Year Five of the Giacomini Wetland Restoration Project: Analysis of Changes in Physical and Ecological Conditions in the Project Area

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Executive Summary

In 2007-2008, the National Park Service (Park Service) and Point Reyes National Seashore Association (PRNSA) implemented an approximately 613-acre wetland restoration project in the southern end of Tomales Bay in Marin County, California. The Giacomini Wetlands Restoration Project (restoration project) is within the Golden Gate National Recreation Area, but is managed by Point Reyes National Seashore (Seashore). The restoration project principally focused on conversion of a former dairy ranch into tidal wetlands, as this area was once historically. However, rather than try to recreate historic conditions, the Park Service focused on restoring natural hydrologic tidal and freshwater processes, thereby promoting restoration of hydrologic and ecological functions. Natural hydrologic processes are the cornerstone of many hydrologic and ecological functions and economic “services” associated with wetlands such as floodwater retention, flood energy dissipation, water quality improvement, and wildlife habitat that benefit both wildlife and humans. These hydrologic and ecological functions are particularly important in Tomales Bay. While it is generally perceived as pristine, this rural coastal watershed still suffers from negative anthropogenic influences such as agriculture, home and road development, leaking septic systems, mercury mining, landfills, and oil spills.

The project objective of establishing processes and functions in the restored Giacomini Wetlands Restoration Project Area (Project Area) similar to those occurring in reference wetlands meets one of the requirements imposed by the California Coastal Commission (CCC) in its mitigation agreement with California Department of Transportation and the Park Service. In this agreement, the CCC stipulated that mitigation requirements would be satisfied if “density of flora and fauna” in the Project Area after restoration is “comparable (to) that in surrounding or nearby habitat areas of the same type.”

As an integral component of the restoration project, the Park Service has implemented a comprehensive long-term monitoring program to assess whether restoration is successful. To facilitate analysis of restoration progress, the Long-Term Monitoring Program relies on a modified BACI (“Before-After, Control-Impact”) sampling framework. The Giacomini Wetlands and another restoration site, Olema Marsh, represent the Impact Area, while Reference or Control Areas were established in natural tidal marshes in Tomales Bay and adjacent watersheds to facilitate a comparative evaluation of changes in the restored system relative to ambient conditions. In addition, for certain variables, sampling was also conducted on the upstream perimeter of the Project Area (Upstream Areas) to more clearly understand the quality of inflow waters to the Project Area. Monitoring of these areas occurred from two - to five years before and four years after restoration, as well as two years in between these periods when cows were removed, but levees were still present (Passive Restoration).

This monitoring program incorporates a number of variables representative of conditions within the restored wetland, including hydrology, topography, sedimentation, water quality, vegetation, invertebrate and fish communities, and avian communities. The proposed monitoring program will meet the five (5) years of post-project monitoring required by the CCC. The Park Service is also required to conduct five years of monitoring as part of its permitting requirements of the U.S. Army Corps of Engineers (Corps). This report presents results for five years of post-restoration monitoring, with the exception of with the exception of some of the variables that have only been monitored or analyzed through Year 3 or 4, including hydrology, topography, zooplankton, benthic invertebrate, fish, and spring birds.

One of the most dramatic changes immediately after restoration was the incredible amount of water that began flowing across what was once dairy pasture. The volume of water far exceeded what would have been predicted under fully evolved or mature marsh conditions, because the restoration project design emphasized an evolving system: only a few new tidal channels were created, and these were undersized to allow the channel to evolve naturally in response to tidal and creek flow. In addition, the lack of vegetation on the marshplain following death of the pasture vegetation and the remaining natural levees along Lagunitas Creek also acted to slow down draining of water during low tides. These factors led to approximately 109.4 acres of the East Pasture being permanently ponded during even very low tides in the first year after levees were breached: This is more than three times what was predicted to remain subtidal under fully evolved conditions (26.5 acres).

However, during the last five years, waters do appear to be draining more quickly from the restored marshplains under low tide conditions (KHE 2009 and 2011b). Indeed, the extent of permanent inundation on marshplains dropped by more than half to 51.0 acres between 2009 and 2011 in the East Pasture. Some of these changes may result from the fact that several of the created tidal channels have deepened significantly near their outlets, thereby conveying more water both in and out of the marsh (KHE 2011a). New tidal channels are also being naturally created. Evidence of channel scouring can be seen in the shoals that have developed just outside the mouth of these creeks in Lagunitas Creek. The system will continue to evolve in future years, although dramatic changes may not occur until large-scale flooding events can re-work the landscape. Since the levees were fully breached in fall 2008, the creek banks have been only overtopped once by flood flows due to lower than average to average rainfall volume. Floods can not only reshape the landscape through erosive scouring, but through sedimentation or deposition of sediment on marshplains. Despite the lack of floods, sedimentation does appear to be taking place, probably due to sediment being carried in by tides and smaller flood flows. This factor may prove quite important in the future to help counter the "drowning" effects of sea level rise associated with climate change.

With a few exceptions, water quality parameters have shown significant, positive improvements in conditions between the dairy ranch and full restoration periods. Dissolved oxygen levels increased 16%, while nitrate, ammonia, phosphate, phosphorous, and fecal coliform levels decreased at least 23%, with some of these parameters falling quite substantially (phosphates; 900%). Nitrate, ammonia, fecal coliform, and, to a lesser extent, phosphate levels were expected to drop with removal of cows and discontinuation of manure spreading. Somewhat expectedly, turbidity and loading or conveyance of nitrates and fecal coliform in streamflow have increased since restoration, although the former was not as dramatic as was anticipated (44%). Loading was certain to increase, given that the ranch was largely diked before and not contributing fully to pollutant loading to Lagunitas Creek except during extreme flood stages. Indeed, some of the parameters -- including temperature, ammonia, phosphates, phosphorous, and nitrates -- appear to be at equivalent levels to those in Reference Areas.

Ultimately, restoration is expected to improve water quality conditions not only in the former dairy ranch, but in the Tomales Bay watershed due to trapping of pollutants on the now hydrologically reconnected floodplains. Over time, as the marshplains become more vegetated and hydrologic re-working of the marsh slows down, the wetland may play a more active role in improving downstream water quality in Tomales Bay.

The EIS/EIR prepared for this restoration project originally predicted that 10- to 20 years would elapse before the restored Project Area began to resemble the salt marsh vegetation communities present in adjacent reference or natural tidal marshes in Tomales Bay and Limantour (NPS 2007). However, this transition from managed pasture to salt marsh and other salt-influenced communities occurred incredibly more rapidly than anyone expected. Pasture grasses rapidly died in response to increased inundation and salinization of soils, leaving large expanses of bareground with either thatch or sparse vegetation, particularly at lower elevations. By 2010, the areal extent of Wet Pasture – or grassland typically managed with irrigation – had plummeted by approximately 212 acres, while Sparsely Vegetated Mudflat/Panne had jumped by approximately 139 acres. This mudflat/panne condition has persisted largely due to the fact that marsh channels have yet to naturally evolve to the point to allow tidewaters to completely flow off the marshplains during low tides. At mid- to high marsh elevations, Tidal Salt Marsh and Tidal Brackish Marsh communities now dominate, increasing by approximately 90 acres by 2010. Cover of non-native species dropped dramatically after restoration, with most of the approximately 30% decline coming from loss in non-native grasses. However, non-native cover still remains considerably higher than natural marshes due to the continued persistence of brackish species such as bentgrass (*Agrostis stolonifera*) and fathen (*Atriplex triangularis*). The more brackish nature of the restored ranch relative to the reference marshes has led to lower soil and water salinities and a greater extent of Tidal Brackish Marsh and brackish species. Despite this, the similarity of the Project Area to natural tidal marshes in vegetation species composition has increased considerably since levees were breached, climbing from 28% pre-restoration to 63% in 2013, with even higher levels (72%) present at mid/high marsh elevations. Species richness and diversity in Giacomini actually decreased slightly in Year One, but has increased almost every year since then.

Tidal marsh mitigation and restoration projects are often accompanied by very quick changes in the numbers and types of fish using restored tidal wetlands, with much slower changes in the invertebrate community, particularly those that burrow in the mud. However, results from the first four years of monitoring after restoration of the Giacomini Wetlands shows a slightly different pattern. Prior to restoration, fish, zooplankton, and benthic invertebrate communities in the Giacomini Wetlands differed considerably from those in natural marshes.

With restoration, both the zooplankton and benthic invertebrate communities responded strongly, shifting species assemblage to the point that some convergence with communities in natural marshes was apparent. Density of benthic invertebrates, which had quite low during ranching management, also climbed considerably, closely approximating numbers found in most of the undiked areas prior to restoration. Zooplankton densities actually declined in the Project Area after restoration, although median densities were equivalent with lower abundances reported in natural marshes. Interestingly, shifts in species composition and increases in benthic invertebrate numbers did not necessarily result from heavy colonization by opportunistic, often non-native polychaete species that have been found in other newly restored systems. Primary benthic invertebrate taxa included oligochaetes and amphipods, the latter of which are associated with less polluted or disturbed systems.

Fish communities appeared to respond the least to restoration, with multivariate statistical analyses showing very little change between pre- and post-restoration species assemblages. What did change, however, was that the number of non-native fish species dropped abruptly following removal of the levees.

Waterbird numbers have increased every year since restoration, particularly for dabbling ducks, climbing from 5,552 observations in Year 1 to 45,022 observations in Year 5 (ARA 2013). However, shorebird numbers have been more variable. Numbers were low in the first winter after restoration, moderate in the second year and first half of the third year, and then declined sharply in the latter half of the third year, rebounding in Years 4 and 5 to approximately Year 2 levels (ARA 2012, 2013). Giacomini is still evolving as potential shorebird habitat, with this evolution strongly dependent on invertebrate prey bases in marsh muds. In the early years, the restored wetland primarily attracted species that forage in the water column, on the ground, and in shallow sediments (J. Evens, ARA, *pers. comm*.), however, the increase in numbers since fall 2011 of Marbled godwits and Dowitchers, which forage more deeply in muds, may be yet another indicator that numbers of benthic invertebrates in the restored wetlands are increasing.

As with waterbirds, densities of spring breeding birds increased following restoration, particularly for some of the wetland-dependent passerines. Overall, species richness and the total number of observations were actually highest in Year 2 or 2010 than in Year 1, 3, or 4 (ARA 2012). The newly restored wetlands supported 65 breeding bird species in 2009, 74 in 2010, and 48 in 2011 (all detections; ARA 2011a). In general, Song sparrows were the most common breeding bird species, followed by Savannah sparrows, Marsh wren, Red-winged blackbird, and Common yellowthroat. Two species with relatively strict habitat requirements – “Salt-marsh” common yellowthroat and “Bryant’s” savannah sparrow – showed some of the highest increases in Year 4 since restoration. Bald eagles are occasionally sighted at the restoration site foraging, and California black rails appear to have expanded their range from the Undiked Marsh to the north into the restored wetland (ARA 2011b).

Ultimately, habitats and species assemblages are evolving within the restored wetlands, and, in many cases, appear to be drawing closer to conditions and communities present in natural marshes within Tomales Bay and adjacent watersheds. This evolution appears to be occurring on a much more rapid timeframe than originally anticipated. However, while restoration projects are often predicated on the so-called Gradual Continuum Model of ecosystem evolution that assumes a linear model of change from the disturbed condition (diked pasture) to a more natural state (tidal marsh), this successional trajectory does not always hold true. Since breaching of the levees, it is apparent that the transition to tidal marsh may not necessarily follow a linear trajectory, but rather occur in a more discrete, step-like fashion, often with abrupt transitions between different states or conditions that require certain “thresholds” to be passed for movement from one state or condition to the next. In the case of Giacomini, some of these thresholds directly relate to how quickly created channels will evolve and natural channels will form, which would affect the extent of permanent inundation in the restored wetlands.

This change in our ecosystem evolution model has distinct implications for evolution of the system. What this means is that the evolution of conditions and species assemblages that we have witnessed so far may never necessarily reach a steady-state condition characteristic seemingly of natural marshes, but may continue to evolve over time in a discrete or stage-type fashion in response to internal and external factors such as tidal channel evolution or climate change-related factors such as sea level rise.

Ultimately, this restoration project will provide an invaluable insight into the issues that affect the path or trajectory of restoration of degraded wetlands back to non-degraded conditions – insights that will prove valuable for planning of future restoration projects. In addition, it will help ecologists to better understand the processes of change for all communities, whether they be part of a restoration project or not.

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1 Introduction

In 2007-2008, the National Park Service (Park Service) and Point Reyes National Seashore Association (PRNSA) implemented an approximately 613-acre wetland restoration project in the southern end of Tomales Bay in Marin County, California. The Giacomini Wetlands Restoration Project (restoration project) is within the Golden Gate National Recreation Area, but is managed by Point Reyes National Seashore (Seashore). The restoration project principally focused on conversion of a former dairy ranch into tidal wetlands, as this area was once historically. However, rather than try to recreate historic conditions, the Park Service focused on restoring natural hydrologic tidal and freshwater processes, thereby promoting restoration of hydrologic and ecological functions. Natural hydrologic processes are the cornerstone of many hydrologic and ecological functions and economic “services” associated with wetlands such as floodwater retention, flood energy dissipation, water quality improvement, and wildlife habitat that benefit both wildlife and humans. These hydrologic and ecological functions are particularly important in Tomales Bay. While it is generally perceived as pristine, this rural coastal watershed still suffers from negative anthropogenic influences such as agriculture, home and road development, leaking septic systems, mercury mining, landfills, and oil spills.

Restoration began in 2006. From 2000 (date of sale of land to the Park Service) to 2006, the Giacominis continued to operate a full-scale dairy operation under a Reservation of Use Agreement. There were at least three dairy herds, and the ranch was actively maintained through manure spreading, haying, and flood and spray irrigation of certain pastures in the summer. This period is referred to in data analyses as Pre-Restoration as it pre-dates any restoration efforts. In 2006, the Giacominis sold the dairy string and instead grazed a much smaller herd of dairy heifers. Maintenance activities were also scaled back, with reduced haying, manure spreading, and irrigation of pastures during the summer. Because most of the restoration achieved during this period probably resulted from passive measures such as discontinuation or scaling back of active dairying and ranch management, the 2006-2008 period is sometimes referred to as Passive Restoration, because removal of agricultural management potentially could have led to some improvement or “restoration” of water quality conditions within the ranch, even without active restoration.

In 2007, the first phase of active restoration of the Giacomini Ranch was implemented. However, as most of this restoration focused on removal of dairy barns and other infrastructure and agricultural conditions and did not substantially alter hydrologic conditions, the ecological changes arising from this phase were comparatively small. The second and more intensive phase of restoration commenced in July 2008 and was completed with the final levee breach in October 2008. This phase involved full-scale levee removal, construction of new tidal channels, realignment of leveed channels, and removal of drainage ditches, although, due to the need to maintain dry working conditions, final hydrologic reconnection with Lagunitas Creek and other streams did not occur until the final levee breach in late October 2008. In addition, some hydrologic improvements occurred in the adjacent Olema Marsh, with lowering of a small berm that constrained outflow of this system to Lagunitas Creek.

As an integral component of the restoration project, the Park Service has implemented a comprehensive long-term monitoring program to assess whether restoration is successful. The monitoring program includes assessment of both the Giacomini Wetlands Restoration Project Area (Project Area) and nearby reference wetlands. This framework will enable us to better determine whether restoration has improved condition and increased functionality of the restored Project Area relative to conditions and functionality present in the Project Area prior to restoration and whether these efforts have brought the Project Area closer to supporting conditions and functionality similar to those in existing natural undiked marshes or reference wetlands. The Park Service anticipates that restoration will either reintroduce processes and functions that were lost through diking or enhance functions that are already present due to the fact that the pastures are largely already “wetland.” The monitoring program will enable the Park Service to evaluate how successful removing, modifying, or minimizing infrastructure and agricultural practices have been in reintroducing or enhancing wetland processes and functions.

The objective of establishing processes and functions in the restored Project Area similar to those occurring in reference wetlands also meets one of the requirements imposed by the California Coastal

Commission (CCC) in its mitigation agreement with California Department of Transportation (CalTrans) and the Park Service. In this agreement, the CCC stipulated that mitigation requirements would be satisfied if "density of flora and fauna" in the Project Area after restoration is "comparable (to) that in surrounding or nearby habitat areas of the same type." The proposed monitoring program will meet the five years of post-project monitoring required by the CCC. The Park Service is also required to conduct five years of monitoring as part of its permitting requirements of the U.S. Army Corps of Engineers (Corps). This report presents results for five years of post-restoration monitoring, with the exception of some variables that have only been monitored or analyzed through Years 3 or 4, including hydrology, topography, zooplankton, benthic invertebrate, fish, and spring birds.

In 2007, the Park Service issued an Environmental Impact Statement/Environmental Impact Report (EIS/EIR) that analyzed short-term and long-term changes anticipated to occur in the former Giacomini dairy ranch with restoration (NPS 2007). In general, conditions and functionality of the wetlands present were expected to improve dramatically over the long-term with conversion of the ranch to tidal wetland, even if the actual extent or acreage of wetlands would only change minimally due to the fact that diking had reduced drainage of waters and encouraged establishment of freshwater wetlands. The rate of change predicted varied considerably depending on the parameter: for example, conversion of pastures to a salt marsh vegetation community more characteristic of natural marshes was estimated by the EIS/EIR to possibly require 10-20 years (NPS 2007). Other measures of wetland condition were expected to change rapidly within the first five years, with the rate of change leveling off in subsequent years.

In addition, the dramatic disturbance associated with restoration can also result in short-term negative impacts, even if, over the long-term, conditions and functionality are expected to improve (NPS 2007). For instance, initial pulses in sediment and nutrients might occur from re-working of exposed soils by tides, floods, and decomposition and mineralization of pasture vegetation, with variables such as pH and dissolved oxygen (D.O.) responding accordingly to the resulting flux in nutrients. With breaching of the levees and reintroduction of saltwater, large-scale die-off of glycophytic or non-salt-tolerant pasture grasses and herbs would create a transitional landscape of barren marshplains with high amounts and detritus and minimal cover of establishing salt marsh plant species.

Over the long term, ecological conditions and wetland functionality are expected to improve not only in the Project Area, but potentially within the watershed itself. More than 66 percent of the inflow to Tomales Bay comes from the Lagunitas Creek watershed (Fischer *et al.* 1996), and the creek flows directly through the Project Area. Previously, levees funneled flood flows and associated pollutant discharge directly to Tomales Bay, but with removal of the levees, the creek is now reconnected to its historic floodplain. Therefore, this restoration project could have watershed-scale benefits to water quality and to the flora and fauna that inhabit the estuary.

Information from this monitoring will enable the Park Service to measure the success of its efforts in restoring or improving hydrologic and ecological processes and functions and, thereby, help the Park Service determine whether the Project purpose and objectives have been achieved. It will also help Park Service managers recognize when adaptive management or remedial measures might be necessary to improve the success of restoration efforts. Lastly, we believe that lessons learned from this restoration project through will prove invaluable to managers of other future wetland restoration projects.

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2 Summary of Monitoring Approach

This technical memorandum summarizes changes in physical and ecological conditions and wetland functionality within the Project Area during Year Five of the monitoring program.

As an integral component of the restoration project, the Park Service has implemented a comprehensive long-term monitoring program to assess whether restoration is successful. Similar to many other restoration projects, our monitoring program relies on a modified BACI sampling approach (Stewart-Oaten *et al.* 1986, Underwood 1991). “BACI” refers to monitoring of an “impact” (I) area both “before” (B) and “after” (A) an activity is implemented, with concurrent monitoring of “control” (C) areas. Based on this sampling design, we are evaluating the Project Area before and after restoration is implemented and are using three (3) reference wetlands to better discern the effects of the restoration “impact” relative to natural variability.

Consistent with the structure of the modified or asymmetrical BACI sampling design (Smith 2002), we have conducted monitoring both in the Project Area (impact) and in selected reference wetlands (control; Figure 1). The current channel course of Lagunitas Creek naturally bisects the Giacomini Ranch portion of the Project Area (PA) into two different areas or sections, called the East and West Pastures: the East Pasture (EP) borders the town of Point Reyes Station, and the West Pasture (WP) is adjacent to Inverness Park. Because pre-restoration site conditions and restoration activities implemented in the East and West Pasture differed, the Project Area was split into two replicate “impact” sites for the purposes of the sampling design. Olema Marsh (OM) represents a third “impact” site. In addition, for some of the more aquatic-based variables, sampling was also conducted on the upstream perimeter of the Project Area (Upstream Areas) to more clearly understand the quality of waters flowing into the Project Area. The leveed portions of Lagunitas Creek (LAG) and Tomasini Creek (TOM) were also assessed separately in some instances, as these were largely hydrologically isolated from the dairy ranch prior to restoration.

Reference Areas or natural marshes monitored concurrently included the Undiked Marsh (UM) directly north of Giacomini, Walker Creek Marsh (WCM) in Tomales Bay, and Limantour Marsh (LIM) in Estero de Limantour. These natural marshes do not necessarily represent “reference standard wetlands.” Reference standard wetlands is the term developed by the Corps for its Hydrogeomorphic Method (HGM) to describe the highest performing wetlands of a particular type within the region, with the understanding that few areas in the United States have wetlands that could be described as “pristine.”

While Tomales Bay is often viewed as “pristine,” there are really no pristine wetlands within the watershed. A substantial percentage of the existing or former coastal wetlands developed as a result of dramatically increased sedimentation rates within the Bay during the period 1860-1950. Many of these, particularly along the eastern and southern portions of the Bay, have been diked or otherwise impacted by construction of roads, railroads, and berms. Those wetlands that have not been diked may

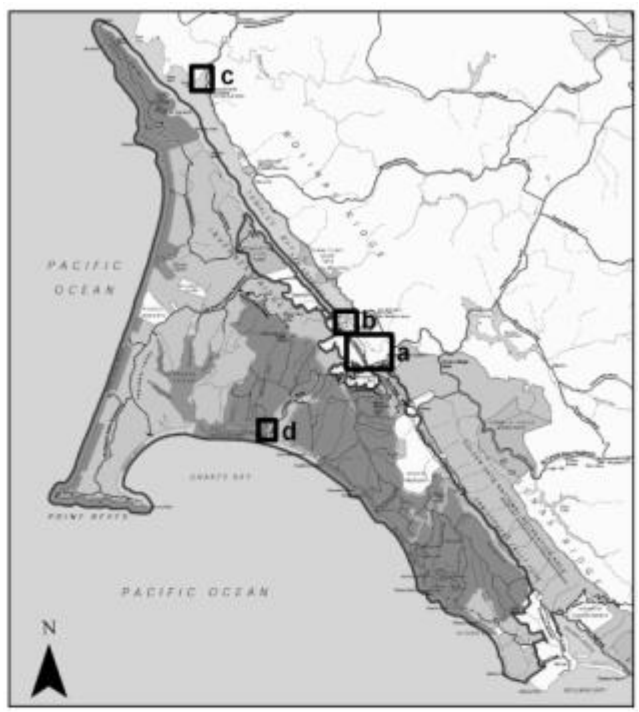

Figure 1. Study Areas
Location of (a) Giacomini Wetland Restoration Project Area (PA/ Giac), (b) Lagunitas Creek Undiked Salt Marsh (UM), (c) Walker Creek Marsh (WCM), and (d) Limantour Marsh (LIM) within the Point Reyes National Seashore

be negatively affected, as well, by impairments in watershed water quality. There are also few sites within Tomales Bay that are relatively similar to the Project Area not only in watershed type, size, and land use (i.e., agriculture, presence of dams, etc.), but site history, as well.

Two marsh systems that are similar are the Undiked Marsh and Walker Creek Marsh (Allen 2005 *in* Parsons 2005). Concurrently or slightly after the Giacomini Dairy lands were formed by excessive sedimentation from poor land use practices during the late 19th–early 20th centuries, the Lagunitas Creek delta continued to extend northwards into Tomales Bay, forming at least another 100- to 200 acres of marsh that were never diked. This same trend of excessive sedimentation also resulted in the formation around the turn of the 20th century of a deltaic marsh at the mouth of Walker Creek in Tomales Bay that has also been only partially diked. Lagunitas Creek is Tomales Bay's largest watershed, while Walker Creek is the second largest. The third potential reference site is also a young marsh – Limantour Marsh – but it is not in Tomales Bay and has a much smaller watershed and a slightly different formation and land use history. There are some issues with using these areas as reference sites. For example, these marshes may themselves change as a result of restoration: the Undiked Marsh may be affected by the restoration project due to its proximity just downstream of the former dairy ranch, and conditions within Limantour Marsh could be strongly influenced by another wetland restoration project that was implemented just upstream of the marsh in 2008. In addition, while it looks pristine, Walker Creek Marsh is believed to have been impacted by mercury from an upstream mine that was improperly capped following closure, which allowed mercury-laden sediment to wash downstream and increase mercury levels in this portion of Tomales Bay (Ridolfi *et al.* 2010).

The Park Service is focusing monitoring efforts on those hydrologic and ecological processes and functions that are expected to be either improved or reintroduced through restoration. These key processes and functions will be assessed either by directly measuring some variable or realized component of function (e.g., wildlife density for wildlife habitat) or by using indicators that relate to the capacity or opportunity for a function to occur (e.g., measuring floodplain width rather than total water storage for floodwater retention). In instances where assessing function is too difficult, specifically water quality improvement, we will focus on functional potential and establishment of optimal ecological or water quality conditions similar to those present in reference wetlands. Monitoring of these variables and indicators incorporate both field- and office-based components, such as mapping, field surveys, sample collection and analysis, and aerial image and map interpretation using Geographic Information System (GIS) software.

The decision on which variables and indicators should be included was largely guided by regional and Park Service estuarine monitoring programs and HGM guidebooks for assessing functions of tidal fringe and riverine wetlands. Most of the variables incorporated represent traditional components of estuarine monitoring programs such as vegetative cover, density of avian species, etc. To maximize comparability, we looked for opportunities to incorporate variables or indicators and monitoring methodologies that are widely used by other local, regional, statewide, or Park Service monitoring programs. Our intention with this monitoring program was to develop a core suite of variables or indicators that are 1) strongly related to key processes, functions, and conditions; 2) considered high priority because of the expected changes with restoration or other reasons; 3) expected to show distinct and clear patterns of change with restoration (either decreases or increases); 4) not duplicative with other variables or indicators; and 5) relatively simple to measure.

Using the criteria listed above, we eliminated variables and indicators that potentially would show no distinct and clear patterns of change with restoration or that were logistically too complex, expensive, or infeasible to monitor. For example, sediment transport, soil oxygen, and contaminants were considered too complex or expensive to monitor for the level of effort proposed. Some variables and indicators that are not expected to show clear and distinct patterns of change with restoration, such as soil salinity and seasonal patterns in water table depth, were also not included in the evaluation of project success. For example, soil salinity will change following restoration, but, as a complex mix of freshwater, brackish, and tidal habitats are expected to develop, restoration is not necessarily expected to result in a net increase or decrease in soil salinities, nor is soil salinity strongly correlated with any particular hydrologic or ecological function or condition.

Intra-annual and inter-annual monitoring frequency varies depending on the variable or indicator, with some assessed several times annually, and others, only once. Overall, inter-annual monitoring has occurred annually prior to restoration and at Years 1, 2, 3, 4, and 5 after restoration is implemented. Monitoring in future years such as Years 7, 10, 15, and 20, which were recommended in the Monitoring Framework document (Parsons 2005), will depend on whether funding can be obtained. For analysis purposes, the monitoring timeframe has been broken into three separate periods: Pre-Restoration (2002-fall 2006), Passive Restoration (fall 2006-fall 2008), and Full Restoration (fall 2008-2013). As discussed earlier, Passive Restoration refers to the period when cows were either lower in number or removed entirely, and many of the agricultural management practices had been discontinued, but the levees had not been breached. In some instances, for certain types of data, analyses have been collapsed into Pre-Restoration (2002-2007/2008) and Post-Restoration (2008/2009 – 2013).

Through statistical analysis of the data, the Park Service will assess whether key hydrologic and ecological processes, functions, and conditions of the restored Project Area 1) exceed those of the Project Area prior to restoration and 2) begin to approach, over time, those of nearby reference marshes, given the potential for some natural range of variation in functionality, even among unimpacted wetlands.

Preliminary reports or memoranda have been prepared annually during Years 1-3 for water quality and avian monitoring and in Year 4 for a broader spectrum of variables. This report constitutes the final report and will focus on a summary of results through Year 5 for variables associated with the following monitoring focal areas: water quality, sedimentation, vegetation, and waterbirds. Results for zooplankton, benthic invertebrates, fish, and spring birds are summarized through Year 4, as hydrologic, topographic, benthic invertebrate, and spring bird monitoring was not conducted in Year 5, and analysis of Year 5 data for zooplankton and fish communities has not been completed. The final report will include at least a qualitative assessment of the overall success of the restoration project in improving hydrologic and ecological processes, functions, and conditions. The Monitoring Framework (Parsons 2005) set forth some preliminary objectives for the 5- and 10-year post-restoration periods. These included that 70 percent of the variables assessed that were intended to represent key processes, functions, and conditions would show improvement in the Project Area relative to pre-restoration conditions (Parsons 2005). Convergence with processes, functions, and conditions present in natural or reference marshes was not expected to occur as rapidly, with only 40 percent of those variables expected to be comparable to reference systems by Year 5, although that number was expected to climb to 80 percent by Year 10 (Parsons 2005).

BACI can be implemented either with or without subsampling (Smith 2002). However, for most variables, some type of subsampling will be necessary due to logistical constraints, with further subsampling being conducted within subsampling areas based on a stratified random approach that varies depending on the variable or indicator or class of variable or indicator (e.g., hydrology, vegetation) being assessed. Because of the hydrologic complexity of wetlands in Tomales Bay, which are influenced not only by large creeks and small drainages, as well as numerous seeps along the upland periphery, we elected to subdivide the Study Areas based on fresh to saline transitions along major creeks, small drainages, and major groundwater interfaces, using drainage divides or subwatersheds, when possible, to create the boundaries. Sampling stations and transects were located using a random stratified approach based on these hydrologic subunits. Many variables such as water quality, zooplankton, and, to a degree, benthic invertebrates were sampled at the same station, although not always concurrently. Fish sampling stations also overlapped with some of the water quality ones.

Whenever possible, original sampling stations have been retained throughout the monitoring program, with some simply renamed to reflect changed status after restoration. However, some sampling stations were eliminated as part of the restoration process, including some of the ditch sites in the East Pasture. In these cases, new sampling stations were chosen as close as physically possible to the former station. Also, as the restoration resulted in more tidal channels and a freshwater marsh being constructed in the East Pasture, additional stations were randomly selected to increase coverage of these new wetland features.

A complete description of sampling methodologies and statistical analysis techniques are available in Appendix A of this report.

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3 Results and Discussion: Changes in Climatic Patterns

Overall, a comparison of local precipitation records showed that the pre- and post-restoration periods were pretty equivalent in terms of rainfall, although the post-restoration period (fall 2008 to fall 2013) started slightly wetter than most of the pre-restoration period (fall 2004 to fall 2008). Both sampling periods had, on average, higher than normal rainfall, ranging from 0.70 inches above normal (Pre-Restoration) to 1.0 inches above normal (Post-Restoration; Western Regional Climate Center (WERC), Olema Valley).

The first year after restoration was actually slightly drier than average (31.5 inches; WERC), but the driest year during the monitoring period actually occurred in the early years of monitoring, while the dairy was still in operation (WY 2004: 22.29 inches; NPS; Bear Valley). In contrast, Years 2 and 3 (WY 2010 and WY 2011) were quite wet, with rainfall in WY 2011 totaling 49.1 inches (WERC). Starting in Year 4, winters became drier, and this trend continued in Year 5, with rainfall approximately 6.3 inches below normal in WY 2013. In addition to Years 2 and 3 post-restoration being wetter than some of the other years, visual examination of the monthly precipitation chart also suggest that there were differences in monthly rainfall distribution, with precipitation being more evenly distributed over the “rainy season” during those two years than in some of the other years (Figure 2).

While levee removal during restoration in 2008 resulted in lowering of creek “bank” elevations to that sufficient to allow overflow of a 2-year-flood event, no overbank flooding from storm flows occurred during either WY 2009 or 2010, even with higher rainfall levels in 2010. In WY 2011, at least one overbank flooding event occurred. There were no overbank flooding events in WY 2012 or in WY 2013.

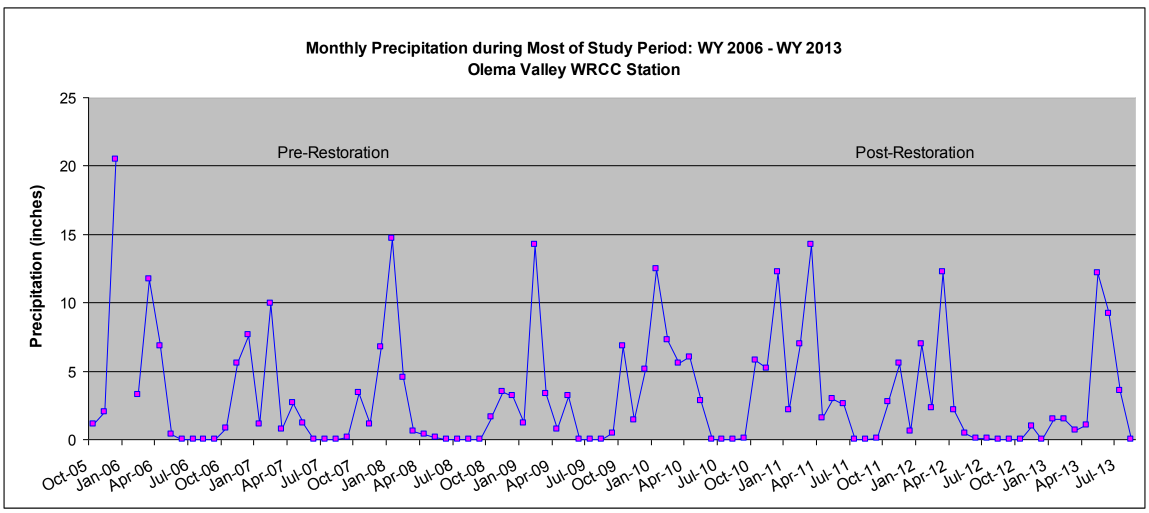

Figure 2. Monthly Precipitation between WY 2005-WY 2013.
Western Regional Data Climate Center: Olema Valley station. Note that sampling actually started in WY 2003, but data not available for this period. Data to left of line is pre-restoration; data to right of line is post-restoration.

4 Results and Discussion: Changes in Hydrology, Topography, and Sedimentation

Abiotic and biotic conditions within the Project Area are strongly swayed by – and tied to – changes in hydrology. One of the most dramatic changes in the Giacomini Wetlands after restoration was the sweeping expanse of water that spread almost immediately across the former dairy pastures with the twice-daily flooding of the tides. This change was predicted. However, what was less well understood was the process by which hydrology within the restored Ranch would evolve, similar to that of vegetation.

Aerial oblique photograph of Giacomini Wetlands taken after levee breach. Northern portion of East Pasture or Shallow Shorebird

During planning for the restoration project, computer hydraulic modeling conducted as part of planning for the restoration project estimated that, based on existing and proposed elevations, 256 of the 550 acres in the East (area adjacent to Point Reyes Station) and West (area adjacent to Inverness Park) Pastures of the Giacomini Ranch would be inundated by tides daily or close to daily (KHE 2006). This modeling assumed that no levees would remain and that some tidal channels would be created. Larger tidal channels were built to jump-start marsh evolution, but only a few smaller tidal creek channels were constructed, with the assumption that most of the smaller channels would develop naturally over time. While levees were removed, the Undiked Marsh that had developed on the outboard of the levees was, in many cases, higher in elevation than the marshplains or former pastures. These marsh shelves, then, represent mini "levees" that can direct – or even constrain – flow within the former pastures.

Hydrologic changes were not notable after final removal of the West Pasture levees and completion of preliminary restoration activities in Olema Marsh in mid-October 2008 (KHE 2009a). However, very dramatic changes occurred almost immediately after final removal of the East Pasture levees on October 25, 2008 (KHE 2009a). Within days, much of the East Pasture -- and the very southern portion of the West Pasture -- was seemingly flooded.

During the first few months after breaching, Park Service staff monitored and mapped tide levels during higher high tide and low tide events to determine how well water levels in the Project Area corresponded with those predicted by hydraulic modeling. Additional monitoring was conducted in December 2009 of extreme high tides and low tides during neap tide conditions, with follow-up monitoring of low tides during spring tide conditions conducted in 2010, 2011, and 2012 and extreme high tides in 2011.

During 2008, 2009, and 2011, the upper extent of higher high tides appeared to match very well with that predicted by computer modeling, often mimicking the contour patterns exactly (Figure 4). In winter 2008, during tides ranging from 6.3 feet to 6.6 feet MLLW (5.8 feet to 6.1 feet NAVD88), tidewaters extended up to the 7.0 to 7.5 feet NAVD88 elevations in the East Pasture (NPS, unpub. data). Computer modeling predictions estimated maximum tidal elevations in the East Pasture of 7.15 feet NAVD88, with the area exposed to tidal influence at extreme tides being close to 282 acres (KHE 2006; Figure 3). However, while the maximum tidal elevations were sometimes greater than predicted, the amount of area in the East Pasture subject to extreme tidal influence turned out to be slightly lower than that predicted (251.7 acres), a difference of 11 percent (NPS, unpub. data; Figure 4). In 2009, the acreage of areas subject to high tides was only slightly less (242.5 acres), which probably resulted largely from the fact that mapping was conducted during an approximately 1 foot lower tide level (5.8 feet to 6.3 feet MLLW or 5.3 feet to 5.8 feet NAVD88; Figure 4). Otherwise, there appeared to be good agreement between 2008 and 2009 high tide extent. The maximum elevations reached by extreme high tides were a little more variable in the

West Pasture. During tides ranging from 6.3 feet to 6.6 feet MLLW (5.8 feet to 6.1 feet NAVD88), tidewaters extended up to the 5.75 feet to 7.0 feet NAVD88 elevations in the West Pasture (NPS, unpub.data; Figure 4). During extreme tides, tidal waters were predicted to reach a maximum elevation of 7.13 feet NAVD88 (KHE 2006; Figure 4). The variability in elevation extent in the West Pasture may result from the influences of freshwater, which may "push back" tidal influence where freshwater surface or groundwater flow is strongest.

Where computer modeling predictions diverged substantially from actual water levels was with low tide conditions. While mapped high tide water levels correspond pretty closely with predicted levels, water levels during lower low tides immediately after restoration were much higher than predicted by computer modeling. In other words, more area remained subtidal or permanently inundated than would have been predicted on elevation alone. Because channels were deliberately undersized to allow natural evolution, the width and density of channels were not large enough currently to fully accommodate flows, and waters were not fully draining on low tide, leaving a significant amount of water in East Pasture channels and marshplains even on the lowest low tides. Drainage problems were exacerbated by the fact that the outboard marsh shelves, which functioned as mini-levees, were funneling flows exclusively through the two primary tidal channel outlets that were created -- the Tomasini Slough, which flows into Lagunitas Creek near Railroad Point in the northern portion of the East Pasture, and, to a lesser extent, the new side channel for Lagunitas Creek, which drains the new Marshplain Enhancement area in the southwestern portion of the East Pasture.

Immediately after restoration, mapping of the permanently flooded areas during extreme low tide conditions indicated that water levels were not dropping below 4 ft NAVD88 in the East Pasture and approximately 3.75 ft -<4 ft NAVD88 in the West Pasture (NPS, unpub. data). Water level patterns in Lagunitas Creek were also affected: a flattening of the water level curve below 3.5 feet suggested that water levels in the creek were also dropping more slowly because of the added volume of water being conveyed by the marshplain (KHE 2009a). Prior to restoration, the morphology of gravel bars in Lagunitas and Fish Hatchery Creeks suggested that, after restoration, subtidal conditions would persist in portions of the Project Area below 2.0 ft NAVD88 in Lagunitas Creek and the East Pasture and 3.4 ft NAVD88 in the West Pasture due to the weir- or dam-type effect these gravel bars have on channel water levels (KHE 2006). In addition to changes on water level patterns, restoration also affected timing of tides, resulting in delays of low tides relative to predicted conditions at the nearby Inverness tide station by as much as 2 hours or more (NPS, unpub. data).

These dramatic hydrologic changes were most evident after restoration in the amount of subtidal area or areas that remained permanently inundated immediately after restoration. In the East Pasture, subtidal area under extreme low tide conditions (-1.7 ft to -0.4 ft MLLW or -1.2 ft to +0.1 ft NAVD88) totaled 109.4 acres immediately after restoration, compared to the 26.5 acres of subtidal area predicted under fully evolved conditions (NPS, unpub. data; KHE 2006; Figure 5). The discrepancy between restored and fully evolved conditions was not quite so great in the West Pasture, where subtidal extent predicted under fully evolved conditions (2.2 acres; KHE 2006) was only slightly lower than actual (7.4 acres; NPS, unpub. data; Figure 5). Interestingly, subtidal extent was actually smaller under neap tide conditions – when the difference in elevation between low and high tides is substantially compressed – than under spring tide conditions, when low tides reach some of their lowest levels. In December 2009, subtidal areas totaled 52.9 acres in the East Pasture and 4.5 acres in the West Pasture when tides ranged between 1.9- to 2.7 feet MLLW (1.4- to 2.2 ft NAVD88). This represents almost a 51 % reduction in subtidal area with only a 1- to at most 3- foot difference in tidal water elevation. These results suggested that drainage was being constrained by the larger volume of water that flowed into the newly restored wetland on a spring tide, when high tides are very high, than on a neap tide, when high tides are lower.

Circulation and drainage patterns are expected to be further altered in the future by changes in Lagunitas Creek and interior tidal channel geometry. Immediate post-project surveys in 2009 had indicated a uniform increase (1.0 ft) in bed elevation of the mainstem Lagunitas Creek channel mid-way through the Project Area relative to elevations in 2003 (KHE 2009b). In contrast, channel elevations at the downstream end of the Project Area (near the former North Levee area) remained fairly comparable in 2009 to those measured in 2003 by the USGS (KHE 2009b).

Since restoration, elevations within the Lagunitas Creek cross-sections have not changed appreciably, with the exceptions of shoals at channel outlets (KHE 2011a, KHE 2012a). In 2009, ebb shoals or gravel bars developed at the mouth or downstream of the mouth of the newly constructed channels draining the East Pasture, with accretion during the first year totaling more than 1 to 2 feet (KHE 2009b). These deltaic-type shoals had encroached into the mainstem Lagunitas Creek channel, reducing the cross-sectional flow area, although they did not span the full width of the channel (KHE 2009b). While both of these shoals rapidly formed after restoration, their evolutionary paths have diverged somewhat. The horseshoe-shaped Tomasini Slough outlet shoal remained relatively consistent in elevation between 2010 and 2011. It is comprised of an inner and outer shoal that range in elevation from 1- to 2 feet NAVD88 (KHE 2011a), although, based on water level results, it may have increased in elevation in 2012 (KHE 2012b). Conversely, the shoal at the mouth of the new side channel off Lagunitas Creek has continued to accrete or build in elevation with estimated deposition rates of 0.7 feet in WY 2010 and 0.75 feet in WY 2011 (KHE 2011a). Shoal build-up slowed in 2012, with elevations increasing only by a few inches (KHE 2012a). With post-restoration winters being relatively dry, little energy in the way of flood scour has been available to counteract deposition of sediments at the mouth of new tributaries to Lagunitas Creek, which has led to a net depositional environment. Should flood flows continue to be reduced, shoals will continue to build in Lagunitas Creek and perhaps change circulation and drainage patterns in the creek and wetland.

While elevations may have increased at the mouth, both of the newly constructed channels have actually deepened since restoration was completed. By 2010, the downstream portions of Tomasini Slough had decreased in elevation relative to constructed elevations by as much as 1.8 feet, with an additional drop of 0.75 feet during 2011 (KHE 2011a). Between 2011 and 2012, channel bottom elevations in Tomasini Slough remained relatively stable (KHE 2012a). The one upstream station on Tomasini Slough for which pre-restoration elevation data exist showed little elevation chance since pre-restoration conditions (KHE 2011a). A similar pattern of channel incision occurred at the newly created side channel off Lagunitas Creek in the East Pasture. At least 1 foot of both channel deepening and widening took place in downstream portions of this small tidal creek, while upstream portions widened, but actually became more shallow through deposition of approximately 1 foot of sediment (KHE 2011a). Channel incision rates slowed in 2012 (KHE 2012a). Unconstructed channels are also becoming deeper: these are naturally developing channels on the marsh floodplain.

Unfortunately, the lack of vegetation, particularly in the northern portion of the East Pasture, may slow down this process somewhat by encouraging overflow of tidal waters and floodwaters onto the marshplain rather than keeping them in channels (KHE 2010a). Indeed, in WY 2012 or Year 4, there was minimal to no change in tidal creek channel morphology, with only some channel widening at two of the cross-sections (KHE 2012a). With little flood flow available for landscape reworking, the tides must do the work, and tidal energy within the restored wetlands is dampened by the shoals off Tomasini Slough and the Lagunitas Creek side channel (KHE 2012a). As downstream reaches deepen in future years, steepening channel gradients will propagate upstream, and channels will scour, if needed (KHE 2012a).

Interestingly, marshplain areas increased in elevation in both the East and West Pastures, despite the lack of sediment input from flood flows and the massive vegetation die-off in the East Pasture that would have been expected to compact soils due to loss of root volume below the soil surface (Appendix A; Parsons and Ryan 2014). In both pastures, elevation gains between 2008 and 2010 exceeded sediment deposition rates measured through use of feldspar markers, with elevation gains ranging during that period from 12.7 mm in the West Pasture to 19.2 mm in the East Pasture and sediment deposition rates for that same period ranging between 0.9 and 5 mm annually in the East and West Pastures (Appendix A). In 2011, the trends in elevation shifted somewhat, with elevation again increasing in the West Pasture (4.5 mm) relative to 2010 elevations, but decreasing in the East Pasture (-3.6 mm), although, overall, elevations were still considerably higher in the East Pasture post-restoration than pre-restoration (Appendix A). In 2012 and 2013, elevations in the East Pasture increased again, with annual elevation gains ranging from 8.0 to 10.1 mm (Appendix A). Interestingly, some of the other sampling sites that had not appeared to gain in elevation since 2008 such as the ones at Walker Creek Marsh in the northern end of Tomales Bay had positive elevation increases for the first time in 2011 (4.5 mm), and this trend continued in both 2012 and 2013 (Appendix A). Considering that elevation gains typically exceed sediment deposition rates, most of the elevation gains in the restored wetlands appear to result from changes in subsurface processes, with reintroduction of tides potentially increasing porewater volume in the soils and slowing down subsurface oxidation rates of organic matter (Appendix A).

As discussed earlier, most of the sediment deposition occurring in Lagunitas Creek and the Project Area appears to come from re-working of soils from the Project Area, which are now exposed and vulnerable after construction grading and decay of pasture vegetation. With the first winter being a dry one, sediment inputs from the upper watershed were probably minimal, particularly as there were no overbank flooding events. While the Year 2 winter was much wetter, there was still not enough flow volume during storm events to cause overtopping of creek banks in the Project Area, and, thereby, any deposition of sediment on newly restored marshplains. Some overtopping did occur during the winter of Year 3, when rainfall totals were even higher than Year 2, but there was only one very brief event. Even during large storms, most of the peak flood flow and sediment generated are trapped by upstream dams, reducing flood volume and sediment loading to downstream areas. The winters of Years 4 and 5 have generally been drier than normal, so flood flows have also been lower than normal, with the exception of December 2012, when almost 12 inches of rain fell, but most of the surface run-off generated by this series of storms was captured behind the dams.

Despite this lack of overbank flooding, sedimentation monitoring has shown that sediment was still deposited on Project Area marshplains during the last five years (Appendix A). Dry winters and reduced flood flow volume have led to a net depositional environment both within the Project Area and other marshes, except where flow velocity is high enough to counteract this trend, such as in the downstream portions of the Tomasini Slough and the Lagunitas Creek side channel. With overbanking flooding during storms being minimal, much of this sedimentation may derive from re-working of Project Area soils (KHE 2012a). Any existing tidal energy in these systems is being used primarily to transport sediment rather than scour channel banks and beds (KHE 2012a).

Annual deposition rates since restoration in the northern portion of the East and West Pastures appeared higher than in other sites in Tomales Bay and lower than in Limantour Estero, however, differences were not statistically significant (GLM, df=3, F=2.12, P=0.11; Appendix A). Average annual sedimentation rates since restoration in the Giacomini Wetlands have ranged from 0.9 to 8.1 mm/year, compared to 0.7 to 3.0 mm/year in other Tomales Bay sites (Appendix A). Since restoration was implemented, the lower-elevation, more-deeply-subsided East Pasture area appeared to have a higher average annual sedimentation rate (6.1 mm/year) than the higher-elevation West Pasture area (1.9 mm/year; Appendix A). The Limantour sites had a higher range of annual sediment deposition rates than Tomales Bay ones, with average rates ranging from 1.6 to 4.2 mm/year (Appendix A). These numbers are still seemingly lower than historic sediment deposition rates in Tomales Bay, which were estimated to average 5 mm/year (Rooney and Smith 1999). In San Francisco Bay, current short-term accretion rates appear to range between 3.1 and 5.9 mm/year, with higher rates at lower-elevation marshes (Callaway *et al.* 2012).

Developing tidal channel

Essentially, the Giacomini Wetlands are in the process of hydrologic evolution. The conditions predicted by hydraulic modeling represent a later phase in wetland development. Over the coming years, existing and created channels will continue to increase in size to accommodate flood flows, and new tidal channels will develop, increasing exchange between the restored wetland and Lagunitas Creek and creating closer to equilibrium between tidal inflow and outflow. In addition, some portions of the higher elevation Undiked Marsh outboard of the levees may continue to erode (as they have been doing prior to restoration), allowing more tidal waters to sheetflow across the marshplain back into Lagunitas Creek. Some of these changes may be accelerated during flood events, although storms so far have not been of sufficient magnitude to dramatically alter the wetland landscape.

This evolution appears to be already well underway. Hydrologic data suggested that the marsh was draining slightly faster during outgoing or ebb flows in 2009 than 2008 (KHE 2010a), and drainage improved slightly again between 2009 and 2011, at least during spring tides (KHE 2011b). Low tide elevations continue to be constrained as they were prior to restoration by the presence of gravel and sand bars at the mouths of creeks, which keep water levels at about 2.0 feet NAVD88, although, in 2011, water levels repeatedly dropped below 2.0 feet NAVD88 (KHE 2011b). In 2012, however, maximum low tide elevations increased slightly within the East Pasture, possibly due to a build-up of the Tomasini Slough shoal, with water levels almost never dropping below 2.0 feet NAVD88 (KHE 2012b).

This improvement in drainage efficiency can be seen in the dramatic declines over the last few years in the extent of subtidal areas during an extreme low tide. Acreage declined from 109.4 acres in the East Pasture immediately after restoration to 68.1 acres in summer 2010 under approximately equivalent tide conditions (-0.44 to -1.74 ft MLLW in 2008 vs. -1.54 to -1.67 ft MLLW in 2010, Figure 5). This represents a 38% decrease in extent of permanent inundation during extreme low tides within two years. In 2011, this trend appeared to continue (Figure 3): acreage of subtidal areas in the East Pasture dropped to 51.0 acres, even though water levels may have been influenced somewhat by the unusual rainfall pattern in WY 2011, where precipitation extended well into the summer. (Summer stream discharge flows averaged 10 cfs compared to the median estimate of 6 cfs, which may have kept the marsh from fully draining during low tide events.) This situation demonstrates that ecosystem evolution following restoration is not a linear process, but can occur in distinct stages or phases that involve triggering or exceeding thresholds before the wetland moves into the next evolutionary stage or phase.

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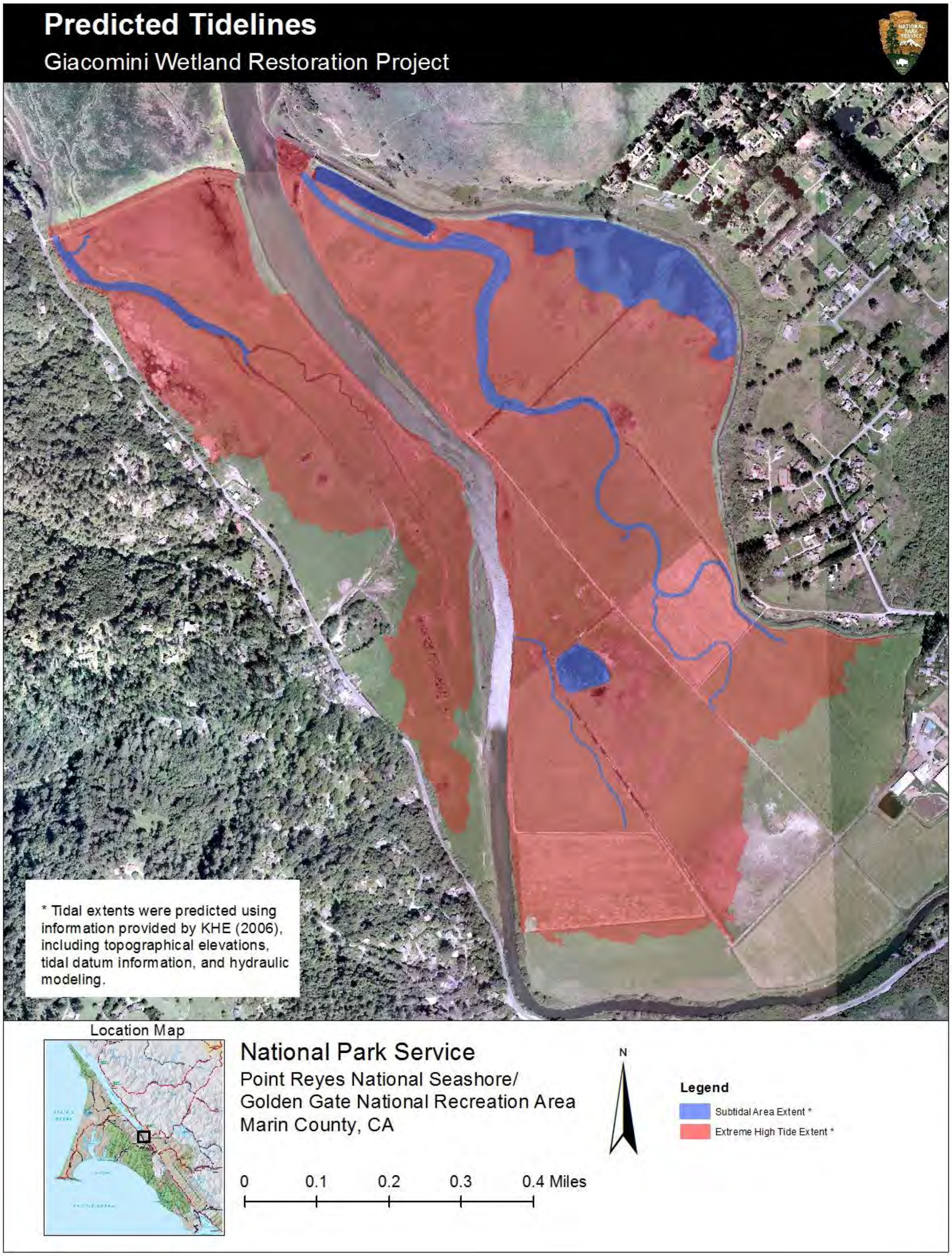

Figure 3. Predicted Tides

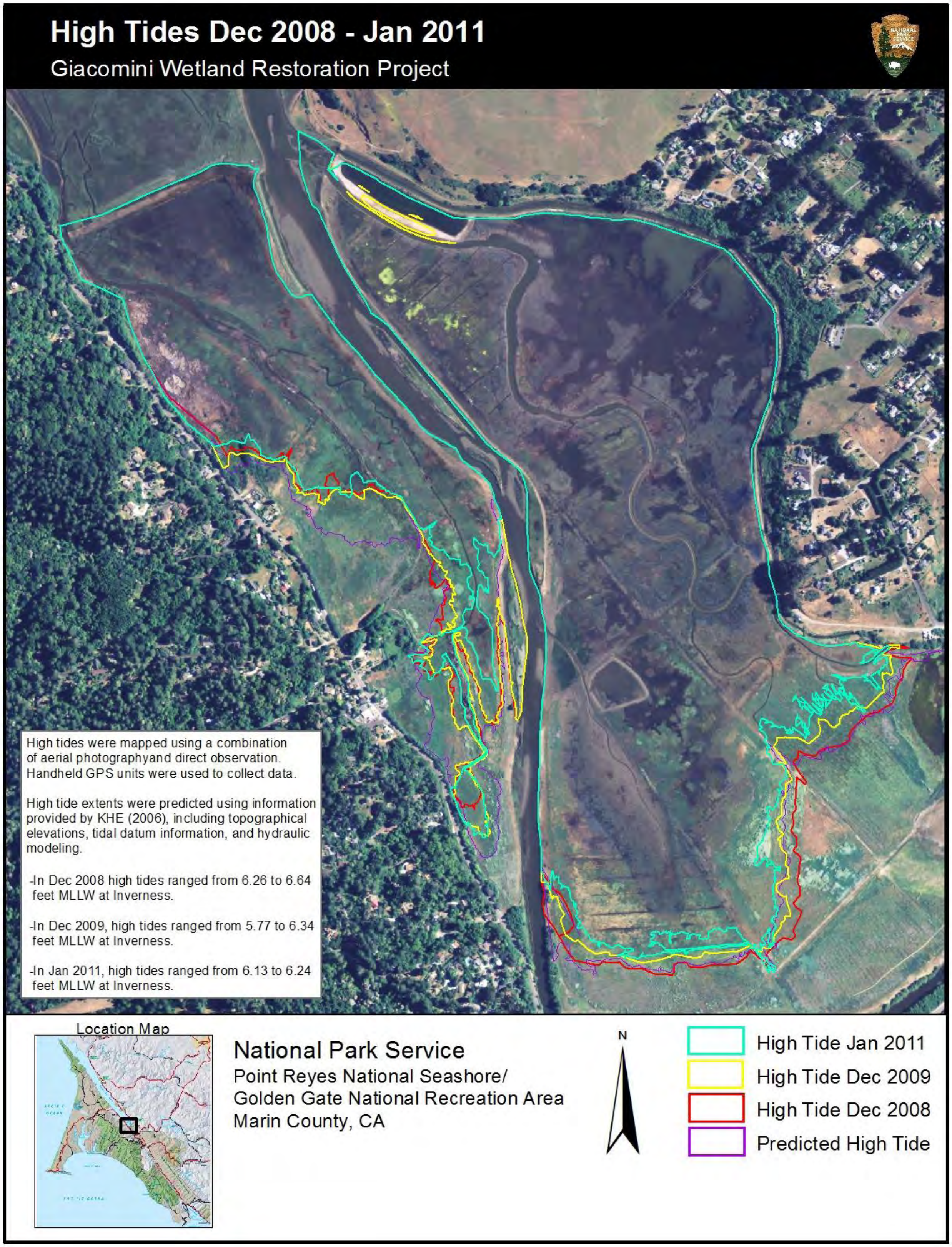

Figure 4. High Tides

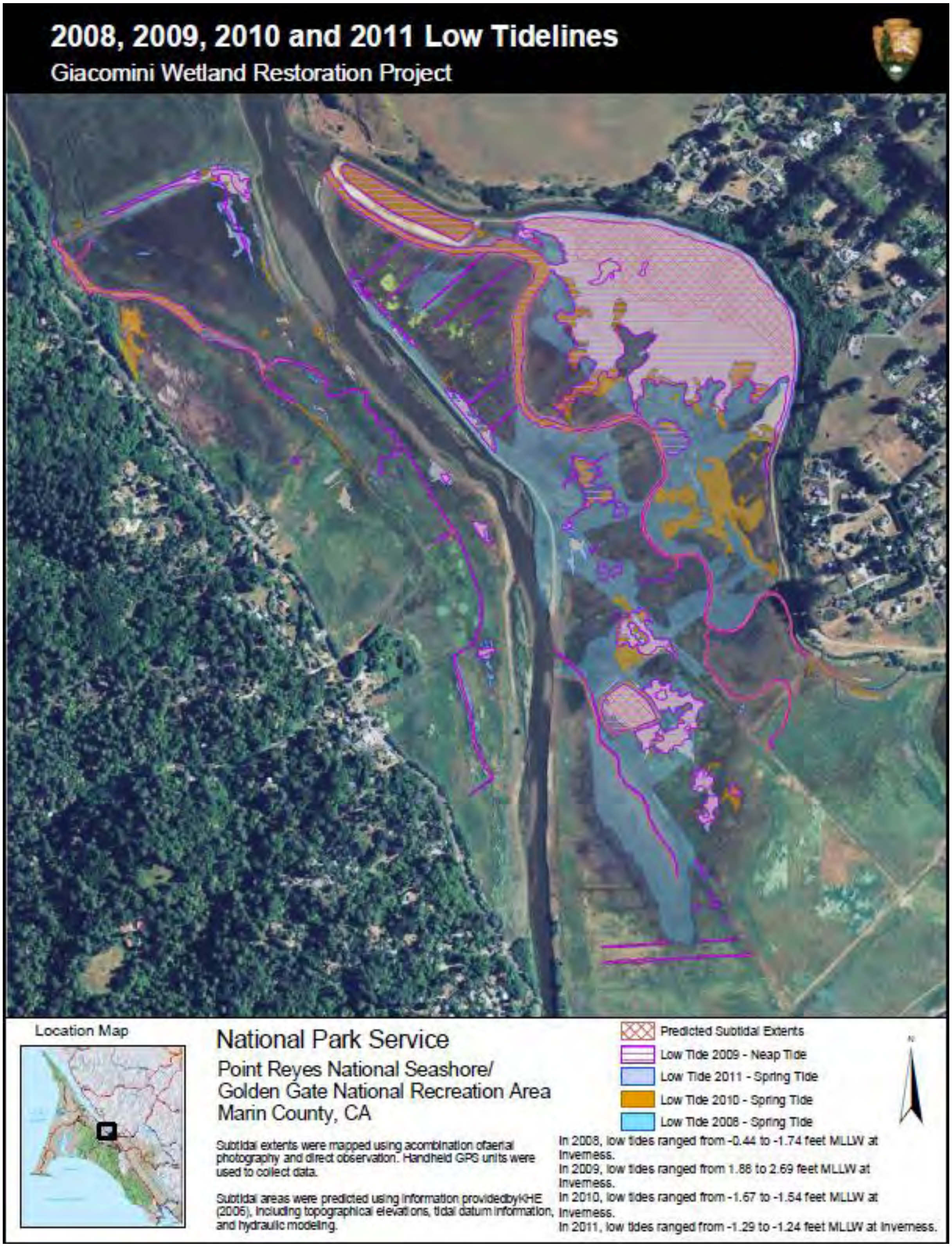

Figure 5. Low Tides

Low tidelines under spring and neap tide conditions since restoration was implemented. Colored areas represent inundated areas under extreme low tide conditions. Predicted refers to areas that were predicted by hydrologic modeling to remain subtidal or inundated under fully evolved marsh conditions.

5 Results and Discussion: Changes in Water Quality

One of the primary objectives of the Giacomini Wetland Restoration Project was to improve not only the quality of water within the former dairy ranch itself, but the quality of water flowing downstream to Tomales Bay. For more than 60 years, the levees constructed by the Giacominis essentially funneled all flows from the upper part of the Lagunitas Creek watershed directly to Tomales Bay. Three major creeks actually merge just upstream of the Giacomini Wetlands: Lagunitas Creek, Olema Creek, and Bear Valley Creek, which flows through Olema Marsh. These creeks represent a sizeable amount of the freshwater inflow—and potential pollutant load—to Tomales Bay, with more than 66% or two-thirds of Tomales Bay's freshwater input coming from this drainage (Fischer *et al.* 1996). Additional inflow to Lagunitas Creek comes from Tomasini Creek, which flows through the East Pasture adjacent to Point Reyes Station, and Fish Hatchery Creek, which flows through the West Pasture adjacent to Inverness Park. These creeks also represent potential sources of pollutant loading from non-point surface run-off, leakage from septic systems, agricultural drainage, and, in the case of Tomasini Creek, potentially leaching from an improperly closed landfill.

Water Quality Conditions within the Wetlands Prior to Restoration

Of course, not all the pollutants came from upstream portions of the watershed. The dairy also represented a sizeable source of potential pollutants to Tomales Bay. During full-scale agricultural operation, the East Pasture was intensively managed through construction and frequent dredging of ditches, manuring of pastures, and grazing of several active dairy herds. The West Pasture, which was located across Lagunitas Creek from the dairy facility, was less intensively managed and grazed. Practices such as frequent dredging of ditches led to some of the pasture ditch waters, particularly in the East Pasture, being very low in oxygen -- low enough to impact and perhaps even kill aquatic organisms (Parsons 2009).

Cows at Giacomini Ranch prior to restoration

Not surprisingly, concentrations of nutrients and pathogens were typically highest in the dairy relative to levels in Lagunitas Creek, other creeks flowing into the Giacomini Ranch, or natural tidal marshes in Tomales Bay (Parsons 2009). One of the predominant nutrients in Tomales Bay is nitrates. The relatively well oxygenated conditions present in most of the Study Areas -- except the East Pasture ditches -- may have contributed to the dominance of nitrates as the primary source of nutrients (Parsons 2009). For pollutants such as nitrates, pulses appeared to substantially drive nutrient dynamics more than consistent loading, as median concentrations (central values) of nitrates within the dairy (0.83 mg/L) were not much different from those in natural marshes (0.7 mg/L; ibid). However, mean nitrate concentrations, which reflect the influence of "spikes" or "pulses," showed greater disparity, with concentrations averaging 7.25 mg/L (NO_3^-) for the East Pasture and dropping below 1.1 mg/L for other areas (Parsons 2009). Another form of nitrogen, ammonia, typically present in more low-oxygen environments, was only infrequently detected during pre-restoration sampling, but of the 64 detections of ammonia during the study, more than 47% of them occurred in the East Pasture (Parsons 2009).

In general, pathogens represent one of the major water quality issues facing Tomales Bay. While seemingly pristine, the Bay and its surrounding watershed generate a considerable volume of pathogen indicator bacteria, total and fecal coliform, because of the large amount of land in agricultural use, leaking septic systems in the many rural residential communities perched on the Bay's edge, and other factors such as bilge discharge from boats. With Giacomini Ranch supporting a considerable number of dairy cattle during its operation, pathogen concentrations were definitely elevated relative to natural waters.

Prior to restoration, the Project Area had substantially higher estimated median concentrations of fecal coliform (1,600.9 mpn/100ml) than natural marshes (72.0 mpn/100 ml), although, based on results of statistical analyses, seeming differences with upstream areas or source creeks (705.6 mpn/100 ml) might have been obscured by high variability in the data (Parsons 2009). Not surprisingly, the heavily managed East Pasture had significantly higher estimated levels (6,298.8 mpn/100 ml) than most of the other areas sampled, with the possible exception, from a statistical standpoint, of Olema Marsh (1,821.4 mpn/100 ml; ibid). Estimated geometric means or medians for all other areas ranged between 356.9 mpn/100 ml for downstream Lagunitas Creek to 1,131.7 mpn/100 ml for the West Pasture (ibid).

While concentrations of nutrients and pathogens were highest in the dairy, loading or the volume of pollutant was actually highest in the creeks or upstream areas flowing into the restoration Project Area and, eventually, into Tomales Bay (Parsons 2009). These results are not surprising. The East Pasture was entirely leveed and was only hydrologically connected to Lagunitas Creek during periods when the pump was actively operated or when floodwaters overtopped the levees and flowed through the pasture to drain out via the spillways at the ranch's northern end. The West Pasture functioned as a muted tidal system in that a somewhat leaky one-way tidegate on Fish Hatchery Creek's downstream regulated inflow and outflow into the pasture. There were some exceptions. For example, for fecal coliform, estimated loading rates for the restoration Project Area (mean=249,389 mpn/s) were lower than upstream area or source creeks (mean=3.86 million mpn/s), but higher than natural marshes (mean= 60,094.1 mpn/s; ibid). Conversely, natural marshes had the highest loading rates for phosphates (0.15 mg/s), with rates for the restoration Project Area (0.03 mg/s) and upstream areas or source creeks (0.06 mg/s) much lower, which may relate to the more substantial marine influence on natural marshes (ibid).

Pipe conveying waters from former Giacomini Ranch

As with many other parameters, loading rates or pollutant volumes were typically driven by pulses or spikes, particularly during storm events. The importance of storm events to downstream loading is evident in the disparity between mean (10.11 mg/s) and median (0.66 mg/s) instantaneous loading rates for nitrates on Lagunitas Creek: during an April 2006 storm, rates reached as high as an estimated 220 mg/s (Parsons 2009). Research on other agricultural watersheds has also documented the highest export of nutrients and pathogens in stormflow, with levels generally higher in the wet season than the dry season (Vanni *et al.* 2001, Lewis and Atwill 2007). Ironically, storms were the least sampled due to inherent planning and logistical difficulties, however, we increased efforts in later years to capture storm events in the monitoring record.

Water Quality Conditions After Restoration

Restoration of the Giacomini Wetlands has actually occurred in several distinct phases. While the Giacomini family had a Reservation of Use Agreement that allowed them to operate through fall 2007, the family elected to sell off their main dairy herds in 2006 and instead a graze a smaller number of dairy heifers or young cows. From that point in 2006 through construction in fall 2008, the restoration process began to occur, but in a more passive manner. Most of the active agricultural management -- including ditching and manure-spreading -- was discontinued in 2006 with limited irrigation and grazing continued until fall 2007. Phase I of construction in fall 2007 involved removal of most of the agricultural infrastructure -- barns, pipelines, fencing, and surface scraping of the manure slurry disposal pasture -- as well as removal of the southernmost levee in the East Pasture. However, full restoration, which hydrologically reconnected the former historic marsh to Lagunitas and Tomasini Creeks, was not realized until October 2008, when the levees on both the East and West Pastures were fully removed.

Analyses performed as part of environmental compliance predicted that water quality conditions within the restored wetlands would remain similar or even worsen immediately after restoration before showing improvement over the long-term (NPS 2007). While removal of cows, manure, and agricultural practices such as ditching would be expected to improve water quality, improvement was not expected to happen overnight due to high levels of nutrients and pathogens in present. In addition, conversion of the pasture back to marshland would cause a massive die-off in mostly non-native pasture grasses and herbs that would, upon breakdown by bacteria, release large amounts of organic and inorganic nutrients into overlying waters. Changes in water salinity and hydrologic regime would also affect soils, most of which have high nutrient -- and possibly pathogen -- levels from decades of agricultural management. Nutrients would be expected to flux or move out of the soils into overlying waters, as well, in response to changes in soil chemistry. With accelerated breakdown of organic matter both above and below-ground would come higher demands on oxygen in water and soils, leading to possible episodes of oxygen depletion in overlying waters. Breakdown of organic matter could also lead to decreases in pH from release of humic acids. The die-off of vegetation and resulting increase in bare ground would also be expected to exacerbate turbidity issues resulting from disturbance of soils during construction for creation of new tidal channels and excavation of certain areas to lower intertidal elevations.

Interestingly, during the first water year after restoration (October 2008 through September 2009), most of the anticipated short-term negative impacts to water quality within the restored wetlands were not observed (Parsons 2010). Rather, there was a general improvement in overall water quality conditions that occurred at a much more rapid pace than initially anticipated (ibid). Remarkably, despite the fact that movement of sediment from the wetland was evident in shoaling at creek mouths in Lagunitas Creek (KHE 2009), turbidity levels with wetland waters remained equivalent to pre- and passive restoration conditions during Year 1 or the first year after restoration (Parsons 2010). Lower turbidity levels may have been driven to some degree by the dry conditions during the 2008–2009 winter, but sampling events did capture at least two small to moderately sized storm events (ibid).

The influence of storm events was perhaps more evident in Year 2 (October 2009 through September 2010). While there were no statistically significant differences in turbidity levels between pre-and post-restoration in Year 1, differences did exist between pre and post-restoration in Year 2, with mean turbidity levels more than tripling from 15.7 NTU to 60.8 NTU during Year 2 (Parsons 2011). In WY 2009/2010, rainfall totals jumped, and 50% of the sampling events occurred moderate to large storm events, although there was still no overbank flooding, at least from Lagunitas Creek. The fact that turbidity levels were significantly higher in the Project Area than in the Reference Areas suggests that turbidity levels in the restoring wetlands exceeded those that would be expected in mature marshes simply based on normal sediment resuspension pulses during storm events. Therefore, Year 2 may better represent the short-term increase in turbidity levels immediately after restoration that was predicted in the environmental compliance analysis documents.

Turbidity during subsequent years dropped relative to Year 2, but still remained higher in Year 3 (21.3 NTU), Year 4 (32.6 NTU), and Year 5 (28.2 NTU) than in Year 1. Overall, turbidity has increased by 44% after the levees were breached, however, differences in the Project Area between treatment periods was only weakly significant (GLM, df=2, F=2.37, P=0.09; log-transformed). A statistically significant interaction existed between year and Study Area (GLM, df=4, F=4.98, P=0.001; log-transformed), suggesting that turbidity differences do exist between the restored marsh and natural marshes, but that this difference is mediated is somewhat by temporal differences. Turbidity levels in natural marshes actually dropped following restoration approximately 20%.

Following levee breaching, mean oxygen levels in the Project Area increased 16% from 7.30 during Pre-Restoration to 8.55 mg/L during Passive Restoration and to 8.50 mg/L during Full Restoration (GLM, df=2, F=31.2, P<0.001; log-transformed). Lower D.O. levels occurred in Year 2 than Years 1, 3, 4, and 5: both Years 2 and 3 were quite wet, although the Year 2 sampling approach may have captured more storm events. Cold temperatures and strong flow conditions could suppress biological activity in waters relative to warmer, more quiescent periods. Improvement was especially marked in the East Pasture, where mean oxygen levels climbed 72% from 4.98 mg/L pre-restoration to 8.58 mg/L after restoration.

With restoration, oxygen concentrations might have been expected to decrease -- or only increase slightly overall – due to the abundant organic matter that die-off of pasture vegetation that has been released into Project Area waters during the first year and even second year of restoration. With high levels of organic matter, bacteria become extremely active and rapidly deplete oxygen levels in overlying waters, particularly during the night, when oxygen stores are not replenished through primary production. While pasture vegetation went through multiple stages of die-off in the first year with some die-off in the second year, the effect of this die-off was not evident in Project Area oxygen concentrations, and, in fact, by the fifth year following restoration, oxygen levels in the Project Area were equivalent to those in Reference Areas or natural marshes (GLM, df=1, F=0.01, P=0.91; log-transformed).

The temperature of waters in the restored wetlands depends to some degree on the temperature of the waters flowing into it, as well as what happens to that water once within the wetland. Temperatures might have been expected to increase perhaps with reintroduction of warmer tidal waters to the former freshwater-dominated dairy, although an increase in daily exchange of waters with adjacent once levees were removed could also act to lower water temperatures. Since breaching of the levees, mean temperatures dropped by 4% within the Project Area from 15.9 degrees Centigrade Pre-Restoration to 14.2 degrees Centigrade during Passive Restoration and to 15.3 degrees Centigrade during Full Restoration (GLM, df=2, F=7.40, P=0.001; log-transformed). This temperature decline could have simply reflected the change in hydrologic regime within the restored wetland. However, an even larger decline (10%) occurred in natural marshes following 2008 -- even in those that are too distant to have been affected by restoration. This suggests that more regional or watershed-scale factors affected temperature regimes such as climatic patterns, which is supported by the fact that temperature varied significantly between years following restoration among the Study Areas (GLM, df=4, F=4.84, P=0.001; log-transformed). Years 2 and 3 had higher-than-average rainfall, and stormwater-influenced creek flows are colder than tidal waters. These regional factors may have narrowed any differences that might have otherwise existed between the newly restored wetland and natural marshes, as temperature did not differ between Study Areas during the post-restoration period (GLM, df=1, F=0.03, P=0.87; log-transformed).

Similarly, the alkalinity or pH of waters also reflects both the pH of source waters, as well as conditions within the wetland. Most creeks feeding into the Project Area actually have fairly high or basic pHs (range = 7.7–8.1) regardless of differences in geologic substrate between the granitic Inverness Ridge and the Point Reyes Mesa coastal marine terrace and surrounding Franciscan Formation hills, which are separated by the San Andreas Fault that created this tectonic estuary (Parsons 2009). This pH range is similar to that of tidal waters. Lower pH waters (~5.9–6.6) only occur in areas where there is more extensive influence from groundwater (ibid). Groundwater substantially influenced the hydrology of the former ranch, because the levees impounded this subsurface flow.

While introduction of full tidal flows to the Project Area might have been expected to boost pH, the median pH in the Project Area has actually decreased as much as 4.6% since dairy operation from 7.60 during Pre-Restoration to 7.30 during Passive Restoration and to 7.25 during Full Restoration (Figure 6). A significant interaction existed between Study Area and treatment period, with pHs dropping most in Upstream Areas (6.6%) since pre-restoration (GLM, df=4, F=4.35, P=0.002). Median pH in the Project Area during the first five years after restoration was 7.25 during Year 1, 7.14 during Year 2, 7.26 during Year 3, 7.21 during Year 4, and 7.37 during Year 5 (Figure 6).

The seeming drop in pH could be attributed a number of factors. First, the increase in hydrologic exchange and decrease in water residence time may have reduced the influence of phytoplankton blooms on pH relative to pre-restoration conditions: phytoplankton or algal blooms tend to drive up the alkalinity of waters (Parsons 2010). Secondly, breakdown of organic matter from extensive die-off of pasture vegetation can also increase release of humic acids into overlying Project Area waters, resulting in a decrease in pH. In addition, other biogeochemical processes can cause release of acidic substances into overlying waters when soils of tidal wetlands are dry for long periods of time. For example, flushing of sulfuric and iron-associated acids from oxidation of reduced sulfur and iron in soils into overlying waters can also decrease pH: sulfuric and iron-associated acids are generated when pyrites or other reduced or anoxic forms of sulfate and iron in the soil are oxidized and broken down or converted during drawdown or low-water periods, with soluble acids from oxidation then released into overlying waters when tidal exchange is reintroduced. The Project Area was deliberately dried out before and during construction to improve constructability conditions, resulting in even drier conditions than when the Project Area was ranched.

Most of these factors relate to possible changes in physical and biological conditions that may have occurred as a result of restoration. However, a similar slight, but significant, decrease in mean pH was also observed in natural marshes, including in areas that were not necessarily affected by the restoration (Figure 6). In Reference Areas, pH dropped from a mean of 7.67 and 7.53, respectively, during Pre-Restoration and Passive Restoration sampling periods to 7.34 during Full Restoration (Figure 6). In evaluating the data more closely, it appears that the median pH for Reference Areas started declining in WY 2007 prior to restoration and continued declining each year until WY 2010, when median values at least appear to have roughly stabilized around 7.30 – 7.35 (Parsons 2012). The largest declines were observed at the Undiked Marsh directly north of the Giacomini Wetlands and Limantour Marsh in Limantour Estero (ibid).

The changes in these two natural marsh systems could relate directly to the fact that, in both cases, upstream areas have been restored, and restoration could be affecting the pH of downstream marshes, as well as that of the Project Areas. However, pHs in more distant areas of Tomales Bay appear to have also been lower during this period, as well. University of California, Davis, (UC Davis) researcher Ann Russell and her colleagues temporarily reoccupied the Tomales Bay sampling stations established by the Land Margin Ecological Research (LMER) program in the 1980s as part of a current research effort to understand the impacts of ocean acidification and climate change on estuarine invertebrates. During LMER, sampling was conducted at 10 stations from the outer Tomales Bay near the mouth to the southernmost one some distance north of the Undiked Marsh between 1987 and 1995. Russell reinitiated sampling in fall 2008 just when the restoration project was almost complete. During sampling efforts from 2008 to 2010, Russell has found no difference in most of the field parameters between the historic LMER data and recently collected data, however, pH did appear to have declined in both the outer and inner Bay by as much as 0.25 pH units (A. Russell, UC Davis, *pers. comm*.).

While apparent decreases in pH in Tomales Bay and its marshes might lead to questions about the effect of ocean acidification on pH of tidal waters flowing into estuaries, there are several factors that argue against the findings of our monitoring and Russell's being connected to ocean acidification. While Russell and colleagues did observe larger decreases in pH in the Outer Bay relative to the Inner Bay (Russell *et al.* 2010), in our results, pH decline appears to have been greatest furthest from the mouth of the estuary. Russell also believes that the change observed in pH for Tomales Bay was too large to be attributable to dissolution of CO^2 from the atmosphere into estuarine waters (A. Russell, UC Davis, *pers. comm*.). However, not all carbon inputs into the estuary come from the atmosphere (ibid). In addition to changes in pH, concentrations of dissolved organic carbon (DOC) and, in the Outer Bay, dissolved inorganic carbon (DIC) also appeared higher relative to the LMER program sampling period (Russell *et al.* 2010).

Some of these differences may relate to the fact that the sampling period during the study implemented by Russell and her colleagues was wetter than the LMER sampling one (J. Largier, UC Davis, *pers. comm*.). This would also affect transport of carbon from the upper watershed into the Bay. At least in terms of the Park Service's dataset, cumulative rainfall volume did appear equal to or very slightly higher during the post-restoration sampling period (mean for Oct 2008-Sept 2013 =1.00 in > normal) than the pre-restoration one (mean for Oct 2004-Sept 2008=0.70 in>normal). Higher rainfall would increase freshwater inflow from both surface water and groundwater sources, which can have lower pH values than tidal waters in our Study Area: this is evident in the generally lower range of values for the Upstream Areas during the post-restoration sampling period (Figure 6). In particular, groundwater outflow in this area tends to have a slightly lower pH: these factors could affect pH within the restored wetlands, above and beyond any restoration effect. In addition, while these pH changes may be unrelated to ocean acidification, it does not rule out that we may begin to see changes related to climate change in future years, although pH in estuaries is normally more highly variable than that of oceans even without the influence of climate change.

As noted earlier, nitrates are one of the predominant nitrogen sources in the Tomales Bay watershed, and levels were elevated in the former dairy ranch prior to restoration. Immediately following breaching of the levees, nitrates did actually show a sharp pulse, with levels in newly restored waters averaging 3.44 mg/L (Parsons 2010; Figure 7). However, concentrations dropped dramatically in subsequent sampling events, which included two storm events (Figure 7). By January 2009, estimated concentrations had dropped to an average of 0.18 mg/L and median of 0.13 mg/L, with May 2009 (average=0.02 mg/L) and August 2009 (average=0.06 mg/L) even lower (ibid).

Since restoration, mean nitrate concentrations (1.02 mg/L) have fallen at least 68% relative to those recorded prior to restoration (3.22 mg/L; $P<0.0001$) and during Passive Restoration (4.52 mg/L; $P=0.06$; GLM, df=2, $F=16.5$, $P<0.0001$; Figure 7). Similar to average values, estimated median nitrate values dropped 70% from 0.83 mg/L Pre-Restoration to 0.37 mg/L during Passive Restoration and to 0.25 mg/L during Full Restoration (Mood Median, df=2, Chi-Square=49.8, $P<0.0001$).

The strong disparity between mean and median levels for nitrates in the Project Area suggest that certain sites or sampling events are substantially influencing results by elevating mean nitrate levels. One factor affecting results is a non-point source discharge from Point Reyes Station into one of the restored wetland features, which sporadically causes large spikes in nitrate concentrations. This “upstream” sampling location exceeded the U.S. Environmental Protection Agency maximum standard of 10 mg/L during every sampling event in Years 2, 3, 4, and 5 and 75% of the events in Year 1 (Figure 7). Not surprisingly, the highest nitrate levels in the restored wetland during Years 3, 4, and 5 occurred in the Tomasini Triangle Pond, which is a created freshwater marsh acting as the receiving water for this non-point source discharge. The pond may also be affected by septic- influenced groundwater from the Point Reyes Mesa.

In addition to non-point source run-off, nitrates do appear influenced – if not completely driven – by watershed-scale factors such as weather patterns. The possible influence of larger scale factors such as precipitation patterns may account for the observed drop in nitrate levels in natural marshes, as well as the restored one, after 2008. Estimated mean nitrate levels in natural marshes decreased after the Pre-Restoration period from 0.88 mg/L to 0.36 mg/L during Passive Restoration (~2007 – 2008) and then climbed slightly again to 0.50 mg/L during the full restoration sampling period (Oct 2008 to Sept 2013; GLM, df=2, $F=29.3$, $P<0.0001$). Nitrate pulses do show a strong relationship with precipitation. During storm events, nitrate concentrations in Lagunitas Creek can reach as high as 2.0 – 2.5 mg/L, which is notably higher than the peak nitrate concentrations of approximately 1.5 mg/L (24uM) documented off the Point Reyes coast that is potentially exported into Tomales Bay during upwelling events (Largier *et al.* 2006, Wilkerson *et al.* 2006). The relationship of precipitation to average or median nitrate levels is a little more cryptic. Mean nitrate levels showed no strong relationship with “wet/dry” year classification, with average levels during “wet” years ranging from 0.63- to 1.02 mg/L and average levels during “dry” years (Years 1, 4, 5) ranging from 0.47 – to 2.00 mg/L. The same was true of median values, as well. This suggests, as was hypothesized even prior to restoration, that factors other than watershed loading may influence nitrate levels such as internal cycling of nutrients from breakdown of organic matter during the dry season.

Post-restoration declines were even more dramatic for fecal coliform than nitrates. Following restoration, the estimated geometric mean or median fecal coliform concentrations decreased significantly in the Project Area, dropping 93% from 1,600.9 mpn/100 ml during Pre-Restoration to 90.1 mpn/100ml during Full Restoration (MLE, df=2, Chi-Square=124.9, $P<0.0001$; Figure 8). A decline in fecal coliform levels was also observed in natural marshes, but the drop was not as large (31%; MLE, df=2, Chi-Square=85.1; $P<0.0001$; Figure 8). Fecal coliform decreased in reference wetlands from a geometric median of 72.0 mpn/100ml to 49.6 mpn/100 ml (Figure 8). A significant interaction existed between Study Area and sampling period, which suggests that differences between the restored and natural marsh show some temporal variability (MLE, df=5, Chi-Square=226.8, $P<0.0001$).

The dramatic decline in fecal coliform concentrations following restoration is also evident in changes in the frequency of exceedance of Basin Plan or TMDL standards. Approximately 35% of samples collected after restoration exceeded the contact water recreation standards of 200 mpn/100 ml, compared to approximately 78% Pre-Restoration, at least a 55% decrease. Only 5% of samples collected after levees were breached exceeded 2,000 mpn/100 ml, the standards for non-contact water recreation, whereas 47% exceeded before levee removal. Exceedance of municipal water supply thresholds of 20 mpn/100 ml dropped from 95% of all samples collected in the Project Area Pre-Restoration to 79% of all samples collected during Full Restoration, respectively.

One of the established sites for fecal coliform monitoring in the TMDL program is Lagunitas Creek at the Green Bridge, just upstream of the restoration period. The restoration project was not expected to directly influence fecal coliform concentrations at this location, because it is upstream of the restoration project, and fecal coliform levels are strongly associated with watershed loading. However, there may be indirect effects due to changes in hydrologic circulation and other factors. Coliform levels at the upstream end of the Project Area boundary on Lagunitas Creek at the Green Bridge dropped slightly relative to Pre-Restoration conditions, with the TMDL standard of 200 mpn/100 ml being exceeded approximately 57% of the time, compared to 72% of the time prior to the restoration period. The 90th percentile of 400 mpn/100 ml standard was exceeded approximately 39% of the sampling periods during the post-restoration period, as opposed to 58% of the time Pre-Restoration. Exceedances of the 95 mpn/100 ml TMDL load-based allocation for the Green Bridge sampling site dropped somewhat from 100% during Pre-Restoration to 73% after restoration.

Perhaps, some of the most interesting results came from the ammonia data. In the first year following restoration, the number of ammonia detections decreased 43% from 22.8% of the samples in restoration Project Area prior to restoration to 13.0% of the samples after restoration (Parsons 2010; Figure 9). Estimated East Pasture concentrations fell even more dramatically from 2.61 mg/L pre-restoration to 0.44 mg/L in Year 1 post-restoration, a decrease of 83% (ibid).

The number of total ammonia detections decreased even more dramatically in Year 2 of Full Restoration (6.8%), but increased in the subsequent two years, with detection levels ranging from 11.1% (Year 3) and 15.9% (Year 4; Parsons 2012; Figure 9). However, mean concentrations continued to decrease 73%-88% after restoration from 1.26 mg/L Pre-Restoration to 0.34 mg/L during Year 1, 0.15 mg/L during Year 2, and 0.52 mg/L during Year 3 (ibid). Interestingly, the number of detections was lowest during the passive restoration period. Overall, then, while detections decreased between pre- and passive-restoration periods, they increased between Passive (4.6%) and Full Restoration (11.9%; Contingency Table, df=2, Pearson Chi-Square = 11.438, P= 0.003; ibid; Figure 9). The increase in ammonia detections from passive to full restoration could be attributable to restoration-related changes: increase in ammonia following breakdown and conversion of decomposing organic matter into organic and inorganic nutrients and flushing of ammonia from soils into overlying waters.

One interesting caveat to this hypothesis is that ammonia detections increased in all of the Study Areas following restoration, even those natural marshes that were distant from the restored wetland (Parsons 2012). The number of detections in natural marshes jumped from 3.9% of the samples pre-restoration to 10.3% in Year 1, while detection frequencies in natural marshes during the passive restoration period were roughly equivalent to Pre-Restoration (4.0%; ibid; Figure 9). During Years 1-4, ammonia detection in natural marshes totaled 11% of samples, a 182% increase (Contingency Table, df=2, Pearson Chi-Square = 6.2, P=0.044; ibid; Figure 9). As with the restored wetland, while the number of detections increased, the estimated average ammonia concentrations for natural marshes appeared roughly equivalent pre-restoration and post-restoration (ibid).

The overall increase in the detection, if not concentrations, of total ammonia within both the restoration Project Area and natural marshes -- some of which are distant from the Project Area -- suggests that the increase in ammonia detections documented post-restoration does not all result from the effects of restoration. Similar increases in ammonia detections were observed at other sampling sites in the Tomales Bay during recent years (Rob Carson, TBWC, *pers. comm*.).

One possible explanation for the increase in ammonia detections, at least in Year 1, may have been the dry winter, which allowed tidal influence to extend further upstream due to the lack of a strong countering force from freshwater flows. Recent research on salinity intrusion associated with sea level rise on the East Coast found that intrusion of even weakly saline waters into formerly freshwater tidal areas -- areas where tides affect rise and fall of water level, but do not affect salinity -- mobilized ammonia into overlying waters, causing a net efflux or outflow from the system. In these areas, ammonium, phosphate, and silicate fluxes or transport from the system increased by 20 to 38% (Joye *et al.* undated). Most of this increase probably results from cation exchange of the strongly ionic salt in seawater (sodium chloride or NaCl) for the more weakly ionic ammonium (NH4-; Craft *et al.* 2009), but ammonia may also be produced through increased mineralization of organic matter in soils.

Salinity data collected in Year 1 showed increases in salinity not only in the restoration Project Area understandably, but in natural marshes, so this supports the potential for increased upstream tidal influence to have caused biogeochemical changes that resulted in more frequent ammonia detections (Parsons 2010). In Years 2 and 3, wetter conditions drove down salinities below Pre-Restoration median levels by as much as 9-11 ppt, so higher ammonia detection frequencies in Years 2 and 3 relative to Passive Restoration periods are harder to explain (Parsons 2012).

Because levees essentially precluded or minimized export of pollutant loads from the ranch pastures, with full levee removal, the contribution of the restored wetlands to downstream loading would be expected to increase, even if concentrations within the wetlands dropped dramatically. For example, for fecal coliform, estimated geometric mean or median loading rates jumped 274% from 57.5 mpn/s during Pre-Restoration to 242.8 mpn/s during Passive Restoration and to 215.2 mpn/s during Full Restoration (Parsons 2012). Conversely, estimated arithmetic mean loading rates, which were higher than median values, seemingly decreased from 249,389 mpn/s during Pre-Restoration to 52,716.9 mpn/s during Year 1, to 33,657.5 mpn/s during Year 2, and to 34,036 mpn/s during Year 3 (ibid). This suggests that peak loading during high flow events may have dropped relative to Pre-Restoration conditions, but the average loading rate for fecal coliform during non-storm flows has apparently increased. For nitrates, loading increased between Pre-, Passive, and Full Restoration, with estimated means climbing from 0.60 mg/s pre-restoration to 1.17 mg/s during Passive Restoration and to 1.44 mg/s after restoration (ibid). In comparison, within natural marshes, loading rates decreased for fecal coliform and remained equivalent for nitrates during the post-restoration sampling period (ibid).

Conclusions

Some improvement in water quality conditions were expected immediately following restoration due to decreases in residence time for leveed waters. However, these improvements were expected to be tempered to a large degree initially by pulses in sediment and nutrients from re-working of exposed soils by tides, floods, and decomposition and mineralization of pasture vegetation, with variables such as pH and dissolved oxygen (D.O.) responding accordingly to the resulting flux in nutrients. During the first several years after restoration, the speed with which conditions improved within the Project Area for variables such as dissolved oxygen and nitrate and fecal coliform concentrations far exceeded our expectations. Expected issues as discussed above such as large increases in turbidity and temporary decreases in dissolved oxygen did not materialize or were not as dramatic as anticipated.

At least initially, some of this may have partially resulted from the fact that Year 1 or WY 2009 was a dry year, and few large storms occurred that would have contributed to reworking of this evolving landscape, even though some of the few larger storm events that did occur were captured. Even in the second year, which was much wetter, there were no overbank flooding events. Year 3, which was also wet, did have at least one, very brief overbank flooding event. Since then, both Years 4 and 5 have been much drier. Despite the lack of storms, reworking of the landscape did occur, largely due to reintroduction of tidal action, with shoals evident at the mouth of newly created tidal channels due to sediment efflux from the marsh (KHE 2009). , Despite the vagaries of climatic conditions, water quality parameters have shown significant, positive improvements in conditions between Pre- and Full-Restoration phases, with only a few exceptions. Dissolved oxygen levels increased 16%, while nitrate, ammonia, phosphate, phosphorous, and fecal coliform levels decreased at least 23%, with some of these parameters falling quite substantially (phosphates; 900%). With restoration, vegetated waterways within the East Pasture of the Project Area were no longer regularly dredged, which increased organic matter available for breakdown by bacteria that depleted oxygen in the relatively stagnant waters within the dairy ranch. Elimination of this management practice, coupled with a decrease in residence time of waters with removal of levees and tidegates, has increased oxygen levels. Nitrate, ammonia, fecal coliform, and, to a lesser extent, phosphate levels were expected to drop with removal of cows and discontinuation of manure spreading. After only a short time (five years), at least some of the parameters, including temperature, ammonia, phosphates, phosphorous, and seemingly at least median concentrations of nitrates were actually statistically equivalent to levels in natural marshes.

Restoration appears to be the primary factor driving changes in levels of dissolved oxygen, turbidity, and and loading rates of nitrates and fecal coliform in the Project Area since the levees were breached. Dissolved oxygen levels between Study Areas showed very dissimilar patterns following restoration, as did turbidity, with sharp changes in turbidity levels after levee breaching within the restored wetland seemingly unrelated to trends in natural marshes (Figure 10). However, the situation is not as clear cut for some of the other variables. Restoration also appeared to affect fecal coliform levels, but, immediately after restoration, the pattern and even the range of fecal coliform values appeared similar to those in natural marshes, even if there has not been total convergence from a statistical standpoint (Figure 11). Similarly, reintroduction of tidal flow and cessation of freshwater impoundment dramatically altered salinities in the former dairy ranch, but once the levees were breached, the pattern of salinity changes strongly mirrored those in reference wetlands, although the range of salinity values remains quite disparate between Study Areas due to fundamental hydrologic differences (Figure 12).

As noted earlier, pH and temperature dropped following restoration in the Project Area, but they also dropped in natural marshes, as well. In the case of pH, the rate of decline was identical between the Study Areas, strongly suggesting that external factors such as the volume and distribution of rainfall and its effect on lower-pH freshwater inflow may have more influence on pH in the restored wetlands than any restoration-related factors such as release of acids from decomposition of organic matter or oxidation of soils (Figure 13). In year 5, pH in the restored wetland begins to show some convergence with natural marshes (Figure 13). The same appears true for temperature (Figure 14), the levels of which have also already fully converged with those of reference wetlands. .

The differences in the magnitude of change between the Project and Reference Areas for variables such as nitrates, total ammonia, and particularly phosphates suggests that both restoration and climatic factors may be playing a role in shaping current water quality conditions in the restored wetland. Others factors may also have an effect, including non-point source run-off: reductions in nitrates might even have been dramatic had mean levels not been artificially inflated by a non-point source run-off that flows into the created freshwater marsh discussed earlier.

Ultimately, restoration of more than 600 acres of historic floodplain/marshplain is expected to not only restore water quality conditions within the Project Area, but Tomales Bay itself. Therefore, one of the most important indicators of the success of this restoration project will be changes in concentrations and, even more importantly, loading between upstream and downstream sampling locations. As was expected, during the first four years after restoration, loading rates of pathogens and presumably nitrates actually increased in the Project Area relative to pre-restoration conditions, because, prior to levee removal, the pastures had either no direct connection to Lagunitas or other creeks (East Pasture) or only muted tidal connection (West Pasture) and, therefore, were only very infrequently in a position to contribute to downstream “loading.” It is likely that watershed-scale benefits will take time to be realized due to the continuing evolution of the restored wetland.

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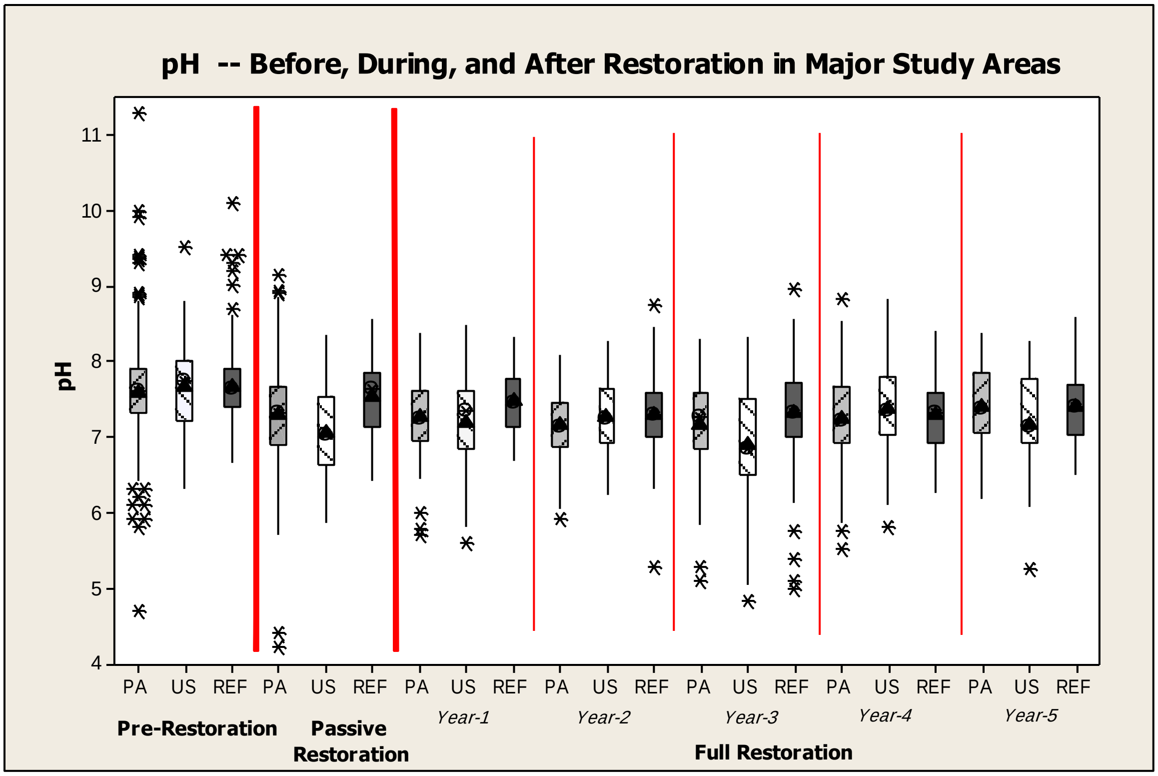

Figure 6. Average, median, and other summary statistics for pH for Study Areas Pre-Restoration, during Passive Restoration, and in the first five years of Full Restoration.

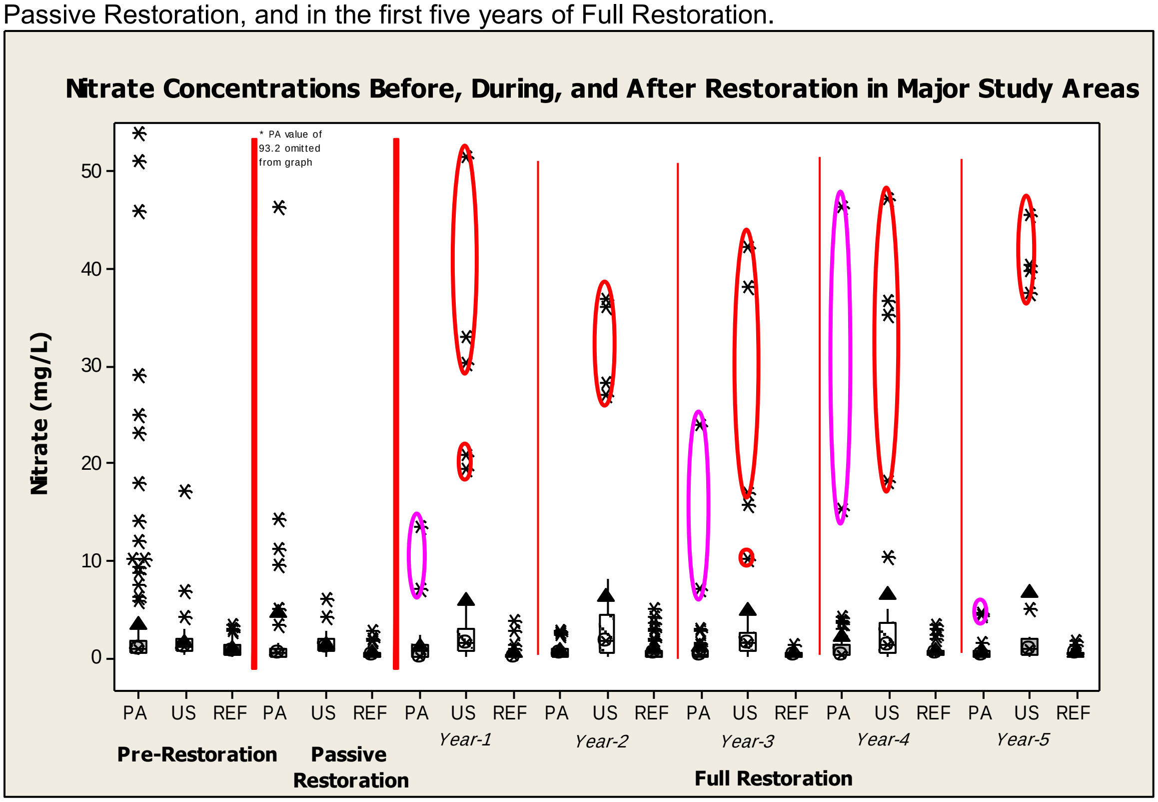

Figure 7. Average, median, and other summary statistics for nitrates (NO3-) for Study Areas Pre-Restoration, during Passive Restoration, and in the first five years of Full Restoration.

Light-shaded grey boxplots indicate first and third quartiles (25%, 75%). Lines indicate 10^{th} and 90^{th} percentiles. Medians are indicated by diagonal-hatched circle, with means designate by black triangles.

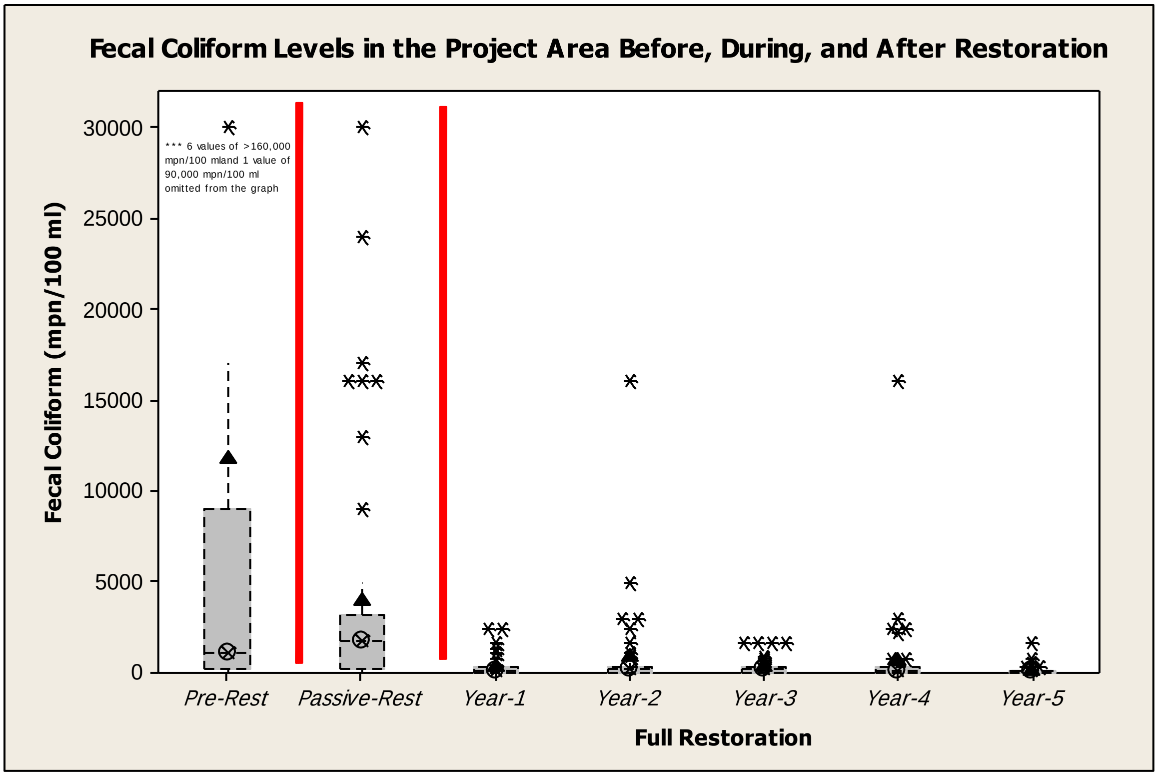

Figure 8. Average, median, and other summary statistics for fecal coliform concentrations for the Project Area Pre-Restoration, during Passive Restoration, and in the first five years of Full Restoration.

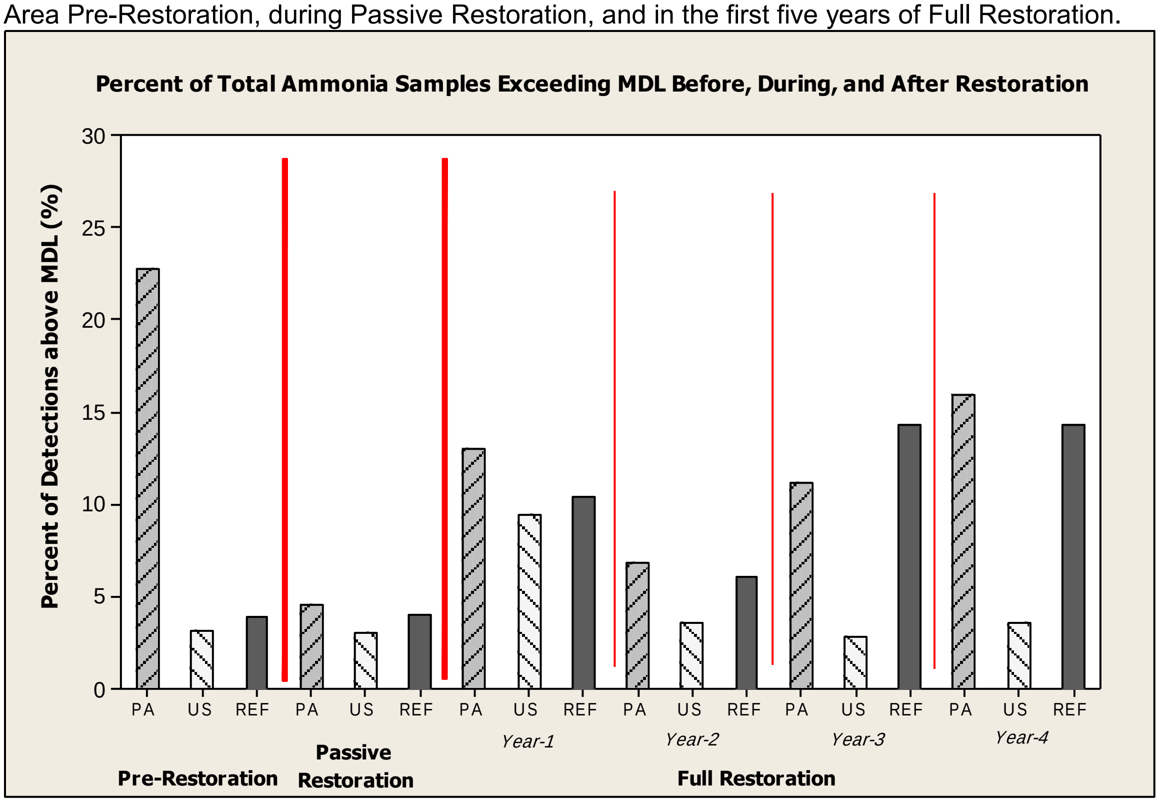

Figure 9. Percent of samples above Total Ammonia concentration detection limits for Study Areas Pre-Restoration, during Passive Restoration, and in the first four years of Full Restoration. Total ammonia was not sampled in Year 5.

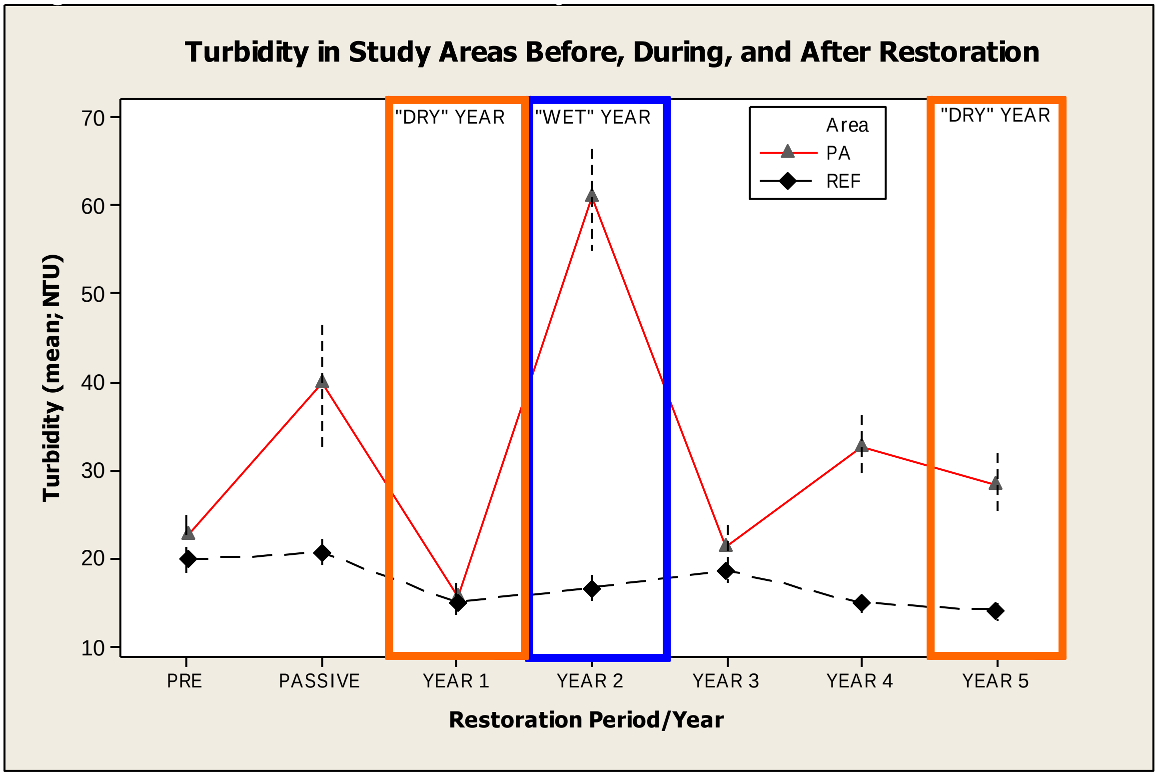

Figure 10. Turbidity in Study Areas Pre-Restoration, during Passive Restoration, and in the first five years of Full Restoration. Dashed lines represent S.E. of the mean.

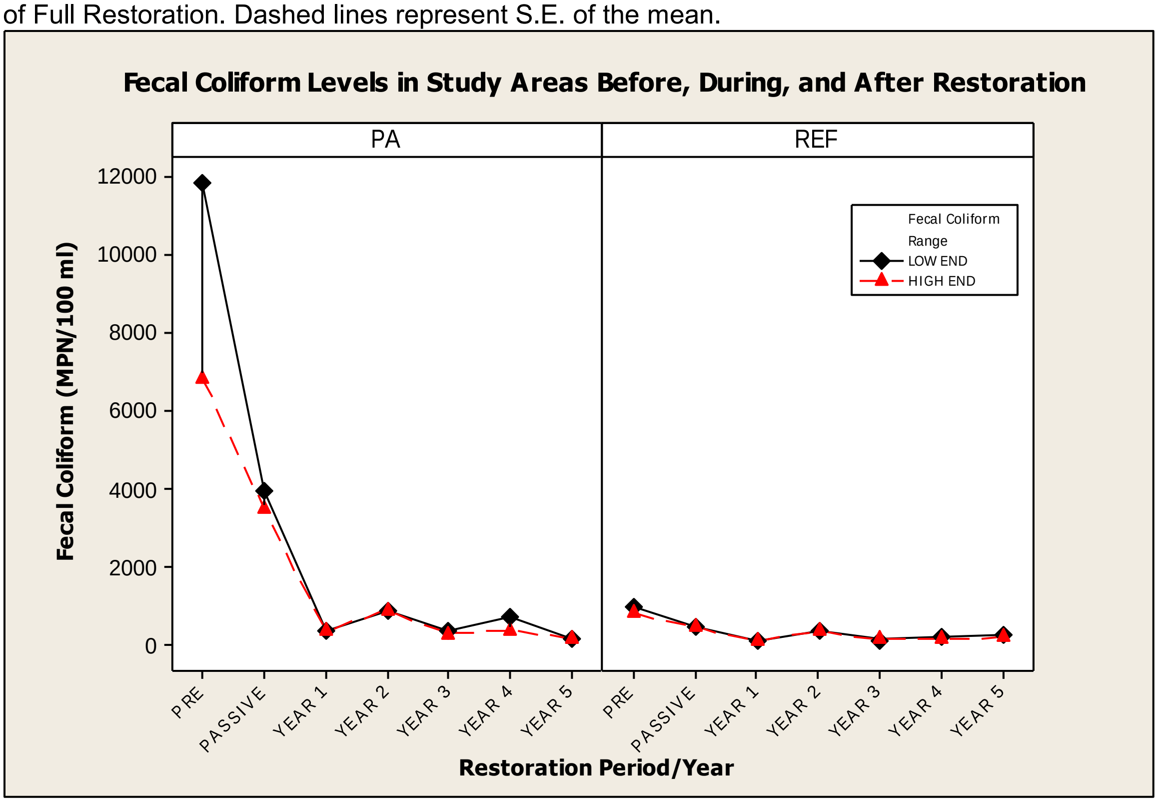

Figure 11. The upper and lower range of fecal coliform levels in Study Areas Pre-Restoration, during Passive Restoration, and in the first five years of Full Restoration.

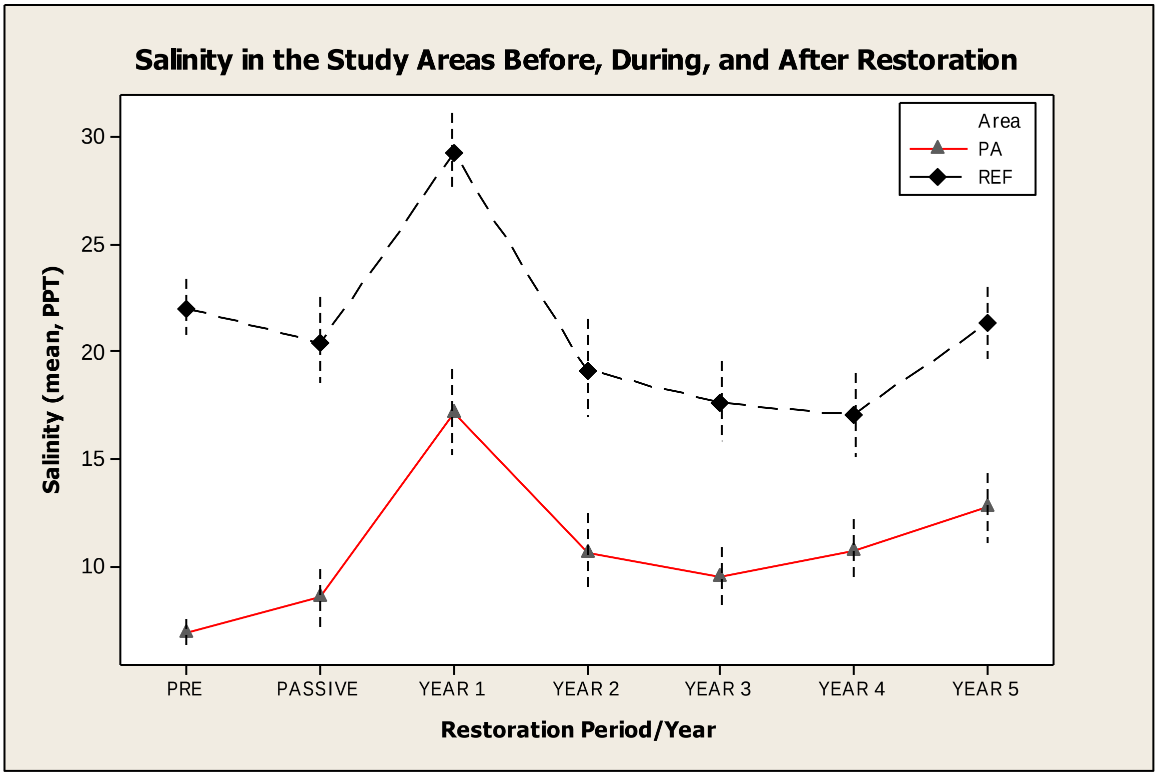

Figure 12. Salinity in Study Areas Pre-Restoration, during Passive Restoration, and in the first five years of Full Restoration. Dashed lines represent S.E. of the mean.

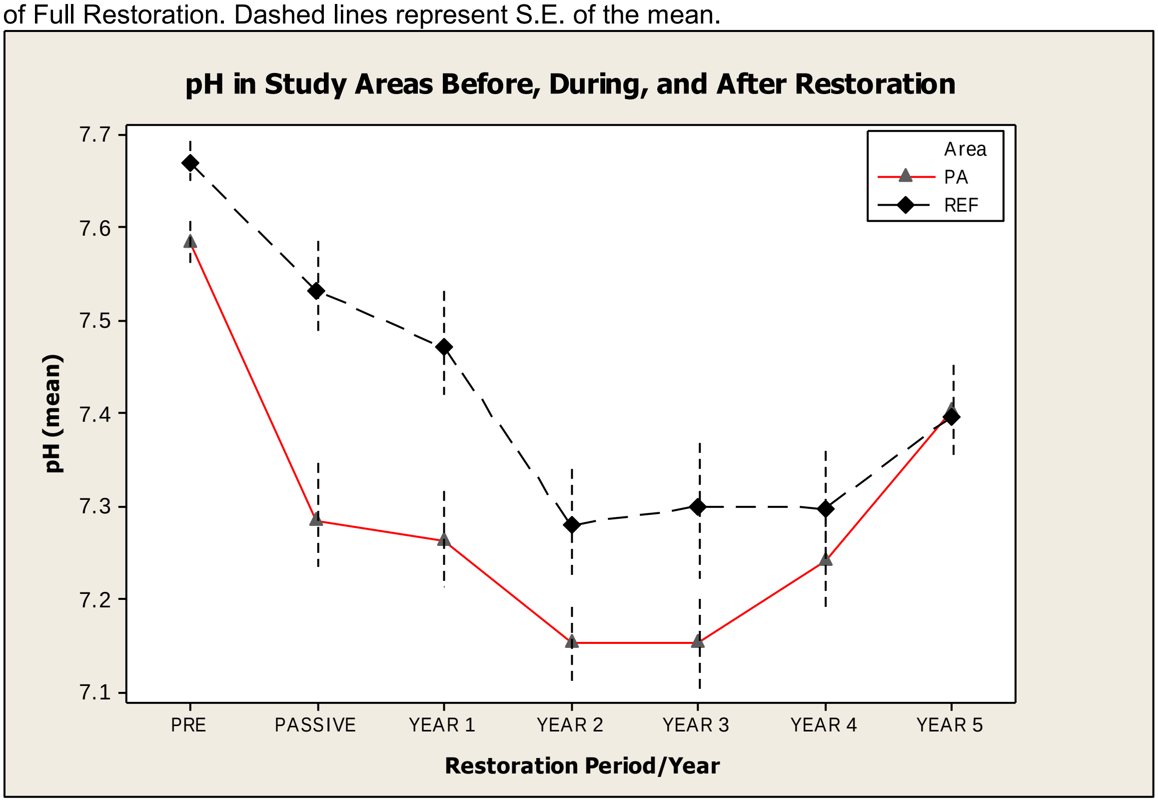

Figure 13. pH in Study Areas Pre-Restoration, during Passive Restoration, and in the first five years of Full Restoration. Dashed lines represent S.E. of the mean.

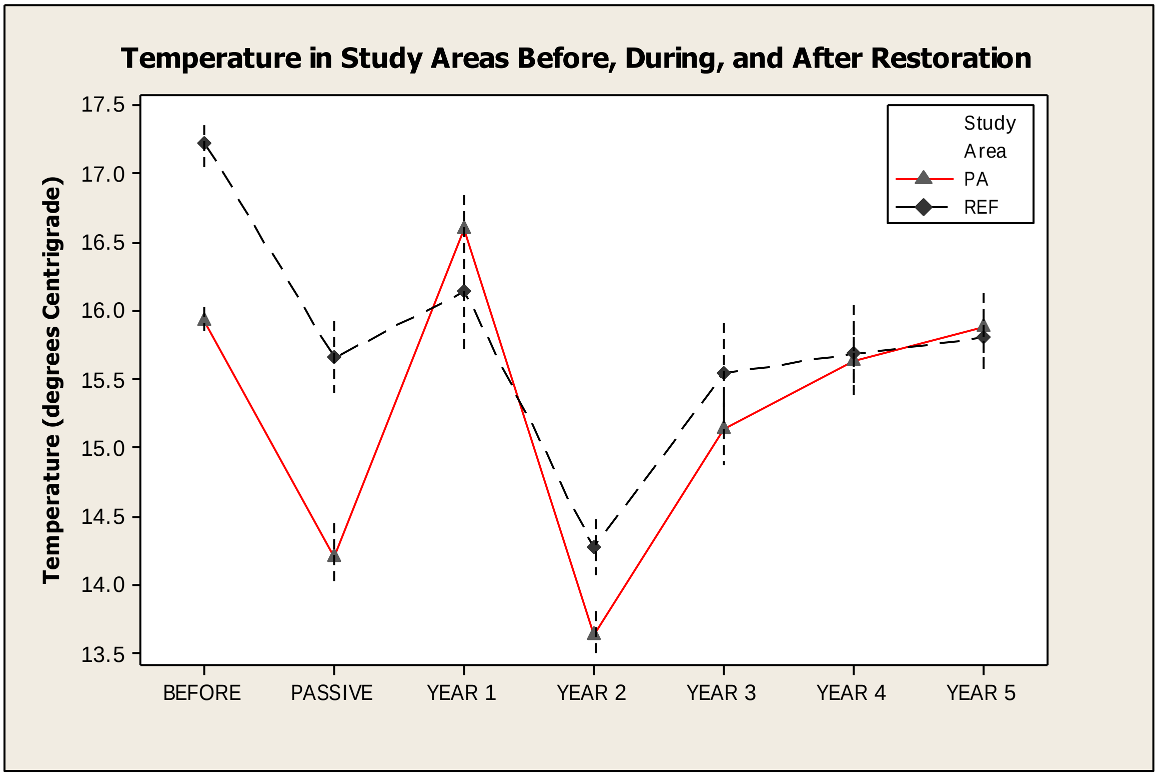

Figure 14. Temperature in Study Areas Pre-Restoration, during Passive Restoration, and in the first five years of Full Restoration. Dashed lines represent S.E. of the mean.

6 Results and Discussion: Changes in Zooplankton, Benthic Invertebrate, and Fish Communities

Adapted from Parsons, L.S., Sanders, L. Ryan, A. and Reichmuth M. Changes in the food web linked to restoration effort intensity and watershed conditions. Natural Resources, 2015, 6, 344-362

Introduction

The most widely studied indicators of change following wetland restoration are often parameters in which change is readily visible—changes in vegetation communities or bird populations. However, changes in higher trophic orders of fish and birds are often dependent on changes in their prey base following restoration. In a wetland ecosystem, this prey base includes pelagic macro- and benthic invertebrates, as well as smaller fish. Successful restoration requires restoration of the entire food web, not just vegetation and higher level organisms.

Tidal wetland restoration efforts to date appear to have been relatively successful in rehabilitating fish populations. Fish abundance, species composition, and species diversity can change dramatically in the first few years after restoration, with densities and species richness quickly resembling conditions observed in reference or natural marshes (Williams and Zedler 1999; Desmond *et al.* 2002; Havens *et al.* 2002; Warren *et al.* 2002; Raposa and Roman 2003; Konisky *et al.* 2006; Kimball and Able 2007). Some restored marshes have even supported higher fish densities and species richness than natural marshes (Desmond *et al.* 2002; Havens *et al.* 2002; Buchsbaum *et al.* 2006) and developed a species assemblage typical of reference marshes within 5- to 13 years (Warren *et al.* 2002; Desmond *et al.* 2002; Kimball and Able 2007). There have been projects where fish use either did not compare favorably with reference marshes (Chamberlain and Barnhart 1993) or actually declined with restoration (Buchsbaum *et al.* 2006; Raposa 2008). However, in these instances, either the types of habitat present in the natural marsh did not correspond well with those created in the restored site, or the restoration reduced potential fish habitat. Rather than being driven by restoration, age of restoration, or native marsh status, several projects found that differences between fish assemblages depended on the marsh channels' physical structure (e.g., width, depth, slope of bank, marsh elevation), as well as hydrologic and environmental conditions, including hydroperiod, temperature, dissolved oxygen, salinity, and flow discharge (Minello and Webb 1997; Williams and Zedler 1999; Desmond *et al.* 2002; Havens *et al.* 2002; Raposa and Roman 2003).

Marsh status (restored vs. native) and age of restoration has played a more critical role in benthic invertebrate community development. Many studies found differences in benthic invertebrate communities between restored and reference marshes even 10- to 20 years post restoration (Minello and Zimmerman 1992; Minello and Webb 1997; Talley and Levin 1999; Ferguson and Rakocinski 2008). Almost all prior studies have shown differences in community structure between constructed and natural marshes persisting within the first few years after restoration (Cammen 1976; Minello and Zimmerman 1992; Levin *et al.* 1996; Scatolini and Zedler 1996; Bolam *et al.* 2006). There are some exceptions: a restored marsh in southern California supported equivalent levels of macrofaunal densities, species richness, and diversity as that of the nearby reference marsh in 19 months, although species composition remained different (Moseman *et al.* 2004). A strong correlation appears to exist between time since restoration and many benthic parameters, with older marshes showing more structural similarities to natural marshes (LaSalle *et al.* 1991; Minello and Zimmerman 1992; Posey *et al.* 1997; Talley and Levin 1999; Craft and Sacco 2003; Toft *et al.* 2003; Havens *et al.* 2002; Cordell *et al.* 2008). Within five to 10 years, some marshes reached equivalence of invertebrate communities with natural marshes or even surpassed them in some parameters such as species diversity or density (Talley and Levin 1999; Desmond *et al.* 2002; Havens *et al.* 2002; Cordell *et al.* 2008).

Despite these successes, most studies have found that differences between natural marshes and their constructed counterparts persist long after restoration is implemented. At created marshes in Mississippi, community structure still differed between created and natural marshes even 27 years after construction, although species diversity and evenness was higher at created marshes (Ferguson and Rackocinski 2008). In some New England marshes, the high marsh snail, *Melampus bidentatus*, took two decades to reach natural marsh densities (Warren *et al.* 2002). This lack of convergence is attributed to a number of factors, including proximity to natural marshes that act as a source for invertebrate recolonization, elevation, belowground biomass of plants, macro-organic matter (MOM), soil organic C, total N, and substrate, with MOM sometimes considered one of the most important (Cammen 1976; Minello and Zimmerman 1992; Talley and Levin 1999; Havens *et al.* 2002; Craft and Sacco 2003; Moseman *et al.* 2004; Bolam *et al.* 2006; Ferguson and Rakocinski 2008).

In 2008, the National Park Service (Park Service) finished a two-year project to restore more than 223 hectares (ha) of former tidal marsh known as Giacomini Wetlands at the head of Tomales Bay in Marin County, California (Figure 1). As part of the project, the Park Service has been conducting pre- and post-restoration monitoring of hydrologic and ecological variables. For several years prior to restoration, we monitored zooplankton, benthic invertebrates, and fish to determine species assemblages and relative abundance while the Project Area was a dairy ranch. Here, we present comparative results from the first four years after restoration to evaluate how assemblages and abundance of invertebrates and fish have changed in response to the rapid conversion of pasture to salt, brackish, and freshwater marsh. We hypothesized that the structure of invertebrate and fish communities would dramatically change following restoration, but that it would not have fully converged yet with that of natural marshes. Understanding the structure of the prey base available in the marsh before and after restoration will help us to better understand changes in use of the restored habitat by birds and larger fish species and the ecological contribution of this restoration project to a complex watershed ecosystem.

Materials and Methods

Study Areas

The Study Areas are located in Tomales Bay and Estero de Limantour, both of which are located on the central California Coast in Marin County just north of San Francisco, USA. Tomales Bay proper is a 28 km^2 shallow, highly unidirectional, Mediterranean-type, coastal estuary (TBWC 2003). Estero de Limantour is very similar to Tomales Bay, but smaller in size (3.9 km^2; Anima 1991). Despite its proximity to the highly urbanized San Francisco Bay area, the Tomales Bay watershed remains largely agricultural, supporting a number of beef and dairy cattle ranches (TBWC 2003), The Estero de Limantour watershed was once more extensively farmed, although these activities have been scaled back now that watershed lands are part of the national park system (Livingston 1994).

The Study Areas included the Project Area (PA; labeled “a” on Figure 1), Reference Areas (REF), which are natural tidal marshes in Tomales Bay and adjacent watersheds (Walker Creek Marsh (labeled “c” on Figure 1), Limantour Marsh (labeled “d” on Figure 1), and the Undiked Marsh (labeled “b” on Figure 1). In addition, for some variables, additional sampling was conducted in sites upstream of the Project Area (Upstream Areas; US) to determine whether conditions in the leveed dairy ranch were actually more similar to upstream freshwater or mildly brackish creek areas than natural tidal marshes.

The Project Area occurs in the very southern portion or the headwaters of Tomales Bay (NPS, *in prep*.). During establishment of a dairy ranch in 1946, this area was leveed off from adjoining natural marshes to the north and from one of the watershed’s largest sources of freshwater inflow, Lagunitas Creek, which bisects the Project Area into two areas that are approximately 142 ha (East) and 81 ha (West) in size (NPS, *in prep*.). Despite being leveed, the former tidal wetlands had not subsided in elevation relative to adjacent natural marshes more than 0.3 to 0.6 meters (m; KHE 2009).

Three reference wetlands represented the natural variability present in nearby tidal marshes that are somewhat similar to the Project Area in terms of site and watershed size, marsh age, land use, and marsh formation history, although hydrologic conditions may be slightly different due to variation in freshwater and tidal inflow. During the late 19th and early 20th centuries, logging and agricultural development caused a substantial increase in sedimentation within Tomales Bay, and marshes were formed or enlarged at the mouths of Lagunitas and Walker Creeks (the Undiked Marsh and Walker Creek

Marsh, respectively; PWA *et al.* 1993; Allen 2005 *in* Parsons 2005). The Undiked Marsh and Giacomini Wetlands once formed an integrated wetland complex, until levee construction divided the two areas. Limantour Marsh, the third reference site, is adjacent to the Pacific Ocean in a different watershed and has a much smaller watershed size and a slightly different formation and land use history: it was formed in the 1960s after an upstream dam was installed (Allen 2005 *in* Parsons 2005).

Restoration Project Background

The National Park Service bought the Giacomini Ranch dairy for wetland restoration in 2000 (Parsons and Ryan, *in prep*.). Extensive changes in watershed condition including huge influxes of sediment following logging in the late 1800s precluded re-creating historic conditions, so, instead, the Park Service focused on restoring natural hydrologic and ecological processes and functions (Parsons and Ryan, *in prep*.). After seven years of planning, restoration of approximately 223 ha was implemented in 2007 and 2008 (Parsons and Ryan, *in prep*.). Because of constraints related to subsidence (e.g., reductions in vertical elevations after long-term leveeing) or to adjacent infrastructure, many other tidal marsh restoration projects in California (e.g., South Bay Salt Ponds, Hamilton Wetlands, Bolsa Chica) involve elaborately phased designs, construction of interim levees, extensive excavation or fill, or placement of dredgespoil materials. In comparison, restoration efforts for this project were relatively minimal, mainly involving removal of agricultural infrastructure, including 4 kilometers (km) of levee, tidegates, and culverts (Parsons and Ryan, *in prep*.). A linear system of drainage ditches was filled with levee material, and 3.7 km of new tidal channels were created (Parsons and Ryan, *in prep*.). Other than channel creation, the only other major excavation was shallow grading of 6.5 ha of uplands into intertidal zones and floodplains (Parsons and Ryan, *in prep*.).

Monitoring Design

The framework for Park Service's long-term monitoring program was a modified, asymmetrical BACI (Before-After-Control-Impact) sampling approach (Stewart-Oaten *et al.* 1986; Underwood 1991; Smith 2002). This sampling design is well-suited to restoration monitoring in that it evaluates conditions in the Project Area (PA; "impact area") before and after restoration and uses several natural marshes or Reference Areas (REF; "control") to differentiate between effects of restoration and other factors and to determine similarity of the restored marsh to natural marshes. Monitoring took place for seven to eight years depending on the variable, with three to four years of those years generally before restoration ("Before" 2005-2008) and three to four of them after restoration ("After;" 2009 – 2012).

Sampling Methodology: Zooplankton

Invertebrate communities were sampled at approximately 28 sites (median) twice annually: in spring (April/May) and fall (October/November) to coincide with spring peaks in productivity and fall shorebird migration. The number of sampling sites varied due to changes in aquatic habitats present due to restoration, seasonal drying up of some features such as creeks, and other factors, with the number of sites ranging from 10 to 37 over this period. Approximately 12 of these sites were in the Project Area, while the remainder was either in Control or Reference Areas (10) or Upstream Areas (US; 6). Sampling was conducted in a variety of aquatic habitats, including sloughs and creeks, drainage ditches, ponds, and shallowly ponded areas (e.g., flooded pannes). Zooplankton sampling characterized the aquatic invertebrate community within the water column. Vertical tows were conducted using a 63 *u*m mesh plankton Nitex net (Turtox; Wildlife Supply Company; Saginaw, Michigan) with a weighted bottom. Two tows were conducted at each sampling site. Both tows were poured into a 250-ml jar and fixed with 10 percent formalin in the field. The samples were sent to a biologist (Anne Slaughter, San Francisco State University, Romberg-Tiburon Center for Environmental Studies, Tiburon, CA) with expertise in zooplankton taxonomy for identification to either genus or species level. Density data were obtained by calculating the volume of water towed: (Length of tow)(π)(Radius of net opening)^2.

Sampling Methodology: Benthic Invertebrates

Benthic invertebrate samples were collected at low tide once annually in November or December to correspond with fall and early winter shorebird migration. Sampling was conducted at approximately 27 sites (median) -- 14 in the Project Area and 13 in Control or Reference Areas. The number of sampling sites did vary slightly each year due to changes in aquatic habitat features following restoration and other factors, ranging from 23 to 30. Starting in 2007, three samples of the top 5 cm of sediment were collected at each sampling site using a benthic corer (10-cm diameter). Initial sampling in 2005 and 2006 involved more subsamples (up to six) and deeper core depths (up to 15 cm), but sampling depth was reduced in the third year, because, in most cases, the majority of infauna is located within the upper 2- to 5 cm of the sediment surface (LaSalle *et al.* 1991; Levin *et al.* 1998). These subsampling cores were then pooled for sorting and identification. For density analysis, abundances were adjusted to reflect a standardized reporting unit (density/cm^3), which represented a fraction of the total sediment volume sampled in all years. For analysis of species richness and associated indices, the initial sampling years were removed the data, as information available on samples did not allow for possible correction of data using other methods (e.g., rarefaction curves). Low abundance taxonomic groups with either combined with others or deleted, which resulted in 73 groups for analysis.

Samples were rinsed with water within 3-4 hours of collection and left overnight in a preservative of 37% formaldehyde solution and water. After 24-48 hours, the samples were rinsed a second time and placed in a 70+% Ethanol solution for shipping to a biologist with expertise in benthic invertebrate taxonomy (Susan McCormick, Auburn, CA), who sorted and identified to the lowest taxonomic level possible under 8X magnification (i.e., order, genus, or species).

Sampling Methodology: Fish

Fish sampling occurred twice annually: in early summer (June) and in fall (October). Fish sampling dates were intended to coincide with other ecological parameters such as migration of salmonids. Sampling was conducted at approximately 33 sites (median) -- 16 in the Project Area; 14 in Reference Areas; and 3 in Upstream Areas. The number of sampling sites did vary slightly each year due to changes in aquatic habitat features following restoration and other factors, ranging from 23 to 30. Fish sampling within tidal and diked wetlands presented special challenges due to the diversity of aquatic habitats (e.g., small and large creeks, ponds) and types of fish species present (i.e., demersal or bottom-dwelling species, fast-moving species, species present only at certain times of year).

Block nets and beach seines were used in smaller creeks within Study Areas. In wider creeks, a beach seine without blocking net was employed, although this method did not allow for reliable estimation of fish densities. Blocking nets and seines employed a multiple-pass depletion method in which seining was performed until all fish were caught or until numbers caught on three successive seines continued to decrease appreciably after each "pass." Dimensions of the blocking and seine nets varied depending on the size of the channel. Net length ranged from 15 m to 45 m, with mesh size ranging from 0.31 cm to 0.625 cm. Fish were placed into aerated buckets for identification and counting and were released upon completion of measurements at the capture site.

Sampling Methodology – Water Quality

During all zooplankton, benthic invertebrate, and fish sampling events, information was collected on water chemistry including water temperature (°C), salinity (parts per thousand; ppt), dissolved oxygen (mg/L), pH, and water depth (cm) using several models of hand-held YSI multi-parameter instruments (YSI Inc., Yellow Springs, Ohio). Water depth during each sampling event was measured using a rod with markings in centimeters (cm).

Data Analysis

Total densities and species richness, diversity, and composition for zooplankton and benthic invertebrate data were compared between Pre- and Post- Restoration periods in the Study Areas and between the Project Area and Reference Areas. Pre-Restoration analysis of zooplankton data also incorporated areas upstream of the Project Area (US): Because the restoration project objective is to restore the historic tidal marsh to conditions similar to those of other salt marshes, the US comparison was dropped from post-restoration analyses.

For univariate variables, data were analyzed using General Linear Model ANOVA (GLM; MiniTab v15.1, MiniTab Inc., State College, PA) if they met assumptions of parametric statistical methods. Environmental variables were included when appropriate as covariates. We also assessed whether results from successive sampling events were temporally autocorrelated using the ACF (Autocorrelation Function) and PACF (Partial Autocorrelation Functions): data did not show strong auto-correlation (all ACF and PACF >0.05). If data did not meet parametric assumptions, they were either transformed, or non-parametric statistics were used. Non-parametric statistical tests included Kruskal-Wallis and Mood Median Test (MiniTab v15.1, MiniTab Inc, State College, PA). Arithmetic means are presented with standard error (S.E.) of the mean (± S.E.).

We analyzed community data using several multivariate statistical analysis methods (Multi-response Permutation Procedure (MRPP), Indicator Species Analysis, and Non-Metric Multidimensional Scaling (NMS); PcOrd v5.3.1, MjM Software, Gleneden Beach, OR). NMS was used to evaluate differences in community structure in sampling sites between Pre- and Post- Restoration periods and between the Project Area, Reference Areas, and, if applicable, Upstream Areas. NMS was run using Relative Sørenson Distance, 50 runs with real data, 250 runs with random data, random starting co-ordinates, stability criterion 0.00005, and a maximum number of dimensions/ axes = 3. The ordination was re-run manually five times to ensure that final stress scores between each run remained relatively stable and did not exceed 20. NMS results were examined further by running a GLM ANOVA model on the primary axis variables to look for significant differences between sites in terms of treatment groups or Study Areas.

Evaluation of group differences for multivariate data were carried out using MRPP and Indicator Species Analysis, which detects species that “indicate” a priori groups by evaluating relative frequency and abundance among groups using a Monte Carlo randomization procedure to evaluate significance (PcORD v.5.3.1). Statistical analyses also incorporated diversity measures, including species richness (total number of species) and the Shannon-Weiner Diversity Index (H’), which evaluates species richness and the proportion of each species. Diversity was defined as $\mathrm{H'} = -\sum_{\mathrm{i}=1}^{\mathrm{s}} \mathrm{p_i} \ln \mathrm{p_i}$ where *S* is the total number of species in the community and p_i is the proportion of *S* made up of species *i*.

Sampling techniques for fish did not allow for calculation and comparison of total densities or other quantitative abundance measures. However, the same type of sampling approach (e.g., block net; beach seine, etc.) was generally used at each site during every sampling period, so numbers can be considered a comparison of relative abundances at sites during different restoration periods. Similar multivariate techniques were used to analyze these relative abundances.

Results

Zooplankton

Baseline Conditions: Prior to restoration, zooplankton communities in the Project Area, Reference Areas, and Upstream Areas differed in abundance and species richness, diversity, and composition. Densities differed between Study Areas and sampling events (GLM; df=2, F=1.9, P=0.04; log-transformed), with average densities higher in the Project Area (103,279 ± 26,141 (S.E.) indiv/ m^3) than in either natural marshes (64,233 ± 22,154 indiv/m^3) or Upstream Areas (25,811 ± 12,733 indiv/ m^3 ; Figure 15). Median densities were 21,046 indiv/m^3 in the Project Area, 10,823 indiv/m^3 in the Reference Areas, and 3,657 indiv/m^3 in Upstream Areas (Mood Median; df=2, Chi-Square=10.2, P = 0.006; Figure 15). In terms of environmental factors used as covariates, only pH appeared to be associated with differences in densities (F=4.1, P=0.05).

As with abundance, species richness and diversity indices also showed some interaction between Study Area and individual sampling events. Mean species richness averaged 9.2 ± 0.4 in Reference Areas and 8.4 in both the Project Area (± 0.4) and Upstream Areas (± 0.6; GLM, df=2, F=2.65, P=0.004). Shannon-Weiner Diversity Index (H') averaged 1.19 ± 0.06 in the Project Area, 1.32 ± 0.06 in Reference Areas, and 1.37± 0.09 in Upstream Areas (GLM, df=2, F=1.9, P=0.04).

Based on MRPP results, Reference Areas had a very different mix of zooplankton species than either the Project Area (P=0.0005) or Upstream Areas (P<0.0001), with the latter two being somewhat similar in species composition (P=0.12) despite the seemingly large habitat type differences expected between managed pasture and freshwater creek areas (MRPP, df=2, T=-6.22, P<0.0001). Multivariate ordination using NMS showed similar results with the largest separation in species assemblages being between the Project Area and Reference Areas, with Upstream Area sites being intermixed with Project Area ones (NMS, Final Stress=15.66, 3D solution, instab.=0.00005, 400 iter.; Figure 16a). Axes 1 and 2 cumulatively accounted for 55% of the 78% of the total variance accounted for by the data. Axis 2 of NMS differentiated species communities within the Project Area and Reference Area sampling sites, showing a strong salinity gradient between the two Study Areas (Kruskal-Wallis, df=2, H=13.01, P=0.001), while Axis 1 showed more of a separation within Study Areas (GLM, df=2, F=4.62, P=0.015; log-transformed).

Indicator Species Analysis suggested that, prior to restoration, Reference Areas were best separated from other Study Areas by *Acartia* spp. (Calanoida), *Pseudobradya* spp. (Harpacticoida), *Monocorophium* sp. (Corophiidae), *Nippoleucon* spp. (Cumacea), Cumaceans, and Nematodes (Monte Carlo, all P< 0.01) and, to a lesser extent, Cyclopoids, Harpacticoids, and Oligochaetes (Monte Carlo, all P<0.10). Diptera, Insecta, and Rotifera were the only taxa that appeared to distinguish the unrestored Project Area (Monte Carlo, all P<0.04).

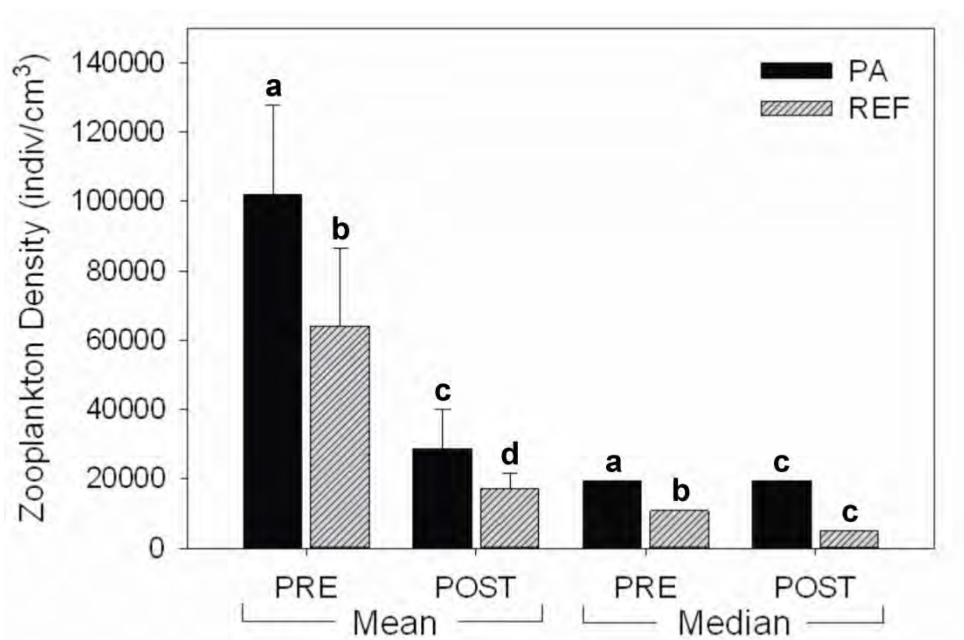

Figure 15. Mean and median densities of zooplankton
Data collected in the Project Area during Pre- and Post-Restoration. Bars on mean densities indicate standard errors. Statistically significant differences as indicated by GLM or Mood Median Test are shown with letters above the density bars.

Changes in Project Area with Restoration: Densities of zooplankton actually decreased following restoration of the Project Area (GLM, df=1, F=16.5, P<0.0001, log-transformed). As noted earlier, pre-restoration, densities of zooplankton averaged 103,279 ± 26,141 indiv/m^3, while, following restoration, densities averaged only 28,748 ± 11,357 indiv/m^3 (Figure 16a). These results appeared to be associated with changes in temperature of Project Area waters (F=10.9, P=0.001).

While zooplankton densities declined, species richness and diversity in the Project Area increased after restoration. The unrestored wetland had significantly lower numbers of zooplankton species (8.4 ± 0.4 than the restored one (11.5 ± 0.4; GLM, df=1, F=13.19, P<0.0001). The Pre-Restoration phase also supported communities with lower mean species diversity indices (H'=1.19 ± 0.06) than the Post-Restoration one (H'=1.42 ± 0.06; GLM, df=1, F=7.66, P=0.006).

Species composition shifted substantially in the Project Area after restoration (MRPP, df=1, T=-6,87, P<0.0001). Multivariate ordination showed separation between communities between the Pre- and Post- Restoration periods (NMS, Final Stress=13.87, 3D solution, instab=0.00005, 88 iter.). Axes 2 and 3 showed the highest incremental R^2 values, cumulatively accounting for 61% of the 78% of total variance accounted for by the data. Most of the group separation came from Axis 3 (Kruskal-Wallis, df=1, H=12.28, P<0.0001), which separated Pre- and Post- Restoration communities. Salinity appeared to most strongly differentiate between restoration phases on Axis 3.

A large number of species demonstrated statistically significant indicator values following restoration of the Project Area, suggesting a considerable shift in species composition or dominance over the restoration period. Based on Indicator Species analysis, Pre-Restoration communities in the Project Area were distinguished from Post- Restoration ones by taxa including *Diaptomus* spp. (Calanoida), *Pseudobradya* sp. (Harpacticoida), *Oncea* spp. (*Poecilostomatoida*), *Hydroida* sp. (Cnidaria), Amphipoda, Corophiidae, Gammaridae, Rotifera, Nematoda, Oligochaeta, and Gastropoda (Monte Carlo, all P<0.03). Only two species distinguished the restored wetland: Copepoda and *Daphnia* spp. (Cladocera; Monte Carlo, all P<0.02).

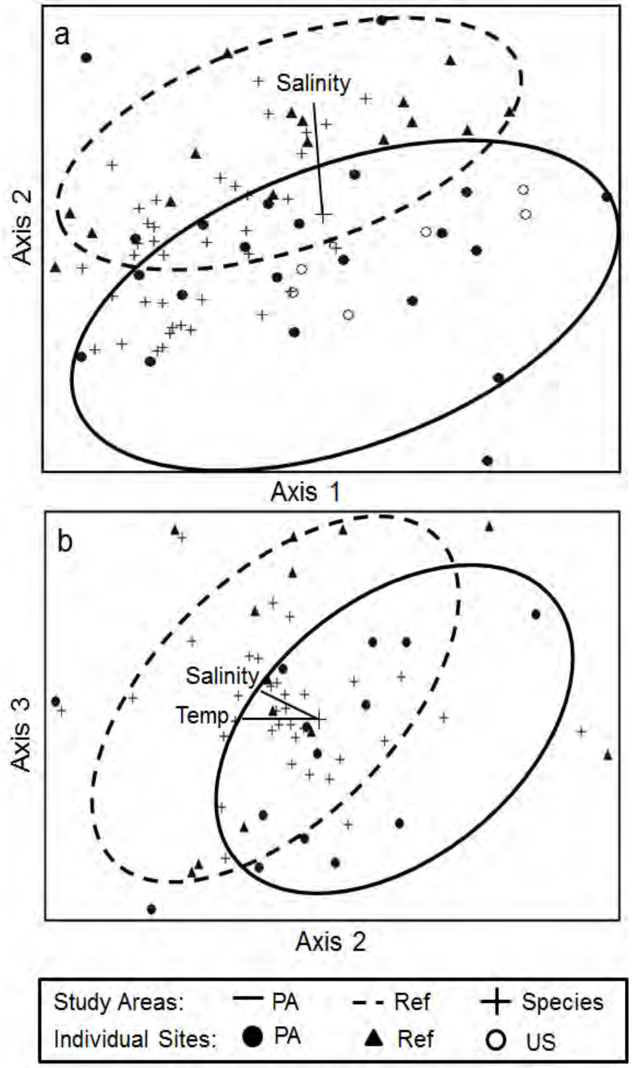

Figure 16. Zooplankton NMS
Three-dimensional (3D) statistical ordination of zooplankton species assemblages and environmental variables using Non-Metric Multi-Dimensional Scaling between Study Areas for a) Pre-Restoration with joint plot (Final stress = 15.66) and b) Post-Restoration with joint plot (Final Stress = 12.33). Salinity accounted for some of the separation between Study Areas during Pre-Restoration period, while salinity and temperature differentiated Study Areas Post-Restoration. Pre-Restoration period includes comparison with upstream freshwater sites (US), as wetlands in leveed ranch were largely freshwater, but this comparison was omitted from post-restoration analyses

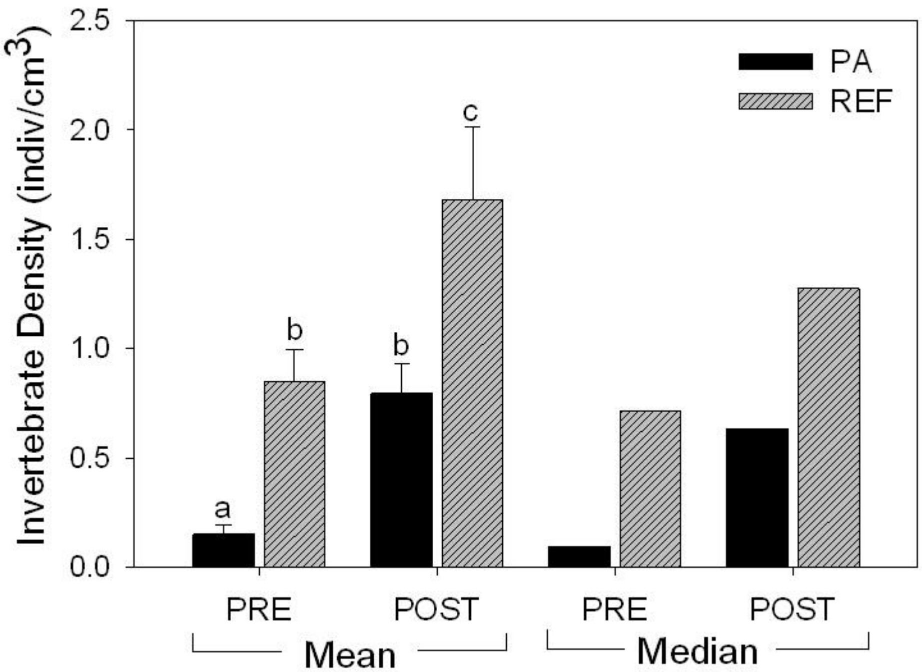

Figure 17. Mean and median benthic invertebrate density
Data collected in the Project Area and Reference Area Pre- and Post-Restoration. Bars on mean densities indicate standard errors. Statistically significant differences as indicated by GLM are shown with letters above density bars

Convergence with Natural Marshes: Following restoration, mean zooplankton densities continued to differ between Study Areas and sampling events (GLM, df=7, F=3.05, P=0.005; log-transformed for analysis; Figure 15). Mean densities were still higher in the restored Project Area (28,748 ± 11,357 indiv/ m^3) than in Reference Areas (17,273 ± 4,171 indiv/ m^3) despite declines in zooplankton abundance in the Project Area after restoration. However, densities also dropped significantly in natural marshes after 2008 (GLM, df=1, F=9.94, P=0.002; Figure 15). Of environmental variables, water temperature had the strongest relationship with Post-Restoration densities (F=17.4, P<0.0001). Median densities in the Project Area (4,990 indiv/ m^3) and Reference Areas (3,567 indiv/ m^3) approximated each other more closely (Mood Median; df=1, Chi-Square=2.29, P= 0.13; Figure 15).

The restored Project Area and natural marshes also supported similar numbers and diversities of species. Following restoration, species richness averaged 11.5 ± 0.4 in the Project Area, compared to 10.5 ± 0.4 in Reference Areas (GLM, df=1, F=1.72, P=0.196). After 2008, species richness increased in Reference Areas (GLM, df=1, F=4.76. P=0.03), as well as the Project Area. For the Shannon-Weiner diversity index, the Project Area averaged 1.42 ± 0.06, while Reference Areas averaged 1.36 ± 0.07 (GLM, df=1, F=0.07, P=0.80). There was no change in diversity indices in Reference Areas from Pre-Restoration (H'=1.32 ± 0.06) to Post-Restoration (H'=1.36 ± 0.07; GLM, df=1, F=0.87, P=0.35).

Species assemblages also became more similar between the restored Project Area and natural marshes (MRPP, df=1, T=-0.50, P=0.25). Multivariate ordination showed overlap between some of the Project Area and Reference Area sites (NMS, Final Stress=12.33, 3D solution, instab=0.00004, 96 iter; Figure 16b). Axes 2 and 3 accounted for 55% of the 78% of total cumulative variance (R^2) accounted by the data. Study Areas were differentiated primarily by salinity and temperature on Axis 2 (Kruskal-Wallis, df=1, H=3.96, P=0.05; Figure 16b).

Despite these multivariate analyses results, Indicator Species analysis still found compositional differences. Taxa including *Diaptomus* spp. (Calanoida), *Bosmina* spp. (Cladocera), *Brachionus* spp.

(Rotifera), Gammaridae (Amphipoda), and Rotifera were indicative of the Project Area (Monte Carlo, all P<0.03), while Reference Areas showed only a few statistically weaker associations with taxa such as *Acartia* spp. (Calanoida) and Cyclopoda (Monte Carlo, all P<0.12).

Benthic Invertebrates

Baseline Conditions: Prior to the restoration, the Project Area had significantly lower average benthic invertebrate densities than Reference Areas (GLM, df=1, F=27.00, P<0.001, square-root transformed; Figure 17). The Project Area averaged 0.15 ± 0.04 inverts/cm^3, compared to 0.85 ± 0.15 inverts/cm^3 within Reference Areas. The Study Areas also showed significant differences in species richness and diversity (H'). The Project Area had significantly lower levels of richness (16.4 ± 2.46) and diversity (H'=1.46 ± 0.10) than Reference Areas (richness=32.14 ± 2.46, GLM, df=1, F=20.47, P<0.001; and H'=1.88 ± 0.08, GLM, df=1, F=10.49, P=0.003).

In addition to abundance and diversity differences between Study Areas prior to restoration, MRPP analysis pointed to species assemblages also being significantly different (MRPP, df=1, T=-6.28, P<0.001). NMS ordination showed natural marshes as tightly grouped in one corner of the ordination plot and distinctly separate from the relatively loosely clumped Project Area sites (NMS, Final Stress=10.49, 2D solution, instab<0.064, 400 iter; Figure 18a). Axis 2 significantly differentiated Study Areas (GLM, df=1, F=95.98, P<0.001). Temperature, salinity, and pH had the strongest effect on spatial distribution of sampling sites (Figure 18a). Only temperature exhibited a significant effect on species composition within sampling sites (GLM, Temperature: df=1, F=3.15, P=0.088, R^2= 20%; Salinity: df=1, F=0.23, P=0.638; R^2=35%; pH: df=1, F=2.75, P=0.109, R^2= 57%).

Pre-Restoration, the Project Area had lower densities and a different mix of nematode, oligochaete, polychaete, and amphipod species than Reference Areas (MRPP, df=1, T=-2.65, P=0.02). There were some similarities: oligochaetes were by far the most prevalent higher order taxonomic group in both the Project Area (0.06 ± 0.02 inverts/cm^3) and Reference Areas (0.28 ± 0.06 inverts/cm^3), while amphipods were the least prevalent group in Reference Areas (0.10 ± 0.02 inverts/cm^3) and one of the least prevalent in the Project Area (0.01 ± 0.01 inverts/cm^3). In terms of lower order taxonomic groups, the benthic invertebrate species assemblage of the Project Area prior to restoration was composed largely of *Potamopyrgus antipodarum* (Gastropoda; 17% of all species), Cyprideis (Ostracoda; 8% of all species), and *Nematostella vectensis* (Actinaria; 6% of all species). Indicator Species Analysis

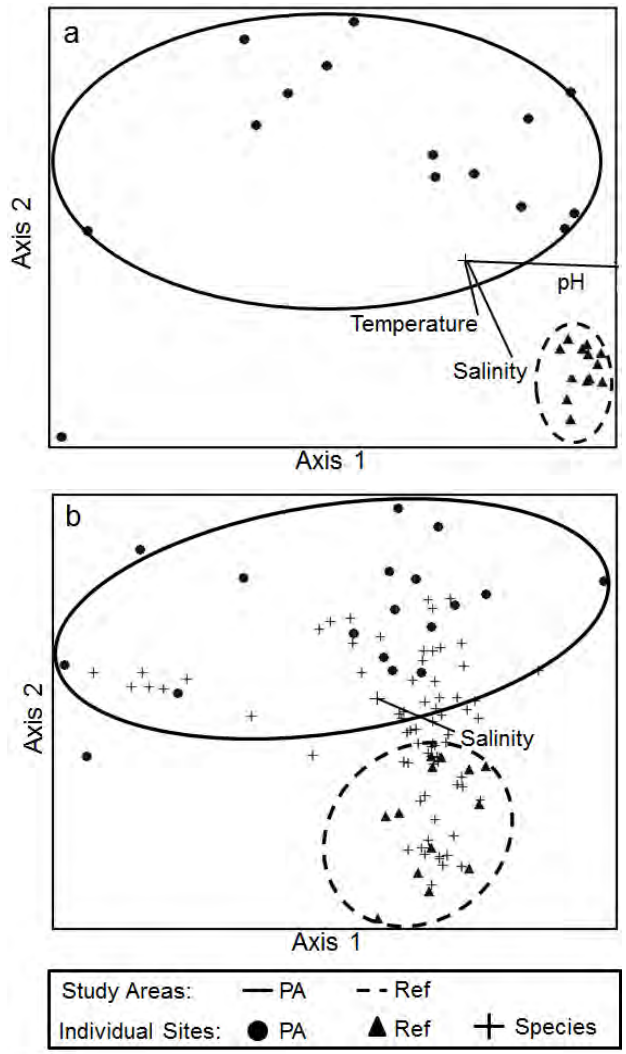

Figure 18. Benthic Invertebrate NMS
Two-dimensional (2D) statistical ordination of benthic invertebrate species assemblages and environmental variables using Non-metric Multi-dimensional Scaling between Study Areas for a) Pre-Restoration with joint plot (Final stress = 10.49) and b) Post-Restoration with joint plot (Final stress = 14.79). Salinity accounted for a moderate amount of variation between Study Areas Post-Restoration

revealed Chironomidae (Diptera) to be the only taxa weakly indicative of the Project Area Pre-Restoration (Monte Carlo, P=0.057). The most common taxonomic groups at the Reference Areas prior to restoration were *Streblospio benedicti* (Polychaeta; 8% of all species) and *Capitella capitata* (Polychaeta; 7% of all species). Indicator Species Analysis found 10 of 73 taxa to be strongly significant indicators of Reference Areas prior to restoration, including *Pseudopolydora kempi* (Polychaeta), *S. benedicti* (Polychaeta), *C. capitata* (Polychaeta), *Monocorophium insidiosum* (Amphipoda), *Grandidierella japonica* (Amphipoda), and *Gemma gemma* (Mollusca; Monte Carlo, all $P \leq 0.001$).

Changes in Project Area with Restoration: Benthic invertebrate densities within the Project Area increased by 530% after restoration (GLM, df=1, F=28.04, P<0.001, square-root transformed). Post-Restoration densities in the Project Area averaged 0.80 ± 0.13 inverts/cm^3, with a median density of 0.63 inverts/cm^3 (Figure 17). Mean species richness in the Project Area increased significantly from 16.4 ± 2.5 species Pre-Restoration to 30.9 ± 1.7 species Post-Restoration (GLM, df=1, F=24.66, P<0.001). Species diversity also increased in the Project Area (H'= 1.77 ± 0.07; GLM, df=1, F=6.7, P=0.017). MRPP analysis suggested significant shifts in species composition between Pre- and Post-Restoration periods in the Project Area (MRPP, df=1, T=-11.27, $P \leq 0.001$) and Reference Areas (MRPP, df=1, T=-8.70, P<0.001).

Convergence with Natural Marshes: Reference Areas also had a large (198%) increase in average benthic invertebrate density after 2008. Post-Restoration, the Reference Areas averaged 1.68 ± 0.3 inverts/cm^3, with a median density of 1.28 inverts/cm^3 (Figure 17). Due to large increases in both Study Areas, abundances within the restored wetland continued to differ significantly from Reference Areas (GLM, df=1, F=6.03, P=0.02, log-transformed; Figure 17).

Changes in species richness in Reference Areas between restoration periods were also significant (GLM, df=1, F=6.52, P=0.017). During Post-Restoration, species diversity between Study Areas was similar (GLM, df=1, F=1.53, P=0.226), while species richness continued to vary (GLM, df=1, F=12.10, P=0.002), with lower levels of species richness in the Project Area (30.9 ± 1.7) than Reference Areas (41.3 ± 2.6).

After restoration occurred, the Study Areas continued to show differences in benthic invertebrate species assemblages (MRPP, df=1, T=-2.96, P=0.014). NMS indicated that, while the group of Project Area sites had moved closer to that of the Reference Areas, they had not fully converged (NMS, Final Stress=14.79, 2D solution, instab<0.0001; 45 iter, Figure 18b). Axis 2 accounted for most of the separation between Post-Restoration Study Areas (Axis 2 GLM, df=1, F=87.13, P<0001). Salinity was the only environmental factor that appeared to affect spatial distribution of Post-Restoration sampling sites in NMS (Figure 18b) and also showed a significant relationship with species composition between sampling sites (GLM, df=1, F=4.18, P=0.05; R^2=31%).

After 2008, the Study Areas had a similar mix of higher order invertebrate taxa (MRPP, df=1, T=-1.28, P=0.10). Oligochaetes remained the most common invertebrate group in Reference Areas (0.40 ± 0.08 inverts/ cm^3) and a common one in the Project Area (0.21 $\pm$ 0.05 inverts/ cm^3), as well. Amphipods were seemingly the most prevalent invertebrate group in the restored wetland (0.39 ± 0.1 inverts/ cm^3). The three most common lower-order taxonomic groups in the restored Project Area were *Paracorophium* (Amphipoda; 34% of all species), *M. insidiosum* (Amphipoda; 11% of species), and *S. benedicti* (Polychaeta; 7% of all species), although the only strong indicator species of the restored wetland were *Eogammarus confervicolus* (Amphipoda) and *Gnorimosphaeroma insulare* (Isopoda); Monte Carlo, all P<0.01).

In Reference Areas, the most common taxonomic groups were *Paracorophium* (Amphipoda; 15% of species), *S. benedicti* (Polychaeta; 14% of all species), *Pygospio elegans* (Polychaeta; 5% of all species), and *M. insidiosum* (Ampipoda, 3% of all species; Figure 19). Despite similarities between prevalent species among Study Areas, Indicator Species Analysis found a considerable number of species that discriminated between the Post-Restoration Project Area and Reference Areas. Eleven taxa were identified as being indicative of natural marshes, with the strongest indicators being Lineidae (Anopla), *P. elegans, Nebalia kensleyi* (Malacostraca), *Cumella vulgaris* (Arthropoda), *Allorchestes angusta* (Amphipoda), *Psychoda* (Diptera), and *G. gemma* (Mollusca; Monte Carlo, all P<0.001)

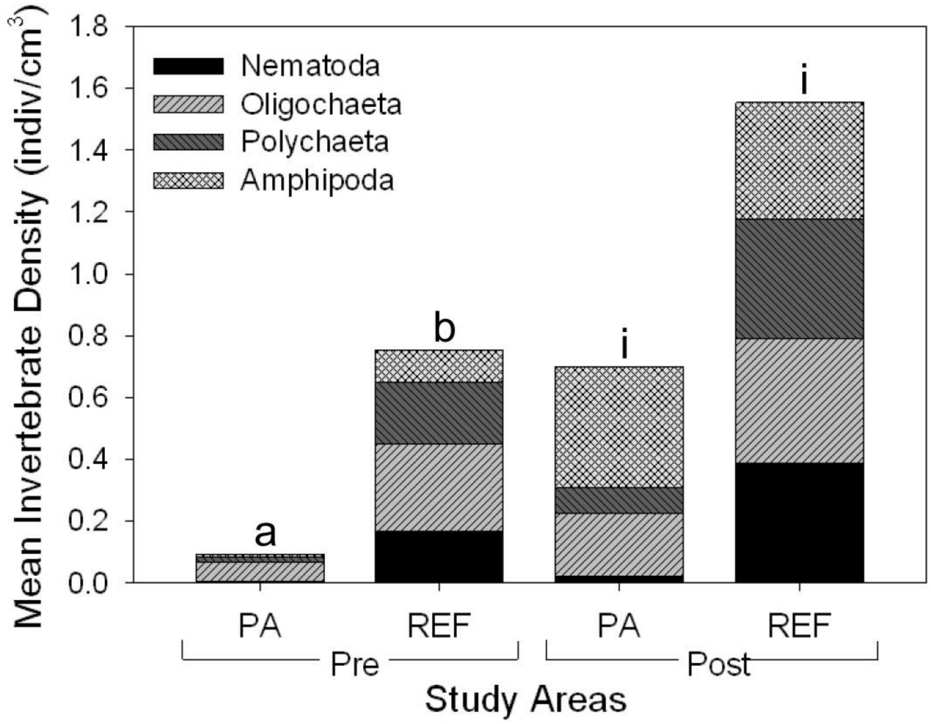

Figure 19. Mean benthic invertebrate density by taxum
Mean density of Nematoda (phylum), Oligochaeta (subclass), Polychaeta (class), and Amphipoda (class) between the Project Area and Reference Areas Pre- (MRPP, T=-2.65, P=0.02) and Post-Restoration (MRPP, T=-1.28, P=0.10). Statistically significant differences within treatment periods are shown with letters above density bars

Fish

Baseline Conditions: The composition of fish species prior to restoration showed strong differences between the Project Area and Reference Areas (MRPP, df=1, T=-11.67, P<0.0001). These differences were evident in statistical ordination results, which pointed to salinity and temperature as the primary factors separating fish assemblages between the unrestored dairy ranch and natural marsh sites (NMS, Final Stress=11.66, 3D solution instab=0.001, 400 iter.; Figure 20a). Axes 1 and 2 accounted for 65% of the 81% of the total variation accounted for by the model, with both axes showing separation between Study Areas (GLM, df=1, F=25.07 and 15.99, respectively, P<0.0001).

Indicator species analysis suggested that the Project Area fish assemblages differed from Reference Area ones by having higher relative abundances of threespine stickleback (*Gasterosteus aculeatus*); the federally endangered tidewater goby (*Eucylogobius newberryi*); and mosquitofish (*Gambusia affinis*; Monte Carlo, all P<0.004), while Reference Areas had higher relative abundances of arrow goby (*Clevelandia ios*), surfperch (Embiotocidae), and goby larvae (Monte Carlo, all P<0.05). In addition to mosquitofish, other non-native species present in the Project Area included yellowfin goby (*Acanthogobius flavimanus*), silver carp (*Hypophthalmichthys molitrix*), and white crappie (*Pomoxis annularis*).

Species richness did not differ greatly between the Project Area and Reference Areas prior to restoration. Before restoration, mean number of species ranged from 3.4 ± 0.2 in the Project Area to 3.2 ± 0.2 in the Reference Areas (GLM, df=1, F=0.06, P=0.81).

Changes in Project Area with Restoration: Restoration resulted in only a weakly significant change in fish community composition in the Project Area (MRPP, df=1, T=-1.66, P=0.07). Species richness in the Project Area remained similar to pre-restoration levels after levee breaching, averaging 3.5 ± 0.2 (GLM, df=1, F=0.07, P=0.79). Similarly, statistical ordination showed no clear separation between restoration periods within the Project Area based on species and environmental variables (NMS, Final Stress=16.00, 3D solution, instab=0.036; 400 iter; Kruskal-Wallis, all P for axes >0.15).

Indicator Species Analysis identified mosquitofish and, to a lesser degree, the Sacramento sucker (*Catostomus occidentalis*) as most indicative of Pre-Restoration Project Area fish assemblages (Monte Carlo, all P<0.07). In evaluating Post-Restoration indicator values relative to Pre-Restoration ones, common estuarine species such as threespine stickleback appeared equally abundant in the Project Area between treatment periods, while arrow goby abundance seemingly climbed during Post-Restoration, although neither result was statistically significant (Monte Carlo, all P>0.21). Relative abundance of tidewater goby, the federally endangered brackish water fish that occurred in the Project Area even prior to restoration, did not appear to change with restoration (Monte Carlo, P = 0.76).

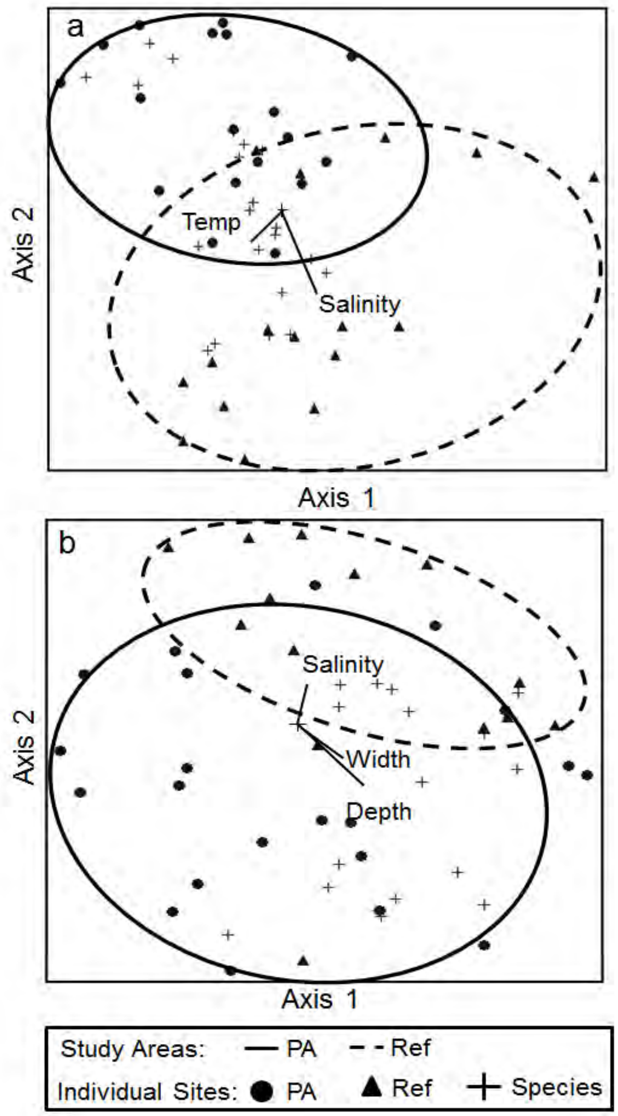

Figure 20. Fish NMS
Three-dimensional (3D) statistical ordination of fish species assemblages and environmental variables using Non-metric Multi-dimensional Scaling between Study Areas for a) Pre-Restoration with joint plot (Final stress = 11.66) and b) Post-Restoration with joint plot (Final stress = 9.03). Environmental variables did not appear to provide any separation between Study Areas

Convergence with Natural Marshes: As with Pre-Restoration, fish composition of the Project Area continued to differ strongly from Reference Areas even after restoration (MRPP, df=1, T=-6.00, P<0.0001). Statistical ordination showed that only a few Project Area sites overlapped with Reference Area ones (NMS, Final Stress=9.03, 3D solution, instab=0.0004, 111 iter.; Figure 20b). Axis 2, which showed the strongest separation of Study Areas, accounted for up to 67% of the 98% of total variation (R^2) explained by the model, suggesting that very strong differences still existed between the Project and Reference Areas four years after levees were removed (GLM, df=1, F=8.79, P=0.006). Some separation of Study Areas also occurred along Axis 1, but this axis differentiated sites more on the basis of salinity and width and depth of sampled area, with salinity generally higher within Reference Area sites and width and depth higher in Project Area ones (R^2=0.20).

Based on Indicator Species Analysis, differences between the restored wetland and natural marshes still revolved around a higher relative proportion of arrow goby, surfperch and longjaw mudsucker (*Gillichthys mirabilis*; Monte Carlo, all P<0.09) in Reference Areas. Similar to Pre-Restoration, the Project Area continued to support a higher relative abundance of tidewater goby and mosquitofish (Monte Carlo, all P<0.04) than natural marshes, as well as sculpin (Cottoidea; Monte Carlo, P=0.02).

Comparisons of species richness in the restored marsh with that in natural marshes (2.8 ± 0.2) continued to suggest that both Study Areas supported roughly the same number of species after 2008 (GLM, df=1, F=0.32, P=0.58).

Water Quality

Baseline Conditions: Water salinities differed significantly between the Project Area and other Study Areas prior to restoration (Kruskal-Wallis, df=2, H=472.6, $P < 0.001$). Median salinities were 1.6 ppt in the Project Area, 25.5 ppt in Reference Areas, and 0.1 ppt in Upstream Areas. Temperatures were also lower in the Project Area (median = 15.1 °C) than in Reference Areas (median=17.3 °C), although not lower than those in Upstream Areas (median=12.7 °C; Kruskal-Wallis, df=2, H=50.0, $p<0.001$). Median dissolved oxygen concentrations also showed considerable dissimilarity between the Project Area (7.58 mg/L), Reference Areas (8.32 mg/L), and Upstream Areas (9.51 mg/L; Kruskal-Wallis, df=2, H=38.6, $p<0.001$). Oxygen levels averaged as low as 4.98 ± 0.24 mg/L in the eastern portion of the dairy ranch. In contrast, median pH did not vary significantly between the Project Area and the other Study Areas prior to restoration (range=7.60 to 7.63; Kruskal-Wallis, df=2, H=5.1, P=0.08).

Changes in Project Area with Restoration: A significant change in average salinities occurred within the Project Area after restoration. Average salinities climbed 70% from 6.9 ± 0.3 ppt pre-restoration to 11.7 ± 0.5 ppt post-restoration (GLM, df=2, F=41.9, $P<0.0001$; sqrt transformed for analysis). Median salinities followed a similar pattern, climbing from 1.6 ppt before restoration to 10.8 ppt after restoration. Following restoration, mean oxygen levels in the Project Area increased 14% from 7.30 ± 0.13 mg/L pre-restoration to 8.30 ± 0.14 mg/L post-restoration (GLM, df=2, F=24.3, $P<0.0001$; log transformed for analysis). Conversely, median temperatures dropped by 6% in the Project Area after restoration from 15.1 °C to 14.1 °C (Mood Median Test, df=2, Chi-Square=8.84, P=0.012). Mean pH in the Project Area also declined after levee breaching, dropping approximately 5% from 7.58 ± 0.02 during Pre-Restoration to 7.21 ± 0.02 during Full Restoration (GLM, df=2, F=70.3, $P<0.0001$).

Convergence with Natural Marshes: Four years following restoration, pHs in the Project Area and Reference Areas still differed from each other (GLM, df=1, F=10.1, P=0.002), with pH averaging 7.21 ± 0.02 in the Project Area and 7.32 ± 0.03 in the Reference Areas. Salinities also remained somewhat dissimilar between Study Areas, although differences were only weakly significant (GLM, df=1, F=4.3, P=0.067). Salinities averaged 11.7 ± 0.5 in the Project Area and 19.8 ± 0.7 in Reference Areas after restoration. Other variables, however, showed some convergence between restored and natural marshes. Oxygen levels between the two Study Areas after restoration were equivalent from a statistical perspective: 8.30 ± 0.14 mg/L in the Project Area and 8.86 ± 0.36 mg/L in Reference Areas (GLM, df=1, F=1.09, P=0.30, log-transformed for analysis). Average temperatures after restoration also did not differ significantly between the Project Area (15.2 ± 0.2 °C) and Reference Areas (15.3 ± 0.3 °C; GLM, df=1, $F<0.0001$, P=0.991).

Discussion

Convergence with Natural Tidal Marshes

Thus far, the effects of restoration on the Giacomini Wetland food web have been considerably dissimilar to those of previous tidal wetland restoration projects. Whereas other projects have often documented a strong and immediate response to restoration within fish communities, restoration at Giacomini has not produced any significant change in fish species assemblages or species richness in the four years since construction, although any changes may have been obscured, in part, by our sampling approach. Conversely, one of the parameters that other studies suggested can take decades to respond -- benthic invertebrates -- has already shown a strong response to restoration, with densities and species richness, diversity, and composition changing considerably post-restoration. Species assemblages, richness, and diversity of zooplankton species also shifted significantly in the restored wetland after levees were removed, but, unlike benthic invertebrates, abundance of zooplankton actually decreased in both the Project Area and Reference Areas after 2008.

With these changes, both zooplankton and benthic invertebrate communities have started converging with those of natural marshes, although convergence is not yet complete. Median densities, species richness, and species diversity of zooplankton were statistically equivalent between the restored wetland and natural marshes after four years, as was diversity of benthic invertebrate species. In addition, based on ordination and other multivariate analyses, species composition of zooplankton and benthic invertebrate assemblages in the Project Area have also moved closer to those in Reference Areas. Benthic invertebrate densities and species richness in Giacomini still differed from those of post-restoration natural marshes, although post-restoration densities in the restored wetland were remarkably similar to pre-restoration ones in natural marshes.

Role of Environmental and Climatic Factors

Salinity was one of the strongest drivers of change in the Project Area species assemblages following restoration, as well as being the strongest driver of continued differences in community assemblages between the restored wetland and natural marshes. Project Area waters have become more saline with reintroduction of tidal flow. Salinity changes may have directly affected invertebrate assemblages, but they could also represent a surrogate for other changes brought about by restoration, including reduced water impoundment, improved water quality, and higher oxygen levels in sediments, which encourages more colonization by benthic invertebrates. Interestingly, higher salinities in the restored wetlands appear to be associated with increases in benthic invertebrate densities, but decreases in zooplankton abundance, although statistical analyses pointed to water temperature having a stronger effect than salinity on zooplankton densities in the restored and natural marshes.

Similar trends in invertebrate density changes between Study Areas after 2008 suggests that restoration may not be the only factor effecting change. At Limantour, increases in benthic invertebrate density could be potentially attributed to a restoration project that occurred upstream in 2008, however, density increases were observed in all of the natural marshes monitored. A more likely explanation for system-wide shifts is that organisms are responding to changes in rainfall patterns. While the post-restoration period was slightly wetter than the pre-restoration one in terms of annual rainfall, rainfall totals in the fall were actually lower after restoration (Western Regional Climate Center, Olema Valley). Lower rainfall means less freshwater entering wetlands, which can increase water salinities. Less freshwater inflow can also decrease erosion of the sediment substrate and minimize fluctuating sediment salinity levels. Benthic invertebrate communities can be greatly depressed by the scouring effect of large storm events and dramatic changes in sediment salinity levels (Wolff 1983; Nordby and Zedler 1991). Therefore, while restoration did influence changes in benthic invertebrate and zooplankton communities in the Project Area, climatic factors may have played a role, as well. The role that climate plays in shaping of these communities may be further intensified in future years with global climate change potentially leading to greater unpredictability in both intra- and inter-annual rainfall (Cayan *et al.* 2012).

While restoration may promote convergence between Study Areas, complete convergence may not be possible due to fundamental hydrologic differences between these systems. For fish, restoration may be less important to community dynamics, at least to this point, than hydrologic factors such as the amount of freshwater inflow. Project Area waters are generally more brackish, even in summer, than those of the natural marshes due to higher groundwater and freshwater inflow rates. The Study Areas supported similar numbers of fish species before and after restoration, but the type of species found continued to differ between them, with the Project Area supporting more species such as sculpin, stickleback, and tidewater goby and natural marshes supporting more arrow goby, topsmelt, and surfperch. Restoration appeared to affect fish communities in the restored wetland primarily through reducing abundance of non-native species such as mosquitofish, yellowfin goby, white crappie, and silver carp. Similarly, while the restored Project Area did support three of the four most common benthic invertebrate species in natural marshes, differences in hydrologic regimes between Study Areas may lead to persistent differences in invertebrate species assemblages.

Role of Invasive Species

Perhaps one of the most notable differences between the restored wetland and its natural counterparts was that the Reference Areas supported a considerable number of opportunistic, invasive invertebrates both before and after 2008, but even four years after levee removal, the Project Area largely did not. Based on results of other studies, newly restored wetlands are often quickly colonized by a large, if not species-rich, group of opportunistic non-native species that eventually gives way to a more stable assemblage dominated by both opportunistic taxa and those more characteristic of natural marshes (Levin *et al.* 1996; Alphin and Posey 2000; Zajac and Whitlatch 2001). Opportunistic species may take advantage of restoration-related disturbance to become the dominant fauna (Levin *et al.* 1996; Alphin and Posey 2000; Zajac and Whitlatch 2001; Havens *et al.* 2002; Moseman *et al.* 2004; Surugiu and Feunteun 2008; Takata *et al.* 2011). In some instances, however, numbers of invasives have been higher in reference wetlands than restored ones (Minello and Webb 1997). Many of the opportunist species found in natural marshes were polychaetes such as *S. benedicti*, *P. elegans*, and *C. capitata*, as well as species from other orders such as Bivalvia (*G. gemma*). Of these species, only *S. benedicti* occurred in any real numbers within the restored wetland by Year 4. Non-invasive taxa such as oligochaetes have a limited dispersal stage (Levin *et al.* 1996), which may hamper colonization of restored areas, although oligochaetes were the dominant invertebrate taxa at both Giacomini and natural marshes. The restored marsh also supported very high numbers of amphipods such as *Paracorophrium*, *M. insidiosum*, and *E. confervicolus*. Amphipods are generally much lower in abundance in created marshes than natural ones (Minello and Zimmerman 1992), which may relate to their pollution intolerance (Nelson *et al.* 2000).

Role of Restoration Approach

Ultimately, evolution of benthic invertebrate and perhaps even zooplankton communities in the restored wetland may have been strongly influenced by the level of disturbance associated with restoration. Newly restored areas in other systems are often defaunated during the restoration process by placement of dredge spoil; extensive excavation of channels, channel bottoms, or marshplains; or other substrate-disturbing procedures (Zajac and Whitlach 2001). Almost every one of the previous studies that evaluated benthic invertebrate communities used marshes that were built with dredge spoil material, which was one of the earliest tidal marsh restoration techniques. Dredge spoil marshes are typically created from excavation of shipping lanes and other subtidal or intertidal areas and placement of dredged material into low to high intertidal zones (Zedler 2000). Dredge spoil soils vary in texture, but many used for marsh creation have tended to be very sandy (HLA 2000, Zedler 2000). Several studies have found that soils of dredge spoil marshes have higher sand, less clay, and less organic matter, including MOM, than adjacent natural marshes (Lindau and Hossner 1981, Craft *et al.* 1988, Langis *et al.* 1991, Minello and Zimmerman 1992, Sacco *et al.* 1994, Ferguson and Rakocinski 2008).

The likelihood that systems with markedly different soils would develop similar infauna communities seems extremely low. Some of the sites where benthic invertebrate communities evolved more quickly were ones where dikes were breached (Eertman *et al.* 2002) or uplands were excavated (Havens *et al.* 2002), although some culvert replacement projects where substrate was not necessarily disturbed still had invertebrate communities that had not converged with reference marshes (Warren at al. 2002).

The minimalistic approach used to restore the Giacomini Wetlands (removal of levees from non-tidal and muted tidal areas with minimal excavation) appears to have placed this wetland on a slightly different evolutionary trajectory than those of more disturbance-intensive projects. The reduced scale of substrate disturbance during restoration may have preserved some remnant of the original benthic community in sheltered areas that were not directly exposed to tidal scour or abrupt increases in water salinity. Recolonization of disturbed areas typically occurs through lateral advection of either adult organisms (often opportunistic invaders) or juveniles, and recolonization dynamics are often governed by the extent of restoration-related ground disturbance and the proximity of source areas or “pools” (Cammen 1976; Alphin and Posey 2000; Zajac and Whitlatch 2001; Ferguson and Rakocinski 2008). The natural marsh that directly adjoins the restored wetland represents a sizeable source of potential colonizing organisms. Rapid recovery of benthic infauna may have also been promoted by the fact that unexcavated marsh soils probably retained considerable organic matter, particularly MOM, which many studies have found to be strongly linked to benthic community development (Cammen 1976; Minello and Zimmerman 1992; Talley and Levin 1999; Havens *et al.* 2002; Moseman *et al.* 2004; Bolam *et al.* 2006; Ferguson and Rakocinski 2008).

Documented changes in invertebrate and, to a lesser degree, fish communities will have dramatic repercussions on higher trophic level organisms such as larger fish and birds. While waterfowl responded almost immediately to restoration of the Project Area, arriving in high numbers only a month after levees were breached, shorebird numbers during the first winter were quite low (ARA 2012). In subsequent years, shorebird numbers have increased, although abundance has been variable intra- and inter-annually (ARA 2012). Some of the more recent frequent visitors to the restored wetland include species such as marbled godwits (*Limosa fedoa*), dunlin (*Calidris alpina*), and dowitchers (*Limnodromus* spp.), which are deeper substrate probers (ARA 2012). These results would seemingly confirm our findings that benthic invertebrate resources are increasing, as the pattern of fish and bird use would be expected to evolve in concert with that of the lower trophic levels.

Conclusions

Our fish and invertebrate monitoring results differ considerably from those of other tidal wetland restoration projects. Studies in other systems have shown that fish communities similar to those of natural marshes can rapidly establish in constructed or restored marshes, while invertebrate communities can take decades to reach equivalency (Minello and Zimmerman 1992; Minello and Webb 1997; Talley and Levin 1999; Ferguson and Rakocinski 2008). These differences may ultimately relate to the restoration approach taken by these projects, with more minimalistic approaches allowing for more rapid development of invertebrate communities and for less establishment by non-native opportunistic species that take advantage of restoration disturbance to rapidly colonize restored marshes.

These non-native species introduce a ‘wild-card’ factor into the evolution of the newly restored Giacomini Wetlands. Invertebrate species assemblages in the restored wetland have so far not become dominated by opportunistic species, but this somewhat atypical dynamic may shift in future years, given the high number of non-native species present in adjacent natural marshes. However, continued differences in hydrologic conditions between the Giacomini Wetlands and its more saline tidal marsh counterparts may at least help to preclude establishment by at least some of these species.

In this case, then, convergence of the restored wetland with natural marshes may not be desirable, at least in terms of invertebrate communities. Non-native invertebrates can severely disrupt the food chain within estuaries by eliminating traditional diet items and supplanting them with lower-quality or unpalatable ones, as has been seen in nearby San Francisco Bay with introduction of the non-native, invasive Asian clam (*Corbula amurensis*) and subsequent dramatic declines in native copepod species and native fish such as northern anchovy (*Engraulis mordax*) and juvenile salmon (Orsi and Knutson 1979, Kimmerer and Orsi 1996, Modlin and Orsi 1997, Kimmerer 2006).

Unlike San Francisco Bay, the historical ecology of Tomales Bay is not well known, so it is difficult to determine how much impact to the food web these invaders have had on the estuary -- and will have on the restored wetland. Ultimately, the success of this and other restoration projects in supporting both lower and higher tropic levels of wildlife may not depend on internal factors such as success of restoration efforts, but instead on extrinsic factors such as short-term or long-term climatic patterns and presence or abundance of non-native species within the watershed.

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7 Results and Discussion: Changes in Vegetation Communities

Introduction

Restoration typically involves primary or secondary succession. Sites that have been created through dredge spoil disposal or complete excavation of uplands to lower intertidal elevations represent primary succession, while those that involve conversion of previously vegetated lands such levee breaches on former pasturelands represent secondary succession.

A substantial percentage of previous restoration projects have shown a rapid establishment or conversion of glycophytic vegetation to halophytic species, often within a few years after restoration was implemented (Eertman *et al.* 2002; Havens *et al.* 2002; Odland and Del Moral 2002; Thom *et al.* 2002; Wolters *et al.* 2005; Konisky *et al.* 2006; Wolters *et al.* 2008; Bowron *et al.* 2009; Pétillion *et al.* 2010). In almost all cases, the turnover to salt marsh from previous vegetation communities or even from no vegetation at all was characterized as “rapid.”

However, despite this rapid recolonization, many of these restoration projects are still not considered entirely successful from a vegetation perspective. This lack of success often stems from dissimilarities with target or reference marsh communities in terms of species composition or zonation patterns, as well as failure to meet other target objectives such as providing habitat for special status species (Onaindia *et al.* 2001; Thom *et al.* 2002; Warren *et al.* 2002; Wolters *et al.* 2008; Garbutt and Wolters 2008). A number of studies from both the United States (Burdick *et al.* 1996, Thom *et al.* 2002, Smith *et al.* 2009) and Europe (Onaindia *et al.* 2001, Wolters *et al.* 2005, Garbutt and Wolters 2008, Wolters *et al.* 2008) found that, no matter how long had passed since restoration was implemented, very few of the created marshes had similar species composition to local or regional natural marshes.

Studies on these systems point to a number of reasons for these issues, including quite a number of physical ones. Elevation in relation to tide level represents one of the most important physical factors affecting salt marsh establishment, because the depth and frequency of tidal flooding (Thom *et al.* 2002, Warren *et al.* 2002; Wolters *et al.* 2005, Wolters *et al.* 2008, Smith *et al.* 2009; Pétillion *et al.* 2010). Should restoration areas be too deeply subsided due to long-term leveeing, or should elevations be too high or too low to support the proper tidal frame, systems similar to natural marshes will not develop (Thom *et al.* 2002; Williams and Orr 2002; Wolters *et al.* 2005). Depth and frequency of tidal inundation, in turn, strongly influences marsh characteristics such as waterlogging/soil aeration, water salinity, and soil salinities, all of which also affect plant distribution (Cooper 1982; Snow and Vince 1984, Armstrong *et al.* 1985).At higher elevations, interspecific competition between species is believed to determine fine-scale vegetation patterns, although interspecific facilitation may actually extend elevational ranges of some species by reducing potential hypersalinity in soils through shading (Bertness and Hacker 1994; Pennings *et al.* 2003). The relative influence of all these factors on plant distribution may vary on a regional basis (Pennings *et al.* 2005).

Tidegates, culverts, and breaches used in some restoration projects can create an improper hydroperiod by compressing the tidal range too severely or restricting outflow and, therefore, causing impoundment of waters (Burdick *et al.* 1996; Warren *et al.* 2002; Smith *et al.* 2009). Even if elevation and hydroperiod are correct, improper soil substrate can hamper marsh creation efforts: many early dredge spoil marshes were built using coarse sand from shipping lanes, or, if muds were dredged, these become highly acidic when exposed to air, leading to either sparsely vegetated conditions or highly dissimilar plant communities (Boyer and Zedler 1996, 1998, L. Parsons, NPS, *pers. comm.*). In addition, in situations where marshes are constructed by deeply excavating uplands, the risk for having poor soil conditions – low in O.M. and bulk density – can be high. However, in projects where levees of former pasture- or agricultural areas are breached, leaving formerly reclaimed lands intact rather than excavating runs it own risk, as well. In the U.K., some believe that the changes in soils from centuries of agricultural management may be affecting distribution of marsh species (Garbutt and Wolters 2008) and preventing natural establishment of tidal marsh channels due to aquacludes or layers of overly consolidated material (Wolters *et al.* 2005). Channel formation can also be hampered by site elevations being too high initially: the presence and distribution can influence salt marsh species establishment (Wolters *et al.* 2005).

Some of the biological issues that have impacted marsh establishment include lack of seed source for marsh recolonization, which is typically due to the fact that there are no natural marshes nearby, and revegetation was not conducted (Onaindia *et al.* 2001). Two Spanish marshes ranging in age from 20 to 35 years old had less than 50% of the species richness and similarity in species composition to natural marshes in the region, which researchers felt may have been partly due to the distance to natural marshes (80 km), as well as differences in edaphic factors (Onaindia *et al.* 2001). Even when seed source is present, created marshes sometimes still bear little resemblance to their natural counterparts, particularly in terms of high marsh species composition or the presence of uncommon species (Garbutt and Wolters 2008, Wolters *et al.* 2008). Garbutt and Wolters (2008) evaluated 18 created marshes in Essex in the U.K. ranging in age from 2 to 107 years since restoration and found that more than half of them (13) still had fewer species than adjacent natural marshes, even the older projects. Some projects, however, are doomed from the start due to hijacking of the restoration process by invasive, non-native species. In England, many newer marshes are now dominated by *Spartina anglica*, a neoendemic that was introduced in the last century: this species can drastically alter sedimentary and drainage characteristics, leading to waterlogging and anoxic soils that discourage other marsh species (Garbutt and Wolters 2008).

Lastly, even if of these other factors are present, the absence or minimal presence of a natural disturbance regime may still lead created or restored marshes not to resemble existing tidal marsh systems. Disturbance factors such as grazing led to seven of a number of sites studied in the U.K., Netherlands, Belgium, and Germany to be the most successful in terms of soil saturation index – which was linked to elevational range – and target species (Wolters *et al.* 2005).

The pattern of vegetation establishment differs between restored systems, but there are some commonalities. A high number of species typically colonized in the first few years (Odland and del Moral 2002). Many of these species were annuals or opportunistic colonizers such as *Salicornia, Suaeda, Sarcocornia perennis, Chenopodium, Atriplex, Salsola, Spergularia marina*, and *Puccinellia maritima* (Pétillion *et al.* 2010). In an analysis of a marsh restoration site in the Blackwater Estuary in the U.K., plant species' tolerance for higher salinity conditions distinguished early colonizing species from later ones (Wolters *et al.* 2008). According to some researchers, *Puccinellia maritima* can be considered characteristic of newly created salt marshes and can lead to competitive exclusion of other species (Pétillion *et al.* 2010): it was the first perennial species at the Tollesbury marsh to reach 100% abundance (Wolters *et al.* 2008). *Spergularia marina* is typical of disturbed soil conditions and occurs in dried-up pannes or along paths and tracks (Rodwell 2000 *in* Wolters *et al.* 2008).

Over time, high species richness can decline somewhat, as opportunistic species begin to disappear from the site, and perennial species gain a stronger control over the newly restored landscape. At a site in the Belgian IJzer Estuary, frequency of *Chenopodium, Atriplex* and *Salsola* decreased by 76%, 51%, and 53% between 2002 and 2007 and were gone entirely by 2007 (Pétillion *et al.* 2010). Colonization by perennial species, in some cases, followed a sigmoidal pattern, which suggests primarily clonal growth patterns (Wolters *et al.* 2008). The speed of perennial marsh species expansion may be higher at lower inundation frequencies (Pétillion *et al.* 2010). Lower inundation frequencies typically occur at the higher intertidal elevations, which may be why factors such as frequency of species in the local species pool and seed flotation time appeared to more important in terms of separating intermediate-arrival species from primary colonizers (Wolters *et al.* 2008).

A number of theories have been developed to try and describe patterns in colonization by plants and other species. Some of these have attempted to use their theories on species assembly to predict what might happen with restoration. The goals of many projects are to create systems that are similar to either historic conditions or to other nearby natural marshes. In general, the proponents of the "If You Build It (marsh), (It/They) will come" school-of-thought believe that reestablishment of the proper physical conditions should result in the return, over time, of either the historic or reference marsh vegetation communities (Suding and Hobbs 2009). Also called the Gradual Continuum Model, this ecosystem evolution theory assumes a linear model of change from the disturbed condition to a more natural state. However, in recent years, ecologists have acknowledged that communities may not have a set endpoint in terms of species assemblages or communities. Chance establishment events can actually lead to a different group or assemblage of species either within the same type of community (i.e., different types of marsh or grassland communities) or even a completely different community (i.e., either coastal scrub or grassland). In addition to the endpoint not being pre-determined, the route to this endpoint may also not be a linear process, but involve a number of alternative stable states in which thresholds need to be passed to move from one state to the next.

One theory that may have incredible influence over the success of restoration project is priority effects. Priority effects revolve around whether early colonizing species can significantly influence the evolutionary trajectory of a system to drive in a different direction than might have otherwise occurred if another species had established. Where this is relative to restoration projects comes from the fact that different species may be present in the watershed currently than were present when natural marshes established, and these species could drive restored marsh evolution on a different trajectory to that of their historic or natural counterparts. Priority effects may also play a significant role if restoration involves conversion of already vegetated areas to salt marsh communities, as the existing species –- whether they be remnant or dying freshwater grasses or herbs or even leftover salt marsh species that colonized under non-tidal conditions –- can influence the extent of salt marsh species colonization and the specific species that can establish within this existing framework.

Vegetation Monitoring Framework

This restoration project involved three types of monitoring: vegetation mapping, vegetation change monitoring, and vegetation assembly monitoring along channels. Each type of vegetation monitoring is designed with slightly different goals in mind – or to answer a different question about the site- but they also have areas of overlap. Vegetation mapping is intended to give us detailed information about the *vegetation communities* at the site, and their *spatial representation*. It tells us overall acreage of different communities throughout the site, allows us to see how they are configured spatially, and visually represents how the spatial extent of different types of community changes over time. Vegetation change monitoring is intended to *quantitatively monitor* the percent cover of different *plant species* along permanent 50-meter transects scattered throughout the Project Area and three references marshes (relatively pristine marshes). Soil and plant biomass data are also collected at each transect. This data has been collected annually since 2006 and allows us to evaluate the changes in plant community between years and before and after the restoration, and connect this to changes in the soil and biomass of each site. This data is used to assess important ecological functions such as plant community diversity, biomass production (a food source for the ecosystem), and the abundance of weedy non-native plant species.

Vegetation assembly data is intended to inform understanding of how the new plant communities were formed in the restoration area. It was oriented along channels, and involved collecting percent cover in three 2m by 1 m quadrats located directly adjacent to the channel, 6 m away, and 23 m away in a perpendicular transect. Sites were set up both areas that had complete removal or die-off of vegetation with the restoration, representing a primary successional state for a marsh community, and sites that had been subject to muted tidal influence prior to restoration and thus represented a secondary successional state. Primary and secondary successional sites were not directly comparable, however, as sites could not be randomly assigned primary or secondary succession. We also examined whether sites that were scraped of vegetation and topsoil differed in vegetation community assembly. Monitoring occurred in 2009,one year post- restoration, and in 2010, 2011, and 2013, five years post-restoration.

Vegetation Mapping - Results

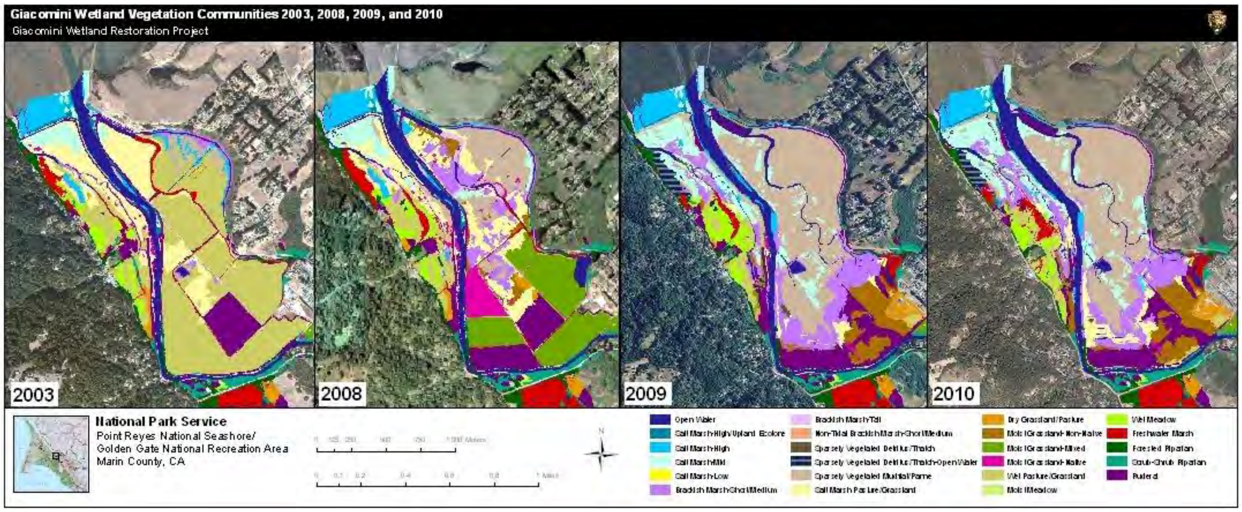

Figure 21. Vegetation Communties on the Giacomini Wetlands – 2003, 2008, 2009, and 2010.

Vegetation communities have changed drastically on the Project Area since restoration (Figure 21). The biggest change is the loss of Wet Pasture, which had decreased by more than 200 acres in 2010 compared to pre-restoration in 2003 (Table1). The largest increase was in Mudflat/Panne habitat, which has been the dominant community in the Project Area since restoration. Most of the mudflat habitat is too low in elevation relative to the tides to support vegetation. However, this habitat has had some colonization: in 2010, this habitat had decreased by almost 20 acres compared to 2009, reflecting that this habitat *is* slowly being colonized by salt marsh plants. Even as Brackish Marsh and Salt Marsh communities moved into some of the bare mudflats, much of the area classified as Brackish Marsh in 2009 was colonized by salt-tolerant grass species and had developed into Salt Marsh Grassland by 2010.

Table 1. Vegetation Community Acreages by Year. Change column represents differences in acreage between 2003 and 2010.

Color	Vegetation Community	2003	2008	2009	2010	Change
	Brackish Marsh-Short/Medium	8.0	37.9	64.9	66.1	58.1
	Brackish Marsh-Tall	6.6	12.1	4.1	4.0	-2.6
	Non-Tidal Brackish Marsh-Short/Medium	0.0	0.0	1.3	0.7	0.7
	Brackish Marsh Total	**14.6**	**50.0**	**70.3**	**70.8**	**56.2**
	Salt Marsh-High/Upland Ecotone	5.0	5.2	5.7	2.6	-2.4
	Salt Marsh-High	28.2	24.0	23.0	22.9	-5.3
	Salt Marsh-Mid	22.3	29.8	59.0	62.8	40.5
	Salt Marsh-Low	0.1	0.0	0.1	0.1	0.0
	Salt Marsh Total	**55.6**	**59.0**	**87.9**	**88.4**	**32.8**
	Sparsely Vegetated Detritus/Thatch	0.0	0.0	7.1	5.8	5.8
	Sparsely Vegetated Detritus/Thatch-Open Water	0.0	0.0	1.9	1.2	1.2
	Sparsely Vegetated Mudflat/Panne	11.4	54.8	167.8	150.4	139.0
	Mudflat/Panne Total	**11.4**	**54.8**	**176.8**	**157.4**	**146.0**
	Wet Pasture/Grassland	82.5	72.4	8.9	23.4	-59.1
	Dry Grassland/Pasture	9.5	12.3	10.7	12.2	2.7

Table 1. Vegetation Community Acreages by Year. Change column represents differences in acreage between 2003 and 2010.

Color	Community	2003	2008	2009	2010	Change
	[illegible]	0.0	11.8	32.3	35.3	35.3
	[illegible]	3.1	89.9	6.5	5.2	2.1
	[illegible]	2.2	20.3	1.7	1.1	-1.1
	[illegible]	233.5	31.1	17.1	21.7	-211.8
	Grassland Total	**330.8**	**237.8**	**77.2**	**98.9**	**-231.9**
	[illegible]	3.3	0.8	0.7	0.9	-2.4
	[illegible]	28.0	13.8	17.2	11.5	-16.5
	[illegible]	55.4	53.8	48.1	52.8	-2.6
	Freshwater Wetland Total	**86.6**	**73.5**	**66.2**	**65.2**	**-21.4**
	[illegible]	41.3	42.7	40.2	40.8	-0.5
	[illegible]	30.6	28.4	28.8	29.8	-0.8
	Riparian Total	**71.9**	**71.1**	**69.3**	**70.6**	**-1.3**
	[illegible]	51.4	58.4	64.4	69.7	18.3
	[illegible]	54.5	78.9	75.6	67.5	13.0

Vegetation Transect Monitoring - Methods

Like most of other aspects of the Restoration Project Monitoring, the vegetation monitoring transects were set-up using a modified BACI (Before-After, Control-Impact) and monitoring was carried out before and after the restoration in both the Project Area (impact) and three references marshes (control). Vegetation monitoring was initiated in the West Pasture, the Undiked Marsh, and Walker Creek Marsh in 2006 and expanded to include the entire restoration area and Limantour Marsh in 2007. Vegetation transects are scattered throughout the Project Area and range in elevation between 3 and 11 feet NAVD88. Reference sites were smaller than the Project Area, and vegetation transects were located on the marsh plain (mid or high marsh) over a much narrower range of elevations: between 4.5 and 7.5 feet NAVD88.

Plant community type was determined by assigning a community type to every species encountered as generally belonging to one of six communities (Freshwater Marsh, Non-native Grassland, Native Grassland, Brackish Marsh, Salt Marsh, and Ruderal). An overall plant community type was assigned to each transect based on whichever community type had the greatest aggregate cover based on the sum of its member plant species. This allowed us to track shifts in plant community over time.

Numerous studies have found elevation in relation sea level to be the most important factor in determining the distribution of salt marsh vegetation (Chapman 1974, Pennings and Callaway 1992, Pennings *et al.* 2005), which forms distinct zones along elevational gradients, and whose upper bound is defined by the highest reach of tidewaters. Therefore, the Project Area was broken into three zones (Figure 22) for the purpose of analysis and discussion: low marsh (n=10 transects, 37% of Project Area), below reference elevation; mid/high marsh (n=20 transects; 37% of Project Area), within the same elevation range as the reference marshes; and upland (n=11 transects, 26% of Project Area), a catchall for elevations above the reference site (not necessarily designating non-wetland status). Results are presented for the restoration area overall and for the three elevation zones within the Project Area. At times, reference sites were compared only to the Project Area sites within the mid-high marsh elevations as those sites located at higher or lower elevations would not be expected to have the same vegetation as the reference marsh.

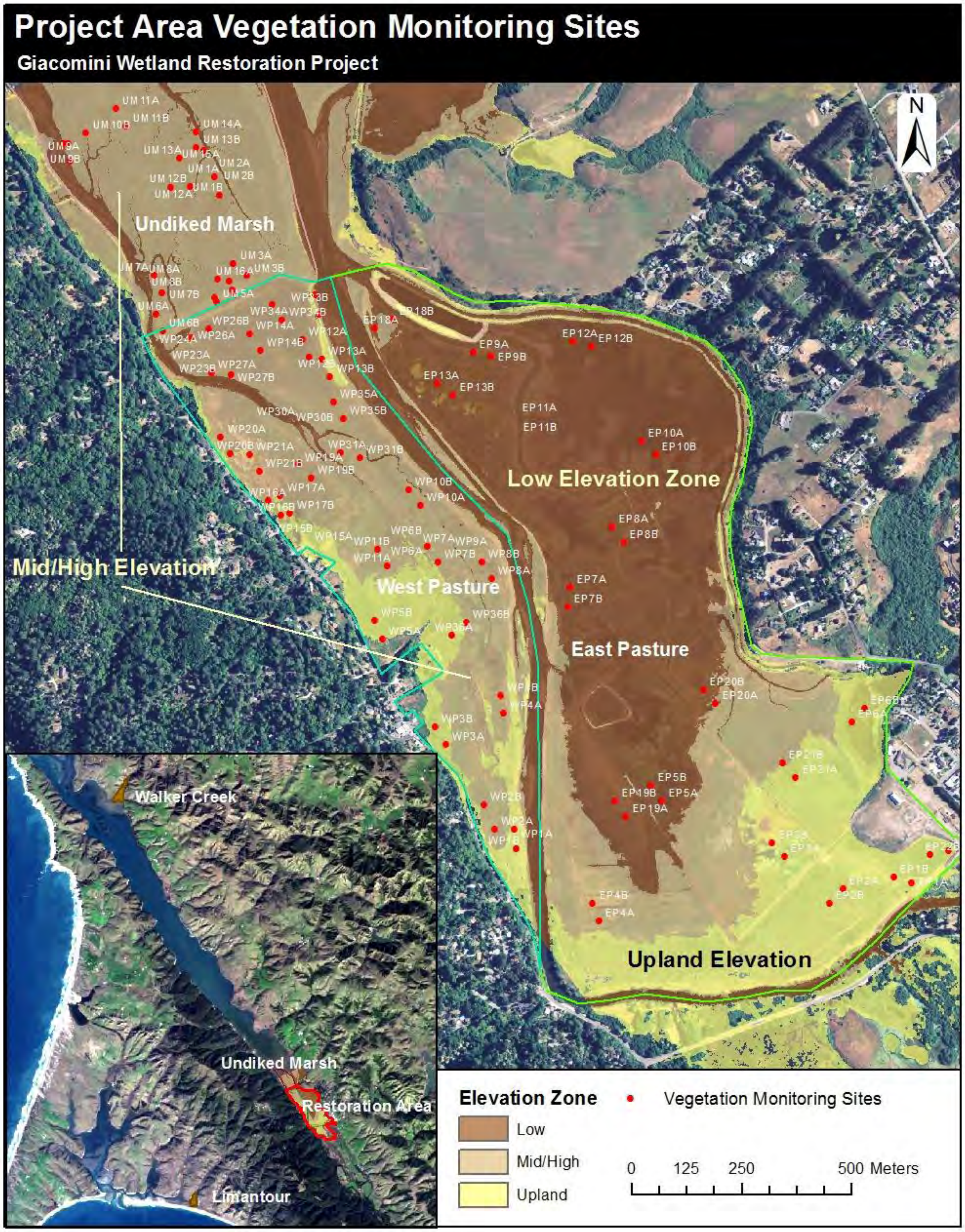

Figure 22. Vegetation Monitoring Transects and associated elevation zonation.
Paired points represent start (A) and end (B) of each transect. Low elevation (brown) is between 3-5 feet NAVD88, Mid/High (tan) is between 5-8 feet NAVD88, and Upland is 8-11 feet NAVD88.

Vegetation Transect Monitoring - Results

General Plant Communities

Overall: Prior to the restoration, most vegetation transects were categorized as Non-Native Grassland, followed by Native Grassland (Figure 23). Three-quartersof monitored transects fell into one of the two grassland types. Interestingly, some transects (15%, n=6) on the Project Area were dominated by Salt Marsh plants before the restoration took place. These transects were located in the northern portions of the East and West Pastures where there was occasional salt water influence through gaps in the levee to allow high-water outflow, and, in the case of the West Pasture, from a leaky tidegate. Also, these transects were located in areas where subsidence had occurred.

During 2008 - when “passive restoration” took place through discontinuation of agriculture, but levees and ditches remained in place - the majority of transects (65%, n=26) were still classified as grassland, but the cessation of pumping and irrigation caused some areas to start reverting to their natural, saltier, condition. Thus we saw an increase in Salt and Brackish Marsh sites from 15% to 25% (2 Brackish and 8 Salt Marsh sites). While grassland decreased overall, Native Grassland transects actually increased from 10% to 15% (n=4 to 6), whereas four Non-Native Grassland transects became dominated by plant species from other habitats.

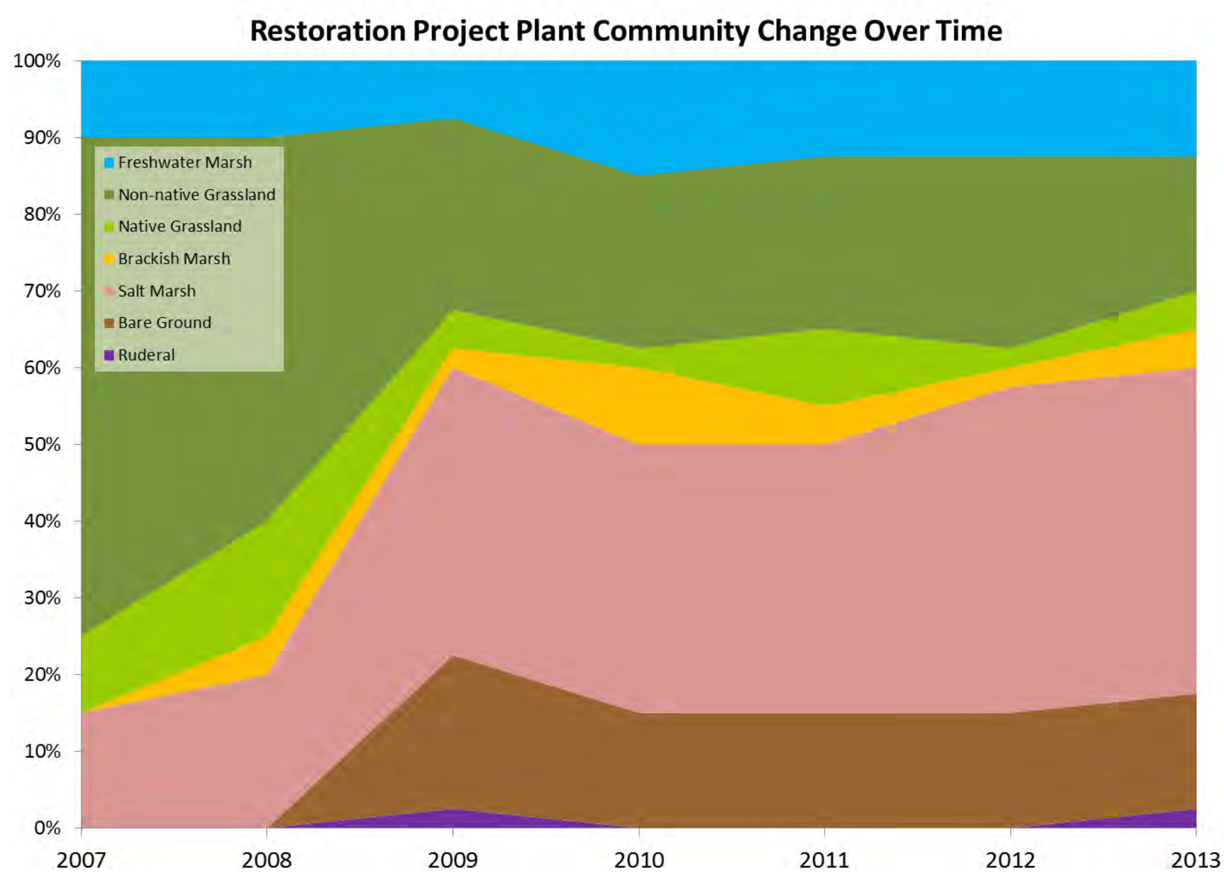

Figure 23. Percent of vegetation monitoring transects in each community category, 2007-2013.

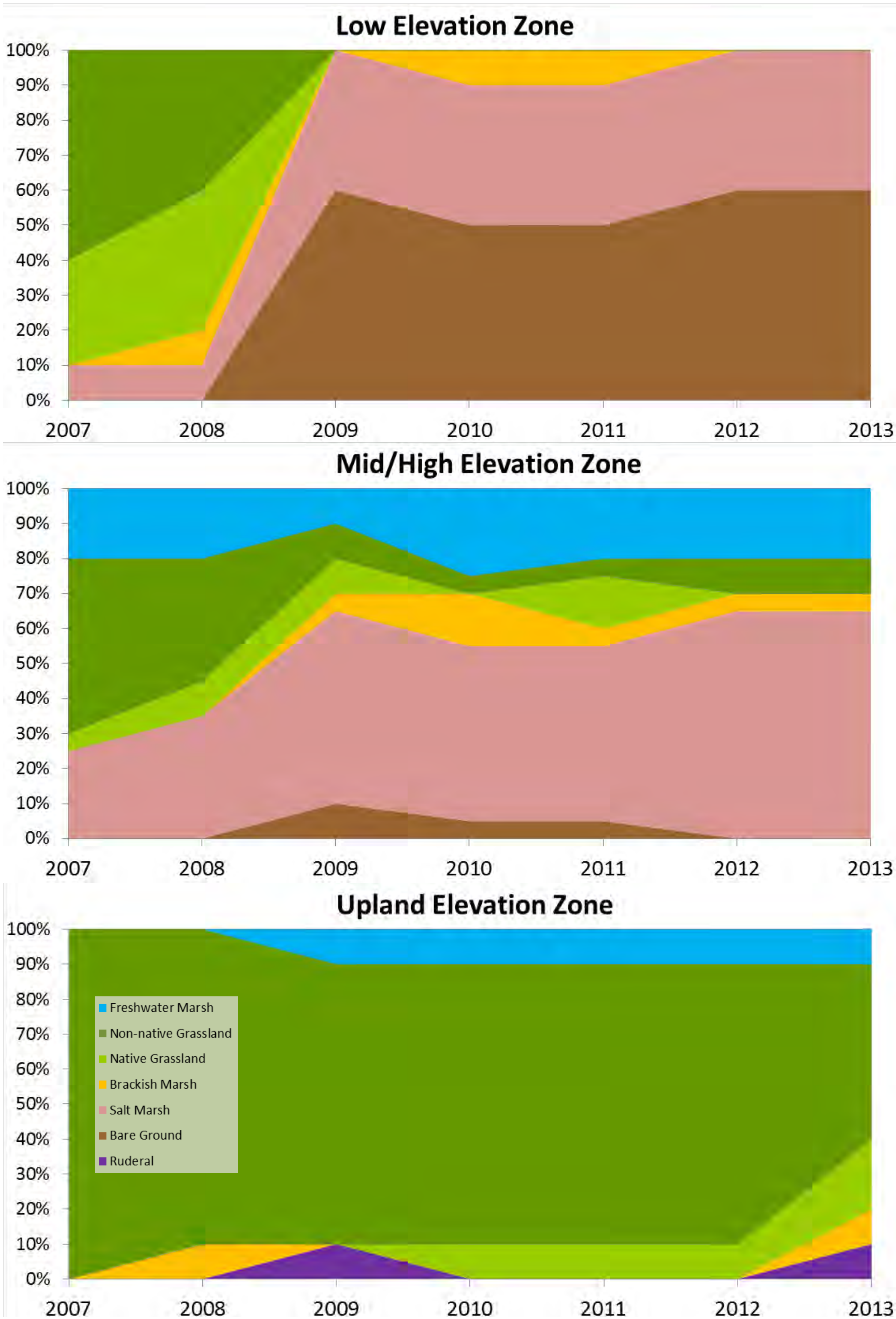

Figure 24. Percent of transects in each community category in the three elevation zones, 2007-2013.

Post restoration, starting in 2009, we saw a sharp (37%, 13 transects) decline in the number of transects dominated by grasslands and a concurrent increase in Salt Marsh (doubling from 20% to 38%, 8 to 15 transects) and Bare Ground (from 0 to 20%, 8 transects) sites. Since 2009, there has been a slight decrease in Bare Ground, as some of the areas impacted by introduction and persistent ponding of saltwater start to recolonize with vegetation, although there continues to be year-to-year variation in the number of sites dominated by Brackish Marsh, Native Grassland, and Ruderal/Weedy species. The Freshwater Marsh sites appear to have increased slightly with one more site added with the construction of the Tomasini Triangle mitigation marsh in a formerly grassland site. This is slightly misleading, however, as two transects that are classified as freshwater due to the continued dominance of cattails are probably brackish rather than fresh. Cattails have some ability to tolerate brackish waters, especially once established, and they have been able to survive albeit with reduced biomass. This may change over time, and more brackish species may come to dominate the area of these transects.

No communities were lost from the Project Area, rather a new community (or habitat type), Mudflats, was added. Further, while the most common community changed from grassland to Salt Marsh, it was not a one-to-one change over the monitoring period. Instead, the restored marsh is a more even mixture of a diversity of communities. This greater diversity of communities can in turn be inferred to support a greater diversity of other types of species, both plant and animal.

***Elevation Zone Differences*:** The low elevation zone transects (n=10) transitioned almost immediately from 90% grassland to 60% bare ground (Mudflat) and 40% Salt Marsh, and the same proportion of these habitats were still present in low elevation zones in 2013 (Figure 24). The years 2010 and 2011 saw some Mudflat replaced by Brackish Marsh: these years were significantly wetter than any other in the study period (Table 2), allowing annual brackish species, in particular brass buttons (*Cotula coronopifolia*), to colonize areas otherwise too salty for them.

Vegetation within the mid-elevation range (Figure 24), representing those transects (n=20) within the elevation of the target reference marsh areas, changed, but the change was less dramatic. Pre-restoration, Salt Marsh accounted for 25% of transects within the mid/high marsh elevation, but grasslands were actually the dominant communities with 55% of cover. In 2009, immediately post-restoration, Salt Marsh vegetation dominated 50% of transects, and 10% and 5% of transects were transformed to bareground (Mudflat) and brackish marsh, respectively. By 2013, salt marsh vegetation had colonized sites that initially transitioned to mudflat, dominating 65% of transects. Brackish and Grassland vegetation expanded slightly in the wet years of 2010 and 2011, but in 2013 they occupied 5% and 10%, respectively. Transects dominated by Freshwater Marsh shifted around somewhat, but ended up representing approximately the same percentage of transects in 2013 as 2007: 20%.

Unlike the Project Area transects, the reference marshes within the mid elevation zone exhibited no diversity in the dominant vegetation of the mid/high marsh zone: 100% of transects were classified as Salt Marsh in every year sampled (data not shown). In part, this is because park staff deliberately selected mid/high marsh areas within natural marshes for reference sites, because, based on anticipated topography of the restoration area, most of the site would fall within these elevation-associated habitat zones after restoration.

The upland elevation zone (n=11; Figure 24) changed the least of the three zones in that it continues to be dominated by grassland, however, the type of grassland has shifted, and other communities are also found. Non-Native Grassland-dominated transects have gone from 100% to 50%, with Native Grassland - dominated transects increasing to 20% and Brackish Marsh, Freshwater Marsh, and Ruderal transects increasing from 0 to 10%.

Table 2. Yearly precipitation totals (Bear Valley)

	2007	2008	2009	2010	2011	2012	2013
Total Precipitation	29.23	31.89	31.36	47.95	49.09	33.26	30.67

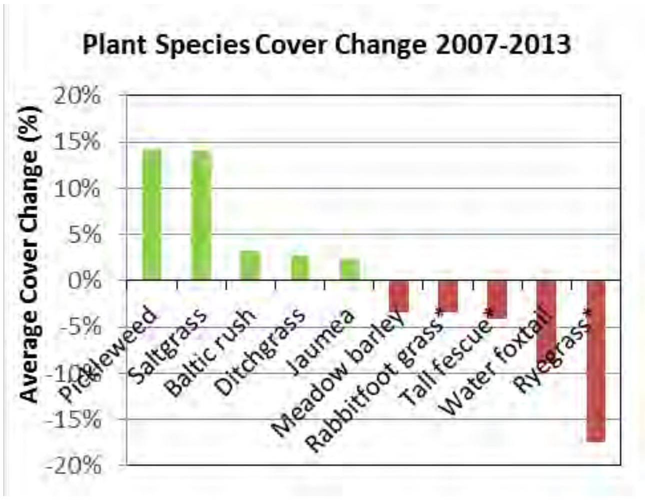

Figure 26. Species most increased/ decreased
The five species that increased (green) and five species which decreased (red) the most between 2007 (pre-restoration) and 2011 (three years post restoration). Non-native species are indicated with an asterisk.

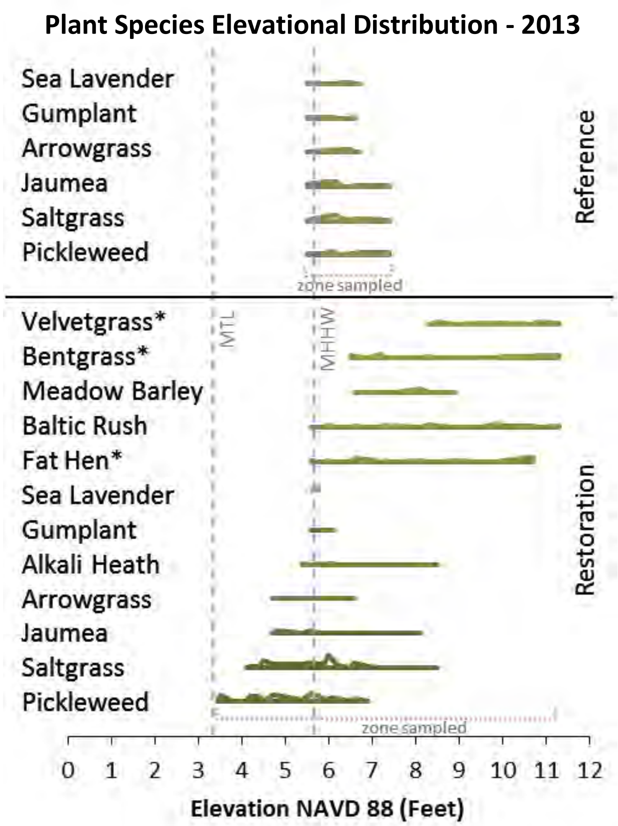

Figure 25. Species distribution by elevation
Comparison of the elevational distribution and abundance of the twelve most common species in the Project Area and the six most common species in the Undiked Marsh reference site. Sampling elevation range is indicated in red below the graphs, as in some instances sampling occurred in only part of the species

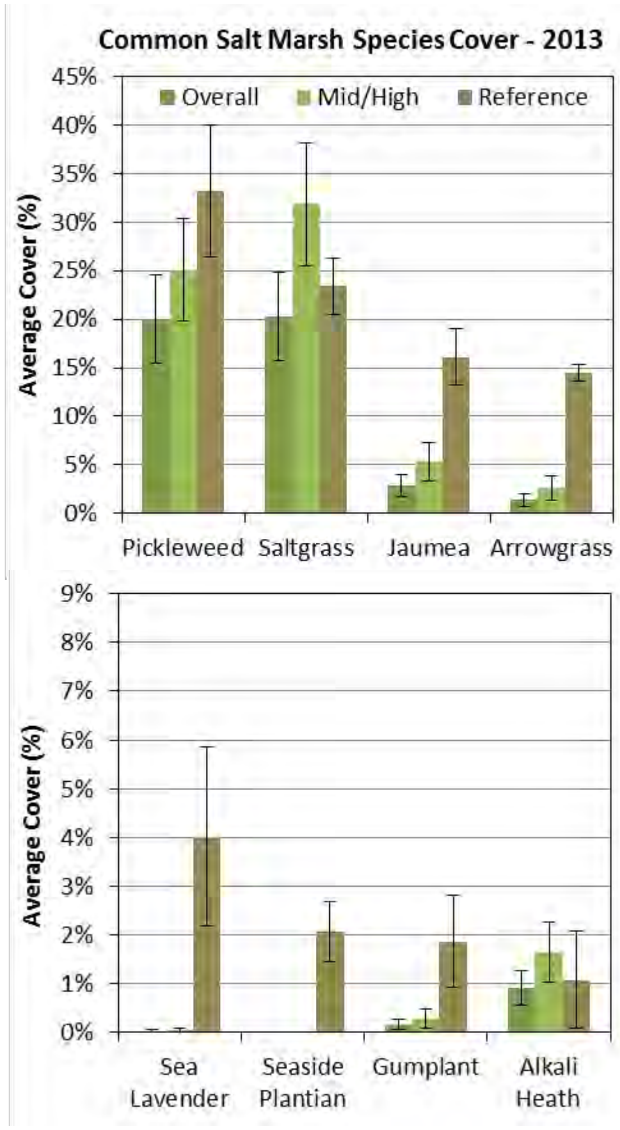

Figure 27. Common salt marsh species by zone
Comparison of the 2013 cover of the eight most common salt marsh species in the Project Area overall, in the Mid/High Marsh zone, and in the reference marshes for (A) dominant salt marsh species, and (B) common, but lower cover salt marsh species.

Individual Plant Species

With the changing conditions at the site, the identities and abundance of the plant species present at the site has changed. All of the species that increased the most – pickleweed (*Salicornia pacifica*; formerly *Salicornia virginica*), saltgrass (*Distichlis spicata*), Baltic rush (*Juncus balticus* x *lescurii*) ditchgrass (*Ruppia maritima*), and jaumea (*Jaumea carnosa*) are native salt or brackish marsh plants (Figure 25). All of the species which have most decreased are grasses, and three of the five, tall fescue (*Festuca arundinacea*), rabbitfoot grass (*Polypogon monspeliensis*), and ryegrass (*Festuca perenne*, formerly *Lolium perenne*) are non-native (Figure 25). Two native grasses, water foxtail (*Alopecurus geniculatus*) and meadow barley (*Hordeum brachyantherum*) - adapted to fresh or brackish conditions, also decreased in average cover.

We compared the cover of the eight regionally most common salt marsh species (as represented in the reference marshes) in the Project Area and in the Mid/High marsh zone to the cover of these species in the references marshes (Figure 27). Four species were very common (14-34% average cover) in the reference marshes (regionally dominant): pickleweed, saltgrass, jaumea, and arrowgrass (*Triglochin* spp.). Four other species were common, but not dominant: sea lavender (*Limonium californicum*), seaside plantain (*Plantago maritima*), gumplant (*Grindelia strica*), and alkali heath (*Frankenia salina*). The Project Area overall had lower cover than the reference marsh of all salt marsh species except saltgrass and alkali heath. The mid marsh zone did not differ significantly from the reference sites in the two most common species (pickleweed and saltgrass), but less common species were significantly lower in cover in the restored marshes, with the exception of alkali heath, which was equivalent in cover to the reference marshes.

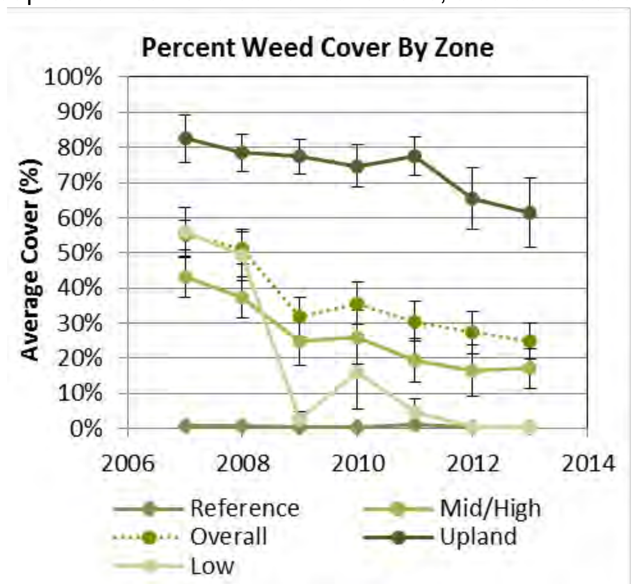

Figure 29. Non-native plant cover 2007-2013
Cover values for the Restoration and Reference sites, overall and by elevation range

Plant species also showed strong zonation trends, with picklweed exibiting the highest tolerance for low elevation site, growing down to mean tide level (MTL), followed by saltgrass, jaumea, and arrowgrass (Figure 26). High marsh plants such as alkali heath, guumplant, and sea lavender were present only slightly below mean higher high water (MHHW), where as meadow barley and bentgrass grew in elevations submerged by only the highest tides. Velvetgrass grew completely above the reach of the tides.

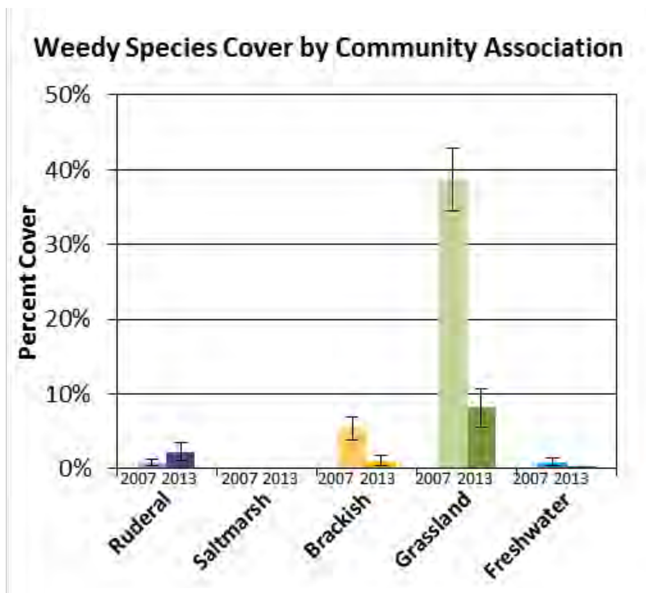

Figure 28. Non-native plant cover by plant community with the Project Area between 2007 and 2013.

Non-Native Plant Cover

Just as the species that increased the most were natives, and most of those that decreased were non-native, the site overall has become less weedy since restoration. The cover of non-native plant species has decreased since 2007 (Figure 28), including during passive restoration in 2008 (p=0.06) and each subsequent year (p<0.0001 for all). In 2013, non-native cover was significantly lower than 2007, 2009, and 2011 ($p<0.0001$, $p=0.006$, $p=0.05$ respectively), but has remained much higher than the reference sites (Figure 28).

In 2010, there was a spike in the weediness of the low elevation zone. This was almost entirely caused by an explosion of a brackish-tolerant weed called brassbuttons (*Cotula coronopifolia*).

As of 2013, the mid/high marsh is still significantly weedier than the references marshes. Two brackish-tolerent weeds, creeping bentgrass (*Agrostis stolonifera*) and fat hen (*Atriplex triangularis*) together accounted for 73% of weeds in this elevation zone. The prevalence of these two brackish weeds may reflect the fact that the average soil salinity of the site is much lower than that of the reference site, averaging 23 ppt +/- 3 SE (range: 0-37 ppt) versus 37 ppt +/- 3 SE (range: 18-75 ppt). In general, water quality monitoring results have indicated that Giacomini is inherently more brackish in nature than any of the reference sites due to the higher influx of freshwater that enters the site from upstream creeks, as well as groundwater from adjacent uplands.

In 2013, average non-native plant cover had decreased in every community except Ruderal relative to 2007 (Figure 29). Salt Marsh communities had the lowest average cover of non-native plant species, but non-native cover was still significantly higher than than in the reference marshes.

Diversity

α–diversity

Five years post restoration, overall α-diversity at the site, as estimated by the Shannon Index, remains significantly lower than in 2007 prior to restoration (paired samples t-test; $p<0.001$; Figure 30A). It has increased significantly (paired samples t-test; $p=0.001$; Figure 30A), however, from the low point in 2009, immediately post restoration. The mid/high marsh elevation of the restoration area had the highest average diversity in 2007, significantly higher than the lower elevation sites pre-restoration, though not differing significantly from the upland sites. Five years post restoration, the upland zone has significantly higher diversity than the mid/high marsh elevation zones (t-test; $p<0.001$; Figure 30A), which in turn is significantly more diverse than the low marsh elevation zone (t-test; $p=0.003$; Figure 30A). The upland zone has approximately the same diversity as pre-restoration, though it has fluctuated somewhat from year to year.

Shannon Index diversity declined sharply post-restoration in the mid/high and low marsh elevations (paired samples t-test; $p<0.001$ for both; Figure 30A), but has increased from the 2009 low. Average diversity increased each year since restoration in the mid/high marsh sites, except in 2013, which did not differ noticeably from 2012. However, average diversity at the reference sites declined slightly over this same time period. Low elevation marsh sites seemingly had highest average post-restoration diversity in 2011, though 2013 was not significantly lower.

As with Shannon Index diversity, pre-restoration low marsh sites had significantly lower species richness (t-test; $p=0.06$; Figure 30B) than mid/high marsh or upland elevation zones. Reference marshes had lower average species richness than the pre-restoration Project Area (t-test; $p<0.001$ in 2007 and 2008). Mid/high- and low marsh elevation zone species richness dropped sharply with the post restoration die-off of vegetation. Five years post-restoration, average species richness across the site averages slightly higher, but not significantly so. Both low marsh and upland elevation zone species richness has fluctuated from year to year, whereas mid/high marsh elevation zone has climbed significantly since 2009 (paired samples t-test; $p=0.03$). The mid/high marsh elevation - and the site overall - did not differ significantly from the reference marshes in species richness in 2013.

Shannon index and species richness decreased most in the freshwater marsh community (Figure 31). Plant communities were characterized by dominant vegetation, but several freshwater marsh sites became slightly brackish post-restoration. Some species such as cattail and bulrush, while most frequently found in freshwater marshes, are able to tolerate oligohaline conditions, where as many

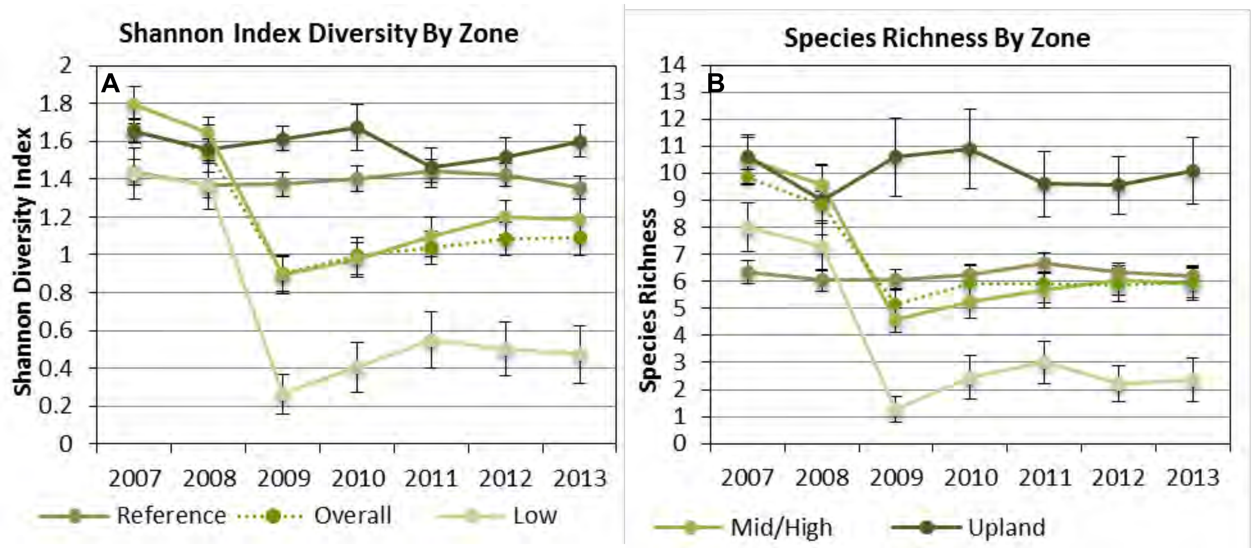

Figure 30. Average restoration Shannon Index site diversity (A) and Species Richness (B)
Project area overall, low- and mid-marsh, and upland values for 2007-2013. Error bars represent +/-1 S.E.

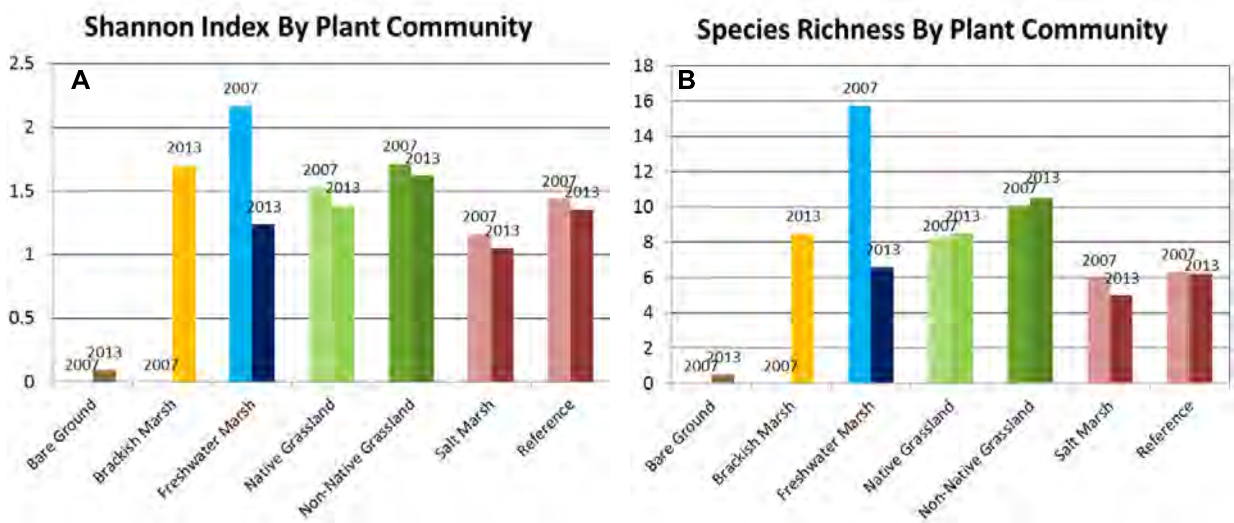

Figure 31. Diversity in terms of Shannon Index (A) and species richness (B) by plant community.
Displayed data is for the Project Area in years 2007 and 2013.

other species are not. These Freshwater Marshes, now apparently low diversity, are in part actually brackish marshes where some freshwater-oligohaline species were able to maintain dominance, but most other, purely freshwater species died off.

The decrease in species diversity at the site is primarily driven by the decrease in non-native species. Figure 32 shows the average diversity of native plants species in the vegetation transects between 2007 and 2013 and the average diversity of non-native species between the same years. After restoration, average non-native species diversity was cut in half, dropping from ~0.9 to ~0.4-0.5 (paired samples t-test; $p<0.001$), where it has remained since 2009. By contrast, average native species diversity was lower than non-native species diversity in 2007, but decreased much less (by ~0.2 on the Index) after restoration and has increased each year since restoration. In 2013, the average native diversity of the West Pasture was equal to that in 2007 ($p=0.75$).

Figure 26 illustrates that plants at this site occur in specific elevational ranges, as is known to be the case in salt marshes in general. Salinity, as well as elevation, is known to be an important driver in salt marsh plant community structure (Cooper 1982, Pennings and Callaway 1992, Pennings *et al.* 2005). Table 3 depicts the correlation between diversity, salinity, and elevation in restored and reference marshes before and after restoration. Before restoration, there was no correlation between any of the factors in the Project Area, but there was a weak positive correlation between elevation and diversity in the reference marshes. In 2013, five years after restoration, diversity and elevation have a positive correlation, diversity and salinity have a negative correlation (higher diversity when soil salinity is lower), and elevation and salinity have a negative correlation (lower salinity at higher elevations). The reference site did not have any correlation between any of the three factors in 2013.

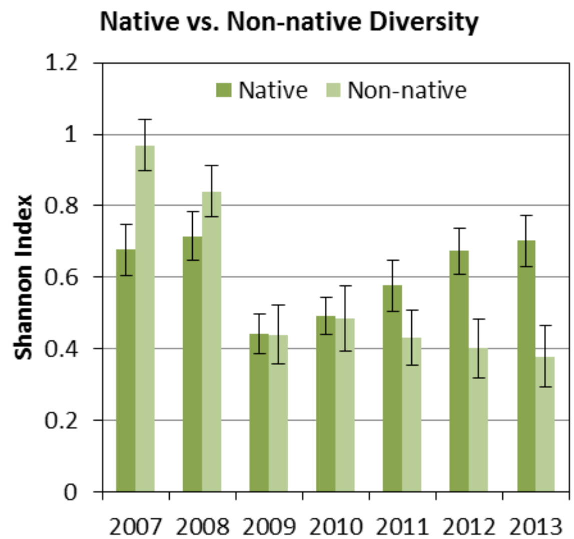

Figure 31. Native vs. Non-native Diversity Contribution to overall diversity of each site from native (left chart) and non-native (right chart) plant species between 2007 and 2011. Error bars represent +/- 1 S.E.

Percent Similarity

Similarity indices compare the overlap between plant species composition at different sites. Both unweighted (Sørenson or Bray-Curtis) and weighted (Czekanowski's) scores were calculated comparing the restoration site to the average of the three reference sites for each year. Unweighted results reflect similarity in species presence/absence data, whereas weighted data also takes abundance into account. A score of 100 indicates that the sites are identical.

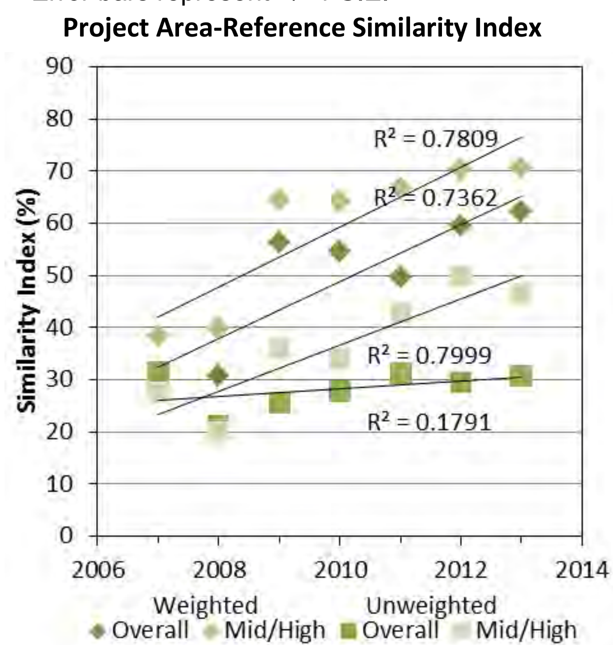

Figure 32. Project Area-Reference Similarity Weighted (Czekanowski's) and unweighted (Sørenson or Bray-Curtis) percent similarity of the Project Area as compared to the Reference Sites years 2007-2013.

The weighted percent similarity of the restoration and reference sites in 2007 prior to restoration was about 28, which indicated relatively little overlap between the sites (Figure 33). Each year since restoration, the overlap has increased, and the overall trend is strongly upward in both the site are a whole and in the mid/high marsh zone (R^2>0.7; Figure 33). In 2013, weighted percent similarity was 63, indicating that the site is becoming progressively more similar to the reference sites. Weighted percent similarity of the mid/high marsh elevation zone in the Project Area, which is the zone at the same elevation as the reference marshes, was 72. Unweighted similarity scores were lower than weighted scores, and the site overall showed no change in similarity over time, but the mid-high zone of the marsh had a strong upward trend (R^2>0.8; Figure 33).

When the Project Area and the reference sites are compared to other sites in the region (based on calculations from data in Vasey *et al.* 2012), some interesting patterns emerge (Table 4). The Project Area as a whole is actually more similar to Rush Ranch (33%), a large brackish marsh fairly far upstream

Table 3. Diversity, Salinity, Elevation Correlations

	2007		2013	
	Restoration	Reference	Restoration	Reference
Elevation Range (ft NAVD 88)	3.4-11.3	5.3-7.4	3.4-11.3	5.3-7.4
Diversity vs. Elevation	0.05	0.21 (+)	0.45 (+)	0.08
Diversity vs. Soil Salinity	0.11	0.06	0.32 (-)	0.00
Elevation vs. Salinity	0.14	0.15	0.62 (-)	0.00

Table 4. Unweighted Percent Similarity (Sørenson or Bray-Curtis) of the Project Area and Reference Sites to San Francisco Bay Marshes, calculated based on data from Vasey *et al.* 2012.

	Size	Salinity	Total Species #	Percent Similarity Restoration	Mid	Reference	UM	WCM	LIM
China Camp	125 ha	28±4	10	20	30	**55**	46	45	50
Petaluma	800 ha	27±4	17	27	30	44	36	34	37
Coon Island	175 ha	21±3	22	28	34	29	26	24	25
Rush Ranch	400 ha	6±2	37	**33**	38	32	34	29	17
Browns Island	200 ha	4±2	53	22	27	17	17	15	10
Sand Mound Slough	25 ha	0±0	55	24	24	11	11	9	3
Giacomini	225 ha	19±3	72	100	n/a	31	25	21	17
Mid	n/a	n/a	37	n/a	100	46	42	37	26
Reference Marshes	n/a	30±5	13(av)	31	46	69	n/a	n/a	n/a
UM	60 ha	26±4	16	25	42	n/a	100	71	53
WCM	30 ha	34±1	12	21	37	n/a	71	100	82
LIM	15 ha	31±3	10	17	26	n/a	53	82	100

Table 5. Comparison of unweighted (Sørenson or Bray-Curtis) and weighted (Czekanowski's) percent similarity between restored marshes and references for the Project Area, for the mid/high marsh area, and as reported in other studies. The similarity between the three reference marshes sampled in this study is also reported.

Location	Project Type	Years restored	Similarity Index Weighted	Unweighted	Source
Giacomini Wetlands, Calif., USA	Tidal, Levee Removal - Overall	5	62	31	
	Mid/High Marsh	5	71	46	
	Reference Marshes	NA	90 (85-96)	69 (54-82)	
Elk River, Wash., USA	Tidal, Levee Breach - planned	4-5 6-11	31-50 57 (47-78)	42-60 67 (52-78)	Thom *et al.* 2002
Bay of Biscay, Spain	Tidal, Levee Breach - accidental	20-35		45-46	Onaindia *et al.* 2001
Essex Estuaries, UK	Tidal, Levee Breach - accidental	2 9-13 51-107		8 39 (35-44) 66 (41-99)	Garbutt and Wolters 2008
Essex Estuaries, UK	Tidal, Levee Breach - accidental	100	95	82	Crooks *et al.* 2002 *in* Byers and Chmura 2007
Bay of Fundy, NB, Can.	Tidal, Levee Breach - accidental	50-60	70-90	50-75	Byers and Chmura 2007
Mass., USA	Mitigation wetlands	0-5 6-15	14 (0-32) 14 (0-78)		Brown and Veneman 1998

in San Francisco Bay-Estuary, than to the reference marshes in Tomales Bay and Limantour Estero (31%, based on average species values, or 17-25%, based on individual marshes).The reference marshes are most similar in composition to China Camp marsh (45-50%), a marsh much closer to the mouth of San Francisco Bay. Vasey *et al.* also reported salinity data for their sites, calculated from summer and fall salinities between 2008 and 2010. Project Area and reference marsh data from the same period show that the reference marshes are more similar in salinity and species number to China Camp marsh than to the Project Area, and the salinity of the Project Area lies between that of Rush Ranch and Coon Island, though close to the latter, which is the site second most similar to the Project Area (28%). Species wise, the Project Area is more diverse than even the most diverse San Francisco bay site – though in part due to the larger range of elevation represented in the sampling of the Project Area as a whole relative to the San Francisco Bay Marshes. The mid zone of the marsh is most similar to the adjacent Undiked Marsh and more similar to the reference marshes as a whole than to either Rush Ranch or Coon Island, but is equivalent to Rush Ranch in species number.

As discussed earlier, various studies have compared restored (sometimes accidentally) marshes to naturally occurring reference marshes (Table 5). Results are highly variable, with some restored marshes remaining very different from reference marshes, even after more than 100 years, and others achieving a high degree of similarity. When weighted percent similarity in the mid marsh and reference marshes is compared to that found in other projects of varying age, similarity was much higher than other sites of <5 years of age (in Massachusetts and Elk River, Washington), and on par with or greater than sites 6-15 years old in Elk River and Massachusetts, and a 50 year old site in the Bay of Fundy in Canada. In unweighted similarity, the Project Area was comparable to sites of similar age in Elk River, 20-35 years old in the Bay of Biscay, Spain, and 9-51 years old in Essex Estuaries in the UK.

Biomass

Aboveground biomass at each vegetation transect is estimated by collecting a 0.25- by- 0.25 meter sample at a random location along, but slightly offset from, the transect. The vegetation is clipped down to the soil surface, and any leaf litter or other organic material is also collected. The samples are then dried and weighed. Plant belowground biomass was not collected due to the difficulty of separating roots from the soil, but it is also an important component of the functions vegetation provides within an ecosystem.

The average biomass of the site since restoration is slightly higher than it was prior to restoration, increasing from an average of 1574 ± 173 g/m² to 1972 ± 311 g/m² (Figure 34; paired samples t-test, p=0.10). Average biomass rose in 2008 (paired samples t-test, p=0.09 compared to 2007) when grazing was removed, but peaked in 2009, the first year after to restoration, which had higher biomass than in 2007(paired samples t-test, p=0.08) and 2010 (paired samples t-test, p=0.05). The peak in 2009 probably is due in large part to a high fraction of dead biomass from the die-off of glycophytic (freshwater) vegetation when tidewater was introduced, rather than live fraction. The decline in average biomass in the three years post restoration likely reflects the decomposition of the dead fraction. This supposition is supported by the peak in cover of shrub layer vegetation in 2008, followed by a sharp decline in 2009.

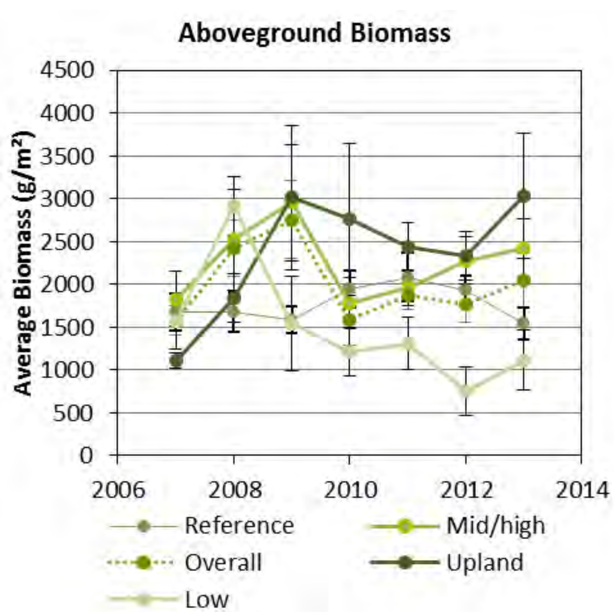

Figure 33. Project Area aboveground biomass Average aboveground biomass within the Restoration Project overall, and by elevation zone, as compared to the Reference Marshes 2007-2011. Error bars represent +/- 1 S.E.

Pre-restoration biomass on the restoration site overall did not differ from that at the three reference marshes (t-test, assuming eq. var., p=0.77), nor did it differ by elevation zone (t-test, assuming eq. var., p>0.15 for all). Five years post-restoration, site biomass overall trended higher than the reference site ((t-test, assuming eq. var., p=0.08), driven by a significant increase in average biomass in the upland zone relative to 2007 (paired samples t-test, p= 0.05) that outweighs a trend of declining biomass in the low marsh elevation zone (paired samples t-test p=0.12). The low marsh elevation zone, which is mostly mudflat, supports some biomass in the form of algal mats as well as some vegetation. While not significantly lower in 2013 than under grazed conditions in 2007, biomass in the low marsh elevation zone is much lower than the passive restoration phase (2008) when the low marsh elevation zone was dominated by ungrazed grassland (paired samples t-test, p=0.005; Figure 24).

Average biomass in the mid/high elevation zone has steadily increased since a 2010 low, but does not differ significantly from pre-restoration biomass (paired samples t-test, p=0.11). Overall, the site appears to be slightly more productive now than in the pre-restoration grazed state, driven by a significant increase in biomass in the upland, while the low and mid/high marsh elevation areas have not changed significantly. These patterns do not change even when average biomass from each zone is weighted by the percent of the Project Area it occupies (Table 3). Still, it is clear that biomass patterns have changed in the Project Area from similar biomass across elevation zones to average biomass increasing with average elevation.

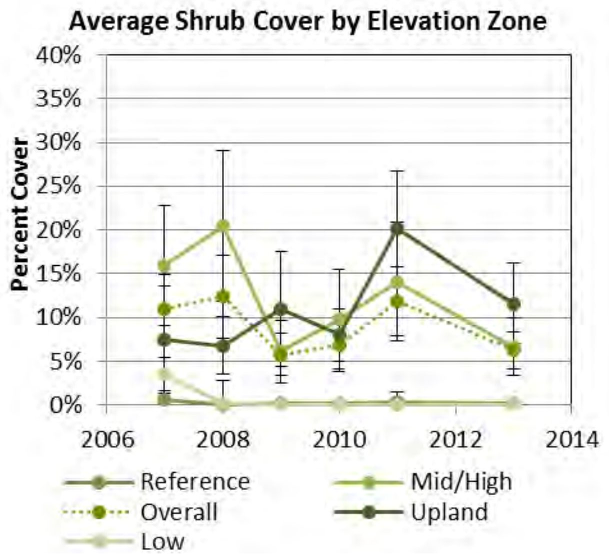

Figure 34. Average shrub cover by zone & year Shrub Cover on the Project Area and Reference sites between 2007-2013. Error bars represent +/- 1 S.E.

Vertical Structure

Vertical structure is a measure of how much of the Project Area is covered by plants of different heights. Most of the Project Area was covered by either herbs (plants less than 0.75m) or shrubs (plants between 0.75-2.5 meters in height). For the purposes of assessing vertical structure, the term “shrub” refers only to the height of the plant, not the presence of a woody stem.

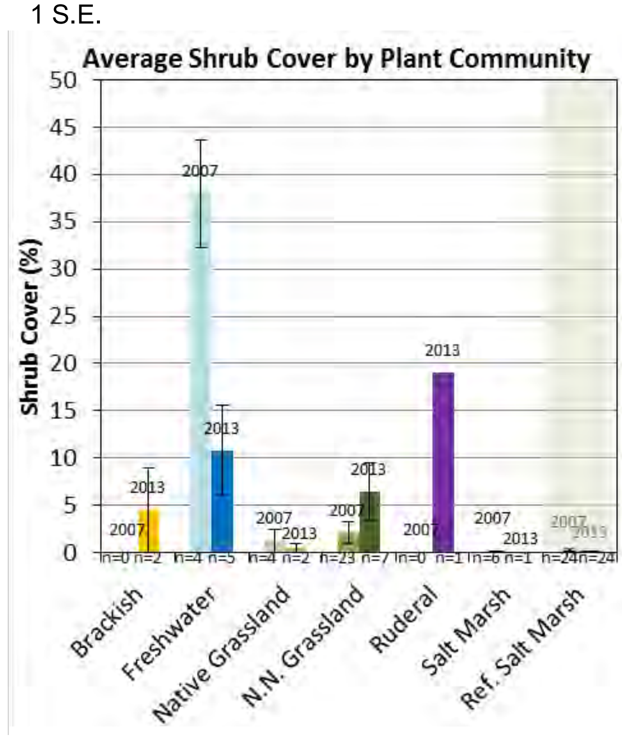

Figure 35. Average shrub cover by community restoration site by community in 2007, prior to restoration, and 2013, five years after restoration. Error bars represent +/- 1 S.E.

Prior to restoration and during passive restoration, shrub cover hovered at around 12% in the Project Area (Figure 35). Shrub cover dropped dramatically in 2009, after the first year after restoration, and then climbed in 2011, which was the wettest year (Table 2) during either the pre or post-restoration monitoring period. However, shrub cover in 2011 was not significantly different from 2007 or 2008, prior to full restoration. Shrub cover dropped to 2009 levels in 2013. The fact that 2013 was the driest year on record in California (CDWR 2014) may have led to less biomass production. The reference marshes, by contrast, are almost completely lacking in shrub cover despite the high visibility of the salt marsh perennial shrub gumplant. Within the marshes, shrub height portions occupy a surprisingly low aerial cover and have remained significantly lower in three-dimensional structure than the restoration site both before and after restoration.

The distribution of taller plant cover communities has changed, however (Figure 36). Areas labeled as salt marsh had low cover in both 2007 (before restoration) and 2013 (after restoration), but the number of transects classified as salt marsh tripled, and these decreased in average shrub cover after the restoration. On the other hand, grassland transects decreased in number after the restoration, but increased in average shrub cover because they are no longer grazed. The number of sites dominated by freshwater marsh plants increased slightly, but there was a slight decrease in average shrub cover in these transects. This may be because the freshwater marsh sites are now subject to occasional intrusion of brackish water, making growing conditions more stressful for freshwater species, such as cattails and sedges that do not grow as tall as they would under glycophytic conditions.

Canopy Complexity

Canopy complexity is another way of referring to the relative cover of transects. Since all species encountered in each layer at each point are recorded, the cover of a particular transect may add up to more than 100%. This reflects the degree of layering of vegetation at each point along the transect, or canopy complexity.
Average transect canopy complexity (Figure 37) declined both overall ($p<0.0001$) and in the mid-elevation zone ($p=0.0008$). Canopy complexity was also significantly lower in the low elevation zone ($p<0.0001$; data not shown), reflecting that this area went from pasture to largely unvegetated mudflat, but did not change in the upper elevation above the tides ($p=0.38$; data not shown). The Project Area overall and the mid-marsh zone had significantly lower canopy complexity than the Undiked Marsh (t-test, un-equal variances, $p<0.0001$ for both), and, marginally lower than Walker Creek as well (t-test, un-equal variances, $p=0.03$, overall and $p=0.07$, mid). There was no significant difference between the Project Area overall or mid elevation zone and Limantour marsh, suggesting that it is within the range of that found in natural marshes.

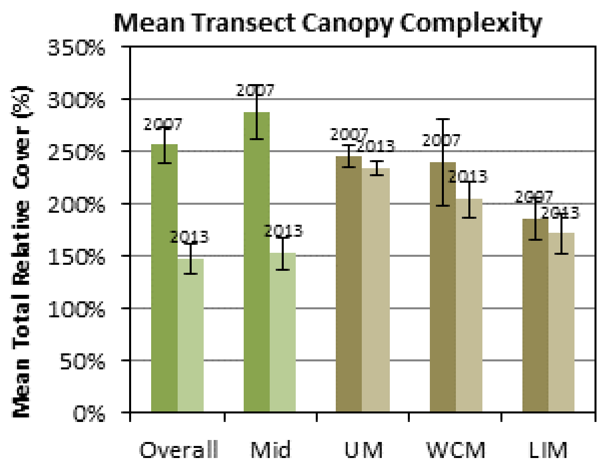

Figure 37. Mean Canopy Complexity, 2007 & 2013
Data from Project Area (overall and mid) and Reference Sites before and after restoration. Error bars represent +/- 1 S.E.

Bareground

The Project Area overall had significantly more bareground in 2013 than 2007 ($p=0.0003$; Figure 38). The high percent cover of bare ground in 2013 is in large part due to the expansive mudflat in the low marsh, but the mid-marsh also had significantly more cover of bareground in 2013 than 2007 ($p=0.05$) and thus contributed to the higher cover in the Project Area overall. Walker Creek and Limantour Marshes also had significantly more bareground in 2013 than 2007 ($p<0.02$), but the Undiked Marsh had no change in bareground.

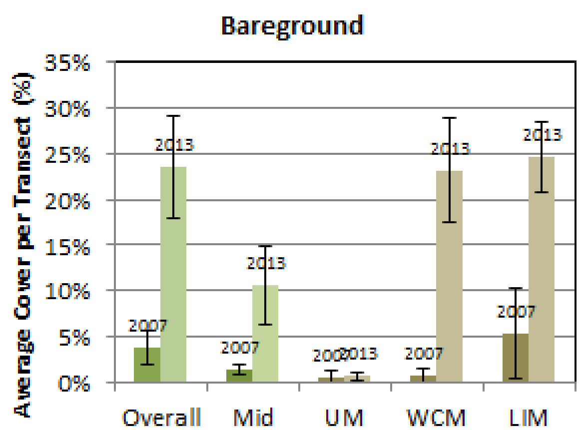

Figure 37. Mean Bareground Cover, 2007 & 2013
Data from Project Area (overall and mid) and Reference Sites before and after restoration. Error bars represent +/- 1 S.E.

Walker Creek and Limantour Marshes appeared to have experience some vegetation die-back and reduced leaf production in 2013 (Amelia Ryan, pers. obs.), which may reflect the exceptional drought conditions. In 2013 the Project Area overall had significantly more bareground than the Undiked Marsh reference site, but not Walker Creek or Limantour marshes.

Results Weighted by Elevation Zone Area

Results presented to this point were averaged by elevation, and across the Project Area as a whole, but while elevation zone heavily influenced plant community, the transect number is not proportional to unit area of each elevational zone. The low marsh zone occupies 37% of the Project Area and contained 10 transects. The mid-marsh zone also occupied 37 percent of the Project Area and had 16 transects. The high marsh occupied 26% of the Project Area and had 11 transects. We therefore examined the way weighting the scores for the whole Project Area for the relative proportion of each of the three zones affected the metrics for describing plant community change (Table 6).

We found that is made relatively little difference to the overall patterns. When weighted for elevation zone area, weediness appears to have declined a bit more (from 58% to 23% pre- to post-restoration) than when transects are averaged unweighted (54% to 25%). Biomass showed a similar trend to weediness. Diversity, species richness, and vertical structure went down very slightly for the sight as a whole: both pre- and post-restoration vales are slightly lower. Overall, weighting by the area of the three zones changed numbers by a fairly small amount and had only a slight effect on overall trend.

Table 6. Comparison of data averages in 2007 and 2013, and unweighted and weighted by percent of Project Area occupied by each zone. Averages in Low and Mid/High zones were weighted by 0.37 and Upland by 0.26.

	Year	Low	Zones Mid/High	Upland	Unweighted Transect Average	Weighted Transect Average	Reference
Weed %	2007	55	43	83	54	58	1
	2013	0	18	61	25	23	1
Diversity	2007	1.41	1.8	1.64	1.64	1.61	1.41
Shannon Index	2013	0.47	1.2	1.6	1.1	1.0	1.38
Species	2007	8	10.5	10.5	9.8	9.6	6.3
Richness (#)	2013	2.3	6.0	10.1	6.0	5.7	6.1
Biomass	2007	1600	1800	1150	1600	1557	1650
g/m^2	2013	1000	2400	3000	1950	2038	1600
Vertical	2007	4	16	7	11	9	1
Structure %	2013	0	6	12	6	5	0

Soil Analytes

Five soil analytes important to vegetation community dynamics (Total Kjeldahl Nitrogen (TKN), phosphorous, organic matter, salinity, and percent clay) are displayed in Figure 39. The reference marshes differed from each other markedly in TKN, organic matter, and percent clay. Even given that, the upland zone of the restoration Project Area was different from the reference marshes in almost all measures. The restoration project overall -- and the mid and low zones -- were within the range of the reference marshes in every measure, except salinity, in which the restoration Project Area continues to average much lower.

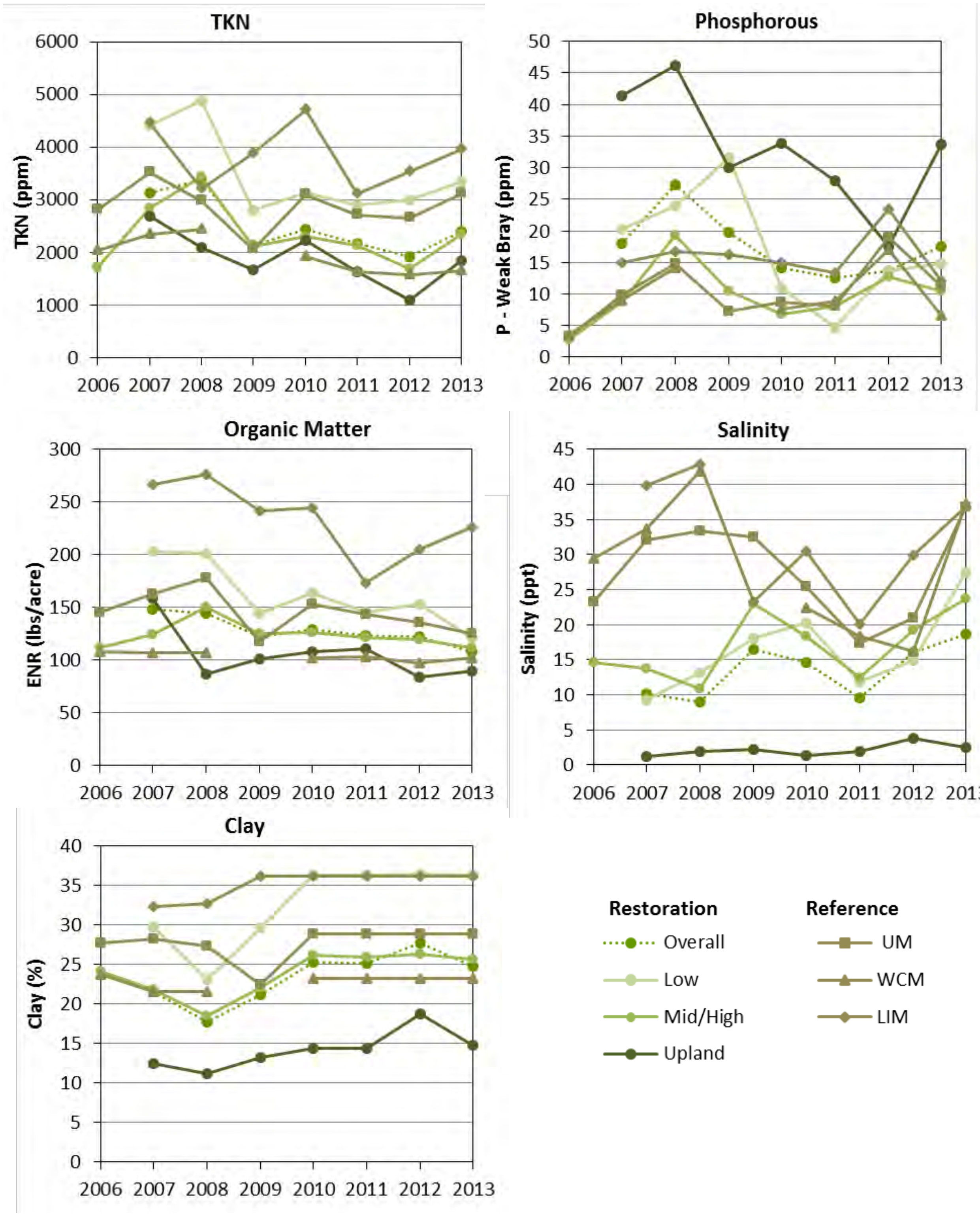

Figure 38. Selected Soil Analytes for the Project Area and Reference Marshes
Mean values displayed from 2006-2013 for the Project Area overall and by elevation zone (low, mid/high, and upland), and for each of the three reference marshes; the Undiked Marsh (UM), Walker Creek Marsh (WCM) and Limantour Marsh (LIM).

Vegetation Assembly Along Channels

There was no significant difference between sites that were scraped or unscraped, nor did plots that were directly adjacent to channels differ signifcanly in diversity from those on the marsh plain (23 m from channels).

The two sites closest to the channel (0-1 m and 6-7 m) were lumped for analysis. Sites were further broken into primary succession and secondary succession based on status immediately after restoration. Sites that had glycophytic vegetation that died off at restoration of tidal flow, leaving a “clean slate” for succession were classified as primary, and sites that had halophytic vegetation already present (albeit with a muted tidal range) pre-restoration were classified as secondary succession sites. These sites are not directly comparable, as they were not randomly distributed. All of the six (6) secondary succession areas occurred in the West Pasture, and all but one of 12 primary succession sites occurred in the East Pasture. The sites were all aligned along major channels within the Project Area or the immediately adjacent Undiked Marsh. The sites used in analysis were all within the mid marsh elevational zone. Primary and secondary successional sites did not differ in average salinity, though they did average slightly lower than reference sites (~1-2 ppt).

Primary successional sites were dominated by pickleweed, followed in cover by saltgrass and jaumea Figure 40). Within one year after restoration, pickleweed dominated 34% cover of the channel zone, and increased each year. By 2013, it had increased by an average of 53%, significantly higher than 2009 ($p=0.01$). By contrast, saltgrass occupied only 6% cover initially, but nearly quadrupled in cover to 41% by 2013, significantly greater than 2009 ($p<0.01$). Jaumea occupied less than 1% cover in 2009, but by 2013 occupied nearly 26%. Arrowgrass and gumplant had nearly zero cover in 2009, but were present in low numbers by 2010. In 2013, they had increased to more than 9%. Secondary succession sites had no significant differences in 2013 from 2009, despite an apparent decrease in saltgrass and increase in jaumea. The reference sites (along channels in the adjacent Undiked Marsh) did not differ between the two sample periods of 2010 and 2013, except in a significant drop in arrowgrass ($p=0.0003$).

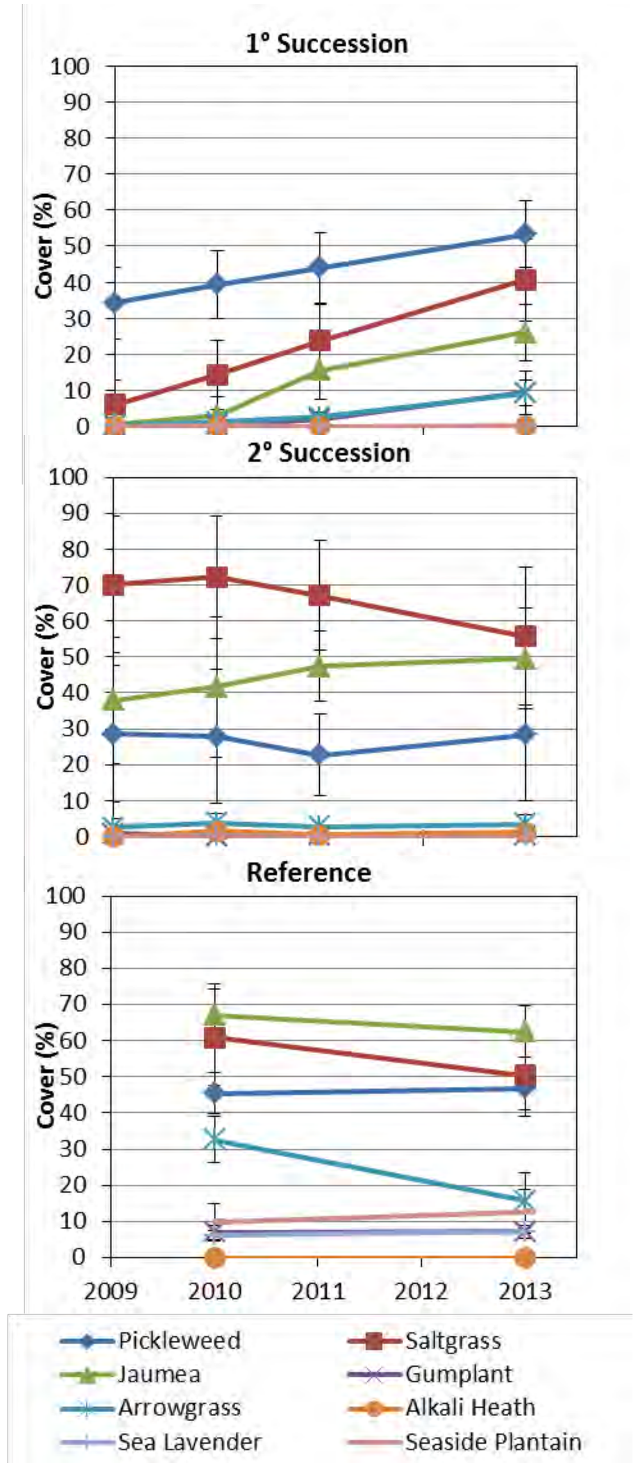

Figure 39. Assembly of common plant species
Assembly data displayed for the eight most common speces, collected along channels in the Project Area in mid-marsh areas of 1° and 2° succession, and in reference sites.

Two years after restoration, in 2010, the primary successional areas had significantly less cover than the reference site of all species except pickleweed and alkali heath, but, by 2013 only jaumea, sea lavender, and seaside plantain differed significantly from the reference site. In 2010, secondary successional sites had less gumplant, jaumea, sea lavender, seaside plaintain, and arrowgrass than reference sites. In 2013, jaumea, along with pickleweed, saltgrass, and alkali heath did not differ significantly from the reference sites, but the other species were still significantly less common. Primary and secondary successional sites differed significantly from each other in saltgrass and jaumea cover in 2009, but, by 2013, did not differ significantly from each other in any of the common species. They likewise did not differ from each other in terms of alpha diversity or weighted percent similarity. They both had significantly lower alpha diversity than the reference site, however, though weighted percent similarity was high (96-97%).

Discussion

As discussed in the introduction, many prior studies have documented the failure of restored marshes to resemble nearby naturally formed marshes sites (Zedler & Lindig-Cisneros 2000, Palmer *et al.* 1997, Wolters *et al.* 2008). This is very often expressed in terms of comparisons of plant community composition and diversity. For example, in the San Francisco bay region, 92% of historic marshes have been destroyed (Goals Project 1999), and the marshes that (partially) replaced them were much lower in diversity (Goals Project 1999, Boyer and Thornton 2012). For example, in their survey of 21 restored and remnant natural marshes in San Francisco Bay, Boyer and Thornton (2012) identified 20 plant species with potential to occur in marshes throughout the bay. Remnant historic marshes contained 14-19 of these species, averaging 17, whereas restored marshes contained only 7-9 of the most common species.

Five years out, the Giacomini Wetland Restoration Project Area as a whole has lower average diversity than it had pre-restoration –but this loss in diversity has primarily been due to a decrease in weedy species - and lower diversity than references marshes. However, weeds on the Project Area as a whole, including in the mid marsh zone, remain significantly higher than the reference marshes. This at least partly reflects the fact that the Project Area is significantly less salty than the reference marshes (Figure 39): none of the weedy species that occur in the Project Area tolerate truly saline conditions, though several are well adapted to brackish conditions (Figure 29). The lower site salinity and the legacy of agricultural management may prevent the site from attaining the extremely low weed cover of the reference sites.

In addition to changes in the soil, changes in the hydrology greatly affected plant establishment at the site. When tidewaters returned to the site, the influx of salty water immediately started killing most of the pasture vegetation that dominated the site, supported by diking, ditching, and irrigation. This created large swathes of mostly unvegetated habitat with large patches of dead or dying plant material. Secondly, due to deliberate undersizing of created tidal marsh channels to allow for more natural evolution, water could not exit the Project Area quickly enough during low tides to allow for complete drainage of waters, which encouraged prolonged ponding on the marsh plains and discouraged vegetation establishment. Another factor was that portions of the Project Area, particularly the northern portions, had subsided relative to the surrounding area when the site was converted from the historic tidal marsh into pasture. Diking and draining cause subsidence by exposing soils rich in organic material to decomposition, which is occurs more slowly when soils are saturated. New channel development may lag in the East Pasture due to the historical influence of agricultural management that has probably compacted dense clays present, similar to what was seen in newly restored agricultural lands in the U.K. (Wolters *et al.* 2005). Together these factors led to the formation of large mudflat pannes, which have persisted five years into restoration.

Yet, as documented in the benthic invertebrate and bird surveys associated with this restoration project (Sections 6,9, and 10), mudflats themselves can be very productive and diverse in benthic invertebrate and algal communities and provide very important food support to shorebirds. The lower plant species diversity in the Project Area compared to the reference marshes (Figure 30) in part reflects the greater *habitat* diversity now present at the site from the presence of mudflats (Figures 21 and 23). The presence of mudflats contributes to the diversity of different types of habitat present within the Project Area, and the lack of vegetation necessarily leads to low diversity within these areas. Overtime, as the hydrology has evolved, and despite historical management of the soils, some of the mudflat has become vegetated (Table 1), and more is likely to do so (though sea level rise will eventually counteract these changes).

When only the mid-marsh is considered (those areas of the Project Area that are in the same elevation zone as the reverence marshes) the Project Area is only slightly lower in diversity and no different in species richness than the reference marshes. The upland area zone is more diverse and species rich than the reference marshes.

Both weighted and unweighted percent similarity of the mid marsh to the reference marshes has risen over time (r^2=0.8), but it is not clear whether this upward trajectory will continue or level off. This project is on par or higher than many other restoration (or reclamation) projects of similar or greater age (Table 5), but in many instances, as in San Francisco Bay, these restored marshes never achieved complete overlap. In the instance of this restoration project, complete overlap, even in the mid marsh zone, only, may not be possible, or even desirable: the reference sites themselves do completely overlap, sharing 53-82% unweighted percent similarity. In addition, this Project Area differs in several key ways from the sites chosen as reference sites: it is larger, more topographically diverse, and significantly less saline, even in the mid-marsh zone (Table 4, Figure 39). Size and elevational variation can matter: Wolters *et al.* (2005b) found higher diversity to be associated with larger, more topographically diverse sites, but in another study size was found to be unrelated to species richness (Garbutt and Wolters 2008). When species richness for the site as a whole is taken into consideration, it is in fact considerably more species rich than any of the reference marshes, which had 10-16 species represented within our transects, whereas 72 species were present within the transects in 2013, 37 in the mid marsh zone alone (Table 4). Of the 20 species listed by Boyer and Thornton (2012), 16 were present in the Project Area, putting the restoration project on par with remnant historic marshes in San Francisco Bay, not other restored marshes. Direct comparison to the largest remnant tidal marshes in San Francisco Bay (Table 4) found that the larges brackish marsh Rush Ranch was more similar to the Project Area as a whole than any of the reference marshes.

Restored marshes are more often more diverse where they are next to prehistoric remnants (Boyer and Thornton 2012). This is one explanation of the relative diversity of the Giacomini Wetlands in comparison to other sites in the region: it is both adjacent to a large, diverse natural marsh directly north of, and contiguous with, the Project Area (the Undiked Marsh reference site), and has functioned in part as its own prehistoric remnant. Proximity to seed source is an important variable in marsh diversity (Wolters *et al.* 2005a). As shown by the vegetation mapping and transect monitoring, pockets of salt and brackish marsh persisted in the Project Area even while actively being managed as agriculture pre-restoration, despite the diking, filling, ditching, irrigating, and seeding to maintain gycophytic pasture. When the site was restored, these areas served as seed sources for primary succession in areas that were “wiped clean” of plants by restoration, and as both seed sources and sites for secondary succession in areas that were.

The post restoration formation of plant communities at the Project Area provides an interesting opportunity to examine the way in which plant communities form or “assemble.” A specific habitat may end up with a specific community (a “set endpoint”) no matter what seeds get to the site first, or the end community at the site may depend on which species colonize first or other chance events (Keddy 1999). One year after restoration, the vegetation assembly plots looked quite different from each other (Figure 40): pickleweed had quickly established significant cover in the primary successional sites, whereas the secondary successional sites already had well-established populations of jaumea and saltgrass. However, despite the documented ability of pickleweed to dominate seed establishment and suppress recruitment of other species into plots (Armitage *et al.* 2006, Bonin and Zedler 2006), saltgrass, jaumea, and to a lesser degree gumplant and arrowgrass recruited into the primary successional plots – primarily in the second year after restoration - and increased significantly over the next four years. Gumplant, arrowgrass, as well as pickleweed and saltgrass did not differ from the reference marshes. In the secondary succession sites, the contrasting lack of change, and the lack of increase of any of the less common species suggests the pre-restoration cover of salt-tolerant species may be preventing recruitment of less common species, even given changed tidal regimes that might otherwise favor them. The snapshot we have of the communities in the primary and secondary successional areas in 2013 is open to interpretation: they did not differ significantly from each other in any species, so one might conclude that order of establishment is not important to the salt marsh community at the site, and that differences with the reference marsh may be ascribed to environmental differences, or, one might look at the very different trajectories of the two areas, and the significant increase in two less common species (even if not yet significantly different the secondary successional sites), and conclude that the primary succession sites have not yet reached a stable state, and thus the final configuration may approach the reference sites, or may be different from either. Like the Project Area as a whole, primary and secondary succession areas had very low cover of two species common in the natural marshes around Tomales Bay: seaside plantain and sea lavender. Unless these species gain very significantly in relative cover as the marsh matures, the restored site will continue to differ significantly from reference marshes.

Restored sites often do not move towards the same “set endpoint” as exist in reference communities. Boyer and Thornton (2012) identify three factors that often separate the communities that assemble in restored communities from reference sites: order of arrival, connectivity to sources of less common seeds, and presence of invasive species. The Undiked Marsh and brackish remnants in the Project Area provide seed source, and invasives have not been a problem in the way they are in marshes with invasive *Phragmites*, *Spartina*, or *Lepidium* species already well established in the area pre-restoration. Where uncommon seed were readily available, salinity can drive order of establishment (Wolters et al, 2008). It may be that the two most underrepresented of the common species – seaside plantain and sea lavender, are more competitive in the higher salinity environment of the reference marshes, or it may be that they are sensitive to other differences. In another study, seaside plantain was present in a reference marsh in England, but failed to recruit into adjacent restored marshes after several years. This species may be sensitive to soil or other changes that can differentiate restored sites on former grazed lands. Differences in abiotic factors after human use were thought to drive failure of reclaimed marshes in England even after more than 100 years (Garbutt and Wolters 2008).

Beyond the identity and diversity of plant species at a site, plants play an important role in the food-web as primary producers, and as structure for nesting, perching, and hiding for animals. Salt marsh habitats generally have low-growing vegetation that does not grow as densely as in grasslands, and has relatively limited vegetation in the shrub strata (>0.75 m). Canopy complexity is important for many perching birds such as song sparrows, and to provide refugia for rallid species such as the endangered Ridgeway’s rail (formerly California clapper rail; Goals Project 2000). Despite the conversion to tidal marsh, the site continues to support a high degree of canopy complexity, primarily due to the diversity of habitats at the sites. Pre-restoration, grazing on the Project Area kept the aboveground biomass on the pastures at a level that was equal to that found at natural salt marshes. Reference marsh biomass ranged between ~1500-2100 g/m², similar to other marshes within the area, which ranged from 1500-2900 g/m² (Ryan 2009), 1300-2100 g/m² (Cuneo 1985) and 1200-1600 g/m² (Mahall and Park 1976) for marshes around San Francisco Bay. Current levels of biomass production are on par with natural marshes, and through the daily action of the tides, and flooding from Lagunitas Creek, the entirety of the Project Area has been hydrologically reconnected to a much larger area and to a much wider range of organisms, making it part of a much larger food web than when the dairy was diked off from the surrounding waters.

Conclusions

Five years since the restoration, the Giacomini Wetlands support diverse and dynamic plant communities. The dominant plant species within the Project Area changed from non-native grasses to native salt marsh plants, and non-native plant species have decreased significantly overall, and in every plant community except Ruderal. The overall α-diversity has decreased since restoration and in almost every habitat type, but the greatest drop was immediately after restoration, and, since then, it has increased significantly. Most of the loss in diversity is due to the loss of non-native species. Percent similarity metrics indicate the plant community of the site is becoming more similar to the reference sites (i.e. more salt-marsh dominated) with each passing year since restoration. Yet, while plant communities have shifted from grassland- to salt marsh- dominated, the restored habitat supports a greater diversity of plant communities, including grassland, and they are more evenly represented than the pre-restoration condition. This greater diversity of plant communities has the potential to provide habitat for a greater diversity of other animal species. As with biomass, average plant height class has not changed since restoration, and the site continues to provide the vertical habitat complexity important to some organisms. While the restoration site has much more vegetation in the shrub strata than the reference marshes, salt marsh communities within the restoration site are similar to the reference marsh communities. Taken together, these measurements suggest a site which provides a diversity of habitats for a diversity of organisms and- within only a few years post-restoration- is on par with reference site conditions in many measures of plant community function.

However, the goal of re-creating a marsh similar to that of the references marsh sites is not assured. The site has lower diversity and higher weediness than the reference marshes. Years of pasture management and the presence of pasture and salt marsh vegetation at the time of restoration have created very different conditions from the ones in which the reference marshes formed. Yet this may also reflect the limitations of the reference marshes themselves, which are smaller, saltier and less variable in elevation. By some metrics the Project Area more closely resembles the large, diverse remnant brackish marshes of San Francisco Bay. Only time will tell the final destination of the still actively evolving trajectory of the Giacomini Wetlands.

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8 Results and Discussion: Rare Plants

Introduction

Four species of plant found on the Project Area are considered to be “rare” plant species. These plants are Point Reyes bird’s beak, Humboldt Bay owl’s clover, Lyngbye’s sedge, and Marin knotweed. More information about each species can be found below.

Rare plants species are plant species of management concern, which encompasses species listed by the USFWS as “Threatened” or “Endangered” (i.e. federally listed), listed by the California Department of Fish and Game as “Threatened,” “Endangered,” or a “Species of Concern” (i.e. state listed), and those species listed by the California Native Plant Society (CNPS) as being of potential conservation concern (http://www.cnps.org/ cnps/rareplants/ranking.php). None of the species found on the Projected Area are federally or state listed species: rather they are all species that have been designated as rare by CNPS. Though these species do not have legal status, the National Park Service is directed to “proactively conserve listed species and prevent detrimental effects on these species,” and, “inventory, monitor, and manage state and locally listed species in a manner similar to its treatment of federally listed species to the greatest extent possible,”(NPS Management Policy 4.4.2.3).

CNPS ranks rare plant species using a list divided into five main categories, ranging from 1A (presumed extinct) to 4 (of limited distribution). Each category (except 1A), is further broken into three categories based on degree of threat to that plant. These categories are summarized in Table 7, below.

Table 7. CNPS Rare Plant Ranking Categories

Categories	Description
Rarity List	
1A	Plants Presumed Extinct in California
1B	Plants Rare, Threatened, or Endangered in California and Elsewhere
2	Plants Rare, Threatened, or Endangered in California, But More Common Elsewhere
3	Plants About Which We Need More Information - A Review List
4	Plants of Limited Distribution - A Watch List
Threat Ranks	
.1	Seriously threatened in California (high degree/immediacy of threat)
.2	Fairly threatened in California (moderate degree/immediacy of threat)
.3	Not very threatened in California (low degree/immediacy of threats or no current threats known)

Methods

To monitor rare plants, we map the boundary of each rare plant patch using a global positioning (GPS) unit, or if the patch is very small (<1 meter) a point is recorded. If the patch is small, we also count the number of plants within the patch. In larger plots, the number of plants is visually estimated. The relative success of the plant in colonizing the restoration area is judged based on the extend of the area colonized by the species in question.

Results

Point Reyes Birds Beak

Point Reyes Bird's Beak – *Cordylanthus maritimus* ssp. *palustris* (*Chloropyron maritimum* ssp. *palustre*), CNPS 1B.2

Point Reyes bird's beak is a rare herbaceous annual plant in the Orobanchaceae family that occurs in the high marsh habitat. It is a hemiparasite, meaning that while it does photosynthesize, it requires a host plant for additional nutrients. Point Reyes bird's beak was found in natural marshes directly north of the Project Area prior to restoration, but not within the Project Area, itself. Since restoration in 2008, this species has expanded to numerous locations within the Project Area. The maps below show the distribution of Point Reyes Bird's Beak in 2005, prior to restoration, and in 2010, 2011, and 2012, post restoration (Figure 41). In the five year period since restoration, this species increased from being absent within the Project Area in 2005 to occupying 1.6 acres of habitat across both former East and West Pastures in 2013 (Figure 42).

Figure 40. Distribution of Point Reyes bird's-beak prior to restoration in 2005, and in 2010-2012.

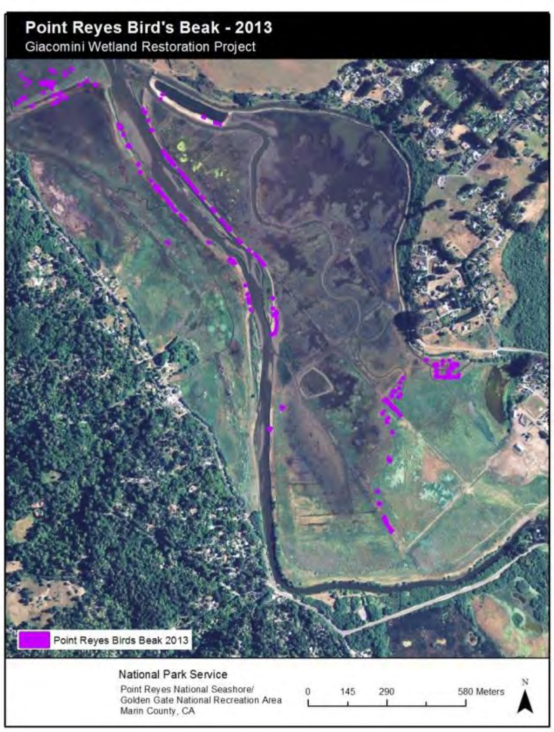

Figure 41. Point Reyes Birds Beak Distribution in 2013.

Humblodt Bay owl's clover

Humboldt Bay Owl's Clover – *Castilleja ambigua* ssp. *humboltiensis,* CNPS 1B.2

Humboldt Bay owl's clover, like Point Reyes bird's beak, is a rare herbaceous annual plant in the Orobanchaceae family that occurs in high marsh habitat. It is also a hemiparasite, as are all members of the Orobanchaceae. Though classified as Humboldt Bay owls clover, the individuals of this subspecies found in Tomales Bay look fairly different to those found in Humboldt Bay, having white- rather than pink-tipped bracts and lighter colored flowers. This has led some botanists to suggest that the Tomales Bay form should be given its own name. Prior to restoration, Humboldt Bay owl's clover was found in natural marshes directly north of the Project Area and outside the levees along Lagunitas Creek, but not within the ranch pastures. Since restoration in 2008, this species has expanded to several new locations within the Project Area. The maps below show the distribution of Humboldt Bay owl's clover in 2005, prior to restoration, and in 2010 and 2011, post restoration (Figure 43). It was not possible to map the entire distribution of this species in 2012 or 2013, but the southern portion of the east pasture, where this species is most abundant, was mapped in these years. Figure 44 shows this area in 2011, 2012, and 2013. The acreage Humboldt Bay owl's clover appeared to peak in 2012, when it occupied 3.7 acres of high marsh/upland ecotone habitat. In 2013 the owl's clover occupied on 1.9 acres of habitat, however it colonized a new area in the north east corner of the mapped area (Figure 44).

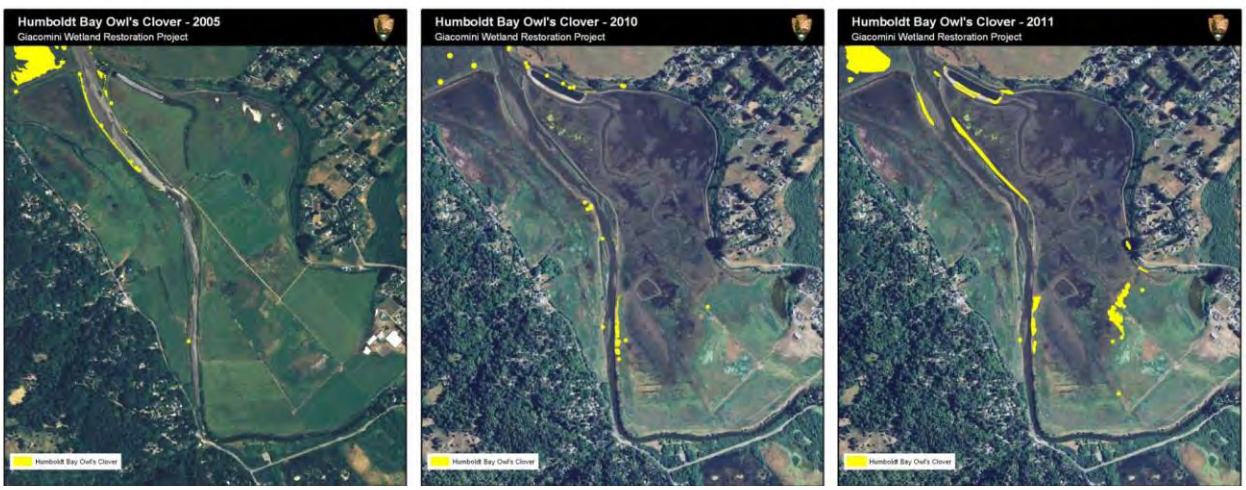

Figure 42. Distribution of Humboldt Bay owl's-clover prior to restoration in 2005, and in 2010-2011

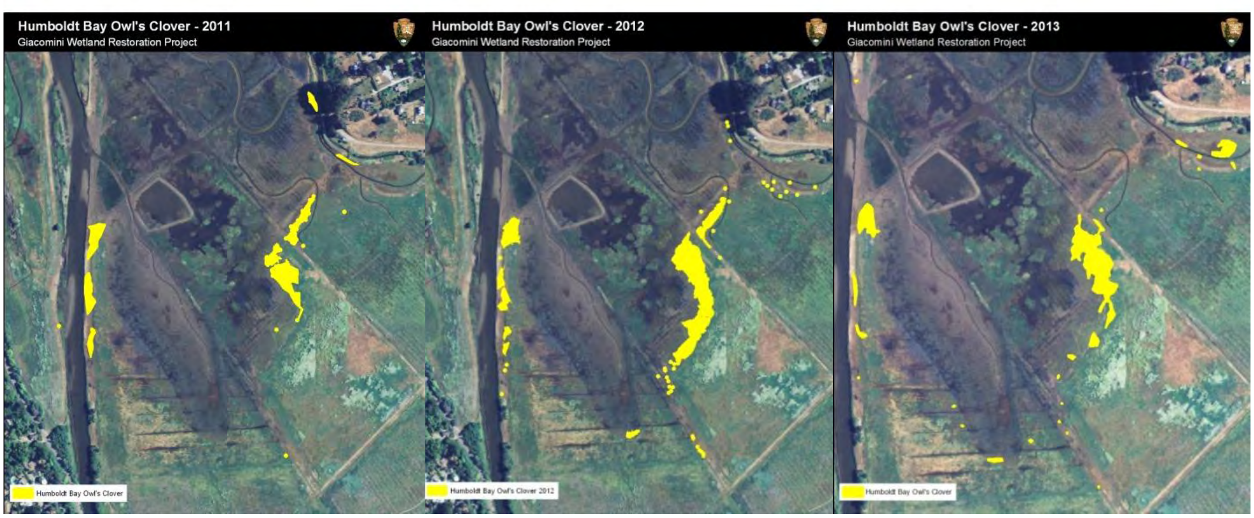

Figure 43. Distribution of Humboldt Bay Owl's Clover in the southern portion of the East Pasture, 2011- 2013.

Lyngbye's sedge

Lyngbye's sedge – *Carex lyngbyei*, CNPS 2.2
Lyngbye's sedge is an herbaceous, rhizomatous perennial sedge which grows in brackish (low salinity) marshes and creek banks. Lyngbye's sedge is found in two patches in the Project Area. Both patches were originally just outside the levees that separate the former dairy pastures from adjacent creeks and marshes. Though they were near the former levees, neither patch was disturbed when the levees were removed from the Project Area. The larger patch, 0.2 acres in size, has remained healthy and vigorous since restoration, but the smaller patch, only ever a few feet in diameter, was not relocated in 2013.

In early 2013, two new introductions of this species were planted along Tomasini Creek, one near the mouth and one near the entry point into the Project Area. As of fall 2013 the mouth population was surviving and the upstream site was observed with vigorous new growth in spring, 2014 (see picture, below). A third site was planted with this species in spring, 2014.

New growth on Lyngbye's sedge individuals in an introduced population

Marin Knotweed – *Polygonum marinense*, CNPS 3.2
Marin knotweed is a small herbaceous annual in the Polygonaceae family found in brackish and salt marches. Though listed as rare by CNPS, taxonomists now think this species is probably not a rare plant, but rather a non-native knotweed (*Polygonum robertii*) introduced from Europe. Even if non-native, this species is not considered invasive (i.e. it is not known to spread rapidly or outcompete native plants for habitat). A few individuals have been seen in the Project Area before and after restoration, but given its dubious status as a rare plant, this species has not been monitored regularly.

9 Results and Discussion: Changes in Avian Communities – Fall and Winter

Perhaps, the most visible evidence of the early success of restoration efforts at Giacomini can be seen during the fall and winter, when thousands of birds are scattered across the shimmering expanse of blue. On a still day, the air reverberates with the sounds of flapping wings, splashing water, and the low drone of chattering birds. The water that blankets the newly restored Giacomini Wetlands during the migratory season results in an explosion in waterbird numbers each fall and winter, and the numbers of these migratory converts continue to grow substantially each year. Numbers of waterbirds, particularly waterfowl, have grown steadily since restoration five years ago, although shorebird numbers have varied somewhat since the levees were breached.

Tidal wetlands in the greater San Francisco Bay area region provide especially valuable habitat for migrating and wintering waterbirds (Shuford *et al.* 1989; Goals Project 2000 *in* ARA 2010), and Tomales Bay is part of a network of Bay Area coastal estuaries that support waterbird populations of hemispheric importance (Kelly 2001). Fall is a period of avian movement and migration, the transition of species between nesting territories and wintering grounds (ARA 2009). Overall waterbird use tends to be low in the fall, but migratory flocks "stop-over" to forage for brief periods before moving on (ARA 2009). As one of the key coastal wetland stepping stones for migrating shorebirds along the Pacific coast, Tomales Bay provides a crucial stopover feeding habitat (Kelly and Condeso 2009). Individuals may stay only a few days to refuel before moving on, or they may stay longer (ibid). During fall migration, first year birds select wintering areas they will to return to in subsequent years (ibid). Little is known about this selection process, but most of them will probably make this choice by mid-November (ibid).

Ducks near Dairy Mesa.

Shorebird and Duck Use Prior to Restoration

Before restoration, bird use of Giacomini was seasonally sporadic and rather limited. Pastures were used for roosting by Canada geese, Great blue herons, Great egrets, and mallards and nesting by grassland species such as Song and Savannah sparrows (ARA *et al.* 2002). The limited number of water features left in the dairy such as ditches and managed creeks also supported low numbers of Mallards, Gadwall, Lesser scaup, and Eared grebes (ARA *et al.* 2002). Occasionally, diving ducks such as Buffleheads used some of the old Duck Ponds created by the Giacominis for hunting (ARA *et al.* 2002). During the winter and spring, bird use would climb dramatically, with sustained flooding of lower elevation areas in the northeastern corner of the East Pasture, known as the Shallow Shorebird area. Dabbling ducks such as Gadwall, American wigeon, and Green-winged teal stopped here, but the highest use probably came from shorebirds that would roost and forage here when tides were high in Tomales Bay (ARA *et al.* 2002). The most common shorebird visitors were Dunlin, Dowitcher, Greater yellowlegs, Common snipe, Willet, and Killdeer (J. Kelly, ACR, *pers. comm.*).

With restoration, waterfowl use of the Project Area was projected to remain similar to historic levels or increase slightly, while shorebird use was predicted to increase more dramatically (NPS 2007). As the marsh had subsided – or dropped in elevation very little since first being leveed in the 1940s – the former dairy ranch was expected to convert relatively quickly to a marsh very similar to the one directly north of Giacomini, near Inverness. This tidal marsh is predominantly marshplain with an intricate network of intertidal tidal creeks. This type of habitat would have great value for foraging shorebirds, but, without large ponds or open water areas, ducks would not find the restored marsh that attractive. Also, with conversion of pastures to marsh, numbers of grassland-associated species were expected to suffer the largest decline (NPS 2007).

Shorebird and Duck Use After Restoration

Less than two months after the levees were breached, enormous flocks of waterfowl descended on the waters of the newly restored Giacomini Ranch. In December 2008, more than 3,295 waterbirds were counted during one morning survey (ARA 2011). Approximately 5,552 individuals and 44 different waterbird species were observed in 2008–2009, of which more than 90 percent were waterfowl species (ibid). The density of birds during this season averaged 8.1 per hectare (ibid). During that first winter, some of the species with the highest numbers were American wigeon (47%), Northern pintail (25%), Green-winged teal (6%), and Northern shoveler (3%). Other duck species included Gadwall, Mallard, Cinnamon teal, Bufflehead, and Ruddy duck.

The number of waterbirds has continued to climb dramatically each year after restoration. The restored wetland attracted incredibly high number of waterbirds in Years Two, Three, and Four, with the total number of birds observed climbing from 34,084 in 2010-2011 to 58,892 in 2011-2012, a 73% increase (ARA 2012; Figure 45). Waterbird abundance has typically peaked in December, although, in Year 4, it actually peaked in January and, in Year 6 (2013/2014), it peaked in November, with a new high count of 16,031 individuals (ARA 2014). Species richness also continued to rise slightly from 64 in Year Two to 74 in Year Three and 75 in Year Four (ibid), dropping perhaps slightly in Year 5 to 72 species (ARA 2013). Average number of birds per survey climbed dramatically from 3,408.4 in 2010-2011 to 6,543.6 in 2011-2012, with one less survey actually conducted in Year Four (ibid). Early winter censuses in Year 6 reported 8,480.4 per census (ARA 2014).

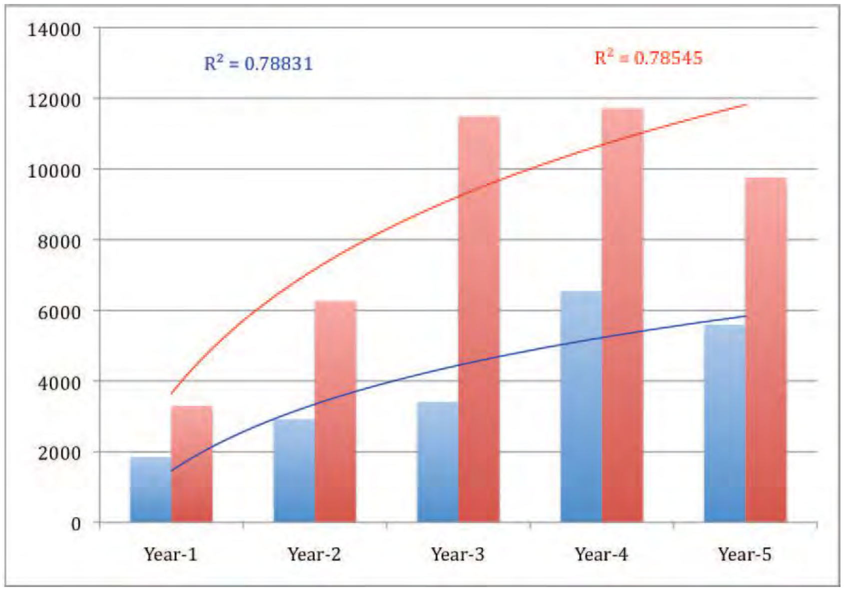

Figure 44. Post-Restoration Waterbird Abundance
Comparison of overall waterbird abundance for each monitoring year, post-restoration.Blue bars indicate mean abundance values, red bars indicate peak abundance values in a given year for all waterbirds counted within restored wetland. Source: ARA .

Waterfowl

In each year since restoration, waterbird composition has been dominated by waterfowl, although the percentages have varied somewhat (Figure 45). In Year 1 and 3, waterfowl accounted for a preponderance of the waterbirds, ranging from 93.3 in Year 1 to 89.8 in Year 3 (ARA 2012). In Years 2 and 4, those percentages dropped substantially to approximately 72% in both years (ibid). In Year 5, the percentage plummeted even lower: waterfowl comprised 58.8 percent, with waders comprising 40.5 percent (ARA 2013). Early winter censuses in Year 6 showed a higher percentage of waterfowl (67%) relative to waders, but this included surveys only through December 2013 (32.1%; ARA 2014). Surface-feeding waterfowl species dominate the wintering avian community (ARA 2012). Bufflehead has been the only diving duck species that has occurred in any notable abundance, increasing in Years Four and Five (ARA 2012, ARA 2013). Interestingly, the mean abundance of dabbling ducks at the newly restored wetlands has greatly exceeded any of the mean values reported by Kelly and Tappen in 1998 for all of Tomales Bay (ARA 2012).

American Wigeons and Northern Pintails congregating in the newly restored wetlands

Between Years 2 and 4, American wigeon represented the dominant species, accounting for at least 5% of the approximately 43,815 ducks counted in Year 4. In Year 5, Green-winged teal surpassed wigeon as the most abundant waterbird on site (ARA 2013). In Year 6, these two species -- American wigeon and Green-winged teal -- represented 50.2 percent of all waterbirds and 74.6 percent of the waterfowl group (ARA 2014). Ranked roughly in descending order, other dominant species included Northern pintail, and, depending on the year, either Gadwall or Northern shoveler (ARA 2012, ARA 2013).

While information on peak pre-restoration densities are not readily available, some conclusions can be made about the effect of restoration on mean densities of certain species by comparing post-restoration data with information collected by the U.S. Geological Survey pre-restoration monitoring effort (ARA 2011). Prior to restoration, dabblers such as American coot, American wigeon, and Northern pintail had the highest mean density per hectare of 6 per ha in the East Pasture (after American blackbirds; ARA 2011). Averaged for the entire Project Area, pre-restoration dabbler density dropped slightly to 4 birds/ha (ibid). Peak densities of American wigeon, Green-winged teal, and Gadwall have climbed substantially since the ranching period (R^2 between 0.38 to 0.90), although relative abundance has varied from year-to-year (ARA 2011). In Year 3 post-restoration, the same suite of six dabbling duck species – American wigeon, Gadwall, Green winged-teal, Northern pintail, Northern shoveler, and Mallard -- averaged 12.8 birds/hectare, an increase of 320% (ibid).

By Year 4, American wigeon had increased ten-fold from pre-restoration numbers, with Year 4 abundance being 60 percent higher than Year 3 (ARA 2012). Green-winged teal numbers increased three-fold from Year 3 to Year 4 (ibid). One interesting "trend" was displayed by Gadwall, a dabbling duck that increased fairly dramatically the first few winters following restoration: in Year 5, Gadwall nearly doubled over Year 4 (1.86x), but showed a decline (-34.5%) compared to the early winter peak in Year 3 (ARA 2013). Mallard was the only species that actually showed a decrease in numbers relative to pre-restoration abundance (ARA 2011). Interestingly, while these species have remained the dominant ones, the dominance of these five species, however, has decreased steadily over time, dropping from 82.3% of the total in Year 1 to 58.2% of the total in Year 4 (ARA 2012).

A flock of buffleheads in restored wetlands in winter 2009-2010

American coots are not typically categorized as a waterfowl or shorebird, but are considered a unique species with a variety of foraging habitats. During Year 4, exceptional numbers of coots visited the restored wetlands, averaging 645.6 coots per census with a peak count of 2,000 on December 16, 2011 (ARA 2012). The high numbers mirrored a region-wide population "bump" in the winter of 2011-2012 (ARA 2012).

Shorebirds

The potential value of the restored wetland for shorebirds, as well as waterfowl, must have seemed promising when a flock of Dunlins flew into the wetlands on the day the final levee was

Northern pintails on Giacomini Wetland waters.

breached. In early November 2008 surveys, shortly after breaching of levees, monitors from ACR observed some Short-billed and Long-billed dowitchers and scattered groups of Western and Least sandpipers in the newly restored marsh, Kelly said. Evens noted at least 1,000 Least and Western sandpipers around the edges of the flooded marsh in early November 2008 (J. Evens, ARA, *pers. comm.*). However, despite these early promising signs, numbers of shorebirds during the first winter was very low (ARA 2011, Kelly and Condeso 2011), accounting for less than 1% of all waterbirds observed by ARA during its surveys. Most of these species observed were Greater yellowlegs, Wilson's snipe, Killdeer, and Least sandpiper (Kelly and Condeso, unpub. data).

Although shorebird use remained low during the first year after restoration relative to established feeding areas in Tomales Bay, shorebird abundance within the restoration area did seem to increase a bit in the late winter of the first year, though species composition remained similar (E. Condeso, ACR, *pers. comm*.). Later, during the peak migratory season for shorebirds in spring, shorebird numbers increased slightly yet again (ARA 2011, Kelly and Condeso 2011), although dabbling ducks still accounted for 84% of the waterbirds observed (ARA 2011). Though the distribution of shorebirds varied among counts, most of the shorebirds observed occurred in specific areas, including the East Giacomini Marsh (Shallow Shorebird Area in northern portion of East Pasture), the goby pond in the northern portion of the East Pasture, and the Triangle Marsh Area (Tomasini Triangle Marsh in central portion of East Pasture; Kelly and Condeso, unpub. data). These ponded areas of the newly restored Giacomini Wetland provide habitat that seems suitable for a number of species (E. Condeso, ACR, *pers. comm*.).

Wader abundance climbed dramatically in Year 2, with the ACR August 2009 count totaling nearly three times as many birds as the highest winter 2008–2009 estimate (Kelly and Condeso 2009). This increase in numbers during the August survey was mostly due to a spike in the number of Least sandpipers observed (ibid). In addition to the species seen during the winter of 2008–2009, Western sandpipers, Dunlin, and Willets were also observed in spring and fall by ACR (ibid). Dunlins, which are the most abundant shorebird in Tomales Bay, did not visit the restored wetlands until Year 2 (Kelly and Condeso 2011). Dowitchers were recorded in respectable numbers—66 for Short-billed and 5 for Long-billed—for the first time in fall 2009 (ibid). Some of the increases in abundance, density, and species richness in Year 2 may have been due to the greater number of surveys conducted in Year 2, as official surveys doubled from three to six, however, the average number of birds per survey also climbed from 1,850.7 in 2008-2009 to 2,917.8 in 2009-2010, suggesting that increases were not just an aberration from the change in survey approach.

Wader abundance continued to increase during the winter of Year 2 (ARA 2010). Abundance of shorebirds increased 25-fold from 87 individuals in December and January of Year 1 to 2,221 individuals in December and January of Year 2 (ARA 2011). On the first two surveys, shorebirds accounted for 26- to 29 percent of all waterbirds detected, a substantial increase from 2008–2009, when shorebirds represented less than 1 percent of birds observed (ibid). In addition to the expected increase in small calidrine sandpipers (Least sandpipers and Dunlin), high numbers of Dowitchers and Yellowlegs were present (ibid). Some shorebirds occurred in very high numbers during December 2010: Greater yellowlegs peaked at 79 individuals, and Dowitchers (probably Long-billed) climbed as high as 580 birds (ibid). Based on numbers from Kelly (2001), these numbers appeared exceptional for Tomales Bay. In addition, the 500 Red-necked phalaropes that flocked in the northern portion of the East Pasture in “Goby pond” also represented a new high number for this species in this region based on data from Shuford *et al.* (1989) and Kelly (2001; ARA 2011).

As with waterfowl, wader numbers started dropping in January of Year 2 and continued to decrease through April, with very few shorebirds present in spring 2010, although they represented an overwhelming majority (86.5%) of the birds present (ibid). A mid-winter decrease in waterbirds associated with cumulative seasonal rainfall is common in Tomales Bay (Kelly and Tappen 1998, Kelly 2001) and other coastal estuaries (Colwell 1983, Shuford *et al.* 1989, Warnock *et al.* 1995 *in* ARA 2010, 2011). This seasonal decline may be associated with a shift to other now-flooded habitat such as pastures or shallow inland basins or with a seasonal shift in foraging patterns and prey availability (ARA 2010). In post-restoration monitoring, waterbird use of Giacomini Wetlands clearly declined by as much as approximately 63- to 73 percent in Years 2 and 3, respectively, starting in late winter or January, typically when the heaviest rainfalls begin (ARA 2011). Interestingly, Black-bellied plover numbers remained remarkably equivalent throughout the early winter surveys, suggesting that this species may have resided at Giacomini during that winter (ibid).

With such promising results in Year 2, great things were expected in Year 3 relative to continued increases in shorebird abundance. By 2009-2010, waders appeared to have “found” the newly restored wetlands. At first, all appeared to be on track for another record-breaking shorebird year, as numbers were high through through fall and early winter of Year 3 (Kelly and Condeso 2011). However, in mid- and late- winter and spring, wader numbers plummeted relative to the previous year (ARA 2011, Kelly and Condeso 2011; Figure 46). For all ARA surveys combined in Year 3, waders represented only 5.7% of the waterbird community, compared to an average of 27.1% during Year 2 (ARA 2011). In addition, waders averaged only 327.3 individuals/survey in Year 3, compared to 780.7 individuals/survey in Year 2 (ARA 2011).

On at least one count date in Year 3 (11/12/10), shorebirds did account for 20.5% of all waterbirds observed (ARA 2011). This peak number illustrated one of the confounding variables of any biological or environmental monitoring efforts: surveys or sampling may not always catch peak abundance or concentrations. Most of ARA’s survey effort was concentrated in mid- to late winter and spring, so, therefore, averages were skewed by the scarcity of waders during that period. However, this phenomenon was not relegated strictly to Giacomini. Reduced abundances of shorebirds occurred throughout Tomales Bay in late winter and spring of 2010-2011 (Kelly and Condeso 2011). Some of this may be due again to increased precipitation during those periods, as discussed earlier: The decline in numbers corresponds closely to the onset of rainy season (Figure 46). Also, use of coastal estuaries is typically lower in wet years than dry years (ARA 2011).

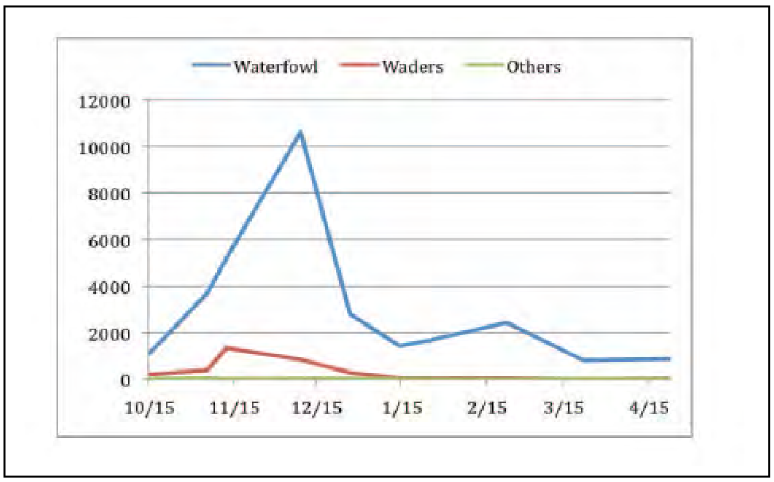

Figure 45. Waterfowl and wader numbers in Year 3
Strong declines occurred in January 2011. Source: ARA.

Despite reduced shorebird numbers overall in Year 3, some species continued to persist at the newly restored wetlands, including Greater yellowlegs, which had a peak abundance in Year 3 of 82 individuals, roughly equivalent to Year 2 (79 individuals; ARA 2011), although ACR surveys suggested a stronger decrease relative to Year 2 (Kelly and Condeso 2011). Greater yellowlegs have always visited southern Tomales Bay in high numbers, leading some birders to consider it “magic” for this long-legged species (Kelly and Condeso 2011).

Western sandpiper occurred in incredibly high numbers during fall 2010 (mean of >800): fall migration of juveniles often determine wintering areas for future years (Kelly and Condeso 2011). Least sandpipers also returned in early winter of Year 3 in roughly equivalent numbers to the prior year (Kelly and Condeso 2011). Wilson’s snipe was one of the few species that remained relatively common during both early winter and late winter surveys in Year 3 (Kelly and Condeso 2011). This species has always been attracted to the seasonally inundated or saturated grassy margins of the Project Area (Kelly and Condeso 2011). Conversely, numbers of species such as Dunlin (the most common shorebird in Tomales Bay), Black-bellied plovers, and Dowitchers fell dramatically during Year 3.

Ironically, in Year 4, this decline in shorebird abundance reversed again. Numbers of waders in Year 4 more closely resembled those of Year 2, with numbers much higher than Year 3 (ARA 2012). Shorebirds accounted or 26.8% of all waterbirds in Year 4, compared to only 9.6% in Year 3; 23.2% in Year 2; and

3.7% in Year 1 (ibid). Least sandpiper, a small shorebird that feeds primarily on the marsh surface, represented about two-thirds or 63% of all waders similar to previous years (ibid). However, Year 4 was marked by an increase in use by long-legged waders, particularly Marbled godwits and Dowitchers (ibid). Although Dowitchers showed a significant presence also in Year 2, Marbled godwits did not arrive at the restored wetland in any large numbers until Year 4, with a high count of 241 on January 26, 2012. A migrant flock of American avocet also dropped in during the fall of Year 4: this species had never sighted here before and is relatively rare on Tomales Bay (ibid).

Both Marbled godwits and Dowitchers were observed roosting and actively foraging in the Giacomini Wetlands. This is notable, because both species are long-billed and forage by probing the substrate. One of the reasons that wader colonization of the restored wetland may have lagged behind that of waterfowl is that many shorebird species might require development of an adequate benthic invertebrate prey base. With these areas formerly being pastures, they obviously did not support many benthic invertebrates, and even the diked channels had only few and relatively monotypic benthic invertebrate communities (Parsons *et al.* , *in prep*.). Studies from southern California showed that polychaetes represented one of the most important fall food items for Marbled godwits, followed by bivalves in the winter, and crabs and bivalves in the spring (Recher 1966, Stenzel *et al.* 1983, Ramer *et al.* 1991 *in* Gratto-Trevor 2000). Dowitchers in the San Francisco Bay area commonly foraged on sedentary polychaetes (*Capitella*), amphipods (*Corophium*), and bivalves (*Gemma* sp., *Macoma* sp., etc.; Stenzel *et al.* 1983; Takekawa and Warnock 2000).

All of these taxa have been documented within the restored wetland (Parsons *et al.* , *in prep*.). After restoration, oligochaetes remained the most common invertebrate group in Reference Areas (0.40 inverts/cm^3) and a common one in the Project Area (0.21 inverts/ cm^3), as well. Amphipods were seemingly the most prevalent invertebrate group in the restored wetland (0.39 inverts/ cm^3; ibid). The three most common lower-order taxonomic groups in the restored Project Area were *Paracorophium* (Amphipoda; 34% of all species), *M. insidiosum* (Amphipoda; 11% of species), and *S. benedicti* (Polychaeta; 7% of all species; ibid), Following restoration, the three most common taxa in the Project Area also ranked in the top four most common taxa of Reference Areas. In Reference Areas, the most common taxonomic groups were *Paracorophium* (Amphipoda; 15% of species), *S. benedicti* (Polychaeta; 14% of all species), *Pygospio elegans* (Polychaeta; 5% of all species), and *M. insidiosum* (Ampipoda, 3% of all species).

Polychaetes -- which, in general, have been associated with newly restored systems -- comprised very little of the invertebrate assemblage at Giacomini and the adjacent Undiked Marsh either pre- or post-restoration, with oligochaetes being more predominant both pre-and post-restoration at Giacomini and the Undiked Marsh (Parsons *et al.* , *in prep*.). Polychaetes accounted for more of the benthic invertebrate community at Limantour Marsh and Walker Creek Marsh both pre- and post-restoration, although, in general, nematodes were numerically dominant (NPS, unpub. data).

Year 5 was the first year since restoration in which waders represented four of the most common species (Dunlin, Least sandpiper, Dowitchers, Marbled godwit; ARA 2013). It was also the first period during which waders outnumbered waterfowl on a given survey, with waders comprising 64.9 percent of the 9,759 waterbirds detected on November 16, 2012 (ibid). High numbers of Dunlin helped to inflate these numbers (ibid). In fact, waders dominated the avian community in fall and early winter (through November), reflecting the earlier timing of their migration patterns of the group as a whole (ibid). As discussed earlier, wader numbers have been variable, but, as a percentage of total waterbird use, Year 5 supported the highest number and the highest percent composition of waders to date (ibid). The waders showing rather dramatic increases in early Year 5 were the substrate-probing shorebirds discussed in depth above as more recent users of the restored wetland -- Marbled godwit, Dunlin, and Dowitchers.

The peak count of Marbled godwit was 1.5 times as high as the previous high, also in Year 4 (ibid). Peak numbers of Least sandpipers -- the most abundant wader overall in Year 5 with 1,810 individuals -- was essentially equivalent to the early winter high in Year 4 (1,816 individuals; ibid). Additionally, shorebirds seem to be more widely and evenly distributed across the East Marsh than in previous years when flocks tended to cluster in “islands” of habitat (P. Pyle, *pers. comm*. *in* ARA 2013). Most substrate probing shorebirds (e.g. Marbled godwit) vacated the restored wetland prior to the January 15 census, but that the surface feeding Least sandpipers persisted, although in diminished numbers, into February 2013 (ibid).

In the early winter of Year 6, approximately one-third of waterbirds counted (32.1%) were waders (ARA 2014). Three small calidrine sandpipers (Dunlin, Least and Western sandpipers) comprised 25.8 percent of all waterbirds and 80.3 percent of all waders (shorebirds; ibid).

Great egrets and a Heron forage at high tides at the newly restored wetlands

Other Species

Different types of species also started visiting the newly restored wetland. Of the 41 waterbird species observed by ARA in fall 2009, 20 species or approximately 38 percent of species were only observed during one census, with approximately 11 of the 20 species or 21 percent of the total represented by observation of only one individual (ARA 2009). Some of these individual sightings represented firsts for the Giacomini Wetlands since breaching of the levees in 2008, including American golden-plover, Solitary sandpiper, and Pectoral sandpiper (ARA 2009).

Other species of note during the latter half of Year 1 and Year 2 included Merlin, Long-billed curlew, Green heron, as well as non-waterbird species such as Bald eagles and Peregrine falcon (*Falco peregrinus pealii*) of the northern race (Kelly and Condeso 2009). Adults and immature Bald eagles, which were once extremely rare in west Marin, started visiting the site in spring 2009 and have remained occasional visitors to the wetlands, with at least four different individuals observed (ARA 2009, ARA 2014).

Raptors (juvenile Northern Harrier shown) continue to visit the newly restored Giacomini Wetlands in large numbers.

Bald eagles on downed tree along Lagunitas Creek

Some of the new noteworthy additions to the restored wetlands during Year 3 included Redhead, Eared grebe, Clark's grebe, Lesser yellowlegs, and Heerman's gull (ibid). Two species also "flew over" the site, but did not drop in – Surf scoter and Wandering tattler (ibid). A rare Yellow rail was also apparently detected in the newly restored West Pasture near Inverness Park during the Christmas bird count (K. Hansen, T. Easterla *in* ARA 2011). Prior to restoration, this species had been observed in the Undiked Marsh north of Giacomini in previous years, but never within the former dairy site (ibid). By Year 5, yellow rails had been sighted within the restored wetland for three consecutive years (ARA 2013). In Year 5, there were several new additions to the species list, including California brown pelican, Common loon, and, nearby, Broad-winged hawk (ARA 2013). A Yellow rail was also detected in the newly restored wetland for the third consecutive year, and two federally endangered California clapper rails stopped over at the natural marsh just north of the restored wetland in November 2012 (Erik Grijalva, Invasive Spartina Project, *pers. comm. in* ARA 2013). This observation is noteworthy, because this is only the sixth record for Tomales Bay over the last decade (ARA files). This taxon is a "target" species for the tidal marsh restoration, and the presence of two birds suggests the possibility of nesting during the 2013 nesting period (January thru

August) or in the future (ARA 2013). During the early part of Year 6, California black rail were also detected in the restored wetland, as they have been now for several years (ARA 2014).

Conclusions

As with all biological species, increases and decreases from year-to-year cannot be evaluated without considering all the other potential factors that could affect these animals, such as climatic variation, impacts to non-wintering and other habitat, disease, and natural population dynamics or cycles. Bird use of the newly restored wetlands is dependent on a number of factors, including migratory birds finding the site and the site providing what the birds need to eat. The taxa observed so far have a wide range of dietary needs, from species that largely consume seeds and other plant matter to those such as Greater yellowlegs who eat fish and other aquatic prey. Most of the waterfowl have been dabbling species, with bufflehead being the only diving duck occurring in any numbers (ARA 2012).

In terms of shorebirds, the species that showed up during the early years were species that forage on the surface (e,g., Black-bellied plovers), shallow probers (e.g., Least sandpipers), and those that forage in the shallow water column (e.g., Greater yellowlegs; J. Evens, ARA, *pers. comm*.). However, by Years 4 and 5, two long-billed species that probe more deeply in the mud became more abundant: Marbled godwit and Dowitchers (ARA 2012), Analysis of benthic invertebrate data suggests that numbers of benthic invertebrates within the restored wetlands are increasing and becoming more diverse, and this is perhaps borne out by higher abundance in recent years of deeper substrate probers (ARA 2012). We are continuing to analyze changes in the zooplankton, benthic invertebrate, and fish communities in hopes of understanding how the prey base for many target organisms is changing with restoration.

To evaluate how much of an effect the restoration itself might be having on higher bird numbers in southern Tomales Bay, Kelly and Condeso (2011) conducted an analysis to adjust shorebird abundances to take into account inter-annual variation observed for these species in other areas of Tomales Bay. What they found was that overall shorebird numbers in Tomales Bay closely match what might have been predicted even without restoration (Kelly and Condeso 2011). However, abundances of particular shorebird species did appear to be higher than would be predicted without restoration, including numbers of Least sandpiper, Western sandpiper, Dowitchers, Black-bellied plovers, and Greater yellowlegs (Kelly and Condeso 2011).

In general, then, despite somewhat uneven trends for shorebirds, the restoration is supporting higher numbers of waterbirds after restoration (ARA 2012, ARA 2013, ARA 2014, Kelly and Condeso 2011). The future for waterbird use ultimately will depend on both extrinsic factors and intrinsic ones, including the trajectory taken by the marsh in its continued evolution, both from a physical and biological standpoint.

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10 Results and Discussion: Changes in Avian Communities – Breeding Birds

Spring is the time for breeding, and many neotropical migrants or birds that may migrate from as far as South and Central America flock to this area to establish nests. During spring, Marin hosts as many as 150 species of breeding birds, and most of these occur on the coast (ARA *et al.* 2002). Tomales Bay cannot compete with San Francisco Bay in terms of sheer numbers, but Tomales Bay supports higher densities of many species (J. Kelly, ACR, *pers. comm.)* and accounts for a large proportion of many species' statewide population numbers. Tomales Bay is one of only three sites along the Pacific Flyway that support more than 100 Red knots (*Calidris canutus*) during spring migration (C. Hickey, PRBO, *pers. comm.).* Ten of the 17 Partners-in-Flight Riparian Focal Species breed in the Point Reyes region (C. Hickey, PRBO, *pers. comm.).* A large factor in the avian diversity found in Point Reyes comes from rare or extremely rare species, which account for nearly one-third of species observed (Evens 2008).

Even prior to restoration, the Giacomini Ranch and surrounding area supported impressive breeding bird activity. The large freshwater marsh and adjacent riparian habitat in the West Pasture adjacent to Sir Francis Drake Boulevard was one of the breeding hotspots, supporting Saltmarsh common yellowthroat (*Geothlypis trichas* var. *sinuosa*, former FSC, CSC), Virginia rail (*Rallus limicola*), Song sparrow (*Melospiza melodia*), Marsh wren (*Cistothorus palustris*), Brewer's Blackbird (*Euphagus cyanocephalus*), and other marsh-riparian associates (ARA *et al.* 2002). Two other areas that also attracted high number of breeding birds were the riparian-grassland habitats of Green Bridge County Park and the riparian-mesic coastal scrub habitats found on some portions of the Point Reyes Mesa bluff (ARA *et al.* 2002). Yellow warbler (*Dendroica petechia brewsteri*, CSC) has been observed breeding in the Green Bridge County park area and near Inverness Park and is a common fall migrant through riparian corridor (ARA *et al.* 2002). During baseline surveys, the Point Reyes Mesa bluff attracted a diverse array of the breeding birds observed in the area during spring, including species such as Swainson's thrush (*Catharus ustulatus oedicus*, CSC), Bewick's wren (*Thryomanes bewickii*, CSC), Wilson's warbler (*Wilsonia pusilla*), Warbling vireo (*Vireo gilvus*), and Allen's hummingbird (*Selasphorous sasin*, CSC; ARA *et al.* 2002). Saltmarsh common yellowthroat occurred in riparian habitat along the former Tomasini Creek channel (ARA *et al.* 2002).

Olema Marsh was once considered "one of the most diverse habitats for breeding, wintering, and migrating birds in the Point Reyes area" (Evens 2008). Between 1991 and 1994, total number of species ranged from 77 to 81 during the winter and from 44 to 49 species during the spring (Evens and Stallcup 1991, 1992, 1993, 1994). In 2004, this species richness trend reversed, with the number of species totaling 74 in spring and 65 in winter (Stallcup and Kelly 2004). Red-winged blackbird (*Agelaius phoeniceus*), Marsh wren, and Song sparrow represented the most abundant species in autumn, winter, and spring, along with Saltmarsh common yellowthroat, which had at least 12 nesting territories during spring 2004 (Stallcup and Kelly 2004, 2005). California black rails (*Laterallus jamaicensis coturniculus*, ST) occasionally occurred in Olema Marsh, but Virginia rails were more common, along with Sora (*Porzana carolina*) sometimes in fall (Stallcup and Kelly 2004, 2005).

California black rail

It is no coincidence that the areas showing the highest breeding bird activity prior to restoration occurred on the perimeter of the Giacomini Ranch, where the juxtaposition between managed pasture and groundwater inflow from the Inverness Ridge and Point Reyes Mesa created a diverse mosaic of habitats. In contrast, the large expanse of monotypic habitats represented by 550 acres of managed pastures was "relatively depauperate in terms of supporting breeding birds in general and special status species in particular" (ARA *et al.* 2002). Savannah sparrows (*Passerculus sandwichensis*) were observed attempting to nest in some of the managed pastures, however, as had occurred for decades, many fields were mowed during the height of nesting, thus excluding perhaps a third of their population (ARA *et al.* 2002). During baseline surveys, the only avian species using managed pastures as its primary habitat was Grasshopper sparrow (*Ammodramus savannarum*; former FSC, S2), but it arrived late and did not breed (ARA *et al.* 2002). Large flocks of migratory swallows sometimes foraged low over pastures and marshes, especially early and late in the day (ARA *et al.* 2002).

Juvenile Northern harrier

In 2008, then, breeding in the then dormant pastures reached a new zenith with discontinuation of agricultural management. (Mowing for construction purposes was only conducted after breeding had stopped, and chicks had fledged). Red-winged blackbirds, Song sparrows, and Savannah sparrows took advantage of the lush green fields, redoubling their nesting efforts and producing two broods in a single season, Evens noted. A pair of Northern harriers or Marsh hawks (*Circus cyaneus*) also nested in some of the fallow fields. White-tailed kites hunted the pastures by day, while Barn owls (*Tyto alba*) hunted them by night, feasting on the booming vole population, which also increased in response to the decrease in mowing and an increase in seed production by grasses. With the decrease in disturbance that had been caused by mowing and cattle, numbers of Wilson's snipe (*Gallinago delicata*) increased, with birds foraging and roosting in the moist pasture in fall, winter and spring. (Snipe do not nest locally.)

In the four years of monitoring that have been conducted since restoration, breeding bird activity has increased relative to both ranching days and the period immediately before construction. Figure 47 shows density changes between pre-restoration (2005-2007) and post-restoration (2009-2012): Density of key wetland-dependent passerines has increased universally after restoration (ARA 2011a). Ninety-three (93) avian species have been found during point counts in spring surveys (ARA 2012). Slightly less than two-thirds of these species (65.3%) are considered local breeders – that is, they either nest on-site (e.g., Mallard, Common yellowthroat, Song sparrow), or they forage on-site and nest nearby (e.g., Great blue-heron, Red-tailed hawk; 2012).

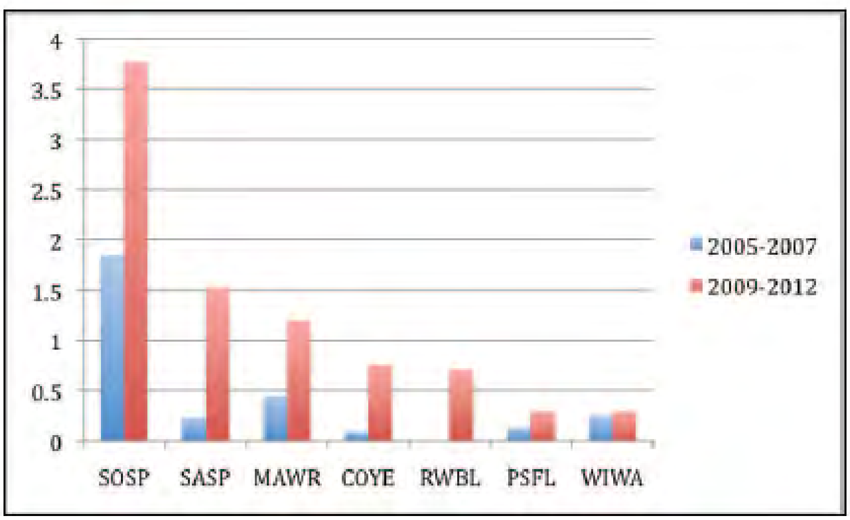

Figure 46. Average Breeding Bird Density, Pre- & Post-Restoration
Relative changes in average densities (D=territories/ha) between pre-restoration (2005-2007) and post-restoration (2009-2012) for the six most common wetland-dependent passerines: SOSP (Song sparrow), SASP (Savannah sparrow), MAWR (Marsh wren), COYE (Common yellowthroat), RWBL (Red-winged blackbird), PSFL (Pacific-slope flycatcher), and WIWA (Wilson's Warber). Source: ARA (2012).

Overall, species richness and the total number of observations was actually highest in Year 2 or (2010) than in Year 1, 3, or 4 (ARA 2012). Approximately 501 birds were sighted in 2009, compared to 940 in 2010, 729 in 2011, and 890 in 2012 (all detections; ibid). The newly restored wetlands supported 65 breeding bird species in 2009, 74 in 2010, and 48 in 2011 (all detections; ARA 2011a). The seeming decline in both numbers and species in 2011 may have been an artifact of field effort timing and weather (ibid). Surveys were conducted in late April and mid-May during Years 1 and 2 and mid-May to early June in Year 3 (ibid). Compounding the potential effect of the difference in timing was the unusual amount of late rains that occurred in Year 3, which may have suppressed or disrupted breeding activity (ibid).

While Year 3 had lower numbers and species richness than preceding years, numbers of specific wetland-dependent focal species continued to increase in Year 3 (ARA 2011a) and again in Year 4 (ARA 2012). These species include Song sparrow, Savannah sparrow, Marsh wren, Common yellowthroat, Red-winged blackbird, Pacific-slope flycatcher, and Wilson's warbler (ARA 2012). Wilson's warbler, which showed a decrease in territories per point in 2011, increased in abundance again in 2012, more than doubling in number since the pre-restoration period (ARA 2012).

In Year 4, two species with relatively strict habitat requirements – "Salt-marsh" common yellowthroat and "Bryant's" savannah sparrow – showed some of the highest increases since restoration. Density of common yellowthroat climbed from 0.09 territories per point between 2005 and 2007 to 1.10 territories per point in 2012 and 0.73 territories per point between 2009 and 2012 (ARA 2012). Density of Savannah sparrow territories also jumped sharply after restoration from 0.23 territories per point before levee breaching to 1.44 territories per point in 2012, a slight drop from the overall average of 1.66 territories per point for all years after restoration (ibid). Both of these subspecies are considered "California Bird Species of Special Concern" by the California Department of Fish and Game (Shuford and Gardali 2008 *in* ARA 2011a).

In general, Song sparrows were by far the most common breeding bird species during the spring in the newly restored wetlands, with densities reaching an average of 3.71 territories per count between 2009 and 2011 (ARA 2012). Savannah sparrows were the second most common (1.66 terr/pt), followed by Marsh wren (1.13 terr/pt), Red-winged blackbird (0.92 terr/pt), and Common yellowthroat (0.73 terr./pt; ibid). Salt marsh common yellowthroat and other wetland-dependent species have taken advantage of the newly establishing riparian thickets that were suppressed in years past by grazing. Species that had nested only along the edges of the wetland expanded their territories into the lowlands. With this increase in available habitat, larger numbers of songbirds are nesting and have improved reproductive success. The vegetation along lower Fish Hatchery Creek has developed into a lush fresh to brackish transitional wetland lined with marsh monkey flower, watercress, reeds, and rushes growing waist high. In Year 1, Marsh wrens, Song sparrows, and Virginia rails were particularly abundant here, and a rare California black rail was calling on territory here in late April.

Bald eagle

Perhaps, some of the most compelling visitors to the newly restored coastal wetland since restoration have been the Bald eagle (*Haliaeetus leucocephalus*) and the California black rails. Bald eagles showed up only after restoration was completed. They are not very common in the Tomales Bay watershed, although sightings have increased in recent years. The adult Bald eagles tend to show up when the bulk of the ducks arrive -- late-October into January (J. Evens, ARA, *pers. comm*.). Immatures (1-3 yr olds) pass over irregularly in the fall, but they are not common (ibid). Bald eagles were sighted again in Year 6, but during the winter (ARA 2014).

California black rail is a state-threatened species that has all but disappeared from many of its historic habitats and has been barely clinging on to its formerly extensive habitats in Tomales Bay. Prior to restoration, almost all of the rail sightings in Tomales Bay were in the Undiked Marsh directly north of the Giacomini Ranch. In Year 1, three California Black Rails were heard in the newly created marsh in April and May, however, they were not calling on territories: focused surveys were not conducted, so estimated numbers were not able to be determined (ARA 2009). In Year 3, focused rail surveys were conducted, and rails were detected calling on territories at three of the 17 census points, with all of the detections in the northern portion of the West Pasture adjacent to the Freshwater Marsh (ARA 2011b). Although focused surveys were not conducted in Year 4, rails were still estimated to have established a minimum of three territories in the West Pasture during that year (ARA 2012), and they have continued to frequent this portion of the marsh since then (J. Evens, ARA, *pers. comm*.). Rails also still occur in the Undiked Marsh (ARA 2011b). The amount of potential rail habitat in the restored Giacomini Wetlands appears greater than is currently utilized, so rail numbers could increase in the future (ibid).

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11 Conclusions

In general, the Giacomini Wetland Restoration Project has greatly exceeded the Seashore's expectations in terms of the speed and extent to which natural habitat and wetland conditions have evolved. Native salt marsh plants are rapidly colonizing the former non-native-dominated pastures, and the diversity of common, uncommon, and rare plant species present becomes more similar every year to adjacent natural marshes. Water quality improved greatly immediately after – and, in some cases, even before – restoration was implemented, with remaining issues almost exclusively emanating from non-point source run-off near Point Reyes Station. Less than a month after levees were removed, thousands of waterfowl and shorebirds flocked to the newly flooded marshplains and mud/detrital flats, and the numbers of these species, as well as spring migratory breeding birds, has increased each and every year since then. Shorebirds have shown a slower, but relatively steady, increase in numbers, perhaps due to the fact that the density of benthic invertebrates -- which some of the deeper probing species feed upon -- within the restored wetlands has jumped since restoration.

Photograph of restored marsh by Louis Jaffe

Assessing Restoration Success

In keeping with the BACI framework, the monitoring program stipulated that project success be evaluated by assessing a suite of variables and indicators representative of key wetland function and conditions in the Project Area both before and after restoration ("BA") and then comparing the restored wetland to natural or reference marshes ("CI"). Due to funding and staffing constraints, not all of the variables and indicators originally proposed to assess progress towards Project Purpose, Goals, and Objectives were sampled or assessed. The majority of variables and indicators not evaluated were GIS-based ones adopted from the HGM method developed by the Corps. However, the monitoring program did assess some aspect of the following processes, functions, and conditions – Freshwater – Surface Flows (Freshwater), Tidal – Surface Flows (Tidal), Tidal Surge/Flood Attenuation-Dissipation (Flood Attenuation), Sediment Deposition and Nutrient/ Contaminant Retention (Water Quality), Characteristic Plant Community (Vegetation), Food Chain Support-Invertebrates (Invertebrates), Wildlife Habitat and Use-Nekton (Nekton), and Wildlife Habitat and Use-Avifauna and Amphibians (Wildlife). Carbon Export and Freshwater – Groundwater/Seeps were the two processes and functions not evaluated.

The park established two sets of Performance Goals for Year 5 in the monitoring framework. First, 70 percent of the 10 assessed processes, functions, and conditions in the Project Area should show improvement relative to pre-restoration conditions (Parsons 2005). According to the framework plan, three (3) of the seven (7) should be Freshwater, Tidal, and Water Quality (Parsons 2005). Secondly, at least 40 percent of these processes, functions, and conditions should be comparable to reference systems in terms of the natural range of temporal and spatial variation observed within "Control" sites (Parsons 2005).

Changes in Project Area with Restoration – Year 5

While formal quantitative evaluation of Performance Goals may not be possible, at least within the framework originally established by the monitoring program in 2005, a qualitative comparison can be made based on post-restoration conditions and monitoring results from those variables and indicators that were assessed. Based on this qualitative comparison, all eight (8) of the assessed processes, functions, and conditions showed improvement relative to pre-restoration conditions.

Removal of levees, berms tidegates, culverts, ditches, and other “manmade constrictions” increased the width of the stream corridor available for lateral migration on Lagunitas Creek, Tomasini Creek, and Fish Hatchery Creek, as well as increasing tidal prism and connectivity to these creeks’ floodplains (Freshwater, Tidal, Flood Attenuation, Vegetation). Removal of levees also increased the distance that floodwaters could travel during storm events, improving Flood Attenuation capacity. In general, the planform or configuration of old and developing creeks appear consistent with the geomorphic landscape such that channel development processes are probably most strongly influenced by upstream fluvial flow despite some very dry years since the restoration project was implemented. The length of unfragmented riparian corridor was not specifically analyzed, but active revegetation efforts along the southern bank of the East Pasture definitively increased the extent of unfragmented riparian habitat by “filling in” a gap between two established stands. In addition, riparian species have rapidly established along central sections of Fish Hatchery Creek in the West Pasture, extending the length of already existing riparian stands near Sir Francis Drake Boulevard.

Only a few variables were incorporated in the Water Quality function, specifically pathogens, turbidity/clarity, and surface hydraulic connections with channel. The latter GIS-based variable was not assessed, but results of pre- and post-restoration monitoring of water quality documents a sizeable reduction in pathogen levels in the Project Area following restoration, although turbidity also increased 44% after restoration, probably due to reworking of soils left barren following die-off of pasture vegetation. Fecal coliform loading – or the amount of fecal coliform transported downstream – did increase, as well, which was somewhat expected as the ranch was largely diked before and not contributing fully to pollutant loading except under when the dairy ranch was inundated during very large floods. However, in general, water quality conditions within the Project Area have improved dramatically: Dissolved oxygen levels increased 16%, while nitrate, ammonia, phosphate, and phosphorous decreased at least 23%, with some of these parameters falling quite substantially (phosphates; 900%). By Year 5, the only “spikes” recorded in nutrients and fecal coliform occurred in the area near Tomasini Triangle Marsh where a non-point water source continues to contribute high levels of nutrients and pathogens to the freshwater system created to mitigate impacts to the California red-legged frog.

Tremendous changes took place in vegetation communities within the restoration area. Tidal inundation and increasing salinization of the soils very rapidly killed off much of the non-native grass and herb species within the former pastures, leading to establishment of a large expanse of sparsely vegetated panne/mudflats at the lower elevations fringed by tidal salt marsh and tidal brackish marsh at more intermediate elevations. Grasslands – both non-native and native dominated – continue to persist at the highest elevations. This transition led to lower total percent vegetation cover and vegetation canopy complexity compared to pre-restoration conditions, but increases in vegetation community patchiness and appropriate shifts in species composition, including decreases in the percent cover of non-native plant species. Special status plant species within the restored area also jumped both in terms of numbers and areal extent.

For tidal marshes, invertebrates represent one of the cornerstones of food chain support. The transition from pasture to tidal salt and brackish marsh that includes large expanses of sparsely vegetated panne/mudflat has definitely boosted the amount of aquatic edge or boundary between water and non-water habitats. Zooplankton and benthic invertebrate communities have responded to the change from impounded freshwater dairy ranch to open tidal system by showing definitive shifts in species composition and increases in species richness and diversity. However, while benthic invertebrate densities have increased following restoration, zooplankton densities have actually decreased, perhaps because the impounded freshwater conditions naturally promoted higher abundances, particularly of specific freshwater species.

Nekton densities were not specifically assessed during monitoring, but methods allowed for some qualitative evaluation of community changes. Restoration only resulted in small changes in species community composition in the Project Area, with species richness remaining largely the same. Numbers of mosquitofish and Sacramento sucker – two non-native species that were characteristic of the unrestored ranch from a multivariate statistical standpoint – did seemingly drop after levees were breached, but monitoring data are somewhat inconclusive. Perhaps, the biggest change in nekton variables came from the rapid expansion of the federally endangered tidewater goby into the restored wetland, as well as into adjacent creeks such as Lagunitas.

Waterbird numbers have increased every year since restoration, particularly for dabbling ducks, climbing from 5,552 observations in Year 1 to 45,022 observations in Year 5 (ARA 2013). However, shorebird numbers have been more variable. Numbers were low in the first winter after restoration, moderate in the second year and first half of the third year, and then declined sharply in the latter half of the third year, rebounding in Years 4 and 5 to approximately Year 2 levels (ARA 2012, 2013). Giacomini is still evolving as potential shorebird habitat, with this evolution strongly dependent on invertebrate prey bases in marsh muds. In the early years, the restored wetland primarily attracted species that forage in the water column, on the ground, and in shallow sediments (J. Evens, ARA, *pers. comm*.), however, the increase in numbers since fall 2011 of Marbled godwits and Dowitchers, which forage more deeply in muds, may be yet another indicator that numbers of benthic invertebrates in the restored wetlands are increasing.

As with waterbirds, densities of spring breeding birds increased following restoration, particularly for some of the wetland-dependent passerines. Overall, species richness and the total number of observations were actually highest in Year 2 or 2010 than in Year 1, 3, or 4 (ARA 2012). The newly restored wetlands supported 65 breeding bird species in 2009, 74 in 2010, and 48 in 2011 (all detections; ARA 2011a). In general, Song sparrows were the most common breeding bird species, followed by Savannah sparrows, Marsh wren, Red-winged blackbird, and Common yellowthroat. Two species with relatively strict habitat requirements – “Salt-marsh” common yellowthroat and “Bryant’s” savannah sparrow – showed some of the highest increases in Year 4 since restoration. Bald eagles are occasionally sighted at the restoration site foraging, and California black rails appear to have expanded their range from the Undiked Marsh to the north into the restored wetland (ARA 2011b).

Convergence with Natural Marshes – Year 5

For decades, success of restoration projects has been strongly linked with how quickly restored marshes come to resemble nearby natural or reference wetlands. However, use of natural marshes in establishing success criteria is not without its drawbacks, because few so-called “natural” marshes exist that haven’t been altered themselves. This is one of the reasons why the developers of the Hydrogeomorphic Approach (HGM) established the terms Reference wetlands and Reference Standard wetlands, with the latter representing “the highest sustainable level of functions (and are generally) the least altered wetland sites in the least altered landscapes” (Smith *et al.* 1995). While Tomales Bay would appear to embody this description to a high degree, this seemingly pristine watershed still has endured many impacts, the legacies of which persist to this day. Some of these impacts such as invasion by non-native invertebrates or non-native plants are, in large part, due to the proximity of this estuary to San Francisco Bay.
In establishing reference wetlands for comparison with Giacomini, we acknowledged that there are really no pristine wetlands within the watershed (Parsons 2005). Therefore, our reference wetlands were really intended to provide some sense of the natural variability in processes and functions found in existing hydrologically connected tidal wetlands that share strong similarities to the Project Area in terms of site and watershed size, land use history, and structure and patterns of natural and anthropogenic disturbances. Even though two of the three reference wetlands fall in the same Tomales Bay watershed as Giacomini, enough subtle differences in land use history and other factors still exist between these sites that they may be on very divergent evolutionary paths: For example, Walker Creek Marsh has been heavily impacted by historic mercury mining in the Walker Creek subwatershed and improper closure of the mine. To date, the southern portion of Tomales Bay does not appear to have been impacted by mercury, although mercury-laden sediments may eventually work their way southwards. Even the most similar reference wetland – the Undiked Marsh directly north of Giacomini which was once physically connected before the levees were installed – suffers from the fact that the restoration project may also alter the Undiked Marsh, thereby hampering its reliability for use as a reference wetland.

The reference wetland-related Performance Goal established in the Monitoring Program focuses more on similarities between the restored wetland and natural marshes in the range of natural variation than on absolute differences such as means or median. However, ranges are not as frequently tested using statistical methods as means and medians, so most of the results reported in this document discuss differences in the means or median values for key indicators or variables between the two groups. In identifying key indicators for comparison, the Monitoring Program took into account the potential for the restored marsh to converge with reference wetlands. Again, while the Monitoring Program attempted to select wetlands that were as similar as possible, there are still some key differences, including hydrology. The Project Area will always be more highly influenced by freshwater inflow than many of the other sites, as it is the meeting point for three very large creeks and two smaller ones, along with having a sizeable groundwater influence along its perimeter.

Nowhere is this more evident than in the key indicators for Water Quality. After only a short time (five years), at least some of the parameters, including temperature, ammonia, phosphates, phosphorous, and seemingly at least median concentrations of nitrates were actually statistically equivalent to levels in natural marshes. In year 5, pH in the restored wetland begins to show some convergence with natural marshes. Similarly, reintroduction of tidal flow and cessation of freshwater impoundment dramatically altered salinities in the former dairy ranch, but while the pattern of salinity changes once the levees were breached have strongly mirrored those in reference wetlands , the range of salinity values remains quite disparate between Study Areas due to fundamental hydrologic differences between these systems.

Hydrology has also influenced the potential for convergence in invertebrate and fish communities. Following restoration, mean zooplankton densities and benthic invertebrate densities continued to differ between Study Areas, however, the pattern of variation was different. For zooplankton, mean densities were still higher in the restored Project Area (28,748 indiv/ m^3) than in reference wetlands (17,273 indiv/ m^3) despite declines in zooplankton abundance in the Project Area – and natural marshes -- after 2008. This continued disparity appears to be potentially related to the lower salinity waters present in Giacomini, even after restoration. Benthic invertebrate densities increased in the Giacomini Wetlands after the levees were breached, with densities reaching those of pre-restoration natural marshes, but convergence did not occur, because densities also increased 198% in natural marshes. Natural marshes may continue to support higher benthic invertebrate densities, because there is less freshwater inflow: lower freshwater inflow levels can decrease erosion of the sediment substrate and minimize fluctuating sediment salinity levels. Benthic invertebrate communities can be greatly depressed by the scouring effect of large storm events and dramatic changes in sediment salinity levels (Nordby and Zedler 1991).

Following restoration, the restored Project Area and natural marshes did support similar numbers and diversities of zooplankton species, and species diversity of benthic invertebrates was also similar between Study Areas. However, species richness continued to vary, with lower levels of species richness or total number of species in the restored marsh than reference wetlands. The Study Areas continued to show differences in benthic invertebrate species assemblages, although zooplankton species composition did become more similar between the two groups.

Fish composition of the restored Giacomini Wetland continued to differ strongly from natural marshes even after restoration. The Study Areas supported similar numbers of fish species before and after restoration, but the type of species found continued to differ between them, with Giacomini supporting more species such as sculpin, stickleback, and tidewater goby and natural marshes supporting more arrow goby, topsmelt, and surfperch. Essentially, the restored marsh supports a higher number of estuarine or brackish water fish species, while reference wetlands attract more salinity tolerant species that are often found in the open water and more highly saline areas of the estuary.

While avian composition was not compared between the restored marsh and these particular reference wetlands, strong differences may also exist between these groups in terms of the types and numbers of birds. Unlike those in the reference wetlands, the marshplains of Giacomini flood up for long periods of time in the winter and spring due to the increased amount of freshwater inflow and the decreased capacity of the still-evolving to discharge those waters to Lagunitas Creek and Tomales Bay. Marshplains in natural marshes are probably only flooded during extreme tide or storm events and, even then, only for a very short period of time. The prolonged inundation of Giacomini's marshplains undoubtedly attracts species that would not necessarily use the only features that are routinely flooded in reference wetlands such as tidal channels or marsh ponds.

While the goal of many restoration projects is to restore impacted areas to a state in which conditions and functions are similar to those in nearby natural marshes, ultimately, total convergence may not be desirable. Perhaps one of the most notable differences between the restored wetland and its natural counterparts was that the reference wetlands supported a considerable number of opportunistic, invasive invertebrates both before and after 2008, but even four years after levee removal, the Project Area largely did not. Newly restored wetlands are often quickly colonized by a large, if not species-rich, group of opportunistic non-native species that eventually gives way to a more stable assemblage dominated by both opportunistic taxa and those more characteristic of natural marshes (Levin *et al.* 1996, Alphin and Posey 2000, Zajac and Whitlach 2001). In some instances, however, numbers of invasives have been higher in reference wetlands than restored ones (Minello and Webb 1997). Given the high number of non-native species present in adjacent natural marshes, this atypical dynamic may shift in future years. However, continued differences in hydrologic conditions between the brackish Giacomini Wetlands and its more saline tidal marsh counterparts may help, at least in part, to preclude or minimize establishment by some of these species. However, salinity intrusion associated with sea level rise and inter-annual climatic variability – which may only increase in future years (Cayan *et al.* 2012) – could ultimately homogenize hydrologic differences between these systems.

Future monitoring efforts should continue to track convergence between the restored Giacomini wetland and natural marshes, but total convergence may not be either feasible due to continued hydrologic differences or even desirable. Therefore, results of these analyses will need to be interpreted carefully.

Marsh Evolution – What does the Future Hold?

Based on analyses in environmental documents, it was originally estimated that conversion to natural tidal marsh would take a minimum of 10- to 20 years – possibly longer – and would involve establishment of some transitional habitats. However, these expectations shifted somewhat before restoration was completed, because rapid conversion of habitats was already occurring prior to levee removal. Because of this, by fall 2008, we had readjusted our estimates on how quickly tidal marsh would develop to as little as 5 to 10 years.

Some of this acceleration in development may be due to the fact that the former dairy operations had not drastically degraded conditions, nor was the land deeply subsided or much lower in elevation relative to adjacent natural marshes even after 60 years of diking. For this reason, the restoration project did not require extensive land manipulation for restoration, and no phasing was required due to counter possible adverse hydrologic effects of historical subsidence. In a few areas, such as the very northern portions of the West and East Pastures, tidegates had failed, creating a muted tidal regime that already begun reconversion to more natural habitat conditions. In addition, relatively unimpacted natural marshes existed nearby to act as a propagule and organism source for rapid reestablishment of natural salt marsh vegetation, invertebrate, fish, and wildlife communities.

Indeed, for variables in which little change appears to have occurred, such as fish community assemblages, the lack of change may be directly attributable to the fact that fish communities had already transitioned even prior to restoration into a community more reminiscent of natural marshes due to the influence of leaking tidegates and occasional flood overflow into the diked ranch.

Ultimately, habitats and species assemblages are evolving within the restored wetlands, and, in many cases, appear to be drawing closer to conditions and communities present in natural marshes within Tomales Bay and adjacent watersheds. This "successional trajectory" of pasture evolving slowly into natural marsh is predicated on the Gradual Continuum Model of ecosystem evolution. This model assumes a linear model of change from the disturbed condition (diked pasture) to a more natural state (tidal marsh) with removal of agricultural infrastructure. While not necessarily envisioned as a classic successional process, the evolution of the Giacomini Wetlands was anticipated to occur more or less in a

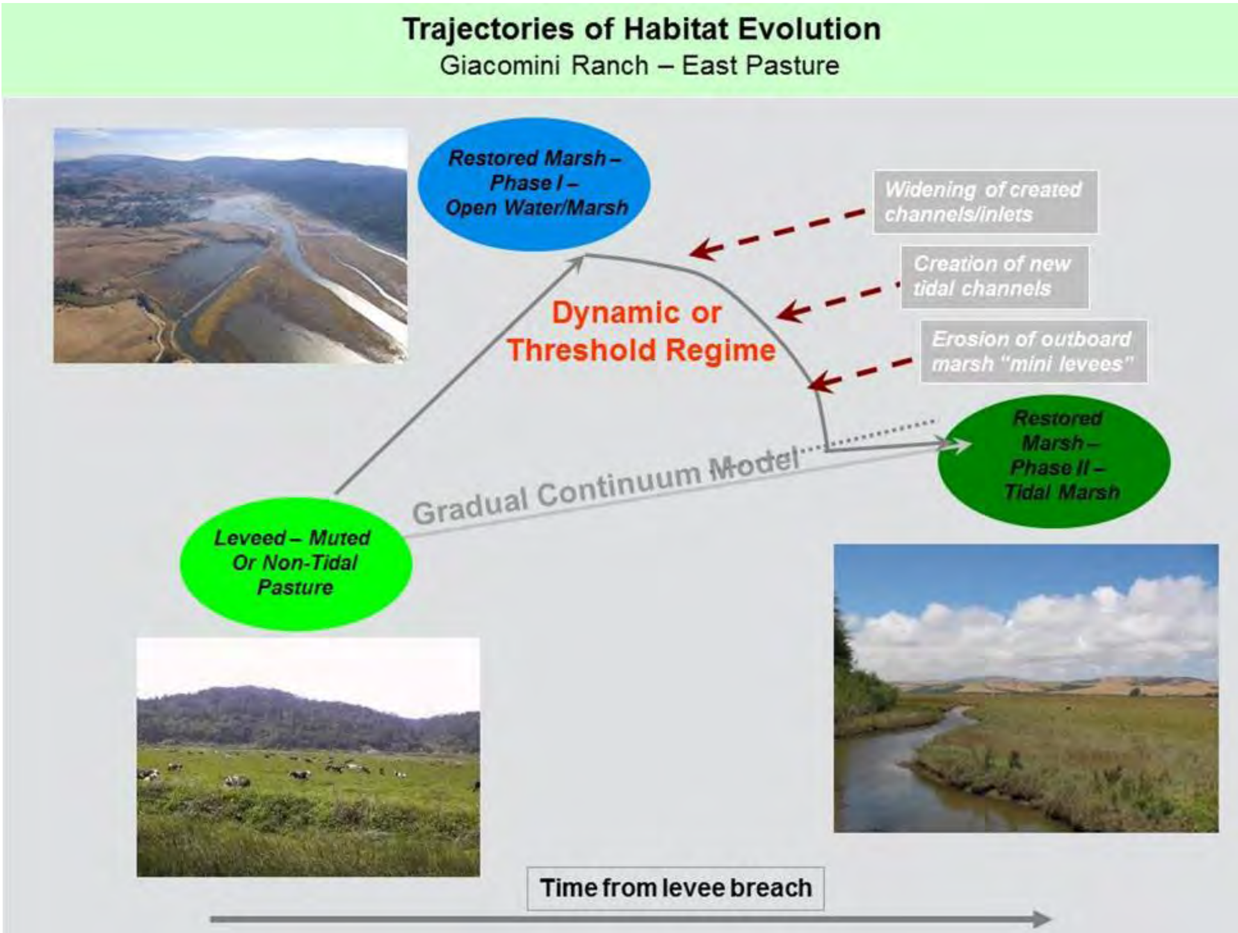

Figure 47. Trajectories of Habitat Restoration

linear fashion or along some type of continuum of change (i.e., A to AB to B). During planning, the trajectory or endpoint for habitat evolution at the Giacomini Wetlands always remained the same. Disturbed conditions or pasture would convert to more natural conditions or marsh. By 2008, the timeline was simply changed to reflect a shorter timeframe between breaching and expected endpoint conditions.

Since breaching of the levees, however, it is apparent that the transition to tidal marsh will not necessarily follow this Gradual Continuum Model of habitat evolution, but more of what is being called a Threshold or Dynamic Regime Model (Suding and Hobb 2009). While certain threshold-type models also assume progression from a disturbed condition to a more natural state with removal of disturbance factors, the progression is not necessarily assumed to be linear or to occur along a continuum of conditions. Rather, under these models, change occurs in a more discrete, step-like fashion, often with abrupt transitions between different states or conditions that require certain "thresholds" to be passed for movement from one state or condition to the next.

In terms of the Giacomini Wetlands, the factors causing more extensive subtidal conditions in the East Pasture and, to a lesser extent, in the West Pasture than predicted originally under levee removal computer modeling scenarios also affect successional trajectories. As was discussed earlier, permanent ponding of waters during even very low tide conditions is occurring currently in a greater percentage of the wetlands than would be expected under fully hydrologically evolved conditions due to the fact that inflow exceeds outflow, because constructed channels were deliberately undersized to allow for natural evolution.

Because of these factors, the wetlands, rather than progressing linearly on a continuum from diked pasture to tidal marsh, are evolving first into a slightly different state or condition characterized by more extensive ponding under subtidal conditions. This different state or condition is depicted in the graphics in Figure 48, which compares the two different models of habitat or marsh evolution. As is shown in Figure 48, the change in successional trajectory has distinct implications for evolution of the system. The extent of intertidal marsh vegetation will be reduced under the Phase I Trajectory, and, so, to some extent, will be development of intertidal mudflats. The larger expanse of open water areas will benefit waterfowl, particularly certain types or groups of waterfowl species such as dabbling ducks, and perhaps encourage more use by non-resident or transient fish species, including some of the larger, open water species such as topsmelt.

Over time, however, this state or condition will continue to change or evolve, as created tidal inlets widen naturally to better accommodate inflows/outflows, and additional tidal channels evolve naturally in the marshplain to better convey flows to sloughs. These changes may start to move this system closer to the original successional or evolution trajectory of tidal marsh (Phase II Trajectory; Figure 48). The transition between the Open Water/Tidal Marsh and Tidal Marsh systems or conditions is expected to change more slowly in the East Pasture than in the West Pasture, because the dense, clay substrate in the East Pasture will slow down the process of inlet widening and natural channel evolution (Figure 48). The West Pasture is underlain by coarser soils that are expected to allow for faster response in inlet widening and channel development to the new hydrologic regimes.

At a certain point, a “threshold” condition will be crossed, probably during a season with high stormflow that encourages loss of the precarious high-elevation outboard marsh shelves (Figure 48). Crossing of this threshold will result in increased drainage of the new wetlands during extreme low tide conditions to a degree more in keeping with that originally predicted by computer modeling and will begin to move the system towards a new “state” or “condition” – tidal marsh system characterized by a lower extent of open water, but increased intertidal marsh and mudflat. This system will have potentially greater benefits for shorebirds and resident marsh bird species, but may have fewer benefits for waterfowl and non-resident and larger estuarine fish species (Figure 48). What this means is that the evolution of conditions and species assemblages that we have witnessed so far may never necessarily reach a steady-state condition characteristic seemingly of natural marshes, but may continue to evolve in a somewhat staged fashion to internal and external factors such as channel evolution or climate change-related factors such as sea level rise.

Ultimately, this restoration project will provide an invaluable insight into the issues that affect the path or trajectory of restoration of degraded wetlands back to non-degraded conditions – insights that will prove valuable for planning of future restoration projects. In addition, it will help ecologists to better understand the processes of change for all communities, whether they be part of a restoration project or not.

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12 Where Do Monitoring and Restoration Efforts Go From Here?

Monitoring

This report summarizes the results of pre- and post-restoration monitoring through Year 5 following implementation of the Giacomini Wetland Restoration Project. The original long-term monitoring program framework developed in 2005 called for additional monitoring in Years 7, 10, 15, and 20 (Parsons 2005). According to the document, monitoring for Year 7 would be spearheaded by the Seashore, while Year 10, the final year of formal – but not required – reporting, might be contracted out to a university or consultant (Parsons 2005). It was anticipated at that point that monitoring for Years 15 and 20 might ultimately be incorporated into the San Francisco Bay Area Network's Inventory & Monitoring Program (Parsons 2005). The intensity of monitoring efforts was projected to remain the same for Years 1-5, 7, and 10, but possibly be reduced during Years 15 and 20, as "a scaled-down approach to assessing Project success will undoubtedly be necessary," given that resources will need to be shared with other monitoring priorities (Parsons 2005).

Following completion of monitoring in Year 5, it became apparent that, if monitoring of the restored Giacomini Wetlands is to continue at any level in the future, the intensity of efforts will probably be far less than what was envisioned at the outset of the Giacomini Long-Term Monitoring Program. The difficulty in securing external funding for long-term monitoring efforts means that those efforts that are undertaken will need to be extremely cost-effective and require relatively little investment in terms of financial or staff resources. Also, it is unlikely that the Inventory & Monitoring Program will assume responsibility for any future monitoring efforts, given that financial resources for this program are extremely limited, as well.

We may also need to shift our paradigm in terms of what should and should not be monitored. Originally, we planned to discard long-term monitoring of variables that would not show a "clear and distinct pattern of change with restoration" such as groundwater levels or variables that would not necessarily converge with Reference Areas such as water salinity. Salinity conditions were never expected to totally converge with those of Reference Areas, because the Project Area receives more direct, abundant, and perennial freshwater input than the natural marshes monitored. However, while salinity may not represent a good indicator for evaluating improvement in Project Area conditions or convergence with Reference Area conditions, it may be important as a harbinger of direct and indirect effects resulting from climate change, including changes in pH, water level, extent of high tides, and salinity. We also considered only incorporating those variables with a low coefficient of variation among all reference sites, but climate change is likely to increase the range of variation, even among variables that were relatively consistent previously.

In selecting variables for possible future monitoring, we have focused on those that provide the highest informational value for the lowest cost in terms of staff and contractor time. These variables speak strongly about evolution of the marsh, both in response to restoration and to non-restoration-related factors such as climate change. In addition, as the Park Service is directed to assist with recovery of threatened and endangered species, variables related to abundance of or extent of habitat use by listed species are also included. The following table outlines some of the proposed variables for future monitoring (Table 8). It ranks variables considered of moderate to high informational value by both cost and logistical intensity (e.g., does it require considerable staff time? processing? Hiring of a contractor? etc.). Internal and external costs are factored into levels that range from 1 (low cost, easy to implement) to 4 (high cost, more effort-intensive). The table also notes which of these have the potential to provide information about the effects of climate change on the restored wetland and watershed.
In general, most of the variables selected focus on areas where continued evolution of the marsh is expected, such as the aerial extent of inundation at low tide and vegetation, benthic invertebrate, and waterbird communities (Table 8). The design for Giacomini relied on construction of only a few primary and secondary tidal creek channels, with establishment of a full tidal channel network expected to occur over time in response to tidal and freshwater inflows. Because of this, water levels right after restoration

was implemented remained unnaturally high at low tide, because the constructed channel network was unable to fully drain before the next high tide. As tidal channels began to evolve, however, the aerial extent of inundation at low tide decreased, particularly in the East Pasture. This hydrologic evolution has important implications for vegetation establishment, benthic invertebrate (and zooplankton) communities, as well as waterfowl and shorebird use. In addition, it could drive patterns of use by species such as tidewater goby, the federally endangered resident brackish water fish species.

All of these variables are also subject to being influenced by climate change-related factors, as well. The response of the restored wetland to sea level rise and increased wave- and storm surge may depend greatly on sedimentation and organic matter deposition on marshplains and changes in marshplain elevation; where contributions are low or non-existent, marshplains may be unable to keep up with sea level rise, thereby leading to loss of vegetated habitat or a conversion in habitat type. Loss of habitat for vegetation and wildlife may also result from sea level rise and decreased freshwater inflows increasing salinities within the predominantly brackish habitats that currently occur in Giacomini, thereby greatly reducing optimal habitat for brackish and freshwater species such as tidewater gobies and California red-legged frogs. Following restoration, most of the water quality issues associated with diking of the former marsh for agriculture resolved very quickly, with the exception of a pollutant "hot spot" associated with a non-point source discharge near the created Tomasini Triangle Marsh. Continued monitoring of water quality in the marsh or pond and of the discharge point is also suggested.

Table 8. Variables Proposed For Future Monitoring.

Monitoring Target	Variable(s) measured	Description	Internal Resources	External Cost	Level	Climate Change Factor
Hydrologic Processes – Tidal – Surface/ Freshwater - Surface	Aerial Extent of Inundation at Low Tide (East Pasture only)	The volume of water remaining at low tide indicates how well the marsh is exchanging water with Lagunitas Creek/Tomales Bay due to development of tidal channels and	Low (staff, GPS, GIS)	$0	1	Y
Sediment Deposition/ Nutrient-Contaminant Retention – *Sediment Deposition*	SET + Feldspar	SET stations already installed show changes in marshplain elevation over time, while paired feldspar monitoring plots indicate how much sediment has accumulated on marshplains.	Low-Medium (staff, eqmt)	$0	2	Y
Sediment Deposition/Nutrient-Contaminant Retention - *Water Quality (WQ)*	Salinity	Water salinities in Project and Reference Areas monitored during summer using continuous water quality instruments such as HOBO temps. Patterns should indicate how salinities changing in response to climate change and how changes might affect habitat for special status species.	Medium (staff for deployment, download/analysis of data)	Low	1	Y
Sediment Deposition/Nutrient-Contaminant Retention - *Water Quality (WQ)*	Nitrates, Fecal Coliform – Tomasini Triangle Marsh	Nitrates and fecal coliform would be monitored quarterly in Tomasini Triangle Pond and inflow locations along the pond perimeter to track pollutant patterns in the one pollutant "hotspot" that remains following restoration.	Medium (staff for sampling, download/analysis of data)	Medium (lab costs for sample analysis)	2	N

Table 8. Variables Proposed For Future Monitoring.

Monitoring Target	Variable(s) measured	Description	Internal Resources	External Cost	Level	Climate Change Factor
Maintain Characteristic Plant Community – *Vegetation (VG)*	Vegetation Community Patchiness – Vegetation Mapping	Vegetation communities within the restored wetland would be mapped using broad-level classification such as Salt Marsh, Brackish Marsh, Freshwater Marsh, Grassland – Native and Non-Native. Changes would be expected in response to restoration and sea level rise.	High (staff for mapping, GPS, GIS)	$0	3	Y
Maintain Characteristic Plant Community – *Vegetation (VG)*	Vegetation Community Composition - Quantitative change	Established vegetation transects would be re-occupied to quantitatively measures vegetative community change over time, including changes in broader plant communities (salt marsh, grassland, freshwater marsh), similarity of salt marsh areas to reference sites, and measuring the effects of changing climate and sea level rise on the communities. Monitoring would occur in late spring to summer.	Medium (staff for sampling vegetation communities, conducting comparative analysis)	$0	2	Y
Food Chain Support-Invertebrates – *Benthic Invertebrates*	Benthic Invertebrate Species Composition and Diversity	Benthic invertebrate communities changed greatly after restoration and represent the cornerstone of the food web for many species, including migrant shorebirds. Also, this is one of the most vulnerable variables in terms of potential impact from invasion by non-native species present already in Tomales Bay. Also, shifts in salinity from climate change could affect species composition and abundance.	High (staff for sampling, processing of samples, download/ analysis of data)	High (costs for sample processing and species identif-ication)	4	Y
Wildlife Habitat and Use – Nekton/*Tidewater Goby (TWG)*	Tidewater Goby Aerial Extent and Relative Abundance	Tidewater goby is a federally endangered brackish water fish whose numbers and aerial extent appeared to expand after restoration. However, potential increases in salinity resulting from climate change could impact this species. Monitoring would be conducted once a year in early summer.	Medium (staff for sampling, download/analysis of data)	$0	3	Y

Table 8. Variables Proposed For Future Monitoring.						
Monitoring Target	**Variable(s) measured**	**Description**	**Internal Resources**	**External Cost**	**Level**	**Climate Change Factor**
Wildlife Habitat and Use – Avifauna and Amphibians – *California red-legged frog (CRLF)*	California Red-Legged Frog Aerial Extent and Abundance	California red-legged frog is a federally threatened species adapted to breeding in freshwater conditions. Frogs moved into at least one of the mitigation marshes created, but the other two remain poorly used, possible due to the presence of bullfrogs. Potential increases in salinity resulting from climate change could impact this species. Monitoring would be conducted during the breeding season.	Low (staff for contracting, data analysis)	Medium (contractor to perform surveys, data reporting)	2	Y
Wildlife Habitat and Use - Avifauna and Amphibians – *Avifauna*	Waterbird Composition and Density	One of the most dramatic changes following restoration was the explosion in waterbird use during fall, winter, and early spring. Waterbird use patterns could continue to change in response to continued marsh evolution and changes in salinity resulting from climate change factors.	Low (staff for contracting, data analysis)	Medium (contractor to perform monitoring, data reporting)	2	Y

Using Monitoring Information for Better Management and Restoration

One of the values of this monitoring program is that it enables the Seashore to pinpoint areas where remedial action and further future restoration might be necessary. Described below are some of the issues identified through monitoring that could be addressed through future mitigation or adaptive restoration efforts.

Tomasini Triangle Marsh: Even after restoration, consistently high nitrates and, at times, fecal coliform levels have been detected flowing into the southern side of the newly created Tomasini Triangle Marsh. Prior to restoration, approximately 7 percent of the samples exceeded 44 mg/L – NO3 equivalent to the 10 mg/L nitrate-N EPA standard for human consumption – and all of these exceedances came from this inflow sampling point. In addition, fecal coliform levels consistently exceeded 160,000 mpn/100ml. It was hoped that removal of agricultural management as part of restoration would improve conditions in this area, particularly as the marsh was constructed as habitat for federally threatened California red-legged frog. However, as discussed earlier in this report, while pollutant levels have dropped dramatically after restoration elsewhere in the Giacomini Ranch, they have remained high in this area, accounting for quite a few of the outlier points in graphs (Figure 7). This sampling site continues to show elevated nitrates even after restoration and exceeded 10 mg/L during every sampling event in Years 2, 3, and 4 and 75% of the events in Year 1.

Some of these waters being conveyed to the marsh appear to come from a ditch on the Point Reyes Mesa that funnels stormwater run-off during periods of heavy rainfall from the southwestern portion of the town into a swale that flows into the marsh. The Regional Water Quality Control Board had sampled this ditch in 2001 as part of a study of pathogens in Tomales Bay (RWQCB 2002) and found that fecal coliform levels were elevated during storm events, with levels ranging from 333 to 4,100 mpn/100 ml depending on the storm event and time of sampling during the event (RWQCB 2002).

While this stormwater run-off source accounts for some of the inflow into the Tomasini Triangle Marsh, site investigation has revealed that there are other sources of pollutants to the newly restored wetlands. A PVC pipe was found upslope of the Seashore's sampling point that conveys a considerable amount of water to the marsh throughout the year. Sampling of this non-point source discharge over 6 months in Year 2 showed consistently high nitrate levels ranging around 30 mg/L, with concentrations occasionally as high as 53 mg/L (R. Carson, TBWC, unpub. data). The source of this discharge is not entirely certain, but, based on some planning documents that were reviewed, this pipe may have been installed originally in the 1980s to improve overall drainage of groundwater in the Point Reyes Mesa and, thereby, improve conditions for installation of septic systems associated with new residential development. Unfortunately, either current or past sources of pollutants are apparently being "captured" by this groundwater flow diversion and diverted into the newly restored wetland, thereby greatly elevating nutrients in this freshwater marsh created as special status species habitat.

Interestingly, fecal coliform levels in outflow from this pipe were typically low, and MBAS – the surfactant found in detergents – was only detected in trace amounts (R. Carson, TBWC, unpub. data). In contrast, fecal coliform levels at the sampling site on the marsh boundary during dry sampling events still continue to be high, although they have dropped to some degree after restoration. These results suggest that, even if the quality of stormwater run-off was improved, and the pipe outflow was eliminated, there would still continue to be inflow of pollutants into the marsh, probably due to the influence of nearby septic systems on the groundwater table.

While no one action may solve this issue, any management or restoration actions undertaken could reduce pollutant levels and improve quality of the Tomasini Triangle Marsh, which supports numerous birds, fish, and amphibians, some of which are federally listed species. In addition, reductions in nutrients may reduce spread or establishment of invasive non-native species that are now present in the marsh such as cattails (*Typha angustifolia*; *Typha Xglauca*) or even native floating emergent species that can establish monocultures in high nutrient conditions (*Hydrocotyle ranunculoides; Azolla filiculoides*).

Olema Creek Frog Ponds: Several seasonally flooded ponds totaling 0.73 acre were created in 2007 on the west side of Olema Creek less than 0.5 miles from Olema Marsh to offset potential short- and long-term impacts to California red-legged frog breeding habitat in Olema Marsh. The lower reaches of the Olema Creek watershed just above its confluence with Lagunitas Creek started supporting breeding red-legged frogs after the creek reestablished connectivity with its historic eastern floodplains and converted pasture to a complex marsh system with both permanently and seasonally flooded habitats.

As part of the pond design, the pond bottom was "excavated to create varying water depths that would support emergent and open water habitats." However, USGS amphibian biologists have concluded from monitoring conducted after construction that the ponds are not being utilized as well by California red-legged frogs as they could be due to the fact that the deeper holes left in the pond bottom are encouraging use by bullfrogs instead, which may be competing with red-legged frogs. USGS biologists have advocated that these deeper holes be filled with sediment to the relative elevation of the surrounding pond bottom in efforts to promote more successful establishment and use by red-legged frogs.

The Park Service has received an amendment to its Biological Opinion to allow for filling of the deep holes. Further permitting is required before this project can proceed.

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RWQCB (2002). Total Maximium Daily Load for Pathogens in Tomales Bay, Preliminary Report. California Regional Water Quality Control Board, San Francisco Bay Region. November 15, 2002.

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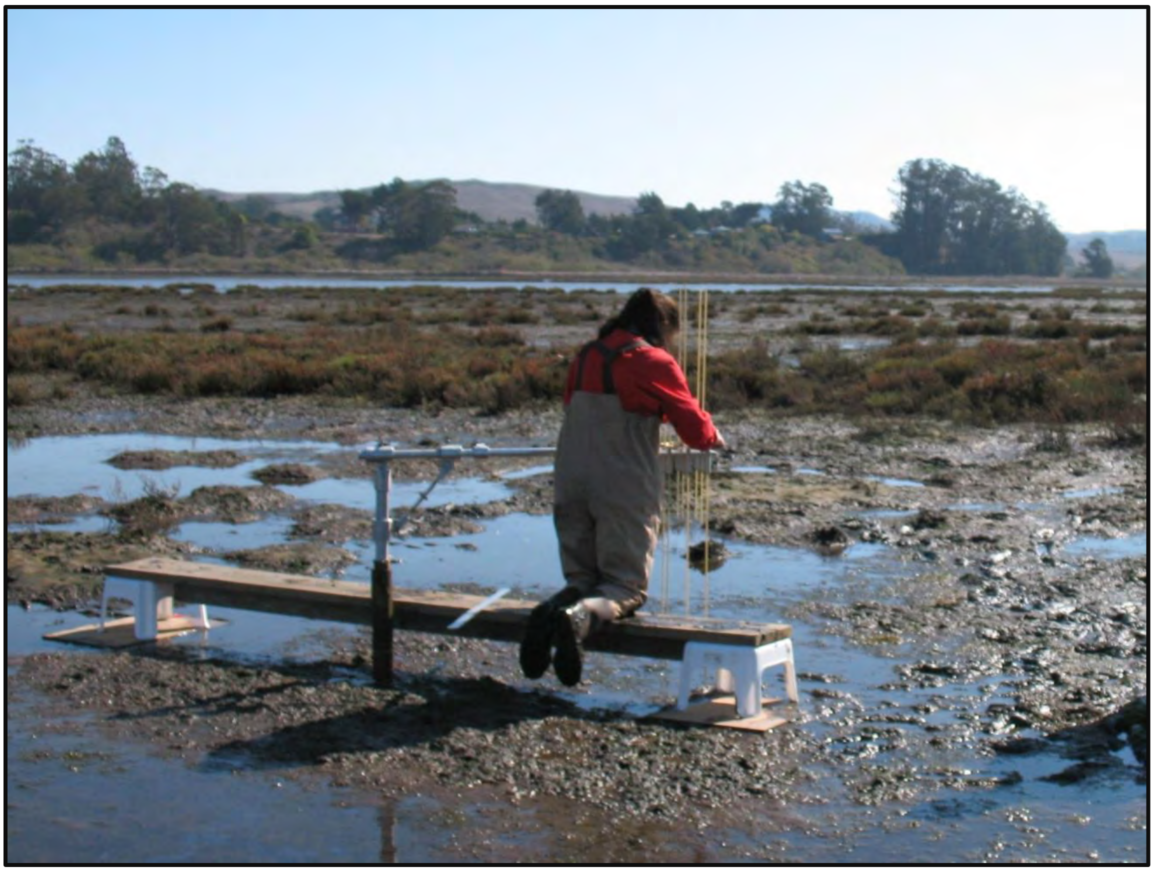
Appendix A. Results of SET and Marker Horizon Monitoring



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Keeping Pace with the Tides
Results of SET and Marker Horizon Monitoring – Giacomini Wetland Restoration Project, Tomales Bay

Lorraine Parsons and Amelia Ryan
Point Reyes National Seashore
March 18, 2014

Introduction

For more than 100 years, sedimentation has been a significant issue in the Tomales Bay watershed. Historically, attention has focused on the problems of having too much sediment, which has shallowed the Bay, converted mudflat to marsh, and potentially impacted salmon fisheries by smothering eggs in gravel streambeds within the upper watershed. The Regional Water Quality Control Board, in fact, is currently developing a sediment management program under the auspices of a Sediment Total Maximum Daily Load (TMDL) plan for Tomales Bay's largest contributing watershed, Lagunitas Creek. However, with the specter of sea level rise now looming on the seemingly not-too-distant horizon, sediment may ultimately become more of a critical resource than a pollutant.

This attitudinal "sea change" towards sediment has already occurred in managers overseeing tidal wetland resources in San Francisco Bay. For decades, San Francisco Bay was characterized as a low productivity estuary relative to other temperate-latitude counterparts, partly due to its turbid waters, which decreased light penetration and clarity and reduced phytoplankton densities. However, studies have shown that the delivery of suspended sediment to San Francisco Bay from the Sacramento River decreased by about half during the period 1957 to 2001 (Wright and Schoellhamer 2004). The erodible pool of sediment that was deposited in the San Francisco Bay in the 1800s from a quarter of a billion cubic meters of sediment being washed down the Sacramento River from hydraulic gold mining operations in the Sierra Mountains apparently finally became depleted in the late 1990s (Schoellhamer 2011, Cappiella *et al.* 1999, Fregoso *et al.* 2008, Jaffe *et al.* 1998). With this depletion, suspended sediment concentrations in San Francisco Bay waters dropped by 40% (Schoellhamer 2009). The result has been not only a less turbid estuary whose phytoplankton productivity rates have climbed considerably, but heightened concern about whether current sediment supplies can support the extensive wetland restoration planned or already being implemented, as well as counteract the future impacts of sea level rise.

Sediment issues in Tomales Bay are somewhat different from those in San Francisco Bay, but no less critical to the long-term persistence and future viability of its wetlands. Similar to San Francisco Bay, sediment issues in Tomales Bay date back to the 1800s. Starting with ranching development and logging in the mid- to late 1800s, the pace of sediment delivery from the surrounding hills and mountains greatly intensified. While historic baseline rates of sediment transport are unknown, they were probably high due to the natural geologic instability of the surrounding watershed, which is shaped substantially by the San Andreas Fault that runs through its center.

Comparison of hydrographic charts prepared in 1861 and 1994 and correction of the charts for changing sea level indicated that, baywide, Tomales had an average infilling rate of about 0.2 in/yr (5 mm/yr; Rooney and Smith 1999). This is equivalent to a watershed erosion rate of approximately 80,000 tons/yr (Rooney and Smith 1999). The highest rates of sedimentation at the southern end of Tomales Bay occurred between 1861 and 1908 (PWA *et al.* 1993), but, apparently for other portions of the Bay, the largest sediment influx to the bay seems to have been in the decades between about 1930 and 1960 (Rooney and Smith 1999). Between 1861 and 1931, sedimentation accumulation rates within Tomales Bay averaged 94 tons/k^2/year (yr), increasing to 357 tons/k^2/yr between 1931 and 1957 and decreasing to 101 tons/k^2/yr between 1957 and 1994 (Rooney and Smith 1999). These sedimentation patterns contrast somewhat with findings from the PWA *et al.* (1993) study of southern Tomales Bay and delta expansion, which characterized the 1861-1931 period as having the highest sedimentation rates.

Rooney and Smith (1999) noted, however, that sediment yield in the Bay is not necessarily synonymous with erosion and that there can be “decades long delay between maximum level of soil surface disruption and maximum sediment deposition.” During these decades, sediment is typically stored in streambeds, gradually moving towards the Bay through episodic resuspension during storms. Another storage reservoir for sediment is stream deltas such as Lagunitas Creek: “A similar delay was found between initial deposition of sediment at stream deltas and subsequent redistribution other areas of the Bay more geographically remote from deltas” (Rooney and Smith 1999).

Sedimentation resulting from erosion induced by agricultural development of the watershed is likely to have been highest first at the southern end or mouth of the Bay, with the rapidly accreting delta and construction of the Giacomini Ranch levees eventually shifting the primary area still available for sediment deposition downstream and more into the Bay itself. It has been estimated that the peak sedimentation period between 1860s and 1910 resulted in deposition of almost 5 vertical feet of sediment (PWA *et al.* 1993) and 250 -300 acres of new intertidal marsh in very southern portion of Tomales Bay, principally the Giacomini Ranch and area directly north of the ranch (PWA *et al.* 1993).

Unlike its large neighbor to the south, the Tomales Bay estuary did not appear to have historically the extensive network of fringing salt marshes that were once present in San Francisco Bay (Parsons and Allen 2004). U.S. Coast Survey maps from the 1860s and 1870s depict small amounts of marsh habitat along the edges of Tomales Bay, with the largest extent in the southern portion of Tomales Bay in what are currently the East Pasture, Olema Marsh, and the Bear Valley and Olema Creek floodplains (Parsons and Allen 2004). The existing Undiked Marsh currently north of Giacomini Ranch appeared to be largely unvegetated or sparsely vegetated subtidal and intertidal mudflats (Parsons and Allen 2004). Walker Creek Marsh, one of Tomales Bay’s other large undiked marshes, does not even exist in the Coast Survey maps, with the marsh area shown as subtidal area and intertidal flats (Parsons and Allen 2004).

The dramatic increase in sedimentation associated with logging and agricultural development had the inadvertent effect of also dramatically increasing deltaic aggradation at the mouths of creeks such as Lagunitas and Walker, which are Tomales Bay’s two largest subwatersheds. Between 1863 and 2001, wetland acreage almost doubled to 944.2 acres due to this sedimentation (Parsons *et al.* 2004), and the Lagunitas Creek delta more than doubled in acreage and length during this period, with the tip of the delta extending another 2,100 feet beyond its 1863 boundaries by 2001 (NPS 2007). The 1982 flood alone caused deposition of 160 acre-feet of sediment on the Lagunitas and Walker Creek alluvial deltas (Anima *et al.* 1983 *in* PWA *et al.* 1993).

Some of this sedimentation resulted in conversion from what appeared to be open estuarine systems with large embayments and little to no marsh habitat into salt marsh estuaries with significant marsh plain and tidal channels (Parsons *et al.* 2004). This increase in intertidal wetlands and decrease in subtidal and intertidal mudflats was accompanied by an overall “shallowing” of Tomales Bay that effectively ended use of ships for transport of goods to San Francisco markets. Until the 1880s, 100-ton steamships reportedly navigated Lagunitas Creek – formerly known as Papermill Creek – on high tides to the old paper mill located near the existing Green Bridge (PWA *et al.* 1993).

Since 1957, sedimentation rates have dropped, slowing the pace of Bay infilling (Rooney and Smith 1999, PWA *et al.* 1993). This is most likely due to construction during the 1950s of five Marin Municipal Water District (MMWD) dams within the Lagunitas Creek watershed, which control 70 percent of the runoff for this subwatershed (PWA *et al.* 1993). Similarly, construction of dams in the Sacramento-San Joaquin watershed has also dramatically reduced the Central Valley contribution to the San Francisco Bay sediment budget, potentially accounting for large erosional losses in shallow areas in San Pablo and Suisun Bays observed between 1942 and 1990 (Jaffe *et al.* 1996, 2001 *in* McKee *et al.* 2002).

Because dams tightly regulate some of the smaller flood flow events and their sediment loads, the highest rates of sedimentation in this subwatershed may now come with catastrophic flooding just as is seen currently in the Santa Clara River (PWA *et al.* 1993, Stillwater Sciences 2005). A 1979-1980 Tomales Bay study, however, asserted that there were still substantial sediment contributions from unregulated tributaries and their watersheds, even with the dams (H. Esmaili & Associates 1980). Current trends in the upper portions of the Lagunitas Creek watershed have not been formally studied, but several researchers have reported problems with "fining" or excessive deposition of fine sediments such as clays and silts relative to coarse materials such as gravel and cobble; poor sediment recruitment below the dams; and armoring of smaller gravel and fine sediments (Stillwater Sciences 2004).

To determine trends in sediment transport processes within the Giacomini Wetlands prior to restoration, KHE conducted computer modeling, as well as sampling gravel bars in Lagunitas Creek (KHE 2006). The coarse-grained nature of these surficial gravel bed deposits indicated that Lagunitas Creek possessed a relatively high sediment transport capacity through the Giacomini Ranch even when it was still leveed (KHE 2006). Modeling results indicated that creek flows were sufficient to mobilize and transport fines, coarse sand, and fine gravel observed within the Giacomini Wetlands (KHE 2006) despite flattening of the creek gradient or slope. However, based on modeling results, stream energy did not appear high enough to convey larger material such as coarse gravel and cobble (KHE 2006).

Prior to restoration, fine or suspended sediment appear to be deposited within adjacent floodplains when flows were sufficient to crest the levees (KHE 2006). One of the highest depositional areas in the East Pasture appeared to be the southwestern corner opposite White House Pool, where the Giacominis apparently deliberately removed or lowered levees to preferentially direct flooding (KHE 2006), perhaps because of repeated flooding problems in the past during more frequently occurring flood events (~ 3 to 5 years). The 1942 aerial photograph shot just prior to establishment of the Giacomini Ranch and following an average winter without excessive flood scour or sedimentation clearly shows overbank scour and sediment deposits within the southeast portion of the East Pasture, along Lagunitas Creek, and within the West Pasture, from historic overbank flooding events (KHE 2006).

Based on modeling, sediment transport rates during overbank flooding events were probably highest just at the point of entry near the south levee of the East Pasture, with flow velocity dropping sharply throughout the remainder of the pasture (KHE 2006). Modeled flow velocity in the south was high enough to transport coarse sand and fine gravel, which, then, would be deposited in the southernmost fields. This modeling-based conclusion agreed with information from sediment coring and aerial photographs (KHE 2006). Prior to restoration, sediment transport in the northern portion of the East Pasture was hindered by persistent ponding of floodwaters caused by reduced outflow, which was limited by the concrete spillway and culvert capacity (KHE 2006). In the West Pasture, flow velocity during highly infrequent overbank flooding events (> 12 years on average) did not appear sufficient to transport sediment through the pasture, with most sediment probably deposited immediately on the floodplain after cresting the levee (KHE 2006).

While watershed sediment contribution has decreased in the last 50 years, Tomales Bay still continues to become shallower through sediment inputs. In addition, colonization – or re-colonization – by native Pacific cordgrass (*Spartina foliosa*) appears to have caused conversion of shallow intertidal mudflat to vegetated marsh in some areas of the Bay. The most recent estimate of the sedimentation rate in the bay, which is based on both bathymetric changes since 1957 and sediment yield measurements, is about 0.04 to 0.08 in/yr (Smith and Hollibaugh 1998).

Now that Giacomini Wetlands are restored, the hydrologically reconnected floodplains could potentially reduce sediment transport to Tomales Bay. Based on earlier estimates, as much as 19% of the sediment load in Lagunitas Creek during a bankfull or ordinary high water storm event (return interval of ~2 yr) could be diverted into the Giacomini Wetlands by overbank flooding, where sediment could be potentially trapped and kept from entering the Bay (NPS 2007). Some of the sediment deposition dynamics depend on a complex interaction between storm frequency, magnitude, intervals between storms, and upstream reservoir levels, such that smaller storms often do not trigger overbank flooding events, because dams capture most of the flood volume and diminish peak flows.

Ultimately, these episodic overbank flooding events could prove crucial in terms of long-term sustainability of the restored wetlands. Some recent hydrologic analyses indicate that sea level rise could encroach substantially into the newly restored wetlands, converting mudflat and lower intertidal areas into subtidal habitats and higher intertidal habitats into mudflat, particularly if sedimentation and other marsh-building processes such as organic matter accumulation cannot keep pace with sea level rise rates (KHE 2009). In 2009, the Pacific Institute published a report that warned that sea level rise rates in California could climb as high as 1.4 m (4.6 feet) by 2100 (The Pacific Institute 2009). Should sea level rise that high, the report estimated that as much as 41 sq. miles or 26,000 acres of California coastline could be lost due to increases in wave erosion and amplified tides (The Pacific Institute 2009).

The threats from sea level rise are exacerbated to some degree by the fact that at least minor subsidence also took place in the Giacomini Wetlands after levees were installed. While not as severe as that occurring in San Francisco Bay or Sacramento Delta wetlands, where subsidence or compaction of soils lowered elevations by as much as 7- to 20 feet, northern portions of the marsh do appear to have dropped at least 1- to 2 feet relative to the adjacent Undiked Marsh (PWA *et al.* 1993). The southernmost portions of the Giacomini Wetlands are actually aggraded well above elevations of the adjacent Undiked Marsh to the north of the Giacomini Wetlands, which range from +3 (low marsh) to +7 feet (high marsh/upland ecotone) NAVD88, with the marsh plain at approximately +5 to +6 feet NAVD88 (USGS 2003). Some of the aggradation may result from land leveling and deposition of fill and manure, but, as discussed earlier, the Giacominis also removed the southwestern portion of the East Pasture levee deliberately to preferentially direct flood flows into this portion of the property (KHE 2006).

Brief Methods Summary

As part of the Giacomini Wetland Long-Term Monitoring Program, a series of Sediment Elevation Tables (SET) stations were installed in the Project Area (Giacomini Ranch), as well as reference marshes in Tomales Bay (Undiked Marsh, Walker Creek Marsh) and Limantour Estero (Limantour Marsh) to measure relative elevation changes (Figure A-1). In addition, feldspar marker plots were set up directly adjacent to these stations to track sedimentation rates.

This monitoring was initiated for most areas in 2006, although the East Pasture South SET station was not installed until 2008. Due to some issues with protocol, comparisons between all years is not necessarily available, although, in 2010, a complete reading of all SET angles was performed, enabling evaluation of changes between 2006 and 2010 and 2008 and 2010. Subsequently, these problems were resolved, enabling analysis of annual changes in elevation. A more complete description of the SET and marker horizon methodologies can be found in the *Long-Term Monitoring Program: Part I. Monitoring Framework* document (Parsons 2005).

Study Area Description

Tomales Bay - Giacomini – West Pasture: While both the West and East Pastures of the former Giacomini Ranch were leveed for dairy operation, by 2000, the West Pasture had already become somewhat of a muted tidal system due to a leaky one-way tidegate on Fish Hatchery Creek that allowed some inflow, as well as outflow. This tidegate malfunction increased tidal influence within the West Pasture, which already had converted back to salt marsh in some areas, particularly the northern areas. This slow conversion back to tidal marsh communities probably resulted from compaction of marsh surface soils over time after being leveed in the 1946, which increased exposure of these soils to the underlying subsurface groundwater table that contained residual salts dating back to pre-levee periods. While the West Pasture was largely isolated from Lagunitas Creek, the pasture did receive some flood flows – and sediment – from Lagunitas Creek during extreme flood events, as well as sediment from Fish Hatchery Creek, which flows from the Inverness Ridge through the pasture, exiting at the northern perimeter. Since 2006, one SET station has been monitored in the northernmost portion of the West Pasture.

Tomales Bay - Giacomini – East Pasture: Unlike the West Pasture, the East Pasture remained leveed off from Lagunitas Creek and tidal flow until restoration. As with the West Pasture, however, the East

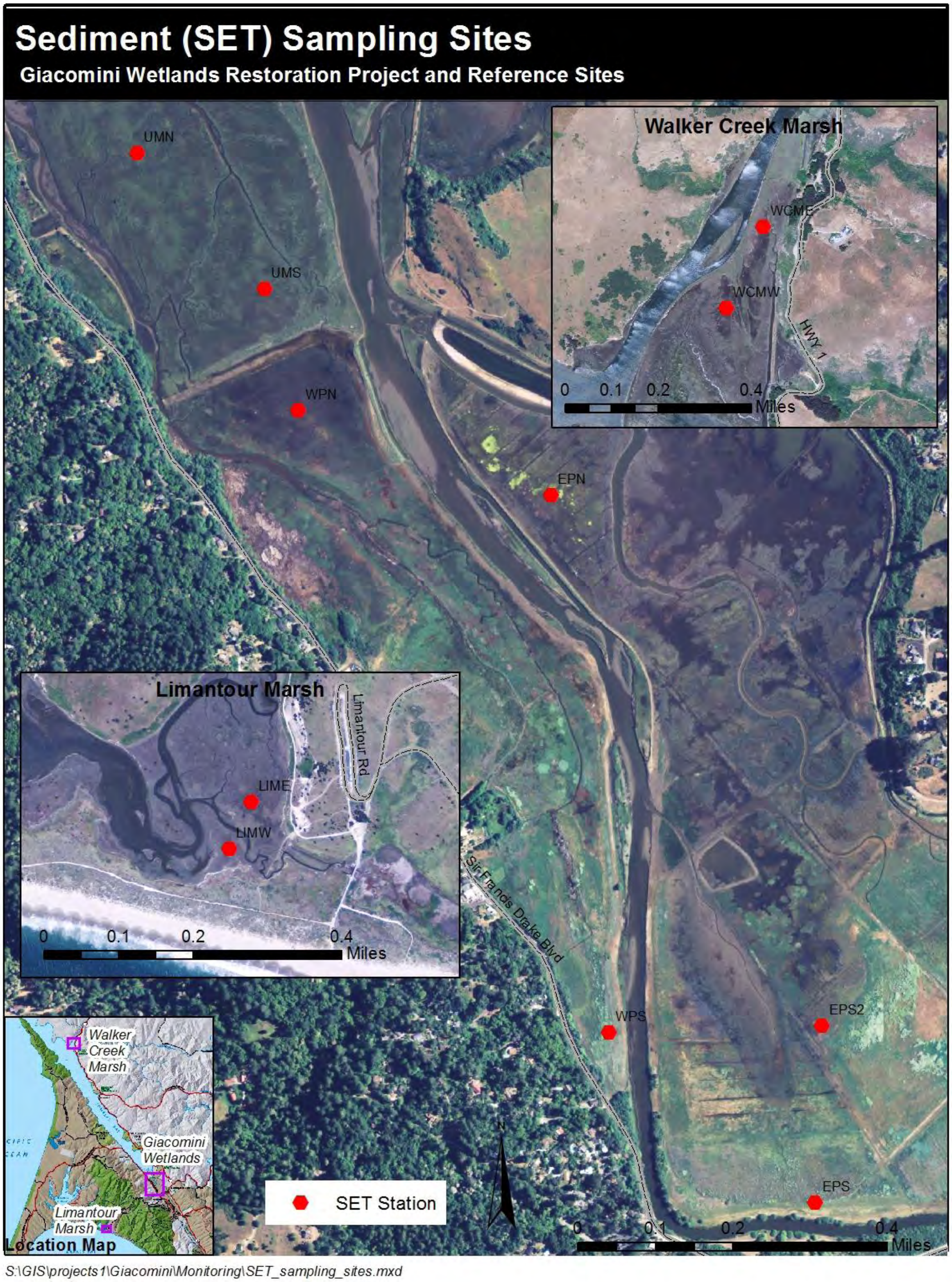

Figure A-1. Sedimentation (SET and Feldspar) Sampling Sites.

Pasture had subsided by as much as 1- to 2 feet: the East Pasture is underlain by more estuarine mud in the subsurface strata than the West Pasture, which is underlain by more granitic alluvium. This may account for higher subsidence rates in the East Pasture. This subsidence, combined with interaction of the now higher groundwater table with residual salts in the former marsh soils, had also converted some of the northern East Pasture to salt marsh. The ranchers attempted to stave off this conversion and maintain pasture by irrigating many of the pastures during the summer through either spray or flood irrigation. The East Pasture was leveed off from both Lagunitas and Tomasini Creeks, but as with the West Pasture, the East Pasture did receive some flood flows – and sediment – from Lagunitas and Tomasini Creeks during extreme flood events. Two stations have been monitored in the East Pasture – one at the northern end of the pasture, and other at the southern end -- although the southern station was not installed until after restoration.

Tomales Bay – Undiked Marsh: The Undiked Marsh is a natural marsh directly north or downstream of the Giacomini Wetlands. The Giacomini Wetlands and Undiked Marsh were once an integrated deltaic wetland until the Giacominis leveed the southern portion in the 1940s. After that, the vegetated intertidal "wedge" continued to extend northwards into Tomales Bay, with progradation slowing with construction of dams by Marin Municipal Water District in the upper Lagunitas Creek watershed after the 1950s. There are two SET stations, one of which is located at the northernmost edge of the deltaic wetlands at their junction to the open water portions of Tomales Bay. The other station – the southern one -- is approximately 550 feet north of the Giacomini Wetlands.

Tomales Bay – Walker Creek Marsh: Walker Creek Marsh is very similar to the Giacomini Wetlands/Undiked Marsh in that it is a deltaic wetland formed at the junction of Walker Creek and Tomales Bay, but this wetland is much closer to the outlet of Tomales Bay to the Pacific Ocean. This wetland is also similar to Giacomini/Undiked Marsh in age such that most of the active marsh development occurred in the early 1900s due to excessive sedimentation caused by upstream land disturbances in the watershed. Two SET stations have been monitored at this marsh: one at the eastern end, which is located further upstream along Walker Creek, and one at the western end, which is closer to the outlet to Tomales Bay.

Limantour Estero – Limantour Marsh: Limantour Marsh occurs on the outer Point Reyes coast in a sheltered estero where Muddy Hollow Creek flows into Limantour Estero. It is a relatively young marsh in that, prior to construction of the Muddy Hollow dam in the 1960s, much of the marsh was actually intertidal mudflat and subtidal areas. With construction of the dam, a vegetated intertidal marsh quickly developed outboard of the dam despite the reduction in sediment supply, probably due to the reduced streamflow velocities during both average and flood flow periods. Two SET stations have been monitored at this marsh: one at the eastern or upstream end of the marsh where Muddy Hollow Creek flows into the estuary and one at the western or more oceanward end of the marsh.

Results

Pre-Restoration: Elevation increased in the Giacomini West Pasture North even prior to levee breaching (~3 mm). Changes in elevation of Giacomini East Pasture North were actually negative in the first year of monitoring (-8.6 mm), but did increase the second year (8.1 mm): monitoring efforts during Year 2 occurred only one month after levees were removed, so they are not considered to reflect restored conditions. Feldspar markers were not installed until later, so no sediment deposition data were available for the first year of monitoring, but in year 2, between 0.6 mm and 3.6 mm of sediment appeared to have been deposited in the East and West Pastures, respectively.

Interestingly, prior to restoration (2006-2008), negative changes in elevation were recorded at many of natural marshes either in Year 1 or Year 2 or both. During Year 2, five of the six natural marsh monitoring sites showed net elevation decreases, with one of those showing declines in both years (Walker Creek Marsh-East). Even though elevations decreased, sediment deposition actually increased, with rates ranging from 0.8 mm in 2008 at Undiked Marsh-South to 5.0 mm in 2007 at Limantour Marsh-East. Sediment deposition prior to restoration averaged around 1.5 mm in Tomales Bay marshes and 2.7 mm at Limantour Marsh.

Immediate Post-Restoration: After restoration, marshplain areas increased in elevation in both the East and West Pastures, despite the lack of flood flow-associated sediment input and the expected compaction of soils from massive vegetation and root die-off in at least the East Pasture. In both pastures, elevation gains between 2008 and 2010 exceeded sediment deposition rates measured through use of feldspar markers, with absolute or total elevation gains during that period ranging from 6.8 mm in the West Pasture to 12.6 mm in the East Pasture and sediment deposition rates for that same period generally ranging between 2.2 and 5.0 mm annually in the East and West Pastures.

Between 2008 and 1010, elevations at other Tomales Bay sites largely continued on their downward trend, with elevation at the northernmost Undiked Marsh site decreasing by potentially as much as 4.5 mm in 2009 relative to installation elevations. Conversely, elevation during this period generally increased at both Limantour Marsh sites, with the easternmost one gaining as much as 14.6 mm relative to 2006 elevations. Sediment deposition rates appeared similar to those recorded prior to 2008, averaging 1.5 mm at Tomales Bay reference wetlands and 2.2 mm at Limantour Marsh. Even Walker Creek Marsh-East, which dropped in elevation by as much as 3.3 mm by 2010, had net deposition within feldspar plots averaging 0.7 mm both years.

Subsequent Post-Restoration Years: In 2011, elevation gains in the East Pasture stalled somewhat, with elevations apparently decreasing 3.6 mm relative to 2010 elevations, although gains in the West Pasture continued (+4.5 mm). Overall, however, in 2011, elevations were still considerably higher in the East Pasture post-restoration than pre-restoration. In 2012 and 2013, elevations in the East Pasture increased again, with annual elevation gains ranging from 8.0 to 10.1 mm. Since 2006, the elevation gain at the East Pasture-North site totaled 35.2 mm or an average of 5.0 mm/year, while, since 2010, elevation gains at the newly installed East Pasture-South site totaled 17.5 mm or an average of 5.8 mm/year. Conversely, elevations in the West Pasture-North have actually decreased a little since 2011, with decreases ranging between 0.1 mm in 2013 and 0.4 mm in 2012. As with the East Pasture, however, elevation gains overall since 2006 have been positive (14.0 mm).

Interestingly, previous trends for other marshes in Tomales Bay reversed somewhat during the 2011-2013 period, with all of the other marshes increasing in elevation, except for Walker Creek Marsh-East. Absolute elevations in the Undiked Marsh-North were 15.2 mm higher than they were in 2006, while those in the Undiked Marsh-South and Walker Creek Marsh-West were approximately 7.4 – 8.0 mm higher than they were initially. Even Walker Creek Marsh-East, which was 1.5 mm below elevations recorded in 2006 in 2013, showed elevation gains of 0.8 mm in 2013 and 1.2 mm in 2011. The most dramatic increase in elevation in natural marshes occurred at Limantour Marsh East, with elevations climbing 28.5 mm since 2006.

In the last three years, annual sediment deposition rates have appeared slightly higher than in earlier years, averaging 3.2 mm at Limantour Marsh and 2.1 mm in Tomales Bay. The highest rates continued to be within the restored wetland, with annual rates ranging from 1.5 mm at the East Pasture-North station in 2011 to 13.2 mm at this same low-elevation, more-subsided station in 2013. This high deposition rate was not an anomaly: Deposition rates of 11.3 to 11.5 mm were also recorded at other East Pasture station/sampling years during this period. Rates at the higher-elevation West Pasture-North station were much lower, ranging from essentially 0 (2013) to 3.0 mm (2011). The highest deposition rates outside the restored wetland were consistently at Limantour Marsh-East, with rates estimated at 5.0 mm in 2013 and 5.3 mm in 2011.

In general, during the monitoring period, annual median sediment deposition rates did not differ significantly between the Project Area (2.9 mm) and natural marshes (1.8 mm; Mood Median, df=1, Chi-Square=1.97, P=0.16) or between monitoring years within all Study Areas (Mood Median, df=6, Chi-Square=3.54, P=0.74). However, a weakly significant difference existed between individual Study Areas (e.g., Giacomini, Walker Creek, etc.; Mood Median, df=3, Chi-Square=6.7, P=0.08). Between 2006 and 2013, median annual sediment deposition rates were highest in the Project Area (see above), followed by Limantour (2.5 mm), Undiked Marsh (2.0 mm), and Walker Creek Marsh (1.3 mm).

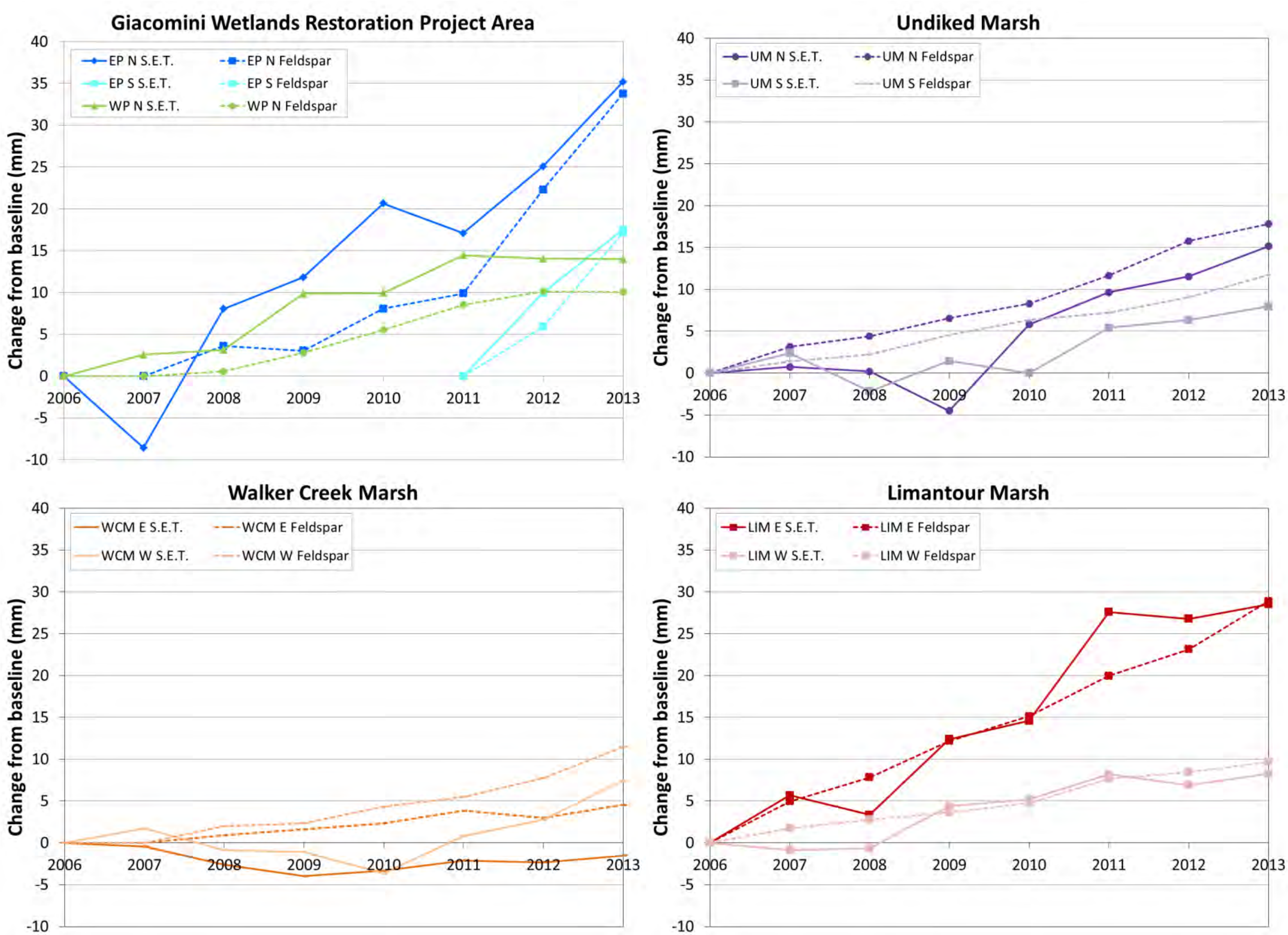

Figure A-2. Sedimentation Monitoring (SET and Feldspar) Change from Baseline, 2006-2013

Discussion

During the seven-year monitoring period, there was generally an increase in elevations within all the marshes monitored, except the eastern or upstream end of Walker Creek Marsh. Elevation increases ranged from 7.4 mm at Walker Creek Marsh-West to 35.2 mm at Giacomini East Pasture-North. During the first few years, many of the reference wetlands actually had net decreases in elevation, but this trend seemingly reversed starting in 2010/2011.

In some cases, however, elevation gains actually did not keep up with sediment deposition rates, suggesting that, overall, subsidence or lowering of topographic elevations might be occurring. This trend was apparent in all of the natural marshes monitored in Tomales Bay, including the Undiked Marsh north of Giacomini and Walker Creek Marsh. Figure A-2 shows a comparison of cumulative elevation changes relative to cumulative sediment deposition rates. This pattern of sediment accretion rates exceeding elevation gains is consistent with those observed in other tidal marsh systems (J. Callaway, USF, *pers. comm*.). Because all of these marshes are relatively young (<150 years old), they may still be evolving, and some reduction in marsh surface elevation could reflect consolidation of recently deposited sediments (J. Callaway, USF, *pers. comm*.).

Conversely, at Giacomini in Tomales Bay and at Limantour Marsh-East, absolute elevation gains typically exceeded – and sometimes greatly exceeded -- sediment deposition rates (Figure A-2). At Limantour Marsh-West, elevation gains typically approximated sediment deposition rates except in later years, when deposition rates were higher than elevation changes (Figure A-2).

In situations where elevation gains exceeded sediment deposition rates, most of the increase in elevations probably resulted from changes in subsurface processes. At Giacomini, reintroduction of tides potentially increased porewater volume in the soils and slowed down subsurface oxidation rates of organic matter. Much of the vegetation in the northern portion of the East Pasture quickly died after the levees were removed due to increases in extent and duration of inundation and soil and water salinities, leaving very sparsely vegetated mudflats with extensive thatch present. To some extent, some surface soil compaction or consolidation might have been expected due to the loss of root/rhizome volume in the soils, which, for many of the pasture grasses, was quite extensive. However, this loss of root volume appears to have been largely offset by other subsurface processes. In the West Pasture, plant communities have not changed as dramatically, and there have not been large die-offs of old vegetation that would at least temporarily reduce root/rhizome volume in the soils.

Most of the sediment deposition that has occurred in Lagunitas Creek and the Project Area since restoration appears to come from re-working of soils from the Project Area, which are now exposed and vulnerable after construction grading and decay of pasture vegetation. With three of the five winters since restoration being "dry" years, sediment inputs from the upper watershed have been minimal, particularly as there has been only one very brief overbank flooding event. As noted earlier, even when large storm events have occurred, most of the flood flow – and accompanying sediment – have been captured behind dams in the upper portions of the Lagunitas Creek watershed.

Despite the lack of overbank flooding, sediment was still deposited on Project Area marshplains during the last five years, as feldspar monitoring results attest. The dry winters and reduced flood flow volume have led to a net depositional environment both within the Project Area and other marshes, except where flow velocity is high enough to counteract this trend, such as in the downstream portions of the Tomasini Slough and the Lagunitas Creek side channel. With overbank flooding during storms being minimal, much of this sedimentation may derive from re-working of Project Area soils (KHE 2012a). Any existing tidal energy in these systems is being used primarily to transport sediment rather than scour channel banks and beds (KHE 2012a).

Limantour Marsh-East is close to the outlet for Muddy Hollow Creek, which is the main source of sediment supply for Limantour Marsh other than ocean- or estuarine-derived sediments. In 2008, a restoration project was conducted that removed the Muddy Hollow dam and pond that had historically trapped most of Muddy Hollow Creek's streamflow -- and sediment supply. Since the restoration, sediment supply to downstream marsh areas has undoubtedly increased due to mobilization of disturbed, open sediments on the former pond bottom. In addition, restoration may have affected subsurface processes relating to soil bulk density, porewater content, or plant/root establishment patterns in the downstream tidal marsh: all of these factors can influence marsh surface elevations.

Interestingly, since monitoring at Limantour was initiated, Pacific cordgrass has rapidly colonized the marshplain, replacing pickleweed (*Salicornia pacifica*), and the amount of ponded area or "marsh ponds" has increased. As Pacific cordgrass is considered a low-marsh species, which typically colonizes the edges of tidal creeks, its expansion into marshplains formerly dominated in large part by a mid-marsh species such as pickleweed potentially suggests that elevations of the marshplain have subsided, perhaps because of subsurface processes relating to the fire that swept through the marsh and surrounding areas in 1996. However, at least since 2006, the marshplain appears to be actively accreting sediment. It is possible that, if any subsidence has occurred, it pre-dated 2006, when monitoring was initiated.

As with Limantour, the proximity of the Undiked Marsh to the Giacomini restoration area may have also affected elevation and sediment deposition trends. In general, because this wetland is a delta, the southern portions of the Undiked Marsh are considerably higher in elevation that the northern portions adjacent to Tomales Bay. This may mean that the southern Undiked Marsh soils were drier and, therefore, had a higher bulk density, lower porewater volume, and were less likely to accumulate organic matter than soils in northern portions. With restoration of the Giacomini Wetlands, the very southern portions of the Undiked Marsh appear to have gotten wetter based on visual observations of prolonged surface ponding in this area (A. Ryan, L. Parsons, NPS, *pers. obs*.). Over time, marsh surface elevations may be affected by changes in subsurface processes resulting from upstream restoration, just as restoration is affecting the Project Area itself.

Conclusions

Based on our monitoring, sediment deposition rates in Tomales Bay marshes are seemingly lower than historic sediment deposition rates, which were estimated to average 5 mm/year (Rooney and Smith 1999). The one exception to this appears to be Giacomini, which has averaged 4.8 mm of sediment deposition per year after restoration was implemented. Sediment deposition rates at the various natural marshes in Tomales Bay and Limantour were lower than Giacomini, ranging from 1.3 mm to 2.7 mm. In comparison, in San Francisco Bay, current short-term accretion rates range between 3.1 and 5.9 mm/year in natural marshes, with higher rates at lower-elevation marshes (Callaway *et al.* 2012). Currently, this sediment deposition rate is enough to enable San Francisco Bay marshes to just keep pace with the rate of sea level rise (Callaway *et al.* 2012). As sea level rise rates would be expected to be pretty similar between these adjacent estuaries, the sedimentation rates documented in our study would suggest that many outer coast marshes are not keeping pace currently with sea level rise, and this problem will only be exacerbated by the expected increases in the rate of sea level rise in the future.

The long-term resilience of coastal marsh systems will be dependent to a large degree on continued sediment inputs from upstream portions of the watershed. Without these inputs, marshplains will sink below the steadily creeping rise in sea level and convert to non-vegetated types of wetlands such as intertidal mudflat or subtidal or open water habitat (San Francisco Bay Joint Venture 2008). Systems that are already sediment-starved to some degree such as the San Francisco-Sacramento Delta estuary – which has lost much of its sediment supply to upstream dams -- are in greater danger from sea level rise than those in systems with either natural or even unnaturally higher rates of downstream sediment delivery (Orr *et al.* 2003). Somewhat ironically, the potential for climate change to increase precipitation and run-off rates could actually help to counter the effects of sea level rise for some coastal ecosystems, particularly those where snowpack currently mediates the intensity and timing of run-off.

Without levees to restrict access of flood flows to marshplains, some portion of the sediment yield from the upper portion of the Lagunitas Creek watershed would be expected to deposit on the newly restored Giacomini Wetlands floodplains. As discussed earlier, based on estimates of sediment loads conveyed by lower Lagunitas Creek during higher streamflow events, the Giacomini Wetlands floodplains could trap or retain as much as 19% of the sediment loading in Lagunitas Creek during a 2-year flow event or slightly more than 9,500 tons/day. This sediment deposition would work to counter the effects of sea level rise in the Giacomini Wetlands, particularly for the southern portion of the system, where hydraulic modeling suggests that most of the conveyed sediment would drop out of suspension (KHE 2006). This continued sediment input would continue to build elevations in the southern portion of the Giacomini Wetlands, countering sea level rise pressures, at least in this area, but the northern portion would be dependent on sediment subsidies from smaller drainages such as Tomasini and Fish Hatchery Creeks. This could potentially lead to an exacerbation in the sharply deltaic or wedge-shaped topography of the wetlands: as noted earlier, southernmost elevations currently range between 10- 11 feet NAVD88, while northernmost elevations range only between 3- 5 feet NAVD88 (USGS 2003).

Another factor that may increase resilience of coastal marshes to sea level rise, particularly marshes in sediment-starved systems, is increase in surface elevation associated with organic, rather than mineral, matter. A study by Langley *et al.* (2009) found that the elevation of salt marshes increased in areas simply through an increase in root biomass, even when where the contribution of mineral sediment deposition was negligible. Not only is surface elevation increased through increases in root biomass, but also through accumulation of undecomposed plant matter or peat, and systems exposed to frequent sedimentation or flooding events are likely to develop more layers or strata of peat that contribute to increases in marsh surface elevation. In one study of 55 wetlands in the Gulf of Mexico and three (3) wetlands in Rhode Island, Turner and colleagues found that organic matter accumulation represented the “dominant influence” on vertical accretion in salt marshes (Turner *et al.* 2000), which supported earlier findings published by Callaway and colleagues demonstrated a strong statistical relationship between organic matter accumulation and vertical accretion rates in 5 Gulf of Mexico wetlands (Callaway *et al.* 1997).

The importance of all these factors and how they interact was highlighted in a recently published study by Schile *et al.* (2014). These researchers modeled the persistence of salt marshes in the San Francisco Bay watershed where marshes were either islands or still had adjacent undeveloped uplands under different sea level rise and sedimentation scenarios (Schile at al. 2014). Under higher rates of sea level rise and lower rates of sedimentation, mid/high marsh habitats only persisted in former upland areas, with salt marsh and islands completely converting to low marsh or intertidal mudflat (Schile *et al.* 2014). Under the same sediment concentrations, low salinity brackish marshes containing highly productive vegetation had slower elevation loss compared to more saline sites with lower productivity (Schile *et al.* 2014). This study illustrates “the importance of adjacent uplands for long-term marsh survival” (Schile *et al.* 2014).

For this very reason, the presence of higher elevation uplands on the perimeter of the restored Giacomini Wetlands may prove critical to the persistence of this marsh in the future. Indeed, given the steep topography along the outer coast of California, outer coastal marshes could be just as impacted by sea level rise as those in San Francisco Bay, even though, in some instances, adjacent uplands along the outer coast are less heavily developed than those in San Francisco Bay. Another key factor in the survival of the Giacomini Wetlands will be ensuring a steady source of sediment delivery to the marsh, which may be difficult given the issues with reservoir storage and flow/sediment releases from the upper part of the watershed. In the future, sediment “releases” may become as important to the Tomales Bay Watershed as flow “releases” are now for both salmonids and water supply.

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