Estimating the Change in Ecosystem Service Values from Coastal Restoration

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Appendices

Update August 20, 2014: This report has been updated to include an executive summary and a new layout to better highlight key findings. This updated version contains no new data or factual revisions to the previous version dated March 24, 2014.

Appendix Contents

A Benefit Estimation Assumptions

Benefits of coastal ecosystem restoration have accrued during habitat restoration, accrue in the present year, and will continue to accrue the future. To estimate the present value of all benefits from each case study, we used the same basic set of assumptions at each site, listed in this Appendix.

A.1 Benefits During Restoration Trajectories

We treat annual WTP estimates for coastal ecosystem restoration endpoints as the product of total WTP for a perfectly-functioning coastal ecosystem (predicted from each site's benefit transfer equations) and the percentage of restored habitat function achieved the a given year (as estimated based on coastal restoration project documentation and plans reported throughout these Appendices and the main report). Across all restoration case studies, we assumed:

- Available monetization studies used in this study generally report economic values for goods and services at fully-functional coastal habitats. We assume all transfer functions estimate WTP for coastal ecosystems functioning at 100% capacity.

- But, restored coastal ecosystems do not function at 100% of natural capacity for a number of years post-restoration, but proceed to accrue size and functionality on a positively-sloped trajectory defined by baseline conditions, local restoration success, etc.

- Each restored coastal ecosystem eventually reaches a maximum restoration endpoint, at which ecosystem services level off at a relatively stable ecological maximum. Thereafter, the ecosystem provides the maximized ecosystem services until the analysis endpoint (see Section A.2). Metrics of completeness for each case study and habitat type are described in respective report sections, but generally incorporate the area, stability, and/or robustness of ecological services provided.

- Partially-restored habitats provide ecosystem services in proportion to total restoration success achieved. Although habitats considered in this study take some time to fully recover, they may start to provide some partial level of services before restoration is complete. To account for benefits from partially-restored habitat during the restoration process, we scaled annual benefits estimates for a given service by the percent of total expected restoration achieved.

A.2 Total Benefits Accrued and Annualization

- **Benefit Accrual Period:** To standardize benefit estimation across all case studies in this report, we estimated benefits over a total of 40 years, chosen to approximate the lifetime of the shortest-persisting habitat type across all three coastal restoration sites (oyster reefs). The 40-year period begins with zero services in the year construction of each project began, and ends at the 40th year. This approach is consistent with each project's documentation, and it allows comparison across sites because the number of years included in the analysis is the same for each.

  Because all projects did not begin at the same time, the fixed accrual frame results in a benefits period that covers slightly different calendar years: the San Francisco and Alabama project time frames cover 2010 to 2050, and the Virginia time frame covers 2009 – 2049.

  Further, assumptions about the effective lifetime of restored habitats may over- or under-state actual benefits based on unforeseen changes in the assumed flow of, or unit monetary value of, benefits. For example, in the future, society may have lower or higher values for a habitat type or ecosystem service endpoints than we have used in our study. Alternatively, the absence of ecological disturbance may actually allow some habitats to continue existing beyond the average "lifespan," thus increasing the future flow of services.

- **Discounting future economic benefits:** We assumed, as is standard in economic analyses of the present value of future benefits, that society holds a positive rate of time preference. Because people generally feel that receiving benefits now is preferable to receiving benefits in the future, society discounts the value of future benefits relative to current benefits (Conrad, 2010). We discounted future values using a 3% discount rate (U.S. EPA, 2010; U.S. Office of Management and Budget, 2003). All present and annualized values are reported in present-day currency (US dollars, in the 2013 dollar year), and where necessary were converted using the U.S. Bureau of Labor Statistics' Consumer Price Index. Table B-1 provides details on Social Cost of Carbon discounting.

- **Annualization:** If annual costs are to be compared to annual benefits, it can be informative to examine annualized total present values (TPV). We annualized TPV of the entire future stream of future benefits over a period of 40 years: the same length of time over which we calculate benefits (see above).

B Social Cost of Carbon Values

In valuing carbon sequestration services, we applied social cost of carbon (SCC) values reported by the Interagency Working Group on the Social Cost of Carbon (2013).We first converted all values from 2007 currency to 2013 currency (Table B-1) using the US Bureau of Labor Statistics Consumer Price Index[1] and then interpolated SCC values between 2010 and 2050 (Table B-2). After 2050, we assumed SCC values remain constant at 2050 values. We then estimate annual values of CO_2 removals by restored ecosystems by multiplying annual SCC values by CO_2 removals, and discount future values to the present year (2013) before reporting TPV of benefits.

Table B-1. Revised Social Cost of CO_2, 2010-2050 ($/metric ton of CO_2, 2013$).

Year	Discount Rate			
	5.0% Average	3.0% Average	2.5% Average	3.0% 95th Percentile
2010	12	37	58	101
2015	13	43	65	122
2020	13	48	73	145
2025	16	54	79	162
2030	18	58	85	179
2035	21	64	91	198
2040	24	70	98	216
2045	27	74	103	231
2050	30	80	110	248

Table B-2. Average Annual Growth Rates of SCC Estimates between 2010 and 2050.

	Discount Rate			
Average Annual Growth Rate	5.0% Average	3.0% Average	2.5% Average	3.0% 95th Percentile
2010-2020	1.20%	3.20%	2.40%	4.30%
2020-2030	3.40%	2.10%	1.70%	2.40%
2030-2040	3.00%	1.80%	1.50%	2.00%
2040-2050	2.60%	1.60%	1.30%	1.50%

[1] U.S. Department of Labor, Bureau of Labor Statistics (2013). Consumer Price Index, All Urban Consumers. Retrieved on December 5, 2013 from ftp://ftp.bls.gov/pub/special.requests/cpi/cpiai.txt, (Last updated 11/20/2013).

C Environmental Justice Screening Analyses

This appendix summarizes our statistical methods to assess the presence of Environmental Justice (EJ) concerns in the each study area. Each analysis is designed to examine a reasonable geographic area that captures people who live adjacent to or nearby the coastal restoration site, and are thus likely to hold use and/or non-use values for the site. EPA guidelines for determining population subgroups that may be considered environmental justice communities indicate that focus should be given to groups that in the past have borne a greater share of environmental risk. In each case study, we thus limited the EJ analysis scope to the counties surrounding the restoration site. The geographical limits are designed to focus the EJ analysis on local ethnic and socioeconomic groups which have either (a) historically borne a greater share of costs of coastal degradation, or (b) stand to gain substantially greater benefit per capita from restoration compared to members of the average population. In these comparisons, we use the same two socioeconomic indicators to flag the presence of at-risk EJ communities: the prevalence of low-income households, and of minority racial groups. We then construct a composite index of these values to reflect the combined potential for EJ benefits.

C.1 Methods

At each coastal restoration site, the analysis compares the average demographic characteristics of Census block groups in coastal counties immediately surrounding the site to the average resident characteristics in all block groups of the containing state (i.e., California, Virginia, or Alabama). We obtained demographic data from the 2010 U.S. Census at the block group level, and then analyzed block group averages by county and state.

- The U.S. Census provides data on the percentage of the population whose household income is below the poverty line. We used this metric as a continuous measure of areas that are best defined as "low-income" areas (U.S. Census Bureau, 2010).

- We characterized the presence of minority groups using Census data summarizing the percentage of individuals who are non-white minorities.

Then, we calculated population-weighted average percent minority residents and percent households in poverty in Census tracts each of the affected counties, and across the entire state. We also constructed a composite index of these values to reflect the combined potential for EJ benefits. Finally, we compared low-income populations, minority populations, and EJ index values within tracts in the affected counties to tracts at the general state-wide level (using statistical t-tests). Results of these tests are reported in each of the main case study chapters.

D SBSPRP Tidal Wetland Habitat Restoration

D.1 Description of ARRA-funded Restoration Sites in SBSPRP Areas.

D.1.a Alviso Marsh Complex

The Alviso Marsh Complex (AMC) is the southernmost marsh of South Bay and is the location of the earliest restoration actions by the South Bay Salt Pond Restoration Program. The salt ponds restored to tidal flow with NOAA-ARRA funding in 2009-2011 are ponds A6 and A8. Earlier restoration within the AMC include the "Island Ponds" (A19-A21) that were breached in 2006 and which provide a good reference for comparison of the trajectory of habitat restoration (Borgnis et al., 2013).

Pond A6 was restored to tidal habitat in December 2010 by breaching and lowering portions of the outboard levee and excavating pilot channels through the fringe marsh outboard of the breaches. Reintroduction of tidal action to Pond A6 has initially created large areas of mudflat habitat. Over time, tidal channel and vegetated salt marsh habitats are expected to develop in Pond A6 as tidal channels re-form, sediment accumulates, and vegetation establishes on the emerging mudflats.

Pond A8 is a managed pond, and is tidally muted from June through November, with the water levels dictated by flood-control gates during winter months (Hobbs, Moyle, & Buckmaster, 2012).Water depths are usually between 1-3 meters. The pond is surrounded by rip-rap levees with sparse pickleweed (*Salicornia virginica*) marsh. One reason for the installation of flood-control gates is a concern regarding potential mobilization of mercury from the sediments (Slotton & Ayers, 2013).

D.1.b Eden Landing

Three Eden Landing (EL) ponds (Ponds E8A, E8X, and E9) were restored to tidal action by breaching of external pond levees in three locations in 2010. EL ponds lack a clearly defined borrow ditch (shallow ditch just inside levees), which differentiates them from the AMC and Middle Bair Island (MBI) ponds. The EL ponds have vestigial tidal creeks in some areas that are completely intertidal. As of 2012, the newly breached habitat at these locations had already begun to vegetate, primarily with pickleweed (Hobbs, et al., 2012).

D.1.c Middle Bair Island

Due to lower than expected costs for restoration activities in the AMC and EL areas, remaining NOAA-ARRA funding was used for Phase I restoration work on MBI[2]. This work consisted of importing over one million cubic yards of material to increase bottom elevation since the area within the levees had significantly subsided (by up to 12 feet) relative to natural levels (Mruz 2013). Further restoration, including breaching of the levees, will be

[2] In addition, the 2009 ARRA funding also provided for activities that were not quantifiable for this analysis but which, nonetheless, would provide ecological services to South Bay, including the inventory and control of non-indigenous invasive Spartina species to prevent their expansion. This action supported overall plant biodiversity and habitat variability, which are important for maintaining a greater range of bird and mammal species.

conducted in the future as funding becomes available. However, since the actual breaching and restoration will not be conducted under ARRA funding, we did not credit any future wetland acres restored for our analysis.

D.2 Case Studies of Past Restoration in SBSPRP Area

Three case studies provide examples of vegetation development in wetland restoration areas located in San Francisco Bay. These case studies indicate that vegetation development is tightly linked to base elevation relative to tidal range. Pioneer species such as cordgrass and pickleweed are typically found within the first 1 to 3 years with greater vegetative diversity found over time. We used data from these case studies to project development of the wetland vegetative cover over time in SBSPRP area.

D.2.a Muzzi Marsh

The Muzzi Marsh is located on the west side of South Bay just below Ravenswood. It was the first ecosystem restoration (1976) that relied on the natural establishment of salt marsh plants after breaching the dike. Table D-1 shows the colonization timeline for important plant species. Pickleweed first established at higher elevations of the "inner" marsh plain within a year after the dikes were breached and in the next ten years had spread across most of the "inner" marsh (PWA 2004). Cordgrass established in several places in the "outer" marsh within three or four years after the restoration of tidal action, with a substantial cover developed after ten or 15 years. It was considered fully vegetated after 20 years.

Table D-1. Establishment of salt marsh species years after dike breaching.

	Years after dike breaching			
	1 to 3	4 to 6	7 to 10	11 to 14
Perennial pickleweed (*Salicorna virginica*)	x			
Annual pickleweed (*Salicorna europaea*)	x			
Pacific cordgrass (*Spartina foliosa*)	x			
Salt grass *(Distichlis spicata*)		x		
Grass buttons (*Cotula coronipifolia*)		x		
Sand spurrey (*Spergularaia macrotheca*)		x		
Jaumea (*Jaumea carnosa*)			x	
Alkali heath (*Franenia salina*)			x	
Fat-hen (*Atriplex triangularis*)			x	
Gumplant (*Grindelia stricta*)				x

Source: PWA (2004)

D.2.b Cooley Landing

Cooley Landing is located on the east side of the South Bay, just south of the EL area. The area was opened to tidal flushing in 2001. Overall, the restoration activities have been successful in converting the approximately 115 acres of the former salt pond from degraded mudflat, open water, and muted tidal wetland habitat to fully tidal salt marsh habitat.

The restoration activities were expected to lead to sediment deposition and re-establishment of the tidal marsh surface and tidal marsh vegetation within a 10-year monitoring period. The performance criteria for percent vegetated cover milestone targets were: >10% salt marsh plant cover at end of Year 1, >40% cover at end of Year 3, >60% cover at end of Year 6 and >70% cover at end of Year 10 (Binard et al. 2008). Three years after restoration, approximately 15 percent of the site was covered with vegetation, but included stands of non-native smooth cordgrass (*Spartina alterniflora*) that have since been controlled. The failure to meet the original 10 year restoration program was thought to be due to slower-than-expected sediment accretion rates and an associated less rapid elevation build-up. Currently, the area is considered fully vegetated and a significant ecosystem restoration success.

D.2.c Island Ponds

The Island Ponds (AMC A19-A21) were first breached in 2006 and evaluated via the Habitat Evolution Mapping Project (HEMP) method over approximately three to five years after the initiation of tidal flushing during 2009-20011 (BF&F 2012). The HEMP satellite imagery indicates readily apparent floral colonization along historic and developing slough channels for all three years of imagery (BF&A 2012). The growth of pickleweed is most prevalent in Pond A21 between 2009 and 2010, as is the emerging presence of cordgrass between 2010 and 2011. In 2011, there did not appear to be significant increases in pickleweed or cordgrass within these ponds suggesting that further sedimentation may be required before further vegetation development.

D.3 Critical Factors for Re-establishment of Pond Elevation and Vegetation

D.3.a Post-Breach Sediment Accretion and Elevation Changes

The base or bottom elevation of many former salt ponds has subsided due to sediment compaction. This base elevation determines, to a large extent, how much of the pond will be inundated by tidal flow. Inundation rates at a particular location within the marsh will be affected by the local tidal range (which increases substantially at the south end of the Bay elevation, the proximity to tidal channels, and any local features such as pans and natural levees that may impede drainage (PWA 2006).

For tidal salt marsh plants to develop, the base elevation needs to exceed the mean tidal level (MTL). Sediment accrual helps base elevation exceed the MTL cutoff by a combination of settling of suspended solids in tidal waters and the build-up of organic matter via vegetative growth. As the pond elevation increases to MTL, pickleweed initiates vegetative colonization on tidal mud flats. As the surface elevation reaches the limit of high tides, tidal marsh vegetation becomes established. The lowest zone of marsh vegetation is comprised primarily of native Pacific cordgrass (*Spartina foliosa*) that helps to further sequester sediment for development of higher marsh (Williams & Orr, 2002).

Table A-2 shows sediment accretion rates from newly breached (A6) ponds, older restorations (Pond A-6, the Island Ponds A19-21, and Muzzi Marsh), and natural marshes (Callaway, Borgnis, Turner, & Milan, 2012). There is a progressive decrease in sediment accretion over time as bottom elevation rises and the area of tidal inundation decreases. Natural marshes exhibit relatively low rates of sediment accumulation.

Table D-2. Summary of sediment accretion rates in San Francisco Bay.

Location	Initial Restoration Date	Range of Sediment Accretion Rates (mm/yr)
Pond A6	2010	> 200
Island Ponds (A19-A21)	2006	10 - 100
Muzzi Marsh (Northern SF Bay)	1976	3 to 10
Low Marsh (natural)	NA	$\leq$ 6
Mid and High Marsh (natural)	NA	3

Source: Callaway et al. (2012).

D.3.b Restored Wetland Vegetation Development Trends

Vegetation development is the other critical component of wetland habitat development. Review of vegetation development in older restoration sites within San Francisco Bay indicates that primary production in the restored wetlands would rapidly increase (faster than diversity or coverage) such that within 10 years, biomass and carbon export should start to approach natural marshes. Other ecosystem functions such as support of biodiversity, denitrification and carbon sequestration should be commensurate with the relative vegetation cover and development of organic soils in the restored wetland.

Comparison of data and qualitative description of the three case studies (Appendix D.2) indicates that two locations (Muzzi Marsh, Island Ponds A19-21) had well established vegetation consisting mainly of pioneer keystone species (pickleweed, cordgrass) present within three years (PWA 2004, BF&A 2012). Cooley Landing has lagged somewhat in vegetative development, apparently from lower rates of sedimentation (Binard et al, 2008).

As previously stated in this Appendix, restoration performance criteria proposed for the Cooley Landing were >10% salt marsh plant cover at end of Year 1, >40% cover at end of Year 3, >60% cover at end of Year 6 and >70% cover at end of Year 10 (Binard et al. 2008). While such criteria are usually site-specific, we use these criteria as a relative benchmark to gauge how well the three example restoration sites in the South Bay were performing.

HEMP was a three year project aimed at developing methods for tracking long term changes to marsh habitats and mudflats for the SBSPRP and used satellite imagery to delineate vegetation, mudflats, and other habitats within the study area boundary (BF&A 2012). It compared vegetation trends in South Bay wetlands over the period 2009-2011. Although this is a relatively short monitoring period, floral colonization within restored ponds (e.g. Island Ponds, Cooley and Bair Island) was clearly evident and within project accuracy requirements. Based on comparison to over 1000 validation field surveys, this method was considered particularly accurate for newly restored sites which contain both mudflats and sparse vegetation (BF&A 2012).

HEMP imagery taken at MLLW in 2011 – not long after the initial breach in A6— shows a clear accretion of sediment (as mud) within the pond, including mud with biofilm, and small patches of pickleweed forming on historic levee tops in the middle of the pond. Images also indicate that the fringe salt marsh at the top of A6 is expanding onto the mud flat (BF&A 2012). Review of Eden Landing sediment and vegetation indicates these sites are also progressing rapidly into the first phase of mudflat development and initial development of pickleweed.

Spartina species in general have very high rates of growth and biomass accumulation. For example, aboveground net primary production (NPP) of smooth cordgrass (*Spartina alterniflora*) in three East Coast tidal marshes (GA, NC, SC) averaged 1,402 while belowground NPP was 3,904 g/m^2/yr. (Dames and Kenny 1986). The native Pacific cordgrass (*S. folisa*) does not grow or expand as fast as smooth cordgrass which has led to a biological invasion and hybridization of some areas by *S. alterniflora*. [Note: some of these areas were treated with herbicide under the *Spartina* control program sponsored by NOAA-ARRA funding.]. The height and NPP of Pacific cordgrass is about ½ that of smooth cordgrass and productivity up to 500-650 g/m^2/yr have been reported in San Francisco Bay (Brusati & Grosholz, 2006).

E SBSP Fish Resource Restoration

Coastal tidal marshes provide important habitat for fish but investigators often find large fluctuations in species composition and numbers. This variability is due to a combination of seasonal and tidal movements of fish as well as population responses to local physical stressors (e.g., salinity, temperature) and biotic factors (e.g., prey abundance, predators).

Table E-1 shows recent fishery investigations and/or summaries of fishery communities in South Bay from 1989 to 2013. This period includes both pre-restoration (>2009) and post-restoration (2010-2011) conditions relative to the ARRA-funded wetlands. For purposes of our analysis, we primarily focused on results from the most current studies (URS 2008, USGS 2009, and Hobbs and Moyle 2013), but we briefly describe findings from the other studies below.

Lonzarich (1989) studied six ponds within the AMC (A9-A13 and A15) when these ponds were still being used for salt evaporation, with salinities ranging from 20 to 88 parts per thousand (ppt) and abundant populations of Franciscan brine shrimp (*Artemia franciscana*) as a primary food source for salt-tolerant fish and bird species. Over 12,000 specimens representing 15 species were collected from fall 1985 to fall 1986 using a combination of seines, gill nets, and minnow traps. The catch was dominated by topsmelt (*Atherinops affinis*) and longjawed mudsucker (*Gillichyths mirabilis*) which were resident and able to spawn at all salinities. Other species tolerating high salinities included: threespine stickleback (*Gasterosteus aculeatus*), rainwater killifish (*Lucania parva*), yellowfin goby (*Acanthogobius flavimanus*) and Pacific staghorn sculpin (*Leptocottus armatus*). Nine species, including striped bass (*Morone saxatilis*), were seasonal transients found only in low salinity ponds (i.e., <40 ppt).

Summaries of biological resources in the South Bay and review of prior fishery work are reported in the Draft and Final Environmental Impact Report (EIR) for the SBSPRP (Life Sciences 2003, USF&WS/CDF&G 2008). These report sections provide qualitative descriptions of the historic San Francisco and South Bay fishery community and relevant background literature but do not provide quantitative fish data. However, both reports stress the importance of forage fish and invertebrates found in the wetlands to the overall health of the Bay fishery.

URS Corporation conducted quarterly sampling (March, June, September, October) at multiple sub-sites in four restored South Bay wetlands adjacent to AMC and EL during 2006, employing several methods including beach seine, gill net and otter trawl (URS 2008). They collected a total of 4,937 fish and a total of 18 species of fish (10 native species), ranging from 8 to 11 species identified at each of the 4 wetlands. The number of species captured was highest in spring and decreased in summer, late summer, and fall. The predominant species in the catch was topsmelt (86% of total catch), followed by threespine stickleback (5%), northern anchovy (3%), longjaw mudsucker (2%) and rainwater killifish (2%) (Table E-2). Similar trends are seen in the USGS survey (conducted just two months after breaching) but some flushing has allowed for a large number of stickleback to enter freshwater habitats.

Table E-1. Fishery Studies Conducted in the General Vicinity of Selected Pond Restorations (1985-2011).

Investigator(s)	Sponsor	Sampling Year(s)	Areas Studied	Fishing Methods	Numbers	Species Present	Relative Abundance	Quant. Measure	Biodiversity Index
Lonzarich (1989)	M.S. Thesis	1985-1986	A9-A13, A15	Gill net, beach seine	Yes	Yes	Yes	CPUE	No
Life Science, Inc. (2003)	USFWS & CDF&G	Summary of older studies	General	Various	No	Yes	No	No	No
H. T. Harvey & Associates (2005)	CA Coastal Conservancy	1980-2002	South Bay waters near Dumbarton Bridge	Otter trawl, plankton net	Yes	Yes	Yes	No	Yes
USFWS & CDF&G (2007)	USFWS & CDF&G	Summary of older studies	A6, A8, E8A, E8X, E9	Various	No	No	No	No	No
URS Corporation (2008)	NOAA	2006	Outer Bair Island, adjacent to Eden's Landing	Otter trawl, gill net, seine	Yes	Yes	Yes	No	No
Saiki & Mejia (2009)	USGS	2006	A19-A21	Gill net, minnow traps	Yes	Yes	Yes	Yes	No
Hobbs & Moyle (2012)	SBSPRP	2010-2011	A6, A19-A21, Bair Island, E8A, E8X, E9	Otter trawl, beach seine	Yes	Yes	Yes	CPUE	Yes

References:

Lonzarich, D.G. 1989. Temporal and spatial variations in salt pond environments and implications for fish and invertebrates. M.S. thesis. San Jose State University. SJSU Scholarworks Paper 3087.

Life Science, Inc. Draft South Bay Salt Pond Initial Stewardship Project. Environmental Impact Report / Environmental Impact Statement. Prepared for USF&WS and CA Department of Fish and Game.

H.T. Harvey Associates. 2005. South Bay Salt Ponds Restoration Project - Biology and Habitat Existing Conditions. Prepared for CA Coastal Conservancy.

U.S. Fish and Wildlife Service and CA Department of Fish and Game. 2007. Final Environmental Impact Statement/Report (FEIS/R) for the South Bay Salt Pond Restoration Project. December 2007.

URS. 2008. Fisheries in Restored South Bay Wetlands and Adjacent Habitats. Prepared for NOAA/National Marine Fisheries Service, Sonoma, CA.

Saiki, M.K. and F.H. Mejia. 2009. Utilization by fishes of the Alviso Island Ponds and Adjacent Waters in South San Francisco Bay following restoration to tidal influence. California Fish and Game 95:38-52.

Hobbs, J.A. and P. Moyle. 2012. Monitoring the Response of Fish Communities to Salt Pond Restoration: Final Report. University of California at Davis.

Table E-2. Summary of Pre- and Post-Restoration Fishery Studies Results.

Investigator(s)	Sampling Year(s)	Areas Studied	Total Numbers	Species Present	# Bay-Resident Species[1]	S-W Diversity Index (H')	Simpson Index (D)	Most Abundant Species. (>95% catch)
Lonzarich (1989)	1985-86	AMC Ponds (A9-A13, A15)	>12,000	15	100%	0.67	0.65	topsmelt, Pacific staghorn sculpin, yellowfin goby, longjawed sucker, threespine stickleback, rainwater killifish
URS Corporation (2008)	2006	Four restoration sites in South Bay	4,937	18	78%	0.67	0.74	topsmelt, threespine stickleback, northern anchovy, longjawed sucker
Saiki & Mejia (2009)	2006	Island Ponds (A19-A21) and adjacent sloughs	4,034	18	61%	1.15	0.45	threespine stickleback, topsmelt, northern anchovy, striped bass
Hobbs & Moyle (2012)	2010-2011	Alviso Marsh Complex	23,138	35	49%	1.94	0.22	threespine stickleback, Pacific staghorn sculpin, Pacific herring, English sole, northern anchovy, arrow goby, yellowfin goby, topsmelt, Mississippi silversides, and longfin smelt
	2010-2011	Island Ponds (A19-A21)	13,146	27	57%	1.92	0.22	threespine stickleback, Pacific staghorn sculpin, Pacific herring, English sole, northern anchovy, arrow goby, yellowfin goby, longfin smelt, and starry flounder

Notes: (1): List of bay resident and seasonal species taken from San Francisco Fish Index (CDF&G 2001).

Lonzarich, D.G. 1989. Temporal and spatial variations in salt pond environments and implications for fish and invertebrates. M.S. thesis. San Jose State University. SJSU Scholarworks Paper 3087.

URS. 2008. Fisheries in Restored South Bay Wetlands and Adjacent Habitats. Prepared for NOAA/National Marine Fisheries Service, Sonoma, CA.

Saiki, M.K. and F.H. Mejia. 2009. Utilization by fishes of the Alviso Island Ponds and Adjacent Waters in South San Francisco Bay following restoration to tidal influence. California Fish and Game 95:38-52.

Hobbs, J.A. and P. Moyle. 2012. Monitoring the Response of Fish Communities to Salt Pond Restoration: Final Report. University of California at Davis.

USGS staff conducted a fishery study to document the post-restoration effects of levee breaching on fish utilization of the AMC Island Ponds (A19-A21) and adjacent sloughs (Saiki and Mejia 2009). Sampling was conducted once per month in May-July 2006. A total of 4,034 fish representing 18 species were collected using gill nets and minnow traps (USGS 2009). The proportions of captured species varied significantly between gill nets and minnow traps. Gill nets collected mostly topsmelt (68.7% of total catch), striped bass, (14.7%), and northern anchovy (*Engraulis mordax*) (7.3%), while minnow traps caught threespine stickleback (96.5%). The authors concluded that the fish assemblage composition was most dictated by the salinity gradient. They speculated that the restoration of former salt ponds to tidal marshes was likely to benefit recreational and commercial fisheries by increasing forage production and providing additional foraging and rearing habitats (Saiki and Mejia 2009).

Hobbs et al. (2012) recently completed a two-year fishery study to document fish abundance and species assemblages within the restored salt ponds and adjacent sloughs in both AMC (A6; A19-A21) and EL (E8A, E8X, E9) locations. Methods differed among the areas because of depth and configuration and included otter trawls, beach seines, and minnow trap methods. EL locations were less accessible since newly breached and sampled only by beach seine.

Approximately 30,000 fish representing 41 species were collected during the survey (Hobbs and Moyle 2012). Diversity at both AMC (35 species) and EL (27 species) was higher than seen in earlier pre-restoration surveys (Table E-2). The larger numbers of fish and diversity are at least partially due to the longer sampling effort, but may also reflect greater diversity of habitat.

F SBSPRP Avian Habitat Restoration

Data on the avian community is available from the USGS and other sources. The USGS monitored avian populations in the 53 ponds in the AMC, EL, and Ravenswood complexes from July 2005 to August 2006 (Takekawa, Athearn, Hattenbach, & Schultz, 2006). The period of these surveys overlaps the March 2006 breaching of the AMC Ponds A19-A21 (also known as the "Island Ponds") and provides avian population trends during the early periods of habitat transition.

Pre-inundation, the Island Ponds avifauna were dominated by gulls (87% of the bird population) that used the site primarily for roosting between foraging visits to nearby landfills (Takekawa, et al., 2006). Post-restoration, once tidal fluctuations were in place, the guild composition during high tide remained similar to guild composition before breaching, although relative abundance changed. However, gull composition decreased by 20% of the total bird count while shorebird composition rose 22%, suggesting increasing use of exposed tidal flats for feeding by shorebirds. Overall counts of foraging rates rose 43% during high tides and 26% during low tides (Takekawa, et al., 2006). High-tide population counts were significantly greater in July and August of 2006 compared to the same months in 2005 (Figure F-1), probably due to increased shorebird foraging in the tidal flats exposed by breaching of the levees.

From 2005 to 2006, Ponds A6 and A8 were characterized by high bird counts with populations disproportionately dominated by nesting gulls, similar to the pre-restoration AMC Island Ponds (Takekawa, et al., 2006). A high density of saline specialists gave pre-restoration EL ponds E8A and E9 the highest and third highest bird populations, respectively, with prevalent species including small shorebirds, herons, and phalaropes (Takekawa, et al., 2006).

Long-term post-restoration monitoring data on the ARRA-funded ponds is not yet available, but some data exist for the 2010 and 2011 SBSPRP restoration areas. The breaching of Pond A6 in 2010 coupled with gull hazing (i.e., non-lethal removal activities) removed the single largest gull colony (23,108 California Gulls). However, between 2011 and 2012 the gull population across the South Bay increased by 38% (Donehower, Tokatlian, Robinson-Nilsen, & Strong, 2013). This displacement and population growth helped fuel concerns of gulls impacting sensitive species (i.e., snowy plover) by egg predation or competition for nesting sites. However, these concerns have not been realized as gulls did not appear to nest in any sensitive habitats in 2012 (Donehower, et al., 2013).

Recent regional-level analysis may provide context to current population trends. In the 2013 South Bay Science Symposium, Takekawa and co-workers (Takekawa, Smith, & Moskal, 2013) presented avian population models using a post-restoration scenario with the hypothesized future 50:50 ratio of restored tidal-marsh habitat to ponds across South Bay (Goals Project, 1999). This scenario predicted that water bird populations would decrease 21% during the winter, and increase 213% in the spring (Takekawa 2013). The annual range in avian occupants is a highly variable (-21% (winter) to +213 (spring)), but suggests an overall increase in diversity and population sizes.

A final source of avian population trend data for AMC and EL is observational data provided by Point Rey Bird Observatory (PRBO) and Cornell Ornithology's AKN (Avian Knowledge Network, 2013). Available by region from the Cornell Lab of Ornithology, this data provided bird counts from a network of volunteer bird watching enthusiasts following rigorous, standardized reporting guidelines. The non-systematic volunteer nature of the observations resulted in significant variation in the number of annual observations from year-to-year, including during the 2010 construction closure of EL. While this data does not provide comparable population numbers, it does provide information on relative avian species diversity. Between 2005 and 2013, the observed avian species diversity in AMC and EL tended to increase by 2.6- and 4-fold, respectively (Figure F-1).

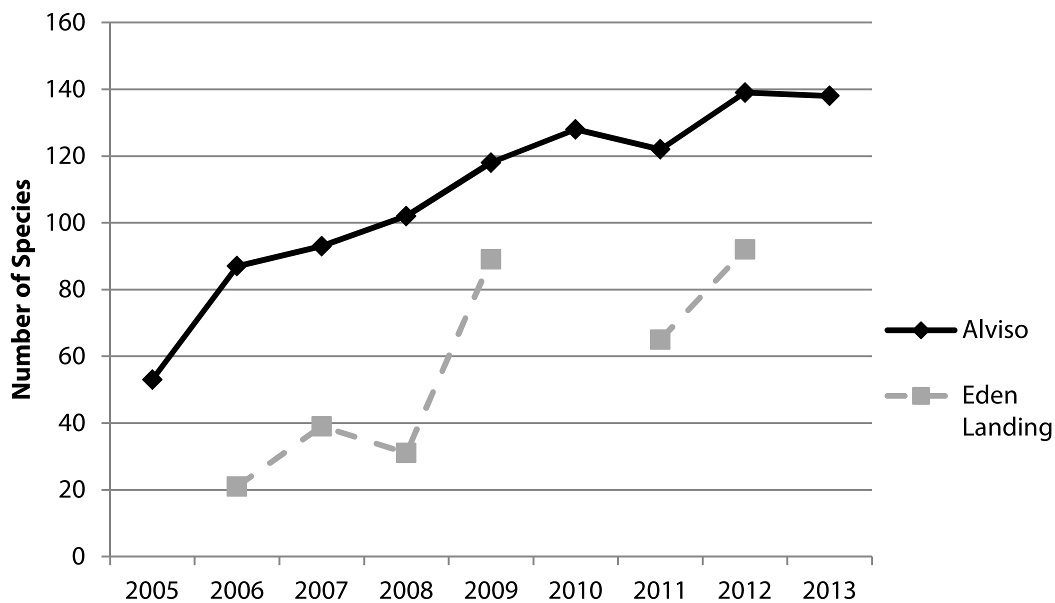

Figure F-1. Species Diversity at Alviso and Eden Landing (2005-2013).

Source: Abt Associates based on PRBO and Cornell Ornithology's AKN.

G SBSPRP Habitat Restoration for Clapper Rail and Salt Marsh Harvest Mouse

G.1 California Clapper Rail

Of critical concern for the long-term survival of the California clapper rail is the fragmented state of the tidal salt marshes around San Francisco Bay. Small isolated fragments of marsh lead to inbreeding in rails with consequent loss of genetic diversity. Isolated fragments also prevent escape from predators. Quality habitat for self-sustaining populations of rails includes large parcels of tidal marsh at least 100 hectares (250 acres) in size and a network of first order channels. Because rails are cautious, they do not go far into a marsh. Resources are more abundant in "high quality" habitat, thus rails don't need as much area to fulfill their life-cycles and population densities can increase. Rails call to each other to make contact, to advertise their breeding status, and to defend their territories. Stable populations fare best with large contiguous marshes, healthy stands of marsh vegetation, and a well-developed network of tidal channels at the bay edge. Deep channels generally support dense vegetative cover nearby and a complex of smaller channels with corridors to refugia for periods of extreme high tides. Other important features include protection from predators that includes refugia during extreme high tides, and well-buffered marshes isolated from predators emanating from neighboring developed upland areas. Low quality narrow marshes give better access to fox and other terrestrial predators. Such marshes are not used as nesting areas. Successful control of the red fox and feral cats is essential in the near future for the long-term survival of the California clapper rail.

G.2 Salt Marsh Harvest Mouse

Salt marsh harvest mice are dependent on the thick perennial vegetative cover of salt marshes and only leave the marshes in late spring and summer if the marsh connects with grasslands, and then only when the plants are green and provide good cover. These mice live primarily in the middle of the pickleweed and upper or high marsh zones of marshes; they need the latter zone to escape from high tides. They are cover-dependent animals that swim well but are exposed to aerial predators when they are forced out of vegetation to swim or to cross bare ground. Their usual method of escape from both tides and predators is to seek the dense cover of the less-flooded upper tide zone of marshes or the bushes along channels within marshes. The upper zone of marshes, the high marsh, was once much more abundant in San Francisco Bay, but is now present in most tidal marshes as 1-2 m (3-6 ft.)-worth of mixed halophytes distributed along the steep sides of outboard dikes. The loss of this essential escape cover has resulted in marshes with sizable pickleweed (i.e., middle) zones that lack populations of this endangered mouse. Upper marsh zones have become highly fragmented, thus making it difficult for populations to persist and movements to occur between marshes. Isolation and reduction in size of habitat usually results in extinction.

The salt marsh harvest mouse prefers large stands of lower marsh pickleweed (an important component of tidal marsh vegetation) along with easy access to upper marsh areas for avoidance of high tides and predation. Their usual method of escape is to seek the dense cover of the less-flooded upper tide zone of marshes or the bushes along channels within marshes (PWA 2004). Development of a fuller mosaic of lower elevation vegetation and supra-tidal habitat with the marshes of levees would provide increased habitat.

H SBSPRP Benefit Transfer Details

Table H-1 and Table H-2 detail the selected benefit transfer functions used in estimating per-household and aggregate willingness to pay (WTP) for wetland restoration. We present results based on two studies (Bauer, Cyr, & Swallow, 2004; Brouwer, Langford, Bateman, & Turner, 1999).

Table H-1. Bauer et al. (2005) WTP function and Independent Variable Values used for SBSPRP.

Variable	Description	Coefficient	Baseline	Post -Restoration	Rationale
cost	cost of the mitigation alternative	-0.01893	n/a	n/a	n/a
acre	acres preserved or restored	0.00913	1,513[(A)]	1,513[(B)]	Total acres restored at Alviso, Eden Landing and Bair Islands
boardwalk	public access via boardwalk	1.02814	1	1	Boardwalks exist in baseline and post-restoration.
viewing tower	public access via viewing tower	0.82109	0	0	No viewing towers in South Bay or under SBSPRP (Don Edwards NWR webpage)
endangered species	endangered species present at mitigation site	0.47982	0	1	Relative to salt ponds restored marshes increase habitat for and observed counts of threatened and endangered species.
no action	no-action alternative	-1.03780	1	0	Signifies no policy action taken in the baseline.
preserve	preservation alternative (as opposed to restoration)	-0.17227	0	0	Coastal restoration context implies restoration, opposed to preservation.
no action*female	Interaction: Tailors WTP for no-action scenario to female respondents.	1.03514	0.483	0	Set to percent of survey respondents that were female.
preserve*female	Interaction: Tailors WTP for preservation to female respondents.	0.64152	0	0	Not used; preservation is not relevant to the policy context.
no action*high income	Interaction: Tailors WTP for no-action scenario to high-income (>$60,000) households.	0.69527	0.5	0	Set to 0.5. Median household income in California is about $58,000.
preserve*high income	Interaction: Tailors WTP for preservation to high-income households.	0.58691	0	0	Not used; preservation is not relevant to the policy context.
no action*graduate	Interaction: Tailors WTP for no-action scenario to respondents with a graduate degree.	-0.34986	0.242	0	Set to percent of study respondents with graduate degree.
preserve*graduate	Interaction: Tailors WTP for preservation to respondents with a graduate degree.	-0.57284	0	0	Not used; preservation is not relevant to the policy context.

Notes: (A): 7,500 acres are restored in the overall South Bay project.

Table H-2. Brouwer et al. (1999) WTP function and Independent Variable Values used for SBSPRP.

Variable	Description	Coefficient	Baseline	Post-Restoration	Rationale
Constant	Intercept term	3.311	0	1	Wetlands provide more biodiversity than salt ponds. This term captures base case of dummy variables; for wetland features, the omitted case is biodiversity supply.
Income tax payment vehicle	Dummy: 1 = income tax; 0 = other	1.576	1	1	Based on best practice. Income tax is better than alternatives (Entrance fee/private fund; product prices; or a combination of these)
Open-ended elicitation	Dummy: 1 = open-ended; 0 = other	-0.376	0	0	Based on best practice. Alternatives (dichotomous choice; iterative bidding, or payment cards) are preferred in the literature.
Country is North America	Dummy: 1 = North America; 0 = other	1.629	1	1	San Francisco, CA is in North America.
Response rate 30-50%	Dummy: 1 = 30-50%; 0 = other	-1.722	0	0	Based on best practice. Use values from studies with higher response rates.
Response rate > 50%	Dummy: 1 = >50%; 0 = other	-1.461	1	1	Based on best practice. Use values from studies with higher response rates.
Flood control	Dummy: 1 = flood control; 0 = other	1.134	0	0[(A)]	Set to “0” for the ARRA-funded portion of the SBSPRP; set to “1” for the overall SRSPRP.
Water generation	Dummy: 1 = water generation; 0 = other	0.441	0	0	Neither salt ponds nor coastal wetlands will aid in groundwater generation.
Water quality	Dummy: 1 = water quality; 0 = other	0.659	0	0	Neither salt ponds nor coastal wetlands will aid in water quality provision.

Note: Brouwer et al. (1999) included random effects; we omit them here for brevity because we did not use the covariance matrix in benefit transfer.
(A): We assume SBSPRP will provide some flood mitigation benefits, whereas the smaller ARRA-funded portion does not.

I SBSPRP Flood Risk Reduction Analysis: Additional Details

I.1 Detailed Methods and Data Sources

As our source of floodplain boundaries for the flood risk reduction analysis, we used US Federal Emergency Management Agency (FEMA) 100-year flood plain boundaries. Based on visual appearance, we determined that the majority of floodplain area in South San Francisco Bay is a coastal flood hazard type 'AE,' but that some 100-year flood areas also included fluvial (river) floodplains. Because the SBSPRP will affect both coastal and riverine flooding severity and risk, we included all types of floodplains in our analysis except for AR and A99. As distance from the coastline increases, SBSPRP impacts on flooding will dissipate. However, available information was not sufficient to document an exact distance at which changes in flood impacts reach zero. As a proxy for the upper-bound spatial extent of flood benefits, we considered relevant flood risk zones within 5 km of the outer border of the coastal floodplain. We also limited the analysis to flood zones south of the San Mateo Bridge, since we do not believe flood impacts are likely to extend north of the South Bay (e.g., and not into the main portion of San Francisco Bay, where changes in flooding due to salt pond conversion are likely negligible relative to the role of other environmental conditions).

Then, we used GIS tools to summarize demographics in Census 2010 block groups and tracts that intersected the chosen flood plain extent.

I.2 Converting House Sale Price to Rental Equivalent Value

We converted the change in the sale price of a home (due to reduced flood risk) to the annual rental-equivalent housing value of reduced risk. This approach is used to avoid the issue of homes selling more than once in our analysis period: we use "rental-equivalent" values to represent a typical homeowner's willingness to pay for the amenities of living in his house, with lower flood risk in *a single year*. In this way, we avoid over- or under-counting housing value benefits that are tied to the sale of a house.

Because the price of a house represents the sum of the present discounted value of the flow of amenities from living in that house in all future years, we calculate the change in annual rental-equivalent flood benefits by multiplying the change in housing values due to reduced flood risk by the 3% discount rate. Flood risk reduction benefits are either present or absent, and cannot be scaled as "partial benefits" while restoration trajectory is still incomplete. So, we assume the annual rental benefits begin accruing in 2060, the year in which the overall SBSPRP is completed and flood risk benefits are achieved. Then, because the SBSPRP analysis considers benefits accrued over a finite 100-year benefit period beginning the year construction started (and not through all future years into perpetuity), we sum the annual rental benefits accrued between 2060 and 2110 – a period of 50 years. Lastly annual benefits in at each year in this period are discounted to 2013$ using a 3% discount rate.

J VSBRP Habitat Quality and Area Calculations

J.1 Oyster Habitat Restoration Notes and Calculations

The methods used for assessing reef area often vary by region, reef types, or investigator. The Oyster Metrics Workgroup (2011) developed uniform assessment targets and recommended monitoring intervals to judge restoration success. These metrics included operational goals dealing with substrate/habitat installation and functional goals relating to oyster growth and production.

The Workgroup's operation goals for oyster reef physical restoration were that shell, alternative substrate, or spat-on-shell should cover a minimum of 30% coverage throughout the target reef area. These goals should be evaluated within 6 to 12 months of restoration.

The target functional goals were an oyster population with a mean density exceeding 50 oysters and 50 grams dry weight/m^2 covering at least 30% of the target restoration area at 3 years post restoration activity. The minimum threshold was an oyster population with a mean density of 15 oysters and 15 grams dry weight/m^2, covering at least 30% of the target restoration area at 3 years post-restoration.

Other metrics are that a minimum of two age classes be present at 6 years post restoration and that a neutral or positive shell budget is maintained. Further evaluation at 6 years and beyond was also recommended to judge ongoing restoration success and guide adaptive management (Oyster Metrics Workgroup, 2011).

Based on these general expectations and observations at VSBRP (Hadley, Hodges, Wilber, & Coen, 2010), a timeline of oyster reef production was estimated, with development over 12 years. During the initial period (1 to 2) years after seeding the reef is not producing harvestable oysters, but the physical structure and sheltering aspect of the restored oyster reef is present from the start.

J.2 Eelgrass Habitat Restoration Notes and Calculations

Table J-1 indicates a yearly 72% geometric mean increase (range of 50- 93%) in eelgrass cover (Orth & McGlathery, 2012). These very high rates of annual increase reflect both ideal growth habitat and facilitation of expansion (i.e., "jumpstarting" restoration) by seeding a very large number of plots set in a mosaic pattern within a carefully chosen target plot (e.g., see Figure 8 in Orth et al. 2010). However, this exponential rate of increase is not sustainable as a long-term average. Indication of a declining rate of increase over time is shown by the reduction in growth for the seedling-2006 rates vs. the 2006-2010 rates. Future expansion would likely proceed at a slower pace as interior habitats are filled in, distance for seed dispersal (most important factor for new expansion) to new habitats increases, and less favorable substrate or depth conditions are encountered.

Table J-1. Post-Implementation Vegetation Monitoring and Rate of Eelgrass Growth in VSB Sites.

VSB Restoration Site	Date Initially Seeded	Total Seeded Area (ac)[1]	2006 Cover (ac)[2]	2010 Cover (ac)	Yearly Areal Increase: Seeding - 2006 (%)[3]	Yearly Areal Increase: 2006-2010 (%)
South Bay	2001	38.8	494.2	2,520.5	166%	150%
Cobb Bay	2003	9.6	101.3	862.4	219%	171%
Spider Crab Bay	2004	0.7	4.0	40.3	239%	178%
Hog Island Bay	2007	-	62.3	449.7	-	193%
				Average annual rates of increase[4] :	**206%**	**172%**

Notes:

(1): Seeded area and vegetation monitoring data from Orth et al. 2012b.

(2): Hog Island Bay was not sampled in 2006; reported cover data are from 2007.

(3): Yearly area increases are adjusted for the number of years between samplings.

(4): Geometric mean of annual rates.

Other examples of eelgrass restoration support our view of rapid expansion into available habitat followed by declining growth at more moderate rates as biomass accumulates.

- Eelgrass acreage at nearby Chincoteague Bay, VA (immediately north of the VSB) increased from1990 until early 2000's (as indicated by mapped polygons; Figure J-1) during a natural (i.e., non-anthropogenic) re-vegetation event. The long-term change in coverage over 13 years of expansion yielded an approximate long-term annual rate of increase of 10.5% per year, with a peak rate of 12% during one 5-year period (1995-1999). Growth leveled off after 2002 and began to decline in the last few years, reportedly due to water quality degradation and ecologically destructive shell fishing practices (Orth, Marion, Moore, & Wilcox, 2010).
- In another study, Greening and Janicki (2006) reported gradual increases in turtle grass (*Thalassia testudinum*) coverage in the Tampa Bay, Florida estuary due to improving water quality over a 24-year period (1986-2010). During that period, turtle grass coverage increased from 2,690 to 11,457 hectares in Tampa Bay estuary over a 24-year period with a long-term annual increase of 6.2% per year.

These examples suggest that the high rates of increase from the VSB data (Table J-1) are likely to overestimate long-term expansion. In projecting the future growth through natural processes we assume an annual long-term rate of increase of 15%, slightly higher than that observed in nearby Chincoteague Bay but consistent with the VSB representing prime, but underutilized eelgrass habitat.

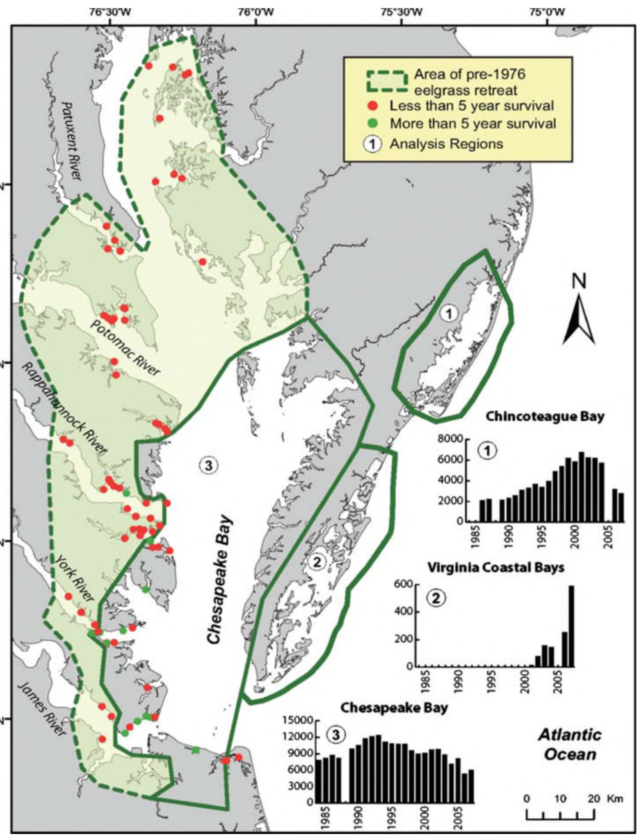

Figure J-1. Map of mid-Atlantic Region Showing Three Analysis Regions and Eelgrass Abundance in Each from 1984 to 2007.

The shaded polygon in the Chesapeake Bay shows the upper extent of eelgrass distribution in the 1960s while the lower, clear polygon represents the current distribution. These regions were derived from all mapped seagrass beds based on field observations of eelgrass and widgeon grass distribution. Red and green dots show eelgrass transplant projects between 1978 and 2006, conducted using a variety of planting and seeding methods.

Source:

Orth et al. (2010). A more detailed figure and table with data from each of the transplant projects can be found at: http://www.vims.edu/bio/sav/online-materials/estuaries-coasts/orth-et-al-2010-eelgrassrestoration transplantprojects.pdf

K VSBRP Recreational Fishing Benefits Transfer Details

Hicks (Personal Communication - 2014) describes several methodological, study site, and recreational fisherman demographic characteristics caveat the benefit transfer from Hicks (2004), who studied Chesapeake Bay, to the Virginia Seaside Bays Restoration Project site on the Atlantic Coast of Virginia's eastern shore. Relevant factors include:

- **Restoration Effectiveness:** Compared to VSBRP, past oyster restoration in Chesapeake Bay has typically been, at most, moderately successful. Because VSBRP restoration has, at least so far, been qualitatively more effective and thus provided larger per-unit fishery benefits, Hicks's original estimates are a lower bound on benefits from VSBRP.
- **Angler Knowledge about Fishing Sites:** Hicks' study used survey data to estimate WTP for increased catch near oyster reefs, moderating the effect of increased catch with anglers' self-reported knowledge that he or she was actually *near* an oyster reef. This knowledge depends in part on the fishers' expertise and in part by visible appearance on the surface of the water. Based on conversations with Dr. Hicks, we believe surface signage locating oyster reefs is similar in the Chesapeake Bay and in VSB: in Chesapeake Bay, signposts mark reefs for navigational purposes; in VSB[3], restored reefs are marked on Seaside water trail maps and are also marked with signs (to note that oysters at the reefs are non- harvestable). In this dimension, Hicks' WTP estimates are likely to roughly approximate values at VSBRP.
- **Baseline Catch Rates:** Incremental catch rate improvements are valued less in areas with higher baseline catch rates than in areas with lower catch rates (Hicks, 2004). If baseline catch rates are higher in VSBRP than in Chesapeake Bay, per-trip benefits will be lower for an equivalent marginal change in catch rates, making Hicks (2004) estimates an upper-bound. If initial catch rates were instead low due to bare sediment, restoration is likely to cause a larger boost in catch rates and thus WTP.
- **Aggregation:** The availability of substitute sites affects WTP for fishing at a given site. There are a lot of substitute fishing sites in the entire Chesapeake Bay, but fewer substitutes considering only the VSB. In terms of site aggregation, the Hicks estimate is a lower bound on WTP for increased catch in the seaside bays.

On net, we believe Hicks' (2004) estimates are a ballpark accurate estimate of WTP for increased catch rates in VSBRP, although slight differences in substitute site availability and restoration effectiveness imply the estimates may very slightly under-estimate benefits.

[3] See: http://www.deq.state.va.us/Programs/CoastalZoneManagement/CZMIssuesInitiatives/SeasideWaterTrail/SeasideWaterTrailWildlifeandHabitatConservat.aspx

L ACR Ecosystem Restoration Site Locations

The general location of oyster reef/breakwater sites constructed in Mobile Bay (Alabama Port, Coffee Islands, are shown in Figure L-1 below. An example of the "living shoreline" constructed at Bon Secour is shown in Figure L-2.

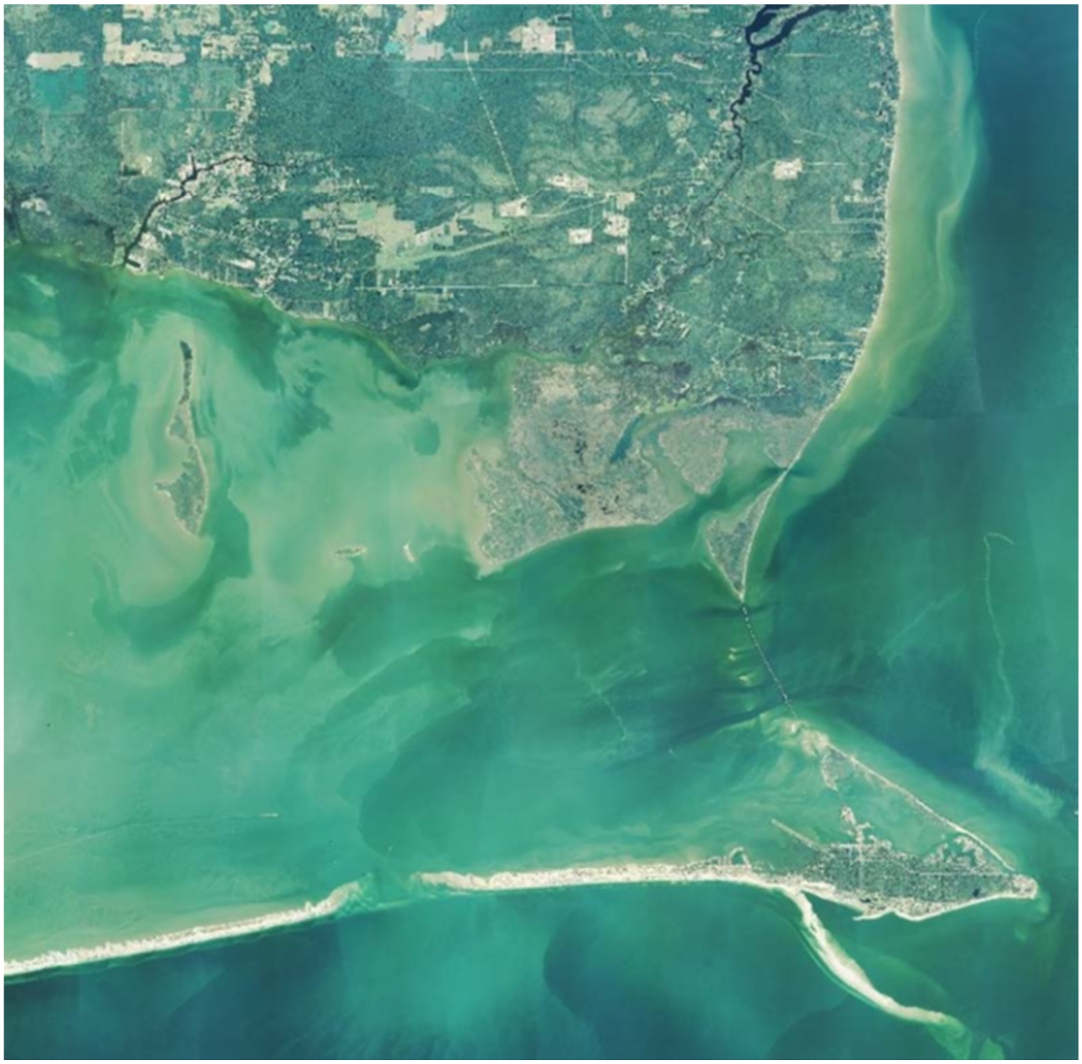

Figure L-1. Location of ARRA-funded reef restoration in Mobile Bay.

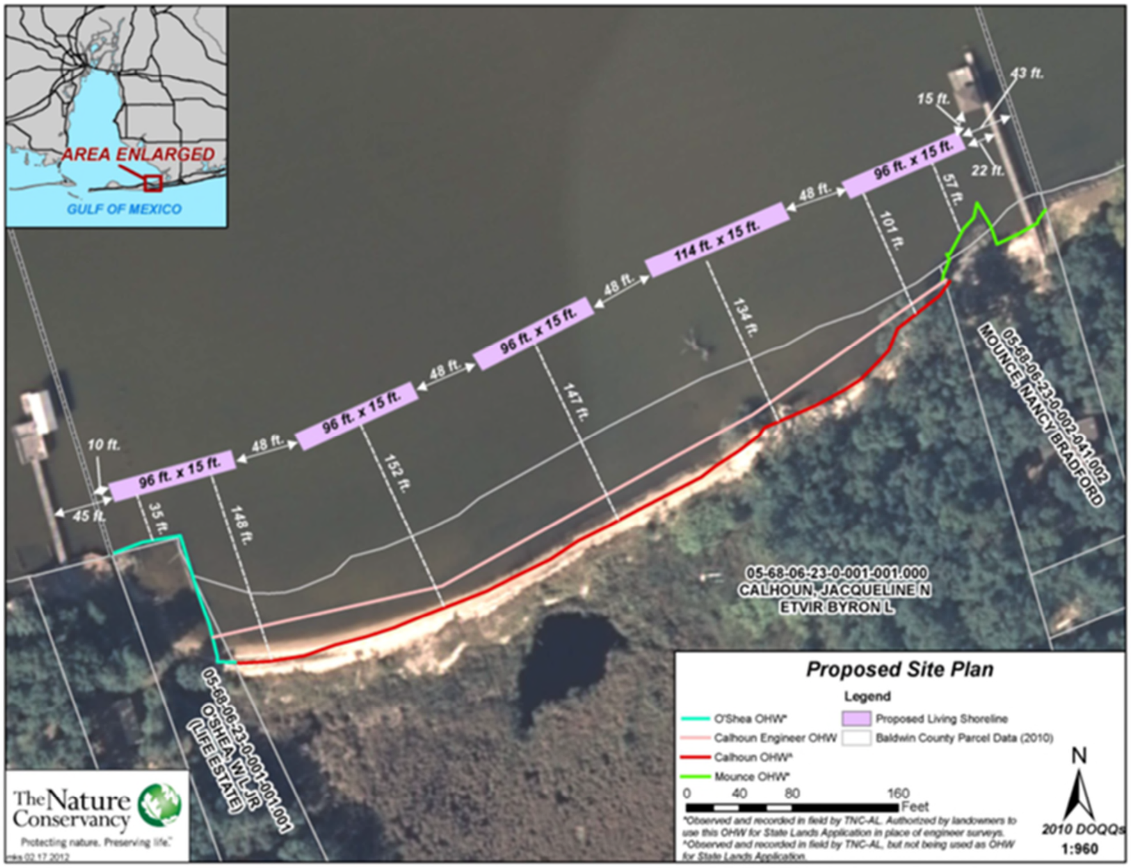

Figure L-2. Example of "living shoreline" application on Bon Secour Bay coastline.

M ACR Oyster Densities

This Appendix reports oyster density monitoring data at Alabama Port and Coffee Island (Figure M-1).

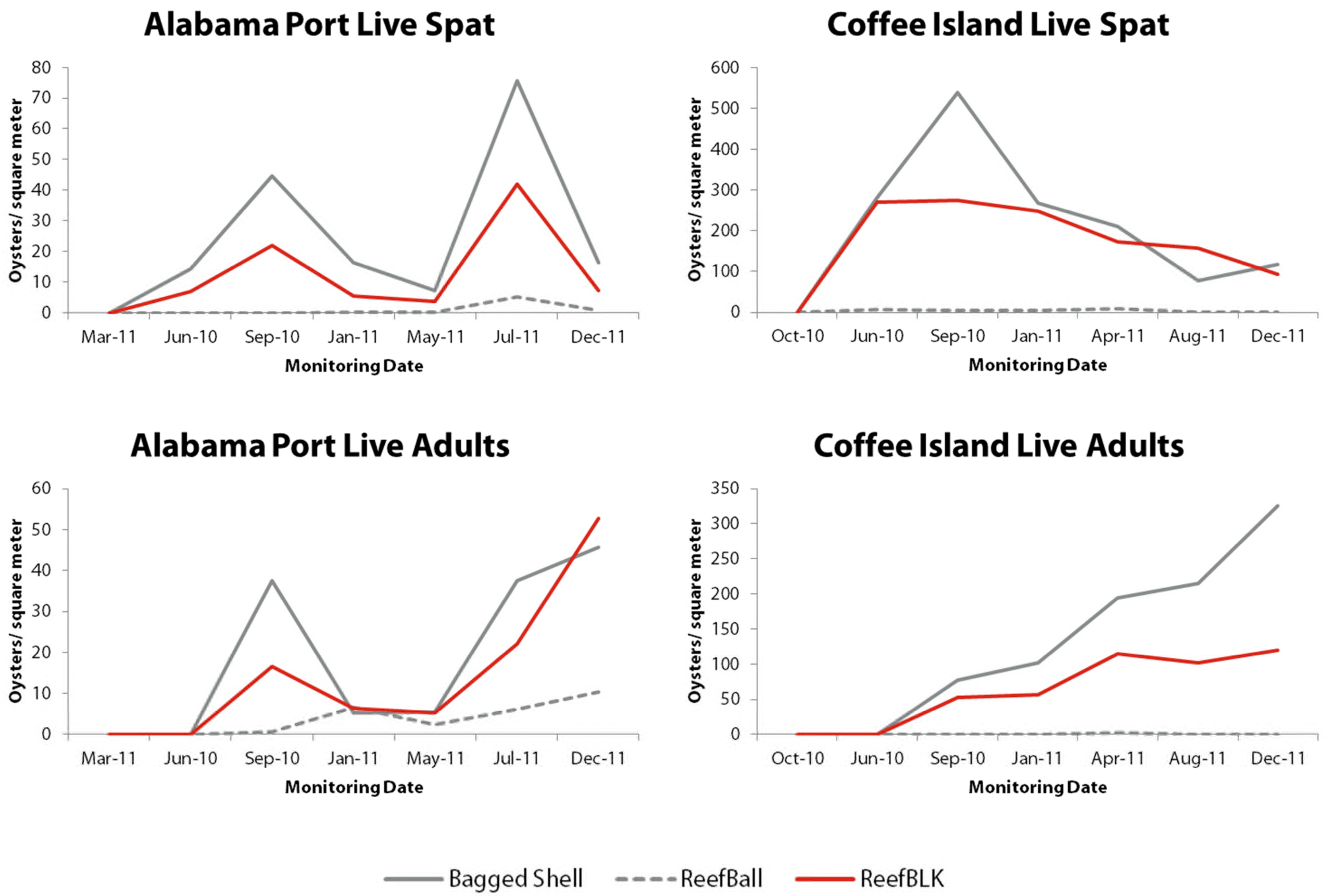

Figure M-1. Oyster densities (oysters/m^2) observed at Alabama Port and Coffee Island breakwater reefs after installation. Reef installation was completed in March 2011 (Alabama Port) and October 2010 (Coffee Island). Source: DeQuattro (2014).

N ACR Fishery Valuation Details

N.1 Commercial Fisheries Production from Oyster Breakwater Structural Habitat

Table N-1 lists additional details of fishery enhancement rates per area of artificial oyster breakwater structure following methods and assumptions in Kroeger (2012) and using commercial fisheries statistics from National Marine Fisheries Service. Per-weight harvest values are assumed constant over the analysis period, although discounting conventions mean that benefits in future years are perceived as less valuable in today's dollars than benefits in current years. Table N-2 lists the net benefit ratios used estimating consumer surplus; the net benefit ratios are used to remove from total landed (ex-vessel) values the typical commercial harvest cost incurred when harvesting additional fish of specific types.

For recreationally-targeted species, Small Game Fish are valued at $6.09/fish and Other Saltwater Fish are valued at $3.01/fish (2013$) (U.S. EPA, 2013). We applied Other Saltwater Fish benefits for crabs.

Table N-1. Fishery enhancement extrapolations from Kroeger (2012) and benefits transfer.

Species	Harvestable Production Enhancement (kg/10 m²/year)	Total Harvestable Enhancement at Site (kg/year)	Percent of Fish Caught Commercially	Commercial (Producer) Surplus per Year (2013$)[A]	Recreational Target Type	Recreational Benefits Per Year (WTP) (2013$)[B]
Atlantic croaker	0.003	3.95	0.09	$0.35	Small Game Fish	$79.25
Black sea bass	0.013	18.19	0.5	$14.74	Small Game Fish	$85.35
Red drum	0.025	34.23	0	-	Small Game Fish	$128.02
Gray snapper	0.033	45.09	0.03	$2.73	Small Game Fish	$426.73
Sand sea trout	0.046	62.05	0.21	$9.74	Small Game Fish	$1,225.33
Spotted sea trout	0.053	72.83	0	-	Small Game Fish	$847.36
Gag grouper	0.053	71.93	0.4	$63.50	Small Game Fish	$73.15
Sheepshead	0.305	415.57	0.34	$165.91	Small Game Fish	$1,469.17
Spottail pinfish	0.001	0.89	0.08	$0.27	Other Saltwater	$15.06
Black drum	0.003	4.64	0.46	$0.88	Other Saltwater	$6.02
Southern flounder	0.015	20.59	0.3	$15.81	Other Saltwater	$57.23
Silver perch	0.020	27.82	0	-	Other Saltwater	$674.65
Blue crab	0.229	312.04	0.8	$309.08	Other Saltwater	$0
Stone crab	0.509	694.64	0.25	$1,238.28	Other Saltwater	$0
Silversides (mullet)	0.002	2.54	0.97	$6.08	Not Targeted	$0
Toadfish	0.003	3.60	0.5	$3.33	Not Targeted	$0
Bay anchovy	0.009	12.18	0	-	Not Targeted	$0
Pig fish	0.018	23.93	0.5	$105.96	Not Targeted	$0
SUM	1.340	1,826.71		$1,918.00		$4,923
					Total Per-year	**$6,841**
					Total Over Time	**$138,737**

Notes follow on next page.

Notes for Table N-1:

(A): Commercial landings values were converted to Commercial Producer's Surplus by multiplying landings values by the species or species group-specific Net Benefit Ratio from US EPA (2013) that best matched each species or group of fish (U.S. EPA, 2013). Net Benefit Ratios ranged from 0.46 – 0.84 (mean 0.61); see Table N-2.

(B): Recreational landings were converted to recreational benefits first by dividing total harvestable enhancement by average weight per recreationally-caught fish in US coastal waters in 2010 (National Marine Fisheries Service, 2010), then by multiplying the resulting number of fish by recreational values of the appropriate target type.

Table N-2. Commercial Fishery Net Benefit Ratios.

Species	Net Benefit Ratio	Source
Atlantic croaker	0.74	Atlantic Croaker, Mid-Atlantic(B)
Black sea bass	0.72	Sea Basses, Gulf Coast
Red drum	n/a	
Gray snapper	0.46	Other Fish, Gulf Coast
Sand sea trout	0.54	Drums, South Atlantic(A),(B)
Spotted sea trout	n/a	
Gag grouper	0.46	Other Fish, Gulf Coast
Sheepshead	0.84	Sheepshead, Gulf Coast
Spottail pinfish	0.46	Other Fish, Gulf Coast
Black drum	0.69	Black Drum, Gulf Coast
Southern flounder	0.65	Flounders, Mid-Atlantic(B)
Silver perch	n/a	
Blue crab	0.72	Blue Crab, Gulf Coast
Stone crab	0.71	Stone Crab, Gulf Coast
Silversides (mullet)	0.79	Striped Mullet, Gulf Coast
Toadfish	0.46	Other Fish, Gulf Coast
Bay anchovy	0.46	Other Fish, Gulf Coast
Pig fish	0.46	Other Fish, Gulf Coast

Notes:

(A): Sand sea trout are similar to drums.

(B): No data were available for the Gulf Coast, so we applied data from the most proximate region.

Source: Benefits Analysis for the Final Section 316(b) Existing Facilities Rule EPA 820-R-13-003. July 26, 2013Table 6-7: Gulf of Mexico Region, Species-Specific Gear Type, Status of Stock, and Net Benefits Ratio (p. 6-9)

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