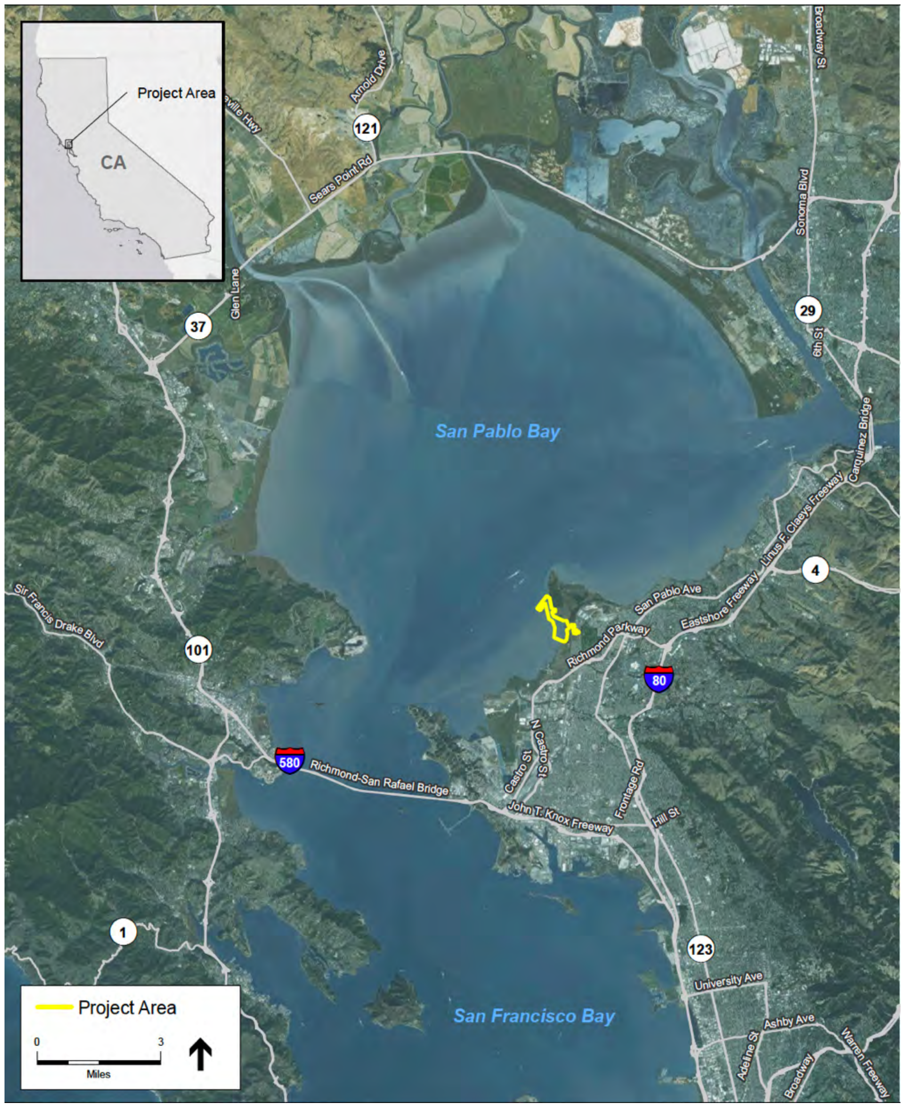


Figure 1. Map showing location of Living Shorelines Project at Giant Marsh (ESA, SCC 2017).

1.1 Goals and Objectives

The overall goal of the Giant Marsh Living Shorelines Project was to restore ecological function and ecosystem resilience through the creation and enhancement of a range of biologically rich shoreline habitats, from the subtidal to the estuarine-terrestrial transition zone. Objectives to accomplish this primary goal included the following:

1. Create or enhance a variety of habitats ranging from the shallow subtidal to the tidal marsh to the estuarine-terrestrial transition zone.
2. Experimentally evaluate techniques to advance restoration practice for each of these habitat types.
3. Assist recovery of particular species of concern, including Pacific cordgrass, eelgrass, Olympia oysters, and endangered species such as California sea-blite, California Ridgway's rail, and salt marsh harvest mouse.
4. Evaluate the use of restored habitats for wildlife, including invertebrates, fish, and birds.
5. Evaluate the efficacy of nearshore restoration treatments in attenuating wave energy and reducing shoreline erosion.

Expected benefits of the project included:

1. Vegetative, seed and rhizome material and oyster larvae will travel over the restoration acreage and continue to expand habitat and self-propagate each year.
2. Plantings and oyster reef installation will re-introduce specific, critical, functional components (nesting, breeding, foraging, and high tide refugia habitat) that contribute to a fully functioning, complete estuarine ecosystem over the full enhancement area.
3. Plantings will increase food resources that benefit multiple species, by increasing seed and detrital resources that spread over the full tidal area and the estuarine-terrestrial transition zones of the project.
4. Synergistic effects of oyster reef structures will help reduce wave energy and accrete sediment, making revegetation of native cordgrass and eelgrass more successful. Reefs will also support numerous small invertebrates and fish that provide resources for birds and larger fish, and installing oyster reefs adjacent to eelgrass is expected to enhance species richness.
5. Providing a variety of habitat types in close proximity will provide food and habitat resources that benefit more species than could be accomplished by restoring any single habitat in isolation.
6. Other anticipated indirect effects of the project included measurably different wave energy and sedimentation patterns in the immediate vicinity of the constructed reefs, which could benefit adjacent eroding shorelines and help to accrete sediment and create more ecologically complex physical structure on the mudflats.

1.2 Site Selection

Seven candidate sites in San Francisco Bay were assessed in 2015 - 2016 before selecting the Giant Marsh site (a site managed by East Bay Regional Park District (EBRPD) and part of the Point Pinole Regional Shoreline) for the next phase of living shorelines projects in the bay. The assessment included a survey of existing populations of oysters on hard substrates as well as monitoring of recruitment and survival of oysters on standardized substrates (porcelain tiles) for one year. In addition, the presence of the Atlantic oyster drill *(Urosalpinx cinerea),* a major predator on oysters, was monitored. The project site at Giant Marsh ranked highly among the seven candidate sites, as it had high densities of oysters on existing hard substrate, high recruitment and survival of oysters to the test substrates, and Atlantic oyster drills were not present at the site. Further, previous work over the past 10+ years indicated that the north Richmond shoreline at Giant Marsh is a favorable location for native oysters (Grosholz et al. 2008, SF Bay Subtidal Habitat Goals Project 2010 and Appendix 7-1, Wasson et al. 2015).

The Giant Marsh shoreline also ranked highly for eelgrass habitat suitability in 2015-16. Test plots of transplants thrived over the year after planting, a good indication that plantings at a larger scale would succeed. Further, surveying and mapping by Merkel & Associates (2014) indicated extensive existing eelgrass beds offshore of Giant Marsh, with coverage ranging from 2,900 acres to 3,700 acres (1200 to 1500 hectares) between 2003 and 2014. Ample acreage was present in the project area within the elevation suitable for eelgrass; i.e., at least as deep as -1.4’ NAVD88. Test plantings of native cordgrass and marsh gumplant also performed well. All of these factors contributed toward our conclusion that the Giant Marsh site would be favorable for eelgrass, tidal marsh, and transition zone plantings, and in combination with promise of oyster recruitment, prompted our interest in a living shoreline at this site.

We also sought a site that would permit tidal marsh enhancement for resiliency of habitat with sea level rise. Observations and surveys of the Giant Marsh site indicated that it contained suitable elevations, substrate, shoreline orientation to wind/wave energy, and hydrology to support growth and persistence of a variety of marsh and estuarine-terrestrial transition zone plant species. Incorporation of native cordgrass plantings could dovetail with the ongoing Invasive *Spartina* Project efforts to revegetate areas where invasive *Spartina* was being removed (led by State Coastal Conservancy and federal lead US Fish and Wildlife Service, with East Bay Regional Park District and other baywide landowners). In addition, CA sea-blite was historically present in the region and the Point Pinole Regional Shoreline was identified in the plant’s reintroduction plan as a site with high potential to support the plant (Baye 2006). Perennial pickleweed is abundant at the site but is undergoing considerable erosion along portions of the marsh edge (up to 3.5 feet [1.1 m] per year, more than 130 feet [40m] of erosion since mid 1900s); promoting this plant’s presence is desirable for maintaining the marsh shoreline as well as the habitat it provides for numerous birds and mammals including the endangered salt marsh harvest mouse. Further, habitat restoration within the Point Pinole marsh complex is a Priority 1 action to recover Ridgway's rail in the Recovery Plan for Tidal Marsh Ecosystems of Northern and Central California (USFWS 2013).

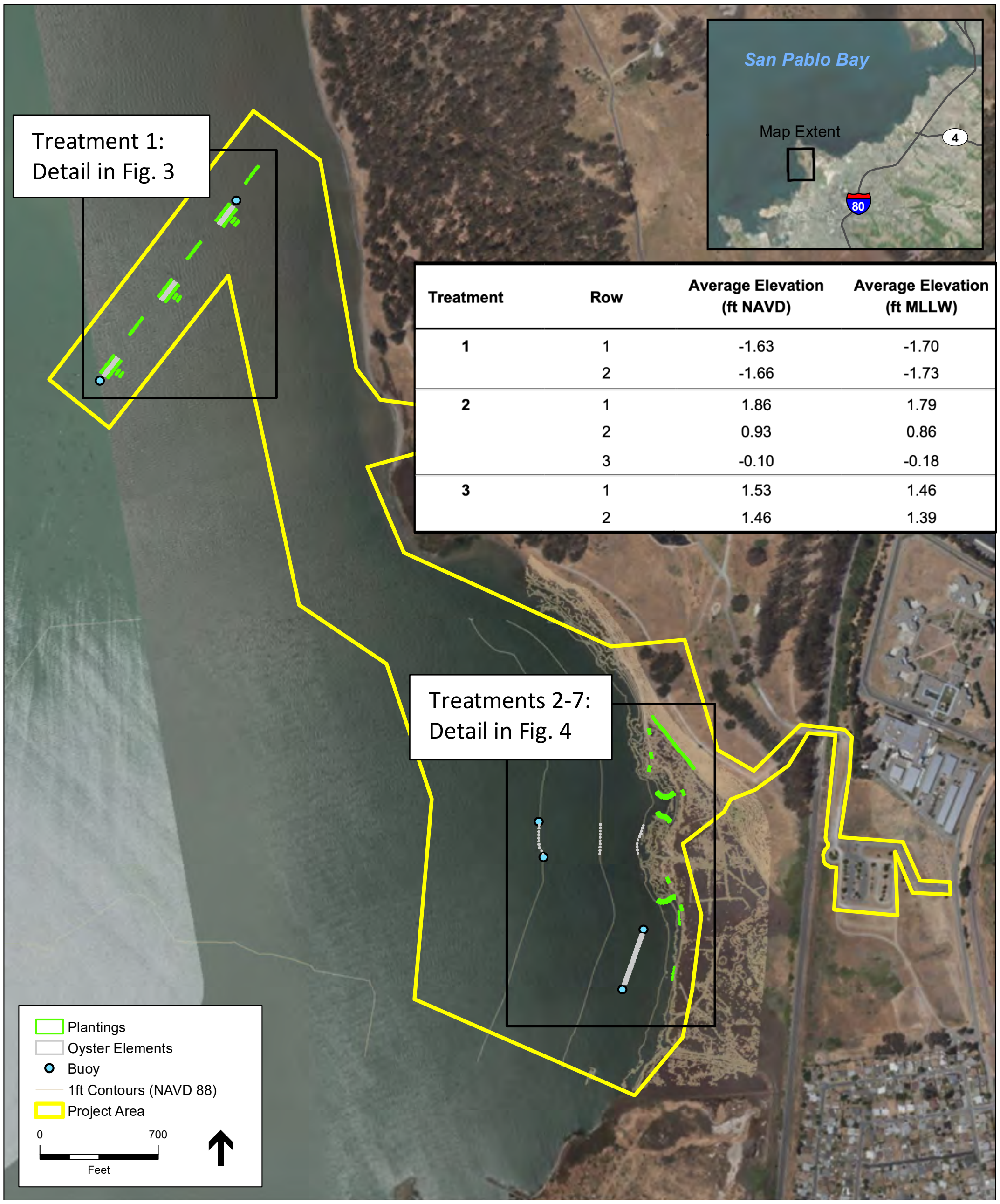
Finally, we hoped to identify a site meeting our biological criteria that could also promote shoreline protection or other benefits to a nearby disadvantaged community. We considered Giant Marsh and the Point Pinole Regional Shoreline favorable in this regard as the nearby Parchester Village Housing Community, an underserved community in north Richmond, has historically used the Giant Marsh and former Breuner Marsh (now restored Dotson Family Marsh) shorelines for recreation. A demonstration project of restored and protected marshland and resilient shorelines might be scaled up in the future to buffer this community against impacts associated with sea level rise and provide improved access to the bay for recreation, open space, and community gathering.

1.3 Project Location and Design

After finding Giant Marsh to be the top candidate site to meet our multiple objectives, the project team defined seven treatments to include in the project:

- Treatment 1: Offshore eelgrass and oyster/habitat reef
- Treatment 2: Oyster/habitat reefs at a range of elevations (with test of rockweed shading)
- Treatment 3: Nearshore oyster/habitat reef near the marsh edge
- Treatment 4: Pacific cordgrass revegetation inshore of nearshore reef
- Treatment 5: Pacific cordgrass revegetation adjacent to existing Pacific cordgrass
- Treatment 6: Marsh scarp revegetation and arboring of CA sea-blite and pickleweed
- Treatment 7: Marsh-upland transition zone enhancement

Figure 2 shows the positioning of these treatments within the project footprint, and Figures 3 and 4 provide a closer view of the treatments. The treatments are described in more detail in the project description (ESA, SCC 2017), and summarized briefly below as we highlight the project results.



Treatment	Row	Average Elevation (ft NAVD)	Average Elevation (ft MLLW)
1	1	-1.63	-1.70
	2	-1.66	-1.73
2	1	1.86	1.79
	2	0.93	0.86
	3	-0.10	-0.18
3	1	1.53	1.46
	2	1.46	1.39

Figure 2. Layout of the treatments, with inset table of elevations of the oyster/habitat reef treatments. Figures 3 and 4 show detail of the inset boxes. Adapted from ESA, SCC (2017).

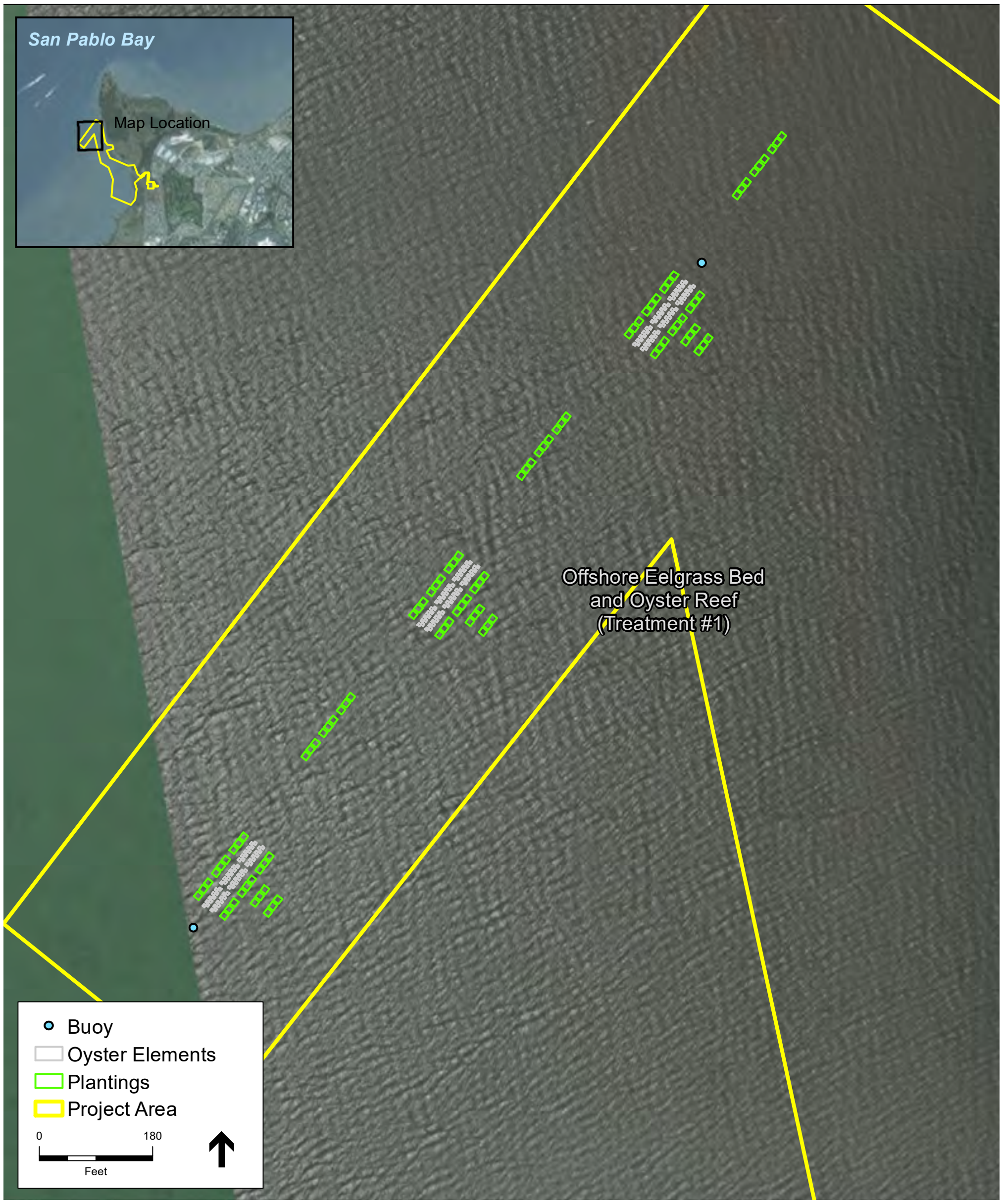


Figure 3. Treatment 1 detail showing three arrays of oyster/habitat reefs, with eelgrass planting on the bayward and shoreward side of each (the latter at three distances from the reef), and three arrays of eelgrass plantings alone (ESA, SCC 2017). See elevations in inset of Fig. 2.

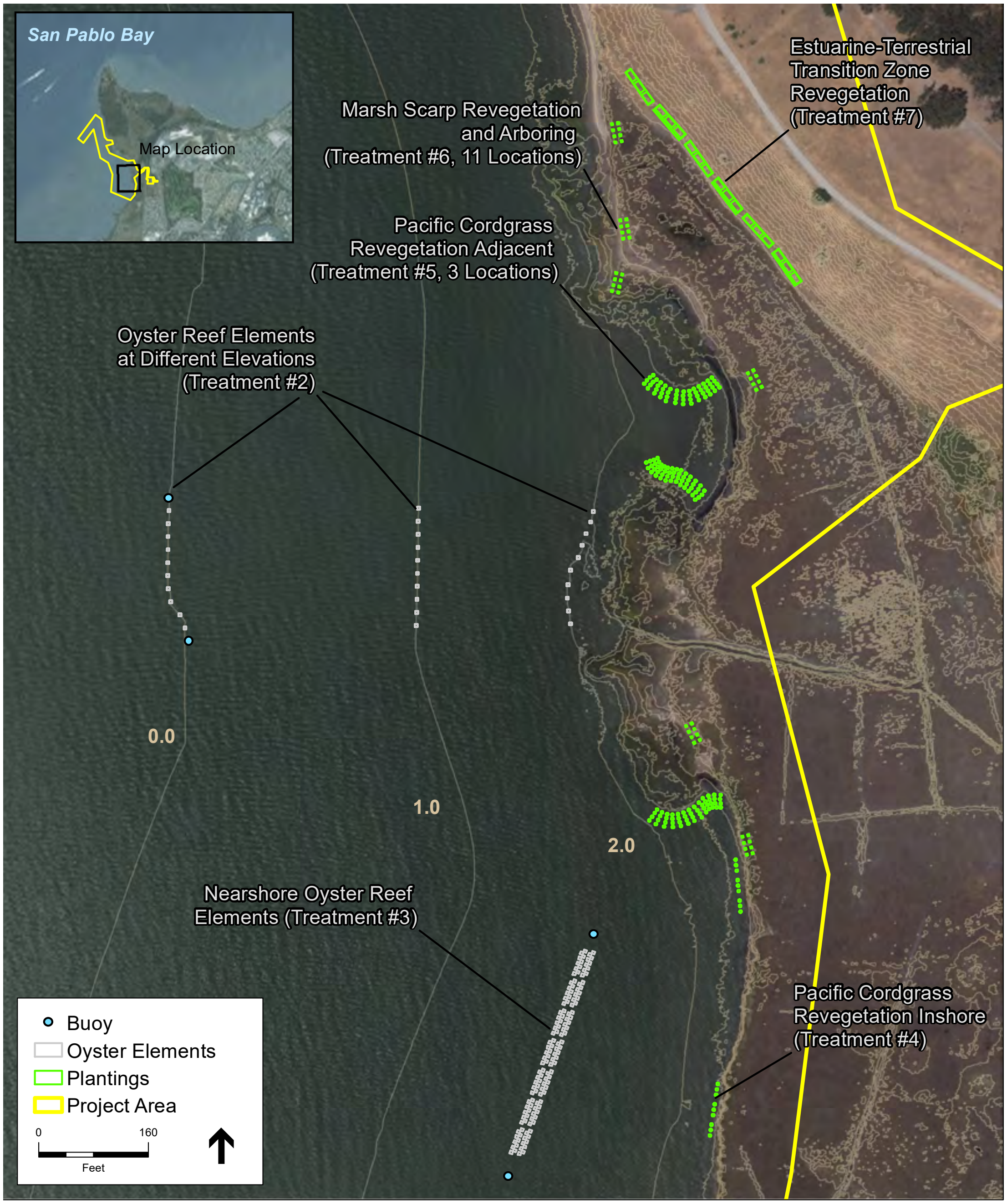


Figure 4. Detail of positioning of Treatments 2 through 7 as originally planned (ESA, SCC 2017). See inset of Fig. 2 for elevations of reef elements. Note that the actual locations of Treatment 6 varied somewhat from this plan (see Fig. 21) and Treatment 7 was ultimately placed northwest of the planned location by about 600 feet (180 m; see Fig. 26).

1.4 Success Criteria

While a primary objective of this project is to conduct scientific experimentation on best techniques for living shoreline approaches through pilot-scale oyster, eelgrass, marsh, and estuarine-terrestrial transition zone treatments, it is anticipated that the project will provide lasting habitat for numerous organisms in the deep intertidal through the estuarine transition zone. Considering natural inter-annual variation of the desired biological and physical responses, we developed the following success criteria, with the project to be deemed successful if one or more of the following criteria were met within the 5-year period following construction:

- Native oysters will recruit, with densities greater than 10 adult oysters per square meter of substrate at the offshore oyster reef.
- Invertebrate species richness will increase by 15% relative to control plots with no physical structure and initial measures collected prior to construction.
- The number of visits by fish species to the offshore eelgrass and oyster reefs will increase by 50%, relative to control areas with no physical structure.
- Eelgrass will establish and spread to at least twice initial planting densities.
- Shoreline erosion immediately landward of the nearshore reef will not exceed erosion rates at adjacent unsheltered areas.
- Offshore reefs provide wave attenuation in landward eelgrass planting areas for mean sea level tide conditions, compared to control areas with only eelgrass plantings or mudflats.
- Pacific cordgrass plantings will establish, spread to at least twice initial planting densities, and expand into previously unvegetated areas.
- California sea-blite will establish in at least three planting areas, thus adding Giant Marsh as a new reintroduction site for San Francisco Bay in the recovery of this species.

At the end of this report, we return to these success criteria and the extent to which they were met.

2. Results

Full write-ups of each of the Giant Marsh Living Shorelines Project teams' work can be found in their individual final reports (Zabin and Blumenthal 2024, Buchbinder et al. 2024, 2025a, Hammond 2025, Graham et al. 2024, ESA 2024). Here we have summarized key findings.

2.1 **Oysters and Associated Measurements** (Smithsonian Environmental Research Center and University of California Davis)

Three of the treatments at the Giant Marsh Living Shorelines Project involve Olympia oysters:

- Treatment 1 provides the largest amount of oyster restoration substrate for this project, three "reefs" comprising 60 reef balls each (made of 'baycrete'), topped with 2-3 mesh bags of clean Pacific oyster half shell (Fig. 5). This is the deepest of the three treatments (see Fig. 3 for all elevations); bases of the reef balls were placed at -1.6 to -1.7' NAVD88, with the shell bags at about 0' NAVD88. The objective of this treatment with regard to native oysters was to increase the existing population of oysters at Giant Marsh through the provision of hard substrate at a suitable tidal elevation. Based on our results from the living shorelines project at San Rafael, we anticipated that ~0' would be the ideal tidal elevation, and we targeted this height for the shell bags, which provide a greater amount of settlement space per unit area than reef balls. We

Figure 5. Left: An array of oyster/habitat reefs in Treatment 1, showing reef balls topped with bags of Pacific oyster shell. Eelgrass in the foreground was present prior to the project installation. Right: Monitoring with a quadrat (top) and acetate sheet (bottom).

anticipated that oysters would also settle on reef balls, but that over time these substrates would be less optimal as other organisms would also colonize them and compete with oysters for settlement space. As clean recycled half-shell material is in limited supply, our goal was to use it only at the best tidal elevation for oysters. We monitored this treatment through removal of 10-12 sample shell bags (four per reef array), and counting and measuring oysters and other sessile organisms on all shells. In addition, we used a 10 x 10 cm quadrat to count and measure numbers of oysters on the same reef balls where the shell bags were collected, on vertical surfaces on the north and south (one each) on the exteriors of each reef ball (Fig. 6). We also used a 10x10 cm acetate sheet marked at 25 point locations to estimate cover of oysters and other sessile organisms (Fig. 6). Finally, we monitored temperature using continuously logging sensors (see details in Zabin and Blumenthal 2024).

- Treatment 2 consists of three rows of 10 oyster block units each (14 blocks per unit), set at three tidal elevations: a low-elevation row at -0.10’ NAVD88, a mid-elevation row at 0.93’ and a high-elevation row at 1.9’ (Figs. 3, 4). Treatment 2 had three main objectives: 1) determine how elevation influences recruitment and establishment of native oysters and other species, 2) develop and refine methods for transplanting the native brown alga, Pacific rockweed (*Fucus distichus*, hereafter *Fucus*) to restoration projects, and 3) experimentally examine the impacts of *Fucus* presence on the development of the sessile community, particularly oysters. We randomly designated five oyster blocks per row as *Fucus*-added blocks (cobbles or driftwood with *Fucus* present, epoxied onto the blocks); the remaining five were designated as controls (Fig. 6). Each oyster block had three vertical levels, or tiers, and *Fucus* was added at the center and top tiers and on the north and south faces of the blocks (Fig. 6). The latter was informed by our earlier San Rafael Living Shorelines Project which showed an effect by substrate aspect (specifically whether a substrate faced north or south) on early development of the sessile communities (Zabin et al. 2014). We monitored oyster recruitment using the quadrats and acetate sheets described above, *Fucus* survivorship following several transplant attempts (see below), and temperature (see Zabin and Blumenthal 2024).

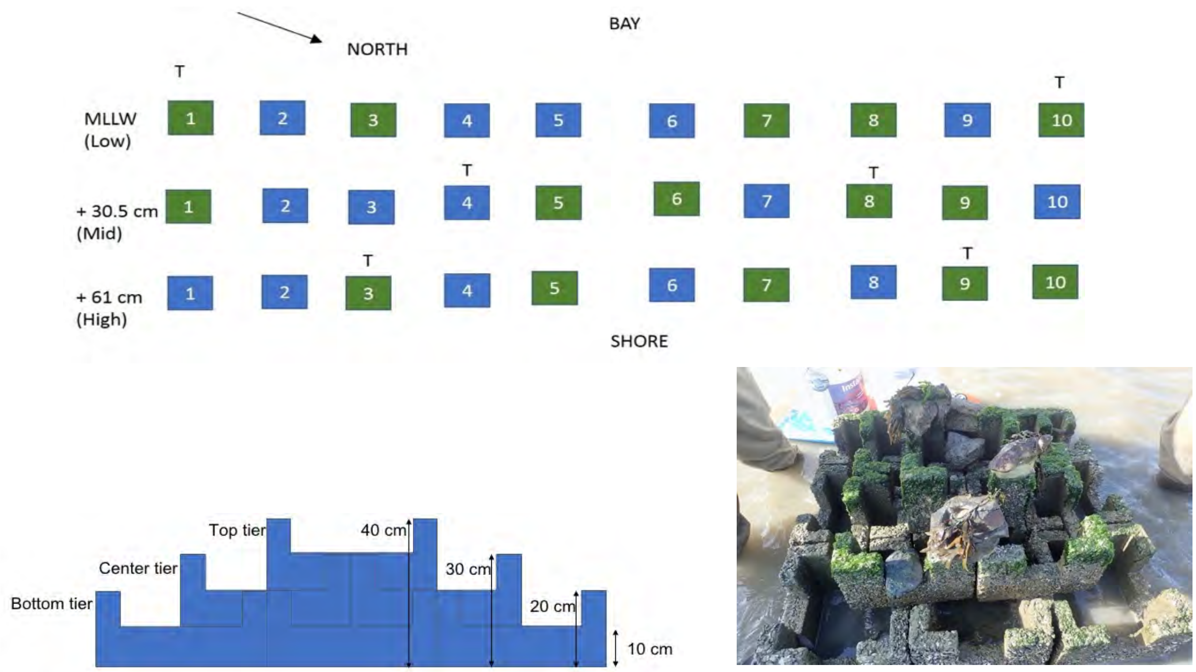


Figure 6. Top: Schematic of the experimental set up for Treatment 2. Green blocks indicate *Fucus* added; blue blocks are controls. “T” designates blocks with temperature loggers. Bottom left: Schematic of a set of oyster blocks, showing the three tiers and height of each tier off the benthos. Bottom right: *Fucus* on cobbles affixed to the top and center elevations of a set of oyster blocks.

- Treatment 3 consists of 140 oyster block elements made of “baycrete” (14 oyster blocks each) parallel to the shoreline, set in two staggered rows separated by a corridor, with oyster block bases at ~1.5’ NAVD88 (Fig. 3). The main objective for Treatment 3 was to test the ability of the nearshore oyster reefs to protect the marsh from erosion; the lower portions of the blocks were expected to serve as additional settlement space for oysters. We monitored Treatment 3 for oyster metrics beginning in 2020 using the same methods described above for the vertical faces of oyster blocks in Treatment 2.

- In addition, we monitored oyster recruitment to standardized recruitment tiles at Giant Marsh and at a nearby reference site, Pt. Orient, as well as populations of oysters, *Fucus,* and other sessile organisms on existing shoreline substrates at both sites (Fig. 7).

Below we briefly summarize the results of all the oyster-related monitoring through 2025. For a more detailed review of methods and findings, see the final monitoring report by Zabin and Blumenthal (2024).

Figure 7. Left: parallel transect lines at Pt. Orient in the *Fucus* (higher) and oyster zone (lower). Right: Recruitment collectors set at two tidal elevations at Giant Marsh.

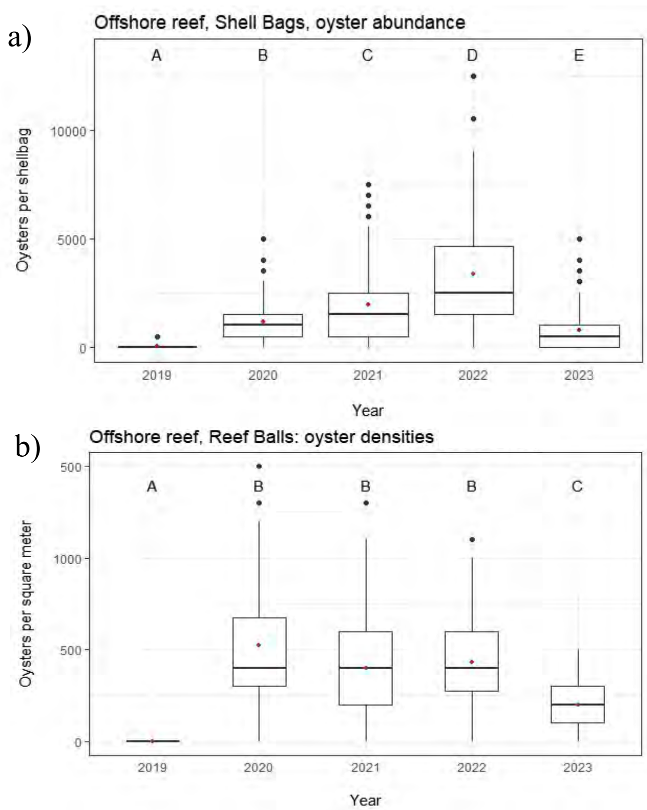


Figure 8. Results of Treatment 1 oyster surveys, 2019-2023. a) Estimated numbers of live oysters per shell bag and b) density of oysters on reef balls from quadrat surveys. Box edges represent the interquartile range of the data; horizontal lines are the median, black dots are outliers, and red dots are the average. Groups that share a letter are not statistically different.

Treatment 1. In August 2019, there were only six oyster recruits (1-3 mm length) from 120 shells in the 12 shell bags examined in the laboratory (Fig. 8). No oysters were found in the quadrat surveys (n=24) in the field. Both the shell bags and reef balls were notably absent of other organisms; surfaces were mostly bare, with a few barnacles, solitary tunicates, encrusting bryozoans and the green seaweed *Ulva* sp. The low number of oyster recruits to Treatment 1 shell bags was not completely unexpected. Spring air and water temperatures were cool in 2019, following a relatively wet winter (See Section 2.7); both factors can delay oyster reproduction in San Francisco Bay (Chang et al. 2016). Oyster recruitment did not begin until late August/early September at other North Bay sites in 2019 (unpublished data from A. Chang and A. Deck). In addition to less-than-optimal conditions in 2019, the historically heavy rains in winter 2017 decimated oyster populations in the North Bay in particular, and populations had still not recovered to pre-2017 levels (authors' unpublished data). There were also few settlers of other organisms, suggesting that recruitment may have been delayed for multiple taxa in 2019.

In Treatment 1, oyster abundance on shells increased from 2019 to 2022. In July 2020, oyster densities were substantially higher than in 2019 and included both larger adults, likely settled in 2019, and a smaller number of new recruits. There were 275 oysters total on the 120 shells randomly drawn from 10 sample shell bags and examined in the laboratory in 2020 (mean of ~two oysters per shell). This is equivalent to ~1,100 oysters per shell bag (Fig. 8a). Scaling up, these estimates suggested an oyster population of 427,000 oysters on the shell bags (Table 1). Numbers rose steadily from 2020 to 2022, reaching 7 oysters per shell in 2022, equivalent to ~1.26 million oysters in the shell bags alone. Oyster counts from the shell bags declined in 2023 to about 2 oysters per shell and 304,348 total oysters, similar to 2020 numbers and slightly less than 25% of the 2022 estimate (Table 1). The average size of oysters was similar across years, ~22 mm (Table 1), with both new recruits and oysters in larger size classes (41-50 mm or larger) present. In addition to oysters, there were over 30 taxa present in the shell bags, with notable additions of small Dungeness crab (*Metacarcinus magister*) as well as saddleback gunnels (*Pholis ornata*) in 2023. Rock crabs (*Romaleon antennarium*), present in other years, were not found in 2023, and tunicates, sponges and many mobile species (amphipods, isopods, polychaetes) were noted to have decreased as well. For the first time in 2023, we also noted several instances in which mussel byssal threads were holding together clumps of both live and dead oysters, creating a matrix of oysters and mussels.

Table 1. Comparison of means of various measurements of oyster performance in shell bags and on Reef Balls in Treatment 1, 2020-2023. Standard error in parentheses.

Substrate type	Year	Avg. oyster abundance	Avg. oyster cover (%)	Avg. oyster size, mm	Total number of oysters
Shell bag	2020	2 oysters/shell (± 0.2)	15.6 (±1 .3)	22 (± 0.5)	**427,361 (± 56,412)**
Shell bag	2021	4 oysters/shell (± 0.3)	24.7 (±2 .1)	23 (± 0.4)	**732,269 (± 94,926)**
Shell bag	2022	7 oysters/shell (± 0.42)	23.1 (± 1.4)	21 (± 0.4)	**1,264,211 (± 134,961)**
Shell bag	2023	2 oyster/shell (± 0.16)	11.8 (± 1.3)	22 (± 0.6)	**304,348 (± 31,227)**
Reef Ball	2020 (high quadrats)	520 oysters/m^2 (± 67)	14.5 (± 2.1)	20 (± 0.6)	**≥ 20,760 (top 10 cm)**
Reef Ball	2021 (high & low quadrats)	397 oysters/m^2 (± 33.6)	17.83 (± 1.5)	24 (± 0.5)	**32,670 oysters (± 2,998) (top 20 cm)**
Reef Ball	2022 (high & low quadrats)	429 oysters/m^2 (± 32.4)	17.7 (± 1.3)	26 (± 0.5)	**35,303 oysters (± 2,664) (top 20 cm)**
Reef ball	2023 (middle quadrats)	200 oysters/m^2 (± 24.5)	9.4 (± 1.5)	28 (± 1.0)	**16,408 oysters (± 2,051) (top 20 cm)**

On the reef balls, average oyster densities were similar in 2020, 2021, and 2022, with roughly 450 oysters per m^2 in the upper 10-20 cm band that could be accessed for monitoring during low tides (Fig. 8b); this was similar to densities at the same tidal elevation at the Point Orient reference site. It is difficult to scale this oyster density to full reef balls, and because the higher elevation of the balls was most likely to support native oysters, we conservatively estimated ~20,000 additional oysters were present on the reef balls in that upper 20 cm band in summer 2020, 33,000 in 2021, and 35,000 in 2022, with a drop to about half of that in 2023 (Table 1). Oyster sizes were somewhat higher on the reef balls than on the shells, reaching a high of 28 mm in 2023 (Table 1). In 2023, canopy cover, at 35%, was greatly reduced from 2022 on the reef balls, with a loss of sponges, upright bryozoans, and red seaweeds (see Zabin and Blumenthal 2024).

The extreme low salinity events of winter-spring 2023 (See Section 2.7), which were followed by high air temperatures, clearly took a toll on oysters and other organisms in Treatment 1. Intriguingly, losses of oysters on the Reef Balls were proportionally not as high as those on the shell bags. Taken with the results from Treatment 2 (see next section), this suggests that conditions slightly lower in the intertidal may have been more benign in 2023. We can imagine two scenarios under which this would be the case: 1) oysters higher in the intertidal zone may have had longer exposure than oysters deeper in the intertidal to any freshwater lens that may have developed on top of the salty Bay water, and/or 2) oysters higher in the intertidal were exposed for a longer time to warm air during low tides. This underscores the idea that providing substrates over a range of intertidal elevations provides some insurance against the various factors that can limit intertidal populations. It is also notable that even at a ~50% loss, oyster densities on Reef Balls were slightly higher than those at a similar elevation at our reference site at Point Orient in 2023. This indicates that the restoration project was not impacted to a greater extent than a natural shoreline. On Reef Balls and in shell bags, oysters in 2023 occurred across multiple size classes. While overall densities were lower, the size distribution was unchanged (Zabin and Blumenthal 2024), suggesting that the salinity stress impacted oysters of all sizes. That the average size was higher on Reef Balls than on shell bags in 2023, as it has been in past years, may indicate greater feeding opportunities lower in the intertidal zone.

Oysters clearly were not the only species impacted by the winter low salinity in 2023. Cover of all other organisms on the shell bags and Reef Balls was down significantly, with a subsequent increase in bare space. Despite this overall decline, the shell bags continued to provide habitat for small mobile taxa, including important native and fisheries species. In 2023, we recorded individuals of the *Metacarcinus magister* (Dungeness crab) and *Pholis ornata* (saddleback gunnel), neither of which had been present in shell bags previously. Surprisingly, the rock crab *Romaleon antennarium* and the porcelain crab *Petrolisthes cinctipes*, which were consistently found in shell bags in 2020-2022, were absent from bags in 2023, perhaps indicative of a shift in community composition due to the low salinity events.

The gradual increase of oysters to the shell bags in Treatment 1 from 2019-2022 contrasts with the results from the Living Shorelines Project at San Rafael, where millions of oysters had settled within the first year of the project (by 2013). Our experience with this earlier project led to expectations of rapid and high settlement of oysters to the Giant Marsh project. Indeed, pre-project surveys at Giant Marsh (2015-2016) revealed high oyster cover on every available bit of hard substrate in the middle to low intertidal zone, giving the impression that lack of suitable substrate was the major limitation to oyster populations in the vicinity of Giant Marsh. However, with 10+ years of data from multiple sites, it is now clear that 2012 and 2013, the first two years of the Living Shorelines Project at San Rafael, were years of very high oyster recruitment. Recruitment declined across sites monitored by the San Francisco Bay NERR and by the San Rafael Living Shorelines Project from 2014-2017 (Margulies 2023, Zabin et al. 2018). Recruitment rates increased in 2018 and have been rising over the past few years but are still low relative to 2012-2013 (Marguiles 2023). Numerous factors can affect oyster recruitment in the bay, including salinity and water temperatures (Chang et al. 2016). The extreme wet years in 2017 and 2019 dramatically reduced adult oyster populations with knock-on effects on recruitment in those years, and these lingering effects are also likely following the wet winter of 2023.

Treatment 2: Oyster responses. From 2019-2021 the densities of oysters increased across Treatment 2 (Fig. 9). In 2022, oysters declined somewhat, with losses mainly on the higher two intertidal rows. In 2023, following the wet winter, oyster densities decreased by an order of magnitude (Fig. 9, Table 2).

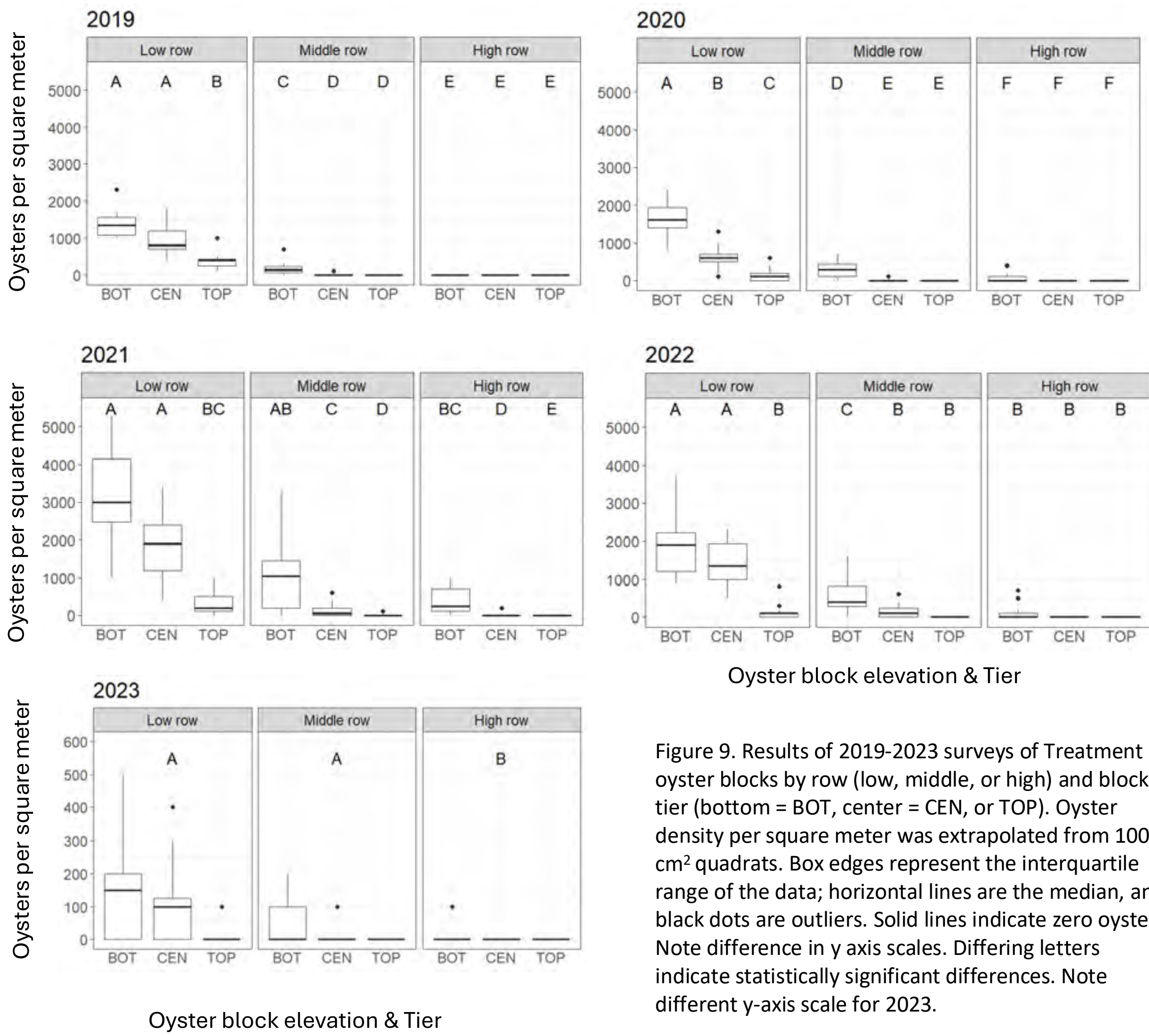


Figure 9. Results of 2019-2023 surveys of Treatment 2 oyster blocks by row (low, middle, or high) and block tier (bottom = BOT, center = CEN, or TOP). Oyster density per square meter was extrapolated from 100 cm^2 quadrats. Box edges represent the interquartile range of the data; horizontal lines are the median, and black dots are outliers. Solid lines indicate zero oysters. Note difference in y axis scales. Differing letters indicate statistically significant differences. Note different y-axis scale for 2023.

We estimated the total population on Treatment 2 in 2023 at 1,670 oysters (SE ±548), down from 21,560 (SE ±2,556) the previous year. Cover of oysters also decreased by an order of magnitude (Table 2). Losses occurred across all tidal elevations. As in previous years, quadrats on the lowest row had the most oysters (Fig. 9, Table 2). There were too few oysters in 2023 to determine whether there were differences at the level of tier within row, but differences between rows were statistically significant (GLMM pairwise contrasts: low-middle row, z = 3.48; p<0.01, middle-high row, z = 2.98, p<0.01; low-high row, z = 5.26, p<0.001). Differences in aspect, which had been significant early in the project (2021) but not afterward, were also not statistically significant in 2023 (Zabin and Blumenthal 2024). We note that despite reduced oyster densities in 2023, they were still higher on the restoration structures than on the reference natural shoreline or the old pier pilings at the same tidal elevations (For more on patterns in these reference locations, see Zabin and Blumenthal 2024). Further, we recorded a range of size classes, which suggests that the population that remains may be comprised of oysters that are better adapted to these stressors.

For the first four years of the project, there were higher densities of oysters on the lowest row of oyster blocks at Treatment 2 than on the Reef Balls in Treatment 1, which is lower in the intertidal zone. These differences were striking: In 2021-2022, there were three times more oysters on structures on the lowest row of Treatment 2 oyster blocks than on the Treatment 1 Reef Balls. In 2023, however, the oysters higher in the intertidal zone (Treatment 2) appeared to be impacted more strongly by the wet winter than oysters in very low intertidal (Treatment 1). This illustrates that there are multiple stressors than can impact oysters and that these vary by tidal elevation and year. Over the course of five years of monitoring, oysters did the best within the range of approximately +0.4 to -0.7' NAVD88 (+10 to -20cm MLLW), with higher elevations faring better in most years, but those slightly lower doing better in Year 5 (2023). Presumably, oysters lower in the intertidal zone face more competition for space given the higher cover of other species, especially soft-bodied organisms on which oysters cannot settle (see evaluation of other species present on the structures in Zabin and Blumenthal 2024). However, patterns in oyster densities by tidal elevation and aspect are probably driven largely by differences in exposure to high air temperatures during low tides; the highest row had the greatest number of time points above 35° C (a lethal temperature for oysters, Bible et al. 2017), and the lowest row had the least (Zabin and Blumenthal 2024). Higher intertidal oysters might also spend more time in low-salinity waters (due to the lens effect of low-density, low-salinity water) than oysters at lower elevations, depending on how well mixed and/or extensive the freshwater input is.

Table 2. Means and standard error (±) of various measurements of oyster performance on oyster blocks (vertical faces) by tidal elevation in Treatment 2, 2020-2023. Total number of oysters across all the rows shown in bold.

2020

Row	Avg. oyster abundance	Avg. oyster cover (%)	Avg. oyster size (mm)
Low	774 oysters/m^2 (± 95.9)	25.3 (± 3)	21.8 (± 0.3)
Mid	110 oysters/m^2 (± 25.4)	2.5 (± 0.7)	20.2 (± 2.1)
High	16.7 oysters/m^2 (± 7.9)	0.2 (± 0.1)	14.7 (± 1.6)
Total	**14,610 oysters (± 1,376)**	**8.7 (± 1.3)**	**21.4 (± 0.3)**

2021

Row	Avg. oyster abundance	Avg. oyster cover (%)	Avg. oyster size (mm)
Low	1213 oysters/m^2 (± 96.5)	39.6 (± 3.8)	24.2 (± 0.4)
Mid	228 oysters/m^2 (±45.0)	14.1 (±1.8)	15.4 (± 0.4)
High	61 oysters/m^2 (±15.6)	8.5 (± 1.1)	15.8 (± 0.1)
Total	**36,270 oysters (± 4,548)**	**15.0 (± 1.9)**	**21.2 (± 0.3)**

2022

Row	Avg. oyster abundance	Avg. oyster cover (%)	Avg. oyster size (mm)
Low	1128 oysters/m^2 (±118.2)	26.7 (±3.3)	23.9 (± 0.4)
Mid	233 oysters/m^2 (±47.0)	5.0 (±1.3)	16.5 (± 0.5)
High	33 oysters/m^2 (±15.0)	0.7 (±0.3)	13.7 (± 1.0)
Total	**21,560 oysters (±2,556)**	**11.0 (±1.5)**	**21.8 (± 0.4)**

2023

Row	Avg. oyster abundance	Avg. oyster cover (%)	Avg. oyster size (mm)
Low	87 oysters/m^2 (±16.2)	1.2 (±0.39)	28 (± 1.0)
Mid	17 oysters/m^2 (±5.9)	0.2 (±0.15)	15 (± 1.6)
High	3 oysters/m^2 (±2.3)	0	13 (± NA)
Total	**1,670 oysters (±458)**	**0.5 (±0.14)**	**26 (± 1.1)**

<u>Treatment 2: *Fucus* experiment.</u> In July and August 2019, there were three efforts to transplant *Fucus* on cobble to the restoration substrates, with the goal of having a minimum of one thallus across 60 sublocations within the treatment (3 rows x 5 blocks x 2 tiers per block x 2 aspects, north and south, per block = 60). While cobble was successfully affixed to the substrates, *Fucus* survival was low after each attempt, with many individual thalli broken or ripped off of the cobble by the following day. This

suggests that individual thalli were weakened, perhaps a result of a stress due to the heat wave in June 2019 (See Section 2.7) combined with transplant stress. In November 2019, there were 49 live *Fucus* thalli across 21 of the 60 sublocations, with 29/80 transplants from the August 2019 planting surviving (Table 3). There was disproportionately higher survival of *Fucus* on the two lowest rows and on north-facing sides of the oyster blocks (Table 3), sublocations that are potentially more protected from desiccation/thermal stress and wave impact.

Table 3. Number and percent survival (±SE) of *Fucus* thalli transplanted on August 30-31, 2019 as of November 25, 2019.

Row	Total	North	South
High	3/31 (10%)	2/8 (25%)	1/23 (.04%)
Mid	6/17 (33%)	2/5 (40%)	4/12 (33%)
Low	20/31 (64%)	12/16 (75%)	8/15 (53%)
Grand total	29/80 (36%)	16/32 (55%)	13/48 (33%)

Very few (about 1/3) of the *Fucus* transplants from 2019 survived into 2020. By late April 2020, only about 18 thalli remained from 2019, and by June 2020, there were only 10. Those that did survive into 2020 tended to be single individuals on the lowest rows and on south-facing sides, and these did not provide much canopy cover. Unsurprisingly, there was no difference in oyster densities (or any other sessile taxa) in plots under *Fucus* compared to plots without *Fucus* when surveyed in June 2020. Another major effort to transplant *Fucus* was conducted in early January 2021. Over the course of three days, we transplanted 140 cobbles with a total of ~380 individual thalli attached and an equal number of control cobbles (no *Fucus* attached) to the high and mid tiers of oyster blocks across all rows. Major changes to the 2019 approach including transplanting *Fucus* at higher densities and collecting and transplanting within a single tide cycle to reduce transplant stress. Biweekly visual checks indicated no loss of transplanted *Fucus* through mid-May 2021. Individual thalli looked healthy, and many become reproductive, including one recruit from the 2019 transplant. However, *Fucus* began to decline in late May 2021, with individuals appearing tattered and rust brown and then disappearing in subsequent checks from the end of May through July. Losses occurred first on the highest row, with individuals persisting for longer on the mid and low rows; only two transplants remained in April 2023. Notably, new recruits were present on two oyster blocks in July 2021, some of which were still present in summer 2023. We did not attempt another transplant as *Fucus* populations at nearby donor sites appeared to be lower than they were in 2018 when first surveyed. Although the low success rate of the transplants is disappointing, the survival of some thalli indicates that *Fucus* can successfully reproduce and grow at Giant Marsh.

Temperatures measured while *Fucus* was still abundant in July 2021 were lower, and relative humidity was higher, under *Fucus* relative to bare controls on the low row (Zabin and Blumenthal 2024). Strikingly, *Fucus* had a positive, statistically significant effect on both oyster and barnacle density (Fig. 10), with ~three times more oysters (Wilcoxon rank sum test W = 2.5, p = 0.046) under *Fucus* compared to control cobbles with no *Fucus* and ~three times greater barnacle cover (Wilcoxon rank sum test W = 1, p = 0.016). Oyster sizes were not

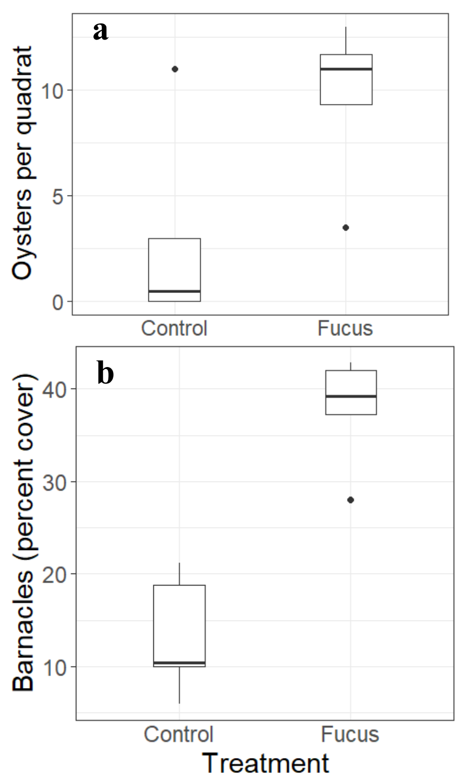


Figure 10. Effect of *Fucus* canopy on a) oyster densities, and b) percent cover of barnacles.

affected by *Fucus* cover (Wilcoxon rank sum test W = 8, p=1). *Ulva* was abundant (70% cover) on controls and absent under *Fucus*. Although the mechanism for these community-level differences isn't completely known, the data indicate that *Fucus* modifies conditions in the under-canopy microhabitat and influences sessile community development in a way that benefits oysters.

Treatment 3. The main objective of the nearshore reef in Treatment 3 is to protect the marsh from erosion. Because the Treatment 3 oyster blocks are located so high in the intertidal zone (oyster block bases at 1.5' NAVD88 [43cm MLLW]), we did not expect them to be rapidly or heavily colonized by native oysters. We monitored this treatment in 2020 and 2022, and in the final year, 2023. Not surprisingly given the intentionally experimentally high elevation, which is about midway between that of the middle and high rows in Treatment 2, we found few live oysters in 2023. We estimated that perhaps 588 oysters (SE ± 302) were present across all of Treatment 3 in 2023, about 25% of what we estimated in 2022 (Table 4).

In 2020 and 2022, most oysters in Treatment 3 were found on the bottom tier where the density was more than an order of magnitude higher than on the center tier (Table 4). No oysters were found on the top tier. There were also far more oysters on the north faces compared with those on the south (Table 4). Consistent with this pattern, the oysters recorded in 2023 were on the bottom tier and north sides of blocks. These patterns were predictable given our experience with Treatment 2. Although we did not install temperature loggers on these blocks, we expect they are experiencing a temperature regime somewhere in between the middle and high row of the oyster blocks of Treatment 2, including exposure to some very high air temperatures during low tides. Nonetheless, given the large number of blocks, even at low densities, Treatment 3 created an opportunity to locally increase the oyster population by tens of thousands of oysters during the drier years. This demonstrates that even high intertidal structures, which might be desirable for shoreline protection purposes, can also support modestly high populations of oysters.

Table 4. Means and standard error (±) of various measurements of oyster performance on the bottom and center tiers of oyster blocks for all three surveys of Treatment 3. Summary statistics for the top tier have been omitted because no oysters were found there in any year.

2020

Tier	Avg. oyster abundance	Avg. oyster cover (%)	Oyster size (mm)
Bottom	116 oysters/m^2 (±15)	2.8 (± 0.5)	17.8 (±0.6)
Center	11 oysters/m^2 (±4)	0.3 (± 0.1)	13.0 (±1.7)
North	60 oysters/m^2 (±11)	1.2 (±0.3)	17.1 (±0.7)
South	25 oysters/m^2 (±7)	0.8 (±0.3)	18.0 (±1.1)
Overall	**42 oysters/m^2 (±7)**	**1.0 (±0.2)**	**17.4 (±0.6)**
Total	**10,350 oysters (±1,219)**		

2022

Tier	Avg. oyster abundance	Avg. oyster cover (%)	Oyster size (mm)
Bottom	234 oysters/m^2 (±42)	5.4 (±1.1)	16.4 (±0.5)
Center	16 oysters/m^2 (±6)	0.3 (±0.1)	15.6 (±1.8)
North	163 oysters/m^2 (±90)	3.8 (±0.8)	16.4 (±0.5)
South	4 oysters/m^2 (±3)	0.1 (±0.1)	15.7 (±3.0)
Overall	**83 oysters/m^2 (±16)**	**1.9 (±0.4)**	**16.4 (±0.5)**
Total	**20,552 oysters (±3,836)**		

2023

Tier	Avg. oyster abundance	Avg. oyster cover (%)	Oyster size (mm)
Bottom	7 oysters/m^2 (±3.6)	0.30 (±0.04)	9 (±1.7)
Center	0	0	0
North	5 oysters/m^2 (±2.4)	0.20 (±0.02)	9 (±1.7)
South	0	0	0
Overall	**2 oysters/m^2 (±1.2)**	**0.1 (±0.01)**	**9 (±1.7)**
Total	**588 oysters (±302)**		

Although not quantified with our survey methods, we note that water is retained in the lower tier of some blocks, which creates tide pool-like habitat in which oysters have settled. In some cases, oysters have settled in small clusters of two and three individuals not attached to the substrate. It could be instructional to investigate what the optimal conditions are for oysters to grow in this way, without the need for attachment to large substrates, as native oysters do in many locations in the Pacific Northwest. Additionally, it interesting to note that even high intertidal structures could potentially be engineered to encourage colonization of oysters through the inclusion of moisture-retaining and shade-creating features.

2.2 Eelgrass and Associated Measurements (Estuary & Ocean Science Center, San Francisco State University)

Eelgrass restoration in the Giant Marsh Living Shorelines Project was limited to Treatment 1: Offshore eelgrass bed and oyster reef (See Fig. 3). Specifically with regard to eelgrass, this treatment addressed the hypotheses that: 1) oyster reefs located in the low intertidal zone can benefit physical processes that enhance survival and growth of eelgrass planted shoreward of reefs; 2) planting eelgrass along with oyster reefs increases community diversity relative to eelgrass-only treatments; 3) planting eelgrass more densely than in other San Francisco Bay projects will increase establishment success through positive feedbacks such as firmer rooting or reduced turbidity; and 4) patchy, naturally-occurring eelgrass in the proposed planting area at Giant Marsh can be enhanced through active planting.

In designing this treatment, we applied and expanded upon lessons from the State Coastal Conservancy's previous living shoreline project conducted at San Rafael from 2012-2017 (Boyer et al. 2017a, b). A 2016 eelgrass planting of the San Rafael LSP resulted in higher eelgrass establishment and persistence shoreward of oyster reefs, compared to eelgrass on its own or bayward of oyster reefs (Boyer et al. 2017b). Data from the San Rafael project also showed that oyster reefs reduced wave energy by 30% at mean tide levels (Boyer et al. 2017a). For the Giant Marsh LSP, we aimed to further test the concept that reefs can protect eelgrass planted on the shoreward side, with replicate plantings on either side of each reef versus plantings alone.

We expected the effects of reefs and eelgrass presence to extend to changes in the animal communities, having found that presence of reefs increased the variety of invertebrates in restored eelgrass at the San Rafael LSP (Pinnell et al. 2021). We assessed the invertebrate communities associated with the eelgrass at Giant Marsh Treatment 1, both on (epifauna) and in the eelgrass (mobile invertebrates like crabs). Further, through our own monitoring and a contract to FISHBIO, we assessed fish use of eelgrass with and without reefs and in comparison with bare mudflats and naturally occurring eelgrass, using multiple methods including sonar, eDNA, and seining/trapping.

Additionally, Treatment 1 of the Giant Marsh LSP tested the role of planting density on restoration success, as had been found important in other regions and ecosystems (Silliman et al. 2015). Eelgrass plantings at nearby Elkhorn Slough (Monterey Bay) had been very successful while using much higher-density eelgrass than has recently been used in San Francisco Bay (80 shoots m^{-2} compared to 11 shoots m^{-2} at the San Rafael LSP [Boyer et al. 2017a] and ~10 shoots m^{-2} in Cosco Busan oil spill restoration plantings [Merkel & Associates and EOS Center 2022]); however, a range of densities was not compared in the Elkhorn Slough project (Beheshti et al. 2022) nor in the aforementioned SF Bay projects. Also pertinent to our design was an experiment in the Dutch Wadden Sea that found greater survivorship of eelgrass seedlings at densities of 14 shoots m^{-2} compared to 5 shoots m^{-2}, albeit only at the higher wave exposures tested (Bos and van Katwijk 2007). Although planting at higher densities is more material- and labor-intensive, we incorporated a range of planting densities into the project design at Giant Marsh, with and without adjacent reef structures, to understand the role of planting density/effort in establishment and spread of eelgrass over time.

Finally, though not originally planned as part of this project, funding from the Ocean Protection Council supported two master's theses at the EOS Center that permitted further understanding of the effects of the Treatment 1 reefs on physical processes that could feed back to ecological functioning. Building on findings at the San Rafael LSP that determined flow reduction shoreward of one of the reefs (Boyer et al 2017a), Carl Hendrickson evaluated detailed patterns of flow attenuation attributable to the Giant Marsh Treatment 1 reefs and also explored early changes (first year) in sediment characteristics, including organic matter and sediment texture. Carl conducted the same evaluations at the San Rafael reefs, the results of which are reported in full in his thesis (Hendrickson 2024) and in a report to the Ocean Protection Council (Buchbinder et al. 2025b). Tessa Filipczyk's thesis focused on determining patterns in organic carbon that could be attributable to the reefs and also compared carbon storage in sediments in natural and restored eelgrass beds in three additional locations in San Francisco Bay, in comparison to mudflats without eelgrass. The results specific to Giant Marsh are reported here; see the complete study in Filipczyk (2025) and summarized in the OPC report (Buchbinder et al. 2025b). Installation of Treatment 1 began in April 2019, when reef balls topped with bags of Pacific oyster shell were placed in three large arrays (see Fig. 5). Eelgrass was harvested from within and adjacent to the

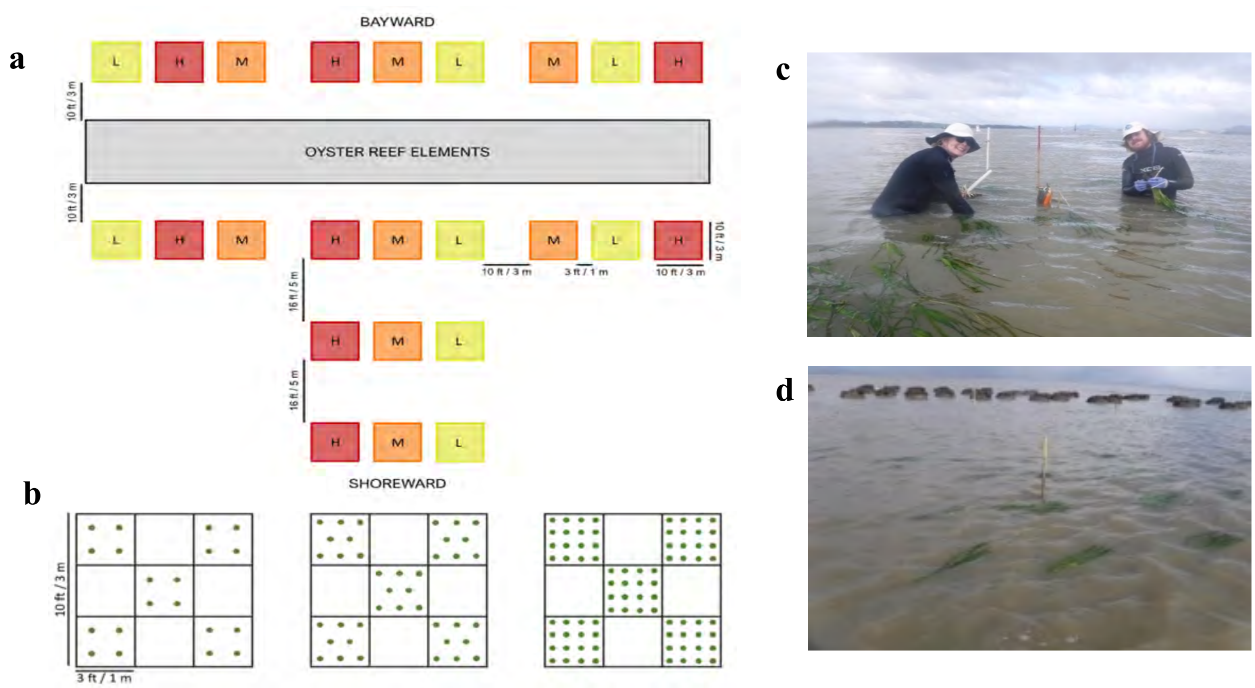


Figure 11. a) Example of the configuration of an oyster-eelgrass combined array in Treatment 1. Planting density arrangement varied at each oyster reef; densities are represented as high ('H', red) for 40 shoots m^{-2}, medium ('M', orange) for 20 shoots m^{-2}, and low ('L', yellow) for 10 shoots m^{-2}. Configurations of the eelgrass only plot arrays duplicated the shoreward row nearest to the oyster reef of the adjacent eelgrass-oyster unit to the south. b) Schematic representing planted eelgrass plots in the densities above, with low to high from left to right. Green dots represent planting units containing 2-3 shoots each. c) Planting eelgrass. d) Immediately after planting.

Treatment 1 area using careful conservation methods to limit collection amounts and areas so as not to damage existing eelgrass, and replanted into the eelgrass plots in May and June 2019 using paper sucker sticks as anchors (a technique commonly used in Boyer Lab and Merkel & Associates eelgrass restoration projects elsewhere in the bay). Plots of low, medium, and high density (10, 20, and 40 shoots m^{-2}, respectively) were planted on either side of each oyster reef or in eelgrass-only arrays (Fig. 11). Monitoring in August and November 2019 revealed large losses of eelgrass, particularly in the plots planted in June, just a few days before a major heat wave that coincided with low tides. Despite the losses observed, by the end of 2019, many plants recovered on their own and planned monitoring resumed in June 2020. Declines were observed again in winter 2021, which is somewhat expected for the season; however, shoot counts were especially low around the southern-most reef (See Fig. 3), where they had previously waned following the June 2019 heat wave. A replant of eelgrass around only that reef occurred in spring 2021.

Beginning in 2021, we began to see evidence that planting eelgrass more densely led to increased densities and total number of shoots present, and these patterns continued through the final summer (2023) of monitoring (see density in Buchbinder 2025a; total shoots by position and planting density in Fig. 12). Importantly, the magnitude of the difference was not proportional to the initial difference in planting density, indicating that the benefits of planting densely must be weighed with the increased effort and resources that are necessary to take on this larger-scale effort. Eelgrass abundance and density relative to reef location were variable by year, with some declines following the storms and low salinity of early 2023 (see Section 2.7 and ESA 2024). There was evidence of the reefs benefitting eelgrass shoreward of the reefs, as seen in previous studies at the San Rafael living shoreline project (Boyer et al. 2017b), particularly in summer 2022.

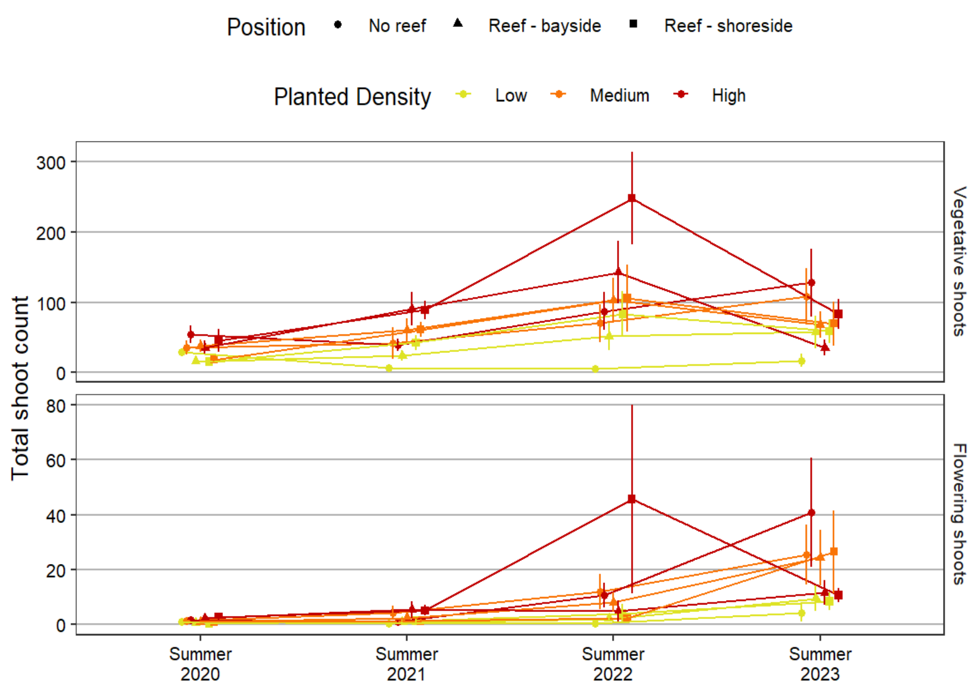


Figure 12. Mean total eelgrass vegetative and flowering shoots by planted density and location relative to reefs in summer 2019 – 2023. Summer 2022 and 2023 total shoot counts were estimated through shoot density and patch sizes. Error bar show ±1 standard error. Mid and far plots shoreward of reefs (See Fig. 3 and 11a) not included (See Fig. 13 for those results).

Further, significant increases in shoot counts (Fig. 13) and vegetated patch area in 2023 in plots further shoreward of the reefs, at the mid and far positions (See planting design in Figs. 3 and 11a), indicate that reefs may provide benefits to eelgrass located upwards of 20 m shoreward of the reef. In contrast, the

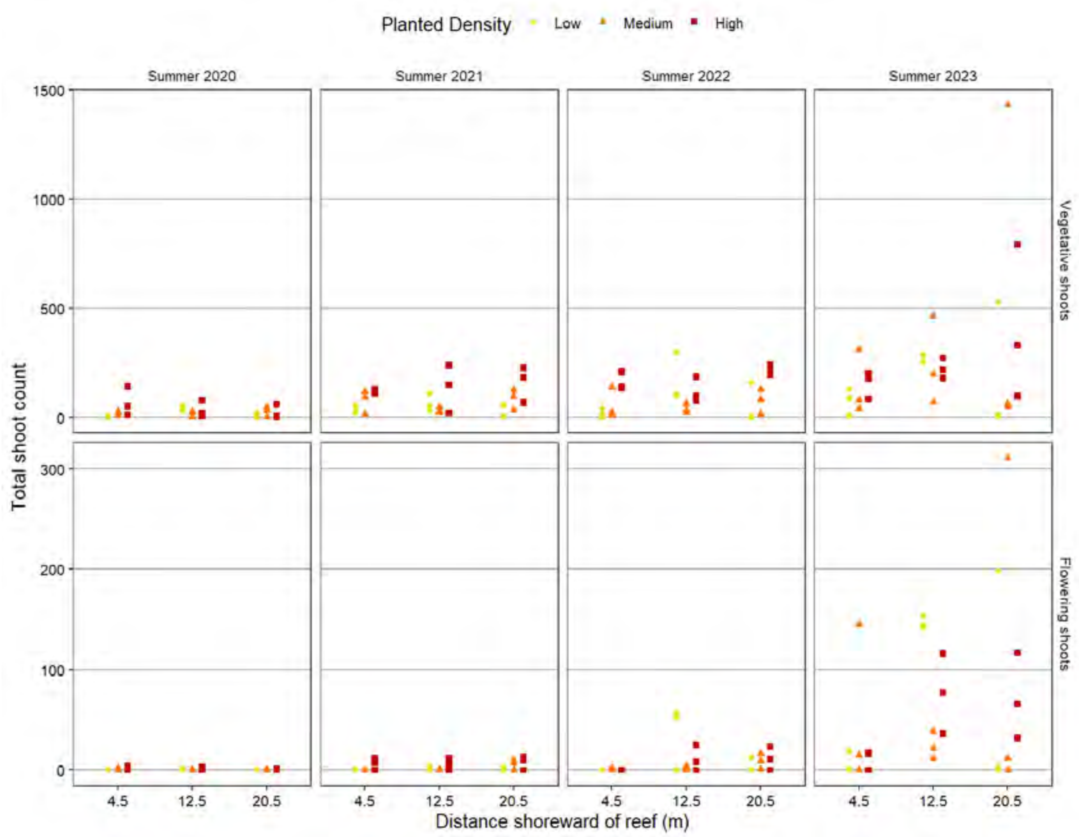


Figure 13. Eelgrass total vegetative and flowering shoots by distance shoreward of the reefs in summer 2020 – 2023. Plot includes data from the shoreside near, mid and far locations (See design in Figs. 3 and 11a); for the near location, only the three center plots were included. Total shoots were calculated from patch sizes and densities in 2022 and 2023, when there were too many shoots to count.

worst outcomes in both patch sizes and total shoots (Fig. 12) came from low density plantings with no association with reefs, suggesting that eelgrass planted on its own may benefit from higher density plantings. Overall, the success criterion of eelgrass achieving at least twice initial densities was met, with densities more than doubling across the site by summer 2023.

Invertebrate communities, including both epifaunal invertebrates and large mobile invertebrates, benefitted from oyster reefs as well, with species richness, diversity, and abundance often higher in reef-associated eelgrass compared with eelgrass growing alone (Buchbinder et al. 2025a, see Fig. 14 for diversity metrics of epifauna). The success criterion of invertebrate species richness increasing by 15% was assessed relative to the natural reference Point San Pablo bed; this goal was met for both total epifauna and large mobile invertebrate species richness (Buchbinder et al. 2025a). We documented persistent differences in invertebrate community structure between the natural bed and restored eelgrass (Fig. 14), as well as strong interannual shifts in dominant species (Fig. 15).

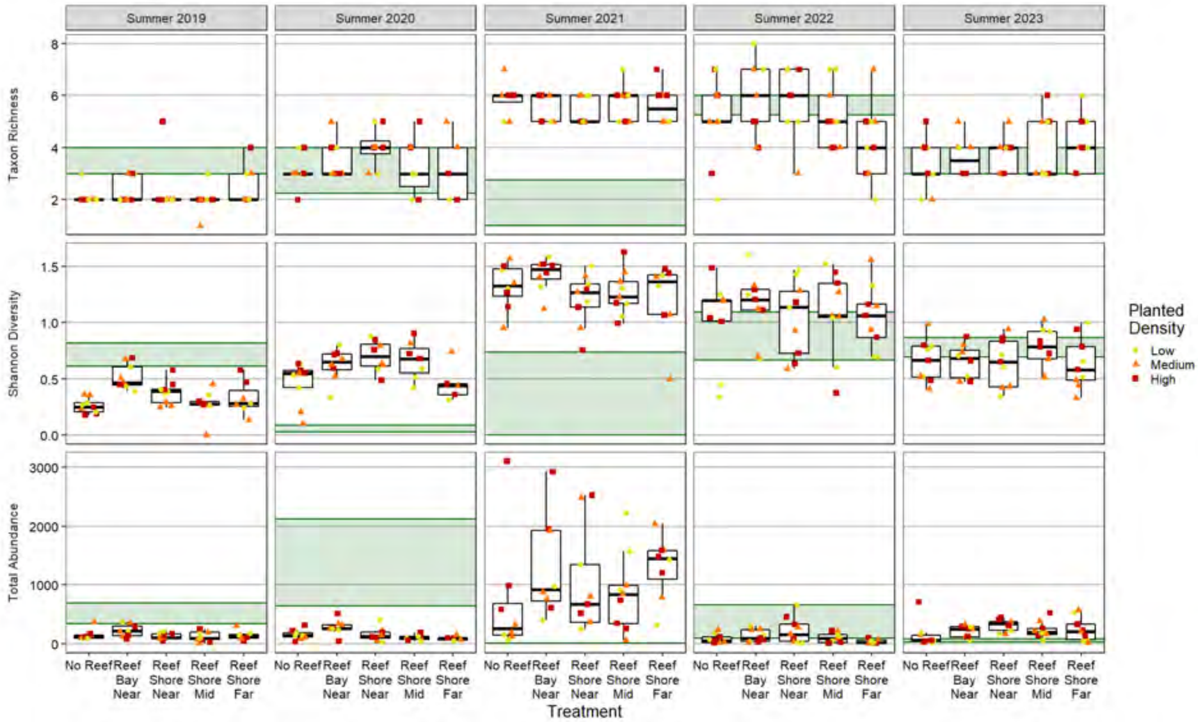


Figure 14. Diversity metrics for Giant Marsh Treatment 1 plots and PSP reference bed epifaunal communities, for 2019 - 2023. Boxplots show abundances at Giant Marsh and green shaded areas show the interquartile range at the reference eelgrass bed, known as Point San Pablo. Abundances are denoted as individuals shoot^{-1}.

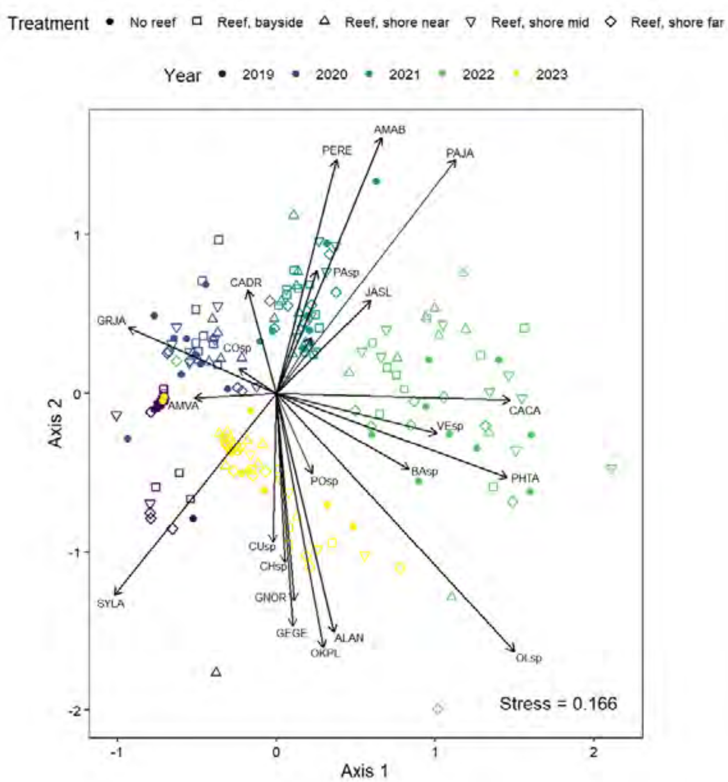


Figure 15. Non-metric Multi-dimensional Scaling (nMDS) of epifaunal invertebrate communities on eelgrass in Treatment 1, 2019-2023. Shapes denote treatment (location within Giant Marsh) and colors denote year. Points represent assemblages found on individual eelgrass shoots, and arrows show the gradients driven by the species present. Species abbreviations: ALAN – *Allorchestes angusta;* AMAB – *Ampelisca abdita;* AMVA – *Ampithoe valida;* ARSE – *Arcuatula senhousia;* BAsp – Barnacle spp.; CACA – *Caprella californica;* CADR – *Caprella drepanochir;* CHsp – *Chromopleustes* spp.; COsp – *Corophiidae* spp.; CUsp – Cumacean sp.; GEGE – *Gemma gemma;* GNOR – *Gnorispaeroma oregonensis;* GRJA – *Grandiderella japonica;* JASL – *Jassa slatteryi;* OKPL – *Okenia plana;* OLsp – *Oligochaete* spp.; PAsp – *Paradexamine* spp.; PAJA – *Paranthura japonica;* PERE – *Pentidotea resecata;* PHTA – *Phyllaplysia taylori;* POsp – *Polychaeta* spp.; SYLA – *Synidotea laticauda;* VEsp – *Velutinidae* sp.

Figure 16. Photographs of A) Hoop net deployed adjacent to reefs at Giant Marsh; B) FISHBIO staff and SFSU student and staff bring in a seine; C) ARIS imagery showing fish swimming between oyster reefs; and D) ARIS imagery showing a leopard shark swimming in eelgrass.

We included multiple methods of fish sampling (Fig. 16) to maximize understanding of fish usage considering known limitations of each technique singly (Buchbinder et al. 2025a, Eschenroeder et al. 2022). We found hoop nets captured larger fish on average than seines, which represented fish of a broader number of sizes, while eDNA picked up additional species not detected by other methods. No strong patterns were detected by treatment, but there was a trend of greater catch per unit effort (CPUE) in restored eelgrass with hoop nets and in natural eelgrass with seines. ARIS sonar captured more fish than physical sampling methods, and showed a trend of greater CPUE in eelgrass associated with reefs compared to restored eelgrass alone. However, neither had a greater CPUE than in bare mudflat, thus we were not able to demonstrate a 50% increase in fish visits to the Treatment 1 reefs and eelgrass relative to control areas with no physical structure, as specified by the success criteria. Overall, we see fish sampling as an ongoing challenge in the complex structures of living shorelines projects.

Relative water motion measured via plaster block dissolution in winter 2020 showed that flow was reduced by 36% shoreward and close to the reefs, declining to 17% and 6% by 8 and 16 m shoreward of the reef, respectively (Fig. 17). Such effects were not detected in a summer 2020 plaster block deployment, perhaps due to overall higher wave energy in the summer or presence of developing eelgrass patches obscuring reef effects.

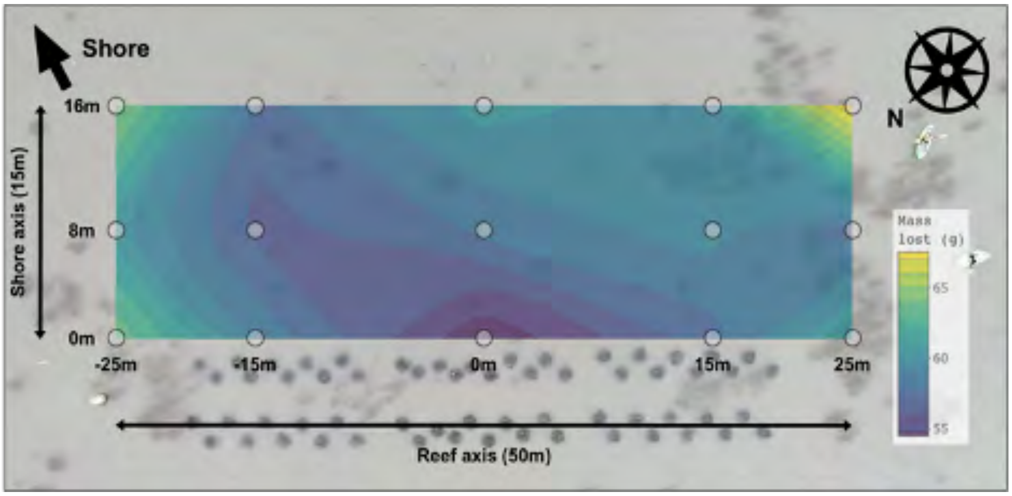


Figure 17. 'Heat map' showing plaster mass loss overlaid on drone imagery (from August 2020) in winter 2020. Darker colors show lower mass lost. Circles indicate locations of plaster blocks during deployment.

Summer 2020 measures of sediment characteristics found few patterns in organic matter, organic carbon, or percent fines, though organic carbon showed a trend of a reef effect at the middle reef within 1.5 m as well as 16.5 m shoreward of the reef (Hendrickson 2024, Buchbinder et al. 2025a,b). In a 2022 assessment of carbon and related variables, reef-associated eelgrass showed trends of increased organic carbon at depths of 4-12 cm, though this effect was not found in the surface sediment (Fig., 18; Filipczyk 2025, Buchbinder et al. 2025a,b). These two early evaluations of sediment changes indicate reef presence may aid sediment carbon accumulation; however, erosion at the surface, perhaps from increased turbulence as water moves through the reef elements, may somewhat counteract this. Further studies will be needed to understand longer-term carbon storage trends attributable to the presence of reefs shoreward of eelgrass plantings at the Giant Marsh living shorelines project.

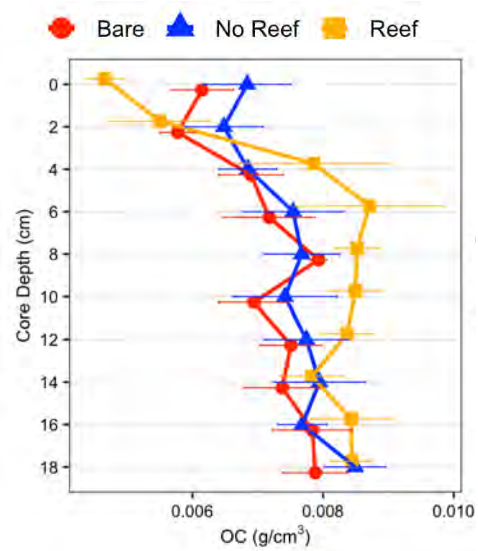


Figure 17. Sediment core profile of organic carbon density (OC g/cm^3) with depth (cm). The y-axis indicates the depth of the top interval; e.g., "2" represents the 2-4 cm deep section.

2.3 **Native Cordgrass Revegetation** (Olofson Environmental, Inc.)

Planting of native Pacific cordgrass occurred in 2022 as part of Treatment 4: Pacific Cordgrass Revegetation Inshore from Nearshore Reef and Treatment 5: Pacific Cordgrass Revegetation Adjacent to Existing Pacific Cordgrass. For Treatment 4, Pacific cordgrass was planted in conjunction with the nearshore oyster reef described briefly above in Section 2.1, to evaluate whether the wave attenuation provided by the oyster reef could help establish planted Pacific cordgrass (*Spartina foliosa*). For

Treatment 5, Pacific cordgrass was planted adjacent to existing cordgrass to test if this could help to facilitate plant establishment and expansion into unvegetated areas.

The Baylands Habitat Goals Science Update (2015) recommended that native cordgrass, as a low marsh species that grows at the leading tidal edge of the bay, be incorporated into restoration projects including living shoreline projects that seek to protect shorelines. A discontinuous fringe of Pacific cordgrass (*Spartina foliosa*) grows along the shoreline within the project area at Giant Marsh, which provides the opportunity to experiment with planting methods that could facilitate plant establishment along unvegetated portions of the shoreline at appropriate tidal elevations. Habitat restoration within the Point Pinole marsh complex is identified as a Priority 1 action to recover California Ridgway's rail within the San Pablo Bay Recovery Unit of the US Fish and Wildlife Service Recovery Plan for Tidal Marsh Ecosystems of Northern and Central California (USFWS 2013).

Invasive hybrid *Spartina* (hybrid *S. alterniflora* x *foliosa*) is a continuing problem in the San Francisco Estuary that is actively managed by the San Francisco Estuary Invasive *Spartina* Project, a regional coalition of partners led by the State Coastal Conservancy, US Fish and Wildlife Service, and California Invasive Plant Council (https://spartina.org). After consultation with Invasive *Spartina* Project staff, with caution, native Pacific cordgrass was included in the project design. The native cordgrass was sourced from Port Sonoma Marina in Sonoma County and propagated by the Watershed Nursery Cooperative.

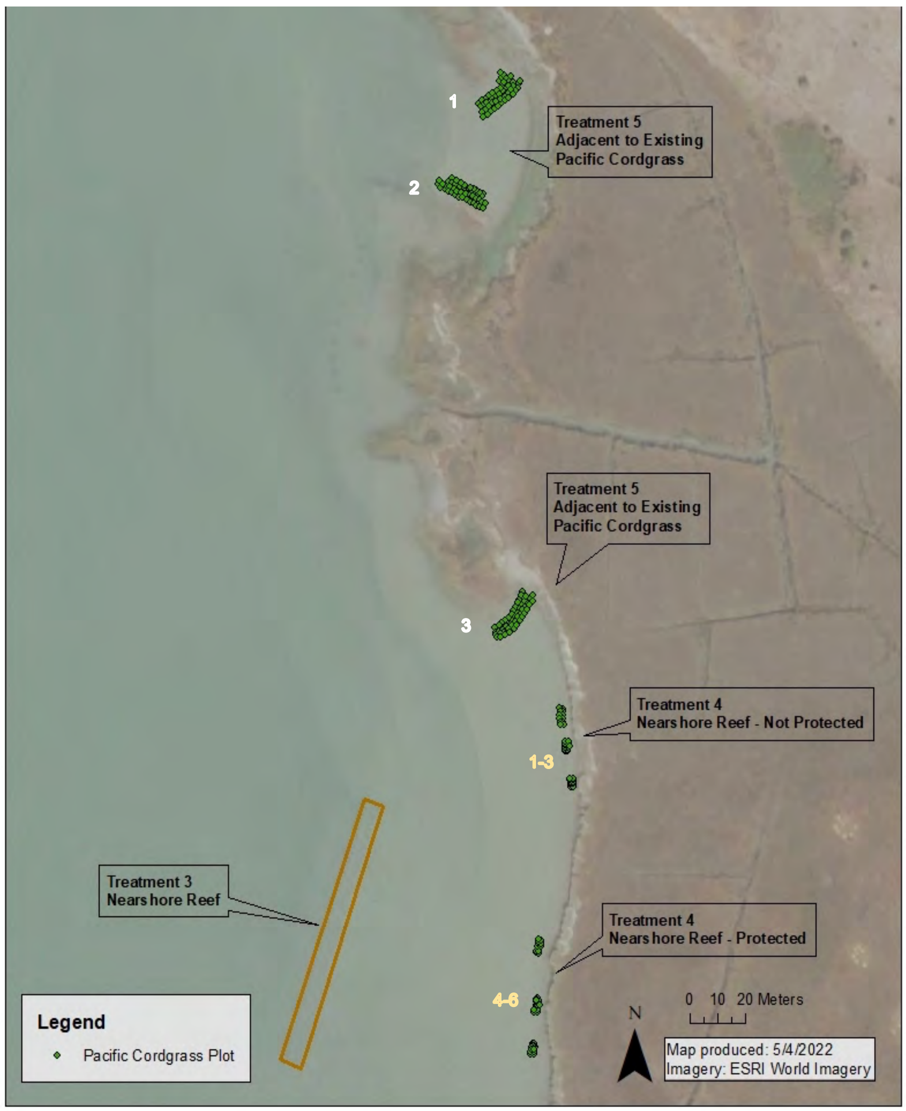


Figure 18. As-built locations of the cordgrass blocks planted in Treatments 4 and 5. Treatment 4 blocks (labels in yellow) 1-3 were placed northeast of the Treatment 3 nearshore reef in an area not expected to receive protection from the reef, while blocks 4-6 were directly landward of the reef and expected to receive more protection. Treatment 5 "transect" numbers 1-3 are denoted in white.

Treatment 4. In January 2022, Pacific cordgrass was planted near the Treatment 3 nearshore reef (see as-built locations of cordgrass blocks in Fig. 18, which differ somewhat from originally planned locations shown in Fig. 4), both landward from the reef in a location expected to receive some wave-sheltering, and somewhat to the northeast in a location expected to not receive as much protection. Three cordgrass planting methods were tested (Fig. 19): bare-root plugs (~5-7 stems with rhizomes) planted in clusters of five (total of twenty plugs), 2) four larger-sized (0.3 x 0.3m, and <0.3m deep) square-shaped "sod" pieces placed together into a larger (0.6 x 0.6m) square in shallow trenches and anchored using one wooden state per sod, and 3) two bare-root plugs planted into each of four ~0.4 x 0.7m burlap bags half-filled with clean nursery soil, placed into shallow trenches and anchored using two wooden stakes per bag. All three planting methods were installed together in a planting block and six planting blocks total were installed, three in the presumably less protected area north of the nearshore reef (blocks T1-3) and three landward of the reef (blocks T4-6). Additional details of methods can be found in Hammond (2025).

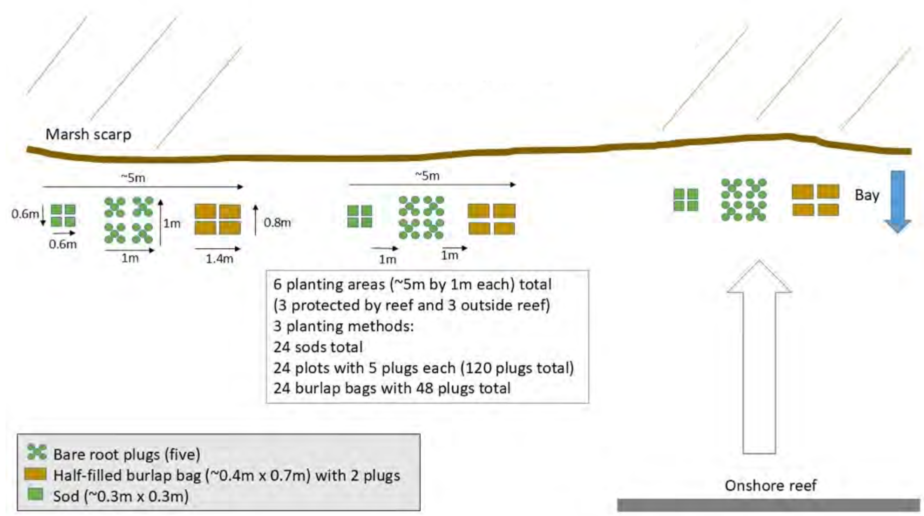


Figure 19. Schematic of planting design for Treatment 4. Distances between planting blocks not to scale.

Monitoring in December 2022 showed that the planting method with the highest survivorship was sods; all were still present in all six planted areas (Table 5). For bare root plug plots, of the 120 plugs planted across the 6 plots, only 28 were present during this monitoring (23%). For burlap bags, which were planted with two plugs in each bag, 15 of 48 plugs survived (31%). There was no pattern of higher survivorship behind the inshore reef; i.e., between blocks T1-T3, which were planted in an area expected to be less protected by the nearshore reef, and blocks T4-T6, which were planted shoreward of the nearshore reef and expected to receive some protection from wave erosion.

Monitoring in May 2024 (proxy for the 2023 growing season) revealed that all burlap bags had failed and there were no surviving plugs that were installed with the bags. For bare root plug plots, there were survivors in only one location, block T5, landward of the nearshore reef. In contrast, sod survivorship was highest in blocks T1 - T3 (although these blocks were expected to be less protected) and somewhat reduced in T4-T6 (expected to be more protected).

Table 5. Treatment 4 monitoring results. Planting blocks T4-T6 were due east of the nearshore reef and T1-T3 were placed to the northeast (See Fig. 18).

Survivorship (%)					
Block	**Type**	**Number Plant Units Installed**	**2022 Initial Surv (Dec 2022)**	**2023 (Proxy in May 2024)**	**2024 (Nov 2024)**
T1	Sod	4	100%	100%	100%
	Plugs	20	35%	0%	0%
	Burlap (Plugs)	8	25%	0%	0%
T2	Sod	4	100%	100%	100%
	Plugs	20	15%	0%	0%
	Burlap (Plugs)	8	25%	0%	0%
T3	Sod	4	100%	100%	100%
	Plugs	20	35%	0%	0%
	Burlap (Plugs)	8	38%	0%	0%
T4	Sod	4	100%	75%	50%
	Plugs	20	0%	0%	0%
	Burlap (Plugs)	8	38%	0%	0%
T5	Sod	4	100%	100%	100%
	Plugs	20	55%	55%	55%
	Burlap (Plugs)	8	63%	0%	0%
T6	Sod	4	100%	50%	50%
	Plugs	20	0%	0%	0%
	Burlap (Plugs)	8	0%	0%	0%

By November 2024, most sod plots continued to persist with only one additional sod lost (at block T4). Minor rhizomatous expansion beyond the original sod footprint was recorded for most plots, including the one remaining plugs plot, indicating that the cordgrass in these plots may continue to persist.

Comparing the three planting methods tested as part of Treatment 4, sods survived in all planted blocks, burlap bags failed, and bare root plugs survived in only one of the blocks (T4, landward of the reef). Minor expansion via tillers (less than one meter) beyond the original sod footprint was observed in all surviving planted blocks. Lateral growth via rhizomes was noted to be progressing slowly and plant persistence may be tenuous due to the small size of each fairly sparse "patch". Still, in all but one of the planted blocks, the area covered by cordgrass originating from sods more than doubled (mean 128% increase in cover across all six blocks; Hammond 2025).

Based on Treatment 4 monitoring results to date, planting method was a better predictor of cordgrass survivorship than whether the plants were installed behind the nearshore reef. This may have been in part because the mudflat area where blocks T1-3 were placed, northeast of the expected reef protection, experienced considerable sediment accretion since the project inception in 2019 (Section 2.7; additional detail in ESA 2024), presumably as a result of the reef's presence. This area was also nearer to existing cordgrass, which accreted a substantial amount of sediment during the project period (Section 2.7) and may have provided some additional protection. Erosion of the marsh scarp east of this broad mudflat, which led to loss of naturally-occurring pickleweed at the scarp edge during the project (Section 2.4 and Buchbinder and Boyer 2024), may have been a localized source of sediment supporting cordgrass establishment along this shore, perhaps with trapping by the nearshore reef providing positive feedback.

<u>Treatment 5</u>. Pacific cordgrass was planted in three bands (also referred to as "transects") adjacent to existing cordgrass in January 2022, with each band running roughly perpendicular to the shoreline (Figs. 18, 20). Two planting methods were compared to see which was more effective at establishing cordgrass. The two planting methods included in Treatment 5, bare-root plugs and sods, were as described above for Treatment 4.

Monitoring occurred in December 2022, May 2024 (proxy for 2023 growing season), and November 2024. Survivorship after the first growing season (2022) was fairly low. Sods survived much better than plugs, ranging from 36% for T2 to 66.7% for the most southern transect, T3 (Table 6).

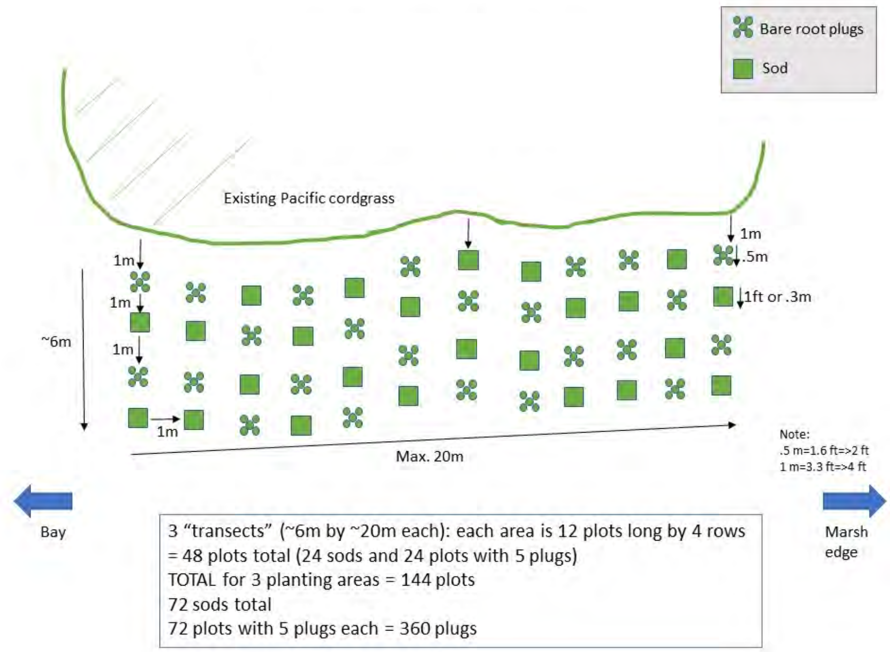


Figure 20. Schematic of planting design for Treatment 5.

Plots that were planted with 5 plugs had much lower survivorship than sods, with only 2 out of 24 plots (8%) having surviving plugs in T1 and T3 and no plugs surviving in T2. Within the two plots in T1 and 3 that had surviving plugs, plug survival ranged from 1 to 4 (T1: 1 and 2 plugs, T3: 1 and 4 plugs). During monitoring in 2022, no expansion beyond the plots was observed but tillers were noted in multiple plots (both in plugs and sod plots). In May 2024**,** we found T1 sod survivorship was similar to previous years. Transect 2 had the lowest survivorship in 2022 and sod survivorship declined further by May 2024 but did not decline further through November 2024. Transect 3 sod survivorship declined in May 2024 but did not decline further through November 2024. In 2024, no bare root plug plots persisted in transects 2 or 3 (Table 6).

Table 6. Treatment 5 cordgrass survivorship. T1, T2, and T3 indicate the three planted bands, or "transects" marked in white labels on Fig. 18.

	2022						**2023**		**2024**	
	Installed Jan 2022		**Initial Survivorship Dec 2022**				**Survivorship May 2024 (Proxy)**		**Survivorship Nov 2024**	
	Number of Sods	**Number of 5 Plug Plots**	**Number of Sods**	**Number of Plots with Surviving Plugs**	**Sods (%)**	**Plots with Surviving Plugs (%)**	**Sods (%)**	**Plots with Surviving Plugs (%)**	**Sods (%)**	**Plots with Surviving Plugs (%)**
T1	24	24	10	2	41.7%	8.3%	41.7%	8.3%	41.7%	8.3%
T2	25	22	9	0	36.0%	0%	10.6%	0%	10.6%	0%
T3	24	24	16	2	66.7%	8.3%	54.2%	0%	54.2%	0%
	73	70	35	4						

Overall in Treatment 5, sods survived much better than plugs across the three planted bands, ranging from 11-54% across the three bands. In general, plots survived if they were closer to the marsh, at slightly higher elevation. Using the project's 2016 survey depth contours (in feet NAVD88), in Fall 2024, fewer plots survived below the 3' contour (9 out of 30). Initial survivorship of sods did not appear to be affected by distance from existing established Pacific cordgrass. However, during the years of monitoring, the existing cordgrass appeared to be expanding into the planted plots and the planted plots expanded laterally as well so that it is now impossible to distinguish between them. Sediment accumulation in all three locations (See section 2.7 and ESA 2024) since the start of the project in 2019 likely aided in cordgrass expansion, with the planted cordgrass sods leading to trapping of additional sediment.

Sod survivorship was much better than plugs (as with Treatment 4) but still only low to moderate, ranging from 11-54% across the three bands. We cannot say definitively that the planted bands aided in spread of existing cordgrass into unvegetated areas; however, in two of the three bands (T1 and T3) existing and planted cordgrass from the sods coalesced, filling in ~10m^2 such that they became indistinguishable. This filling in along the existing cordgrass as well as lateral spread of ~35m^2 in those two planted bands indicate that sod plantings enhanced the cordgrass overall in these locations.

2.4 **Marsh Scarp Revegetation and Arboring for Sea-blite and Pickleweed** (Estuary & Ocean Science Center, San Francisco State University)

Treatment 6 addressed the hypotheses that 1) California sea-blite (*Suaeda californica*) can be established at Giant Marsh to aid in the recovery of this federally endangered plant; 2) CA sea-blite will establish and grow with greater vigor if plantings are provided with wrack additions; and 3) marsh vegetation (sea-blite and pickleweed, *Salicornia pacifica*) can be trained to grow taller with support, to increase high tide refuge and cover for birds and small mammals during flooding events. A focus on plant species with a propensity to climb that are also drought-tolerant could help to provide such refuge, even in drought periods when other high marsh plants such as *Grindelia stricta*, which has been used extensively in marsh restorations to provide nesting substrate and cover (Rohmer and Kerr 2023), does poorly, losing its ability to provide effective refuge (Peter Baye, pers. comm.).

California sea-blite (hereafter sea-blite) is a federally listed endangered species that was extirpated from San Francisco Bay in the early 1960s (Baye 2006) and recommended for recovery action (USFWS 2013). It has since been the target of re-introduction trials as well as experiments to inform these efforts (e.g., Santos 2020, Santos et al. 2020, Buchbinder and Boyer 2023). It is a salt-tolerant, fleshy-leaved coastal wetland shrub that can grow several feet tall, and therefore has the potential to provide high tide refuge for birds and mammals. Additionally, it has been observed to climb on structures such as logs and fences in its extant populations in the marshes of Morro Bay, leading to the recommendation that these structures could be proactively simulated in restoration projects (Baye 2006, USFWS 2013). Such "arboring" using tree branches inserted into the marsh might increase plant height and enhance bird and mammal refuge services. In addition, observations of increased vigor and seedling establishment in accumulated wrack (eelgrass, algae, and marsh plant debris at the high tide line) in Morro Bay (Baye 2006) led to the hypothesis that adding wrack to planting holes could be used to provide a nutrient-rich organic matter amendment to help establish sea-blite in restoration settings.

Pickleweed is the dominant species in the saltmarsh plain of San Francisco Bay marshes (and throughout California). It has also been observed climbing structures in marshes, including logs, similar to sea-blite (authors' personal observations, Santos et al. 2020). These observations led us to propose testing arboring in existing pickleweed at Giant Marsh as a way to enhance high tide refuge, particularly with the salt-marsh harvest mouse in mind, as pickleweed is a preferred habitat for this federally listed endangered species (USFWS 2013).

Figure 21. Map of Treatment 6 sea-blite (*Suaeda californica*) blocks (S1 – S8 in blue) and pickleweed (*Salicornia pacifica*) blocks (P1-P5 in orange). Sea-blite blocks noted include the full arboring and wrack treatments in their crossed design; additional plantings after 2020 (that did not follow this design) not shown.

Planting blocks for both species were established in January 2019 (Fig. 21). Arboring was accomplished using *Eucalyptus* branches with diameters of 2- 6 cm collected from downed trees at Point Pinole Regional Shoreline. At arbor locations in both pickleweed and sea-blite blocks, the primary branches of the arbors were inserted approximately 0.5 m into the sediment (See example in Fig. 22b). Branches were trimmed so that arbors did not extend over 0.75 m above the sediment, to prevent them from being used as raptor perches. Two to three smaller branches were enmeshed with the arbors and inserted into the soil to add additional fine "lattice" structure within the main arbor. Arbors were installed in January 2019 for both the sea-blite and pickleweed blocks, totaling 44 arbors.

<u>CA sea-blite</u>. In March 2019, sea-blite propagated at the Estuary & Ocean Science Center (see details of methods and sources in Buchbinder et al. 2024) was planted with arbor or wrack additions, both treatments, or neither treatment (controls), in a series of six experimental blocks along the Giant Marsh shoreline (blocks S1-S6, Fig. 21). Arboring is described above; wrack consisted of primarily fresh eelgrass leaves collected at the high tide line, roughly chopped using scissors and trowels, with a volume of ~500 cm^3 (two dense handfuls) mixed into the sediment at the bottom of the planting hole. This experiment was supplemented to replace lost plants later in 2019 and again in 2020; the latter included planting into existing blocks as well as into two new blocks selected to test survivorship in locations presumably more protected due to bayward marsh vegetation (S7 and S8, Figs. 21, 22). Two previous blocks, S1 and S6, were not replanted due to repeated loss of plants and arbors, and observations of conditions that

may make these locations unsuitable for sea-blite. For example, large amounts of sediment (which was 96-99% sand across plots, Buchbinder et al. 2024) movement at block S6 suggested that this location may be too dynamic for sea-blite to successfully establish. Figure 22 details the planting dates and surviving numbers over time in the plots with the full experimental design. Survivorship continued to decline over subsequent monitoring periods, and in September 2023 overall survivorship was 0% for plots S1, S2, S3, S6, and S8; 13% for plots S5 and S7; and 75% for plot S4, with 8 total plants present representing 13% of 64 planted locations (eight blocks with eight planted locations each) and an overall survivorship of 9% of all 92 individuals planted from 2019-2020 (Fig. 22).

a

		Sea-blite blocks from N to S								
		S1	S2	S3	S4	S5	S6	S7	S8	
Date	**Activity**	**Count of sea-blite individuals**								**(Total)**
3/11/2019	planted	8	8	8	8	8	8	-	-	48
4/19/2019	# surviving:	3	8	7	8	6	4	-	-	36
	replanted	5	0	1	0	2	4	-	-	48
9/5/2019	# surviving:	0	7	5	5	2	3	-	-	22
1/17/2020	# surviving:	0	7	5	4	2	2	-	-	20
3/4/2020	# surviving:	0	7	4	4	1	-	-	-	16
	replanted	0	1	4	4	7	0	8	8	48
9/22/2020	# surviving:	0	3	1	8	5	1	5	0	23
1/22/2021	# surviving:	0	3	1	8	5	0	5	0	22
9/13/2021	# surviving:	0	2	1	8	3	0	5	0	19
1/25/2022	# surviving:	0	2	0	7	2	0	4	0	15
9/27/2022	# surviving:	0	2	0	7	2	0	4	0	15
1/3/2023	# surviving:	0	1	0	7	2	0	4	0	14
9/18/2023	# surviving:	0	0	0	6	1	0	1	0	8

Figure 22. a) Timeline of planting and survivorship of sea-blite in the eight blocks planted with the full experimental design in 2019 and 2020, b) a sea-blite plant beginning to climb an arbor, January 2020, c) a bright blue-green individual that had been treated with wrack in the planting hole, and d) an arbored plant at right with meter stick for scale, and an un-arbored plant at left, September 2020.

Ongoing losses of plants led to additional planting efforts in 2021, 2022, and 2023, which deviated from the original experimental design; i.e., no arboring was completed in these replanting years, as we decided to delay this labor-intensive activity until it was determined that plants remained in place (see details of additional planting in Buchbinder et al. 2024). Plant losses continued, mostly during winter months, indicative that exposure to wind and waves during storm events, as well as sediment movement, may exceed the plants' tolerance for physical stress in many locations along this shoreline. Qualitative monitoring in March 2024 showed that all sea-blite present in September 2023 from the original 2019 – 2020 plantings remained, totaling 9 plants, and along with additional plantings and one recruited seedling (see below), a total of 14 plants were still present. Overall, the most successful planting sites were protected areas where sea-blite had limited exposure to direct wave action, primarily coming from the northwest. As such, persistent plants were often located in the north edge of protected coves with openings to the west or southwest, with marsh or beach areas located to the north in a position that allows for the interception of waves, and which appeared to protect plants from excessive sediment movement.

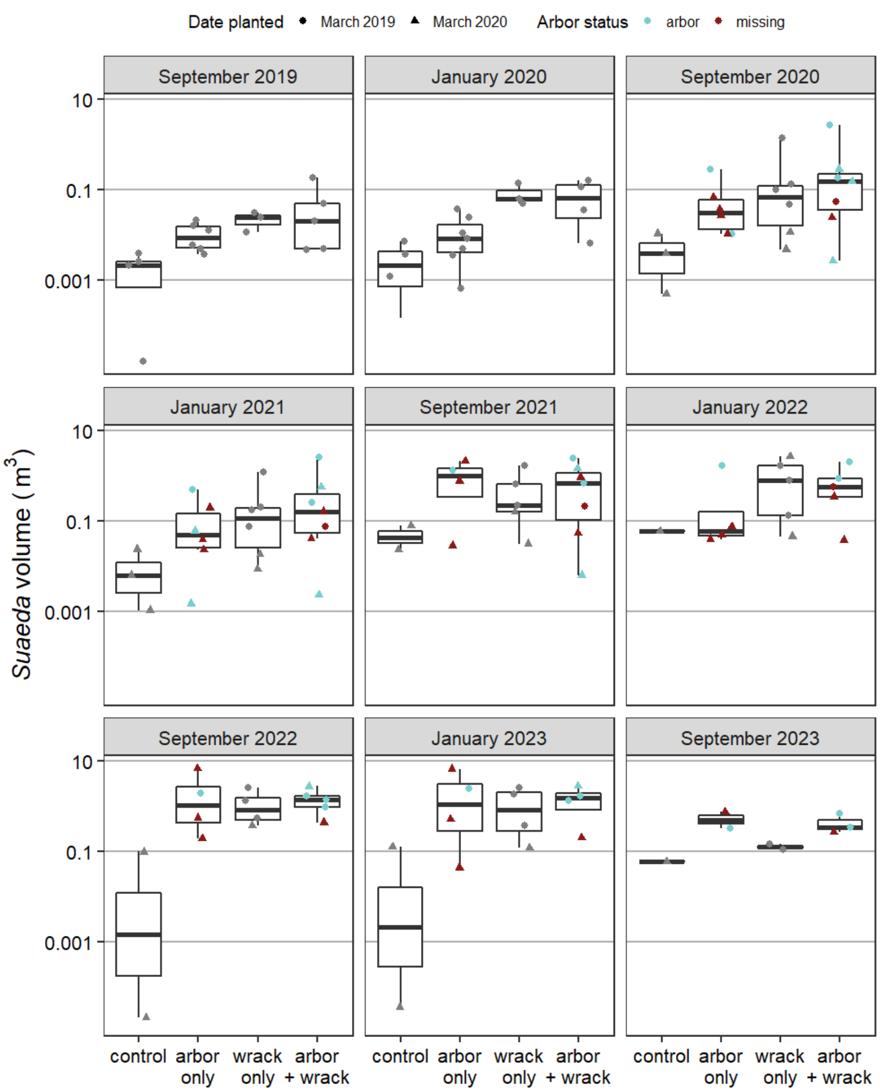


Figure 23. Volume of surviving sea-blite from the original experimental blocks (S1 -S8), by treatment in fall 2019-fall 2023. Circles show sea-blite planted in March 2019 and triangles represent March 2020 replacement plantings. For treatments with arbors, blue points represent plants where the arbor persisted; red indicates the arbor was missing. Points in grey indicate that no arbors were installed (control and wrack treatments), or that arbor condition was not reported (September 2019 and January 2020). Plant volume is shown in cubic meters (half-ellipsoid volume calculated using the plant length, width, and height). Values that appear close to zero are not dead plants, but rather relatively smaller plants. Note the log scale.

Because of high mortality, we were limited in our ability to test the effects of wrack and arbors on the sea-blite. However, we observed that the addition of wrack to planting holes led to bright blue-green plants (Fig. 22c) and significantly increased the height and volume (linear mixed effects model including results through January 2020: $F_{1,16} = 12.68207$, $p = 0.0026$; Fig. 23) of the sea-blite, an effect that visually persisted (replication too low for statistical analysis) in surviving plants into early 2023. Further, the addition of arbors led to a trend of increased height and volume (analysis through January 2020, Figs. 22d, 23). Even as arbors were lost over time, volume of once-arbored plants visually exceeded controls, including on the last sampling date when wrack effects were no longer apparent (September 2023, Fig. 23).

Twenty-two sea-blite plants were found to have recruited naturally to the site beginning in September 2022 (7 noted then, and 15 more in January 2023), indicating that the established population is capable of successful reproduction necessary for creating a persistent population. However, the recruits were impacted by storms in late winter 2023 and only a single self-recruiting individual was found to have persisted as of September 2023.

We conclude that, 1) It is possible to establish sea-blite at Giant Marsh although the site's bayfront edge has high exposure to wind and waves during storms, making the establishment of a long-term population uncertain. The success criterion of establishing three areas of sea-blite plantings at Giant Marsh was met, but with difficulty. 2) The addition of wrack to sea-blite plantings supported early growth as hypothesized, with effects visible for greater than three years; we recommend this as a planting augmentation for future projects with this rare plant. 3) Arboring tended to enhance growth of sea-blite plantings but the high losses of plants made it difficult to assess the benefits; visually, effects were apparent after four years. Combined with other studies of arboring we have conducted (a previous experiment at Brunini Marsh in Tiburon [Santos 2020] and a concurrent experiment at Heron's Head Marsh in San Francisco [Buchbinder and Boyer 2023]), locations with protection from the worst storm surge conditions show very promising results of arboring on both sea-blite height and volume. 4) Reintroduced sea-blite has the potential for self-seeding but will require protected locations for seedlings to persist and contribute to expansion of restored populations.

<u>Pickleweed</u>. Five pickleweed blocks were demarcated in January 2019 on the bayward edge of the existing marsh plain (Fig. 21); each block contained eight plots, with four arbored as described above. Pickleweed blocks were monitored in September and January of each year from September 2019 through September 2023. At each arbor, we measured the maximum height of living plant tissue in the pickleweed canopy within the arbor footprint. In plots with no arbors, we measured the maximum canopy height of living plant tissue within a 50 cm radius of a control point.

Pickleweed arbors held up well over time and appeared to be more stable than the *Suaeda* arbors, which were installed into more mobile sand and in locations with higher wave energy. Notably, the southern pickleweed blocks, P4 and P5 (Fig. 21), were located adjacent to the receding marsh edge when established in 2019. Beginning in January 2022, erosion began to impact some paired arboring points. Between January and September 2023, erosion had progressed such that both of the southern blocks were entirely missing at the later time.

We saw strong evidence that pickleweed was climbing the arbors within the first year (Fig. 24). When looking at paired locations for individual timepoints, the arboring effect was detected in January 2020, and became more significant over time. It is possible that the delay in treatment effect is a result of the disturbance to the pickleweed treatment area during arbor installation, or that it simply takes some time for the climbing to manifest. The positive response of increased pickleweed canopy height with arboring at Giant Marsh was mirrored in an experiment on the pickleweed plain at Corte Madera Marsh Ecological Reserve (CMMER) in Corte Madera, CA. At CMMER, there was no measurable difference in canopy height after two growing seasons; however, in subsequent years, the pickleweed canopy became higher with arboring (Authors' unpublished data), indicating that it could take time to see results, with disturbance during the installation of arbors potentially contributing to the delay.

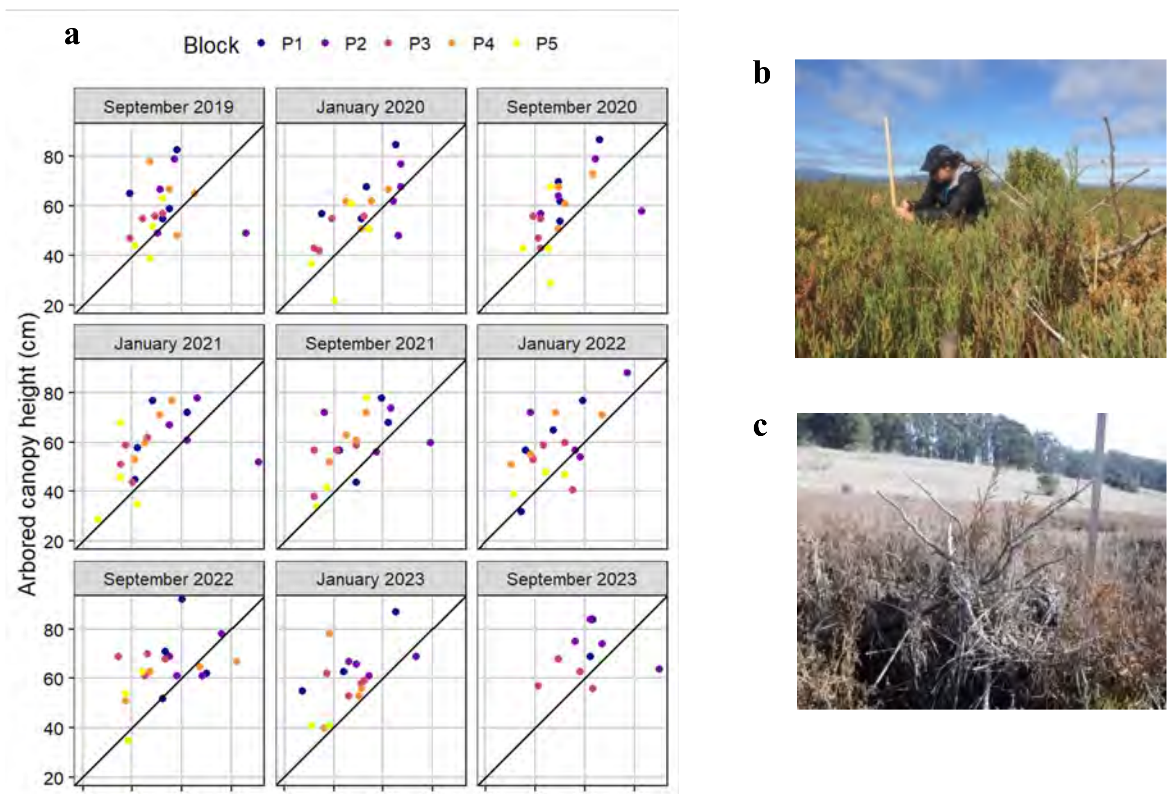


Figure 24. a) Pickleweed canopy height across paired treatment and control points, 2019 - 2023. Each point represents canopy height at a pair of measurement points, with control points on the x axis, and adjacent arbored point on the y axis. The diagonal line is y = x. Points falling above this line represent pairs where the arbored plant was taller. Colors represent blocks (See Fig. 21). Absence of points indicates a loss of the arbored pair, due to shoreline erosion or sediment deposition over the arbor and/or control point. b) Pickleweed climbing an arbor in September 2019 and c) in January 2020.

Pickleweed canopy height benefited from arbors even through natural fluctuations in the canopy of the marsh plain; in September 2022 through 2023, there were similar fluctuations in the maximum canopy height of the non-arbored marsh plain and arbored locations (Fig. 25). The strong effects of this treatment indicate that this technique could effectively be used as a method to increase marsh plain canopy height in the future.

We recommend adding arbors to the pickleweed canopy in locations where an increase in canopy height may be beneficial to wildlife escaping flooded conditions. We only added *Eucalyptus* branches once, and

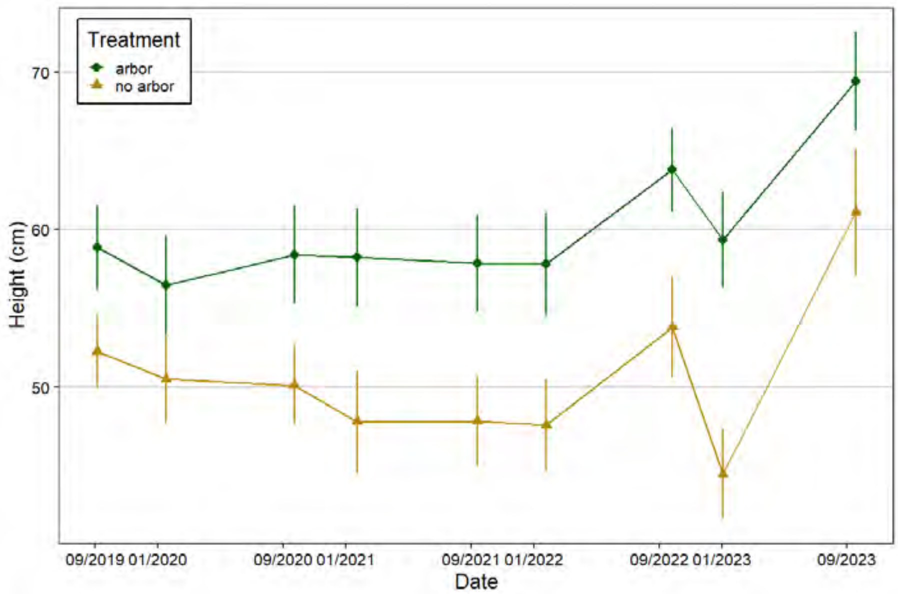


Figure 25. Maximum pickleweed canopy height in arbored and non-arbored (control) points, during fall and winter monitoring dates in 2019 - 2023. Points represent mean heights for each treatment and error bars are ±1 standard error.

limited their height to avoid attracting raptors; canopy height increases greater than the 10 cm average we achieved would likely require repeated additions of taller arbors as the pickleweed grows. Further, as the increases in pickleweed canopy height remained very localized to the arbor location, additional arbors across larger areas of pickleweed are likely to be needed in order to expand effects across larger scales.

2.5 Marsh-Upland Transition Zone Enhancement (Olofson Environmental, Inc.)

Native estuarine-terrestrial transition zones enhance the biological value of tidal marsh and shoreline habitats by providing high tide refuge habitat for wildlife during winter storms and other extreme high tide events. As a refuge from high tides, this zone is identified as an important habitat feature for the protected species California Ridgway's rail and salt marsh harvest mouse (Baylands Habitat Goals Science Update 2015, USFWS 2013). In addition, transition zones provide multiple ecosystem services including buffering the landward effects of tidal processes and the bayward effects of terrestrial processes such as controlling pollution, erosion, and flooding. Native transition zones also tend to be biologically diverse, and in San Francisco Bay support a variety of plants and animals of special management concern (Baylands Habitat Goals Science Update 2015). Transition zones also support the movement and dispersal of plant and animal species along the shore between patches of preferred habitat.

The existing transition zone and uplands adjacent to Giant Marsh are actively treated to control invasive Fuller's teasel (*Dipsacus sativus*). Treatment 7 included planting multiple native species, primarily perennials, to increase both species diversity and habitat complexity by providing additional types of available plant structures that could be used as refuge cover from predators for wildlife. These native perennials will also provide additional flower nectar and seed sources for foraging wildlife during more of the year than the non-native annual grasses present in the upper transition zone that die off in late spring.

The marsh-upland transition zone planting design included 1,000 native plants of five species sourced from the Watershed Nursery Cooperative and planted in dispersed small clusters at elevations from 9-20’ NAVD88 (2016 project survey contours; ESA 2024). Note that the location of Treatment 7 was shifted about 180 m (600 feet) to the northwest from its original planned location (Figs. 4, 26; Hammond 2025). The different plant species were planted across the transition zone as follows (Fig. 26, 27): 1) High marsh edge (9-11’ NAVD88) – clustered plantings of marsh Baccharis (*Baccharis glutinosa)* and western goldenrod (*Euthamia occidentalis*). Several creeping wild rye (*Elymus triticoides*) plots were interspersed among the clusters. 2) Steep slope/scarp (9-12’ NAVD88) – linear plots of creeping wild rye planted along the slope contours. 3) Upland “meadow” (14-20’ NAVD88) – clustered plantings of western ragweed (*Ambrosia psilostachya*) and Pacific aster (*Symphyotrichum chilense*) were dispersed throughout teasel treatment area. Several creeping wild rye plots were interspersed among the clusters of ragweed and aster. These meadow plots were planted in disturbed areas and mulched to help suppress weed species. The meadow plots were located adjacent to a trail heavily used by humans and dogs; they were fenced using PVC poles and rope to protect plants from trampling, and temporary signage was posted (Fig. 27b). 4) We also opportunistically removed a small area of non-native ice plant located along the high marsh edge.

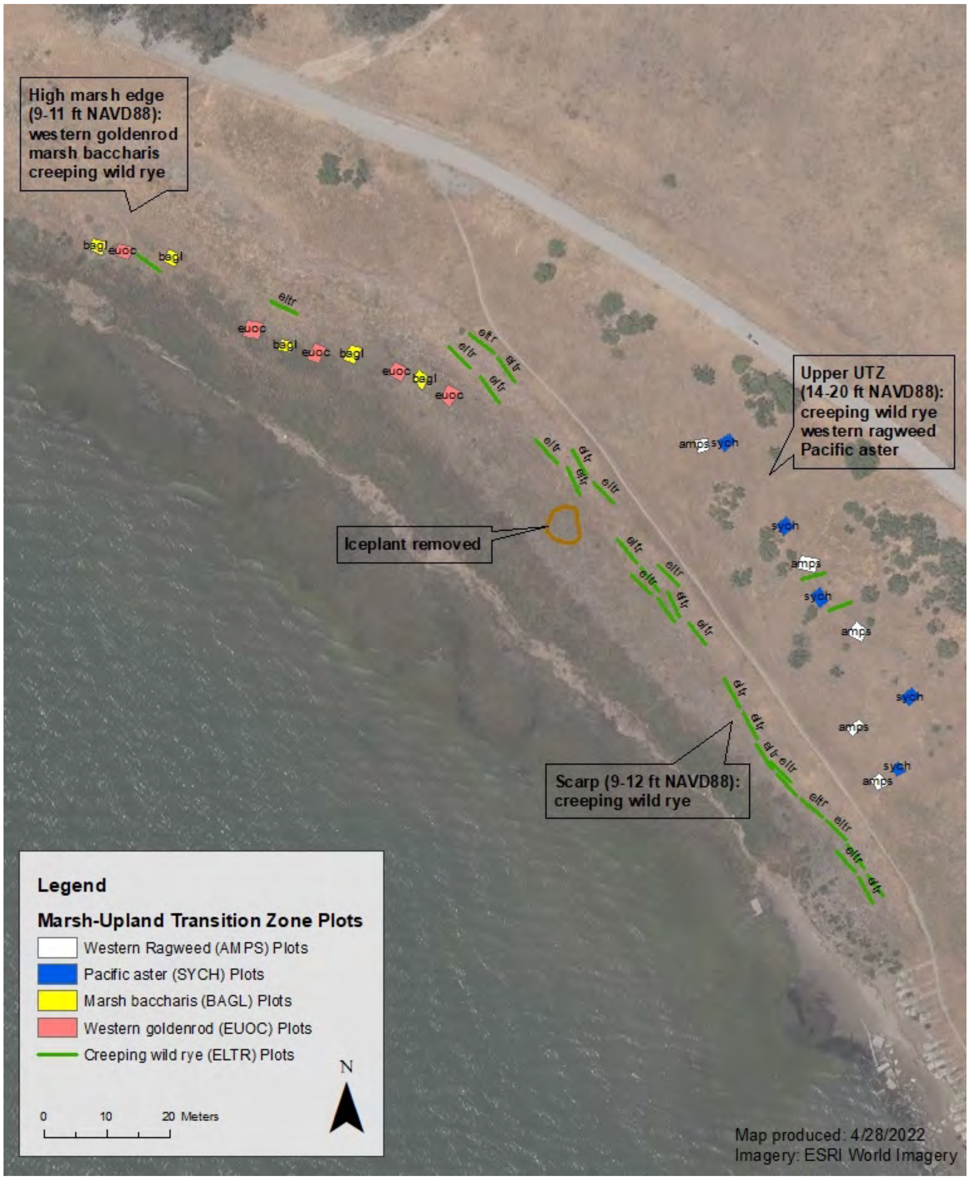


Figure 26. As-built Treatment 7 location, ~180 m (600 feet) NW of planned location, with the various planting treatments.

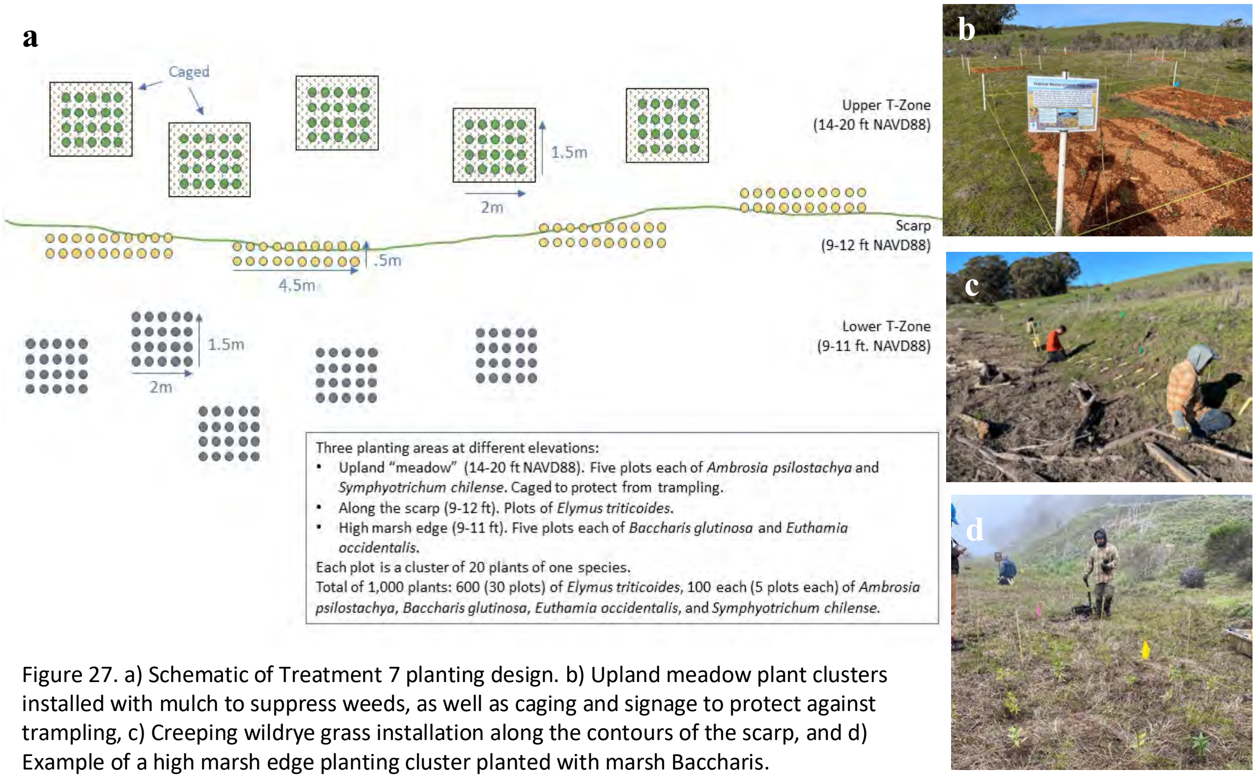


Figure 27. a) Schematic of Treatment 7 planting design. b) Upland meadow plant clusters installed with mulch to suppress weeds, as well as caging and signage to protect against trampling, c) Creeping wildrye grass installation along the contours of the scarp, and d) Example of a high marsh edge planting cluster planted with marsh Baccharis.

Due to drought conditions after planting in early 2022, we watered planted plots using a water wagon as needed through June 2022. In addition, Hydretain was applied during watering after plant installation and three months later during the final watering event. Hydretain attracts soil moisture vapor and converts it into water droplets that can be absorbed by plant roots, which can reduce evaporative loss and extend the time needed between watering events. Supplemental watering was discontinued during the first growing season to encourage deep root growth as recommended by several local restoration practitioners (e.g., Peter Baye and Save The Bay staff). Opportunistic weeding occurred during plant installation, during watering events in 2022, and annually in 2023 and 2024 to ensure teasel was not encroaching in planted plots. Spot herbicide treatment of teasel in the vicinity of the upland "meadow" planted areas was conducted by East Bay Regional Park District staff.

High marsh edge plots had variable survivorship, with no surviving western goldenrod plants but thriving marsh *Baccharis* in two of five planted plots (Table 7). Steep slope/scarp creeping wildrye grass survivorship declined from 37% to close to 0% between 2023 and 2024, while plots in the high marsh edge and upland meadow declined slightly. High marsh edge and steep slope/scarp plots were not mulched or caged and the lack of protection from weed competition and trampling by dogs may have contributed to the lower survivorship observed as compared to the upland meadow plots. In addition, EBRPD did not treat teasel in/near the high marsh edge and steep slope plots so presence of that weed species continued near plots. In general, the weed species present in/near plots differed across the transition zone. A particularly aggressive weed, perennial pepperweed (*Lepidium latifolium*), was present along the high marsh edge potentially contributing to low survivorship. Overall, the creeping wildrye grass plugs planted into steep slope/scarp plots likely experienced the worst growing conditions with disturbed, eroding soils, heavy weed competition, and close proximity to the unofficial trail.

Upland meadow plots of all three species had very high survivorship, likely due to supplemental watering and the addition of mulch to reduce competition with weed species to promote plant establishment (Fig. 27b, Table 7). The simple rope caging installed around the upland meadow plots also likely reduced trampling from dogs and people while plants were establishing.

Table 7. Treatment 7 survivorship over time.

Survivorship (%)					
Planting Type	**Species**	**Number Plant Units Installed**	**April 2023**	**Dec 2023**	**Oct 2024**
High Marsh Edge	Marsh Baccharis	100	39%	49%	49%
	Western goldenrod	100	0%	0%	0%
	Creeping Wildrye	40	50%	42.5%	42.5%
Steep Slope/Scarp	Creeping Wildrye	520	n/a	37.4%	0.2%
Upland "Meadow"	Western Ragweed	100	99%	100%	100%
	Pacific Aster	100	98%	99%	99%
	Creeping Wildrye	40	85%	77.5%	77.5%

Natural estuarine-terrestrial transition zones around SF Bay are rare and frequently degraded by human activities. Most transition zones are actually the result of construction for flood protection and typically vegetated with weed species. Despite being disturbed by heavy use, the natural transition zone areas present at Point Pinole are unique and warrant more active management to ensure that native species and habitats persist. We recommend active invasive species removal in the high marsh edge and steep scarp areas to reduce weed competition. The low survivorship of creeping wild rye grass on actively eroding steep scarp areas may indicate that a larger-scale green engineering approach (e.g., sediment addition) may be needed to slow erosion.

2.6 Avian and Benthic Invertebrate Response to Treatments (USGS Western Ecological Research Center, San Francisco Bay Estuary Field Station)

The USGS conducted avian and benthic invertebrate pre-treatment and post-treatment monitoring of waterbirds and benthic macroinvertebrates to help address project Objective 3*: Evaluate the use of restored habitats for wildlife, including invertebrates, fish, and birds*. Specific objectives for birds and benthic invertebrates were to 1) Assess abundance, community composition, and behavior of waterbirds using eelgrass and oyster bed restoration treatments in comparison to untreated controls, and 2) Evaluate how treatments may influence trophic structure and prey resources for migratory waterbirds.

Our overall design followed Before-After, Control-Impact (BACI; Stewart-Oaten et al. 1986, Stewart-Oaten 2003) methodology. We recorded bird numbers and locations within a 125 x 125-m grid system (Fig. 28), a spatially explicit design that allowed for birds to be counted across the entire site, including pre-defined treatment grid cells containing eelgrass and oyster installations, as well as control grid cells where no treatments were placed (Fig. 28). Each control grid cell was selected to be similar in depth and distance to shore as paired treatment grid cells.

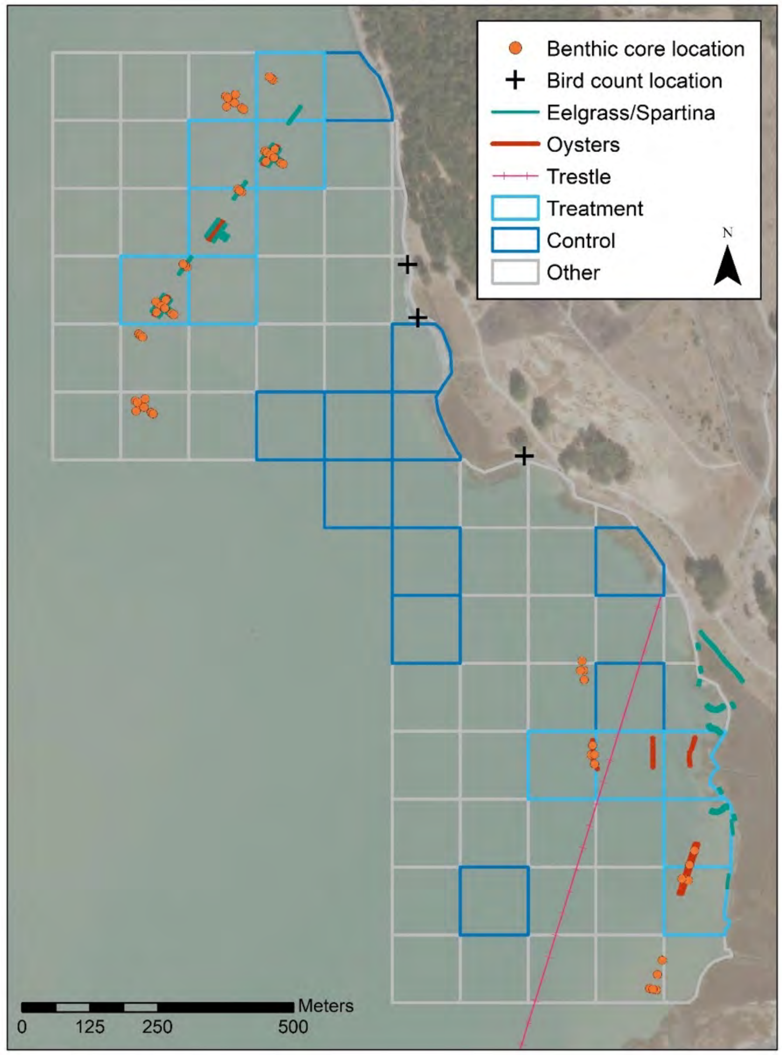


Figure 28. The Giant Marsh study site with 75, 125 x 125-m grid cells. Eleven treatment cells surrounded eelgrass, eelgrass/oyster, and oyster enhancements. Eleven control cells did not contain treatments but had similar water depths and distances from shore compared to the treatment cells. Benthic cores were collected to assess invertebrate abundance, and trained observers conducted high and low tide surveys for birds from fixed locations along the shore.

To compare abundance, trained observers used spotting scopes at fixed points for low and high-tide avian surveys monthly during pre-treatment from Jan to May 2017, and Oct 2017 to May 2018, quarterly during post-treatment from Nov 2019 to May 2021, and monthly during post-treatment from Jan to May 2023 and Oct to Dec 2023. High-tide surveys were conducted on a falling tide with a minimum allowable tide of 3.0 ft based on the Pinole Point tide station, and low-tide surveys began at a 1.0 ft tide to coincide with the start of shoreline exposure. We examined the effects on waterbird abundance of the eelgrass and eelgrass/oyster enhancements together in Treatment 1, and separately considered oyster enhancements at both Treatments 2 and 3 using BACI analyses of waterbird count data from treatment and control grid cells collected before and after treatments were installed.

After completion of abundance surveys, we collected information on behavior of birds, such as roosting and foraging. We used scan sampling (Altmann 1973, Borgerhoff Mulder and Caro 1985) to record the behavior of individuals of each species observed during abundance surveys (see details in Graham et al. 2024).

After each scan survey was completed, we conducted focal observations of foraging individuals from three target guilds: diving ducks, shorebirds, and wading birds. Foraging behavior, including number of dives (waterfowl) or steps (shorebirds) is linked to prey density and can provide information on prey availability for different species (Rocha et al. 2016, Jackson et al. 2024). Ten birds were randomly selected (using methods described above) from each guild in both treatment and control grid cells.

To determine if bird density or species richness on treatments was different during time periods not captured by observer surveys, and to evaluate avian use across the entire tidal cycle, we deployed field cameras quarterly at the south end of the nearshore reef (Treatment 3). Cameras were deployed for the two-week period surrounding observer-based avian surveys in May and November 2020, January, March, and May 2021, and April 2023. Images were captured during daylight hours at five-minute intervals beginning 15-mins prior to sunrise and ending 15-mins after sunset. Downloaded images were trimmed to include only those surrounding the period of the tidal cycle when the treatment was at or near exposure (≤3.8 ft at NOAA Point Pinole gauge). A bounding box was placed around the Treatment 3 nearshore reef area and every bird within the bounded area was enumerated and identified to species whenever possible. We worked with STEP-UP (Secondary Transition to Employment Program - USGS Partnership, a transition program for students with cognitive and other disabilities) students from Fremont Union High School District, who helped to determine presence and numbers of birds.

To evaluate how eelgrass, eelgrass/oyster, and oyster enhancements each separately (not combined as in bird surveys) may influence trophic structure and prey resources for fishes and migratory birds, we collected benthic invertebrate samples in fall 2017 and spring 2018 (pre-treatment), and in fall 2019, fall 2020, spring 2023, and fall 2023 (post-treatment). We collected 604 benthic core samples (10-cm deep, 10-cm diameter) in treatments (eelgrass, eelgrass/oyster and oyster) and localized control areas to measure benthic invertebrate density, diversity, and biomass. Sampling locations were arranged at 5-m intervals and aligned such that they bisected treatment plots (Fig. 28). Localized control areas were randomly chosen from a subset of grid cells identified to be of similar depth and have similar eelgrass coverage as the treatment grid cells; note that vegetated controls were used instead of no-eelgrass areas due to natural recruitment of eelgrass around Treatment 1. Core samples were rinsed through a 0.5-mm sieve and preserved in 70% ethanol with Rose Bengal tissue dye. Invertebrates were identified to the lowest possible taxonomic class, enumerated, measured, and weighed (dry biomass). Ash-free dry weight was determined using published conversion factors (Ricciardi and Bourget 1998).

We found that across the entire survey area, waterbird species richness increased between the pre- and post-treatment period, with a 21% increase in species observed at high tide and an 18% increase at low tide. Overall, we observed 55 species of waterbirds from 12 guilds in the survey area during the 2017 – 2023 study period.

BACI analyses indicated there was no change overall in waterbird abundance as a result of the installation of Treatment 1, with eelgrass and mixed eelgrass and oyster/habitat reef areas taken together. However, we did detect a difference earlier in the project (2020) when a statistically significant interaction between grid cell type and treatment period indicated that eelgrass and mixed eelgrass and oyster/habitat areas had a positive effect on waterbird abundance at low tides.

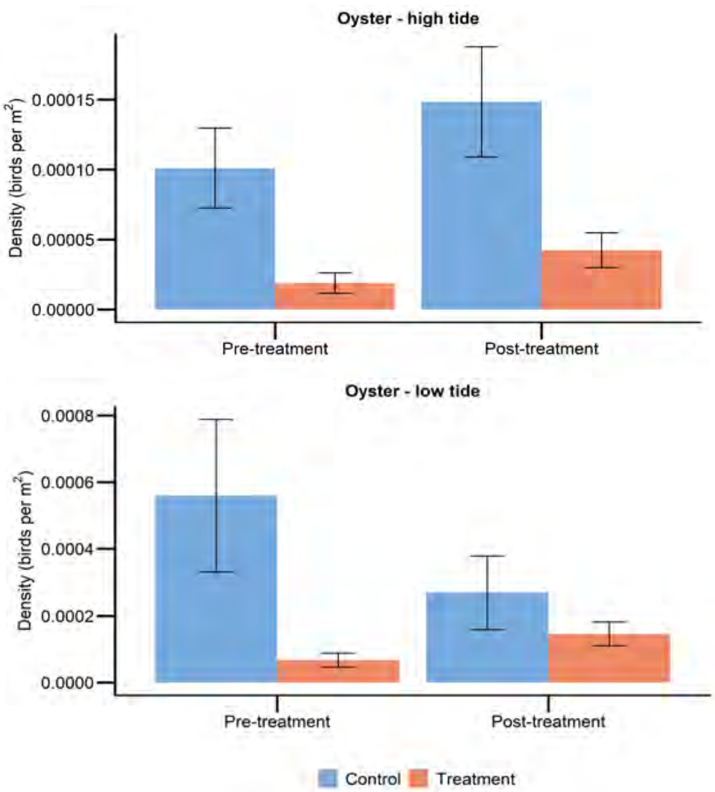


Figure 29. Mean waterbird densities (± SE) in paired control and treatment grid cells for the Treatment 2 and 3 oyster/habitat reef additions at (a) high and (b) low tides before (pre-treatment) and after (post-treatment) installation of reefs.

Waterbird abundance at low tide increased significantly after the installation of Treatment 2 and 3 oyster/habitat reefs but did not change at high tide (Fig. 29). Based on instantaneous behavioral scan data from waterfowl and shorebirds, there was no change in the proportion of individuals engaged in different behaviors (foraging, movement, sleep, or comfort) between the pre- and post-treatment installation periods.

Focal behavioral observations of foraging individuals from specific species showed that ruddy ducks spent significantly more time foraging, and less time engaged in other behaviors (alert, comfort, drinking, sleeping, preening, resting, and social) during the post-treatment as compared to the pre-treatment period (Fig. 30). Time spent on foraging and other behaviors did not differ significantly between the pre- and post-treatment periods for greater and lesser scaup (Fig. 30) and willets (Graham et al. 2024).

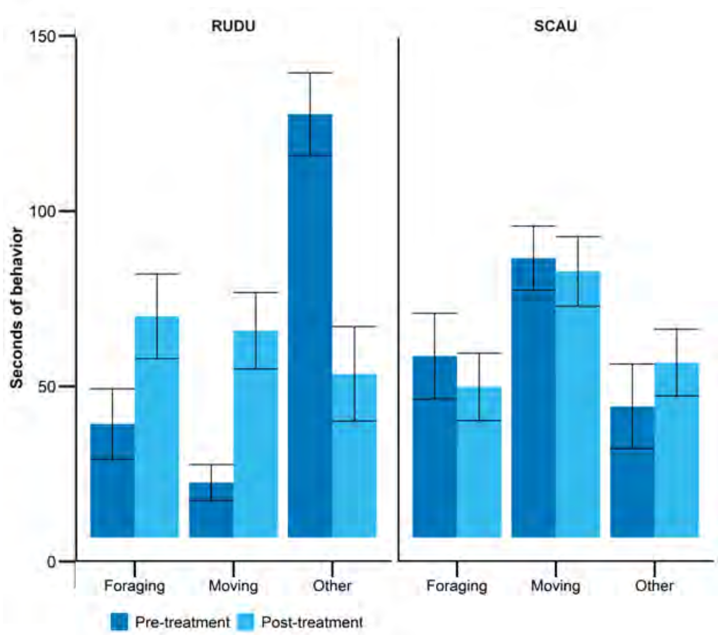


Figure 30. Mean duration in seconds (± standard error) of three behaviors (foraging [diving], moving, and other) exhibited by ruddy duck (RUDU), and greater and lesser scaup (SCAU) during the pre- and post-treatment periods in Giant Marsh study area at low tide.

Remote field camera data suggested avian use of the Treatment 3 nearshore reef varied by species guild and tide height (Fig. 31). Dabbling ducks were most abundant when the treatment was partially exposed (1.6 – 3’), whereas diving ducks were most abundant when the treatment was nearly inundated (3.3 – 3.6’). Gulls were common throughout the tidal cycle but generally increased in abundance as the tide height increased. Large and small shorebirds were both most abundant in images when the tideline ebbed or flowed across the treatment (1 – 2.3’), with large shorebirds displaying species-specific differences in use of the treatment area, particularly during winter.

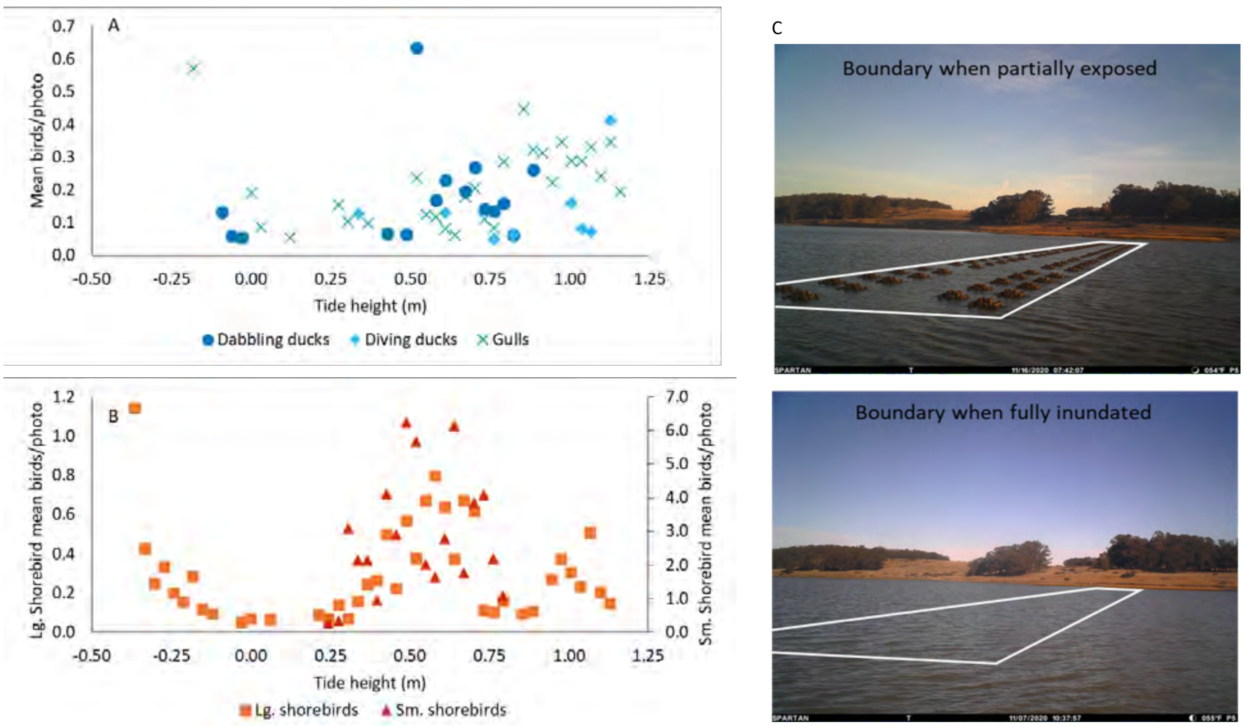


Figure 31. Mean number of birds observed per photo by tide height (m) for each of the five most abundant guilds observed at the Treatment 3 nearshore reef. A) Dabbling ducks – dark blue circles; Diving ducks – blue diamonds; Gulls – blue X. B). Large shorebirds (orange squares) are plotted on the left Y-axis, and small shorebirds (red triangles) on the right Y-axis. Note: Y-axes differ in scale. C) A view of the Treatment 3 nearshore reef when partially exposed versus fully inundated.

We identified 77 unique invertebrate taxa in benthic cores collected across all treatment types during the 2017 – 2023 study period. Overall, we did not detect a significant effect of any treatment type (eelgrass, eelgrass/oyster, oyster) on total invertebrate density or biomass. We did observe significant changes in invertebrate density and biomass in both control and treatment cores between study periods (pre- and post-treatment installation), which may have been driven by changes in regional abiotic conditions such as salinity and other factors.

Taken together, our results suggest that treatments installed at the Giant Marsh Living Shoreline site may be providing enhanced habitat for some waterbird taxa. This may be particularly true for oyster enhancements at Treatment 3, where we saw a significant increase in waterbird abundance during low tide. Waterbird species and guild use of treatment elements varied by tide height and season, highlighting the importance of considering these factors when designing and evaluating avian response to living shoreline enhancements.

2.7 **Physical Processes** (ESA)

Over the monitoring period, ESA collected water quality and physical processes data, including wave measurements, ground-level photo documentation of shoreline conditions, topobathymetric survey data, and high-resolution aerial imagery of site conditions (Table 8; ESA 2024). Monitoring activities were specific to the following treatments (See Figs. 2, 3, 4, and 32):

- Treatment 1: Offshore eelgrass and oyster/habitat reef
- Treatment 2: Oyster habitat reefs at a range of elevations, and
- Treatment 3: Nearshore oyster/habitat reef near the marsh edge

Table 8. Physical processes monitoring dates by type of monitoring activity.

Monitoring Activity	Pre-Construction	Year 1 (2019)	Monitoring Date(s)		
			Year 2 (2020)	**Year 4 (2022)**	**Year 5 (2023)**
Water Quality		6/28/19 – 12/31/19	1/1/20 – 8/10/20	11/30/22 – 12/31/22	1/1/23 – 12/31/2023
Wave Monitoring – Sonic		8/6/19 – 8/21/19 8/21/19 – 9/14/19 9/30/19 – 10/11/19 1/17/20 – 1/20/20 2/28/20 – 3/25/20 3/31/20 – 4/7/20 4/22/20 – 5/11/20			
Wave Monitoring – Video		8/6/19 – 8/10/19 8/21/19 – 8/29/19 1/17/20 – 1/29/20			
Wave Monitoring – Photo		6/18/19 – 6/21/19 7/6/19 – 7/9/19 8/6/19 – 8/9/19 8/21/19 – 8/22/19 9/30/19 – 10/4/19 1/17/20 – 2/2/20 2/18/20 – 3/20/20 4/22/20 – 5/11/20			
Bathymetric Survey	3/18/19 – 3/19/19	10/28/19 – 10/29/19			10/30/23 – 10/31/23
Topographic Survey	9/19/16 – 9/21/16	10/25/19			10/25/23
Drone Ortho-Photo	9/19/16	12/9/19		1/19/23[1]	10/25/23
Drone Elevation Survey	9/19/16	12/9/19		1/19/23[1]	10/25/23
Flight Times	9:15 – 9:45 AM	2:39 – 3:30 PM		4:00 – 5:00 PM	4:30 – 5:30 PM
Photo Documentation		Station A-NE: 6/19/19 Station A-NW: 1/17/20 Station B: 1/22/20 Station C: n/a		All Stations: 1/19/23[1]	All Stations: 10/25/23

NOTES:
[1] Drone orthophotography and related photo documentation for Year 4 (2022) were delayed to January 2023 due to weather conditions.
[2] Predicted tides provided by the National Oceanographic and Atmospheric Administration (NOAA) Point Pinole tide station (station ID: 9415056) (NOAA, 2023).

Water Quality: The purpose of the water quality data collection was to provide ambient water quality conditions at the Giant Marsh LSP site to inform biological monitoring results by others on the project team. ESA deployed a multi-parameter sonde adjacent to the project site to collect ambient water properties (temperature, salinity, dissolved oxygen (DO), pH, and turbidity) of biological relevance from June 28, 2019, through August 10, 2020, and from December 1 to December 31, 2023. The sonde was deployed directly adjacent to the easternmost oyster reef of the Treatment 1 Offshore Eelgrass Bed and Oyster Reef at an elevation of approximately 0.87' NAVD88, one foot above the seabed. For all years monitored, water quality data collected at Giant Marsh Treatment 1 were generally in good agreement with other data collected in San Pablo Bay at the National Estuarine Research Reserve, China Camp station (Figs. 33 and 34).

Figure 32. Physical processes instrument deployment locations.

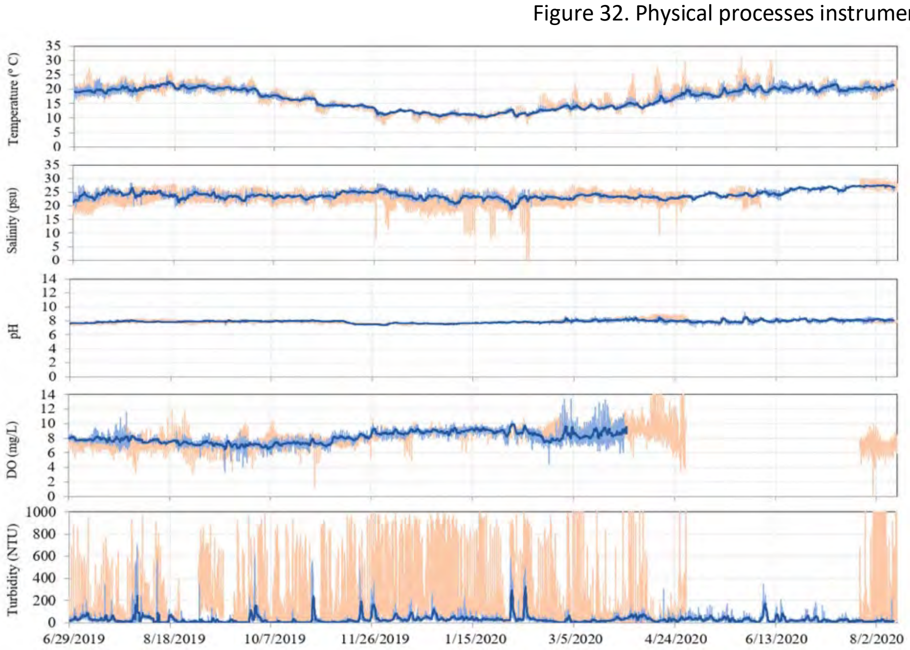


Figure 33. Water quality data from the sonde near Treatment 1 in blue (dark blue line = 24-hr moving average), late-June 2019 to early-August 2020. Orange points represent data from across San Pablo Bay at the China Camp NERR sonde, for comparison.

Water quality parameters including temperature, salinity, pH, DO, and turbidity showed little fluctuation in the monitoring period between June 2019 and August 2020 (Fig. 33), indicating a strong marine influence and generally good conditions for organisms of interest, particularly Olympia oysters and eelgrass. Apart from two occasions during January 2020, salinity remained above 20 practical salinity units (PSU), and only very minor seasonal freshening was observed during the winter months. Salinities reached as high as 27 PSU in August. pH levels remained between 7 and 9 throughout the year. pH levels dropped slightly during the winter months before steadily climbing back up throughout spring and summer. Turbidity generally was below 200 NTU throughout the monitoring year, with periodic spikes that reached above 500 NTU. These spikes, in large part, coincide with nearby high wind events measured at the California Irrigation Management Information System (CIMIS) Point San Pedro weather station (CIMIS 2020; Fig. 35).

In contrast, a series of atmospheric rivers from late December 2022 until mid-January 2023 and others in March 2023 made water year 2023 one of the wettest in recent history. Following the mid-December and January storm systems, salinity was below 10 ppt for approximately 19 days in the latter half of January (Fig. 34). This same freshening phenomenon can be observed in March/April and May after the onset of storms, though average salinity values were not as quick to recover to antecedent conditions after May. Due to the abundant snowpack generated from the winter storms, there was a moderately paced melt-out in the Sierra Nevada (CDWR, 2023). The effects of the steady flow of fresh water into the San Francisco Bay are prominent through early July where average salinity values remained below 20 ppt (Fig. 34). Presumably related to these spikes and extended low salinity, oyster monitoring by the Smithsonian showed dramatic declines in oyster numbers following very high densities the previous year (Figs. 8, 9, Tables 1, 2, 4). The oyster team also noted periods of high temperature following the fresher periods according to temperature loggers at the Treatment 2 reefs (Zabin and Blumenthal 2024); these differences were not picked up by the sonde deployed in deeper water at Treatment 1 (Fig. 34), suggesting that declines in oysters in the Treatment 1 offshore reefs in 2023 may have been more related to low salinity than to high temperature. The difference in temperature results between Treatment 1 and Treatment 2 also highlights the need to monitor temperature at each reef depth in future projects.

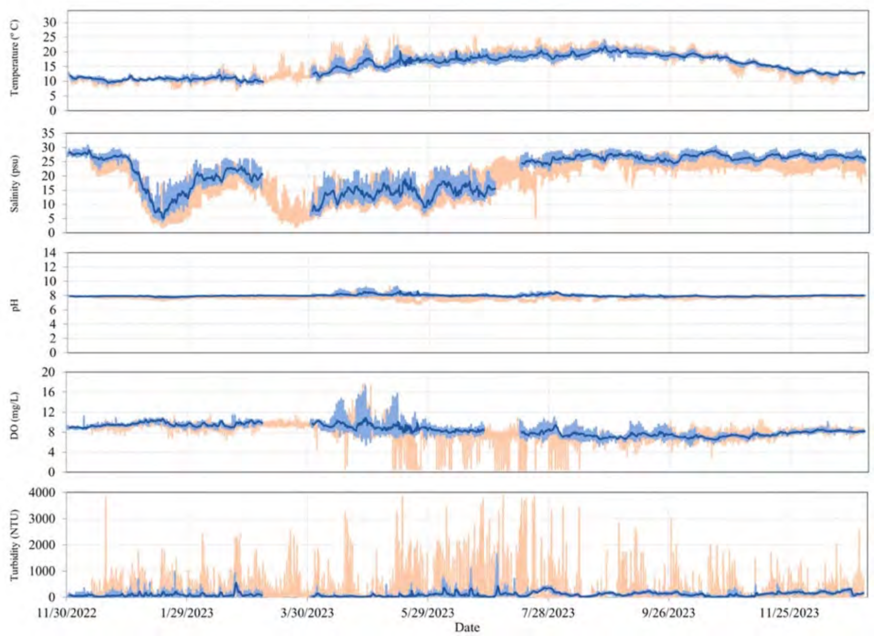


Figure 34. Water quality data from the sonde near Treatment 1 in blue (dark blue line = 24-hr moving average), late-November 2022 to late December 2023. Orange points represent data from across San Pablo Bay at the China Camp NERR sonde, for comparison.

It is notable that turbidity was much lower than recorded by the China Camp sonde for both periods (Fig. 33, 34). This is likely due in large part to the low height of the China Camp sonde above the mudflat but could also be due to geographic location differences, local substrate differences, and/or density of eelgrass in the vicinity of Giant Marsh, which tends to reduce sediment resuspension.
The highest observed turbidity at Giant Marsh occurred during the 2023 monitoring period, where values were generally below 400 NTU throughout the year with some values spiking between 800 and

1,700 NTU. The highest overall turbidity measured was 1,641 NTU on July 2, 2023. In 2023, observed spikes in turbidity did not appear to coincide with nearby high wind events measured at the CIMIS Point San Pedro weather station (Fig. 35). In comparison to previous years of data collection, turbidity at Giant Marsh in 2023 remained relatively elevated from May through the end of the year. This sustained increase in turbidity could have been due to a greater amount of suspended solids delivered to the San Francisco Bay during the significant precipitation water year.

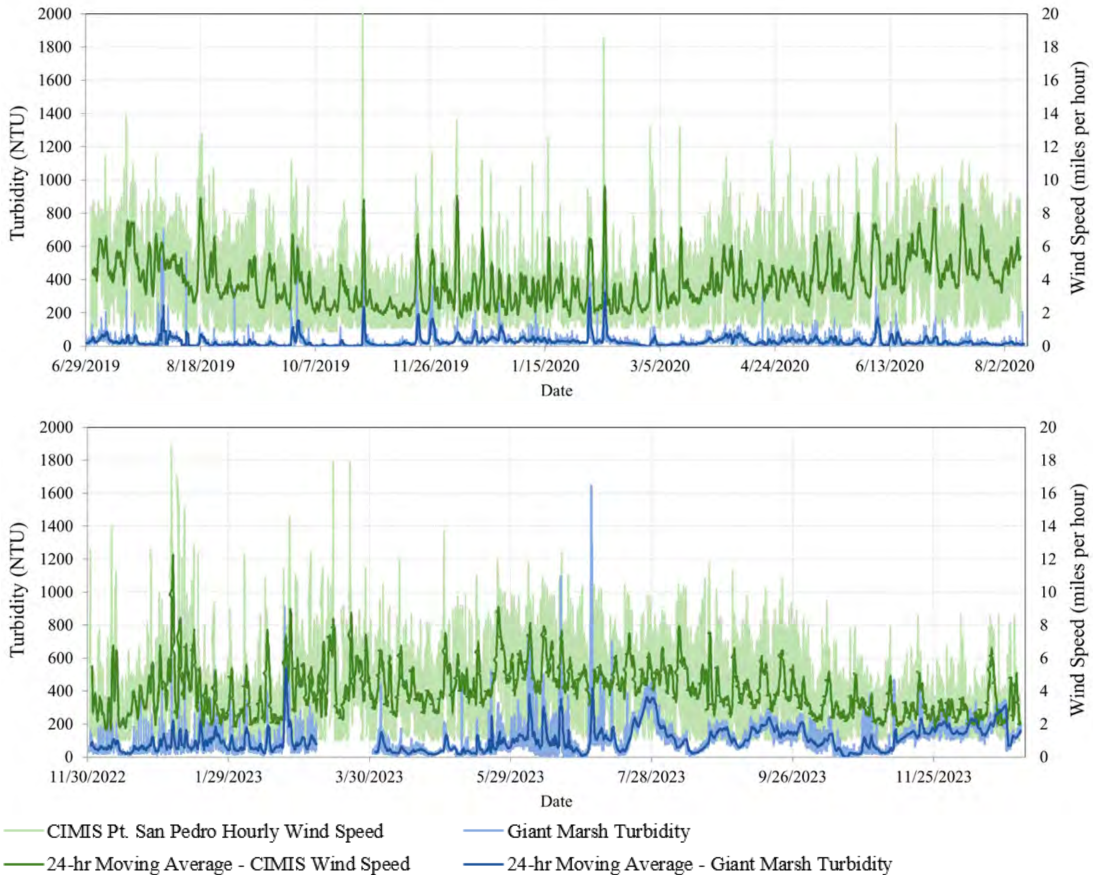


Figure 35. Turbidity measured by the sonde at Giant Marsh Treatment 1 in comparison to wind speed measured at the CIMIS Pt. San Pedro station.

Wave Monitoring: ESA developed protocols for and deployed both sonic and image-based wave monitoring equipment in the vicinity of the near-shore Treatment 3 reef structure in order to quantify the ability of the reef to dissipate (attenuate) wave energy (Fig. 32). Key wave parameters such as wave energy and wave height were compared between sensors to quantify wave dissipation due to the oyster reef. Additionally, a numerical wave model was used to assess the effectiveness of the Nearshore Oyster Reef in attenuating wave energy for a variety of water levels and conditions. Wave dissipation of approximately 30% was observed when water levels were around the center of the Nearshore Oyster Reef elements or lower (Fig. 36). Wave dissipation declined to negligible compared to the control with increasing water depth. The top of the Treatment 3 nearshore reef elements sits approximately 1.5 feet above their underlying bed and no wave dissipation benefits were observed for water depths greater than approximately 2.5 feet, or approximately 1.7 times the height of the reef. When the findings of the wave measurements are extrapolated to a wind-wave hindcast record from 2012 to 2023, the analysis estimates that the reef will dissipate approximately 10 percent of the total wave energy, per year. Is important to note that most of this wave energy dissipation will occur during low to mid tides. Oyster reefs would be most effective at dissipating wave energy with their crests placed at or above mean higher-high water (MHHW) where oysters do not thrive. Therefore, based on the Giant Marsh oyster reef design, the location most beneficial to wave dissipation in San Francisco Bay is not necessarily the location that is most suitable for oyster habitat.

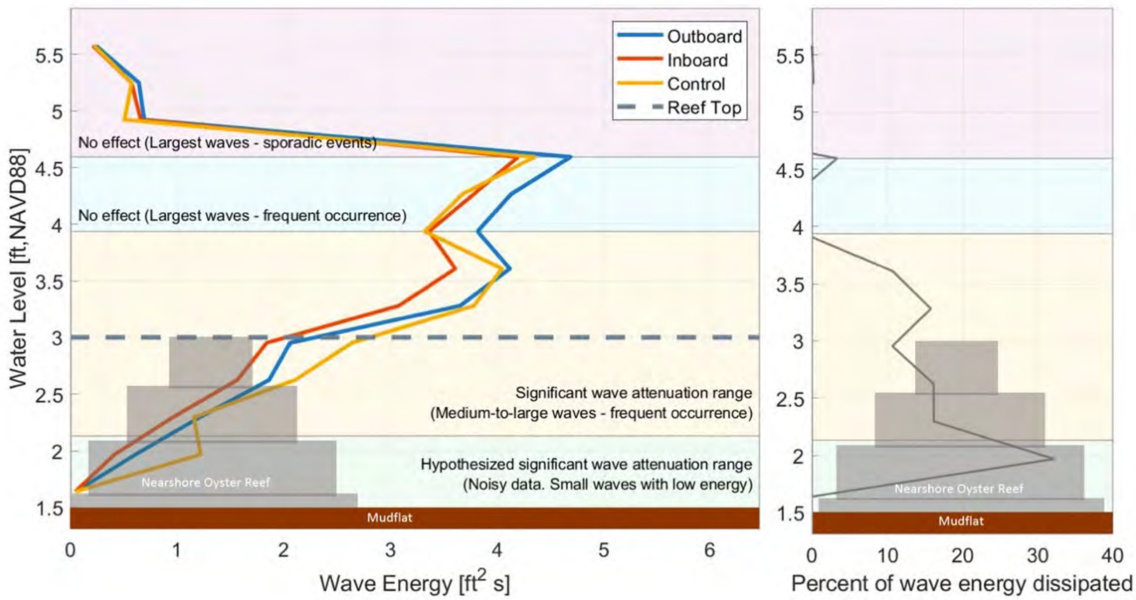


Figure 36. Wave energy distribution and nearshore oyster reef wave dissipation potential as a function of water depth outboard and inboard of Treatment 3 reef and compared to control sensors.

Survey and Basemap: Site elevations were measured using a combination of bathymetric surveys (below water) and drone-based surveys (above water). ESA combined the bathymetric and drone-based surfaces together to form one site-wide mixed Digital Elevation Map (DEM) surface and used a ground-based elevation survey to QA/QC the combined surface results. Ortho-rectified high-resolution drone-based imagery from 2016 (pre-construction), 2019 (Year 1), 2022 (Year 4), and 2023 (Year 5) were created (ESA 2024). Elevation survey data were also used to evaluate vertical movement of the Treatment 1 offshore reefs and the Treatment 3 nearshore reefs. The elements in both reefs remained stable and upright in the first five years following implementation, and perhaps even grew in height slightly due to barnacle and/or oyster growth. Subsidence was different between the two reefs. The Treatment 1 offshore reef elements subsided by an average of 0.4 feet (12 cm) in the five years while the immediately adjacent mud they were placed in subsided by an average of 0.6 feet (18 cm); both the Treatment 3 nearshore reef elements and the sandier (compared to Treatment 1) substrate around them subsided by an average of 0 feet (no subsidence).

Shoreline Change Analysis: Based on regional studies conducted by SFEI (2015 and 2020), ESA digitized the shoreline for 2019, 2022, and 2023 using the high-resolution imagery captured for this project. The complete set of shoreline data, which stretches back to 1993, was used to compute shoreline change over time at a set of shore-perpendicular transects. Over the entire project shoreline, typical marsh erosion rates pre-project were about 1.5-5 feet per year. Figure 37 shows the vertical and lateral change behind the two reefs by combining a raster showing the 2022-2023 elevation change, and analysis transects showing the difference between post-project (2023) and pre-project (2019) erosion rates. Though it is too early in site evolution to assess shoreline change with certainty, early monitoring shows reduced marsh edge erosion in areas landward (in the lee of) the Treatment 2 mid-shore and Treatment 3 nearshore reefs compared to areas further from the reefs. Landward of Treatment 2, a pre-project trend of marsh erosion changed to a trend of marsh progradation in the years after the project was implemented. Landward of the Treatment 3 nearshore reef, marsh erosion rates were slightly lower (dark blue band thinner in Fig. 37) directly in the wave shadow of the reef compared to areas immediately north and south (see details in ESA 2024).

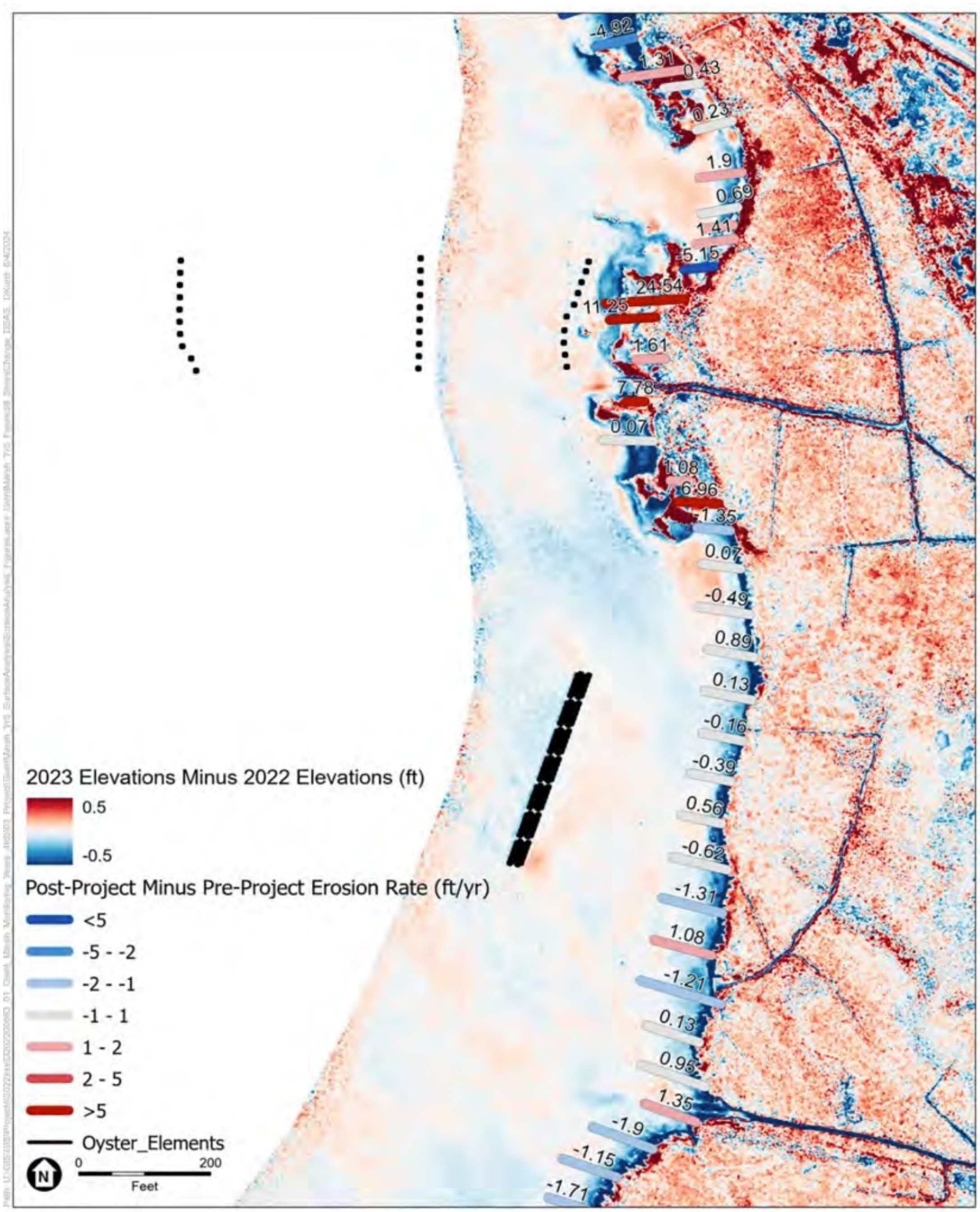


Figure 37. Shoreline and elevation change analyses along the shoreline in the vicinity of the Treatment 2 and Treatment 3 reefs. Field color indicates accretion (red) or erosion (blue) between 2022 and 2023. Bars showing accretion/erosion along the shoreline edge indicate change between 2019 (pre-project) and 2023 (5 years post-installation).

Spatial Patterns of Deposition and Erosion: The basemaps that emerged from the survey and basemap task were used to assess the change in elevation over time at the site by subtracting elevations in one map from another. This analysis indicated that the mudflats are generally accreting over time, on the order of 0-0.2 feet between 2019 and 2023. Sediment accretion (red coloration) was observed in the lee of the Treatment 3 nearshore reef (Fig. 37 and see ESA 2024), consistent with the hypothesis that the reef attenuates wave energy. Higher mudflats can, in turn, further attenuate wave energy at the marsh edge.

3 Findings Relative to Objectives and Success Criteria

Here we assess accomplishments of the Giant Marsh Living Shorelines Project relative to our five objectives:

Objective 1: Create or enhance a variety of habitats ranging from the shallow subtidal to the tidal marsh to the estuarine-terrestrial transition zone.

The Giant Marsh Living Shorelines Project featured seven treatments across a range of elevations, including oyster/habitat reefs, eelgrass beds, cordgrass meadows, beach/marsh edge vegetation, and terrestrial transition vegetation. In all cases we satisfied Objective 1 by creating or enhancing these foundational habitats, as follows:

- Installing oyster/habitat reefs greatly increased the native Olympia oyster population along the Point Pinole/San Pablo shoreline, even with the substantial variation seen between years of high (2023) and low (2022) rainfall (320,000 – 1.3M total oysters present on the Treatment 1 offshore Reef Balls topped with shell bags, 1700 – 36,000 on the Treatment 2 oyster blocks examining the effects of depth, and 600 - 21,000 on the Treatment 3 nearshore oyster block reef). Despite reduced oyster densities in 2023, Treatment 2 reefs still supported higher densities than the reference natural shoreline or old pier pilings at the same tidal elevations, and oyster densities on the Treatment 1 Reef Balls were slightly higher than those at a similar elevation at the reference site. In general, oysters at higher elevations had more exposure to air and high temperatures during low tides; however, even the lower oyster densities at the Treatment 3 nearshore reef increased the local oyster population by tens of thousands during drier years. The oyster/habitat reefs supported a range of oyster sizes each year, indicating that they are growing and also consistently reproducing on the reefs, and this should benefit the population in the Point Pinole region well beyond the project footprint and over time. Very high oyster numbers in 2022 likely served as a major source of propagules outside the project area.
- Eelgrass became well established in the Treatment 1 plantings, with evidence of the reefs benefitting densities and total shoot counts (see Objective 2, below). As with oysters, there was interannual variability in eelgrass measurements; however, densities still more than doubled from initial plantings as of summer 2023, with some planted patches exceeding 1000 shoots.
- Though delayed in their installation until 2022, cordgrass plantings via sods in both Treatments 4 and 5 established and showed evidence of spread through rhizome extension.
- Along the beach and marsh edge, CA sea-blite was difficult to maintain through a series of storms that shifted sediments, but surviving plants represent a new location (the fifth) in its reintroduction to San Francisco Bay. We incorporated planting methods that increased sea-blite size as well as that of existing pickleweed, which should be beneficial to mammals and birds seeking cover and canopy height during high tide flooding.
- Terrestrial transition zone plantings established variably but with very high survivorship in the upland meadow, which appeared to benefit from weed control, signage, and fencing, and moderately high survivorship at the high marsh edge.

Objective 2: Experimentally evaluate techniques to advance restoration practice for each of these habitat types.

Embedding trials and comparisons of techniques in this project led to numerous findings that can guide future restoration of the foundational habitats we restored. These findings include:

- Reefs placed at a wide range of depths supported recruitment of native Olympia oysters, but greatest abundances occurred within the range of approximately +0.4 to -0.7’ NAVD88 (+10 to -

20 cm MLLW). Higher elevations in that range fared better in most years, but those slightly lower did better in the high rainfall year of 2023.

- Reef Balls comprising the Treatment 1 offshore reefs, and oyster blocks installed in the Treatment 2 and Treatment 3 reefs nearer to shore, were all structurally sound and showed no signs of degrading, moving, or listing. Reef Balls in soft sediments at Treatment 1 subsided by about 0.4 feet (measured at their tops), slightly less than sediments around the Reef Balls (0.6 feet subsidence) that may represent some localized scouring around the structures. Oyster blocks on sandier sediments at the Treatment 3 nearshore reef did not show subsidence or evidence of scour.
- Although only the top 10-20 cm of the Treatment 1 Reef Balls could be monitored during low tides (thus densities are likely somewhat underestimated), the attached shell bags supported much greater numbers of oysters (at least one order of magnitude in each of the years), presumably through higher availability of interstitial space.
- There was a trend of higher oyster densities on the north sides of structures in 2020, which became statistically significant in 2021-2022. In 2023, likely due to low numbers of oysters, there was no difference by aspect.
- While it was difficult to establish *Fucus* on the oyster reefs (Treatment 2), this first attempt to include an alga in a living shorelines project in San Francisco Bay led to increased densities of both Olympia oysters and barnacles, an effect correlated with cooler temperatures and higher humidity under the canopy.
- It was notable that water was retained in the lower tier of some Treatment 3 oyster blocks, creating tide pool-like habitat in which oysters settled in small clusters of two and three individuals unattached to the substrate. Thus, even high intertidal structures could potentially be engineered to encourage colonization of oysters through the inclusion of moisture-retaining and shade-creating features.
- Planting eelgrass at medium or high densities (20 or 40 shoots per square meter) produced more shoots than the lower densities typically used in eelgrass restoration projects in San Francisco Bay (10 shoots per square meter), but increases were not proportional to numbers of shoots required to plant at the higher densities.
- There was evidence that eelgrass planted on the shore side of oyster reefs grew more vigorously, as found in the previous San Rafael living shoreline project. Reef presence was especially important to eelgrass planted at the lowest density.
- Planting cordgrass as sods, with an intact sediment layer (~0.3m deep) was far more successful than via bareroot plugs (even those plugs planting into soil in a burlap bag).
- Additions of wrack-line debris to planting holes increased the size of CA sea-blite.
- Arboring increased the size of both sea-blite plantings and existing stands of pickleweed.
- Controlling weeds and adding fencing and signage may have aided in success of transition zone plantings relative to plantings at the marsh slope/scarp and high marsh edge.

Objective 3: Assist recovery of particular species of concern, including Pacific cordgrass, eelgrass, Olympia oysters, and endangered species such as California sea-blite, California Ridgway's rail, and salt marsh harvest mouse.

Pacific cordgrass, eelgrass, Olympia oysters, and the endangered California sea-blite all established through the treatments incorporated at the project site. Ultimately, we expect these species, as well as the arbored pickleweed, to enhance habitat for wildlife including endangered species, although evaluating use by Ridgway's rail or salt marsh harvest mouse was out of the scope of this project.

Objective 4: Evaluate the use of restored habitats for wildlife, including invertebrates, fish, and birds.

We documented a range of benefits of the restored foundational habitats for associated species:

- In addition to native oyster recruitment, we noted 28 sessile and small mobile taxa on the oyster/habitat reefs, compared to only seven present on existing pier pilings at the site. These taxa included a multi-canopy layer of seaweeds which had developed by 2022 on the Reef Balls in Treatment 1, and juvenile rock crabs and fish, which were present in the shell bags each year. In 2023, despite declines in many species with extended low salinity, the shell bags continued to provide habitat for small mobile taxa, including important native and fisheries species; there were notable additions of small Dungeness crab (*Metacarcinus magister*) as well as saddleback gunnels (*Pholis ornata*). Also in 2023, for the first time, we noted several instances in which mussel byssal threads were holding together clumps of both live and dead oysters, creating a matrix of oysters and mussels. Community shifts following extreme events are expected, and the species and abundances found over the monitoring period are a good representation of long-term expectations over years of varying conditions.
- Invertebrate communities, including both epifaunal invertebrates and large mobile invertebrates, benefitted from oyster reefs, with species richness, diversity, and abundance often higher in reef-associated eelgrass compared to in eelgrass growing alone.
- Adaptive Resolution Imaging Sonar (ARIS) showed a trend of greater catch per unit area in eelgrass associated with reefs compared to restored eelgrass alone.
- Though we did not find an overall change in waterbird abundance with the offshore reef/eelgrass treatments, the shallower reefs (Treatments 2 and 3) showed increased use at low tides, and dabbling ducks, diving ducks, and both large and small shorebirds frequented the nearshore reef (Treatment 3).
- It is notable that while restoring and creating habitat by increasing native species presence and coverage, we did not see an increase in non-native species beyond background levels present within the project area or in nearby reference sites.

Objective 5: Evaluate the efficacy of nearshore restoration treatments in attenuating wave energy and reducing shoreline erosion.

We showed that the Treatment 3 nearshore reef's effects on wave dissipation may have a beneficial effect on mudflat sedimentation rates in their lee, thereby further attenuating wave energy at the marsh edge. Mudflat elevation change is episodic and can be inconclusive at five-year time scales. Conclusive results will require additional data collection over a longer time period. The same is true for the nearshore oyster treatment's effects on reducing shoreline erosion, which indicated potential benefits; i.e, evidence of slowed erosion of the marsh scarp directly in the lee of the Treatment 3 nearshore oyster reef relative to pre-project and adjacent shoreline areas. To verify the nearshore oyster treatment's effects on reducing shoreline erosion and increasing mudflat elevations, measurements are needed over a longer time period.

Overarching objective and success criteria

We met our overarching objective to conduct scientific experimentation on best techniques for living shoreline approaches through pilot-scale oyster, eelgrass, marsh, and estuarine-terrestrial transition zone treatments. We also met our success criteria of achieving one or more of the following within the 5-year period after construction as follows:

- *Native oysters will recruit, with densities greater than 10 adult oysters per square meter of substrate at the offshore oyster reef.*

  Despite limited oyster recruitment in summer 2019, this milestone was far exceeded by summer of the second project year (2020) and continued through the five-year project period, with 200-520 oysters/m^2 on the Reef Balls.
- *Invertebrate species richness will increase by 15% relative to control plots with no physical structure and initial measures collected prior to construction.*

  Although there were strong interannual shifts in dominant species of invertebrates associated with eelgrass, the success criterion of invertebrate species richness increasing by 15% was met for both epifauna and large mobile invertebrate species relative to the natural reference Point San Pablo bed. Invertebrate species richness also increased on the reefs relative to pier pilings at the site.
- *The number of visits by fish species to the offshore eelgrass and oyster reefs will increase by 50%, relative to control areas with no physical structure.*

  We included multiple methods of fish sampling, finding all had pros and cons with regard to documenting abundance or species richness resulting from the reef and eelgrass installations in Treatment 1. No methods found a greater catch per unit effort (CPUE) in the treatment areas than in bare mudflat, thus we were not able to demonstrate a 50% increase in fish visits relative to control areas with no physical structure. Overall, we see fish sampling as an ongoing challenge in the complex structured habitats in living shorelines projects.
- *Eelgrass will establish and spread to at least twice initial planting densities.*

  Eelgrass planted in May and June 2019 generally established well, although losses of plants at the southern-most reef of Treatment 1, which had been planted a few days after a June 2019 heatwave, led us to replant that eelgrass in spring 2021. The success criterion that eelgrass achieve at least twice initial densities was met, with densities more than doubling across the site by summer 2023.
- *Shoreline erosion immediately landward of the nearshore reef (Treatment 3) will not exceed erosion rates at adjacent unsheltered areas.*

  Marsh scarp erosion continued or increased over time, but marsh edge erosion rates were slightly lower directly in the wave shadow of the Treatment 3 nearshore reef compared to areas immediately north and south. And, although this criterion did not refer to Treatment 2, a pre-project trend of marsh erosion changed to a trend of marsh progradation landward of Treatment 2 reefs over the project period.
- *Offshore reef provides wave attenuation in landward eelgrass planting areas for mean sea level tide conditions, compared to control areas with only eelgrass plantings or mudflats.*

  Measures of relative water motion via plaster block dissolution in winter 2020 showed that flow was reduced by 36% immediately shoreward of the reefs, declining to 17% and 6% by 8 and 16 m shoreward of the reef, respectively. Though eelgrass abundance and density were variable by year, there was evidence that this flow attenuation by the reefs benefitted eelgrass densities and facilitated large patches, even 20 m shoreward of reefs.
- *Pacific cordgrass plantings will establish, spread to at least twice initial planting densities, and expand into previously unvegetated areas.*

  Although cordgrass plantings did not occur until early 2022, establishment using sods succeeded very well in Treatment 4 and patches were noted to be spreading slowly as of fall 2024 after two growing seasons. In all but one of the blocks the area covered by cordgrass originating from sods had more than doubled, with a mean 128% increase in area covered across all six blocks; hence, the success criterion was met for sods in Treatment 4. Further, in Treatment 5, existing cordgrass and planted cordgrass sods in two of the three bands planted could no longer be distinguished after two seasons post-transplant. This area filled in, plus lateral spread from the sods, was estimated to be ~45 m^2 in these two locations, thus indicating that the sod plantings enhanced the cordgrass at the site overall.

- *California sea-blite will establish in at least three planting areas, thus adding Giant Marsh as a new reintroduction site for San Francisco Bay in the recovery of this species.*
  Although there were significant losses of planted sea-blite during storms over the course of the project, we met this criterion starting in 2019 and continued to do so through the five-year monitoring period. Further, evidence of self-seeding suggests that the population could be self-sustaining, particularly in the more protected planting locations (e.g., shoreward of a marsh fringe).

Overall, our major findings through the fifth project year (2023) were:

- In Treatment 1, offshore combination reef balls topped with oyster shell bags were successful at recruiting abundant native Olympia oysters, although low recruitment in the first year, 2019, and the final year, 2023, point to the importance of interannual variation in salinity and temperature on oyster densities. This variability in native oyster abundance suggests the importance of installing reefs in multiple places in the bay to spread risk, and that natural recruitment might be supplemented in future projects with laboratory-reared spat.
- Providing suitable substrate can quickly lead to valuable habitat aside from Olympia oysters, as the offshore reefs supported 28 mobile and sessile taxa within two years.
- Planting eelgrass in the cooler spring months is advisable but does not always avoid heat waves that come sporadically at other times, often coinciding with low tides (early June 2019).
- High-density eelgrass plantings produced more shoots but not proportionately more shoots, indicating that the benefits of planting densely may not balance favorably against the increased effort and resources needed to do so.
- Planting eelgrass adjacent to oyster reefs is beneficial, as evidenced by greater eelgrass abundance on the shore side of oyster reefs and greater numbers of mobile invertebrates in eelgrass when adjacent to reefs.
- Nearer to shore (Treatment 2), the highest oyster densities (and largest oysters) were found on the deepest oyster blocks (-0.10’ NAVD88) and on the deepest part of those blocks, probably due to less exposure to high air temperatures during low tides.
- Oyster numbers were higher on the north sides of the oyster blocks in most years, suggesting benefits of cooler temperatures.
- Transplanting Pacific rockweed to the oyster blocks resulted in low survival but proved favorable to both Olympia oysters and barnacles while the alga was present.
- At the Treatment 3 nearshore reef at ~1.5’ NAVD88, native oysters recruited to the lower portion of oyster blocks at numbers intermediate to the Treatment 2 high (1.9’) and mid-elevation (0.93’) blocks, as expected.
- The Treatment 3 nearshore reef was most effective at reducing wave energy at mid-tide elevations of 3.6-5.5’ NAVD88.
- There was no overall change in waterbird abundance with the offshore reef/eelgrass treatments, despite early evidence (2020) of increased abundance with these treatments.
- The shallower reefs (Treatments 2 and 3) showed increased waterbird use at low tides, and dabbling ducks, diving ducks, and both large and small shorebirds frequented the nearshore reef (Treatment 3).
- Waterbird species and guild use of the Treatment 3 nearshore reef varied by tide height and season, highlighting the importance of considering these factors when designing and evaluating avian response to living shoreline enhancements.

- Planting cordgrass using sods worked best to establish this native species, including along existing areas of cordgrass that may be encouraged to spread and coalesce with the plantings.
- Storm-driven dynamic sediment movement led to repeated losses of planted CA sea-blite; however, additional plantings in more protected locations (behind fringing pickleweed marsh) showed that survival is possible at this site, and self-seeding may reinforce population persistence.
- Surviving sea-blite plants grew substantially larger where wrack was added to planting holes, presumably due to moisture retention and nutrient availability as the wrack decomposed.
- Arboring of both planted sea-blite and existing pickleweed along eroding marsh edges produced larger plants, suggesting this simple method can be used to increase habitat value for birds and mammals during extreme high tides and storm-generated flooding.
- Transition zone plantings should be pursued from the high marsh edge to upland meadow zones to enhance wildlife refuge and foraging, but weed control will be needed, and fencing and signage should be added near locations where people and dogs are common.
- Steep scarp areas may require sediment augmentation (e.g., coarse beach addition to support wave-built berms) or other erosion control interventions to slow erosion, as indicated by poor performance of planted creeping wildrye along scarps in Treatment 7 and loss of two naturally-occurring pickleweed plots in Treatment 6.
- Across a wide array of treatments, we did not find that implementation of the Giant Marsh Living Shorelines Project increased non-native species beyond background levels or nearby reference sites (see summary below, Section 5).

4 Project Considerations in Ephemeral Habitats with Ongoing Climate Impacts

Marine species and habitats can be ephemeral in estuarine environments; and this and many other efforts locally, nationally, and globally show that exact footprints, locations, and survival of species such as native oysters and eelgrass can show wide fluctuations both within and between years. This has been especially observed during heavy winter flood events in San Francisco Bay (winter 2006-07, 2010-11, 2016-17, 2019-20, 2022-23), which can create low salinity conditions (less than 10 psu) for weeks or months, with resulting die off of oyster and eelgrass beds observed during heavy flood years- often with natural rebound of the species observed at sites within one recruitment season. The sources of propagules (oyster spat, eelgrass plants, etc.) do not necessarily get generated at the same site as the future location of the healthy adult populations. This speaks to the need to better understand and protect both source site populations as well as adult habitat populations, and connectivity between sites. It is important to continue to assess both short- and long-term survivability of these habitats in context with water quality data. It is also important to continue assessing the short-term and long-term habitat benefits and ecosystem services generated by these ephemeral habitat types, as the value of even short-term benefits (such as reproductive substrate for plants, oysters, invertebrates, and fish; juvenile Dungeness crab rearing on the reefs) may make the overall long-term restoration worthwhile -- even if regular maintenance (i.e., replanting eelgrass beds) and additional resources and funding may be needed over time. Many entities are changing how we think about this work - not as a “project” or even a “restoration” that needs maintenance - but rather as a process of intervention to restore function that will only be successful with continued engagement and investment.

5 Enhancement of Native Species and No Increase in Non-native Species Habitat

Our restoration and creation of habitat enhanced native species presence and coverage at the Giant Marsh Living Shorelines project, with added substrate and plantings leading to substantially more of the targeted native habitat-forming species including oysters, seaweeds, eelgrass, cordgrass, CA sea-blite, and upland transition vegetation. We did not find evidence of increased non-native species beyond background levels at the project site or in nearby reference areas. Below we summarize our findings related to this question for various groups of taxa, in bold.

All the **waterbirds** in our dataset are migratory native species. In addition, nearly all the **fish** using the project area are native species. We did not discern that the presence of reefs and eelgrass in Treatment 1 changed the fish species composition through the numerous sampling methods employed to detect use by highly mobile species. Although we have noted the difficulty in monitoring the highly structured reef habitat we created, in general the fish using this region of San Francisco Bay are native, and we have no reason to believe that our reef and eelgrass installations changed that. Among the less mobile species, we did find that Chameleon gobies were fairly common in the shell bags, though we also found that high numbers of native gunnels were attracted to this added interstitial space among the Pacific oyster half shell.

Considering **crabs** in the Treatment 1 reefs, we occasionally observed an adult green crab around the reef balls, but mostly we observed large native cancer crabs. Within shell bags in Treatment 1, we recorded at least 4 species of native crabs, including juvenile *Metacarcinus magister* and *Romaleon antennarium*, and we did not find juvenile green crabs. Thus, the shell bags were excellent habitat for small native crabs, with no evidence that they were being used by non-native crabs. In minnow traps deployed in eelgrass, native crabs far exceeded abundance of the non-native green crab, which was only found, in low abundances, in one year, 2020. Non-native **shrimp** were found in both restored and natural eelgrass.

Native **algae** were very abundant on the Treatment 1 reefs. The flora was dominated by *Chrondracanthus exasperatus* and *Cryptopleura* spp., both native. We noted the non-native *Gracilaria*, but it was rare. Observationally we had some of the non-native *Caulacanthus okamurae* on Treatment 2, but it was rare enough that it was not recorded in our quadrat counts. *Caulacanthus* is found nearly everywhere in the intertidal zone in San Francisco Bay. We did not see it at levels higher than elsewhere, including our reference site at Pt. Orient. The non-native *Gracilaria* is also very common and widespread in the Bay; we did not see it at levels that were higher than any other shallow-water site we have worked in.

With regard to invertebrates found on the eelgrass (**epifauna**), there were large interannual shifts in the dominant species in both the restored eelgrass and the natural reference bed, and no obvious patterns of increased non-native species abundance with the restoration. The most abundant epifaunal species on eelgrass, the non-native amphipod *Caprella drepanochir*, in its most abundant year of 2020, reached much greater densities in the natural bed (~1000 individuals/shoot) than in the restored eelgrass (~50 per shoot). A similar pattern (but with lower overall densities) was found in 2020, while in 2021 the restored eelgrass had higher *C. drepanochir* densities, and in 2019 and 2023 densities were quite low and comparable between natural and restored eelgrass. *Corophiidae* spp., also very abundant, could not be discerned to the species level; these similarly switched back and forth in abundance when comparing between natural and restored eelgrass across years. The non-native amphipod *Ampithoe valida* did as well, sometimes reaching greater numbers in the natural beds (as high as ~500 individuals per shoot) and sometimes in the restored eelgrass (as high as ~100 per shoot). The cryptogenic (uncertain native versus non-native status) amphipod *Jassa slatteryi* was generally more abundant on restored eelgrass (up to ~500 individuals per shoot in 2021), but was similar between restored and natural in 2022 (~50 per shoot in both), and nearly absent in both in 2019 and 2023 in restored and natural eelgrass.

For benthic invertebrates from cores (**infauna**), some taxa could not be discerned to the species level thus making it impossible to determine their native versus non-native status. Among the taxa that we could identify as non-native, they were prevalent in control and treatment areas both before and after treatment installation, indicating that the project did not increase their presence.

Non-native species were present in the transition zone planting areas, but there was no evidence that they expanded as a result of project activities, and in fact we actively reduced their abundance locally through our actions.

6 Key Recommendations for Future Living Shorelines Projects

An important recommendation, considering the variation in abundances of key species such as Olympia oysters over the years of the project, is that restoration of living shorelines be conducted over a range of sites and years to buffer across conditions that may become detrimental at any one site in a particular year. Within-site heterogeneity in restoration design can also help practitioners to diversify and maximize components that can help spread risk against physical and biological stressors across time and location, to increase chances of survival and growth. Including a variety of habitat approaches within a project can result in higher biodiversity at the site, more food resources and reproductive substrates, beneficial interactions between species, and increased spatial and temporal coverage of treatments across the site and in different growing seasons.

A second key recommendation is to conduct monitoring projects and restoration projects with methods to better understand both source site populations that are generating propagules, as well as adult habitat populations, and connectivity between sites. This applies to eelgrass and oysters as well as many species of seaweeds and invertebrates. These factors all influence the success of some project components and can help to put the results at restoration sites into context. In some cases, siting and timing of restorations may be maximized to increase dispersal and connectivity benefits between sites, and for long term goal diversification to ensure success.

A third recommendation is to support efforts to assess both short- and long-term survivability of these habitats in the context of tidal, wind, wave, water quality, and air temperature data. Atmospheric river events, heat waves, and sea level rise impacts can all influence outcomes and performance during particular time periods, that are often hard to accurately predict within the project timeframe. Project goals and timeframes should be flexible enough to account for these impacts and response mechanisms and realistic timing. Restoration and living shoreline projects should not be expected to perform better than existing conditions in relation to these dynamic and stochastic stressors.

Fourth, we recommend continuing to test ways to maximize facilitative interactions between species and habitat types and co-deployed restoration methods in future projects. We designed for and observed multiple examples of positive interactions such as seaweed shading and moisture on oysters, and benefits with strategically siting oyster reefs resulting in eelgrass establishment and spread especially inshore of the reefs.

The State Coastal Conservancy made substantial efforts to fundraise local, state, and federal funds to support seven years of robust and frequent pre-and-post construction monitoring as part of this voluntary, experimental restoration project. With the need to look at more specific and long-term interactions, this was just sufficient to document patterns and trends. We recommend longer-term assessments of this and other restoration projects where possible, and strong support for policy improvements and other needed avenues towards funding such monitoring efforts. We also recognize that this can be a burden or barrier to entry for some landowners and smaller-scale community project efforts, and we support clear and specific monitoring that is matched to the scale, needs, and timeframe of pilot projects. The Giant Marsh Living Shorelines Project included substantial experimental components in the design with a related commitment to monitoring these components, which required substantial funding and technical expertise in different disciplines. This should not be replicated at all future living shorelines projects but should be developed thoughtfully and commensurate to the goals, scale, risk, and key questions of each project.

Finally, we also want to make a minor but important note on terminology, because language matters and sets the tone for agency and public perception of goals and outcomes. Our team, and many other partners and efforts including agencies involved in the San Francisco Bay Subtidal Habitat Goals Report, have been using the term Olympia oyster reefs and we have used the term oyster/ habitat reefs in this report. In the future, we recommend the terms **nearshore reefs and offshore reefs** to better describe the multiple species and multiple goals and benefits associated with this living shoreline approach. Native Olympia oysters remain a key species we care about and want to increase focus on and include in enhancement efforts in the bay, but this was never meant to be a single species or single habitat goal. Nearshore and offshore reef structures provide habitat for a wide variety of invertebrates and seaweeds, which in turn provide benefits such as foraging and reproductive substrate that enhance ecological health for fish, birds, and marine mammals. Reefs also provide physical benefits including wave attenuation and sediment accretion that help buffer shorelines against sea level rise and erosion. There are times when Olympia oysters may be more or less abundant at a site over time, due to factors like fluctuations in baywide annual oyster spawning and recruitment, atmospheric river events causing prolonged periods of low salinity, and non-native Atlantic oyster drills in some regions that can prey on oysters and cause mortality. Even during periods of lower oyster presence on the reefs, they provide habitat for many plants and animals, increase biodiversity and food resources, and provide physical structure that helps to buffer shorelines and prevent shoreline erosion as one tool in an adaptive toolbox of living shoreline methods.

7 Capacity-Building Challenges - Pace, Scale, and Participation

This project and other pilot living shorelines projects led by the State Coastal Conservancy and many partners have resulted in new information gained about promising nature-based shoreline adaptation techniques. Forward momentum and enthusiasm for this work is building with each person and organization engaged, who are working collaboratively to develop capacity in areas such as engineering and ecological design planning, scientific and technical guidance, expertise in environmental compliance and regulatory consultations, landowner awareness and involvement, local workforce development in materials, fabrication, and construction, and public and funding support. This momentum will have bigger and more immediate growth in each of these areas if it can continue to advance without interruption between projects, as too few projects with larger gaps of time between projects causes gaps in participation and sequential learning if an entity has to stop and start and remobilize. Like any business, most entities engaged in San Francisco Bay restoration work need drivers like upcoming project work to have the incentive and funding to maintain and grow the right staff, expertise, technology, equipment, and general resources needed to advance living shorelines work. The small number of project opportunities often limit forward progress and participation, since there are gaps in time between living shorelines projects as even small pilot projects can take years to fund, develop, permit, and implement.

In addition, there are a relatively small number of design engineers and marine contractors in San Francisco Bay with the appropriate skills, training, and contractor expertise (down to the type and size of equipment) that can be best applied to living shorelines. Many engineering and design firms, and marine construction firms, have formed to respond to civic infrastructure needs like airports, roads, and housing, and economically driven activities like dredging or placement of bay fill, in the bay's nearshore environments. There is a need to develop more projects more consistently over time, engage new practitioners and construction contractors, and further develop innovative and local approaches to design, fabrication and installation of living shoreline elements that include the most appropriate local green-grey materials, equipment, and contractor knowledge and commitment to environmental goals and conservation measures. There is a huge need for local industry development and local workforce training related to materials, fabrication, and construction of living shorelines project components within the preferred and allowable seasonal and tidal windows, elevations, materials type, and access.

The Conservancy has developed strong working relationships with a great number of public and private entities as a necessary part of its work on wetland and subtidal restoration efforts over the past 25 years. In particular, the regional network of partners in the San Francisco Bay Living Shorelines Projects at San Rafael and Giant Marsh, and the San Francisco Estuary Invasive Spartina Project (ISP) are recognized as some of the most extensive and diverse collaborations of public and private agencies and landowners engaged in restoration projects in San Francisco Bay. The current project included substantial education and outreach with these partners, as our goal is to share results from the projects with multiple agency staff and other project managers, so that these techniques can be incorporated into multiple restoration projects to enhance endangered species habitat, pilot climate adaptation techniques, and integrate habitat connectivity and functions. This project helped to advance goals of those networks, and the regional habitat recommendations in the San Francisco Bay Subtidal Habitat Goals Report (2010), Baylands Habitat Goals Science Update (2015), the Adaptation Atlas (SFEI and SPUR 2019), and the San Francisco Estuary Blueprint (2022).

8 Next Steps to Regionally Advance Living Shorelines in San Francisco Bay

The Giant Marsh Living Shorelines Project and others have led to the development of the Regionally Advancing Living Shorelines (RALS) Project to continue building collective momentum and regional involvement. The RALS project is intended to help increase the pace and scale of shoreline restoration work, transfer knowledge between regions and engage new entities, develop regional design and constructability guidance, and programmatically work with a coalition of landowners in the Central San Francisco Bay to design and permit ten new living shorelines sites. The project is being led by the State Coastal Conservancy with support from the San Francisco Estuary Institute and includes participation from many of the partners involved in pilot projects to date as well as many new partners to increase uptake of living shorelines approaches in San Francisco Bay. The effort was designed to help share and make project information accessible, advance successful living shorelines methods and outcomes through inclusion in larger scaled-up future projects, and build capacity with practitioners, regulatory agencies, the marine construction community, and others to normalize and increase living shorelines projects and benefits in San Francisco Bay.

9 Thank You to Our Funders and Partners

This project could not have occurred without the tenacious and forward-thinking hard work by staff and organizations involved in this first-of-its-kind work in San Francisco Bay. We are grateful for the collaboration, contributions, and support by all partners, funders, landowners, practitioners, and others involved:

- State Coastal Conservancy
- Estuary & Ocean Science Center

- Smithsonian Environmental Research Center
- Environmental Science Associates
- US Geological Survey Western Ecological Research Center
- Olofson Environmental
- Ducks Unlimited
- Reef Innovations
- Dixon Marine Services
- CiviCorps
- Conservation Corps North Bay
- The Watershed Nursery Cooperative
- CA Wildlife Foundation
- Triton Marine
- US Fish and Wildlife Service
- NOAA Fisheries Restoration Center
- Cosco Busan Oil Spill Fund
- East Bay Regional Park District
- State Lands Commission

10 Literature Cited

Altmann, J. 1973. Observational study of behavior: sampling methods. Behaviour 49:227–266.

Baye, P. 2006. California sea-blite (*Suaeda californica*) reintroduction plan, San Francisco Bay, California. Prepared for US Fish and Wildlife Service, Sacramento Fish and Wildlife Office.

Baylands Habitat Goals Science Update. 2015. The Baylands and Climate Change: What We Can Do. California State Coastal Conservancy.

Beheshti, K. M., S. L. Williams, K. E. Boyer, C. Endris, A. Clemons, T. Grimes, K. Wasson, and B. B. Hughes. 2022. Rapid restoration of multiple ecosystem services associated with a coastal foundation species. Ecological Applications 32(1), e02466.

Bible, J. M., B. S. Cheng, A. L. Chang, M. C. Ferner, K. Wasson, C. J. Zabin, M. Latta, E. Sanford, A. Deck, and E. D. Grosholz. 2017. Timing of stressors alters interactive effects on a coastal foundation species. Ecology 98:2468-2478.

Borgerhoff Mulder, M. and T. M. Caro. 1985. The use of quantitative observational techniques in anthropology. Current Anthropology 26:323–335.

Bos, A. R. and M. M. van Katwijk. 2007. Planting density, hydrodynamic exposure and mussel beds affect survival of transplanted intertidal eelgrass. Marine Ecology Progress Series. 336:121-129.

Boyer, K., C. Zabin, S. De La Cruz, E. Grosholz, M. Orr, J. Lowe, M. Latta, J. Miller, S. Kiriakopolos, C. Pinnell, D. Kunz, J. Moderan, K. Stockmann, G. Ayala, R. Abbott, and R. Obernolte. 2017a. San Francisco Bay Living Shorelines: Restoring Eelgrass and Olympia Oysters for Habitat and Shore

Protection. Chapter 17 in D. M. Bilkovic, M. Mitchell, J. Toft, and M. La Peyre, eds., Living Shorelines: The Science and Management of Nature-Based Coastal Protection. CRC Press Marine Science Series.

Boyer, K., M. Patten, J. Miller, C. Pinnell, J. Modéran, G. Ayala, S. Kiriakopolos, and K. Stockman. 2017b. Living Shorelines Project: SF State Progress Report through 2017. Report submitted to the California State Coastal Conservancy.

Buchbinder, M., M. Patten, and K. Boyer. 2024. SFSU Final Report through 2023 for Treatment 6: Marsh Scarp Revegetation and Arboring for California Sea-blite and Pickleweed. San Francisco Bay Living Shorelines Project at Giant Marsh. Prepared for the California State Coastal Conservancy.

Buchbinder, M., M. Patten, and K. Boyer. 2025a. SFSU Final Report through 2023 for Treatment 1: Offshore eelgrass bed and oyster reef (eelgrass portion). San Francisco Bay Living Shorelines Project at Giant Marsh. Prepared for the California State Coastal Conservancy.

Buchbinder, M., J. Bruno, C. Tettelbach, T. Filipczyk, and K. Boyer. 2025b. Restoring eelgrass in Living Shorelines: Opportunities to combine shore protection with amelioration of ocean acidification. Final report to the California Ocean Protection Council.

Buchbinder, M.V. and K.E. Boyer. 2023. *Suaeda californica* growth and survivorship through 2023 (Year 1 monitoring) for the Heron's Head Park Shoreline Resilience Project. Report to Environmental Science Associates and the Port of San Francisco.

Chang, A. L., A. K. Deck, L. J. Sullivan, S. G. Morgan, and M. C. Ferner. 2016. Upstream – downstream shifts in peak recruitment of the native Olympia oyster in San Francisco Bay during wet and dry years. Estuaries and Coasts. DOI: 10.1007/s12237-016-082-1.

California Irrigation Management Information System (CIMIS), Point San Pedro weather station. https://cimis.water.ca.gov/WSNReportCriteria.aspx, accessed 2020.

California Department of Water Resources (CDWR). 2023. Water Year 2023: Weather Whiplash, From Drought to Deluge.

Cohen, A. N and C. J. Zabin. 2009. Oyster shells as vectors for exotic organisms. Journal of Shellfish Research 28(1), 163-167. DOI: 10.2983/035.028.0106.

ESA (Environmental Science Associates). 2024. Comprehensive Physical Processes Monitoring Report (2019-2023), San Francisco Bay Living Shorelines Project. Prepared for San Francisco Estuary Institute and California State Coastal Conservancy.

ESA (Environmental Science Associates), SCC (State Coastal Conservancy). 2017. San Francisco Bay Living Shorelines Project at Giant Marsh, Pt. Pinole Regional Shoreline (Richmond, Contra Costa County). Project Description. Prepared for California State Coastal Conservancy.

Eschenroeder, J., E. Carpenter, T. Lamb, and D. Lee. 2022. Fish monitoring at the Giant Marsh living shorelines habitat restoration project: Evaluation of fish abundance, biodiversity, and habitat use. Final report to San Francisco State University.

Estuary Blueprint (Comprehensive Conservation Management Plan for the San Francisco Estuary). 2016. San Francisco Estuary Partnership.

Filipczyk, T. 2025. Carbon storage in natural and restored eelgrass beds of San Francisco Bay. M.S. Thesis. San Francisco State University.

Graham, T., L. A. Hall, and S. E. W. De La Cruz. 2024. Giant Marsh Living Shorelines Project: Avian and Benthic Invertebrate Monitoring. Final report prepared for the California State Coastal Conservancy.

Grosholz, E., J. Moore, C. Zabin, S. Attoe, and R. Obernolte. 2008. Planning for native oyster restoration in San Francisco Bay. Final report to the California Coastal Conservancy, Agreement # 05-134, 40 pp.

Hammond, J. 2025. 2023/2024 Monitoring Progress Report. Treatment 4: Pacific Cordgrass Inshore from Nearshore Reef, Treatment 5: Pacific Cordgrass Adjacent to Existing Pacific Cordgrass, and Treatment 7: Marsh-Upland Transition Zone (Ecotone) Enhancement. Final report prepared for the California State Coastal Conservancy.

Hendrickson, C. I. 2024. Restored oyster reef effects of flow and related ecosystem services. M.S. Thesis. San Francisco State University.

Jackson, M. V., R. Mott, S. Delean, B. J. Hunt, J. D. Brookes, P. Cassey, and T. A. A. Prowse. 2024. Shorebird habitat selection and foraging behavior have important implications for management at an international important non-breeding wetland. Ecological Solutions and Evidence, DOI: 10.1002/2688-8319.12316.

Margulies, A. 2023. Identifying refuges from climate extremes to enhance conservation in a shallow estuary. M.S. Thesis. San Francisco State University.

Merkel & Associates. 2014. San Francisco Bay Eelgrass Inventory. Submitted to National Marine Fisheries Service, Santa Rosa, CA.

Merkel & Associates and EOS Center. 2022. Merkel & Associates, Inc. and San Francisco State University Estuary & Ocean Science Center, 2014-2022 Monitoring Report to Cosco Busan Trustee Council, Cosco Busan DARP, San Francisco Bay, California.

Pinnell, C. M., M. V. Patten, G. S. Ayala, and K. E. Boyer. 2021. Seagrass and oyster reef restoration in living shorelines: Effects of habitat configuration on invertebrate community assembly. Diversity 13(6), 246; doi.org/10.3390/d13060246

Ricciardi, A. and E. Bourget. 1998. Weight-to-weight conversion factors for marine benthic macroinvertebrates. Marine Ecology Progress Series 163:245–251.

Rocha, A. R., J. A. Ramos, T. Paredes, J. A. Masero. 2017. Coastal saltpans as foraging grounds for migrating shorebirds: an experimentally drained fish pond in Portugal. Hydrobiologia 790:141-155.

Rohmer, T. and D. Kerr. 2023. San Francisco Estuary Invasive Spartina Project 2021-2022 Monitoring and Treatment Report. Report submitted to the California State Coastal Conservancy. Olofson Environmental, Inc., Oakland, California.

San Francisco Bay Subtidal Goals Report. 2010. California State Coastal Conservancy.

SFEI and SPUR. 2019. San Francisco Bay Shoreline Adaptation Atlas: Working with Nature to Plan for Sea Level Rise Using Operational Landscape Units. Publication #915, San Francisco Estuary Institute, Richmond, CA.

San Francisco Estuary Blueprint (Comprehensive Conservation and Management Plan for the San Francisco Estuary). 2022. San Francisco Estuary Partnership, San Francisco, CA.

Santos, K. 2020. *Suaeda californica* in San Francisco Bay: Phenology, germination, and high tide refuge. Master's thesis, San Francisco State University.

Santos, K., M. Patten, P. Baye, and K. Boyer. 2020. Experimental tests of "arboring" as a tool to increase tidal marsh canopy height for high tide refuge. Report submitted to the California State Coastal Conservancy for the New Life for Eroding Shorelines Project of the Advancing Nature-based Adaptation Solutions in Marin County grant program.

San Francisco Estuary Institute (SFEI). 2015. Shifting Shores: Marsh Expansion and Retreat in San Pablo Bay. Developed for the S.F. Bay Water Quality Fund, U.S. Environmental Protection Agency, and the San Francisco Estuary Partnership.

San Francisco Estuary Institute (SFEI). 2020. New Life for Eroding Shorelines: Beach and Marsh Edge Change in the San Francisco Estuary. Developed for the Marin Community Foundation and the California Coastal Conservancy.

Silliman, B. R., E. Schrack, Q. He, R. Cope, A. Santoni, T. van der Heide, R. Jacobi, M. Jacobi, and J. van der Koppel. 2015. Facilitation shifts paradigms and can amplify coastal restoration efforts. Proceedings of the National Academy of Sciences 112(46): 14295- 14300.

Stewart-Oaten, A. 2003. On Rejection Rates of Paired Intervention Analysis: Comment. Ecology 84:2795–2799.

Stewart-Oaten, A., W., W. Murdoch, and K. R. Parker. 1986. Environmental Impact Assessment: "Pseudoreplication" in time? Ecology 67:929–940.

US Fish and Wildlife Service (USFWS). 2013. Recovery Plan for Tidal Marsh Ecosystems of Northern and Central California. Region 8, US Fish and Wildlife Service, Sacramento, CA.

Wasson, K., C. Zabin, J. Bible, S. Briley, E. Ceballos, A. Chang, B. Cheng, A. Deck, T. Grosholz, A. Helms, M. Latta, B. Yednock, D. Zacherl, and M. Ferner. 2015. A guide to Olympia oyster restoration and conservation: environmental conditions and sites that support sustainable populations in Central California. Elkhorn Slough National Estuarine Research Reserve, Watsonville 43 pp.

Zabin, C., G. Ayala, and T. Grosholz. 2018. San Francisco Bay Living Shorelines Project: Oyster performance on constructed reefs and adjacent shorelines: 2021-2017. Report to the California State Coastal Conservancy.

Zabin, C. and J. Blumenthal. 2024. San Francisco Bay Living Shorelines Project Giant Marsh Restoration, Year 5 (2023): oyster performance on restoration substrates and existing substrate. Prepared for the California State Coastal Conservancy.

Zabin, C. and S. Kiriakopolos. 2014. San Francisco Bay Living Shorelines Project: Assessment of oyster performance on constructed reefs and adjacent shorelines. A report to the California State Coastal Conservancy.