Appendix 8-1: Eelgrass Conservation and Restoration in San Francisco Bay: Opportunities and Constraints

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Final Report for the San Francisco Bay Subtidal Habitat Goals Project

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I. Introduction

Seagrasses—rooted aquatic plants in estuaries and along protected shorelines throughout the world—have important influences on biogeochemical cycling, sediment stability, and food web support (e.g., McGlathery et al. 2007; see review by Orth et al. 2006). They provide food and shelter for numerous fish and invertebrates, and serve as a nursery habitat, providing predation refuge for juvenile fishes (Orth et al. 1984; Bostrom and Bonsdorff 2000; Duarte 2000). Worldwide, they support myriad rare and endangered animals as well as commercially important species (Hughes et al. 2009). However, seagrasses are in decline throughout much of the world (Worm, 2006; Halpern 2008), with rates of loss accelerating from a median of 0.9% per year before 1940 to 7% per year since 1990 (Waycott et al. 2009). A large percentage of this decline is attributed to human impacts, including filling of shallow waters, dredging, and eutrophication (Short and Wyllie-Echeverria 1996). Both global and small-scale declines have led to increased interest in developing successful conservation and restoration programs for seagrasses.

The most widespread seagrass in San Francisco Bay, *Zostera marina* L. (eelgrass, Fig. 1), provides valued ecological services (Spratt 1981; Kitting 1993; Hanson 1998), yet eelgrass beds only cover <1200 ha or approximately 1% of submerged land in the bay (Merkel and Associates 2004a). Although Zimmerman et al. (1991) found submarine light levels in the late 1980's to be relatively low and consequently limiting for eelgrass growth and vegetative reproduction, current biophysical modeling efforts indicate that nearly 9490 ha of bottom area may now be suitable habitat for eelgrass (Merkel and Associates 2005). In addition, declines in suspended sediment concentrations measured in the last decade indicate improving water clarity (Schoellhamer 2009), suggesting that conditions in San Francisco Bay are becoming more favorable for eelgrass. Restoration measures could proactively advance population expansion in San Francisco Bay and take advantage of improvements in water quality conditions. Recent experimental work in small plots has led to guarded optimism on the potential for restoration success at a larger scale (e.g., Boyer et al. 2008).

In this report, we review the state of knowledge on eelgrass in San Francisco Bay, including distribution and abundance, population dynamics, and potential limiting factors. Next, we make specific recommendations for strategies, locations, and techniques for restoration, and identify constraints and areas for further study needed. Throughout the document the paucity of information specific to this region will be apparent, suggesting the need for future studies that improve our understanding of how best to manage this habitat. We emphasize the need for a phased approach to restoration activities, so that learning from previous steps informs wise decision-making as restoration projects scale up. This document has been written for the San Francisco Bay Subtidal Habitat Goals (SHG) Project, a collaborative interagency effort to establish a long-term vision for research, restoration, and management of subtidal habitats in the Bay. This final version incorporates feedback on previous drafts from the SHG Project administrative core group and committee members, including several experts on seagrasses, as well as comments resulting from a voluntary public comment period. Hence, it reflects the contributions of many participants in the overall project.

II. Current knowledge of eelgrass in San Francisco Bay

A. Historic distribution

No quantitative information is available on the extent of eelgrass in San Francisco Bay prior to the 1980s. The earliest known studies of eelgrass in the Bay were conducted in the 1920s by William Setchell, a botany professor at the University of California, Berkeley. These studies were not intended to document the area of eelgrass distribution, but they do indicate that at a minimum there were beds present at Keil Cove and Paradise Cove along the Tiburon Peninsula in Marin County (Setchell 1922, 1927, 1929). Setchell stated that flowering shoots were abundant along the shores of San Francisco Bay at this time, but did not provide any other indication of the extent of the beds. There is anecdotal evidence that Brant geese were numerous in south San Francisco Bay in the past, which has led to speculation that eelgrass may have been abundant in that region, as eelgrass is an important food for these geese along the Pacific Flyway (N. Consentino-Manning, NOAA Restoration Center, pers. comm.). Unfortunately, this is the extent of our knowledge of historic distribution at this time. Hence, throughout this report, we use the term "restoration" loosely, as locations where we may choose to create eelgrass habitat may not coincide with historic distribution.

B. Current distribution and abundance

Bay-wide surveys

Bay-wide surveys of eelgrass distribution were conducted in 1987 and 2003 (Fig. 2). Both surveys found that eelgrass is discontinuously distributed throughout the central and southern regions of San Francisco Bay. In the more recent study, the total acreage estimate was 1165.7 ha (2880.5 ac), with individual patches ranging in size from 609 ha (1504 ac) to 0.8 ha (0.2 ac) (Merkel and Associates 2004a). These data are in contrast to the 1987 survey which reported the estimated total abundance to be 127.8 ha (316 ac) with no patch larger than 50 ha (124 ac) (Wyllie-Echeverria and Rutten 1989). Also noteworthy is the detail that, when compared to 1987 data, eight new sites and additional "minor patches" were detected in 2003 (Merkel and Associates 2004a). The nine-fold increase in total abundance and expansion of individual patches is perhaps largely explained by improvements in mapping techniques. The later study utilized sidescan and single beam sonar with integrated GPS along with aerial surveys from a helicopter to help narrow down regions at which to conduct the acoustic monitoring work (Merkel and Associates 2004a). However, it is also possible that within patch and bay-wide population expansion is resulting, in part, from the trend in improved water clarity from reduced suspended sediment concentrations (Schoellhamer 2009).

Notable from both surveys is the large percentage of the total acreage in a single bed located between Point Pinole and Point San Pablo (the North Richmond or Point San Pablo eelgrass bed); approximately half of the eelgrass in each study was found in this location. In addition, most beds were found to be along the east shore, from Point Pinole

to Bay Farm Island. The greatest depth found for any bed was at Richardson Bay (-3.0 m), but this was a very unusual occurrence baywide. 98.8% of all mapped eelgrass in the bay was found between -1.77 and 0.4 m MLLW. 94% was found between -1.6 and 0 m (Merkel and Associates 2004a), which is relatively consistent with earlier predictions by Zimmerman et al. (1991) for depths observed at clear versus turbid sites.

An additional bay-wide survey using the same methods as in 2003 was conducted in October 2009 and suggests further expansion, to 1500 hectares (3700 acres) (Merkel and Associates 2009a; Figure 8-1 in main Goals report). More surveys would be needed to determine whether this change reflects an increasing trend in eelgrass acreage as opposed to simply capturing variation between two time points.

The Seven-Site Survey

A four-year study (2006-2009) of eelgrass bed densities was conducted as part of a CA Coastal Conservancy funded project to assess conditions within extant beds that might help in the planning and evaluation of restoration projects. Dubbed the Seven-Site-Survey, researchers at San Francisco State University's Romberg Tiburon Center (SFSU/RTC) visited the seven beds each year in each of two seasons, late April/early May and late July/early August. The seven beds were chosen to represent a range of conditions and the geographic extent of eelgrass beds in the Bay (Fig. 3).

Seasonal and interannual patterns in densities are described below in part D. Mean shoot densities ranged from <5 to >80/m^2 in the years for which data review is complete, 2006-2008 (Fig. 3). The bed at Keller Beach frequently had the highest densities, while Richardson Bay tended to have the lowest. Two points in time each year represent only a snapshot of densities; more frequent monitoring would be needed to fully describe the range in densities occurring seasonally at any one site.

Through repeated visits to these beds for the survey, it was discovered in August 2006 that the bed at Point Molate had died back some time during the months since the April survey (Fig. 3, top panel shows change from April to August). Only a few isolated shoots and rotted rhizomes were found in a thorough search of the embayment. According to historical air photos, this bed had been present as early as 1949. Monitoring of extant beds is important to observing such changes in the Bay's eelgrass distribution.

Other localized surveys

Surveys of eelgrass extent have been conducted at various individual sites around the bay in recent years. There now exists an 8-year record of eelgrass distribution and cover at the Emeryville bed adjacent to the Oakland San Francisco Bay Bridge, an effort to document eelgrass coverage before, during, and after impacts from seismic retrofit work on the bridge. The resulting images from sidescan sonar surveys provide important documentation of the extreme interannual variation in distribution and cover that can

occur in one of the Bay's beds (Fig. 4; Merkel and Associates 2006). While high turbidity early in the project may account for some of the variation, these extremes in distribution may best be explained by static annual survey timing only sometimes coinciding with high seasonal abundance periods (Merkel and Associates 2006). While we cannot be certain that this level of variability occurs in all beds in the Bay, personal observations suggest that this is not an unusual pattern for some of the beds, and high variation in seagrass bed density and distribution has been similarly observed in other regions (Orth et al. 2006).

Other surveys of individual bed extent have been conducted before and after dredging or marina development. Examples include surveys to assess dredging effects in Horseshoe Bay, Richmond Harbor, and Oakland Middle Harbor conducted in late 1990s to mid 2000s by Merkel and Associates using sidescan sonar and a survey to assess effects of channel and dock construction in Belvedere Cove conducted in the mid-2000s by WRA Environmental Consultants using GPS points to create polygons. Other surveys include Richardson Bay Audubon's GPS survey by kayak of the eelgrass within the Audubon Sanctuary in the northeast portion of Richardson Bay in 2006. In addition, sidescan sonar surveys of Keil Cove, at the tip of the Tiburon peninsula, have been used to assess the effects of oiling and damages from clean up efforts following the Cosco Busan oil spill in November 2007.

Recommendations

Much of the acreage occurring at favorable depths for eelgrass in the mid 1850's is no longer suitable due to sediment accretion, especially that resulting from the large volumes of sediment that began entering the bay through hydraulic mining in the Sierra Nevada during the Gold Rush. In addition, the period of intense development that came with settlement of the Bay Area led to filling or dredging of extensive areas that would have been at suitable depths for eelgrass. Hence, the configuration of the bay is so altered that even if we were able to ascertain the historic distribution and abundance it would make little difference to our current planning efforts at specific sites. However, it would be useful to know the relative abundance among embayments and along shorelines within the bay when setting goals for increasing acreages on these larger scales. It has been suggested that oyster landings records may provide evidence of eelgrass distribution during the latter part of the 19th century and early 20th century, when there was a viable oyster fishery in the bay (J. Thompson, USGS, pers. comm.), and we recommend that investigation of these records be undertaken. Further, it may be possible to evaluate sediment cores from particular areas for evidence of past eelgrass presence through pollen, seeds, or chemical signatures of stable isotopes or lipid biomarkers (e.g., Goman et al. 2008; Zimmerman and Canuel 2000).

Periodic bay-wide surveys should be conducted, as these are critical to our ability to ascertain changes in distribution or coverage that should inform management decisions for protection and restoration. We recommend bay-wide surveys every 3-5 years with sidescan sonar or other technology with high efficacy at detecting plants beneath turbid

waters (e.g., Dobson et al. 1995). If technology used changes, it should be calibrated to the sidescan sonar surveys conducted in 2003 and 2009 so that distribution and cover classes can be compared over time. Similar surveys should be conducted yearly for specific beds of interest as has been done at the Emeryville Flats bed to evaluate bridge-retrofitting impacts, and if possible, multiple surveys should be conducted per year to capture seasonal variation. Candidate beds are those known or suspected to fluctuate dramatically such as Richardson Bay at the Audubon sanctuary and Pt. Molate (authors' observations). Other candidates for yearly surveys are those with particular conservation or restoration interest (e.g., the large bed along the North Richmond shoreline and the annual bed at Crown Beach) and beds at the extremes of the bay-wide distribution, which may serve as indicators of response to climate-related changes (e.g., small beds near Carquinez Bridge and just south of the San Mateo Bridge at Eden Landing). In addition, total density and flowering shoot densities should be continued along transects yearly, as they have been informative at describing the range of fluctuations within and among beds in the bay over time, and can be used to set reference conditions for restoration sites. At select beds with particular conservation interest, such monitoring should be conducted more than the twice-yearly schedule of the Seven Site Survey (e.g., monthly) to avoid the "snapshot" in time produced by less frequent monitoring.

C. Model predictions of distribution

A biophysical model was developed to assess the habitat potentially suitable for eelgrass in San Francisco Bay (Merkel and Associates 2005). The model is an Ecological Limits, Viability, and Sustainability (ELVS) Model, originally developed to evaluate the effects of cooling water discharges and port development on marine habitats. The ELVS model developed for San Francisco Bay eelgrass included data inputs of bathymetry, tides, wind conditions, bottom substrate, fluvial influences, and surface light levels through time. This resulted in predicting intermediate outputs of bottom shear, turbidity, salinity, and light levels near the bay bottom over time. The temporal aggregation of these factors resulted in defining multiple environmental gradients that were then compared with known tolerances of eelgrass to develop a predictive model of site suitability for eelgrass growth and survival. The model predicts that as much as 9,490 ha (23,440 ac) of potential habitat may be suitable for eelgrass within the north, central, and southern regions of the bay (Fig. 5), about an order of magnitude more than currently exists. The largest areas of high suitability are predicted in the Central Bay and the southern portion of San Pablo Bay. The depth range predicted was 0.3 to -5.4 m MLLW. About 94 % of eelgrass was predicted to occur between -1.0 m and -0.2 m MLLW. High current speeds (0.39 m/s) and lower ones (0.15 m/s) were related to predicted eelgrass occurrence, perhaps related to clear water expected at higher speeds and lowered sediment suspension expected at lower speeds. Low bottom shear (≤0.25 pascals of shear) was predicted to be important, as was low levels of fluvial sediment suspension (15-55%) which are likely correlated with clearer water conditions.

The ELVS model can be used as a preliminary tool in site selection for restoration projects, and review of the model output is recommended as a Phase 2 action in

restoration planning and execution (see Section V, Phased Approach). However, any model has its limitations, and the ELVS model should be just one tool used in site selection. Limitations of this model may include an overestimate of suitable acreage at the deeper elevations (the model predicts suitable depths in some areas >2 m deeper than current distributions of eelgrass) and a tendency to over-predict eelgrass distribution in sheltered harbors and coves with high residence times. Good bathymetry for shallow waters is lacking for the Bay, and the estimates used in some areas could have led to under- or overestimates of suitable habitat for some regions of the Bay. In addition, factors such as bioturbation and herbivory are poorly understood and not considered in the model; these and other factors could influence success at specific sites. Further, eelgrass currently occurs in eastern San Pablo Bay (Figure 8-1 in main report from Merkel and Associates 2009a) and even into Carquinez Strait as well as south of the San Mateo Bridge at Eden Landing (K. Boyer, pers. obs., 2010), suggesting that suitable habitat extends well beyond the geographic range predicted by the ELVS model, at least in favorable years.

D. Population dynamics of eelgrass

Seasonal and interannual patterns in density

A four-year survey led by researchers at SFSU/RTC was conducted at seven extant beds in the bay (Fig. 3) in two seasonal periods each year, late spring and summer, from 2006 to 2009. In this "Seven Site Survey", shoot counts were made in twenty 0.25 m^2 quadrats along a 100 m transect line placed through a representative portion (or portions) of each bed and average densities calculated. In the three years of data evaluated so far, eelgrass densities in the bay were highly variable among beds within a season in a given year (Fig. 3). Spring sampling at some beds in some years captured seedling recruitment, evidenced by higher densities in April than in August, by which time some mortality of new plants had occurred (e.g., at Keller Beach in 2007 and 2008). In contrast, April sampling at the annual bed at Crown Beach (see below, Perennial versus annual life history) generally showed lower densities in April than in August as shoots in April were entirely new seedling recruits, rather than existing perennial plants being supplemented by seedlings as at other beds. The Point San Pablo (North Richmond) bed, which has been observed to undergo winter storm damage to leaves, sediment deposition, and even some uprooting of rhizomes (K. Boyer, pers. obs., 2006), tends to have lower density in spring in some years despite observations of new seedlings recruiting to the population. The Point Molate population disappeared in summer 2006 followed by a small degree of recovery in 2007 and much increased densities in some areas of the bed in 2008 (hence high variability; this bed is sampled with several transects due to variation in recovery across the bed). Richardson Bay tended to have low shoot densities relative to other sites.

That there is such high variability in shoot densities across sites and seasons and even within single beds over time is an important lesson for planning restoration and assessing success. It should be noted that densities are one measure used to reflect bed architecture

and habitat structure and function; biomass and productivity rates may be equally important indications of habitat and food production for invertebrates and fishes. These latter measures are more destructive and/or time consuming than density counts, but nonetheless provide a more complete picture of the structure and function of the Bay's eelgrass beds when measured in concert with densities. For example, plants at Richardson Bay tend to be sparse relative to other beds in the Bay but also tend to be quite large, and may make up for low densities with high biomass. We do not know of biomass estimates in the Bay to date, but productivity estimates are underway as part of the Seven Site Survey.

Perennial versus annual life history

There is at least one annual population of *Z. marina* in San Francisco Bay, at Robert Crown Memorial Beach (Crown Beach) in Alameda; plants flower in the late summer, then die in the late fall, and the population recruits from seed each spring (Fonseca et al. 2003; authors' pers. obs.) (Fig. 6). There is speculation that other annual populations may be present in the Bay (Fredette et al. 1987). Biophysical modeling efforts are underway to predict the suite of environmental conditions that contribute to this occurrence (M. Fonseca et al., in prep). Annual populations of *Z. marina* are known to exist at several locations in other regions (Keddy and Patriquin 1978; Bayer 1979; Phillips and Backman 1983; van Katwijk et al. 2000); abiotic stressors such as extreme cold or heat are hypothesized to drive the annual life history observed at these sites. At Crown Beach, where the annual portion of the bed occurs at higher elevations than many beds in the bay, one potential abiotic driver is exposure or related variables such as dessication and higher temperatures. It has also been hypothesized that disturbances from bat rays, whose feeding pits are evident at the site, lead or contribute to the annual life history response (M. Fonseca, pers. comm.). Further, it has been hypothesized that the observed annuality is genetically driven, that the plants in this area have evolved through time to exhibit this life history, perhaps as a result of the environmental stressors above or others.

In the past several years, a series of experiments at the Romberg Tiburon Center and observations and experiments in the field have led to improved understanding of the annual life history observed at Crown Beach. Experiments to evaluate this bed as a potential donor for restoration projects have led to the observation that Crown Beach plants seeded in mesocosms with constant immersion or at subtidal restoration sites persist through the winter like the perennial populations evaluated (Boyer et al. 2007, 2008). Thus, environmental factors appear to contribute to the annual life history expression observed at Crown Beach. A tidal simulator experiment in which Crown Beach plants were subjected to exposure similar to either the annual portion of the Crown Beach bed or the deeper band that persists each winter (Fig. 6) showed that exposure did not lead to mortality in the winter, although growth was slightly reduced (S. Kiriakopolos, SFSU thesis, in prep.).

Observations of intense grazing by Canada Geese in the late fall of 2007 prompted an exclosure experiment to evaluate their effects (S. Kiriakopolos, SFSU thesis, in prep.). Preliminary data from this experiment show that when the geese are excluded from plots in the annual region of the bed during the fall and winter, the plants do not suffer mortality but rather persist as perennials into the next growing season, while the unprotected surrounding areas are completely devoid of plants (Fig. 7) until seedling emergence in spring. This experiment is continuing, but results so far suggest that herbivory by geese is an important contributing factor to the annual life history observed.

Flowering shoot densities

Flowering is generally thought not to occur in *Zostera marina* within the first year of growth (DeCock 1980). In mature, perennial beds, it is difficult to track individual plants to determine whether they recruited from seed the same year. However, seed-based restoration experiments, in which we know that all plants are newly recruited from seed, suggest that flowering may be relatively common in the first year in San Francisco Bay (Boyer et al. 2008).

The Seven Site Survey shows flowering to be highly variable among beds within a single season as well as among seasons within a single bed (Fig. 8). Flowering shoot densities ranged from 0 to over $20/m^2$. While the two sampling periods (April and August) may not have captured peak flowering shoot densities at any bed, the data show a tendency for Bay Farm Island and Keller Beach to flower earlier in the season than other beds. Flowering shoot densities were relatively high at the annual bed, Crown Beach; this bed continues to increase in flowering frequency through at least September (S. Kiriakopolos, unpublished data), perhaps due to the later start from seedlings in the spring.

Recruitment from seed

While most eelgrass beds in the Bay spread clonally and persist through each winter, seedling recruitment is commonly observed among the perennial plants. In addition, the majority of the bed at Crown Beach in Alameda is annual (see above); i.e., it recruits from seed each spring after a near 100% mortality of the bed in the late fall. Genetic diversity, as measured by heterozygosity and allelic richness, is similar in the annual portion of the Crown Beach bed in Alameda relative to five perennial beds that have been compared to date (Ort et al. in review). The fact that this annual bed, which relies on sexual reproduction for population recruitment each year, does not have markedly greater genetic diversity may indicate that the perennial beds also experience substantial recruitment from seed, as also suggested by data on flowering densities (see above).

Recovery from disturbance

In the spring of 2003 a field experiment was initiated to determine the rate of eelgrass recovery following a known disturbance (Fonseca et al. 2003). Three m^2 plots were randomly selected at two locations in the Central Bay (Pt Pinole and Pt. Molate). Eelgrass (leaves, roots and rhizomes) was completely removed from each plot and recovery from rhizomal spread or seed release was tracked in the spring and summer of 2004 and spring of 2005. As a control the surrounding bed was sampled using the arbitrary toss of a m^2 sampling frame. The experiment was terminated in the spring of 2005. Results of this study are still being analyzed, but preliminary analysis shows that beds recovered substantially through vegetative expansion within five months after plots were cleared; percent recovery (the number of shoots in formerly cleared plots) relative to ambient densities was 64-82% (Fonseca et al. 2003).

Recommendations

Seasonal and interannual patterns show a high degree of variability in the shoot densities monitored in the Seven Site Survey, and repeated sidescan surveys within a single site (Emeryville Flats) over many years support that beds in the bay are fluctuating to a great degree both in bottom cover and density. Long-term monitoring is needed for recognizing the extent of these fluctuations. Continued monitoring of both types should be used to assist in establishing goals for restoration sites. Still, it should be clear that even with this somewhat intensive monitoring, two times points for measuring densities within a year or a single time point for sidescan surveys per year represent merely a snapshot of patterns in these beds. More frequent (monthly or even biweekly) monitoring would be needed to determine the peak of seedling recruitment, the phenology of clonal recruitment and flowering, changes in densities due to disturbances, etc.

A better understanding of the annual life history expression at Crown Beach has been achieved in recent years. That plants originating from this population in restoration settings persist through the winter suggests environmental conditions at Crown Beach are important to annuality there, and herbivory by Canada Geese appears to be a contributing factor. We recommend further evaluation of the factors controlling this annual life history expression. If this bed is prone to disturbances (geese or other) due to its shallow distribution, this would provide important insight into limitations of shallow distributions for restoration projects and eelgrass in general in the bay.

E. Potential limiting factors to existing populations/restoration projects

Light

To quantify the influence of turbidity on eelgrass growth in the San Francisco Bay, Zimmerman et al. (1991) conducted experiments at five sites in Central San Francisco Bay, starting in the spring of 1988. Using a combination of field measurement

and laboratory analysis this study found there was a high degree of variation in available light between sites and, in general, eelgrass was adapted for survival in low light, turbid environments. The study theorized that reduced eelgrass growth, detected by Fredette et al. (1987) coincided with very turbid conditions associated with increased river flow in the spring and suggested that if this was a chronic condition, it represented more of a threat to eelgrass survival than persistent low light conditions (Zimmerman et al. 1991).

In response to the Zimmerman et al. (1991) findings, the U.S. Army Corps of Engineers (ACOE), San Francisco District, required continuous submarine light monitoring to determine if dredging necessary to deepen the shipping channel in Richmond Harbor would degrade water clarity over eelgrass adjacent to the channel to the degree where decreased growth or coverage would result (CH2MHill 1998). In addition to monitoring within Richmond Harbor, the program also included data collection at Keller Beach, a reference site considered to be outside the influence of dredging activity. Data was recorded nearly continuously at the Keller Beach site form October 1997 to April 1998. H_{sat} was computed for each day during the sampling interval but exhibited extreme variability; however seasonal signals were evident (CH2MHill 1998). As implied by Zimmerman et al. (1991), this study also found a correlation between rainfall events and lower estimates of H_{sat} at reference and harbor sites. In fact, a common feature of heavy rainfall (>2.5 cm d^{-1}) was H_{sat} values of 0.0, an event possibly exacerbated by the presence of a storm water drain at the site (CH2MHill 1998).

To test the relationship between the reduced submarine light environment and eelgrass transplanting in San Francisco Bay, Zimmerman et al. (1995) launched an experimental transplant in Keil Cove on the Tiburon peninsula. Results from this work indicated that a reduction of carbohydrate reserves occurred in winter and early spring and was likely related to the acclimation of individual ramets to reduced PAR commonly associated with this season (Zimmerman et al. 1995). This finding suggests that the survival of eelgrass planted in winter may be at risk if turbidity events reduce the already low light available in winter and early spring.

Sediment physical characteristics

Sediment texture and organic matter are both characteristics considered to be important in the natural distribution of seagrasses and in selection of sites that result in successful restoration projects (Wicks et al. 2009). Fine sediments can be entrained in flowing water more easily than coarser sediments; thus turbidity and light attenuation can be greater in locations with high clay and silt content than with sand. In addition, there is evidence that seagrass seeds are more difficult to dislodge by waves and tidal currents when in coarse versus fine sediments (E. Koch, University of Maryland, unpublished data presented at the 2009 Coastal and Estuarine Research Federation meeting). Sediment organic content measurements can indicate the contribution of detritus to a particular site and associated low oxygen due to decomposition; it has been suggested that organic matter of $>4\%$ can indicate a site with lower suitability for seagrass restoration (Wicks et

al. 2009). Even lower organic contents (<2%) are found in the most successful restoration sites in New York where macroalgal blooms have negatively impacted restoration projects (C. Pickerell, Cornell University Marine Program, pers. comm.).

In San Francisco Bay, several extant beds sampled in 2005 had sediments lower in organic matter than the potential restoration sites at Marin Rod and Gun Club and China Camp State Park (Boyer et al. 2007). Sediments in existing eelgrass beds had ~2% organic matter on average (Fig. 9). Organic matter content was lowest at Crown Beach and Bay Farm Island (~1%), intermediate at Point Molate and Point San Pablo (~3%), and highest (~5%) at the deep edge of the Point San Pablo bed (sampled separately due to observed differences in sediment along this deeper edge). The selected restoration sites had a greater concentration of organic matter (~7%) than extant beds, with similar levels measured at the two restoration sites (Fig. 9).

In evaluations of sediment texture in the same study (Boyer et al. 2007), restoration sites were substantially lower in sand than extant eelgrass beds. Sediment cores had an average of ~90% sand across all the extant beds sampled (Fig. 9). As with organic matter, the Point Molate bed and the deeper portion of the Point San Pablo bed differed from the other sites in sediment texture, with a lower proportion of sand (~75%). Compared to sediments in extant beds, those at potential restoration sites had a lower proportion of sand on average (Fig. 9); China Camp and Marin Rod and Gun Club sediments were composed of ~65% and 40% sand, respectively.

Hence, sediments in extant eelgrass beds in San Francisco Bay tend to be much lower in organic matter and higher in sand content than sediments in a few restoration sites studied to date. These restoration sites had higher organic content and lower sand than recommended by site selection guidelines in other regions. However, it is important to note that most unvegetated regions of the Bay predicted as suitable habitat for eelgrass have relatively fine-grained sediments (Merkel and Associates 2005, Boyer unpublished data). A positive feature of fine sediments is that they have a greater capacity to retain nitrogen (as adsorbed ammonium cations) than coarser sediments, and thus a greater potential to support plant mineral nutrition. A 2006 microcosm experiment compared growth of seedlings from Crown Beach in sandy sediments from Crown Beach to those grown in relatively fine-grained sediments from Point Molate or the even finer-grained restoration site at Marin Rod and Gun Club (Boyer et al. 2007, Fig. 9). Over the course of the experiment, shoot lengths and numbers of leaves increased for plants grown in either Point Molate or Marin Rod and Gun Club sediments (Fig. 9). In contrast, seedling growth did not occur in the Crown Beach sediments and, in fact, shoot lengths tended to decline over the course of the experiment. As the Point Molate and Marin Rod and Gun Club sediments were finer-grained as well as much higher in organic content, this study suggests there is not a disadvantage to these features of the sediment *per se*. The facts that Point Molate is a naturally occurring eelgrass bed and seed-based restoration of eelgrass at the Marin Rod and Gun Club was successful in 2006/2007 suggest that finer, organic sediments in the bay should not be ruled out in restoration site selection. Support for this suggestion comes from biophysical modeling results predicting that places where flow conditions are somewhat reduced, such that resuspension is minimal, will permit

finer sediments to provide suitable habitat for eelgrass growth (Merkel and Associates 2005).

Nutrients

Seagrasses extract the primary mineral nutrients needed for growth, nitrogen (N) and phosphorus (P), in varying proportions from both the interstitial porewater and the overlying water column (reviewed in Leoni et al. 2008). For *Zostera marina*, most N assimilation occurs in the roots when there is ample light (Zimmerman et al. 1987), but under low light conditions a shift to water column uptake can occur, because uptake by roots and rhizomes is photosynthesis-dependent (Pregnall et al. 1987). N is the primary limiting nutrient in temperate seagrass systems (Howarth and Marino 2006); however, in turbid waters, limited light for photosynthesis is a greater concern, as plants are not able to utilize N despite ample supply. In *Zostera* species (*marina* and *noltii*), N may be assimilated, but cannot be used for growth and is thus stored, as has been observed in increased tissue N concentrations under experimental shading conditions (van Lent et al. 1995; Holmer and Laursen 2002; Peralta et al. 2002).

In general, the highly turbid waters of San Francisco Bay lead primary producers to be light limited, such that nutrients are a secondary concern. Seagrasses are expected to be N-limited if the foliar tissue N content is lower than 1.8% of dry weight (Duarte 1990). Preliminary data for eelgrass in San Francisco Bay show tissue N content ranging from 1.9-3.6% among eight beds evaluated (K. Boyer, unpublished data from 2008); while such percentages permit only a rough evaluation, these data suggest that N is probably not limiting to eelgrass growth at the present time. Preliminary experimental data suggest that N does not limit plant growth under low light conditions common in the bay, but is secondarily limiting; i.e., plant growth rates increase with added nitrogen only when light limitation is alleviated (G. Santos, SFSU thesis, in progress).

Excess anthropogenic nutrients can lead to macroalgal blooms and increased epiphyte loads on eelgrass blades, which can limit light, nutrient, and oxygen availability (Moore and Wetzel 2000; Short et al. 1995); however, these producers may also be limited by light in San Francisco Bay (see Epiphytes and Macroalgae, below) as is thought to be the case for phytoplankton. If submarine light availability increases with decreased sediment loads in the coming years (see sections on Light and Sediments), increased ability to utilize N may shift relative abundance of primary producers and associated functions in these eelgrass beds.

Sediment microbes

There is evidence that microbial action in the rooting zone can influence seagrass growth. Milbrandt et al. (2008) found that an intact microbial community was important in minimizing short-term stress associated with *Thalassia testudinum* transplanting in Florida. Further, Capone and Budin (1982) found root and rhizome associated nitrogen-fixing bacteria may subsidize nitrogen supply to *Zostera marina*. Experiments are needed to determine whether microbial communities should be manipulated to enhance restoration success in San Francisco Bay.

A mesocosm study begun in 2005 at the Romberg Tiburon Center tested the effects of including the microbial community from eelgrass donor sites on eelgrass seedling recruitment, clonal spread, and growth (Boyer et al. 2007). This was part of an experimental test of buoy-deployed seeding (for further information, see Section IV, Restoration Techniques, Seeding) using three donor populations (Point Molate, Crown Beach and Bay Farm Island; Fig. 3); 3 of the 6 tanks assigned to each donor were inoculated with sediment (five small cores, 5 cm diameter and 5 cm deep) collected from the site at the time of flower collection. Inoculation with donor bed sediments led to a significant increase in seedling recruitment across sites (Fig. 10). In addition, shoot growth rates and maximum shoot lengths tended to increase with donor-site sediment inoculation. The increase in shoot densities through clonal growth resulting from donor site sediment addition was pronounced throughout the growing season (Fig. 10). Effects were somewhat reduced, though still significant overall, by the end of the experiment in early December.

We suggest that the enhanced performance of eelgrass in mesocosms that received the inoculation treatment is attributable to increased microbial activity, perhaps through bacteria-mediated nitrogen fixation. However, if increased nitrogen fixation through addition of nitrogen-fixing bacteria in the added sediments is the mechanism, the importance of such an effect in finer sediments with higher available nitrogen remains unclear and demands further testing before sediment inoculation is recommended as a field protocol.

Ephiphytes and macroalgae

Epiphytes, typically diatoms and macroalgae that attach directly to seagrass blades, can negatively impact seagrasses through shading of photosynthetic surfaces of the leaves and interference with foliar nutrient uptake (Short et al. 1995; Moore and Wetzel 2000; Cornelisen and Thomas 2004). Epiphyte biomass is typically quite low on eelgrass in San Francisco Bay compared to other systems such as Chesapeake Bay (Carr et al. in press; G. Santos, SFSU thesis, in progress). At current abundances, we do not believe that epiphytes represent a major stressor to eelgrass growth or persistence at natural or restored sites. Experiments are in progress at the Romberg Tiburon Center to evaluate the degree to which epiphyte growth increases under different nutrient and light regimes; results should help to predict conditions under which epiphytes might play a greater role in eelgrass growth and nutrient dynamics (G. Santos, SFSU thesis, in progress).

Macroalgae can limit seagrass growth by reducing availability of light (Hauxwell et al. 2001; Havens et al. 2001; Brun et al. 2003), or by drawing down dissolved oxygen during decomposition (Koch et al. 1990), which can lead to decreases in redox potential and increases in sediment sulfide and toxic ammonium concentrations (van Katwijk et al. 1997; Terrados et al. 1999; Lamote and Dunton 2006). In Tomales Bay, Huntington and Boyer (2008) found that high abundance (the average maximum biomass measured in a survey of eelgrass beds = 1700 g wet weight/m^2) of the red alga *Gracilariopsis* sp., led to

reduced shoot densities and growth rates of eelgrass in experimental field enclosures. In contrast, average densities surveyed, 325 g/m^2, produced no detrimental effects in the same experiment. In Bodega Bay, Olyarnik (2008) found abundances of the green alga (*Ulva* sp.) of ~2000 g wet weight/m^2 or less to have no measurable effects on eelgrass; however, very high biomass (4000 g/m^2) had negative effects on eelgrass density and growth.

In a quarterly survey of macroalgal biomass at four beds (Point San Pablo, Keller Beach, Crown Beach, and Bay Farm Island) in 2008-2009, Santos (SFSU thesis, in progress) measured macroalgal biomass in replicate meter-squared plots placed along a 100 m transect line. Algal biomass was collected, rinsed and cleaned of invertebrates, dried, and weighed. During this survey, algal biomass was generally quite low, and many sites yielded no biomass (Fig. 11). The greatest density recorded was at the Point San Pablo (North Richmond) bed in August 2008. This density (~120 g/m^2 dry weight or ~1200 g wet weight) approaches the range found to reduce densities and growth rates of eelgrass in nearby estuaries. Overall, these preliminary data suggest that macroalgae are generally too low in abundance to threaten eelgrass in the bay, but that abundances may at certain times and places reach levels that could be detrimental.

Herbivory

It is widely accepted that seagrasses tend to experience low levels of herbivory, due at least in part to their lower nutritional value and higher proportion of structural carbohydrates compared to algal epiphytes and macroalgae available to grazers within the beds (e.g., Cebrian and Duarte 1998). Instead, seagrasses have long been thought to contribute to food webs largely through the production of detritus. However, there are many documented grazers on seagrasses worldwide, ranging from manatees and turtles to urchins and isopods, and direct measures of grazing effects on live tissues show a wide range of impacts, from very low (3% of net primary production) to extremely high (~100%), i.e., leading to local extinction of beds (Heck and Valentine 2006; Rivers and Short 2007). Further, recent analyses suggest that methods for measuring grazing may have greatly underestimated actual values, and thus the importance of herbivory in seagrass systems (Heck and Valentine 2006).

Brant geese are known to exhibit high feeding rates on eelgrass in many other northern and central California bays (e.g., Humboldt, Bodega, Morro); however, their consumption of young leaves without disturbance of belowground structure or meristematic tissue has been shown to have positive effects on eelgrass growth, perhaps due to a thinning of the canopy (leading to increased light availability) or through fertilization by fecal matter (Frimdog 2007; F. Shaughnessy and J. Black, Humboldt State University, unpublished data). Brant geese are not currently found in San Francisco Bay in any numbers, although there have been observations of a single pair recently in the vicinity of the Richmond Marina near Brooks Island.

To our knowledge, there have been no observations or experimental tests of herbivory on

Zostera marina in San Francisco Bay until very recently. Two herbivores may have large, localized impacts on eelgrass: an introduced amphipod, *Ampithoe valida*, and Canada geese, *Branta canadanensis*.

Ampithoe valida is a gammaridean amphipod from the eastern US, thought to have been introduced to San Francisco Bay in the 1920's. On two collection dates (July 2005 at Point San Pablo and August 2006 at Crown Beach) for flowering shoots for seed-based restoration, observations of extremely high densities of this amphipod were made concomitant with observations of a high degree of damage to flowering shoots and seeds (K. Boyer and L. Reynolds, pers. obs.). A preliminary test of the feeding preferences of this amphipod were made in a microcosm experiment in 2006 (Boyer et al. 2007; Fig. 12). Briefly, to each of 20 small containers (240 mL, 10 cm diameter) with a 2 cm layer of sand, a randomly selected spathe was added at each of the following flowering stages (DeCock 1980): early (stage 0-1), maturing (late stage 4), and decomposing (stage 6; seeds gone from spathe). In addition, each container held one leaf from a vegetative shoot and 5 ripe seeds. To ten of the containers, 50 *Ampithoe valida* were added, while the other ten served as controls. All plant material was weighed before and after the experiment, and seeds on the sediment were counted. In addition, the number of seeds present on the stage 4 spathes at the end of the experiment was noted. Figure 12 shows that there was consumption of a small amount of vegetative plant material by amphipods, but not of stage 0-1 spathes. Stage 4 and 6 spathes were clearly the preferred food in the microcosms, as there was 3-4 times less tissue on these spathes when amphipods were present compared to controls. Seeds placed on the sediments were not eaten; i.e., the same number was present at the end of the experiment with or without amphipods present. Interestingly, the numbers of seeds left on the stage 4 spathes was greatly reduced when amphipods were present; as these were not found on the sediments, the amphipods must have consumed them. These results suggest that *Ampithoe valida* was capable of the damage observed to the maturing (late stage 4 and stage 5) spathes in the field. This amphipod may preferentially consume tissues of maturing and decomposing spathes, and also appears to consume seeds when they are still on the spathes. However, abundance of these amphipods varies dramatically among extant (potential donor) beds, as well as seasonally and interannually (Carr et al. in press; K. Boyer, pers. obs.).

Canada Geese can be observed feeding on eelgrass tissues in many of the beds in the bay (e.g., Keil Cove, Keller Beach, Crown Beach). At Crown Beach, abundances in fall can be quite high, perhaps in the hundreds; biweekly counts are underway to evaluate seasonal patterns in abundance (S. Kiriakopolos, SFSU thesis, in prep.). Detailed observations suggest that these geese feed on eelgrass blades to some degree but primarily damage plants by digging their bills into the sediment and removing shoots beneath the meristematic (growing) region (S. Kiriakopolos, pers. comm.; Fig. 7, also see Section IID, Perennial versus annual life history). This damage may be largely responsible for loss of perennating structures in the majority of this bed by winter (see above, Perennial versus annual life history), leading to a reliance on seeds for re-colonization of the population the following spring (i.e., the observed annual life history). Why this particular bed attracts such large numbers of Canada Geese compared to other beds is uncertain, but may be related to the shallow depth of the majority of the bed

(perhaps combined with its sandy sediments which may more easily support these large-bodied birds than finer sediments), the proximity to *Spartina* stands on which the geese also graze (T. Grosholz, pers. comm.), or other factors.

Physical disturbance

Disturbances to seagrass beds can result from human activities such as boating, which can cause scarring of beds from anchors, motors, and mooring lines (Short and Wyllie-Echeverria 1996). Large ships, tug boats, and ferries can produce wakes that disturb sediments and perhaps erode beds. In other regions, polycheates*,* rays, and crabs have all been found to negatively affect seagrass plantings (Davis and Short 1997; DeWitt and Wyllie-Echeverria, in prep.).

In San Francisco Bay, winter storms have been observed to erode sediments in some areas, exposing rhizomes and uprooting some plants at the Point San Pablo Bed (authors' pers. obs.). Sediment accretion can be quite high in some areas of the bay and may have been responsible for failure of one experimental planting at the Albany Mudflats area (Boyer 2008).

Scarring is prevalent in active boat regions such as Sausalito, and mooring line "crop circles" are also evident there. It is not uncommon to see large scars through the beds that occur on offshore shores, i.e., Point San Pablo and Bay Farm Island. A barge used in the clean-up of the Cosco Busan oil spill in 2007 left a large scar through the Keil Cove eelgrass bed. Restoration sites, if not marked by buoys, are likely to also experience damage from small boats.

Large wakes can be observed in some of the extant beds in the bay, including Point San Pablo (North Richmond) and Point Molate (authors' pers. obs.). It is unclear whether they have negative impacts. These wakes, especially from deep draft tugs and swiftly moving ferries, are strong enough to knock over the unsuspecting seagrass researcher. In the restored bed at the Marin Country Day School in Corte Madera Bay, it is very difficult to keep marker buoys in place, presumably due to abrasion of ropes under frequent disturbance by the Larkspur Ferry (K. Boyer, pers. obs.).

Bat rays produce feeding pits within eelgrass beds in the bay. It has been hypothesized that this feeding activity negatively affects eelgrass seedlings, and this possibility has been proposed as a factor contributing to the annual life history observed in the majority of the eelgrass bed at Crown Beach (M. Fonseca, NOAA, pers comm.). Recent studies suggest that a different disturbance, that of Canada Geese removing whole shoots below the meristem, plays a role in the observed annuality at Crown Beach (S. Kiriakopolos, SFSU thesis, in prep.).

Disease

Adverse environmental conditions such as reduced submarine light and high temperature can trigger an epidemic infection of a virulent slime mold (*Labrinthula zosterae*) that leads to a condition known as wasting disease (Short et al. 1987; Muehlstein et al. 1991; Vergeer et al. 1995). This disease can spread by leaf-to-leaf contact and destroy large areas relatively quickly (Short et al. 1987). Disease events have occurred in several locations but the most well know occurrence took place on both sides of the North Atlantic in the 1930s (Renn 1934; Rasmussen 1977, Muehlstein 1989; Short and Wyllie-Echeverria 1996).

To date there is no evidence of a disease outbreak in San Francisco Bay but, because reduced submarine light is known to occur and higher temperatures are a predicted element of climate change, the monitoring of eelgrass populations for disease presence should be considered.

Salinity

The mixing of marine and fresh water determines the salinity environments in coastal and estuarine environments (e.g., Bowden 1967) that can be influenced by seasonal rainfall and snowmelt which can, in turn, influence seagrass distribution patterns (Short and Neckles 1999). There are no studies in San Francisco Bay to date that link changes in extant *Zostera marina* beds to a variation in salinity values. However, the distribution of eelgrass in the Bay may shift as salinity increases up the estuary with sea level rise.

Temperature

Temperature can be very important to the initiation of flowering shoots, which develop through metamorphosis of a mature vegetative shoot (DeCock 1980; Thayer et al. 1984). Along the Atlantic coast of the US, immature flowering structures have been observed at temperatures below 5º C but for further development of reproductive structures, water temperatures must reach from 10-15º C, and water temperatures of 15-20º C lead to completion of flower development and production of seeds. In San Francisco Bay, Setchell (1929) observed that 15º C water temperature was necessary to begin sexual reproduction.

Sustained high temperatures (>30º C) in July-August 2005 in Chesapeake Bay are believed to have been largely responsible for a massive, nearly complete dieback of eelgrass (Moore and Jarvis 2008). In San Francisco Bay, Setchell (1922, 1929) suggested that water temperature influences distribution and phenology of eelgrass and that high temperatures (~30º C) can cause shoot mortality. On the other hand, temperatures up to 35º C for 4-5 hours at a time produce no noticeable damage in New Hampshire eelgrass beds (F. Short, University of New Hampshire, pers. comm.). Temperature data loggers at Crown Beach and Richardson Bay in 2007-2009 occasionally recorded temperatures >30º

C, but this was an unusual occurrence never repeated on consecutive days (S. Kiriakopolos, SFSU thesis, in prep.). Global warming will shift water temperatures upwards in the coming decades, but it is unclear how soon this will be a concern for eelgrass health in the bay. Additional water temperature data is needed, as well as experimentation to test the effects of increased temperature, particularly in combination with other current and potential stressors.

Sea-level rise

Short and Neckles (1999) identified five potential effects of sea level rise on seagrass distribution: reduced light as a result of increased depth, increased tidal range, salinity intrusion into predominantly fresh water environments, salinity influence on asexual and sexual reproductive phenology, and stress from elevated salinities. Based on our current understanding of eelgrass distribution and growth patterns, it seems likely that sea-level rise will move the maximum depth of eelgrass growth closer to the current shoreline. The impact of this potential has not been quantified.

Although only conjecture at this point due to the unknown influence of local effects, if the up-estuary environments of San Francisco Bay become more saline due to rising seas, eelgrass populations may colonize new environments.

Genetic diversity

Efforts to maximize genetic diversity, such as collecting transplants for restoration over large areas of a donor bed, are expected to increase the success of seagrass restoration efforts (Williams 2001; Procaccini and Piazzi 2001). Increased genotypic diversity, as tested by inclusion of multiple genotypes in experimental plots, has been found to enhance resiliency to disturbances such as high temperatures and Brant geese herbivory (Hughes and Stachowicz 2004; Reusch et al. 2005). Hence, methodologies that maximize genetic diversity should be encouraged in restoration projects. Sufficient spacing of restoration collections to include different individuals can help to promote genetic diversity of donor stock. Because seeds provide enhanced diversity via sexual recombination, they are inherently a greater source of variation in comparison to vegetative shoot transplants. Practitioners in other regions often add seed to supplement whole-shoot transplants to increase the potential for genetic diversity in restored populations (C. Pickerell, pers. comm.). As extant populations in San Francisco Bay show significant genetic structure (Talbot et al. 2004; Ort et al. in review), it may be advisable to utilize multiple populations as donors in a single restoration site in order to improve the probability of including genotypes suitable for that site (Ort et al. in review). However, information about the genetics of the potential donors and the habitats to be restored should be used to inform choices of specific donors or donor mixes for any project.

Recommendations

As recognized in many other regions where seagrasses grow, light is probably the single most important limiting factor to eelgrass in San Francisco Bay. However, as noted by Zimmerman, prolonged periods of turbidity may be more important in determining success of eelgrass beds than average light availability. This fact bears consideration when evaluating sites for suitability for eelgrass restoration. Most areas available for restoration in the bay have finer sediments than do most extant eelgrass beds, but these sediments may be helpful in supporting establishment of plants due to their higher nitrogen availability. The test of restoration sites on fine sediments, existing and future, will be their ability to persist through periods of high turbidity resulting from rainfall and high winds. Mesocosm experiments to test eelgrass growth under differing periods of low light would be useful in supporting this evaluation. In addition, we recommend establishment of a bay wide water quality monitoring program capable of determining site specific diffuse attenuation coefficients (K_d). This program would contribute to the creation of restoration criteria at sites not currently under consideration (vis. a vis., the Chesapeake Bay "exclusion zone" delineation; Dennison et al. 1993). Recently documented and predicted continued increases in light availability may have implications for nutrient dynamics in eelgrass beds. Nutrients are probably not limiting to eelgrass at this time, and abundances of other primary producers such as epiphytes and macroalgae within the beds are currently low relative to other regions. However, increased light penetration could lead to enhanced ability to utilize excess nutrients, and shifts in both light and nutrient availability could lead to changes in the relative abundance of eelgrass and the other producers in eelgrass beds. Experiments underway by G. Santos (SFSU thesis, in prep.) should be useful in helping to predict how communities will shift under different scenarios of light and nutrient availability, and should suggest additional research to improve our understanding of these dynamics.

Inoculation of restoration site sediments showed promise in increasing both seedling recruitment and clonal spread in a mesocosm experiment conducted with purchased sand (Boyer et al. 2007). Additional work is needed to determine whether this should be recommended as a practice at restoration sites. Specifically, an assessment of benefits should be made following inoculation with donor bed sediments of finer grained sediments taken from potential restoration sites.

Effects of herbivory and disturbance by animals such as bat rays observed in the field and in preliminary experimental tests warrant further exploration. The evidence that an introduced amphipod targets the spathes of flowering shoots with maturing seeds as well as the seeds has important implications for successful sexual recruitment and bed maintenance. Surveys of amphipod abundance and grazing effects are needed to help determine the extent of effects. We also recommend that future restoration projects consider the abundance of these amphipods when planning flowering shoot collection and that multiple potential sites be monitored to insure both appropriate timing and availability of flowering shoots for seeding activity. In addition, distribution and abundance of Canada Geese should be tracked at Crown Beach and perhaps other sites, and the effects of their consumption of eelgrass further elucidated. Further, experimental

bat ray exclusion should be employed to evaluate the effects on eelgrass restoration, especially at early stages of establishment.

Temperatures known to have detrimental effects on eelgrass growth and persistence occur at least occasionally in San Francisco Bay according to limited data to date. We recommend the placement of data loggers in the shallow depths of multiple existing eelgrass beds to assess the degree to which conditions lead plants to reach their thermal tolerances. This is especially important for predicting responses to increased temperatures in a changing climate. Finer grained areas with higher organic content should be included in such measurements as darker surfaces may lead to increased temperatures in shallow areas.

Genetic diversity at restored sites can and should be enhanced through the use of seeding methodologies (or supplementation of transplants with seeding), by choosing donors with high genetic diversity, by collecting propagules at distances that maximize inclusion of different individuals (e.g., ≥10 m spacing; Ort et al. in review), and possibly through the use of multiple donor populations for seed or transplants if deemed appropriate considering the specific circumstances of the restoration project.

III. Ecosystem functions of eelgrass

Habitat provision and food web support

Eelgrass is well known for its role as a foundation species (sensu Dayton 1972; Kenworthy et al. 2006), i.e., a species whose presence results in provision of significant habitat and food web support for myriad other species. Numerous invertebrate species are harbored among the blades and inflorescences of *Z. marina*, including amphipods, isopods, and copepods (Kitting and Wyllie-Echeverria 1992, Hanson 1998). Figure 23 shows abundances of these epifauna seasonally among five beds in San Francisco Bay in 2007; numbers are high compared to those from studies in other regions (Carr 2008, Carr et al. in press).   These invertebrates provide food resources for resident fishes such as the bay pipefish and shiner surfperch (L. Carr and K. Boyer, unpublished data). Eelgrass is known to serve as spawning and nursery habitat for Pacific herring (Spratt 1981), the primary commercial fishery species in the bay. Eelgrass in the bay is thought to provide food and shelter for outmigrating juveniles of several diadromous fish species as in the Pacific Northwest (Simenstad 1994). Many birds forage for invertebrates, fish, and fish roe in the bay's eelgrass beds, particularly during winter and spring migration, including Forster's, least, and elegant terns, double-crested cormorants, and several shorebird and diving duck species (S. Wainwright-de la Cruz, pers. comm.).

Many studies in other regions have shown an increase in the abundance and species richness of fauna inside compared to outside eelgrass beds (Orth 1977; Hoffman 1991). A study currently underway by Richardson Bay Audubon suggests increased invertebrate and fish use of eelgrass patches in comparison to surrounding areas (W. Norden, S.

Olyarnik, Richardson Bay Audubon, pers. comm.). Increasing eelgrass acreages should enhance food and habitat for numerous organisms at multiple trophic levels. Samples currently being processed are showing high numbers of recruiting invertebrates to eelgrass restoration sites at Marin Rod and Gun Club and Marin Country Day School (K. Boyer and L. Carr unpublished data). Bay pipefish were observed at Marin Rod and Gun Club within the first year of eelgrass recruitment from buoy-deployed seeding. Tagged Chinook salmon and steelhead have been detected (by acoustic monitoring devices) lingering among eelgrass and oyster shell reefs constructed at the Marin Rod and Gun Club (R. Abbott, Environ Corp., unpublished data).

Sediment stabilization

Seagrasses, through root and rhizome structures and reduction in water flow among shoots, can help to stabilize sediments (e.g., studies of *Z. marina* in Chesapeake Bay and *Thalassia testudinum* in Bermuda; Orth 1977). To our knowledge, there have been no local studies of eelgrass effects on sediment stability. San Francisco Bay beds tend to be patchier than many other locations were *Z. marina* occurs; however, the degree to which this distribution influences the plants' ability to hold sediments in place is unknown. If local growth patterns are conducive to stabilizing sediments and reducing erosion, then it may be important to restore beds to areas of the bay where erosion is a concern.

Water flow reduction and sediment accretion

Seagrasses are known for their important role in reducing water flow relative to surrounding unvegetated areas (e.g., Fonseca et al. 1982; Gambi et al. 1990), which can in turn reduce erosion of sediments and increase recruitment of fauna. Baffling by eelgrass shoots can lead to deposition of sediments, which drop from water as the flow is slowed (Orth 1977). To our knowledge, there have been no local studies of the effects of eelgrass on water flow and sedimentation. However, still water is one way in which researchers find eelgrass beds when approaching by boat, and flow reduction is visible around even small eelgrass patches (Fig. 13).

Recommendations

1) Habitat dependency and scaling. Any increase in eelgrass acreage should increase habitat for the eelgrass dependent bay pipefish. However, it is not clear how much benefit other invertebrate or fish species derive from eelgrass compared to other habitats. While salmonids pause to spend time in areas restored with eelgrass, would these fish use any three-dimensional structure available or do they actually prefer eelgrass for the food or shelter it provides? Similarly, herring spawn on macroalgae (e.g. *Gracilaria* sp.) and rocks as well as eelgrass; does eelgrass provide better habitat, and in turn significantly increase herring reproductive success when available? If eelgrass is a preferable habitat for any of these species, how much acreage is needed to significantly increase their

fitness? We recommend that controlled experiments and surveys be conducted that can address these questions. If eelgrass does appear to increase the fitness of these species above other alternative habitat types, then it would increase the urgency with which we move forward with restoration projects as well as protection of existing beds.

2) Native and non-native species diversity. A major goal of many restoration projects is to increase the numbers of native species. The degree to which eelgrass restoration can be used to accomplish this goal in San Francisco Bay is not clear. While a number of native fishes may be benefitted by increased eelgrass acreage (see above), the vast majority of epifaunal invertebrates in extant eelgrass beds are introduced species (Carr et al. in review; Fig. 14).   These species are very abundant and many have been present in the bay for decades; it is unlikely that they can be eradicated. However, a number of the species are live brooders, meaning they do not disperse larvae in the plankton. Thus, it is possible that restoration projects that are distant from existing eelgrass beds could be designed to favor colonization by native invertebrate species. Experiments are needed to test competition among native and non-native species and whether there are "assembly rules" that guide community development depending on order of introduction (J. Lewis, SFSU thesis, in progress). Further, it should be determined whether there is any disadvantage to having non-native species in our eelgrass beds. If all the small crustacean grazers are equally palatable and nutritious to fishes, then the idea that we should try to dissuade non-native dominated assemblages comes into question. It has been suggested that systems that are highly disturbed or that have diverged from a native flora or fauna may be more suitable for use of non-native species in restoration projects; both because there may not be any choice and because non-native species may provide important services that were once provided by native species (Ewel and Putz 2004; Fig. 15).

3) *Effects on flow, sediment accretion, and sediment stability.* Eelgrass plants in San Francisco Bay are both patchier in distribution and larger than plants in many other regions. The spacing of patches may mean that effects on flow across a bed are reduced relative to more dense beds in other areas; however, the size of plants leads to a greater vertical extent through the water column when the tide is high. Surveys are needed to assess the effects of extant patchy eelgrass beds on flow and sediment accretion. Surveys could be conducted across the range of bed configurations (very patchy versus more meadow-like, narrow vs. broad, shoreline vs. offshore shoal), densities, and depths of these beds. Experimental tests of density or size of plants on flow could also be conducted in a flume setting. Surveys of sediment stability inside and out of patches and whole beds would also assist in determining the importance of this function in local beds. If these functions are enhanced by the presence of eelgrass, then it may be desirable to include eelgrass restoration, where feasible, adjacent to shorelines where erosion is a concern.

4) *Eelgrass interactions with native oyster or reef restoration.* Efforts to restore native oysters are underway, and it may be possible that restoring oyster reef structures (that hopefully attract settlement of native oysters) and eelgrass at the same sites could be beneficial. Oyster reef structures such as oyster shell in bags or stacked on pallets can

reduce flow and are sometimes used to protect shoreline vegetation from erosion. If native oysters settle on these structures there could be benefits to eelgrass through oyster filtering of particles from the water and possibly fertilization effects of oyster wastes on eelgrass. Eelgrass may also be beneficial to native oysters if it changes particle sizes available for filter feeding or enhances particulate organic matter as food. Further, there may be benefits to fish, birds, or invertebrates of having both types of habitat in close proximity to perhaps provide a greater diversity of food resources or predation refuges. While the range of light, salinity, and other factors favorable to each species differs, there is an overlap in suitable habitat for the two species. Test plots for eelgrass have found that several sites with high eelgrass transplant performance also recruited greater numbers of oysters to cement oyster recruitment stakes (Boyer 2008).

To date we have limited local evidence that there are benefits to restoring both types of habitat in proximity. A project at the Marin Rod and Gun Club is currently underway to investigate food resources for salmonids in plots with no structure versus eelgrass, oyster reefs, or eelgrass + oyster reefs (Fig. 16). Preliminary data show a trend of greater invertebrate abundance on eelgrass in its first growing season when adjacent to oyster reefs (Boyer and Carr 2009). Additional sample processing is needed to determine if this trend is significant. Acoustic monitoring devices within the whole array of plots do not permit assessment of individual versus combined effects of the two habitats on salmonid visitation, but do suggest greater visitation within the overall array of plots when compared to a large open plot with no structure present at the other end of the site (R. Abbott, pers. comm.). A previous project at the Marin Rod and Gun Club to determine if eelgrass seedling recruitment could be enhanced by the presence of oyster reefs as breakwaters was inconclusive; i.e., there was slightly higher recruitment on the shoreward side of reefs but recruitment was too low on both sides to conclude an effect (K. Boyer, unpublished data).

On the other hand, there is a possibility that these habitats will not interact or will interact in undesired ways. Research from Richardson Bay Audubon showed no increase in oyster settlement on artificial oyster shell reefs (made of oyster shells in nylon mesh bags) located next to eelgrass patches compared to oyster reefs away from eelgrass (S. Olyarnik, pers. comm.). A possible negative interaction could result if oyster settlement stakes in close proximity to eelgrass abrade and damage the plants' blades (Boyer 2008). Clearly, further studies are needed to better understand the potential for synergy among oyster reef and eelgrass restoration.

IV. Restoration techniques for eelgrass

Whole shoot transplants

Bare root planting units. Transplantation of bare root vegetative shoots is a common method for seagrass restoration worldwide (Fonseca et al. 1998). Vegetative shoots are collected with a portion of the rhizome, then planted singly or in bunches, with or without

anchors such as nails, u-shaped metal sod staples, or pieces of steel reinforcing bar (Churchill et al. 1978; Phillips 1990). Biodegradable anchors are often preferred, and include bamboo skewers (Davis and Short 1997; Fig. 17) and popsickle sticks (Merkel and Hoffman 1990). The Horizontal Rhizome Method is commonly employed; it entails using bent bamboo skewers to secure two shoots with rhizomes oriented in opposite directions (Davis and Short 1997). Automated methods of planting are also being tested (Fig. 17).

In San Francisco Bay, whole shoot transplants have been used for several restoration projects or experiments on a small scale with varying success. Results from an experimental transplant in the Richmond Harbor were not encouraging. Preliminary planning for this effort began in the April 1984 with plant harvest and planting following in April of 1985 (Fredette et al. 1987). These studies indicated that eelgrass transplanting should be approached with caution in San Francisco Bay. This warning was put forward to resource agencies because (a) transplant success was marginal; (b) preliminary evidence suggested that annual populations of eelgrass might occur in San Francisco Bay (annual populations are known to exist in Baja California, Mexico and Yaquina Bay, Oregon on the Pacific coast [Wyllie-Echeverria and Ackerman 2003]) and (c) time-series information delineating the population ecology of eelgrass was lacking for San Francisco Bay. Nonetheless, the experiment contributed directly to important ecological considerations for extant eelgrass populations with studies conducted at the transplant donor site: the Richmond Long Wharf (aka Chevron Pier). After harvest, this site was monitored as a control by tracking shoot density along a depth gradient from shallow to deeper water. Control densities were highly variable during the course of the experiment. For example, shoot density in April 1985 was almost 10 times greater than that observed in July 1985, whereas shoot densities for September 1985 and May 1986 were 30% and 48% of the April values (Fredette et al. 1987). When rhizome branching frequency data were examined for the sampling year, virtually no branching was detected during the April 1985 to July 1985 time period. Conversely, after July, branching became more frequent, decreasing in winter and increasing again in the spring of 1986. Because branching frequency is an important indicator of vegetative colonization potential (every shoot in *Z. marina* is an apical that contributes to occupation of space as the shoots migrate across the sediment, leaving an interwoven rhizome mat behind), we can assume that an event or series of events induced stress to the population sometime during the April 1986 to July 1986 sampling interval. It is also noteworthy that increased shoot density in 1986 was most evident in the deeper stations along the control transect. Submarine light levels can be lower in deeper waters, a finding corroborated by Zimmerman et al. (1991) at Paradise Cove just to the west of the Richmond Long Wharf site. A significant finding of Fredette et al.'s (1987) was that flowering shoots were observed in clones of recently germinated seedlings. While the dimorphic expression of vegetative and flowering shoots is common in the genus *Zostera* (den Hartog 1970) when plants are in their second season of growth, its association with early life history stages generally signals the presence of an annual population (Phillips and Backman 1983).

In the final analysis, the experiment at the Richmond Training Wall identified knowledge gaps and provided direction for continued ecological evaluation of the eelgrass resource.

For example, it was clear that the lack of information on life history including seasonality and percent frequency of flowering, seedling ecology and vegetative growth rates, were a severe impediment for the planning of restoration projects, as it has been elsewhere (Fonseca et al. 1998). Also obvious were deficiencies in an understanding of relationship between nearshore environments and eelgrass survival within the Bay. In this regard, knowledge of the influence of the observed, but not quantified, turbid and sometimes very low salinity Bay waters on the geographic range and depth distribution of eelgrass was needed. Without such information, we cannot determine whether declines at Richmond Long Wharf between April and July 1985 were an anomaly or a typical seasonal fluctuation. Either way, identification of such periods of potential stress constitute a fundamental need in the planning (timing) of eelgrass restoration projects (Fonseca et al. 1998).

Merkel and Associates (2006) conducted bare root transplants using planting bundles (shoots attached by their rhizomes with string to a paper stick) at the Emeryville Flats, adjacent to the San Francisco-Oakland Bay Bridge. Survival varied among locations within the study area, but averaged ~35% after 48 weeks. Initial establishment was greater using larger bundles (8-20 shoots) compared to smaller bundles (2-6 shoots), but differences in performance declined with time. The authors recommended that 6 or more shoots be included in a bundle to maximize success. Trimming leaves did not increase rates of survivorship as had been hypothesized due to the damage sustained by eelgrass leaves during transplantation. Bundles collected from four donor locations (Bay Farm Island, Brooks Island, Emeryville Flats and Keil Cove) and transplanted into Emeryville Flats did not differ in percent survival or basal area (Merkel and Associates 2006).

In 2007, SFSU graduate student Stephanie Kiriakopolos developed a new technique that has been used effectively in several small-scale restoration projects around the Bay (Boyer 2008; Boyer and Carr 2009; S. Kiriakopolos thesis, in prep.). In this method, individual vegetative shoots are wrapped loosely at the base with a small piece of burlap and secured ~15 cm from the top of a bamboo stake with a paper-covered wire twist-tie (Fig. 18) (Richardson Bay Audubon is currently testing a similar design substituting hemp twine for twist-ties; S. Olyarnik, pers. comm.). The bamboo stakes are pushed down into the sediment so that they emerge ~10 cm (with 35 cm beneath the sediment surface), thus permitting good root contact with the sediment while securely holding the plants in place until roots establish. The long stake used in this method may be an improvement over other bare root planting methods as it holds plants in place very well, even in unconsolidated sediments, until they are well rooted. However, it should be noted that site selection is important for any restoration method, and plants did not establish at one of the sites where this method was deployed in small scale test plots (Albany mudflats; Boyer 2008). Further, in considering different bare root methods, this technique has not been compared directly to others such as the bundled shoot transplants described above.

Planting frames of mesh fabric anchored with steel pins (Homziak et al. 1982) or wire frames anchored with bricks (Short et al. 2002b) have been used to increase the numbers of vegetative shoots that may be planted at a time. The latter design, known as TERFS

(Transplanting eelgrass remotely with frames system; Short et al. 2002b) is being used currently with success in some regions of the US. These are frames to which vegetative shoots are attached using paper ties, which hold the plants in place just long enough for the frames to be placed at the restoration site. The typical TERFS frame used in the Northeast US is a rigid metal grid fabricated from lobster pot materials (Fig. 19). The frames are held in place with bricks to maintain contact of roots with the sediment until the plants are securely rooted at the restoration site (4-6 weeks), after which the frames are removed. These frames are recommended to reduce bioturbation and erosion around transplanted shoots (Short et al. 2002b) and thus are an attractive restoration tool for SF Bay as erosion has been implicated in poor establishment rates in transplanting efforts (Merkel and Associates 1999) and bioturbation has been noted to be quite high in some vegetated areas (M. Fonseca, pers. comm.). As noted by Short et al. (2002b), use of TERFS allows efficient use of community volunteers, who can help with assembly on land. Criticisms of the technique include that the frames are heavy to deploy and retrieve, and that because well-established shoots will have spread clonally across the frame in the month+ period the frames are left in the field, some shoots are torn back out of the restoration site when the frames are removed. Success rates can be quite high (47-86%) using this method in subtidal areas in New England; TERFS outperformed the horizontal rhizome method in a side-by-side comparison at one site (Short et al. 2002b).

The TERFS design described above has not been used in San Francisco Bay, but two modified versions of TERFS have been tried in the last few years. In 2006, scientists at San Francisco State's Romberg Tiburon Center transplanted whole shoots using frames comprised of a plastic mesh material (Vexar) to which bricks were attached by cable ties (Fig. 19). Paper ties were used to attach 15 adult shoots to each of the 50 x 50 cm frames. In July 2006, these modified TERFS were deployed at three restoration sites. While more affordable and easy to procure in this region than the lobster-pot frame material of the original design, the combination of the paper ties used, which disintegrated very quickly (as planned), and the flexible Vexar frames, which did not securely hold the shoots in place until rooting, led to poor plant establishment. As shoot density the following March was on average only 6% of planted density, this version of the TERFS is not recommended for further use (Boyer et al. 2008).

Another modified version of the TERFS frame, consisting of 1" x 1" square wooden dowels and hand-constructed with burlap mesh and fiber twine (Fig. 19), was tested in a mitigation project at Clipper Yacht Harbor in Sausalito in 2007. This version, developed by the consulting company, WRA, has the advantage of being biodegradable, and was developed in consultation with NOAA resource managers to permit leaving the frames in place, thus avoiding potential damage to established shoots that can occur upon TERFS removal. While time-consuming to make, transplants using this method appeared relatively successful; 5 out of 6 frames had plants present after one year, with an overall survivorship averaging 20% (WRA 2008). These plants are believed to have persisted and spread after two years, although clonal recruitment from the adjacent areas could not be distinguished from transplants within the frames (WRA 2009). While this modified TERFS design was probably even more flexible than the Vexar used in the SFSU project, the use of twine instead of paper ties likely held shoots in place long enough for rooting

of a larger number of shoots to occur.

Sods

Transplantation of sods, chunks of sediment containing seagrass shoots and belowground tissues, is commonly used for restoration in many areas (30% of projects reviewed by Fonseca et al. 1998; Fig. 17). These are usually small plugs removed with PVC pipe, metal cans or similar (Phillips 1990; Fonseca et al. 1998), or shovel-sized sections removed by hand with a flat blade shovel (Addy 1947; Churchill et al. 1978). Efforts to move very large units (0.25 to 1 m^2) are also underway using mechanized planting machines to accomplish greater areas of transplants while preserving greater numbers of intact meristems (Paling et al. 2001 a, b; Uhrin et al. 2008). Sods have the advantage that they are less susceptible to erosion and bioturbation than bare root planting units and that transplantation of donor sediments around root structures can be less stressful to the transplants. The disadvantage is that removal of larger sods (shovel-sized or larger) from donor beds leaves gaps and these can lead to erosion of adjacent areas of the bed.

Several studies have evaluated the effectiveness of sods locally. First, a mesocosm experiment compared small plugs to bare root transplants, and found greater survivorship with plugs (Josselyn and Alberte 1990). Plugs were used successfully to transplant at both Point Molate and Keil Cove in a study during the same period (30-60% survival after one year), although effectiveness was not compared with other restoration techniques (Zimmerman et al. 1991). Experimental tests of plugs in support of the Oakland Middle Harbor Enhancement program could not proceed, as sandy sediments used in the plugs did not remain intact long enough for transplantation (Merkel and Associates 1999). A later experiment comparing the use of plugs to bare root transplant bundles (8 shoots anchored with paper sticks and string tied around the rhizome bundle) found greater survivorship with the bundles, although this conclusion is limited by the facts that plugs typically only contained a single shoot and were placed into poorly consolidated sediments without anchors (Merkel and Associates 2004b).

It is possible that small sods (plugs ~6" in diameter used in the studies above) could be effectively used for eelgrass restoration in San Francisco Bay under certain conditions (e.g., sediments in plugs and transplant location with sufficient fine sediments to remain intact), and we suggest that further experimental tests are needed. We do not recommend use of larger sods (shovel sized and larger) employed in other regions, as most of the potential donor beds in San Francisco Bay are relatively small and their protection is a high priority.

Seeding

Seed-based restoration methods may help to alleviate the problem of genetic diversity decreases in previous restoration attempts using whole shoot transplants (Williams and Davis 1996; Williams 2001) and can be less time consuming than whole shoot transplant

methods. Seeding methods are generally of two types, hand-broadcasting and buoy-deployed seeding. In the first, flowering shoots are collected and placed in tanks to permit mature seeds to drop, and these seeds are broadcast by hand at a restoration site (Granger et al. 2002; Fig. 20); this method is commonly used in Chesapeake Bay and the coastal bays of Virginia (Orth et al. 2003) and is time-efficient if facilities are available for storing the flowering shoots in running estuarine water until dehiscence or storing seeds until spreading. The second is a recently developed technique using harvested flowering shoots suspended in mesh bags buoyed above the sediment of a targeted restoration area (Pickerell et al. 2005; Fig. 21). This buoy deployed seeding (also referred to as "BuDS" or "seed buoys") simulates long distance dispersal of detached reproductive shoots (Harwell and Orth 2002) and takes advantage of the natural slow release of seeds as they mature. This technique permits placing flowering shoots at the restoration site the same day as collection, and thus does not require facilities for seed collection and storage.

Buoy-deployed seeding in mesocosms. Seeds had not been used to restore *Z. marina* anywhere in California prior to 2005, but seemed a good choice for San Francisco Bay restoration considering the relatively high flowering rates and the knowledge of at least one annual bed, which relies on seeds to re-establish each year. The new buoy-deployed seeding technique seemed a good choice, as it did not require an investment in seed sorting facilities. It was first tested in a mesocosm experiment under semi-controlled conditions (Boyer et al. 2007). An array of 18 tanks was maintained with continuous-flow baywater and constant aeration with bubblers at San Francisco State's Romberg Tiburon Center (Fig. 10). This experiment tested seedling establishment with seed buoys using three donor sources of flowering shoots: two perennial beds, Point Molate and Bay Farm Island, and the annual bed, Crown Beach (Fig. 3). Three of the 6 tanks assigned to each donor site were inoculated with sediment collected from the donor site at the time of flower collection; these sediments presumably represented the microbial community found at the extant beds, which was hypothesized to be beneficial to plant establishment in the mesocosm tanks (Milbrandt et al. 2008). Spathes in late stage 4 or 5 (DeCock 1980) were removed from flowering shoots and thirty were placed in each of six seed bags (one per mesocosm assigned to that donor). These mesh bags were suspended just below the water surface in the tanks to simulate buoy-deployed seeding methodology in the field (Pickerell et al. 2005).

All three extant beds serving as donors produced seedlings (Fig. 10), suggesting that, at least under controlled conditions in mesocosms, the seed buoy technique was effective in achieving seedling establishment. Contrary to the hypothesis that the annual donor, Crown Beach, might have particularly high rates of seedling recruitment, mesocosms seeded with Crown Beach flowering shoots had lower seedling recruitment than those seeded from Point Molate and Bay Farm Island flowering shoots. Clonal growth proceeded through the growing season. Inoculation with donor bed sediments led to a significant increase in seedling recruitment and clonal spread for all donors; see Section II E, Sediment Microbes, for additional information on the effects of inoculation.

First buoy-deployed seeding field experiment. Buoy-deployed seeding has been tested through two experiments in the Bay to date. The first (detailed in Boyer et al. 2007) used two donors, the perennial Pt. Molate and the annual Crown Beach at two restoration sites, the Marin Rod and Gun Club (just north of the Richmond-San Rafael Bridge in the Central Bay) and China Camp State Park (southwestern San Pablo Bay). These sites were identified by the Merkel and Associates (2005) ELVS model as potentially suitable for *Z. marina*. Sixty seed buoys were deployed with flowering shoots from either Point Molate or Crown Beach at each restoration site. Thirty flowering shoots with several late stage 4 or 5 spathes (after DeCock 1980) were placed into each pearl net, and these remained at the restoration sites for six weeks to allow maturing seeds to drop from the mesh bags to the sediment surface (after Pickerell et al. 2005).

Recruitment at the two restoration sites was assessed at three time points in spring 2006. All sampling was by feel in extremely low visibility conditions using snorkeling gear. In late-February, 15+ seedlings were detected at the Marin Rod and Gun Club, and seedlings were found beneath both Point Molate and Crown Beach buoys. However, we did not detect seedlings on the later sampling dates. Winter 2006 produced very high rainfall and salinities were consistently low throughout the late winter and early spring, dropping to 6-7 ppt in March and April and 13 ppt in May (R. Obernolte, pers. comm.). No seedlings were detected at the China Camp restoration site on any date. Observations suggest that strong currents at this site could lead to transport of seeds or burial of seeds and/or seedlings. While whole transplants might be possible at this site, it may not be a viable location for seeding. However, it is possible that seeding would be successful in a year with lower rainfall and turbidity during the period of seedling establishment.

Second buoy-deployed seeding field experiment. A second experiment was conducted to test buoy-deployed seeding relative to hand-broadcasting and modified TERFS (see above) in 2006 (Boyer et al. 2008). While the first field trials of buoy-deployed seeding had not been successful, the anomalously high rainfall in that year, and success in the mesocosm experiment suggested another try was in order. The experiment was designed to compare techniques as well as donors at each of three restoration sites. Two perennial donors, Point San Pablo (PSP, also known as North Richmond) and Keller Beach (KB), were used, as well as the annual donor Crown Beach (CB) (Fig. 3). Restoration sites were chosen based on the ELVS model (Merkel and Associates 2005) and the ability to achieve landowner support and permission. Restoration sites included Marin Rod and Gun Club (MRGC) again because it had recruited seedlings in the first attempt and because a microcosm test of different sediment sources suggested it could provide favorable conditions for eelgrass establishment (See Section II E, Sediment Physical Characteristics). Additional sites were chosen in Corte Madera Bay (Marin Country Day School, MCDS) and Richardson Bay (RB) within the Audubon sanctuary.

Buoy-deployed seeding was by far the most successful restoration method (Boyer et al. 2008). Modified TERFS were not effective, probably because they failed to hold individual vegetative shoots in place long enough for them to root (see above). Broadcast seeding had very poor success, even though methodologies outlined in Granger et al. (2002) were closely followed; this method produced only nine seedlings as of March (at

Richardson Bay, from the Keller Beach donor population) and these did not persist beyond April 2007. Seedlings from buoy-deployed seeding were observed from March to May 2007, after which clonal growth increased dramatically (Fig. 22). Success of recruitment and persistence was quite variable by site. In March, initial recruitment was highest at RB but by July 2007, RB had lost nearly all its recruits. MRGC and MCDS seedling recruits spread clonally into sizable patches over time (Fig. 22). By July 2007, MRGC had substantially greater total shoot densities resulting from the seed buoys compared to the other sites (over 1300 shoots), followed by MCDS with 327 total shoots. Donor-specific recruitment in buoy-deployed seeding plots differed, with the annual (CB) donor population producing the fewest seedling recruits and total shoots (Fig. 22). The two perennial donors, KB and PSP, produced many more seedlings and, by the end of the 2007 growing season, the number of individual shoots at both MRGC and MCDS was nearly four times the number recruited from the annual donor, and this difference has continued over time. Average densities per plot were comparable for the two perennial donors at MRGC. However, at MCDS, average densities were significantly higher for PSP compared to KB (Fig. 22). Clonal growth was also strong for the CB donor at both sites, but total numbers were limited due to poor initial establishment in the previous year. Notably, the plants established from the annual donor in 2007 persisted through the winter and continued to spread clonally in 2008; i.e., the plants did not exhibit an annual life history at the restoration sites, suggesting that annularity at CB is largely environmentally-driven. Canopy heights of vegetative and flowering shoots were comparable among donors at the MRGC restoration site when measured in July 2008 (Fig. 23). Canopies were on average shorter at the MCDS site. At MCDS, both vegetative and flowering shoot heights were shorter in the CB plots than in those of the two perennial donors (Fig. 23).

Buoy-deployed seeding performed well at conserving genetic diversity in the restored populations relative to the donor populations. Observed heterozygosity and the average number of alleles per locus tended to decline slightly but nonsignificantly between donor and restored populations (B. Ort and S. Cohen data reported in Boyer et al. 2008).

Overall, the relative ease of use as well as the successes with buoy-deployed seeding lead us to recommend further use of this method for eelgrass restoration in the Bay.

Substrate enhancement

Submarine light availability is generally greater when sediments are coarse (sand) rather than fine (clay and silt) due to greater resuspension of fine particles. Many of the extant beds in San Francisco Bay have sediments with relatively high sand content, and thus low concentrations of finer particles, clay and silt. One possibility for increasing sand content at restoration sites is to bring dredged sand from high-energy areas to muddier restoration sites. In addition, some areas may be more suitable for restoration of eelgrass if their elevations are raised through the use of dredged material fill.

At the Emeryville Flats, adjacent to the San Francisco-Oakland Bay Bridge, six sand plateaus were created with mined sand either from Presidio Shoals or Angel Island, brought by barge, and discharged into a silt curtain rings (Merkel and Associates 2004b). Highest elevations were approximately -0.3 m MLLW. Sediment plateaus lost approximately 0.5 m of elevation over 60 weeks, and transplanted shoot bundles and sediment plugs had <10% survivorship after only 4 weeks (lower in coarser sediments from Angel Island than somewhat finer sediments from Presidio Shoals), perhaps due to shifting sediments. The authors cautioned against future sediment additions of this type in high energy locations without structural enhancements to keep sediment in place (Merkel and Associates 2004b).

Another experiment to assess the use of sandy dredged material to raise restoration site elevation on a plateau was conducted beginning in 2005 at Berkeley North Basin, near the Berkeley Marina (Merkel and Associates 2009b). Sandy dredged material from Presidio Shoal was deposited in a 110 x 50 m area (trucked to site and spread with bulldozers and hydraulic pumping). Initial depth ranged from -3.0 to 0.4 m MLLW. Sediments were contained with plastic mesh fences, and planted with whole shoots (in 5-shoot bundles) collected from extant beds along the East Bay (Berkeley Shoals and Pt. San Pablo). Transplant survival dropped precipitously within the first month and no plants were present by the following January, perhaps related to high sedimentation rates during winter storms. The authors recommended that given the uncertainty of eelgrass restoration on plateaus and the associated high costs of sand delivery and placement ($375,000 in this project), San Francisco Bay eelgrass restoration efforts should focus on sites with unmanipulated sediments.

Living shorelines

In other regions, such as the Chesapeake Bay and the coastal bays of North Carolina, "living shorelines" are used as an alternative to riprap, concrete, and other hard structures to accomplish sediment stabilization and reduce erosion while also increasing habitat values. Typically, hard, man-made structures are removed and sediments added or graded, followed by shoreline plantings of an emergent marsh grass such as cordgrass (*Spartina alterniflora*) or natural recruitment of plants. Natural structures such as oyster shell may be added to replace the erosion control function of man-made materials or to serve as wave break further offshore in the subtidal zone. Submerged grasses, sometimes in combination with oyster reefs or other natural structural features, are included in these projects to enhance shoreline buffering against waves and increase habitat value. There are a number of locations within San Francisco Bay that might be suitable for the application of this concept. These may be existing marshes for which there might be benefit in adding subtidal features, such as enhanced connectivity to resources for organisms that move among habitats, or reductions in flow that might otherwise erode marsh edges.

V. Recommended restoration approach

Strategy for restoration

We recommend an overall strategy for eelgrass restoration in San Francisco Bay that is measured and step-wise, and that takes into account knowledge of eelgrass ecology, physiology, and genetics from this and other systems. In re-creating communities and ecosystem functions at restoration sites we test our understanding of the natural systems we hope to simulate (Jordan et al. 1987). We recommend conducting restoration projects as well-designed, replicated experiments that afford the opportunity to learn what is effective as well as what is not, as opposed to the trial and error approach often employed in restoration efforts (Palmer 2009). Whenever feasible, we recommend restoration activities include:

1) A phased approach, with field surveys, mesocosm experiments, small scale "test plots" (Fig. 35), and pilot-scale field experiments used to make decisions about next steps at progressively larger scales,
2) Experimental treatments, replicated within and among sites, to permit statistical evaluation and support an adaptive management approach to decision-making,
3) Sufficient monitoring (using standard methods, e.g., Phillips and McRoy 1990; Short and Coles 2001) and length of evaluation period (Evans and Short 2005) to assess success.

Eelgrass restoration goals

1. Protect existing eelgrass beds.

Protecting existing beds must be the first goal of any seagrass restoration program. Existing beds provide the propagules for natural recruitment of eelgrass and its associated community of organisms, thus promoting long-term persistence of eelgrass habitats in the Bay. Further, unless large-scale nurseries are established in the future, all eelgrass restoration in the Bay must rely on collection of vegetative or flowering shoots from extant beds. We recommend that a buffer be placed around existing beds to avoid direct impacts of boating, boat anchoring and mooring, construction of new or expanded docks or other structures, and dredging. A buffer of 150 feet (45 m) may be sufficient in many instances to allow for interannual variation in the extent of bed margins but should be determined on a case-by-case basis considering bed configuration relative to slope; in the case of broad flats, the expanded or contracted bed variance between surveys can be many hundreds of meters (K. Merkel, pers. comm.; K. Boyer, pers. obs.). Further, it is imperative that restoration activities are conducted in a manner that minimizes disturbance to extant populations serving as donors. Restoration practitioners should assess the size and density of beds before deciding the number of shoots to collect. In some other regions, a rule of thumb or specific mitigation policy (e.g., in southern California) dictates that less than 10% of the vegetative shoots within an extant bed be removed for transplantation to another location, and this seems a reasonable guideline to

follow in San Francisco Bay as well. In addition, we recommend that collections be spread over a large region of the bed, both to minimize impacts in any one area and to maximize the genetic variation included in the transplants.

2. Create additional acreage of eelgrass in the Bay

While historic information on eelgrass distribution and abundance is scant, biophysical modeling and improvements in water clarity suggest there are unvegetated areas that may be suitable to support eelgrass, and might do so through restoration activities. We suggest that restoration should move forward (albeit cautiously) both to increase the acreage that may support valued functions (e.g., Kenworthy et al. 2006), and to off-set potential threats to extant eelgrass beds.

3. Develop self-sustaining eelgrass beds.

To achieve the most effective and efficient restoration program for the Bay, each individual restoration project should be able to maintain itself with little additional human intervention over time. This means that restored plants will persist over time, spread clonally and establish extensive rhizome systems, flower and contribute seed, and be resilient to storms, herbivory, and other damages each year as well as resilient to changes that occur in climate and human uses of the bay over time.

4. Provide habitat.

Seagrasses are valued as foundation species that provide habitat for numerous fish, invertebrates, and birds. Greater acreage of eelgrass should increase biodiversity and productivity in subtidal habitats of the Bay.

5. Help to stabilize shorelines.

In other regions, seagrasses slow water flow and buffer wave and current energy, perhaps increasing shoreline stability. Such benefits may accrue from eelgrass restoration in San Francisco Bay.

6. Integrate with other habitat types.

As there may be synergy among different types of restoration such as eelgrass and oyster reefs within the subtidal or among these subtidal habitats and intertidal habitats such as salt marshes, we recommend considering the potential benefits of including multiple habitat types in restoration projects where feasible in the Bay.

7. Improve understanding of existing eelgrass beds and functions.

Restoration projects provide an opportunity to increase knowledge related to eelgrass ecology and ecosystem function in San Francisco Bay through an analysis of the extant beds being evaluated as potential donor sites or serving as reference sites. In the evaluation process, we will gain better understanding of the population and community dynamics of extant beds, seasonal and interannual patterns in distribution and abundance, and functions, including physical processes and habitat provision for other species.

8. Use restoration as an opportunity for education.

While subtidal restoration and ecological study are difficult logistically, there are opportunities to involve the community. Citizen volunteers have been involved in many of the local restoration projects to date with coordination through organizations such as Save the Bay and the Tiburon Sunset Rotary Club. Land-based mesocosm and nursery work permits volunteers of all physical abilities to participate and learn about subtidal habitats. Connections with universities permits hands-on training of undergraduate students in classes and graduate students on related research projects.

Phased approach to restoration

We recommend a phased, adaptive management approach to restoration for eelgrass, such that prior knowledge or experience at a site guides the next steps, as follows:

Phase I. Experimental Restoration

Phase I-1: No prior knowledge; conduct basic site survey

Definition: No prior surveys of site

Recommended action: Basic site survey

   a. Detect eelgrass presence through visual surveys, aerial photos, acoustic monitoring or other, and if present, map areal extent using remote sensing survey techniques (e.g., sidescan sonar).
   b. Establish transects either by wading at extreme low tides or SCUBA to provide more detailed investigation (e.g., stem density, flowering frequency, etc.).

Phase I-2: Limited site knowledge; conduct preliminary site evaluation

Definition: Previous mapping or surveys have been conducted

Recommended Action: Preliminary assessment of suitability of the site for restoration.

    a. Determine if there are nearby eelgrass beds and potential for recruitment into site.
    b. Determine if ELVS model (Merkel and Associates 2005) suggests suitability.
    c. Determine land ownership and potential for cooperation.
    d. Assess additional constraints such as conflicting human uses.
    e. Evaluate seasonal and spatial patterns in abiotic factors such as light attenuation, sediment grain size and organic content, wave and current energy, and water temperature and salinity.

Phase I-3: Experimental restoration (small-scale test plots)

Definition: Phase I-2 actions completed; area is unlikely to recruit naturally, area is suitable according to ELVS model, willing landowners, no major conflicting uses of area, no indication of limitations based on abiotic conditions—best judgment suggests further action

Recommended Action: small-scale test plots

    a. Install experimental whole shoot transplants (number to be determined by site size and heterogeneity) using bamboo stake method (Fig. 18) or other in small plots (Fig. 24) and an array of seed buoys (see Recommended Methodologies, below).
    b. Replicate plots in at least three areas of the site at the same elevation.
    c. Repeat with multiple donors if feasible.
    d. Repeat at multiple elevations available at site within range of eelgrass growth.
    e. Monitor plant persistence and spread in late spring or summer 1/yr for at least 2 years (more if mild winters).
    f. Monitor abiotic conditions such as light and temperature that could influence the success of transplants, and monitor use by other organisms.
    g. Report findings of restoration potential and lessons learned.

Phase II: Pilot restoration (0.5 acre or less)

Definition: Phase I-3 completed with positive outcome, site appears suitable for eelgrass

Recommended Action: small restoration project (0.5 acre or less), to further assess site suitability, evaluate methodological questions, and the values of ecosystem functions and services at a larger scale than possible in test plots

    a. Use knowledge gained in Phase I-3 to guide restoration design.

b. Include experimental elements as feasible to test restoration techniques, donor choice, depth, etc. to increase knowledge base for future restoration work.
c. Monitor plant characteristics; e.g., shoot density, canopy height, flowering frequency, growth rates, and areal coverage each year (in summer if only once) for a minimum of 5 years.
d. Evaluate success by comparing monitoring results (including genetic information) from reference sites and time-series analysis of environmental parameters, particularly temperature and submarine light.
e. In the second year of the monitoring program, initiate an assessment of ecosystem function and services (e.g., spawning substrate and nursery and foraging habitat, sediment stabilization, etc.).
f. Report findings, including evaluation of restoration potential, habitat value, and lessons learned.

Phase III: Larger-scale restoration project (1 acre or greater)

Definition: Phase II completed with positive outcome, pilot scale restoration at multiple sites deemed to provide ecosystem services commensurate with the cost and effort involved in the restoration

Recommended Action: larger-scale restoration project (one acre or greater), to begin to increase acreage of eelgrass habitat, to assess how functions and services scale up at larger acreages, and to further develop the science and practice of eelgrass restoration

a. Use knowledge gained in Phase II to guide scaling up.
b. Continue to include replicate experimental treatments that will aid in future restoration designs; e.g., range of techniques, donors, depths, combination of eelgrass with other subtidal or intertidal habitat restoration elements.
c. Monitor plant characteristics; e.g., shoot density, canopy height, flowering frequency, growth rates, and areal coverage for a minimum of 5 years.
d. Evaluate success by comparing monitoring results from reference sites (including genetic information) and time-series analysis of environmental parameters, particularly temperature and submarine light.
e. In the second year of the monitoring program, initiate an assessment of ecosystem function (e.g., spawning substrate and nursery and foraging habitat, sediment stabilization, etc.) using standard methods.
f. Report findings, including evaluation of restoration potential, functions and services, and lessons learned.

Recommended methodologies

Site selection. Site selection is probably the single-most important factor in restoration of seagrasses worldwide (Fonseca et al. 1998). Site selection methodologies are actively used in some regions and help to increase the overall success of restoration programs (Short et al. 2002a). In San Francisco Bay, the ELVS model (Merkel and Associates 2005) can be used to help identify locations suitable for restoration. However, the evaluation of individual sites is needed to address potential site-specific limitations. We recommend the following questions be asked when evaluating a particular site:

1) Is there eelgrass nearby? If so, then conditions overall may be favorable in the region, and an assessment of why eelgrass is not growing at the proposed site should be undertaken. Eelgrass within the same sub-embayment does not necessarily mean that seeds are being delivered to the proposed restoration area. Further, there may be site-specific constraints to recruitment or persistence acting on fine scales (e.g., with depth, proximity to outfalls, etc.) that currently limit eelgrass at the restoration site.
2) If there is not eelgrass nearby, do conditions appear suitable for restoration? In this case, propagule supply is likely to be limiting, and intervention necessary for eelgrass establishment if conditions are favorable. Sites can be evaluated in part through ELVS model predictions (Merkel and Associates 2005), and further through assessment of time series measurements of existing conditions such as light and sediment characteristics.
3) If conditions appear suitable, do small-scale test plots (See Phase 3 above; Fig. 24) suggest that eelgrass can be established at the site on larger scales?

Attention to site selection should help to increase the overall success of the restoration program in the Bay by helping to direct time and funds toward sites that have the greatest potential for success.

Techniques: Seeding. We recommend buoy-deployed seeding for future projects (alone or in combination with whole shoot transplants; see below), as it has been used to establish plants effectively at two restoration sites to date while maintaining donor levels of genetic diversity (Boyer et al. 2008). This technique is relatively easy to conduct and does not require specialized facilities. However, it was not effective in one year (perhaps due to extreme rainfall and associated turbidity) and at two restoration sites (Richardson Bay, when other sites were successful in the same experiment, and China Camp, where poor recruitment may have been related to the site or the high rainfall year). Greater attention to site selection through site evaluation and test plots could help to increase the success of this and other restoration techniques. At this point in time, we cannot recommend broadcast seeding, as it does require special facilities, is more time intensive, and was not found to be effective at sites where buoy-deployed seeding was successful in the same year and with the same donors (Boyer et al. 2008).

Techniques: Whole shoot transplants. Whole shoot transplants have had mixed results locally, but recent tests with a new bamboo-stake technique (Fig. 18) have been

promising. This technique offers a biodegradable anchoring system and a long anchor to hold shoots in place effectively in many sediment types until rooting occurs. Other techniques are likely also suitable, and the vagaries of site selection may have led to the conclusion that the technique was at fault rather than a poor understanding of a site's ability to support eelgrass. Modifed TERFS that include a biodegradable frame are a step in the right direction (WRA work in Sausalito) and this design should be refined for more rapid construction to permit scaling up to larger sites. Further, mesocosm tests of plugs with intact sediments suggest that this technique should be evaluated further (Josselyn and Alberte 1990; Merkel and Associates 2004b).

Techniques: Combined seeding and transplant approach. Where possible we recommend an experimental comparison of techniques on a small scale to aid in selection of a technique that may be best suited to a particular site. Seeding can increase genetic diversity and perhaps resilience of restored beds, but well-anchored transplants may be more suitable for sites with high sedimentation rates or currents. If experimental evaluation is not possible due to time or funding constraints, then a combination of both seeding and transplants may help to "hedge bets" that one or the other will be effective in establishing plants at a particular site and that resiliency expected to be conferred by genetic diversity of new plantings will be maximized as much as possible. We recommend that seeded and transplanted plots be kept separate to help evaluate the success of each at a particular location.

Donor selection. We have learned through experimental comparisons of donors that not all donors are as effective at establishing plants at restoration sites. While genetic diversity among donors was not substantially different, genetic structure is different among all populations tested in the Bay (Ort et al. in review), suggesting that donor choice could matter in terms of matching donors to the conditions present at a particular site. Perennial donors Point San Pablo and Keller Beach resulted in similar recruitment of shoots to the Marin Rod and Gun Club site but Point San Pablo produced significantly more shoots at the Marin Country Day School site. The annual donor Crown Beach was expected to produce greater numbers of seedlings at the restoration sites but instead recruited poorly and also led to shorter vegetative and flowering shoots at one of the restoration sites (Boyer et al. 2008; Fig. 21, 22). In addition, contrary to expectations it does not have greater genetic diversity (as observed heterozygosity and allelic richness) than perennial donors tested (Ort et al. in review). When possible, continued experimental tests of donor effectiveness should be made at restoration sites before scaling up. Use of multiple donor sources of restoration material should be considered on a case-by-case basis; if deemed suitable at a particular site, mixing donors could increase the probability that genotypes best suited to a site will be included, which may increase long-term resiliency of restored populations (e.g., Williams and Orth 1998).

Substrate addition. It is not clear at this point in time whether coarse substrate (sand) addition is needed or desirable at restoration sites. While it is true that most extant beds occur in sandier conditions than most available restoration sites, it is expensive to move coarse sediments, requires lengthy permitting, and in the few cases where it has been attempted in San Francisco Bay (Emeryville Flats and Berkeley North Basin), did not

result in successful transplants. Raising existing elevations appears to require structural containment, and even then sediment movement in the placed material seems to preclude establishment of plants. Sediments with higher clay and organic content at restoration sites appear so far to be capable of supporting restored plants (three growing seasons to date); however, it is possible that plants in fine sediments will have difficulty persisting through heavy winter storms or that resulting turbidity over fine sediments will negatively impact persistence at these sites. We have not had a harsh winter season since these sites were restored, so this possibility remains to be tested.

Monitoring: Metrics for measuring success

It is essential that restoration sites be monitored for extended periods that encompass a range of climate conditions and resulting differences in storm-generated waves, turbidity, etc. We suggest that monitoring should be conducted for at least five years. Otherwise, it will be very difficult to assess the success of various restoration treatments (donors, techniques, sites, depths, etc.) and to apply this learning to new restoration efforts. We find that funding is often available to conduct restoration but not to monitor; this trend has the potential to greatly hamper overall restoration progress for eelgrass in the Bay.

Particular elements of a monitoring program will depend on the goals of the restoration or mitigation project. Monitoring should at a minimum include a measurement of density and acreage estimates to assess structural characteristics developing at the restoration site. In addition, a number of measures that help to assess the provision of functional features of a restored eelgrass bed are highly recommended. These include growth rates, flowering frequency, seed viability, and genetic diversity measures. Epifaunal and infauna surveys are essential to determining if the restored beds are developing the food web support functions of natural beds and an evaluation of fish and bird use is needed to determine if higher trophic levels are being supported by the restored eelgrass bed.

These and other measures should be undertaken in comparison to reference sites, which could be the donor beds used for the project or other extant beds in the region. We recommend that multiple reference sites be chosen for comparison, as the large range in variation in natural beds should be considered when assessing the success of restored sites. In addition, we recommend that reference sites be monitored for these metrics over the same period of time, as interannual differences in a restored bed can best be evaluated within the context of regional variation in eelgrass autecology.

Restoration Costs

To date, eelgrass restoration in San Francisco Bay has been conducted at a small scale (0.5 acre or smaller). Estimates of cost have not been scaled up to larger restoration projects. Short et al. (2002b) estimated that the horizontal rhizome method for bare root transplants costs approximately $100,000 per acre, while TERFS cost approximately $58,000 per acre. In contrast, Fonseca et al. (1998) summarized costs for bare-root

planting across numerous seagrass species and sites at about $10,000-20,000 per acre; however, it should be noted that these data are now more than ten years old. Busch and Golden (2009) estimate that it costs the Maryland Department of Natural Resources between $2,702 to $39,654 to disperse seeds using buoy-deployed seeding and between $17,009 and $67,085 per acre to conduct broadcast seeding. Costs of eelgrass bare root transplants at the Berkeley North Basin site were $34,000 for a 0.79 acre area, which scales to $43,000 per acre (Merkel and Associates 2009b).

Clearly, numbers range dramatically among published summaries of seagrass restoration costs. Notably, none of these costs include monitoring. It is probably safe to estimate that with the difficult logistical conditions found in much of San Francisco Bay, along with monitoring for the recommended five years, total costs per acre would approach or exceed the highest of the estimates above for seeding and bare root transplants.

VI. Acreage goals and site-specific recommendations for restoration

Overall acreage goal

Submerged Aquatic Vegetation Restoration Goal 1 of the Subtidal Habitat Goals Project is to "increase native eelgrass populations in San Francisco Bay within 8000 acres of suitable subtidal/intertidal area over a 50-year timeframe using a phased approach". We arrived at the 8000-acre goal as follows. The predictive model known as ELVS (Merkel and Associates 2005; see section II.C of this report) identified 23,440 acres as potentially suitable habitat for eelgrass. Within this acreage, potential habitat was divided into several classes: 0-33, 34-66, and 67-100% on the habitat suitability index (HSI). The higher two suitability classes (34% or greater on the HSI scale) compose approximately 50% of the total acreage identified by the model, or about 11,700 acres. Subtracting the existing acreage of eelgrass as of 2009, 3700 acres (Merkel and Associates 2009a), from this total leaves approximately 8000 additional acres possible within the areas identified in the moderate to high suitability range in the model. Achieving this long-term goal would lead to approximately three times the acreage of eelgrass present in the bay as of 2009. Considering that 1) we are including only approximately half of the area identified by the model, and 2) the much higher potential acreage that might be deemed suitable if we included all areas at appropriate depths for eelgrass (see potential acreage from shoreline to -1.5 or 2 m depth MLLW in Appendix 1), we find this 8000 acre long-term goal to be conservative. We expect that this goal will be modified over time to reflect natural expansion or contraction of beds, as well as successes or setbacks as restoration projects proceed in the step-wise fashion we outlined above (Section V, Phased approach) and in the next section.

Site-specific recommendations

In a workshop at the Romberg Tiburon Center in December 2008, local and regional experts reviewed maps and discussed segment-by-segment the opportunities and

constraints of eelgrass restoration in the bay. This discussion has led to site-specific recommendations for phasing of restoration activities and needs for protection of extant beds. These recommendations are presented in detail in Appendix 1 of this report and are the basis for Submerged Aquatic Vegetation Restoration Objective 1-1 in the Subtidal Habitat Goals, to "implement phased restoration at 35 key locations to increase native eelgrass habitat by 25 acres within 5 years, 100 acres within 10 years, and up to 8,000 acres within 50 years". Below we summarize sites recommended for various phases of restoration actions. Segments of the Bay denoted below and in Appendix 1 are defined in Fig. 25. Figure 8-7 in the main report (Chapter 8) shows the locations of the proposed restoration activities.

Priority sites for Phase I-1 action (basic site survey)

Several bay-wide surveys have been conducted in the Bay to date, and probably encompass a very high percentage of the total acreage of eelgrass. A survey was conducted in 2003 using sidescan sonar and diver ground truthing (Merkel and Associates 2004a), focusing on the regions where eelgrass was previously known to occur from a 1987 survey (Wyllie-Echeverria and Rutten 1989). An additional survey was conducted in 2009 (Merkel and Associates 2009a). Areas covered in less detail in these surveys are not predicted to support eelgrass (Merkel and Associates 2005), based on environmental conditions such as low salinity (e.g., east of Vallejo) or high residence time (e.g., the far South Bay). It is possible that changing climate conditions such as increased sea level, which could result in higher salinity up-estuary, will lead to revisiting expectations of suitable habitat. It is important to note that many beds and small patches detected in previous surveys have had no additional study, meaning the second recommended part of Phase I-1 action, density surveys through wading or SCUBA, have not been conducted. Such surveys are highly desirable to complement the sidescan sonar-based surveys.

There are three specific locations we currently recommend for this second part of Phase I-1 action. These sites are in the South Bay, at the most southerly extent of the known eelgrass distribution and in eastern San Pablo Bay and western Carquinez Strait at the most northeasterly extent of the known distribution. First is the region near the San Mateo Bridge on the east side (Segment S). High interest is associated with this area due to pending changes in sediment dynamics associated with South Bay Salt Ponds restoration in the Eden Landing region. In addition, this area has been identified as a possible Living Shorelines concept site, thus further information on extant eelgrass distribution is needed. The second region is on the opposite shore in the San Mateo area just north and south of Coyote Point (Segments J and M). This area has supported small patches of eelgrass in past surveys, but may be recruitment limited currently as prevailing winds in the bay would tend to carry detached flowering shoots toward the east side of the bay. More detailed assessment of the extent and density of eelgrass in this region is needed. The third region includes the area east of Point Pinole (in Segment H) and into Carquinez Strait (Segment BA) where eelgrass was found to be present in the summer of 2010 (K. Boyer, pers. obs.). The areas within these segments should be assessed for

eelgrass presence and extent, especially over the next few years to determine whether presence in these areas represents a new eastward expansion trend or if contraction will follow after temporarily favorable conditions. As funding becomes available, other areas that have not been surveyed beyond monitoring with sidescan sonar are likely to become candidates for density surveys that make up the second part of Phase I-1 action.

Priority sites for Phase I-2 action (preliminary assessment of suitability)

We have identified a number of sites for Phase I-2 action. Starting at the top of Appendix 1:

Horseshoe Cove has previously supported eelgrass but currently has very limited coverage (0.5 acres in 2003 survey). It is possible that with its current small distribution, there is limited seed available for bed maintenance. Investigation of constraints at this site is needed.

Richardson Bay has a large natural bed near Sausalito in deep waters, but the ELVS model suggests much additional acreage may be suitable for eelgrass (Merkel and Associates 2005), although this acreage appears on the lower end of the habitat suitability index (0-33%). The Audubon sanctuary in the northeast arm presents opportunity (control of boat access, engaged staff, source of volunteers, potential synergy with other types of restoration). Additional evaluation is needed in this region, as restoration using seed buoys was not successful (when it was in the same experiment at two other locations; Boyer et al. 2008), and poor experimental transplant success at shallower depths than current distribution (S. Kiriakopolos, SFSU thesis, in prep.) suggest limitations. Please see additional notes on this site in Appendix 1.

The ELVS model suggests suitability of habitat for eelgrass west of Point San Pedro in San Pablo Bay along the shoreline of China Camp State Park (Segment G). Buoy-deployed seeding was not successful along this shoreline in 2005-2006, perhaps related to high turbidity from extended winter storms (See section IV, Seeding). Further evaluation of restoration potential in this region is needed.

We recommend preliminary assessment of suitability along the North Richmond Shoreline (in Segment H), shoreward of the large offshore bed, as there may be areas with eelgrass restoration potential especially in conjunction with upcoming shoreline and tidal wetland restoration activities by East Bay Regional Parks District.

The area east of Point Pinole to just past the Carquinez Bridge (in Segment H and BA) is also recommended for Phase I-2 actions in conjunction with Phase I-1b density assessments. That eelgrass is now present (as of 2010) in the Vallejo area (at CSU Maritime) and east of the Carquinez Bridge at the Glen Cove Marina suggests that this region may be more suitable for eelgrass than previously supposed, and if higher salinity waters proceed northeast up the estuary with sea level rise, this region may become more

suitable over time. A more detailed investigation of opportunities and constraints is needed.

The shoreline along Albany and Berkeley may present opportunities, but additional evaluation is needed, particularly along Albany Beach, North Basin, and the Emeryville Crescent area.

The area at the tip of the former Alameda Naval Air Station (in Segment K) may be suitable for eelgrass, and preliminary evaluation is recommended.

Segment T, in the Hayward area, shows potential according to the ELVS model (Merkel and Associates 2005) and should be further evaluated for restoration potential.

Segment S, south of the San Mateo Bridge in the Eden Landing area, is recommended for Phase I-2 action in concert with recommended Phase I-1b density surveys (above) and Phase I-3 test plots (below).

Segment J, north of Coyote Point and Segment M, south of Coyote Point are recommended for Phase I-2 action in concert with recommended Phase I-1b density surveys.

Priority sites for Phase I-3 action (testing for restoration)

We recommend small-scale test plots in several locations.

Areas of Richardson Bay within the Audubon sanctuary should be targeted for test plots, particularly the area adjacent to the channel along the west side of this sub-embayment, which may bring cooler, clearer water into this area. Test plots should proceed only if preliminary assessment suggests this is prudent.

We recommend test plots in Corte Madera Bay near the Corte Madera and Muzzi Marshes. Successful seed buoy restoration in the central portion of this bay leads us to suggest that shallower regions nearer to shore should be assessed for transplant/seeding potential through test plots. This area has been identified as having potential for integration between wetland and subtidal habitat features perhaps through a Living Shorelines type project.

Test plots should also be used to further evaluate restoration potential along the San Rafael shoreline in Segment I. Test plots with bamboo-stake transplants performed very well over two years at one location (See Fig. 24), but other areas along this shoreline should also be tested; e.g., along the quarry near Pt. San Pedro, and the region between the existing test plots and southwest to the Marin Rod and Gun Club.

Along the shoreline of China Camp State Park (Segment G) test plots are recommended along with Phase I-2 site evaluations. Whole shoot transplants may succeed in this

region of swift currents and sediment movement even though seeding did not in the one attempt with buoy-deployed seeding.

Test plots are recommended in conjunction with Phase I-2 evaluations shoreward of the large existing bed along the North Richmond Shoreline (in Segment H).

We recommend test plots be used in locations deemed suitable through Phase I-1 and I-2 evaluations in southeast San Pablo Bay (east of Point Pinole in Segment H) and in the western portion of Carquinez Strait (Segment ED and BA) where recent expansion of eelgrass has been observed (Merkel and Associates 2009; K. Boyer, pers. obs.).

We recommend test plots in the Emeryville Crescent area. Previous test plots in this area established well after one year in 2008 but showed evidence of bird herbivory and did not persist through a second year. Somewhat deeper test plots should be established in this region, as greater depth may reduce access by avian herbivores.

Test plots are also recommended in Segment S (south of the San Mateo Bridge on the east side) near the Eden Landing portion of the South Bay Salt Ponds restoration project. This site is a possible Living Shorelines project candidate and Phase I-1b, I-2, and I-3 actions are all recommended to facilitate rapid preliminary evaluation of the site for further restoration activity.

Priority sites for Phase II and III action (pilot restoration and larger restoration project)

Previous site-specific investigations suggest readiness for scaling up to pilot-scale and larger-scale restoration projects in regions, San Rafael Bay and Corte Madera Bay. Test plots at one location along the San Rafael shoreline have been very successful over two years and we suggest a pilot-scale restoration project for this site. A successful pilot-scale seed-buoy restoration site at the Marin Rod and Gun Club should be expanded to a larger-scale project on adjacent lands. Similarly, in central Corte Madera Bay, successful seed-buoy restoration should be expanded to a larger-scale project. In both of these cases, preliminary data suggest high invertebrate densities and use by fish and birds as well as evidence of flow reduction, all characteristics that we expect to see in eelgrass beds. Additional quantification of these values is recommended, as well as continuing an experimental approach to test specific hypotheses and methodologies, to feedback to decisions about future restoration projects.

In addition, the Oakland Middle Harbor project is currently planned as a large-scale restoration site for eelgrass; sediment fill has been added and final grading and eelgrass planting are planned within the next several years. As at other sites, we recommend a phased approach, with large-scale restoration contingent on success at smaller scales.

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Fig. 1. Eelgrass, *Zostera marina*, is the only seagrass that grows in the soft sediments of the San Francisco Bay proper. Lower photo shows density counts in a 0.25m^2 quadrat along a transect line, part of the Seven Site Survey (see text). Inset shows a flowering shoot with ripening seeds. Photos: top, K. Boyer, inset, S. Kiriakopolos, bottom, J. Stalker.

Location	Abundance in 1987 (Acres)	Abundance In 2003 (Acres)
San Pablo Bay	124	1,504.5
Point Orient	3	2.3
Naval Supply Depot	12	77
Point Molate Beach	26	32
Toll Plaza West	0.5	0
Toll Plaza East	0.5	2.5
Point Richmond, North	7	24
Point Richmond, South	4	65.6
Richmond Breakwater, North	18	19
Richmond Breakwater, South	7	86.3
Brickyard Cove		17.7
Emeryville (breakwater)	13	28.7
Emeryville Flats		21.6
Yerba Buena Island		1.7
Treasure Island		5.1
Alameda	55	269.9
Bayfarm, North	2	4.4
Bayfarm, South	4	127.9
Coyote Point	1	0.6
Richardson Bay	13	436.7
Angel Island West	3	1.6
Angel Island South		0.7
Angel Island East		2.8
Belvedere Cove	5	21.8
Point Tiburon	1	0.2
Keil Cove	10	20.4
Paradise Cove, North	4	13
Paradise Cove, South	30	0.2
Pt. San Quentin		0.5
Pt. San Pedro		1.6
Minor Beds and Patches		90.2
TOTAL	**316**	**2,880.5**

Sources: (WyllieEcheverria et al. 1989); (Merkel 2003a, b)

2003 Eelgrass (*Zostera marina*) Density
(Percent Cover)

0-5%   20-40%
5-20%   >40%

May 2007

Fig. 2. Extent of eelgrass in San Francisco Bay in 2003, with acreage estimates from surveys in 1987 (Wyllie-Echeverria and Rutten 1989) and 2003 (Merkel and Associates 2005). See Figure 8-1 in main report for 2009 distribution (Merkel and Associates 2009).
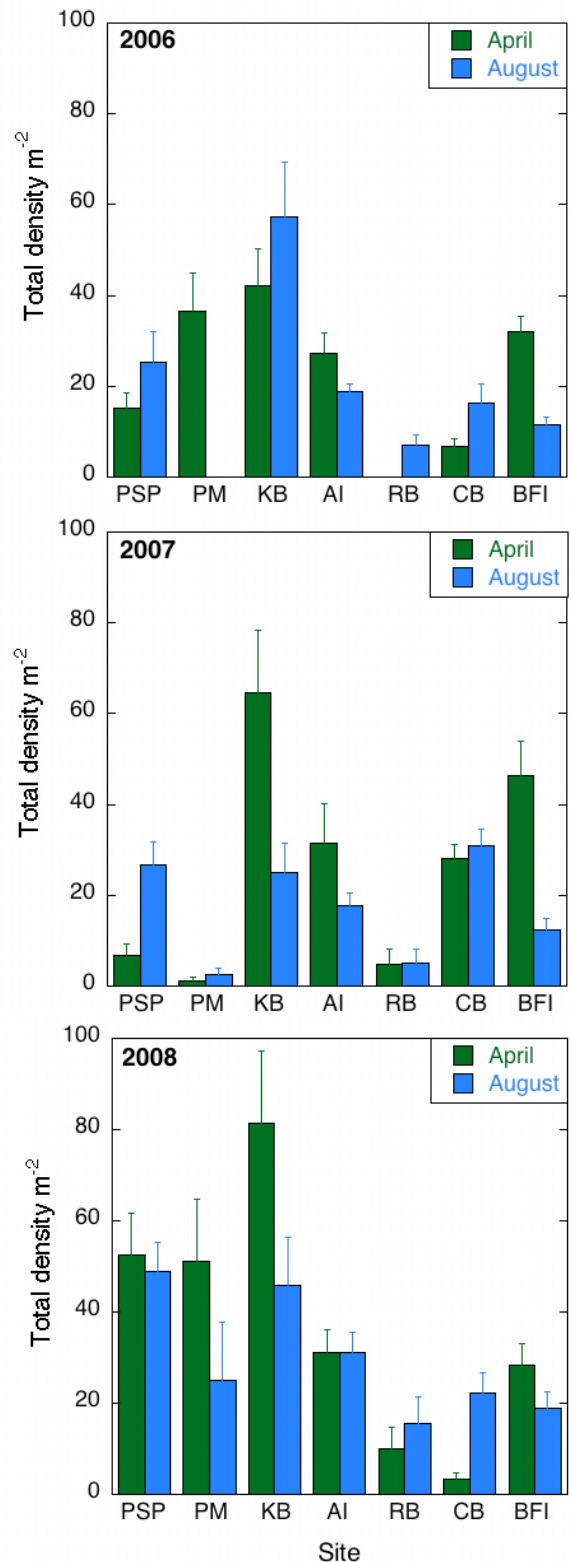
Fig. 3. Total shoot density (vegetative and flowering) measured in the Seven Site Survey from 2006 to 2008, Boyer et al., unpublished data. Bars are ± 1 SE. Eelgrass beds from left to right on graphs represent north to south distribution as shown on map. Note, Keller Beach is the same as Point Richmond in Fig. 2. The Point San Pablo bed is also known as the North Richmond Shoreline bed.
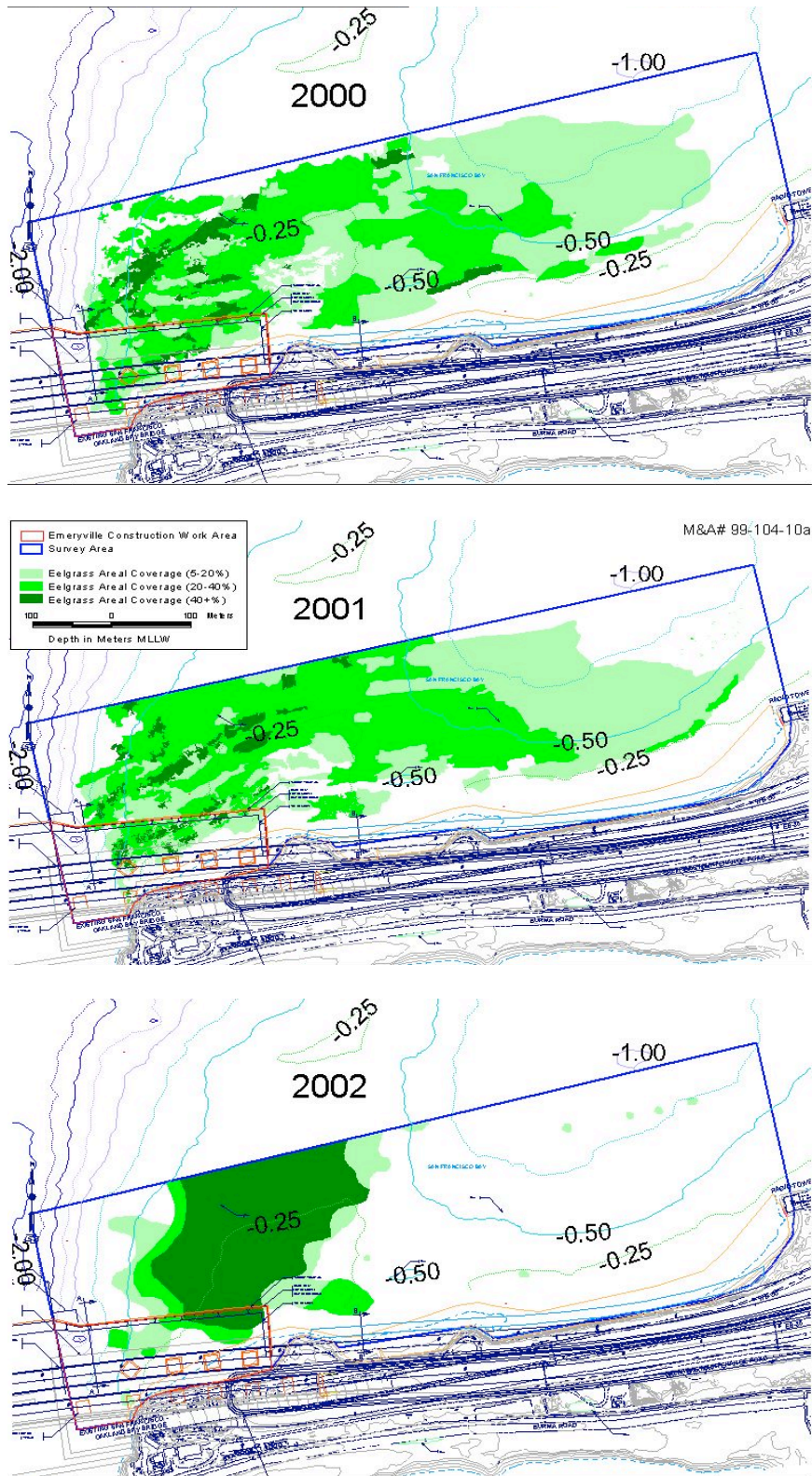
Fig. 4. Images of the Emeryville Flats eelgrass bed (adjacent to the San Francisco-Oakland Bay Bridge) from sidescan sonar, showing large interannual variability in areal extent and cover classes (dark color indicates greater cover). Images from Merkel and Associates (2006).
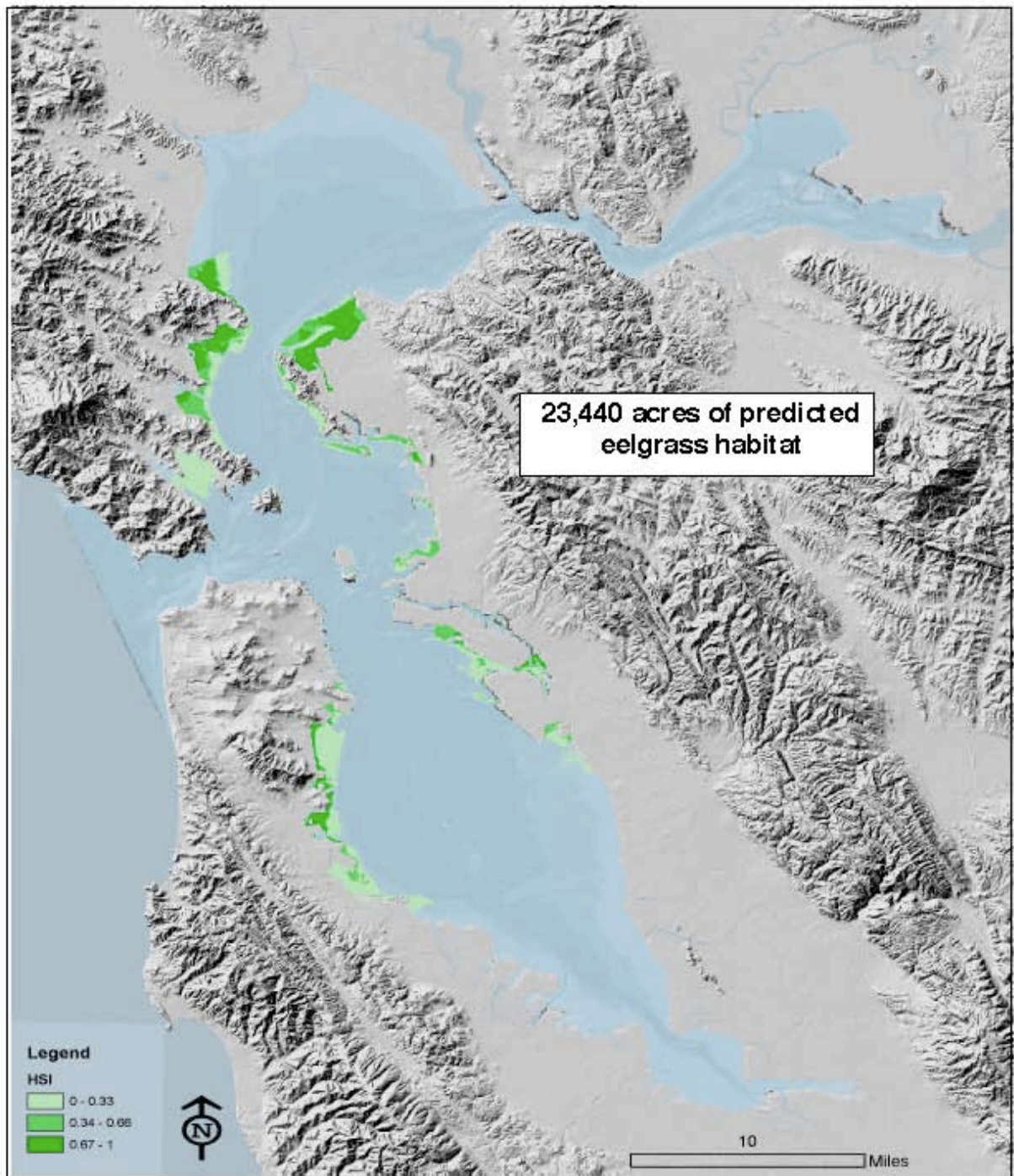
Fig. 5. Ecological Limits, Viability, and Sustainability (ELVS) model output, indicating areas potentially suitable for eelgrass, with darkest green predicted to be most suitable. Reprinted from Merkel and Associates (2005).

Fig. 6. Top: The majority of the eelgrass bed at Crown Beach (Alameda) is annual; i.e., it dies back during the late fall and recruits from seed the following late-winter. New seedlings become visible in mid-February, and by August, the bed appears similar in cover to beds that persist through the winter each year. Photos: K. Boyer.

Bottom: Tidal simulator experiment at the Romberg Tiburon Center by S. Kiriakoplos (SFSU thesis, in prep.), 2008. Top photo shows tank at top with fluctuating tidal exposure to simulate conditions in the shallow, annual portion of the Crown Beach bed, and tank at bottom with continuous immersion. Inset shows seedling in containers at the start of the experiment. Photos: S. Kiriakopolos.

Fig. 7. Canada Geese at Crown Beach frequently remove eelgrass shoots at a point below the growing region (meristem), thus killing the shoot. Middle photos show an uncaged plot with plants lost over a 10-day period of high geese herbivory. A caged plot (bottom) shows plants persisting during the same period when protected from herbivory. Photos: S. Kiriakopolos.
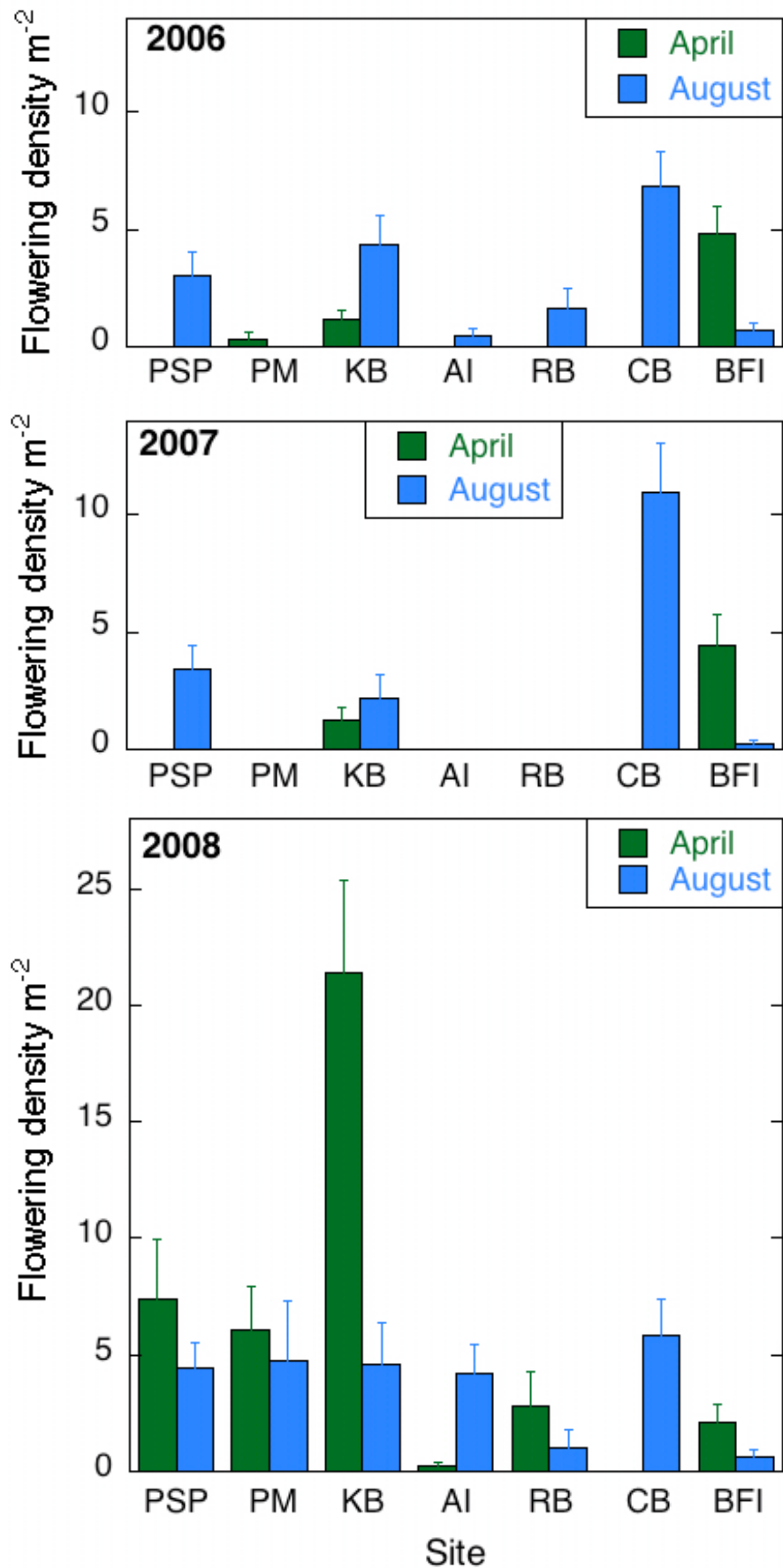
Fig. 8. Flowering shoot density measured in the Seven Site Survey from 2006 to 2008, Boyer et al., unpublished data. Eelgrass beds are as in Fig. 3. Bars are ± 1 SE.
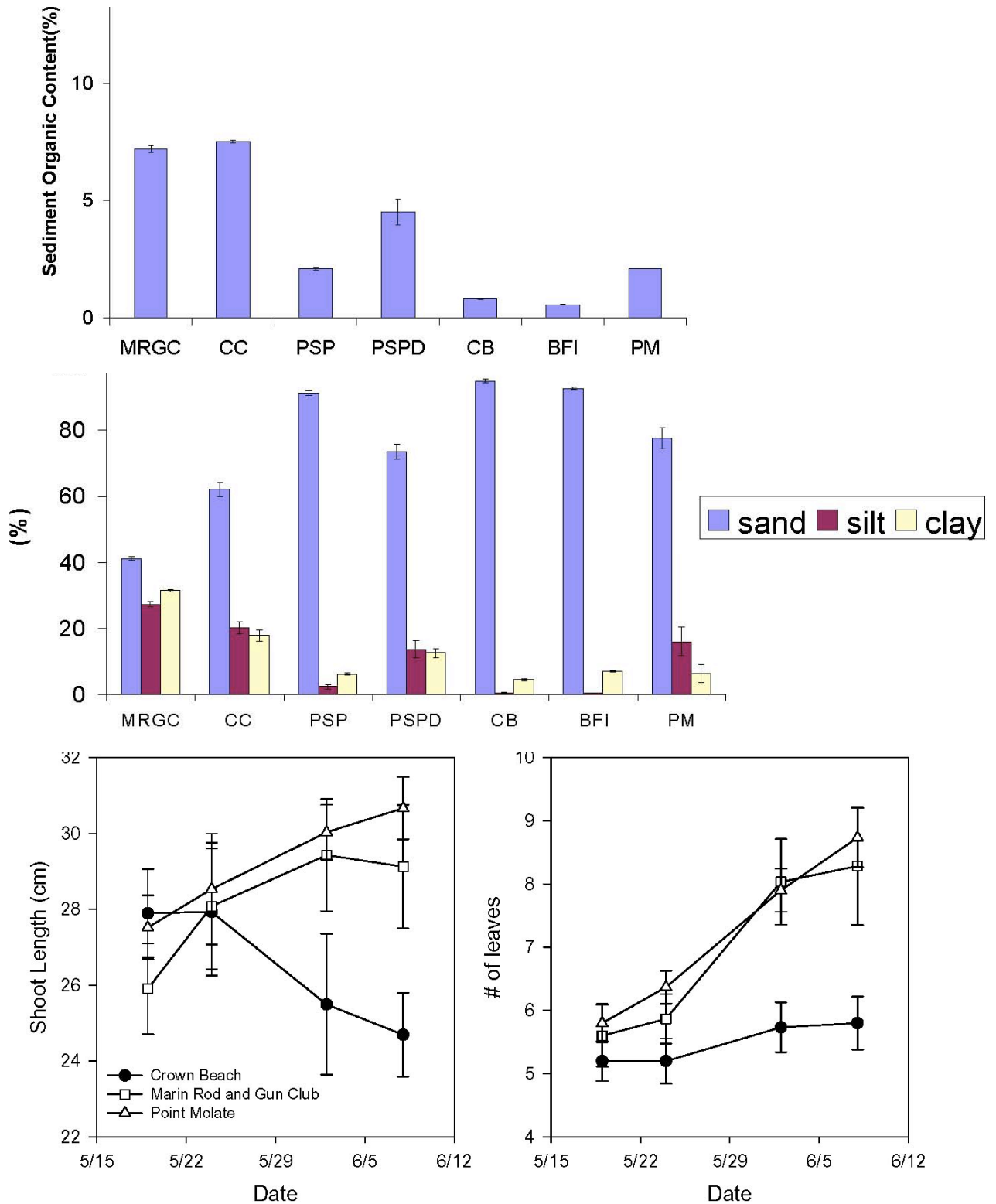
Figure 9. Organic matter (top) and grain size distribution (middle) in sediments collected from extant eelgrass beds (Point Molate = PM, Crown Beach = CB, Point San Pablo = PSP, the deep edge of the Point San Pablo bed = PSPD, and Bay Farm Island = BFI) and from the 2005 seed buoy restoration sites at Marin Rod and Gun Club (MRGC) and China Camp (CC). Bottom: mean shoot length and number of leaves on seedlings grown in sediments from CC, MRGC, and PM. Bars are ± 1 SE. From Boyer et al. (2007).
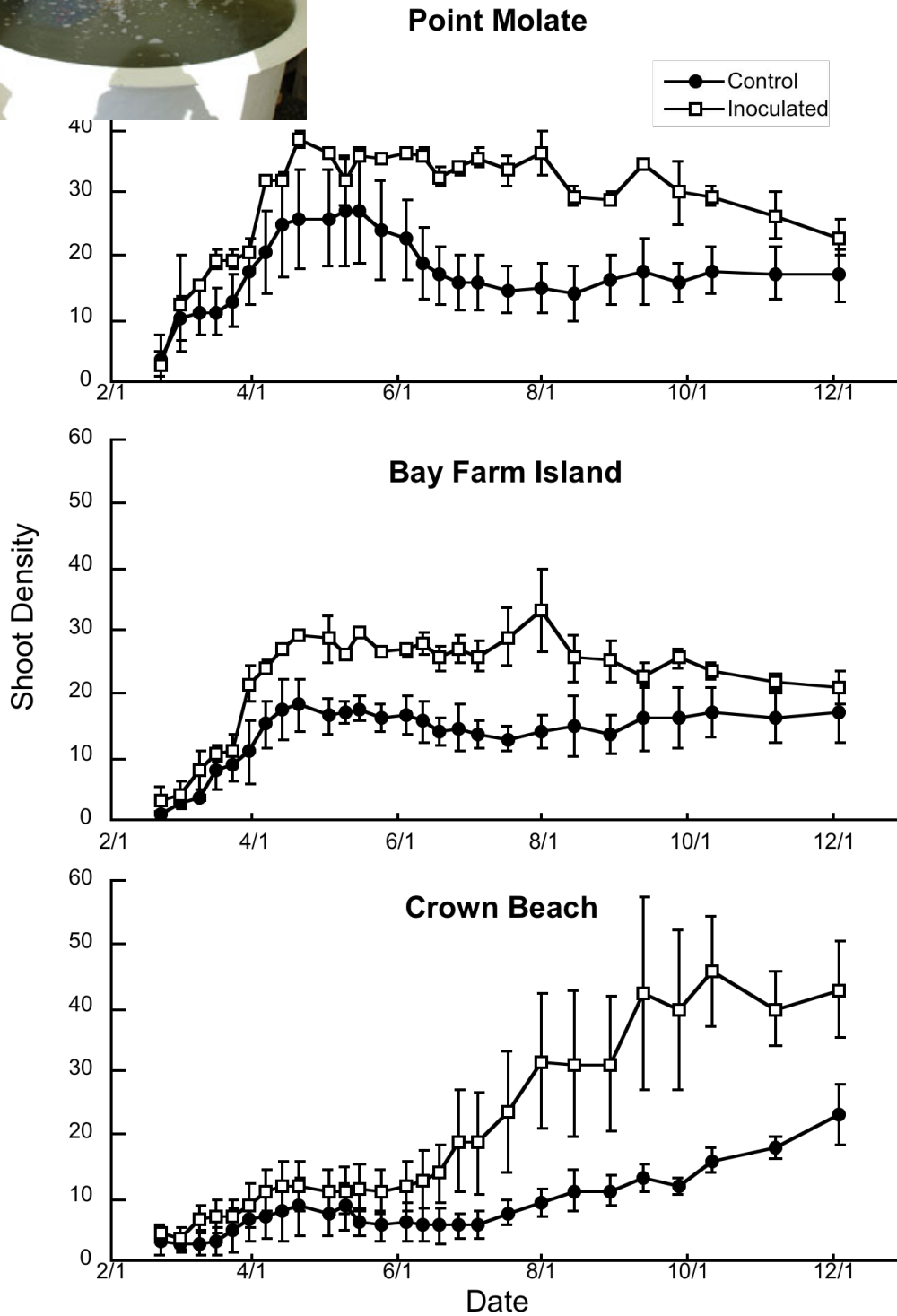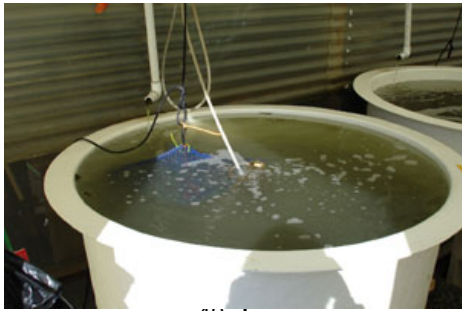
Figure 10. Shoot density of seedlings (through late April) and clonal recruits from the three donor populations used in a mesocosm test of the buoy-deployed seeding technique, with and without donor-site sediment inoculation. Bars are ± 1 SE. Inset shows one of the mesocosms with floating seed bag (photo: L. Reynolds). From Boyer et al. (2007).
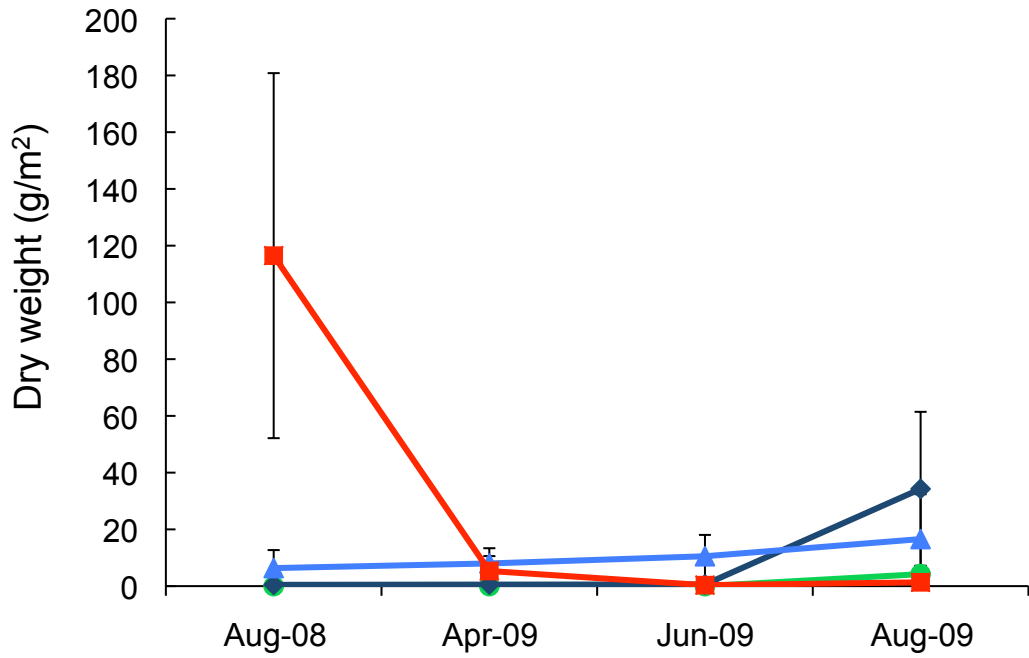
Figure 11. Biomass of all macroalgal species in eelgrass beds at Point San Pablo (red, also know as North Richmond), Keller Beach (blue), Crown Beach (black), and Bay Farm Island (green) over time, August 2008-August 2009. Shown are means of 10 one meter-square quadrats along a 100 m transect line; error bars represent ± 1 SE. Note difference in scale. Unpublished data from G. Santos thesis, SFSU, in progress.
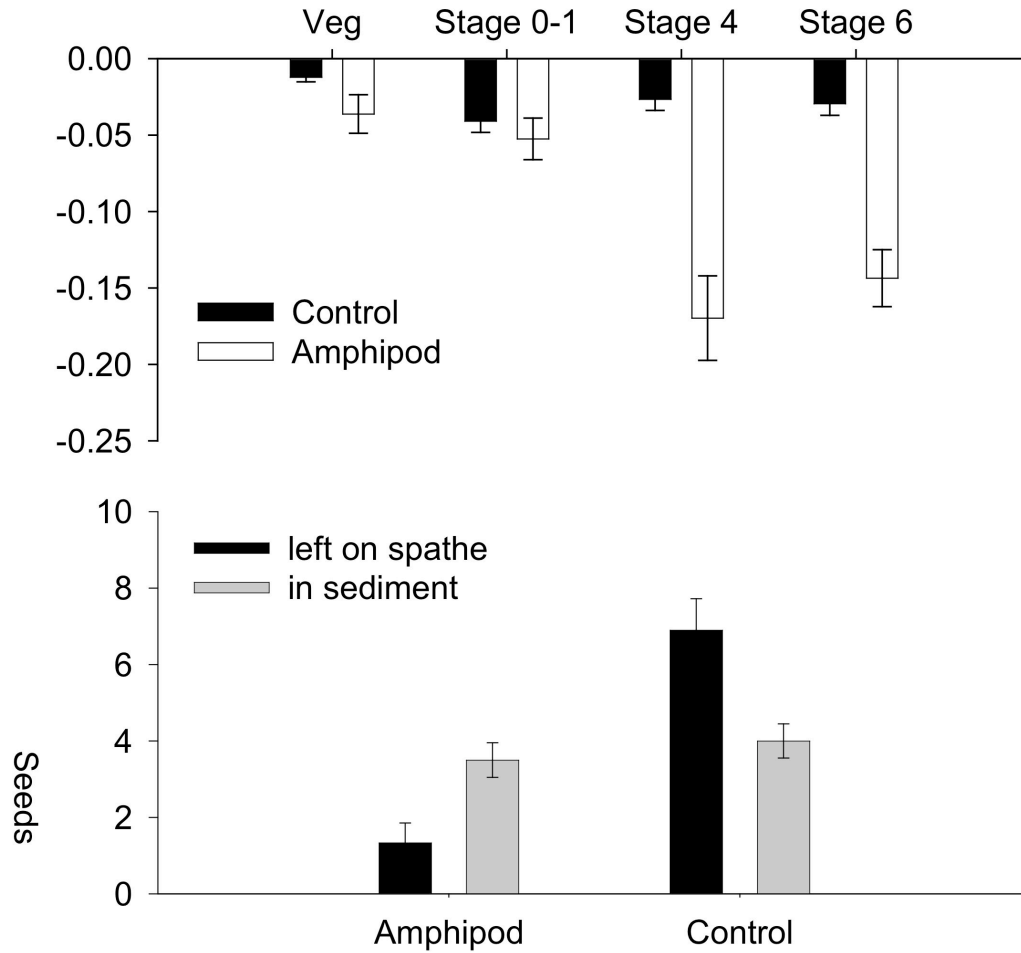
Figure 12. Top: Percent change from initial mass of vegetative blades, and spathes at stage 0-1, late stage 4, and stage 6 (DeCock 1980), in control or +amphipod treatments. Bottom: Number of seeds remaining on late stage 4 spathes (dark bar), and number of seeds on the sediment surface (light bar), with and without amphipods present. From Boyer et al. (2007). Photo of *Ampithoe valida*: K. Boyer.

Figure 13. Flow reduction is apparent even around small patches of eelgrass.
Marin Rod and Gun Club restoration site, 2008. Photo: K. Boyer.
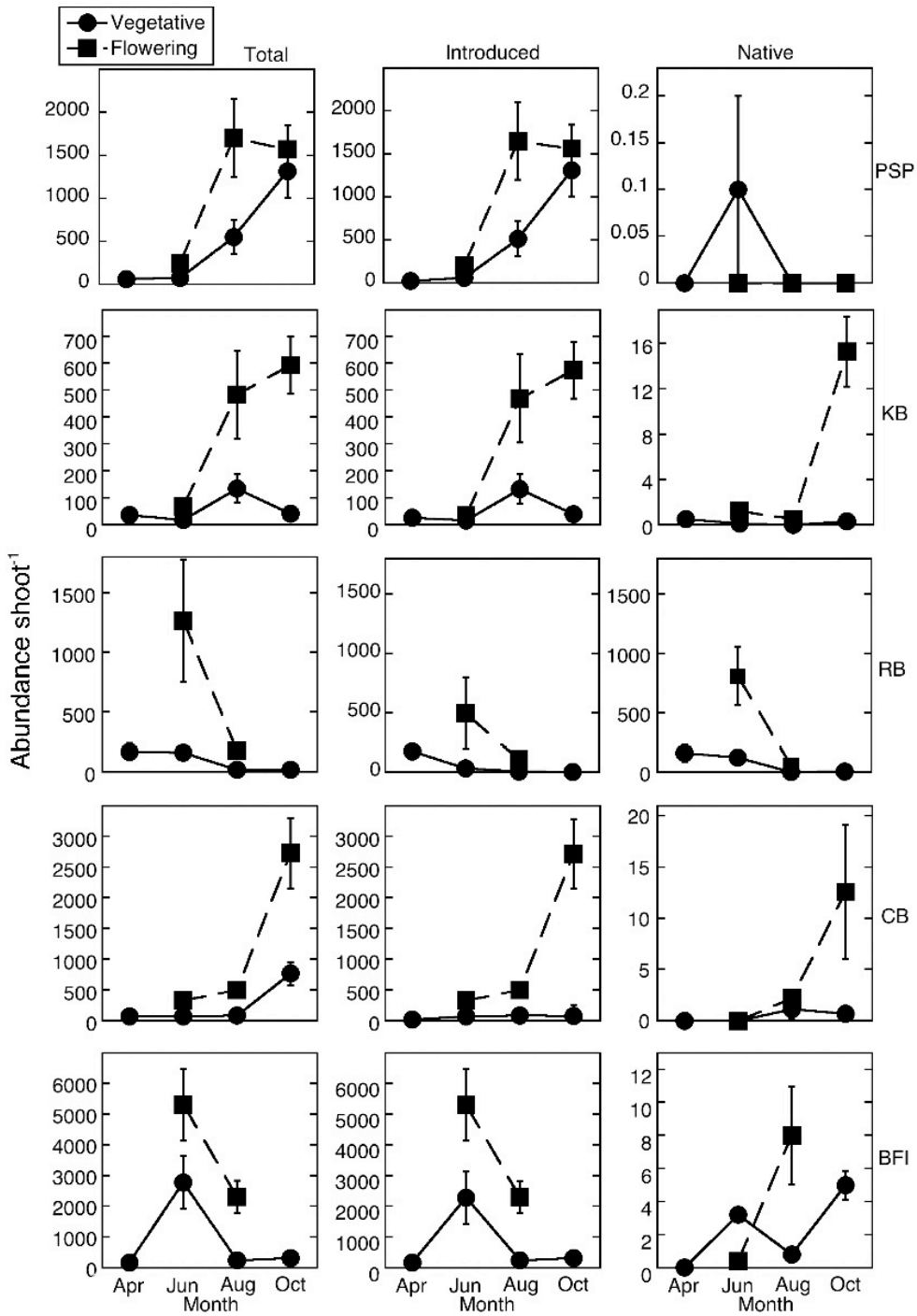
Figure 14. Eelgrass epifaunal abundance on eelgrass shoots at five sites in San Francisco Bay (see Fig. 3) in 2007. From Carr et al. (2011, in press).
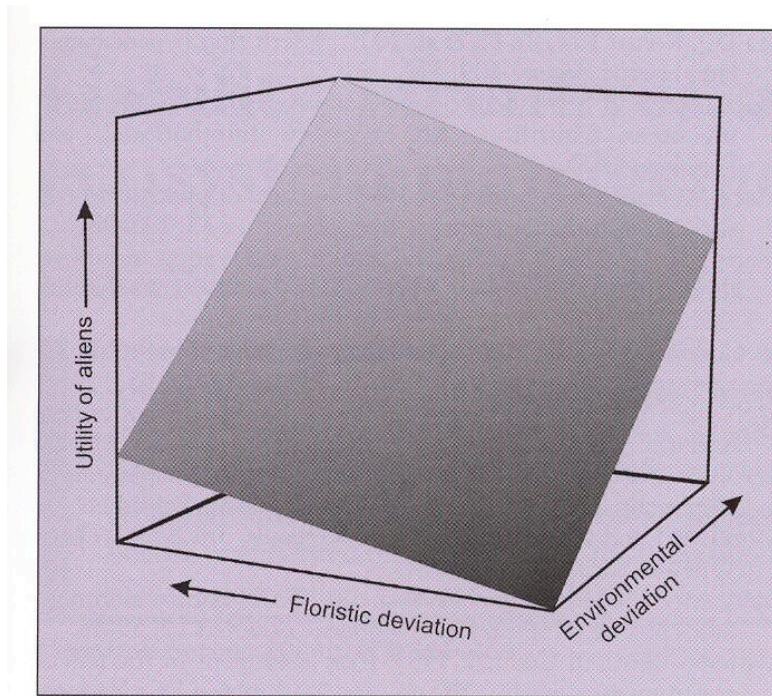
Figure 15. Ewel and Putz (2004) suggest there is "a place for alien species in ecosystem restoration" when either the community of flora or fauna or environmental conditions have deviated dramatically from historic conditions.

Figure 16. Oyster shell bags adjacent to planted eelgrass patch in experiment at Marin Rod and Gun Club to compare food resources for salmonids and other fishes, 2009. Plots contain eelgrass patches alone, oyster shell reef alone, both habitats, or neither. Photo: K. Boyer.

The green, lean, grass-planting machine

Eelgrass Stalks

Wheel

Spoke

Air Hose

CBF scientists and volunteers load bundles of eelgrass onto nozzle-shaped tubes attached to the boat's two metal wheels. As the wheels roll along the bottom of the river, air from a compressor injects the plants straight into the sediment. It would take 20 divers four days to plant the same amount that the boat can plant in one day. CBF hopes to use similar envelope-pushing technology to plant aquatic grasses on a larger scale than ever before on other rivers.

Chesapeake Bay Foundation

Figure 17. Various planting methods for seagrasses have been used in other regions, including planting of bare-root vegetative shoots with bamboo staples. Sods are frequently used to reduce disturbance to roots. Automated equipment has been used for both planting of whole shoots and seeding. Photos: Chesapeake Bay Foundation.
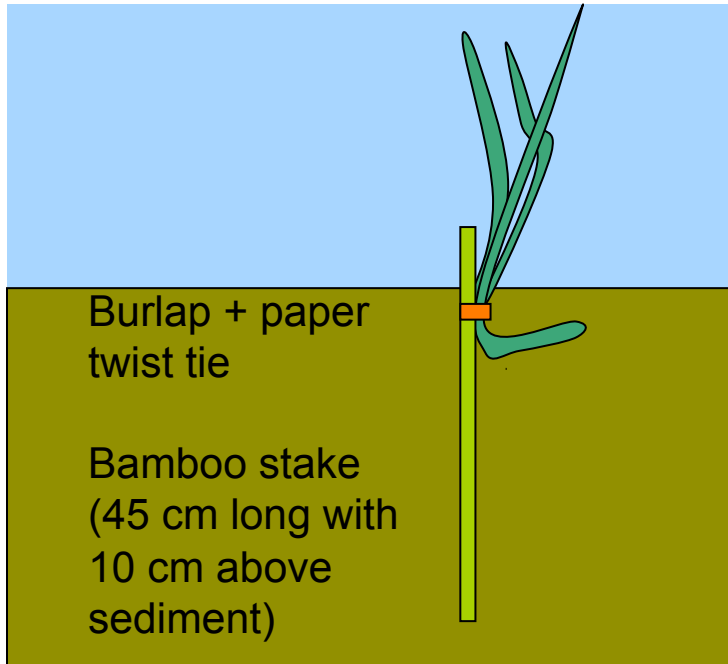
Figure 18. Bamboo stake planting technique developed by San Francisco State University masters student, Stephanie Kiriakopolos. Drawing and photo: S. Kiriakopolos.

Figure 19. Top left, TERFS (Transplanting Eelgrass Remotely with Frame Systems, developed by F. Short, Univ. New Hampshire) frame, showing eelgrass attached with paper ties. Drawing courtesy M. Fonseca. Right, collection of vegetative shoots and attachment to modified frames made of Vexar plastic mesh. Photos: K. Boyer. Bottom left, wood, burlap, and twine frames used in the Clipper Yacht Club mitigation (Sausalito). Photos: J. Semion.

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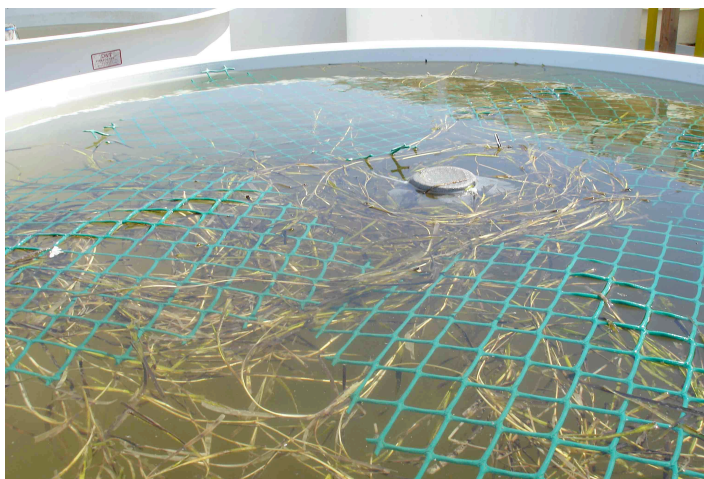
Figure 20. Processing flowering shoots and seeds for hand-broadcasting. Shown at top are flowering shoots with nearly ripe seeds (late stage 4 or 5; DeCock 1980) floating in a tank with flow-through baywater at the Romberg Tiburon Center. Once seeds have dropped to the bottom of the tank they are sorted from other debris, then scattered by hand within marked plots at the restoration site. Processing according to Granger et al. (2002). Photos: L. Reynolds.

Figure 21. Buoy-deployed seeding. Top left: flowering shoot collection at Point San Pablo. Right, top to bottom: Deployment at Marin Rod and Gun Club, Marin Country Day School, and Richardson Bay. Middle, left: pearl nets ready to place on buoys. Bottom left, buoy-deployed seeding in action. Photos: J. Stalker.
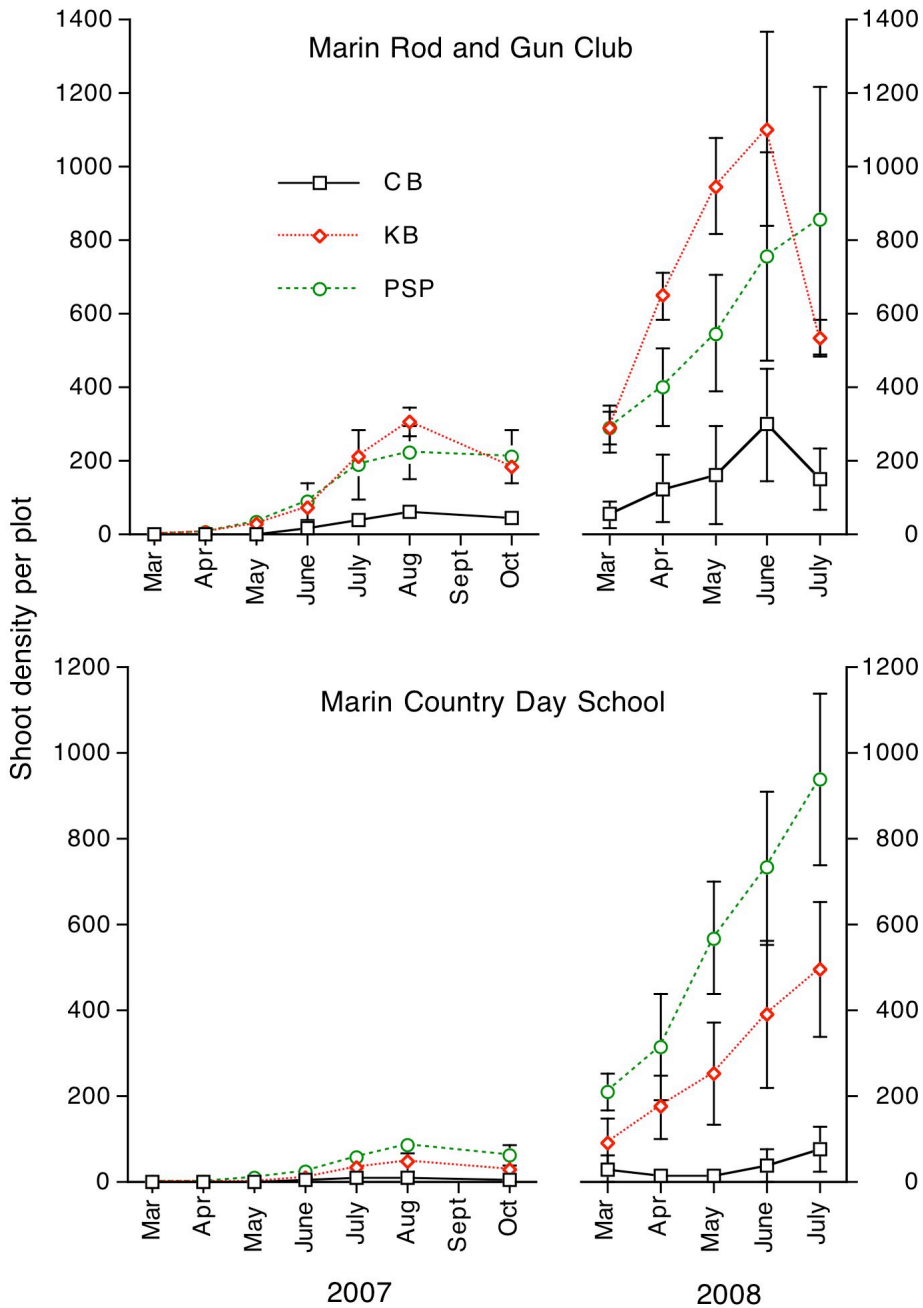
Figure 22. Average shoot density resulting from buoy-deployed seeding using donor populations Crown Beach (CB), Keller Beach (KB) and Point San Pablo (PSP; also known as North Richmond) over time at Marin Rod and Gun Club and Marin Country Day School. Bars represent standard error. From Boyer et al. (2008).
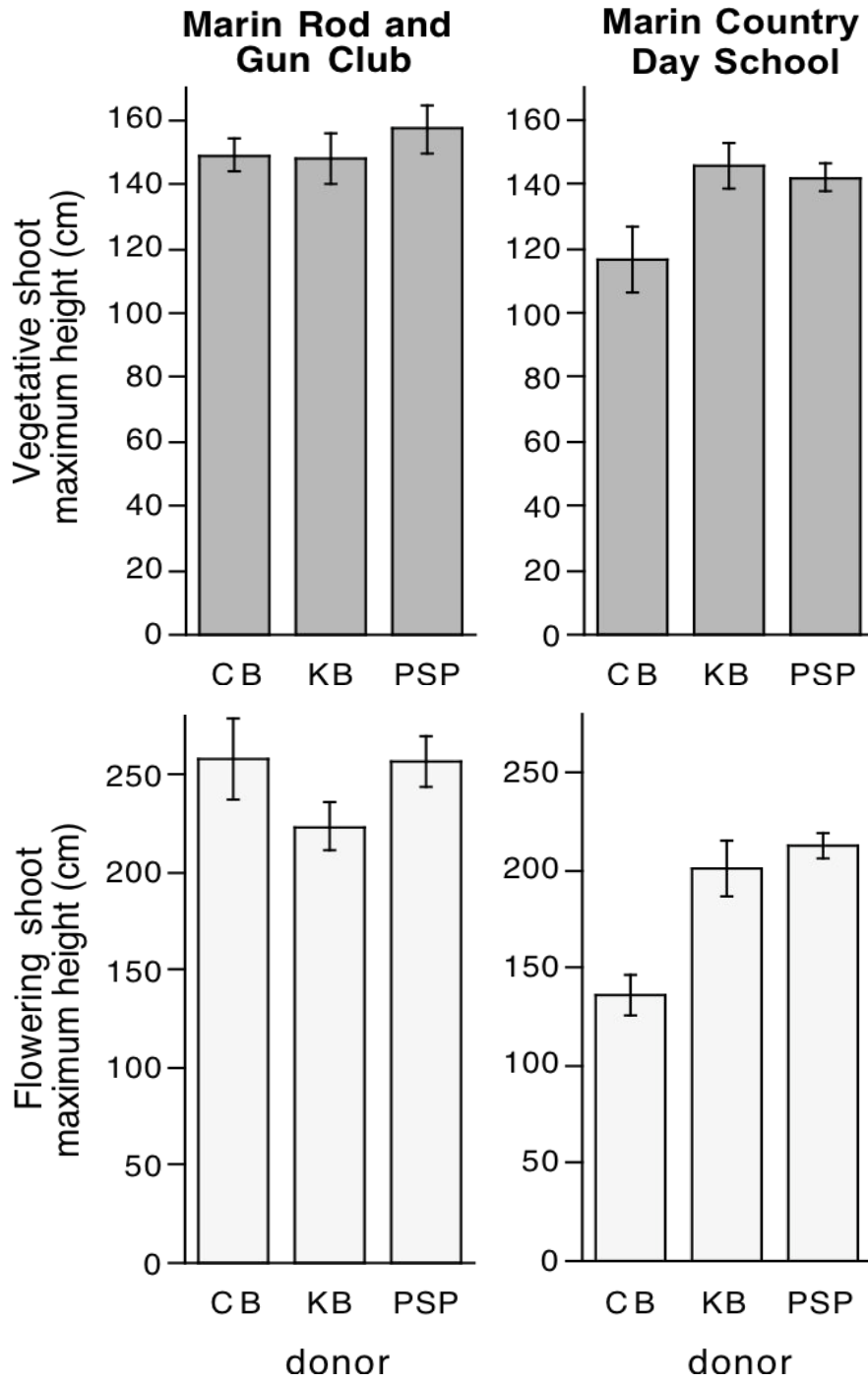
Figure 23. Average shoot heights of vegetative and flowering shoots in the second growing season (July 2008) following buoy-deployed seeding at two restoration sites, Marin Rod and Gun Club and Marin Country Day School. Donor populations were Crown Beach (CB), Keller Beach (KB) and Point San Pablo (PSP; also known as North Richmond). Bars represent standard error. From Boyer et al. (2008).

Figure 24. Test plots in the second growing season at a Nature Conservancy property in San Rafael, July 2009. Nine shoots were planted in each of ten plots in 2007 (using the bamboo stake planting method described in Section IV, bare-root planting units). A ten-fold increase in density occurred by the time of this photo, two years later. From Boyer (2008). Photo: K. Boyer.
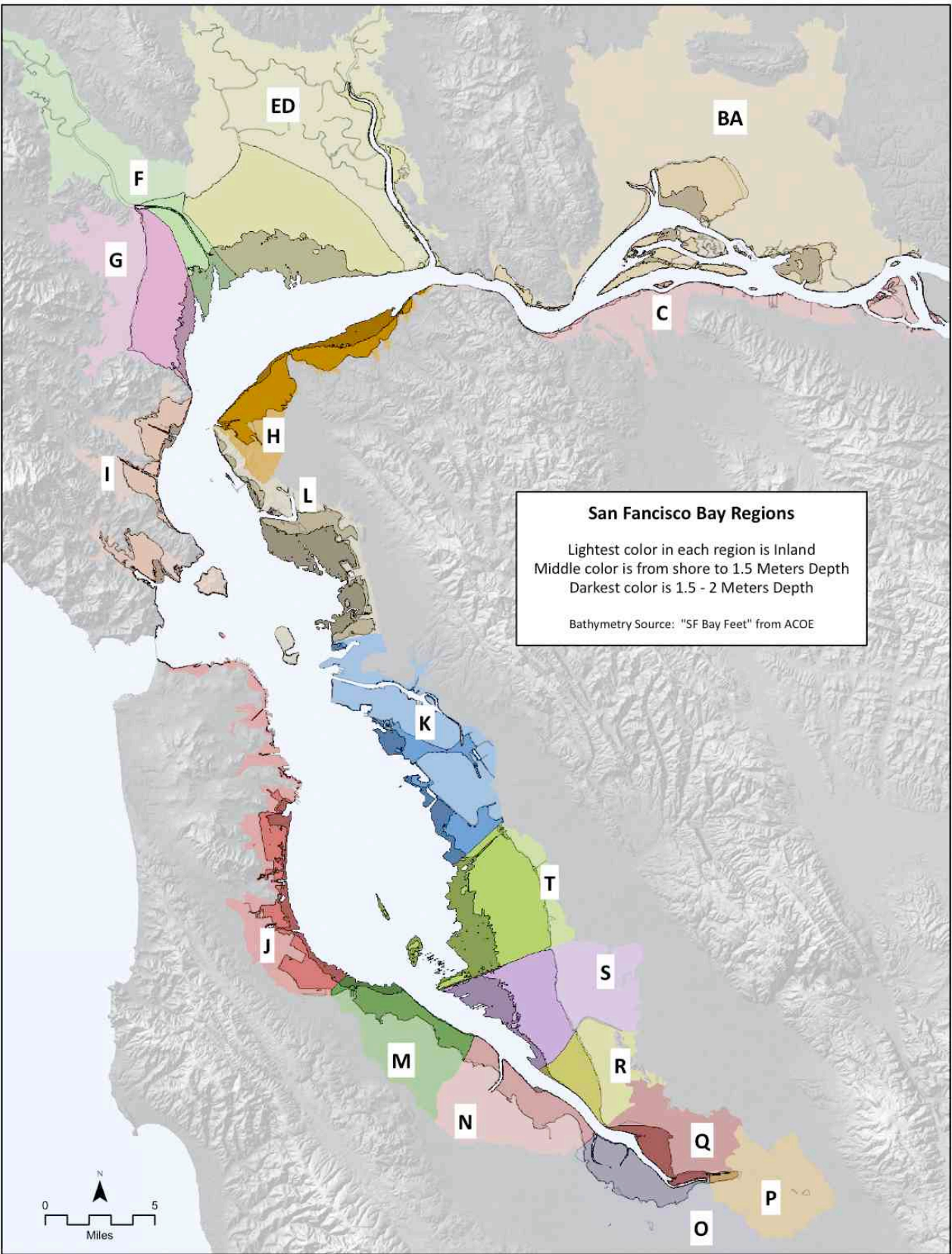
Figure 25. San Francisco Bay regions (segments) used to describe locations of proposed restoration (see section VI and Appendix 1). Note coloration in legend indicating depth to -1.5 or -2 m MLLW.

Segment	Location	GPS	Work to Date						Proposed Action							Eelgrass acreage in 2003 (Merkel and Associates 2004)	Eelgrass acreage in 2009 (Merkel and Associates 2009)	Potential acreage (shoreline to -1.5 m)	Potential acreage (-1.5 to -2.0 m)	Acreage Goals		Notes	
			None	Basic Site Survey (Phase I-1)	Preliminary Evaluation (Phase I-2)	Test Plots (Phase I-3)	Pilot Restoration (Phase II)	Larger Scale Restoration (Phase III)	None	Basic Site Survey (Phase I-1)	Preliminary Evaluation (Phase I-2)	Test Plots (Phase I-3)	Pilot Restoration (Phase II)	Larger Scale Restoration (Phase III)	Protection					5 year	10 year	Opportunities	Constraints
I	Horeshoe Cove to Sausalito Point	37.8320, -122.4776 to 37.8590, -122.4777		x							x					6.01	5.88	21	7			Horseshoe Cove: Sediments sandy, good marine water throughout this segment. Suggest preliminary evaluation and possible future restoration.	Horseshoe Cove: less eelgrass than in past, only 0.5 acres in 2003--unknown constraint, possible dredging impacts. Need bathymetry. Possible change in circulation following build-out of piers and breakwaters. Cove is isolated so loss of plants may mean limited recruitment without restoration. North of Horseshoe Cove, pocket beds; otherwise steep slopes, low potential for expansion.
I	Richardson Bay, Sausalito Point to Peninsula Point	N of 37.8590, -122.4777 and 37.8620, -122.4581		x	x	x	x				x	x			x	448.97	669.70	1,868	321	5	10	Large natural bed near Sausalito. Change to another type of mooring to keep from creating "crop circles". Restoration opportunity to restore these if natural recovery does not occur. Large part of this bay shows as being somewhat suitable habitat in ELVS model. Audubon sanctuary in northeast arm controls boat access in winter, steady source of volunteers, motivated and able staff. Suggest investigation of constraints and test plots. Add channels to bring in cooler water? Important wildlife habitat; herring, waterfowl. Possible synergy with oyster and intertidal restoration near Arambaru Island. Acreage increased in this segment as a whole between 2003 and 2009.	Main bed very patchy in center, more stable along channel by Sausalito. Methylmercury hotspot. In Audubon sanctuary, seed buoy restoration (see Corte Madera and San Rafael Bays) not successful. Extreme variation in depth--due to bat rays? Natural bed relatively deep; small-scale transplants to shallower depths not successful (Kiriakopolos SFSU thesis, in prep.), perhaps due to lower light conditions (sediment resuspension). West arm: high fluvial influences, extremely shallow, probably not much opportunity.
I	Belvedere/Tiburon, Peninsula Point to Bluff Point	37.8620, -122.4581 to 37.8801, -122.4388		x		x									x	42.56	43.62	55	44			Clean, cool oceanic water. Modified TERF frames used by WRA for mitigation along Belvedere relatively successful, and natural recovery also prevalent. Keil Cove: privately owned, shoreline owners make access possible. Relatively pristine, active wildlife area (herring, seals, birds). Scarred by barge in clean up following Cosco Busan oil spill; communication with subtidal owners may lead to better access in future. One of best historical records (Setchell).	Channel dredging and dock maintenance around Belvedere lead to small scale damages, followed by mitigation efforts. Some damages probably not noticed or mitigated; protection needed. Deep channel at Raccoon Strait is sink for sediment. Keil Cove previously difficult to access due to unwilling subitdal landowners, but this may be changing (N. Cosentino-Manning, pers. comm.).
I	Angel Island	37.8612, -122.4464 to 37.8617, -122.4183		x											x	5.20	3.87	66	12			Clean, cool oceanic water. Unique, marine influence, marine species of macroalgae. Heavily utilized by harbor seals. National Park Service.	Narrow space for beds, drop off to deep water quickly. Little opportunity for expansion.
I	Tiburon E. shoreline, Bluff Point to Paradise Cay	37.8801, -122.4388 to 37.9168, -122.4750		x											x	13.56	18.67	159	34			Paradise Cove beds some of oldest known (Setchell). Some willing landowners identified.	Bluff Point to Romberg Tiburon Center, drops off to deep water quickly. Paradise Cove plants very ephemeral. Many owners of subtidal would take coordination effort.
I	Corte Madera Bay, Paradise Cay to Pt. San Quentin	37.9168, -122.4750 to 37.9424, -122.4783		x	x	x	x					x	x	x		0.52	0.78	1,160	107	10	25	Successful seed buoy restoration site ("Marin Country Day School") in 0.5 acre area has persisted and spread over 3 growing seasons. Recruitment from escaped buoys 100+ meters from restoration site suggests more acreage suitable. Extensive additional acreage--recommend large scale restoration. Also additional shallower acreage--recommend test plots and small scale experiments. Identified as Living Shorelines experimental site. BCDC study in Corte Madera marshes to evaluate marsh buffering of sea level rise; potential for collaboration.	Not clear if shallower depths suitable, thus recommend test plots and experimentation. Ferry wakes from Larkspur Ferry. Difficult to keep marker buoys in place. Windsurfing area adjacent to San Quentin prison.

Segment	Location	GPS	Work to Date: None	Basic Site Survey (Phase I-1)	Preliminary Evaluation (Phase I-2)	Test Plots (Phase I-3)	Pilot Restoration (Phase II)	Larger Scale Restoration (Phase III)	Proposed Action: None	Basic Site Survey (Phase I-1)	Preliminary Evaluation (Phase I-2)	Test Plots (Phase I-3)	Pilot Restoration (Phase II)	Larger Scale Restoration (Phase III)	Protection	Eelgrass acreage in 2003 (Merkel and Associates 2004)	Eelgrass acreage in 2009 (Merkel and Associates 2009)	Potential acreage (shoreline to -1.5 m)	Potential acreage (-1.5 to -2.0 m)	5 year	10 year	Opportunities	Constraints
I	San Rafael Bay, Pt. San Quentin to Pt. San Pedro	37.9424, -122.4783 to 37.9854, -122.4471		x	x	x	x				x	x	x			1.65	0.09	2,100	383	10	25	Successful seed buoy restoration site at Marin Rod and Gun Club in 0.5 acre area has persisted and spread over three growing seasons. Additional current experiment to test synergy of oyster reef and eelgrass restoration at Club. Successful test plots (transplants with bamboo stake technique) just south of San Rafael Channel; 10-fold increase in density in 2 years. Extensive acreage available in this region. Willing landowners.	Not clear if shallower depths suitable, thus recommend test plots in shallower zones and also in region between current test plots and Marin Rod and Gun Club.
G	China Camp area, W. San Pablo Bay, Pt. San Pedro to Petaluma River	37.9854, -122.4471 to 38.1101, -122.4899		x		x					x	x				0.00	0.04	8,021	1,342			ELVS model suggests some areas suitable. Whole shoot transplants may be possible with secure planting method (e.g., bamboo stake method); recommend test plots. China Camp is state park and NERR site.	Seed buoys along China Camp not successful in 2006. Strong current. State Park does not want stakes or buoys visible from land.
DEF	N. San Pablo Bay, Petaluma River to Carquinez Bridge	38.1101, -122.4899 to 38.0622, -122.2287	x							x	x	x				0.00	0.15	22,387	8,335			New bed has appeared at CSU Maritime in Vallejo just west of Carquinez Bridge (2008?); suggests area more suitable than previously thought.	Very shallow mudflats in much of north portion of San Pablo Bay probably not suitable.
ABC	Carquinez Strait, Carquinez Bridge to Pittsburg	38.0622, -122.2287 to 38.0643, -121.8249		x							x	x	x			0.00	0.09	9,657	4,321			Small patches inside and just outside Glen Cove Marina (E of Carquinez Bridge) suggests opportunity in this region not previously realized.	Brackish conditions in wetter winters than those of last few years may not support eelgrass (may see a contraction of distribution with wetter winters).
H	SE San Pablo Bay, Carquinez Bridge to Pt. Pinole	38.0622, -122.2287 to 38.0141, -122.3664		x							x	x	x			0.00	135.64	2,321	1,610			Many new patches in this region in last few years. As bay becomes more saline, this region may present opportunities. Large area of potentially suitable habitat not identified in ELVS model (Merkel 2005).	Recruitment may be limited due to protuberance of Point Pinole. Brackish conditions in wetter years.
H	North Richmond Shoreline, SW San Pablo Bay, Pt. Pinole to Pt. San Pablo	38.0141, -122.3664 to 37.9654, -122.4285		x							x	x			x	1,513.10	2,016.81	3,561	398			Largest bed in bay. Good source for collection of restoration material, as minimal impacts due to size. Very important to protect. Large increase in acreage between 2003 and 2009 surveys. Offshore shoal but is area available for shoreward migration--possibility of maintaining bed as sea level rises. Potential to restore shoreward areas of existing bed should be evaluated through phase I-2 and I-3 actions.	Ferry wakes. Enough sediment to maintain over time? Sediment transfer facility for Hamilton Wetlands sink for sediment?
L	Richmond Shoreline, Pt. San Pablo to Pt. Isabel	37.9654, -122.4285 to 37.896, -122.325		x	x	x	x				x	x			x	310.05	402.40	521	219			Eelgrass along much of shoreline and on several offshore shoals. Increase in acreage in this segment in 2009 compared to 2003 survey. Pt. Molate self-recovering now and being restored with seed buoys and transplants (2009-2010). Its damage in 2006 (see right and report text) provided opportunity to develop small scale nursery and test maintenance of genetic diversity in nursery stock at Romberg Tiburon Center/SFSU. Cozy Cove bed privately owned--owners interested in protecting. Keller Beach bed owned by East Bay Regional Parks. Previously was eelgrass beween Brooks Island and directly north--potential for restoration (recommend Phase I-2&3)?	Castro Cove bed contaminated. Pt. Molate Beach subject to spills of hydrocarbons and vegetable oil in past. Bed disappeared in 2006--cause unknown. Planned large casino resort and ferry service impacts uncertain. Keller Beach heavy recreational use--monitoring stakes frequently removed, but most of bed deep enough may not be heavily impacted by beachgoers. Richmond Training Wall transplants only marginally successful in late 80's. Contamination at UC Richmond Field Station--issue for restoration in this area?

Segment	Location	GPS	Work to Date: None	Basic Site Survey (Phase I-1)	Preliminary Evaluation (Phase I-2)	Test Plots (Phase I-3)	Pilot Restoration (Phase II)	Larger Scale Restoration (Phase III)	Proposed Action: None	Basic Site Survey (Phase I-1)	Preliminary Evaluation (Phase I-2)	Test Plots (Phase I-3)	Pilot Restoration (Phase II)	Larger Scale Restoration (Phase III)	Protection	Eelgrass acreage in 2003 (Merkel and Associates 2004)	Eelgrass acreage in 2009 (Merkel and Associates 2009)	Potential acreage (shoreline to -1.5 m)	Potential acreage (-1.5 to -2.0 m)	Acreage Goals: 5 year	10 year	Opportunities	Constraints
L	Albany/Berkeley/Emeryville, Pt. Isabel to Bay Bridge	37.896, -122.325 to 37.8188, -122.3190		x	x	x	x			x	x				x	94.35	96.48	3,088	5,256	5	10	Sites along Albany, Berkeley, Emeryville owned by Eastshore State Park. Possible opportunity to enhance eelgrass along Albany shoreline (south of bulb). EBRPD interested in conducting restoration at Brickyard Cove. Test plots transplanted with bamboo stake technique lasted for one year at Emeryville Crescent (but did not last 2 yrs--see right)--possible that deeper restoration would be more successful. Interest in Living Shorelines concept along Albany and Berkeley shores.	Albany Mudflat test plots not successful in first year--heavy sedimentation may be cause. Pilot scale transplants at Berkeley North Basin on sand fill not successful but seedling recruitment occurring. Possible future ferry terminal in Berkeley area. Bird herbivory heavy on Emeryville mudflat test plot plants in first year, may have been responsible for plants not persisting into second year. Emeryville flats bed shaded by new Bay Bridge span (0.6 hectares). Decline in acreage for segment as a whole between 2003 and 2009 survey.
K	Oakland Area, Bay Bridge to S of Bay Farm Is. at San Leandro Marina	37.8188, -122.3190 to 37.6978, -122.1947		x							x		x	x	x	408.58	311.99	3,681	2,294	15	42	Oakland Middle Harbor sediment grading planned for completion in 1-2 years followed by planting by Port of Oakland and ACOE--42 acres required. Possibility for restoration at tip of former Alameda Naval Air Station? Crown Beach in Alameda owned by East Bay Regional Parks and local staff very supportive of research in bed. Primarily offshore shoal bed at Bay Farm Island isolated from many human impacts.	Heavy boat traffic and shoreline armoring along much of Oakland and Alameda. Large majority of Crown Beach bed annual; may be at greater risk to damages because relies on seed to recruit each year. Heavy recreational use at Crown Beach, windsurfing community resistent to monitoring stakes, buoys. Boat scarring at Bay Farm Island bed; strong ferry wakes. Concern that this offshore bed will not have opportunity to adjust elevation upslope as sea level rises. Decline in acreage for segment as a whole between 2003 and 2009 surveys.
T	Hayward area, San Leandro Marina to Highway 92	37.6978, -122.1947 to 37.6174, -122.1534		x							x					0.00	0.00	10,105	4,805			ELVS model shows potential acreage in northen part of this segment and anecdotal evidence of eelgrass present here in the past (Mark Taylor, EBRPD, pers. comm.). Suggest preliminary evaluation in this area.	No finding of eelgrass in this region by 2003 or 2009 sidescan surveys. Area distant from known eelgrass populations--may be recruitment limited.
S	Eden Landing area, Highway 92 to Alameda Flood Control Channel	37.6174, -122.1534 to 37.5639, -122.1335		x	x	x				x	x	x	x		x	21.59	0.04	6,697	2,880	5	10	Possible recruitment limitation. Need density survey throughout segment, preliminary evaluation, and test plots. Monitoring of existing patches needed as Salt Pond restoration progresses. Potential for integration with Salt Pond restoration activities. Pilot scale restoration recommended following successful test plots in 2010. Potential site for Living Shorelines concept. More acreage present in 2010 than found in 2009 sidescan survey (Boyer, pers. obs. 2010).	Southern most known extent of eelgrass--becoming marginal habitat? Breach of South Bay Salt Ponds levees could influence mudflats in region. Decline in acreage from 2003 to 2009 sidescan surveys (but additional acreage found through airboat search, Boyer, pers. obs.)
NOPQR	South Bay, south of Eden Landing and San Mateo Area	S of 37.5639, -122.1335 and 37.5471, -122.2223	x						x							0.00	0.00	14,186	627			Eelgrass may have been abundant in this region historically--records of heavy Brant Geese use.	ELVS model suggests limited opportunity--high residence time.
M	San Mateo area, Coyote Point to Steinberger Slough	37.5899, -122.3306 to 37.5471, -122.2223		x							x	x				~0	0.03	3,410	678			Tiny patch found just south of Coyote Pt. in 2003. ELVS model suggests some opportunity just south of Coyote Pt. Probably recruitment limited as no other beds in vicinity and prevailing winds to east. Conduct density survey and preliminary assessment.	Presence of only very small patch may suggest marginal habitat. ELVS model suggests limited opportunity south of San Mateo Bridge on west side.

Segment	Location	GPS	None	Basic Site Survey (Phase I-1)	Preliminary Evaluation (Phase I-2)	Test Plots (Phase I-3)	Pilot Restoration (Phase II)	Larger Scale Restoration (Phase III)	None	Basic Site Survey (Phase I-1)	Preliminary Evaluation (Phase I-2)	Test Plots (Phase I-3)	Pilot Restoration (Phase II)	Larger Scale Restoration (Phase III)	Protection	Eelgrass acreage in 2003 (Merkel and Associates 2004)	Eelgrass acreage in 2009 (Merkel and Associates 2009)	Potential acreage (shoreline to -1.5 m)	Potential acreage (-1.5 to -2.0 m)	5 year	10 year	Opportunities	Constraints
J	San Francisco Area, Golden Gate to Coyote Point	37.8096, -122.4730 to 37.5899, -122.3306		x						x	x					2.60	0.02	4,240	1,952			Small patches just north of Coyote Pt. in 2003. Probably recruitment limited as no other beds in vicinity and prevailing winds to east--suggest density survey and preliminary evaluation. Owner of Coyote Flats area subtidal may be supportive. ELVS model suggests some potential between Coyote Flats and north to SFO. Due to prevailing easterly winds, restoration in Pier 94-98 area could provide propagules to east side of bay.	Presence of only very small patches and decline in acreage between 2003 and 2009 surveys may suggest marginal habitat.