

**Appendix A: Year Two of the Giacomini Wetland
Restoration Project: *Analysis of Changes in Water
Quality Conditions in the Project Area and Downstream***

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Introduction and Background Summary

The Point Reyes National Seashore Association (PRNSA) and National Park Service (Park Service) have implemented an approximately 550-acre wetland restoration project in the southern end of Tomales Bay in Marin County, California (Figure A-1). 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. Perhaps, one of the most important functions that wetlands can play -- particularly in Tomales Bay -- is water quality improvement. 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. During the last few decades, poor water quality in the Bay has forced oyster fisheries to close down several times and, in 1998, was associated with a virus outbreak.

As an integral component of the restoration project, PRNSA in collaboration with the Park Service is implementing a comprehensive long-term monitoring program to assess whether restoration is successful. The proposed 20-year monitoring program will include assessment of both the Project Area and nearby reference wetlands both prior to restoration and after restoration is implemented. As part of this program, from winter 2002 to fall 2006, monthly to quarterly systematic sampling of water quality field parameters, nutrients (nitrate, nitrites, total ammonia, total phosphorous, total dissolved phosphates or dissolved orthophosphates), chlorophyll a/phaeophytin, and pathogen indicators (total and fecal coliform) was conducted within the Giacomini Ranch, Olema Marsh, and selected reference sites.

To facilitate analysis of restoration progress, the Long-Term Monitoring Program relies on a modified BACI (“Before-After, Control-Impact”) sampling framework. The program divides the Study Area into the Project Area (PA) or Impact Area (Giacomini Ranch and Olema Marsh) and Reference (REF) or Control Areas (natural tidal marshes in Tomales Bay and adjacent watersheds). In addition, for some analyses, sampling locations on the upstream perimeter of the Project Area were evaluated separately as Upstream Areas (US) to more clearly differentiate the effect of the Project Area on internal water quality and downstream loading conditions. Within these Major Study Areas, subsampling units or sub-groups were also broken out that included the differently managed pastures or areas in the Giacomini Ranch (East and West Pastures and the leveed Tomasini Creek), Olema Marsh, and the individual reference wetlands (Undiked Marsh, Walker Creek Marsh, and Limantour Marsh). It should be noted that one of the Reference Areas (Limantour Marsh) was dropped temporarily in 2008 as a Reference Area, because upstream areas were dramatically altered through active restoration. After a brief hiatus, Limantour Marsh was reincorporated in 2010 as a Reference Area, although we will continue to evaluate its validity as a Reference Area because of potential changes caused by the upstream restoration.

From 2000 (date of sale of land to Park Service) to 2006, the Giacomini 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.

Monitoring was continued on a more limited scale following discontinuation of the full-scale dairy operation in 2006. In 2006, the Giacomini 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. 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. 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 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.

With final breaching of the levees and hydrologic reconnection to Lagunitas Creek and Tomales Bay in October 2008, full restoration of the Giacomini Ranch and, to a lesser extent, Olema Marsh, was initiated. Monitoring continued during the Full Restoration period, with seven (7) scheduled and storm sampling events conducted in Year 1 (2008-2009) and five scheduled/storm sampling events conducted in Year 2 (2009-2010). Year 2 sampling events were conducted in early late October 2009; November 2009; late January 2010; mid-April 2010; and late July/early August 2010. At least two of these sampling events actually coincided with sizeable storm events (January, April), and, in general, Water Year (WY) 2010 (October 2009 – September 2010) was a much wetter year than the previous WY 2009 (Figure 1). While levee removal 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. Timing and scale of monitoring efforts during Year 1 were constrained by loss of funding due to state budget issues, with full-scale monitoring only reinitiated once funding was re-secured.

This technical memorandum summarizes changes in water quality conditions within the Project Area during the second year after restoration for WY 2010. Water quality conditions in the Project Area during the first year after restoration and prior to restoration were summarized in two previous reports (Parsons 2009; Parsons 2010). 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 year 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, and expected issues as discussed above with temporary increases in turbidity and temporary decreases in dissolved oxygen and pH did not materialize. Some of this may have resulted from the fact that 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. 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).

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 project will be changes in concentrations and, even more importantly, loading between upstream and downstream sampling locations. As was expected, during the first year 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." While estimated average nitrate concentrations and median coliform concentrations and loading appeared to show

downstream reductions in the first year of Full Restoration, few of these differences were strong enough to be statistically significant, probably due to the large variance in this type of data. However, it is likely that watershed-scale benefits will take time to be realized due to the continuing evolution occurring within the Project Area, as pasture vegetation continues to die off and convert into more natural salt- and brackish marsh vegetation communities.

A complete description of the water quality sampling methodology is available in the pre-restoration monitoring report (Parsons 2009). In general, sampling methodology has remained consistent with pre-restoration techniques with a few exceptions. Some of the notable changes in sampling since 2006 involve more storm sampling; use of a different laboratory for scheduled nutrient sampling events resulting in a shift in some of the types of nutrients being analyzed; and changes in sampling locations when restoration eliminated some stations, and tidal channel creation created opportunities for new stations, particularly in the East Pasture. Whenever possible, original sampling stations were retained, with some simply renamed to reflect changed status after restoration. One other change is that one station (EUC1) was switched from being a Project Area (PA) to an Upstream (US) site, because waters in this area now derive entirely from downslope run-off and groundwater inflow and, therefore, more accurately reflect the quality of water flowing into the Project Area than Project Area conditions. As the change occurred after restoration, the Year 1 data has been reanalyzed to account for this change in status. Therefore, values for Year 1 in this report may differ slightly from those in the Year 1 report (Parsons 2010).

Starting in November 2007, analysis of nutrient samples was largely switched to a different university laboratory that offered lower MDLs and the ability to detect nutrients at very low concentrations. Because of this switch, analysis of total dissolved phosphates were replaced by dissolved orthophosphates, which are not synonymous, as total dissolved phosphates includes phosphates that are not orthophosphates or biologically reactive phosphates. In addition, total phosphorous was recently added to the analytes list, so there is no comparative pre-restoration data for this constituent: inclusion of TP was intended to determine how much of the phosphorous within the system is in particulate rather than dissolved form.

Most of the field parameter data fell within instrument detection limits. For these parameters, either Excel or a statistical package such as Minitab (State College, PA) or Systat (Chicago, IL) were used to statistically analyze data using either parametric or non-parametric techniques. For non-parametric procedures, the Mood's Median Test was often employed this year rather than Kruskal-Wallis, because, while the Mood tests is powerful in terms of detecting change, it is more robust towards outliers, and there are many valid outliers in this data set. Substitution can be employed if the number of non-detects or "censored" data is relatively low (<15 % of the data; Helsel 2006, *pers. comm. in* Parsons 2009.). However, when the number of non-detects exceeds 15 % of the data, more sophisticated analytical techniques should be used that take advantage of the information provided even if values fall below (or even above) MDL (Helsel 2005). Most of the nutrient and pathogen data showed varying proportions of non-detect data, with some of the most problematic in terms of high numbers of non-detect values being the total ammonia and total dissolved phosphates data. For parameters that had moderate to large number of values that fell either below or above the reporting limit, summary statistics were calculated using statistical methodologies commonly employed in other fields such as the medical and biotechnology industries that fit a distribution to observed values using Maximum Likelihood Estimates (MLE) or other parametric or non-parametric equivalents and then extrapolate a collection of values above and below the reporting limit for use in estimations (Helsel 2005).

Changes Following Restoration – Year Two: Results and Discussion

Changes in Hydrologic Conditions in Project Area

Water quality 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.

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 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 permanently flooded.

Based on data collected during continuous hydrologic monitoring by KHE, water levels at the very northern end of the Project Area in Lagunitas Creek (former North Levee) immediately showed compression in the maximum and minimum water levels during spring tides – that is, the low tides were not getting as low as they did previously during the lowest low tide conditions (KHE 2009a). Because channel width and density was not large enough currently to fully accommodate flows, waters were not fully draining on the 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 it by the marshplain (KHE 2009a). Prior to restoration, the morphology of gravel bars in Lagunitas and Fish Hatchery Creeks suggested that subtidal conditions after restoration would persist 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-type effect these 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. Based on hydraulic modeling, subtidal extent, particularly in the East Pasture, was much greater after the levees were breached than expected under fully evolved conditions. 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 2). The discrepancy between current 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). Interestingly, subtidal extent was actually lower 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 an approximately a 1- to at most 3- foot difference in tidal water elevation. These results suggest that drainage was being constrained by the larger volume of water that flows 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 may be further altered in the future by changes in Lagunitas Creek (and interior tidal channel) geometry. Post-project channel surveys in 2009 indicated the formation of ebb shoals in Lagunitas Creek at the mouth or downstream of the mouth of both of the newly constructed channels draining the East Pasture, with accretion totaling more than 1 to 2 feet (KHE 2009b). These deltaic-type shoals 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). Post-project surveys also indicated a uniform increase (1.0) in bed elevation of the mainstem Lagunitas Creek channel immediately upstream of the former cattle crossing near White House Pool in 2009 relative to elevations in 2003 (KHE 2009b). In contrast, channel elevations immediately upstream of the former North Levee area remained fairly comparable in 2009 to those measured in 2003 by the USGS, (KHE 2009b). In 2010, the shoal at the newly constructed Lagunitas Creek side channel had accreted by 0.3 to 0.5 feet relative to elevations in 2009, but there was no change in elevation in the shoal downstream of the former North Levee and Tomasini Slough outlet in Lagunitas Creek (KHE 2010b). There was also no change between 2009 and 2010 in channel elevations at the former cattle crossing bar near White House Pool or just upstream of the former North Levee, although the nearby gravel bar appeared to have elongated somewhat in 2010 (KHE 2010b).

Within the Project Area itself, most areas showed little change in channel geometry between pre- and post-restoration conditions, particularly in the West Pasture, although upstream areas of Tomasini Slough accreted by as much as 1.2 feet, while downstream Tomasini Slough areas just upstream and downstream of the outlet to Lagunitas Creek had potentially incised or decreased in elevation by as much as 1.7 to 1.8 feet since 2008 (KHE 2010b). Interestingly, marshplain areas appear to be gaining in elevation in both the East and West Pastures, despite the massive vegetation die-off in the East Pasture that would be expected to compact soils due to loss of root volume below the soil surface (Parsons and Ryan, *in prep.*). In both pastures, elevation gains actually exceeded sediment deposition rates measured through use of feldspar markers, with elevation gains between 2008 and 2010 ranging between 19 mm (East Pasture) and 13.5 mm (West Pasture) and sediment deposition rates ranging between 4 mm and 4.5 mm, respectively (Parsons and Ryan, *in prep.*). Most of these elevation gains, then, appear to result from changes in subsurface processes such that reintroduction of tides increased porewater volume in the soils and slowed down subsurface oxidation rates of organic matter (Parsons and Ryan, *in prep.*).

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 and decay of pasture vegetation. With 2008–2009 being a dry winter, sediment inputs from the upper watershed were probably minimal, particularly as there were no overbank flooding events. While the 2009/2010 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. During much of the year, upstream dams trap most of the peak flood flow and sediment, reducing flood volume and sediment loading to downstream areas. Despite this lack of overbank flooding, sedimentation monitoring has shown that sediment was still deposited on Project Area marshplains during the last two years (Parsons and Ryan, *in prep.*). While much of this sediment may derive from re-working of Project Area soils, deposition rates (4.0-4.9 mm) are similar between the northern portion of the East and West Pastures and the very northern portions of the Undiked Marsh near Tomales Bay, which is unlikely to be greatly affected by sediment re-working within the restored wetlands. Not only have dry winters and reduced flood flow volume affected sediment deposition rates, they also affect stream energy and the ability of the creek to erode newly deposited sediments in and at the mouth of tributaries. In general, this leads to a net depositional environment both within the Project Area and Lagunitas Creek, except where flow velocity is high enough to counteract this trend, such as the mouth of Tomasini Slough. 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.

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 increase in size to accommodate flood flows, and new tidal channels will develop, increasing exchange between the restored wetland and Lagunitas Creek and creating more of an 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.

Despite this, change does appear to be occurring. Within months of the levee breach, informal surveys of the newly restored wetlands showed that new tidal channels were developing. 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). However, hydrologic data from 2009 suggested that the marsh was draining slightly faster during outgoing or ebb flows (KHE 2010a). Water level logger data from 2010 is still under analysis. In addition, the extent of subtidal areas during an extreme low tide dropped dramatically in 2010, 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 2). This represents a 38% decrease in extent of permanent inundation during extreme low tides within two years. This situation demonstrates that ecosystem evolution following restoration is not a linear process, but can occur in distinct stages or phases that involves triggering or exceeding thresholds before the wetland moves into the next evolutionary stage or phase.

Changes in General Water Quality Conditions in Project Area

Salinity

The Project Area lies in the Estuarine Transition Zone, the dynamic interface between freshwater and saltwater influences. For this reason, salinity regimes and patterns are understandably dynamic both spatially and temporally. Much of the freshwater inflow comes from the copious

amount of small freshwater drainages and emergent groundwater flow from the Point Reyes Mesa and Inverness Ridge, as well as the larger creeks such as Lagunitas Creek, Olema Creek, Bear Valley Creek, Fish Hatchery Creek, and Tomasini Creek.

Because of these freshwater influences, prior to restoration, salinities and temperatures differed significantly between the Project Area and other Study Areas (Parsons 2009). Salinity averaged 6.9 ppt in the Project Area, 22.0 ppt in Reference Areas, and 0.6 ppt in Upstream Areas, which receive less or no tidal influence and have strong perennial or seasonal freshwater influences (Kruskal-Wallis, $n=1261$, $df=2$, $H=472.6$, $P < 0.001$; *ibid*). Based on statistical analyses, a significant change in average salinities occurred within the Project Area after restoration. Average salinities climbed from 6.9 (± 0.3 ; SE) ppt Pre-Restoration to 8.5 (± 0.9 ; SE) ppt during Passive Restoration to 13.5 (± 0.8 ; SE) ppt after restoration (ANOVA, $n=1,188$, $df=2$, $F=29.94$, $P<0.0001$, Adj. $R^2=0.047$). Median salinities jumped during the first year after restoration (16.6), but dropped during Year 2 (4.3 ppt; Mood Median Test, $n=1,188$, Chi-Square = 52.4, $P<0.0001$). Average salinities were slightly higher than medians, suggesting that either some sampling events (e.g., summer) or areas influenced salinity conditions. Salinities averaged 17.1 (± 1.2 ; SE) ppt in Year 1, dropping to 10.6 (± 1.0 ; SE) in Year 2. Figure 3 shows changes in salinities for each of the sub-sampling units or areas within the Project Area.

Water quality monitoring conducted by Kamman Hydrology & Engineering pre- and post-levee removal showed that salinities increased not only within the former pastures, but within Lagunitas Creek, as well. Average salinity in Lagunitas Creek increased immediately after final removal of the East Pasture levees on October 25, although the West Pasture levee removal and Olema Marsh restoration components completed two weeks earlier appeared to have no immediate discernable effect on Lagunitas Creek salinity (KHE 2009a). In general, average salinity, if not maximum salinity, increased along the entire portion of Lagunitas Creek within and upstream of the Project Area, although salinity levels and the absolute magnitude of the change decreased with distance upstream from the downstream boundary of the Project Area (KHE 2009a). At this furthest downstream location (former North Levee), the maximum salinity remained the same immediately post-restoration, but average salinity increased, because there was an upward shift in the lower limit of salinity, with the range increasing from between 10 and 32 psu immediately pre-restoration to between 18 and 34 psu immediately after restoration (KHE 2009a). While the range of salinity variations in 2009 remained comparable to immediate post-levee breach conditions in 2008, the amplitude in salinities was more compressed in 2009, ranging only from approximately 22 to 35 ppt (KHE 2010a).

Salinities increased in the Project Area primarily in response to the reintroduction of tidal action. However, results definitely reflect the strong influence of intraannual and interannual climatic patterns. In 2008-2009, the dry weather and unusual precipitation patterns led to a much higher average salinity in the Project Area (17.1 ppt) than in 2009-2010 (10.6 ppt), when rainfall was higher and more evenly distributed, with at least 50% of the sampling events last year occurring during a rainfall event. In comparison, average salinities prior to restoration were only slightly lower (6.9 ppt).

The strong influence of dry weather during the first year of restoration can be seen in the change in salinities within Reference Areas. For the four-year period prior to restoration, the median salinity within Reference Areas was 25.5 ppt. Median salinities in Reference Areas actually decreased between Pre-Restoration and Passive Restoration (median=19.3 ppt) sampling periods, but, in 2008-2009, they climbed 20% to 30.7 ppt (Mood Median Test, $n=835$, $df=3$, Chi Square=25.6, $P<0.0001$). In Year 2, salinities in Reference Areas dropped back down to a median of 21.0 ppt. Average salinities in Reference Areas during these sampling periods were 20.4 \pm 1.2 (SE) ppt (Passive Restoration); 29.3 \pm 1.0 (SE) ppt (Year 1-Full Restoration); and 19.1 \pm 1.5 (SE) ppt (Year 2-Full Restoration).

While one of the Reference Areas is adjacent to the Project Area and could have been affected by changes in tidal prism and salinity dynamics in the southern portion of the watershed, the other

Reference Area sampled following Restoration is at the opposite end of the estuary near the estuary's mouth and was unlikely to have been substantially affected by restoration activities. As noted earlier, one of the Reference Areas (Limantour Marsh) was dropped between Pre and Passive Restoration Periods and Full Restoration, which may have affected post-restoration summary statistics. However, salinities within this sampling site prior to restoration (average= 24.0 ± 11.6 (SD) ppt) were roughly equivalent to those of Walker Creek Marsh (average= 24.4 ± 11.8 (SD) ppt) and higher than those in the Undiked Marsh directly north of the Project Area (average= 17.4 ± 10.0 (SD) ppt; Parsons 2009).

Temperature

The influence of freshwater was also evident in water temperatures prior to restoration (Parsons 2009). Before levees were breached, temperatures were lower in the Project Area (median = 15.1 degrees Centigrade) than in Reference Areas (median=17.3 degrees Centigrade), although not lower than those in Upstream Areas (median=12.7 degrees Centigrade; Kruskal-Wallis, $n=1234$, $df=2$, $H=50.04$, $p<0.001$). While diking of the Giacomini Ranch and the culvert-levee road system at Olema Marsh resulted in longer residency time for waters – and more time for sunlight to drive up water temperature – the substantial freshwater influences from both creek and emergent groundwater flow appeared to moderate the effect of these management impacts on water temperature.

With removal of the levees and reconnection of Project Area waters to Lagunitas and Tomasini Creeks, median temperatures increased slightly (2%), albeit significantly, within the Project Area from 15.1 degrees Centigrade Pre-Restoration to 15.4 degrees Centigrade during Year 1, only to drop 16% to 12.9 degrees Centigrade during Year 2 (Mood Median Test, $n=1,152$, Chi-Square =20.1, $P<0.0001$; Figure 4). Temperatures actually also dropped to some degree during Passive Restoration, but not as dramatically (median=14.2 degrees Centigrade). Average salinities were slightly higher than median ones in Year 2 (average= 13.6 ± 0.3 (SE) degrees Centigrade), but lower than averages for previous years: 15.9 ± 0.2 (SE) degrees Centigrade Pre-Restoration; 14.2 ± 0.5 (SE) degrees Centigrade Passive Restoration; and 16.6 ± 0.5 (SE) degrees Centigrade Year 1- Full Restoration.

Comparatively, mean temperatures in Reference Areas actually decreased from 17.2 ± 0.3 (SE) degrees Centigrade Pre-Restoration to 16.1 ± 0.7 (SE) degrees Centigrade during Year 1 and 14.4 ± 0.5 (SE) degrees Centigrade during Year 2 (ANOVA, $n=518$, $df=3$, $F=6.40$, $P<0.0001$, Adj $R^2=0.03$; log-transformed). The drop in temperatures during Year 1 is contrary to what was observed in the Project Area. There was no sampling in Limantour Marsh during Year 1, and, in general, average and median water temperatures in Limantour Marsh prior to restoration (average=17.5 degrees Centigrade) were slightly higher than those of the Walker Creek Marsh (average=16.9 degrees Centigrade) and somewhat equivalent to the Undiked Marsh (average=17.3 degrees Centigrade), although differences were not statistically significant (ANOVA, $n=309$, $df=2$, $F=0.29$, $p=0.751$). Loss of this sampling site could have driven down average temperatures for Reference Areas during Year 1, although Limantour Marsh was reincorporated into the monitoring program in Year 2.

Prior to restoration, Reference Areas exceeded the lethal limit for salmonids of 25 degrees Centigrade (Moyle 2002) approximately 6.7 of the time, and another 17.8% exceeded 22 degrees Centigrade, the suboptimal limit for salmonids (Moyle 2002, Parsons 2009). Comparatively, in the Project Area, before the levees were breached, temperatures exceeded the lethal limit during only 5% of the sampling periods and exceeded the suboptimal limit during approximately 15 % of the sampling periods. In the first year of Full Restoration, temperature exceedance levels in the Project Area actually dropped despite the low rainfall, with temperatures exceeding 25 degrees Centigrade reduced slightly to 4.4% of the sampling events, although the number of exceedances of the suboptimal limit remained the same (15.0%). This compared to a slight drop for Reference Areas in Year 1 to exceeding the lethal limit only 3.3 % of the sampling periods and the suboptimal limit of 22 degrees only 14.9% of the events. Due to the generally lower temperatures

in the wetter 2009/2010 season, exceedances during sampling events for lethal and suboptimal limits were reduced considerably to below 1% for both the Project and Reference Areas.

One of the objectives of the restoration project involves the marsh eventually evolving towards conditions similar to those present in reference natural marshes, specifically for those parameters where, based on site conditions, convergent evolution would be expected. Due to the very different climatic conditions between Year 1 and Year 2 after restoration, average temperature conditions within the Project Area fluctuated sharply, driven by strong variation in the amount of inflow of cold freshwater from the upper watersheds and groundwater. Temperatures might be expected to range a little lower in the Project Area than reference marshes due to strong influence of Lagunitas Creek, the largest creek in the Tomales Bay watershed. However, despite this, average salinities after restoration during Years 1 and 2 did not differ significantly between the Project Area (average = 15.0 ± 0.3 (SE) degrees Centigrade) and Reference Areas (mean= 15.0 ± 0.4 (SE) degrees Centigrade; t-test, n=347, df=247, t=-0.02, P=0.983).

While these results would suggest that the Project Area is converging with conditions in the Reference Areas, the lower salinities in the Project Area during Year 1, which was a dry year, suggest that the Project Area will probably never totally converge with that of the Reference Areas due to its geographic position within the freshwater-saltwater interface zone, although both spatial and temporal pattern of salinities and temperatures will continue to change as conditions evolve after restoration.

pH

Another variable that shows the influence of freshwater is pH. While pH prior to restoration might have been expected to be lower in the freshwater-dominated Project Area compared to the more marine-influenced Reference Areas – pH of ocean waters is typically somewhat alkaline – Pre-Restoration pH did not vary significantly between the Project Area and the other Study Areas prior to restoration (range=7.60 to 7.63 in Upstream Areas; Kruskal-Wallis, n= 1218, df=2, H=5.09, P=0.08; Parsons 2009). Most creeks feeding into the Project Area actually had fairly high 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 (ibid). Muted tidal influence in the West Pasture and Tomasini Creek and high primary productivity during some sampling events also boosted pH (ibid). However, lower pH waters (~5.9 – 6.6) occurred only in areas where more extensive influence from groundwater occurs or, less frequently, where there was organic matter decomposition actively occurring (ibid).

Based on statistical analyses, the geometric mean or median pH in the Project Area has actually decreased consistently since dairy operation, from 7.60 during Pre-Restoration to 7.30 during Passive Restoration, and to 7.25 during Year 1 and 7.14 during Year 2 of Full Restoration despite the increase in tidal influence and the expected decrease in groundwater inflow during at least the 2008/2009 dry winter (Mood Median Test, n=1144, df=3, Chi-Square=93.0, P<0.0001; Figure 5). The increase in exchange and decrease in residence time may have led to decreases in pH associated with phytoplankton blooms. However, breakdown of organic matter from 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, flushing of sulfuric and iron-associated acids from oxidation of reduced sulfur and iron complexes 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 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.

Interestingly, a similar seemingly slight, but significant, decrease was also observed in Reference Areas, although means for pH between Project Area and Reference Areas in Years 1 and 2 were not equivalent from a statistical standpoint (t-test, $n=358$, $df=244$, $t\text{-value}=-2.86$, $P=0.005$; Figure 5). In Reference Areas, pH dropped from a median of 7.62 during the Pre-Restoration and Passive Restoration sampling periods to 7.44 during Year 1 and 7.35 during Year 2. There was a significant interaction factor between year and marsh (ANOVA, $n=513$, $df=3$, $F=2.41$, $P=0.027$, $\text{Adj } R^2=0.14$), with the Pre-Restoration period significantly different from Year 2 (Tukey, $T>-4.29$ and $P=0.0001$). As early sampling efforts only included pH of surface waters, medians from later sampling periods – primarily post-restoration – were re-run using just surface water values to ensure that more recent medians were not being potentially affected by lower pH values in bottom waters. However, the medians for just surface values were identical to those of the averaged surface and bottom waters. While the overall median between Pre-Restoration and Passive Restoration periods remained consistent at 7.62, slight variation did occur between years, ranging from 7.53 in WY 2005/2006 to 7.8 in WY 2001/2002, although the latter did not include the Undiked Marsh. Starting in WY 2007/2008, prior to restoration, the overall median dipped down to 7.50 and has dropped slightly every year since.

The median pH appeared to drop more in the Undiked Marsh, which is furthest from the mouth of Tomales Bay, and Limantour Marsh, which is in another watershed, than in Walker Creek Marsh, which is located close to the mouth of Tomales Bay. In the statistical analysis, the Undiked Marsh differed significantly from Walker Creek Marsh and Limantour Marsh, but Walker Creek Marsh did not differ significantly from Limantour Marsh (Tukey, all $T>-2.44$, all $P<0.039$). During Pre-Restoration and Year 2 (2009/2010) periods, the median pH averaged 7.48 and 7.15, respectively, in the Undiked Marsh; 7.80 and 7.34, respectively, in Limantour Marsh, and 7.70 and 7.51, respectively, for Walker Creek Marsh. As with the overall median, the range in median pHs in Limantour Marsh tended towards the higher end, ranging between 7.68 and 7.93 until WY 2007/2008, when it dropped to 7.44. Limantour Marsh was only sampled once in WY 2008/2009. For the Undiked Marsh, median pH ranged between 7.34 and 7.69 until WY 2007/2008, when it again dropped to 7.34, however, during the pre-2008 period, it was typically higher than 7.34 and never showed a consistent downward trend, as it has done since WY 2007/2008. The drop in median pH in Walker Creek Marsh only occurred this sampling year, with pH consistently averaging 7.71 between fall 2007 and fall 2009, and the geometric mean for pH actually did not appear to drop at all from Pre-Restoration period (7.61) to Year 2 (7.58). Therefore, for Walker Creek Marsh at least, this may represent more year-to-year variation than an indication of some declining trend in pH.

The consistent decreases in median pH in the Undiked Marsh and Limantour Marsh since WY 2007/2008 could potentially result from the fact that, in these systems, upstream areas have been restored, and restoration is affecting the pH of downstream marshes, as well as that of the Project Areas. However, there are indications that pH is declining system-wide, as well. University of California, Davis, (UCDavis) researcher Ann Russell and her colleagues have 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 LMER and recently collected data, however, pH did appear to decline in both the outer and inner Bay by as much as 0.25 pH units (A. Russell, UCDavis, *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 throw this into question. While Russell and colleagues did observe larger decreases in pH in the Outer Bay relative to the Inner Bay relative to the LMER data (Russell et al. 2010), in our results, pH decline appears to have been greatest furthest from the mouth of the estuary,

which suggests that, for our study, changes are not directly related to inflow of lower pH waters from the ocean. Russell believes that the change observed in pH for Tomales Bay was too large to be attributable to dissolution of CO² 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) appeared to have climbed relative to the LMER program sampling period (Russell et al. 2010). Russell and colleagues are planning to construct mass balance models for DIC and DOC to determine relative inputs of carbon to Tomales Bay from both marine and terrestrial sources. Russell noted that some of the discrepancy for her results could come from different methodological approaches than employed by LMER scientists. While we are using a different and more advanced instrument to measure pH than we did initially, we continue to calibrate the instrument in the same way, so it is unlikely that our changes result from changes in methodology. However, even if these pH changes are 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 in oceans even without the influence of climate change.

Dissolved Oxygen

While diking did not appear to negatively impact salinities, temperature, or pH of waters within the unrestored Project Area, diking and other agricultural land management practices did appear to affect oxygen concentrations within drainage ditch and creek waters, often causing hypoxic or even anoxic conditions (Parsons 2009). Most of the extremely low oxygen concentrations occurred in the East Pasture drainage ditches, where frequent ditching increased oxygen demand by filling ditch waters with loose vegetation material that was consumed by oxygen-dependent bacteria (*ibid*). This management practice, coupled with the relatively infrequent exchange or subsidy of ditch waters except during the winter or when irrigation was performed, typically kept oxygen levels below 5 mg/L and often below 2 mg/L.

Prior to restoration, oxygen levels in the East Pasture averaged 4.98 ± 3.86 (SD) mg/L, with median levels actually slightly lower (4.56 mg/L; Parsons 2009). These same factors – copious amount of organic matter and infrequent exchange between the impounded marsh and Lagunitas Creek -- also contributed to consistently low levels of oxygen in Olema Marsh, although levels were not as low as the East Pasture (mean = 5.83 ± 1.21 (SD) mg/L; *ibid*). Median oxygen concentrations in other Project Area subsampling areas or sub-groups – excluding upstream sampling sites -- ranged from 8.64 mg/L in Lagunitas Creek to 7.91 mg/L for Tomasini Creek, with the less heavily managed West Pasture having slightly higher levels (8.50 mg/L; *ibid*).

Following restoration, median oxygen levels in the Project Area increased from 7.58 mg/L during Pre-Restoration and 8.39 mg/L during Passive Restoration to 8.63 mg/L in Year 1 and 8.13 mg/L in Year 2 after Full Restoration (Mood Median Test, $n=1165$, $df=3$, Chi-Square=12.66, $P=0.005$). Average salinities were generally slightly lower, ranging from 7.30 ± 0.13 (SE) mg/L during Pre-Restoration and 8.55 ± 0.35 (SE) mg/L during Passive Restoration to 8.73 ± 0.39 (SE) mg/L in Year 1 and 7.79 ± 0.24 (SE) mg/L in Year 2 after Full Restoration. Perhaps because Year 2 was wetter and 50% of the sampling events during a storm, oxygen production in waters associated with biological activity may have been reduced relative to previous years.

Oxygen concentrations in the East Pasture jumped from a median of 4.56 mg/L pre-restoration to 9.85 mg/L during Year 1 and 8.83 mg/L during Year 2 post restoration, an increase of more than 90% (Mood Median Test, $n=395$, $df=3$, Chi-Square=66.22, $P<0.0001$; Figure 6). In terms of average salinity, oxygen concentrations in the East Pasture jumped from an average of 4.98 ± 3.86 (SD) mg/L pre-restoration to 9.89 ± 2.94 (SD) mg/L during Year 1 and 8.27 ± 2.53 (SD) mg/L during Year 2 post restoration. While D.O. also appeared to increase almost 20% in Olema Marsh between pre- and post-restoration conditions, increasing from 5.83 ± 1.21 (SD) mg/L to 6.86 ± 1.88 (SD) mg/L in the first two years after restoration, this change was not statistically

significant, perhaps because of the relatively low power in the first year of post-restoration analysis (ANOVA, $n=27$, $df=2$, $F=1.52$, $P=0.238$; Figure 6).

In the Project Area, oxygen concentrations prior to restoration fell below the Basin Plan standard of 5 mg/L during 25% of the sampling periods, with most of these exceedances occurring in the East Pasture (Parsons 2009). In contrast, only approximately 8% of the oxygen concentrations recorded in reference marshes fell below 5 mg/L, a difference of 68% (ibid). In the first year of Full Restoration, the number of Basin Plan standard exceedances in the Project Area dropped 43% from 25% to 14.2% and compared reasonably well with the number of incidences recorded in Reference Areas during the same period (12.8%). Incidences of hypoxia (< 2 mg/L) and anoxia (<0.5 mg/L) in the Project Area totaled 2.7% and 0%, respectively, in the Project Area in the first year after Full Restoration, compared to 12.2% and 5.4%, respectively, prior to restoration. This was a 78% decrease in hypoxia incidences within the Project Area. Exceedances of Basin Plan standards in the Project Area dropped even lower in Year 2, with only 4% falling below 5 mg/L; 1.6% falling below hypoxic levels; and 0% falling below anoxic levels. Ironically, the number of Basin Plan exceedances appeared higher in 2009/2010 in Reference Areas, with 21% of the concentrations recorded falling below 5 mg/L, but no values reached hypoxic or anoxic levels.

While differences in Year 2 exceedance rates between the Project Area and Reference Areas would suggest that dissolved oxygen levels in the Project Area have not converged towards those in natural marshes, medians between these two Study Areas in Year 2 were almost exactly equivalent: 8.13 mg/L in the Project Area and 8.14 mg/L in Reference Areas (Mood Median Test, $n=200$, $\text{Chi Square}<0.001$, $P>0.99$). In addition, no significant differences existed between the Project and Reference Areas for Years 1 and 2 combined, with median oxygen levels being 8.35 mg/L for the Project Area and 8.28 mg/L for natural marshes (Mood Median Test, $n=347$, $df=1$, $\text{Chi-Square}=0.11$, $P=0.741$).

With restoration, oxygen concentrations might have been expected to decrease – or only increase somewhat 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 has not been evident in Project Area oxygen concentrations, and, in fact, oxygen levels in Year 2 were almost exactly equivalent to those in Reference Areas.

Turbidity

Prior to restoration, turbidity levels appeared to differ at least slightly between the Project Area (median=10.7 NTU) and Reference Areas (median=12.2 NTU), with Upstream Areas having the lowest levels (median=5.7 NTU; Kruskal-Wallis, $n=1,086$, $df=2$, $H=43.0$, $p<0.0001$; Parsons 2009). This same pattern was apparent with mean turbidity levels, with values estimated at 22.7 ± 2.3 (SE) NTU for the Project Area, 19.9 ± 1.5 (SE) NTU for Reference Areas and 13.4 ± 1.8 (SE) NTU for Upstream Areas (ibid). Based on this, it would appear that turbidity levels were similar between the Project Area and Reference Areas, but much lower in the fluvially dominated Upstream Area portions of the system. Before levee removal, differences also existed within the Project Area itself. Turbidity levels were higher in the heavily managed East Pasture (median=13.5 NTU) than in the other Project Area sub-groups, which ranged from a median of 8.0 NTU in the West Pasture to 11.3 NTU in Olema Marsh (Kruskal-Wallis, $n=658$, $df=4$, $H=24.0$, $p<0.001$; ibid). The disparity between sub-sampling areas was even more apparent with means, with turbidity averaging 36.6 ± 97.0 (SD) NTU in the East Pasture and 13.3 ± 18.25 (SD) NTU in the more lightly managed West Pasture (ibid). These numbers do not necessarily correspond with those discussed earlier in this section, because they exclude upstream sampling sites.

The highest measured turbidity Pre-Restoration occurred at the downstream sampling station near the Giacomini Ranch North Levee in June 2003 with a value of 266 NTU (Parsons 2009). In general, before the levees were removed, turbidity fell below 50 NTU in Lagunitas and Fish Hatchery Creeks and 40 NTU in Tomasini Creek (ibid). Turbidity did show a somewhat unexpected temporal trend, with the highest values in spring, summer, or early fall: turbidity is typically expected to be highest during the winter when sediment is being actively moved by creeks (ibid). The production of suspended particles during these periods may have been due to events such as upstream dam releases, biological activity, cattle activity, tidal action, and other activities within streams, ditches, and other water bodies.

Turbidity would be expected to increase, at least temporarily, following restoration due to the resuspension of sediment disturbed by excavation and other construction activities, die-off of pasture vegetation, and evolution of the marsh surface in response to tides and stormwater flows. In addition, release of decomposing organic matter into overlying waters would decrease clarity. As noted above under Hydrology, sediment efflux does appear to be occurring, based on the formation of ebb shoals at the confluence of newly constructed primary tidal channels in Lagunitas Creek (KHE 2009). Interestingly, however, turbidity levels in the Project Area showed no significant differences between pre- and post-restoration during Year 1 (ANOVA, $n=849$, $df=2$, $F=1.2$, $P=0.30$; Figure 7). Median turbidity levels were estimated at 10.7 NTU in the first year of Full Restoration, compared to 10.7 NTU during Pre-Restoration and 10.5 NTU during Passive Restoration.

However, in Year 2 after restoration, differences did exist between pre and post-restoration, with median turbidity levels almost doubling from 10.7 NTU to 22.2 NTU during Year 2 (Mood Median Test, $n=970$, $df=3$, $Chi-Square=35.70$, $P<0.0001$; Figure 7). Means also seemingly jumped during Year 2, averaging 60.8 ± 11.2 (SE) NTU during Year 2 relative to 22.7 ± 2.3 (SE) NTU during Pre-Restoration; 40.0 ± 13.7 (SE) NTU during Passive Restoration; and 15.7 ± 1.6 (SE) NTU during Year 1. While median turbidity levels in the Project Area did not differ significantly from Reference Areas in Year 1 (median=10.1 NTU; Mood Median Test, $n=185$, $df=2$, $Chi-Square=0.20$, $P=0.906$), they did differ significantly in Year 2 (Reference Area median=13.3 NTU; $df=2$, $Chi-Square=11.32$, $P=0.003$).

The very disparate trends in turbidity levels between Years 1 and 2 following restoration may be largely due to very different climatic conditions between the two years. In WY 2009/2009, conditions were relatively dry due to low rainfall and low-energy storm events, with no overbank flooding occurring that year. With higher rainfall, scour of the new channels and flooding of the still evolving marshplain would at least temporarily increase resuspension of sediment into overlying waters. Because rainfall was so low in Year 1, most of the "re-working" in the Project Area marsh came solely from tides, although they, in conjunction with vegetation die-off, would have been expected to increase turbidity within Project Area waters. As noted earlier, shoaling at creek mouths show that re-working of the landscape was taking place, even without the influence of storm events. 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.

Nitrates Predominant Nutrient Source Particularly in Ranch Prior to Restoration, but Levels Already Decreasing After Restoration

Nitrates

The relatively well oxygenated conditions present in most of the Study Areas -- except the East Pasture prior to restoration -- may contribute to the dominance of nitrates as the primary source of nutrients in the Study Areas (Parsons 2009). In contrast to ammonia and phosphates, nitrates have only very infrequently fallen below detection limits, even at relatively high limits used by commercial laboratories. Results from the LMER/BRIE study conducted a decade earlier -- which were, at least for Bay samples, generally much lower in magnitude than our pre-restoration results -- also showed nitrates as being the predominant source of nutrients (ibid). In our study, average nitrate concentrations did differ prior to restoration between Major Study Area groups, although median concentrations within the Project Area (0.83 mg/L) were actually not considered significantly different from those in the Reference Areas (0.70 mg/L; ibid).

Prior to restoration, the Project Area mean was substantially influenced by consistently high values in the more heavily managed East Pasture, which supported two active dairy herds, as well as being more actively managed in terms of irrigation, manure spreading, haying, land leveling, and other actions. Within the Project Area (excluding upstream sampling sites), estimated nitrate concentrations averaged 7.25 ± 1.83 (SE) mg/L (NO₃⁻) for the East Pasture and then dropped to below 1.10 mg/L for the other sub-groups (Parsons 2009). While nitrate concentrations were lower in less heavily managed portions of the Project Area, these areas were still subject to nitrate inputs from passive agricultural management of the West Pasture (e.g., grazing of dry or less active dairy herds); dairy use of Lagunitas Creek both inside and directly upstream of the Project Area; loading from upstream portions of Lagunitas, Tomasini, and Fish Hatchery Creeks; non-point source run-off and stormwater flow from the town of Point Reyes Station; and potential influence of leaking septic systems into groundwater that flows along the perimeter of the Giacomini Ranch and Olema Marsh (ibid).

The similarity in nitrate concentrations between the Project Area and Reference Areas and even among the different Reference Area units -- all of which occur in different watersheds or subwatersheds -- suggests that nitrogen and other nutrients are strongly controlled by internal, as well as external, factors (Parsons 2009). Indeed, these factors at times appear to override the differences in concentrations and loading that would be expected from the three Reference Area units given the very substantial difference in the degree and type of agricultural and residential development in the respective subwatersheds. While concentrations of nitrates were highest in winter and fall sampling events in the Project Area, there were occasionally spikes or pulses in spring or summer that were unrelated to increases in streamflow with storm events or run-off (ibid). Some of the pulses in nitrates during non-flood periods may result from inorganic nutrients being regenerated "internally" from breakdown of organic matter within marshes (Chambers et al. 1994b; ibid).

Immediately following restoration, a sharp pulse in nitrates did occur. In November 2008, only a few weeks after the levee was breached, estimated nitrate concentrations averaged 3.44 ± 1.59 (SE) mg/L, with median concentrations of 1.60 mg/L, however, by January 2009, estimated concentrations had dropped to an average of 0.18 ± 0.08 (SE) mg/L and median of 0.13 mg/L, which were seemingly higher, but not significantly so from August 2009 (est. average= 0.06 ± 0.04 (SE) mg/L) and May 2009 (est. average= 0.02 ± 0.01 (SE) mg/L) events. Estimated nitrate concentrations showed a statistically significant relationship with sampling date in WY2008/2009, with January, May, and August 2009 sampling results differing significantly from November 2008, and the two February 2009 storm sampling events (MLE, n=43, df=5, Chi-Square=20.0, p<0.0001). So, following the early transitional period after levee breaching, the only recorded surge in nitrates occurred during the two February 2009 storm sampling events, where estimated nitrates climbed to average levels between 1.63 and 1.93 mg/L and median levels between 1.6

and 2.0 mg/L during both events due to strong pulses at certain Project Area sampling sites. It should be noted that average levels recorded during non-storm events between January 2009 and August 2009 in the Project Area were roughly half that of Reference Areas.

Despite these episodic pulses, estimated mean nitrate concentrations did appear to actually decrease from 3.22 ± 0.83 (SE) mg/L pre-restoration to 0.94 ± 0.35 (SE) mg/L during Year 1 of Full Restoration, which also represented a drop from levels during Passive Restoration (4.52 ± 2.35 (SE) mg/L; Figure 8). In Year 2, estimated mean nitrate concentrations dropped even further to 0.63 ± 0.12 (SE) mg/L (Figure 8). A slightly different trend occurred with estimated medians. Estimated median nitrate values dropped from 0.83 mg/L Pre-Restoration to 0.37 mg/L during Passive Restoration to 0.04 mg/L in Year 1 and then climbed slightly to 0.38 mg/L in Year 2 of Full Restoration (Wilcoxon, $n=267$, $df=3$, Chi-Square=27.1, $P<0.0001$; Figure 8). The slightly disparity in trends between Year 1 and Year 2 probably results from the influence of very different climatic conditions and the immediate post-breach nitrate pulse in Year 1. Conditions were wetter in Year 2, with 50% of the sampling periods falling during moderate to large storm events, so estimated median nitrate concentrations in Year 2 were driven up relative to Year 1, which was drier in terms of precipitation. However, pulses of nitrates immediately after the levee breach and during two February 2009 storm events elevated the estimated mean concentration in Year 1 relative to Year 2, despite the fact that most of the other sampling periods in Year 1 had very low estimated nitrate levels.

The influence of storm events is evident in results for Reference Areas, as well. Both estimated mean and median nitrate concentrations were higher in Year 2 (0.76 mg/L and 0.34 mg/L) than in Year 1 (0.35 mg/L and 0.07 mg/L; Figure 8). Interestingly, estimated mean and median nitrate levels appeared to drop after the Pre-Restoration period (~2002-2006) from 0.86 mg/L and 0.70 mg/L, respectively, to 0.36 mg/L and 0.13 mg/L during Passive Restoration (~2007 – 2008; Wilcoxon, $n=278$, $df=3$, Chi-Square=52.3, $P<0.0001$). While one of the Reference Area marshes is located directly adjacent to the Project Area, the other two locations are at the southern end of Tomales Bay and in another watershed completely, so changes in nitrate levels cannot be ascribed entirely to the restoration project, particularly as waters only infrequently discharged from the more heavily managed – and polluted – parts of the Project Area downstream. In fact, both the proximal and distant Reference Areas in Tomales Bay showed similar temporal patterns in nitrate levels, as well as equivalent Pre-Restoration concentrations, with estimated means ranging from 0.87 to 0.89 mg/L and estimated medians, from 0.70 to 0.77 mg/L. Both Walker Creek Marsh and the Undiked Marsh displayed much lower nitrate levels between 2006 and 2009 than they did prior to 2006, with estimated median concentrations ranging from 0.07 to 0.19 mg/L. In fact, in Year 1, the estimated median nitrate concentration for both marshes was 0.07 mg/L. In Year 2, median nitrate concentrations rose slightly again in both marshes (0.30 to 0.46 mg/L), but not to the levels that were observed between 2002 and 2006. Ironically, more storm events were sampled after 2006 than prior to that time, so a higher frequency of storm samples prior to 2006 cannot explain this downward trend in nitrate levels within Reference Areas.

In most of the Project and Reference Areas, nitrates never exceeded USEPA water quality objectives of 10 mg/L as nitrate-N (or 44 mg/L as NO_3) for human consumption, even prior to restoration (Parsons 2009). However, in the East Pasture, approximately 7% of the nitrate samples collected exceeded 44 mg/L prior to restoration, with most of the exceedances coming from a ditch at the base of the Dairy Mesa that receives non-point source run-off from Point Reyes Station, as well as potentially septic-influenced groundwater (ibid). This same upstream boundary sampling site continues to show elevated nitrates even after restoration and exceeded 10 mg/L during every sampling event in Year 2 and 75% of the events in Year 1. In Year 1, at least two other sampling locations displayed sharp pulses in nitrate concentrations, with one site located at the upstream boundary twice exceeding this limit and another site within the Project Area exceeding only once. No Project Area sampling locations exceeded 10 mg/L in Year 2.

Interestingly, nitrites were generally not detected (<0.05 mg/L), in the Project Area prior to restoration, but they were occasionally found in Reference Areas, with Walker Creek and

Limantour Marsh both having six (6) detections, although only three (3) samples exceeded RWQCB recommended thresholds of 0.5 mg/L (ibid). Because nitrates were only rarely recorded prior to restoration, nitrites were not specifically sampled during the Passive Restoration and Full Restoration sampling periods.

Ammonia

Prior to restoration, most of the ammonia pulses in the Project Area occurred in waters with lower oxygen (or pH) levels and appeared more related to cattle grazing and other management practices such as ditch maintenance than with timing of storm inflows or run-off (Parsons 2009). Cattle grazing provided a source of ammonia that would be maintained in low oxygen waters, while ditch maintenance promoted hypoxic conditions by increasing organic matter available for mineral decomposition and creating a surge in biological oxygen demand. These conditions favored retention of nitrogen as ammonia rather than as nitrates.

Within the Project Area (excluding upstream sites), estimated ammonia concentrations Pre-Restoration in the East Pasture averaged 2.61 ± 1.51 (SE) mg/L, which differed significantly from values estimated for the West Pasture (0.45 ± 0.24 (SE) mg/L) and Tomasini Creek (0.20 ± 0.01 (SE) mg/L; Wilcoxon Score, $p < 0.001$; ibid). However, because of the high number of non-detections during Pre-Restoration due to use of a commercial laboratory, a more valid parameter might be the distribution of “detections” among sampling sites. Of the 64 detections of ammonia during the Pre-Restoration period, more than 47 % of them occurred in the East Pasture, a substantial – and statistically significant – difference from the other Project and Reference Area subsampling areas that accounted for no more than 11 % of the detections (Contingency Table, Chi Square, $n=99$, $df=4$, Chi-Square=13.4, $p=0.009$; ibid).

Overall, there was apparently no statistically significant differences in the number of detections between Study Areas Pre-Restoration (Contingency Table, Chi Square, $n=320$, $df=2$, Chi-Square=2.70, $p=0.26$; Parsons 2009). However, before levees were breached, estimated concentrations appeared to be substantially higher in the Project Area (mean = 1.26 ± 0.58 (SE) mg/L) than in the Reference Areas (mean = 0.23 ± 0.01 (SE) mg/L) or Upstream Areas (mean = 0.22 ± 0.01 (SE) mg/L; Wilcoxon, $p < 0.001$; Wilcoxon Score, $n=320$, $df=2$, Chi-Square=22.46, $p < 0.001$, ibid). Ammonia pulses in Reference Areas prior to restoration most likely resulted from decreases in oxygen levels in tidal creek waters due to high primary productivity and subsequent respiration or an increase in water residency time than from point-source loading. Conversely, sporadic pulses in creeks such as Lagunitas and Walker Creek probably related more to point-source loading or an immediately proximal source of ammonia than to the presence of a low oxygen environment.

Following restoration, the number of ammonia detections decreased in the Project Area during Year 1 of Full Restoration, dropping 43% from 22.8% of the samples Pre-Restoration to 13.0% of the samples after restoration (Figure 9). The number of total ammonia detections decreased even more dramatically in Year 2 of Full Restoration, dropping 48% from Year 1 with 6.8% of the samples exceeding detection limits (Contingency, $n=724$, $df=11$, Chi-Square=48.5, $P < 0.0001$; Figure 9). Interestingly, the number of detections was even lower during Passive Restoration, falling to 4.6%. Estimated mean total ammonia concentrations within the Project Area did appear to drop 73% from 1.26 ± 0.58 (SE) mg/L Pre-Restoration to 0.34 ± 0.10 (SE) mg/L during Year 1 of Full Restoration and 88% to 0.15 ± 0.06 (SE) mg/L during Year 2 of Full Restoration. Estimated East Pasture mean ammonia concentrations appeared to drop even more dramatically from 2.61 ± 1.51 (SE) mg/L to 0.44 ± 1.51 (SE) mg/L in Year 1 and 0.24 ± 0.15 (SE) mg/L, a decrease of 83% and 91%, respectively, from Pre-Restoration levels.

The increase in ammonia detections between Passive and Full Restoration periods could be entirely attributable to restoration-related changes: increase in ammonia following mineralization of decomposing organic matter and flushing of ammonia from soils into overlying waters with reintroduction of tidal and creek flows after the deliberate drawdown during construction. Oxygen

and pH conditions within Project Area waters would appear sufficient to promote rapid conversion of ammonia into nitrates, so these detections suggest local, continued production of ammonia at sampling sites from plants and, to some degree, wildlife.

One interesting caveat to this hypothesis is that, during Year 1 of Full Restoration, ammonia detections increased in all of the Study Areas following restoration, even those distant from the Project Area. The number of detections in Reference Areas jumped from 3.9% of the samples Pre-Restoration to 10.3% in WY2008/2009, while detection frequencies during Passive Restoration were roughly equivalent to Pre-Restoration (4.0%). In Year 2, the frequency of total ammonia detections in Reference Areas dropped slightly to 6.1%, but still appeared higher than Pre-Restoration levels. While these changes were not considered statistically significant (Contingency, $n=267$, $df=3$, $Chi-Square=2.0$, $P=0.57$), these negative results must be interpreted with caution because of expected counts lower than 5 in three of the eight cells. Despite the potential increase in ammonia detections during Year 1, estimated average ammonia concentrations in Reference Areas appeared roughly equivalent between Pre-Restoration (0.23 ± 0.01 (SE) mg/L) and Year 1 post-restoration (0.24 ± 0.04 (SE) mg/L) and actually appeared to drop slightly in Year 2 (0.18 ± 0.05 (SE) mg/l).

The overall increase in Year 1 and, to a lesser degree, in Year 2 in frequency of total ammonia detections, if not average concentrations, within both the Project Area and Reference Areas – some of which are distant from the Project Area – suggests that the increases in ammonia detections documented in WYs 2008/2009 and 2009/2010 do not all result from the effects of restoration. One possible explanation for the increase in ammonia detections in Year 1 may be the dry winter, which allowed tidal influence or the “salt wedge” 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 – tides affect rise and fall of water level, but do not affect salinity – mobilized ammonia into overlying waters, causing a net efflux or transport from the system. In these areas, ammonium, phosphate, and silicate fluxes increased by 20 to 38%; reduced iron fluxes increased by ~150%; methane fluxes decreased by 77%; and in situ organic carbon mineralization rates increased by ~110% (Joye et al. undated). Most of this increase probably results from cation exchange of the strongly ionic sodium chloride for ammonium (Craft et al. 2009), but ammonia may also be produced through increased mineralization of organic matter under tidal versus freshwater regimes. Salinity data collected in WY2008/2009 showed increases in salinity not only in the Project Area, which was expected, but in Reference Areas, so this supports the potential for increased upstream tidal influence to have caused biogeochemical changes that resulted in more frequent ammonia detections, at least during Year 1. In Year 2, wetter conditions drove down salinities below Pre-Restoration median levels by as much as 5 ppt, so higher ammonia detection frequencies in Year 2 relative to Pre-Restoration and Passive Restoration periods are harder to explain, although frequencies did decrease relative to Year 1.

Interestingly, despite occasional spikes in ammonia concentrations, only a few sampling locations prior to restoration exceeded the maximum concentration limit for unionized ammonia in estuarine waters of 0.16 mg/L (Parsons 2009). Some of these included East Pasture drainage ditches, where ammonia reached as high as 76 mg/L prior to restoration, and even one sampling location on Lagunitas Creek in April 2003, when total ammonia levels climbed as high as 13 mg/L. While ammonia was obviously detected in lower, but still relatively high, concentrations elsewhere in the dairy ranch, particularly in the East Pasture, temperature and/or pH did not climb high enough to encourage dissociation of ammonia into its unionized ion.

In general, ammonia detection frequencies between the Project Area and Reference Areas in Years 1 and 2 of Full Restoration showed no statistically significant differences (Contingency, $n=224$, $df=1$, $Chi-Square=2.0$, $P=0.636$). These results would suggest that the Project Area is beginning to converge with Reference Areas in terms of nutrient levels, but short-term and long-term climatic conditions and other forces may end up causing system-wide changes in nutrient levels and patterns that will affect both Project and Reference Areas.

Phosphates

Phosphates appeared to be driven more by biogeochemical processes than upstream loading, at least in most of the Project Area (Parsons 2009). While concentrations of phosphates prior to restoration were sometimes high during storm events – as was observed in Walker Creek and Lagunitas Creek -- they also showed peaks during spring and fall (ibid). These spring and fall peaks probably resulted from recirculation of phosphates from sediments into overlying waters when the upper sediment and bottom water layers became anoxic due to low oxygen levels at the soil-water interface, which can occur when plankton respiration rates increase substantially.

Prior to restoration, phosphate concentrations were highest in the Project Area and, specifically, in the East Pasture due to not only the proximity of sources such as cattle and septic-influenced groundwater, but also to agricultural management regimes that caused oxygen levels within ditch waters to frequently be low (Parsons 2009). Before the levees were breached, significant differences occurred between the frequency of detection between Study Areas (Chi Square Test, $n=183$, $df=2$, $Chi\text{-Square}=9.29$, $p=0.010$), with the number of detections disproportionately higher in the Project Area than in the other areas (ibid). Phosphates averaged an estimated 0.99 ± 0.16 (SE) mg/L in the Project Area Pre-Restoration compared to 0.23 ± 0.03 (SE) mg/L for Reference Areas and 0.12 ± 0.01 (SE) mg/L for Upstream Areas (Wilcoxon Score, $n=346$, $df=2$, $p<0.001$; ibid).

The East Pasture largely accounted for the disproportionate number of samples in which phosphates were detected Pre-Restoration (26%; Chi-Square Test, $n=51$, $df=4$, $Chi\text{-Square}=25.47$, $p<0.001$; ibid). It also accounted for 76% of the values recorded in the upper end of the detection range (0.79 – 9.4 mg/L), with detections in other subsampling areas typically falling below 0.79 mg/L (ibid). In the East Pasture, concentrations averaged an estimated 2.40 ± 0.33 (SE) mg/L Pre-Restoration, which was significantly higher than the means for the rest of the Project Area (excluding upstream sampling sites), which ranged from 0.15 mg/L (West Pasture) to 0.24 mg/L (Olema Marsh; ibid).

Low oxygen levels also probably accounted for the higher estimated average phosphate concentrations for Olema Marsh and for the higher estimated average concentration and loading rates during the summer for many of the Reference Areas such as Limantour and Walker Creek marshes. Phosphate levels within Reference Areas would also be influenced by the greater relative proximity of most of these systems to the ocean, where phosphorous is naturally high (Mitsch and Gosselink 2000, Day et al. 1989).

Following restoration, estimated mean phosphate concentrations in the Project Area appeared to drop significantly, decreasing 88% from 0.99 ± 0.16 (SE) mg/L Pre-Restoration to 0.12 ± 0.02 (SE) mg/L during Year 1 and 0.12 ± 0.01 (SE) mg/L during Year 2 of Full Restoration (Wilcoxon, $n=245$, $df=3$, $Chi\text{-Square}=37.7$, $P<0.0001$). Estimated levels after levee removal were even lower than those during Passive Restoration (average= 0.68 ± 0.37 (SE) mg/L). Estimated concentrations in the East Pasture dropped from 2.40 ± 0.33 (SE) mg/L Pre-Restoration to 0.11 ± 0.02 (SE) mg/L during Year 1 and 0.10 ± 0.02 (SE) mg/L during Year 2 of Full Restoration, which represents approximately a 95 % decrease in phosphates after restoration. During Year 2, other sampling areas had estimated mean concentrations ranging from 0.08 mg/L (downstream Lagunitas Creek) to 0.17 mg/L (OM). The concentration differences in Year 2 between sampling areas within the Project Area were apparently minor enough that they were not considered statistically significant (Wilcoxon, $n=44$, $df=4$, $Chi\text{-Square}=3.4$, $P=0.495$). Phosphate concentrations – and perhaps the frequency of phosphate detection – have probably dropped to the discontinuation of active agricultural management and, with the removal of the levees, the improvement in oxygen levels within pasture waters.

Despite plummeting phosphate concentrations after restoration, median phosphate concentrations in Years 1 and 2 of Full Restoration still differed between the Project Area (0.11 mg/L) and the Reference Areas (0.07 mg/L). Therefore, the Project Area has not totally

converged with Reference Areas, at least in terms of phosphate concentrations. The same restoration-related factors that can affect nitrate and ammonia levels can also drive up phosphates, i.e., breakdown and mineralization of decaying pasture organic matter. In addition, frequent to continued inundation of former pasture areas may create an anoxic soil interface that encourages flux of agriculturally related phosphates from soils into overlying waters.

Some caveats must be noted for these results. Analytical chemistry methods were changed between Pre-Restoration and subsequent sampling periods or treatments, with measurement of Total Dissolved Phosphates being changed to measurement of Orthophosphates. Also, the method detection limit decreased greatly, which negates our ability to use Contingency Tables to evaluate changes in the number of detections between treatments. Total Dissolved Phosphates typically incorporates both Orthophosphates, as well as Polyphosphates, so Orthophosphates would be considered to represent a smaller fraction of the dissolved phosphorous component, although Polyphosphates are unstable and will eventually convert over time to Orthophosphate, particularly in low oxygen waters (Murphy 2007). A comparison of Orthophosphate and Total Dissolved Phosphates for several sampling periods during Passive Restoration when both were measured showed typically 94 to 99% correlation, although, during one sampling event, correlation was as low as 48%: as samples were collected in different jars and sent to different laboratories, the dynamic and extremely variable nature of natural waters, which can change rapidly from moment to moment, does not make the latter result extremely surprising.

Pathogens A Major Issue in Project -- and Reference -- Areas, but Levels in Project Area Dropped Dramatically After Restoration

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, the Project Area was certainly located in an area where it could have had maximum impact on downstream water quality.

Prior to restoration, the Project Area had substantially higher estimated median concentrations of fecal coliforms (1,600.9 mpn/100ml) than the Reference Areas (72.0 mpn/100 ml), although seeming differences with Upstream Areas (705.6 mpn/100 ml) might have been obscured to some degree by high variance in the data (MLE Regression, n=379, df=2, Chi-Square=98.5, $p < 0.0001$; Parsons 2009). Not surprisingly, the heavily managed East Pasture had significantly higher estimated geometric means or medians (6,298.8 mpn/100 ml) Pre-Restoration than most of the other sub-sampling areas, 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 subsampling areas ranged between 356.9 mpn/100 ml for downstream Lagunitas Creek to 1,131.7 mpn/100 ml for the West Pasture (*ibid*).

In terms of compliance with Basin Plan or TMDL standards, prior to restoration, more than 95% of all samples collected from the Project Area and Upstream Areas exceeded objectives for shellfish harvesting and municipal water supply of 14 and 20 mpn/100 ml respectively (Parsons 2009). Approximately 78% exceeded contact water recreation standards of 200 mpn/100 ml, and 36-47% of the values actually were higher than 2,000 to 4,000 mpn/100 ml, the standards for non-contact water recreation (*ibid*). Lagunitas Creek exceeded the TMDL standard of 200 mpn/100 ml during 72 % of the sampling events and the 90th percentile standard of 400 mpn/100 ml 58% of the time, with the overall geometric mean and 90th percentile estimated at 584.6 mpn/100 ml and 6,146.8 mpn/100 ml, respectively (*ibid*). The TMDL load-based allocation of 95 mpn/100 ml set for Green Bridge location on Lagunitas Creek was never met during the Pre-Restoration study

period. In comparison, only 34% of Reference Area samples exceeded contact water recreation standards, and less than 12% exceeded non-contact water recreation standards (ibid).

Following restoration, the estimated geometric mean or median fecal coliform concentrations decreased significantly in the Project Area, dropping 94% from 1,600.9 mpn/100 ml pre-restoration to 90.4 mpn/100 ml in Year 1, but climbing slightly (56%) to 141.3 mpn/100 ml in Year 2 (MLE, $n=270$, $df=3$, $\text{Chi-Square}=58.8$, $P<0.0001$; Figure 10). Estimated median concentrations during Passive Restoration fell in-between those of Pre- and Full Restoration (919.9 mpn/100 ml; Figure 10). Not surprisingly, large decreases were recorded in the East Pasture, with estimated median levels dropping more than 98 % from 6,298.8 mpn/100 ml to 64.1 mpn/100 ml in Year 1 and 119.7 mpn/100 ml in Year 2. Estimated mean concentrations in the West Pasture also dropped by more than 95% from 1,137.5 mpn/100 ml Pre-Restoration to 48.7 mpn/100 ml Year 1 post-restoration. Decreases in Olema Marsh were also substantial (125%), with concentrations in Bear Valley Creek at the downstream boundary of the Project Area slipping from 1,821.4 mpn/100 ml to 394.8 mpn/100ml in Year 1 and 376.6 mpn/100 ml in Year 2 of Full Restoration, even though restoration was not quite as extensive in this area during this phase of the project. Concentrations in downstream Lagunitas Creek in Year 1 after restoration (median=141.5 mpn/100 ml) appeared to be lower than those recorded before restoration (median=356.9 mpn/100 ml), but not those in Year 2 (median=341.8 mpn/100 ml).

Estimated median levels in Reference Areas were lower in WY2008/2009 or Year 1 post-restoration (median=21.0 mpn/100 ml) than in the years prior to restoration (72.0 mpn/100 ml; MLE, $n=280$, $df=3$, $\text{Chi-Square}=7.56$, $0.10 > p > 0.05$). This might suggest that lower concentrations in WY 2008/2009 have been affected to some degree by the dry winter, decreased precipitation, and reduced pollutant inflow. This is supported by the fact that, in Year 2 or WY 2009/2010, coliform levels generally increased in the Project Area and in the Reference Areas, with the past year estimated back near 70.6 mpn/100 ml. WY 2009/2010 was much wetter than WY 2008/2009, and almost 50% of the sampling events just happened to occur during storms. For Reference Areas, it is also possible that the drop in Year 1 reflects the absence of Limantour Marsh from Year 1, particularly as higher values occurred again in Year 2, when Limantour Marsh was added back into the sampling frame. Concentrations in this marsh have typically been low as reflected by the estimated mean of 23.1 mpn/100 ml for the Pre-Restoration sampling period. For evaluation purposes, Limantour Marsh was deleted from Year 2, and statistics were re-run: median coliform levels did drop to 59.8 mpn/100 ml with removal of Limantour, but those levels are still higher than Year 1.

The dramatic declines in fecal coliform concentrations in the Project Area following restoration, even during wetter periods, are also evident in changes in the frequency of exceedance of Basin Plan or TMDL standards. 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 80 and 85% of all samples collected during Year 1 and Year 2 of Full Restoration, respectively. Approximately 39% of all samples collected in Year 1 and 44% of samples collected in Year 2 exceeded the contact water recreation standards of 200 mpn/100 ml, compared to approximately 78% Pre-Restoration, at least a 43% decrease. Only 4 to 8% of samples collected after levees were breached exceeded 2,000 to 4,000 mpn/100 ml, the standards for non-contact water recreation, whereas 36 to 47% exceeded before levee removal, an 85- to 89% decrease in exceedances, respectively.

Coliform levels at the upstream end of the Project Area boundary on Lagunitas Creek at the Green Bridge remained roughly similar to Pre-Restoration Conditions, with the TMDL standard of 200 mpn/100 ml being exceeded approximately 60% of the time in Year 1 and 71% of the time in Year 2, compared to 72 % of the time prior to restoration. The 90th percentile of 400 mpn/100 ml standard was exceeded approximately 27% of the sampling periods in Year 1 and 30% of the sampling periods in Year 2 after restoration 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 slightly from 100% during Pre-Restoration to 89% during Year 1 and 98% during

Year 2. The overall geometric mean and 90th percentile in Year 1 were estimated at 275.8 mpn/100 ml and 827.2 mpn/100 ml, while those same parameters were estimated in Year 2 at 366.6 mpn/100 ml and 1,317.0 mpn/100 ml. Both of these levels are substantially lower than that estimated for Pre-Restoration, with the overall geometric mean being 955.3 mpn/100 ml and the 90th percentile being 5,852.7 mpn/100 ml. The frequency of exceedance of contact water recreation standards in Reference Areas decreased from 34% to 9% in Year 1 and 28% in Year 2.

Again, given the similar patterns in fecal coliform levels observed between Project and Reference Areas in the first and second years following Full Restoration, changes in Project Area concentrations before and after levee removal cannot be completely ascribed to restoration, as changes in precipitation and pollutant inflow in dry Year 1 and the wetter Year 2 must be taken into account. While two storm events were sampled in 2009, reduced rainfall may have caused an overall drop in pollutant mobilization or loading for the 2008/2009 Water Year. Conversely, the fact that more than 50 % of the sampling events in the WY 2009/2010 coincided with storm events undoubtedly influenced those results. In some instances, upstream loading inputs did not appear affected by climatic changes. Fecal coliform levels at one non-point source discharge point into the restored wetlands ranged from 800 to 5,000 mpn/100 ml during both years, with this same site having levels as high as 30,000 mpn/100 ml at one point prior to restoration. More sampling post-restoration is necessary to determine whether results from the past years are reflective of climatic conditions or real declines associated with restoration or changes in watershed use or loading patterns.

Loading Rates in Project Area Increase Slightly as Expected After Restoration Due to Hydrologic Reconnection of Diked Former Pasture Lands

Despite high concentrations in the Project Area prior to restoration, loading rates for the Giacomini Ranch and Olema Marsh Pre-Restoration were usually lower or only slightly higher than Reference Areas (Parsons 2009). This trend reversal resulted from the fact that the East Pasture – where concentrations were highest – essentially contributed nothing to downstream loading, because it was diked (ibid). The only potential for loading from the East Pasture came during moderate to large storm events when waters in the pasture overtopped the levees or when the Giacomini occasionally pumped ditch waters into Lagunitas Creek (ibid). However, even if the East Pasture had been operated as a muted tidal unit, the volume of water and, subsequently, loading that these ditches and sloughs could have contributed to downstream flow would have been relatively insignificant (between 0.1 and 1.15 mg/s for nitrate loading), based on rates estimated using average discharge for similarly sized creeks in the adjacent Undiked Marsh: with diking of both Lagunitas and Tomasini Creeks, the East Pasture had no other source watersheds to increase flow and loading volumes (ibid).

Prior to restoration, then, loading rates were generally highest in Upstream Areas, which included sampling locations on the upstream perimeter of the Project Area on Lagunitas, Tomasini, Bear Valley, and Fish Hatchery Creeks (Parsons 2009). There were some exceptions. For example, for fecal coliform, estimated loading rates for the Project Area (mean=249,389 mpn/s) were lower than Upstream Areas (mean=3.86 million mpn/s), but higher than Reference Areas (mean=60,094.1 mpn/s; ibid). Conversely, Reference Areas had the highest loading rates for phosphates (0.15 mg/s), with rates for the Project Area (0.03 mg/s) and Upstream Areas (0.06 mg/s) considerably lower, which, as discussed earlier, may relate to the more substantial marine influence in these areas (ibid).

As with concentrations, estimated median loading rates Pre-Restoration were considerably smaller than mean loading rates, showing the influence of pulses during the winter or wet season sampling events (Parsons 2009). One of the clear findings from our study is the close relationship between rainfall, run-off, streamflow, and loading. While these relationships were not always distinct enough to be linear, with some exceptions, most of the high loading events

occurred during winter or wet-season sampling events, with the highest values usually occurring 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 (ibid). During storm events, concentrations in Lagunitas Creek can reach as high as 2.0 – 2.5 mg/L, which is considerably higher than the peak nitrate concentrations of approximately 1.5 mg/L (24 μ M) documented off the Point Reyes coast that is potentially exported into Tomales Bay during upwelling events (Largier et al. 2006, Wilkerson et al. 2006). While these upwelling events may influence nutrient conditions in the outer portion of Tomales Bay during the summer, when streamflow is lowest, the likelihood that these nutrient reach the inner portion of Tomales Bay is reduced by the fact that, during the summer, hydrologic exchange between the outer and inner portions of the Bay becomes infrequent, occurring only every 120 days, due to changes in estuarine circulation patterns (Hollibaugh et al. 1988) 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 have been the least sampled in Tomales Bay due to inherent planning and logistical difficulties, however, we are increasing efforts to capture storm events in the future monitoring record.

Because levees essentially precluded or minimized export of pollutant loads from the ranch pastures, with full levee removal, the contribution of the Project Area to downstream loading would be expected to increase, even if concentrations within the Project Area dropped dramatically. For example, for fecal coliform, estimated geometric mean or median loading rates appeared to increase from 57.5 mpn/s pre-restoration to 98.3 mpn/s in Year 1 and even higher to 291.2 mpn/s in Year 2, although differences between these two treatments or sampling periods were technically not considered statistically significant based on the strong variance (MLE, n= 217, df=3, Chi Square = 6.06, 0.20>p>0.10; Z-test, p>0.087; Figure 12). Conversely, estimated arithmetic mean loading rates seemingly decreased from 249,389 \pm 369,023 (SE) mpn/s during Pre-Restoration to 52,716.9 \pm 74,464.5 (SE) mpn/s during Year 1 and 33,657.5 \pm 37,936.7(SE) mpn/s during Year 2 (Figure 12). Similar comparative trends of higher medians and lower arithmetic means in Year 1 were also observed for fecal coliform loading when all Study Areas (e.g., PA, REF, US) were incorporated into the pre- and post-restoration comparative analysis, although, again, differences were not statistically significant (MLE, Z-test, all P>0.694; Figure 12). Originally, we hypothesized that the lower arithmetic means reflected the dry conditions during WY2008/2009 or Year 1, but as they were low in WY 2009/2010 or Year 2, as well, which was a wetter year, that cannot completely account for these results. .

For nitrates, loading appeared to increase between Pre-Restoration and Full Restoration, with estimated means climbing from 0.54 \pm 0.35 (SE) mg/s pre-restoration to 0.82 \pm 0.28 (SE) mg/s in Year 1 and 0.91 \pm 0.28 (SE) mg/s in Year 2 (Wilcoxon, n=218, df=3, P=0.0.001; Figure 11). Conversely, nitrate loading within Reference Areas alone appeared to decrease from 0.51 \pm 0.11 (SE) mg/s pre-restoration to 0.09 \pm 0.04 (SE) mg/s in Year 1 and then climb again sharply to 1.63 \pm 1.30 (SE) mg/s in Year 2 (Wilcoxon, n= 222, df=3, Chi-Square=25.3, p<0.0001; Figure 11) . Again, in Year 1, dry weather conditions appear to reduce average nitrate loading within reference marshes, but these reductions were seemingly offset in the Project Area by the reconnection of the formerly diked pastures to Lagunitas and Tomasini Creeks and the increased connectivity of Bear Valley Creek and Olema Marsh with Lagunitas Creek. In Year 2, nitrate loading increased universally with the increase in overall precipitation levels and the frequency of storm sampling events.

Ultimately, the Project Area may not converge with conditions present in Reference Areas. The Project Area bears the full brunt of approximately 66% of the freshwater – and pollutant – inflow to Tomales Bay. While the Undiked Marsh also falls in this system, it is further downstream, and pollutants are more likely now to have been intercepted by the newly restored flood- and marshplains of the Project Area. Walker Creek does receive the full of Walker Creek flows, but this subwatershed -- and potentially its pollutant load – is smaller than that of Lagunitas Creek.

During the first two years after restoration, fecal coliform loading rates in the Project Area still differ from Reference Areas (overall MLE, $n=224$, $df=2$, Chi Square=17.1, $0.001 > p > 0.0001$; $Z=-2.56$, $p=0.01$), as well as seemingly from Upstream Areas ($Z=1.93$, $p=0.054$). Estimated geometric mean coliform loading totaled 166.8 mpn/s for the Project Area after restoration, compared to 44.3 mpn/s for Reference Areas and 495.5 mpn/s for Upstream Areas. The same was true for nitrate loading after restoration, with median nitrate loading rates varying from 0.15 mg/s for Upstream Areas to 0.004 mg/s for Reference Areas and 0.017 mg/s for Project Areas (Wilcoxon, $n=222$, $df=2$, Wilcoxon=17.8, $p < 0.0001$). Interestingly, mean nitrate loading rates were seemingly equivalent between the Project Area (0.87 ± 0.29 (S.E.) mg/s), Reference Areas (1.00 ± 0.78 (S.E.) mg/s), and Upstream Areas (0.76 ± 2.60 (S.E.) mg/s) during the post-restoration period.

Restored Wetlands' Potential to Trap Downstream Pollutant Loads Still Evolving

Because of being extensively leveed prior to restoration, the Project Area was not expected to provide much in the way of downstream reduction in either concentrations or loading of nutrients or pathogens (Parsons 2009). In general, floodplain systems are most effective at removing particulate forms of nutrients and other pollutants, because emergent vegetation “traps” the sediment or organic matter and removes it from water sheetflowing across the floodplain or marshplain surface. Pollutants can also be trapped within creek channels and bays by physical forces related to fluvial and estuarine sediment transport and circulation processes. Sediment laden with nutrients, organic matter, and pollutants are likely to deposit in areas where the creek gradient flattens or velocities decrease sharply.

Downstream reductions in pollutants prior to restoration were evaluated for two parameters – nitrates and fecal coliform (Parsons 2009). Several of the sampling locations are strategically arranged with sites at the upstream boundary of the Project Area and either at the downstream boundary or midway through the Project Area. Ultimately, the most robust evaluations rely on having a closed system such that there are no additional inputs or at least large sources of inputs other than the waters flowing in at the upstream boundary. Also, these analyses require that all sources of surface water inflow be accurately accounted for to reliably estimate both inputs and outputs. In the original analyses, Lagunitas Creek was incorporated using the upstream boundary at the Green Bridge and the downstream boundary at the former North Levee. However, at least two additional major creeks merge with Lagunitas Creek in between – Olema Creek and Bear Valley Creek – and, while pollutants have been measured at the Bear Valley Creek outlet because it was part of the restoration project, they had not been measured at the Olema Creek outlet. Therefore, total upstream loading has been underestimated. In addition, Lagunitas Creek is large enough, particularly at its downstream end near the former North Levee, that estimating outflow reliably from one grab sample and a flow rate taken in one nearshore area is probably not accurate. For these reasons, Lagunitas Creek is not incorporated in current downstream loading analyses, although, at some point, should funding become available, its incorporation would be key to better characterizing inputs and outputs in terms of loading and understanding the true effects of the restoration on watershed water quality, as this creek represents 66% of the freshwater inflow to Tomales Bay (Fischer and Smith 1996).

Before the levees were breached, fecal coliform concentrations and loading showed no statistically significant pattern of downstream reductions for any of the Project Area creeks, although high variability in the data may have masked differences (Parsons 2009). Median pathogen concentrations and/or loading rates actually increased downstream in some areas, including Fish Hatchery Creek and Bear Valley Creek (ibid). For both of these systems, this suggests that there are some additional inputs other than the upper portions of Fish Hatchery and Bear Valley Creek watersheds, such as wildlife use or septic-influenced surface water and groundwater flowing from the adjacent developed portion of Inverness Ridge into the west end of the marshes (ibid).

Despite the fact that soluble nutrients such as nitrates are the least effectively trapped pollutants by floodplain systems, nitrates did show some downstream reductions Pre-Restoration for many of the creeks, including Fish Hatchery Creek, Tomasini Creek, and Bear Valley Creek, all of which were leveed or impounded to some degree (Parsons 2009). Nitrate reductions on Fish Hatchery Creek may have resulted from change in creek gradient as creek flows from the steep slopes of the Inverness Ridge drop abruptly onto the broad, low-gradient floodplains in the West Pasture (Parsons 2009). Fish Hatchery Creek was not leveed directly adjacent to the creek, but contained within the Lagunitas Creek levees, with exchange with Lagunitas Creek and other undiked areas limited to a tidegate at its downstream end (ibid). Trapping also appeared to occur on Bear Valley Creek and Tomasini Creek (ibid). For Tomasini Creek, most of the reduction in nitrate concentrations and mean (if not median) loading rates probably occurred due to the change in creek gradient and flow velocity and trapping of materials within the creek itself, not on the relatively narrow floodplains, which were limited to narrow strips of marshplain within the tightly confining levee system (ibid). (Tomasini Creek was leveed separately by the Giacomini in the 1960s to exclude it from the East Pasture). With its defined inlet and outlet, Olema Marsh, in some ways, resembles a constructed treatment marsh, where long residence times often result in accelerated trapping of nitrates (ibid). However, similar to Tomasini Creek, mean, but not median, nitrate loading was lower at the downstream Bear Valley Creek outlet.

Following restoration in 2008, the Giacomini Wetlands are in a state of evolution and can be expected to take time to reach their full nutrient trapping potential due to the loss of vegetation and larger expanse of bareground during the conversion of pastureland to marsh. Some preliminary evaluations of fecal coliform data collected in the first year of Full Restoration showed no statistically significant differences in estimated fecal coliform loading between upstream and downstream sampling sites in the first year of Full Restoration, although most creeks did appear to show lower estimated geometric means or medians for pathogen levels and loading downstream than upstream. One exception to this was Bear Valley Creek in Olema Marsh, where estimated median loading, if not estimated median concentrations, actually appeared to be higher downstream than upstream, although differences were not statistically significant (MLE, $n=12$, $df=1$, $\text{Chi-Square}=2.83$, $0.10 > p > 0.05$).

In Year 2, both average and median instantaneous nitrate loading rates appeared to be lower downstream than upstream on Fish Hatchery Creek and Tomasini Creek, but not on Bear Valley Creek, although none of the analyses were statistically significant (Paired t-tests and Mann-Whitney, all $p > 0.110$). In contrast to nitrate loading, fecal coliform loading appeared generally higher downstream than upstream, with the exception of average (arithmetic mean) fecal coliform loading on Tomasini Creek, where downstream instantaneous loading rates were lower (Paired t-tests and Mann-Whitney, all $p > 0.116$). However, as with nitrate loading, none of these differences were statistically significant. High variability, combined with low sample size, may obscure the significance of any differences. To improve power, Years 1 and 2 were combined together for analysis. Fecal coliform loading results for the combined years still appeared to conflict with Year 1 results, in which instantaneous fecal coliform loading rates seemed to drop between upstream and downstream sampling locations. As Year 2 post-restoration was considerably wetter than Year 1, and 50% of the sampling events occurred during a storm, this climatic factor may have been enough to “swamp” the potential of this still evolving wetland system to reduce downstream pollutant loading.

Some of the difficulties in analyzing how successful the restored wetlands are in trapping pollutants results from the fact that Fish Hatchery Creek, Bear Valley Creek, and even the East Pasture are not “closed” systems. Inflow from small drainages, groundwater, and non-point source discharge enters these systems in between the upstream and downstream sampling points. Groundwater inflow, in particular, is difficult to characterize in terms of loading due to its diffuse nature, but, based on results of sampling of these areas over the years, groundwater and non-point source discharges could be contributing greatly to pollutant loading. Based on past results, groundwater and non-point source discharges should elevate both fecal coliform and nitrate loading, so the contrasting results from nitrate and fecal coliform loading are puzzling,

particularly as nitrate is much less likely than fecal coliform to be trapped or filtered by marsh soils.

Ultimately, all of these analyses suffer from low power due to the few sampling events that have occurred since restoration and the amount of “noise” present in this type of environmental data. Continued sampling in future years will boost power and perhaps enable us to better evaluate upstream and downstream trends in pollutant loading, at least for most creeks. Over time, the restored Giacomini Wetlands will evolve into a more functional floodplain/marshplain system that will be more effective at trapping pollutants. The ability of this wetland to trap pollutants will be enhanced in years with overbank flood flow: in WYs 2008/2009 and 2009/2010, there were no overbank flood events, despite the fact that Year 2 or WY 2009/2010 was a wetter year. We will continue to look for ways to improve our analytical approach to comparison of upstream and downstream sampling sites in the future.

Conclusions

In the environmental assessment document (NPS 2007), the impact analysis section predicted short-term negative impacts resulting from the conversion of pastureland to marsh, with long-term benefits for water quality conditions within the former Giacomini Dairy Ranch, as well as for downstream water quality and the health of Tomales Bay. In general, 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, and expected issues with temporary increases in turbidity and temporary decreases in dissolved oxygen and pH did not materialize, although turbidity levels did increase during Year 2. Initially, some of these lower-than-expected temporary impacts may have resulted from the fact that Year 1 or WY 2008/2009 was a dry year, and few large storms occurred that would have contributed to reworking of this evolving landscape, even though we captured some of the few larger storm events that did occur. However, Year 2 was a wetter year, and 50% of the sampling events in WY 2009/2010 occurred during a storm event. As a result, turbidity levels almost doubled from 10.7 NTU in Year 1 to 22.2 NTU during Year 2, and fecal coliform concentrations and loading rates and nitrate loading rates also increased during Year 2 from 11% to 196%. In some instances, such as nitrate concentrations and fecal coliform loading, mean concentration or loading levels appeared to decrease between Years 1 and 2, while median levels showed a seeming increase. Despite this, other nutrient-related either decreased from Year 1 to Year 2 or remained roughly equivalent, including total ammonia detections and phosphate concentrations,

As would be expected, loading rates of pathogens and presumably nitrates increased 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). One of the most important indicators, however, of the success of this project in improving downstream water quality conditions in Tomales Bay will be changes in loading between upstream and downstream sampling locations. While nitrate loading did appear to decrease at downstream locations during the first two years after restoration, fecal coliform loading appeared to increase, although none of these differences were statistically significant due probably to low sample size and high variability. In general, Year 2 was wetter than Year 1, so there were generally higher pollutant inputs for this still evolving system to cope with during storm events. Some of these analyses are undoubtedly affected by the fact that these systems are not closed and receive inputs of nitrates and fecal coliforms from smaller drainages, groundwater inflow, and non-point source discharge. Most of these sources would be expected to vary similarly to the marshes such that loading is higher during higher rainfall periods, however, the non-point-source discharge site in Point Reyes Station had consistently high nitrate and fecal coliform levels throughout both Year 1 and Year 2. Estimating pollutant loading from diffuse groundwater and non-point sources is complex, but years of sampling results definitely show that these areas could be contributing significantly to loading in these systems. Continued sampling in future years will boost power and, hopefully, the ability to evaluate trends for downstream loading of nitrates and fecal coliform. In addition to analysis improvements, over time, the restored Giacomini Wetlands will evolve into a more functional floodplain/marshplain system that will be more effective at trapping pollutants.

In addition to the effects of restoration on the Project Area, our monitoring has also revealed some other interesting trends that may pertain changes in the Tomales Bay ecosystem as a whole. In Year 1, total ammonia detections actually increased in the Project Area relative to Passive Restoration conditions, which, in and of itself, was not surprising considering the amount of biogeochemical upheaval the Giacomini Wetlands was undergoing in the first year after breaching of the levees. However, what was more interesting was that, concurrently, the number of total ammonia detections also increased in Reference Areas, after being relatively consistent during the Pre-Restoration period. The number of detections in Reference Areas jumped from 3.9% of the samples Pre-Restoration to 10.3% in WY2008/2009, while detection frequencies

during Passive Restoration were roughly equivalent to Pre-Restoration (4.0%). In Year 2, the frequency of total ammonia detections in Reference Areas dropped slightly to 6.1%, but still appeared higher than Pre-Restoration levels.

One possible explanation for the increase in ammonia detections in Year 1 may be the dry winter, which allowed tidal influence or the "salt wedge" 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 – tides affect rise and fall of water level, but do not affect salinity – mobilized ammonia into overlying waters, causing a net efflux or transport from the system. Most of this increase probably results from cation exchange of the strongly ionic sodium chloride for ammonium (Craft et al. 2009), but ammonia may also be produced through increased mineralization of organic matter under tidal versus freshwater regimes. Salinities were generally higher in WY 2008/2009. In Year 2, wetter conditions drove down salinities below Pre-Restoration median levels by as much as 5 ppt, so higher ammonia detection frequencies in Year 2 relative to Pre-Restoration and Passive Restoration periods are harder to explain, although frequencies did drop relative to Year 1.

The other phenomenon that may be at least partially influenced by system-wide changes is the apparent decrease in pH in certain Reference Areas, as well as the Project Area. Since pre-restoration, pH has consistently decreased every year in the Project Area from 7.60 to 7.14 in Year 2 despite the increase in higher pH tidal waters and expected decrease in the influence of lower pH groundwater inflow on the restored system. Some of this decrease may be attributable to the restoration such that the breakdown of organic matter is generating more humic acids. However, pH has not only declined in the Project Area, but also in Reference Areas. In Reference Areas, pH dropped from a median of 7.62 during the Pre-Restoration and Passive Restoration sampling periods to 7.44 during Year 1 and 7.35 during Year 2 after restoration. Median pH declined more sharply in the Undiked Marsh and Limantour Marsh than in Walker Creek Marsh, where medians remained remarkably consistent until WY 2009/2010, and, even then, geometric means were roughly equivalent to those in years past. Between pre-restoration and Year 2 sampling periods, median pH plummeted from 7.48 to 7.15 in the Undiked Marsh and 7.80 to 7.34 in Limantour Marsh.

The dramatic decreases in median pH in the Undiked Marsh and Limantour Marsh since WY 2007/2008 could potentially result from the fact that, in these systems, upstream areas have been restored, and restoration is affecting pH of downstream marshes, as well as the Project Areas. However, there are indications that pH is declining throughout Tomales Bay. A recent reoccupation of the 10 LMER Tomales Bay sampling stations monitored extensively between 1987 and 1995 by UC Davis researcher Ann Russell and her colleagues found that most water quality parameters had not changed greatly over the past decade, except for pH and dissolved organic (DOC) and inorganic carbon (DIC). The pH of Tomales Bay appears to have dropped by as much as 0.25 pH units since the late 1980s-1990s (A. Russell, UC Davis, *pers. comm.*). While changes in pH invariably lead to questions about the effect of ocean acidification on pH of tidal waters flowing into estuaries, Russell noted that the change observed in Tomales Bay was too large to be attributable to dissolution of CO₂ from the atmosphere into estuarine waters. However, not all carbon inputs into the estuary come from the atmosphere (*ibid*). Russell and colleagues are planning to construct a mass balance model for DOC and DIC to evaluate relative inputs from marine and terrestrial sources. However, even if these pH changes are 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 in oceans even without the influence of climate change.

Where Do Monitoring and Restoration Efforts Go From Here

Monitoring

Ultimately, monitoring of water quality and other hydrological variables will become part of a larger evaluation of the successfulness of restoration efforts. Based on evaluation of preliminary data, predicted restoration changes, and results from some of the progress criteria analyses proposed in the Long-Term Monitoring Program Framework: Part I (Parsons 2005), it appears that some water quality monitoring variables might be more capable of discerning change between pre-restoration and restored conditions and the direction of the evolutionary restoration trajectory (i.e., are restored wetlands becoming more like reference marshes?) than others. For example, the pattern of salinities between the Project and Reference Areas may never totally converge, because the Project Area receives more direct, abundant, and perennial freshwater inputs than Reference Areas. Some factors such as salinity may not seemingly not represent a good indicator for evaluating improvement in conditions within the Project Area and convergence of conditions with those observed in Reference Areas, but may ultimately be important as harbingers of potential future changes in the system from direct and indirect effects of climate change, including changes in pH, water level, extent of high tides, and salinity.



Photograph of restored marsh by Louis Jaffe

For the third year of Full Restoration, we will continue quarterly synoptic sampling of field parameters, nutrients, pathogens, and productivity indicators. We will also continue with our efforts to understand hydrodynamics within the restored wetlands through continued installation of continuous water level and conductivity instruments, as well as cross-sections of the new and previously existing tidal channels. In keeping with the goals outlined in our analysis of Pre-Restoration data (Parsons 2009), we are continuing to improve our monitoring approach by increasing frequency and spatial coverage of sampling during storm events, assessing particulate as well as dissolved nutrients, and better assessing nutrients such as total ammonia and total dissolved phosphates through use of analytical techniques with lower laboratory detection limits. While monitoring is focused on assessing change resulting from restoration, our results show that we will need to constantly take into account more system-wide or even global changes resulting from climate change, which ultimately may have a significant effect on both Project Area and Reference Area systems.

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. 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 Freshwater 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 (*Rana aurora draytonii*). 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 (Figures 8 and 10). This sampling site continues to show elevated nitrates even after restoration and exceeded 10 mg/L during every sampling event in Year 2 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 the Tomales Bay Pathogen Study (RWQCB 2001) 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 2001).

To improve the quality of waters flowing into the restored marsh, the Seashore is partnering with the Tomales Bay Watershed Council, the West Marin chapter of the NOAA-Bay Watershed Education and Training Program (B-WET) program, and the County of Marin to develop a demonstration bioswale project. The goal of this project is to alter the physical and biotic structure of the ditch through installation of gravels and plants so that the filtering capacity of the ditch will be improved, which should, in turn, improve the quality of water flowing into the Tomasini Triangle Marsh in the Giacomini Wetlands. Not only will this project improve the quality of waters flowing into the marsh, it will provide a unique educational opportunity for students in West Marin to learn about watersheds, water quality, and wetland science while fostering stewardship for the environment and their local community. In addition, the project partners also hope to educate the general public about stormwater run-off pollution issues and how community members can help solve pollution problems that negatively impact local wetlands. This project is currently in the planning and development stage, and the partners hope to finish compliance in summer 2011 to allow construction in fall 2011.

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 the 6 months has shown 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 marsh. The Seashore is currently working with TBWC to better understand the source of this discharge and the laws that regulate it.

Interestingly, fecal coliform levels in outflow from this pipe have typically been low, and MBAS – the surfactant found in detergents – has only been detected in trace amounts (R. Carson, TBWC, unpub. data). In contrast, fecal coliform levels at the sampling site on the marsh boundary 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 fillucoides*).

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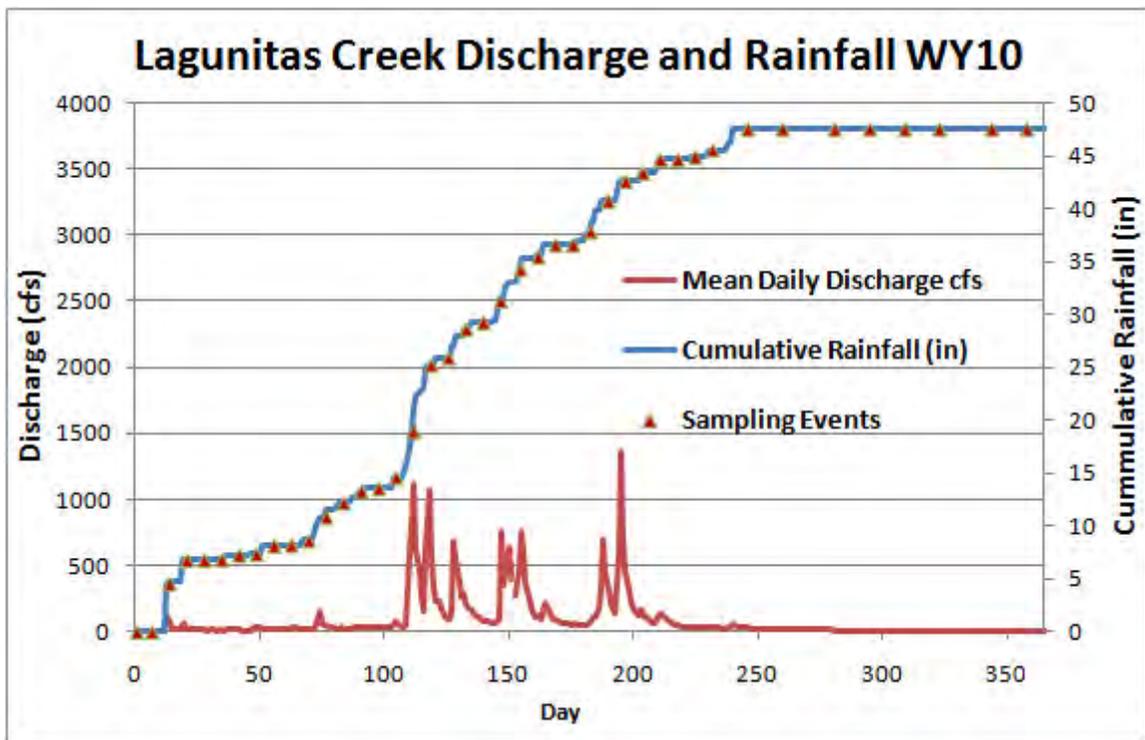
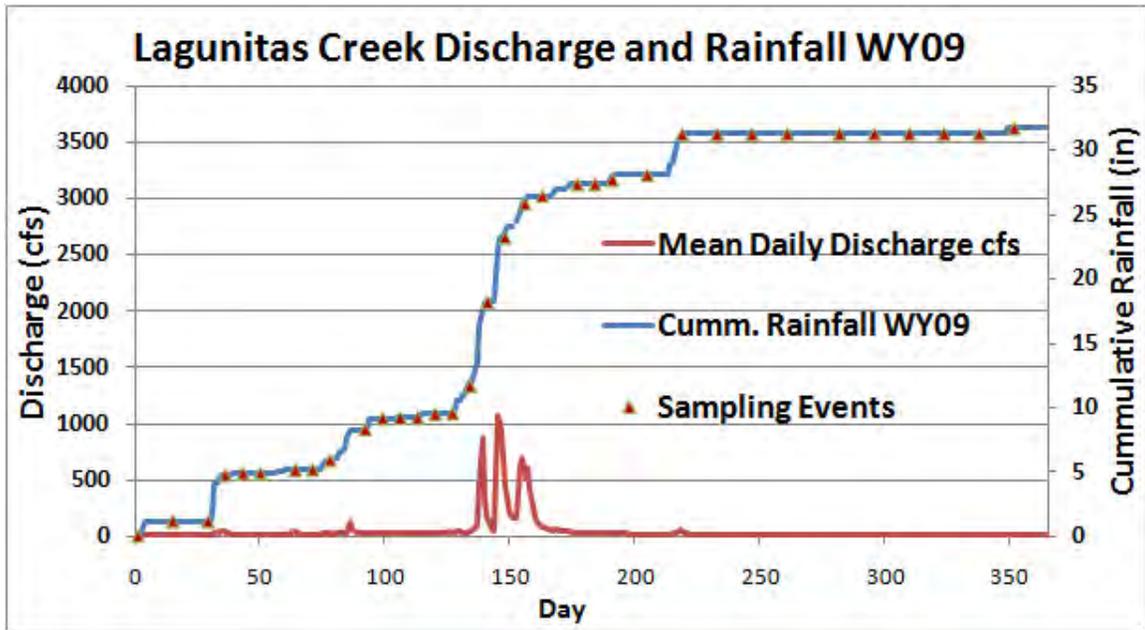


Figure 1. Lagunitas Creek discharge and cumulative rainfall in WYs 2008/2009 and 2009/2010. Graphs courtesy of Tomales Bay Watershed Council.

2008, 2009, and 2010 Low Tidelines Giacomini Wetland Restoration Project



Location Map



National Park Service

Point Reyes National Seashore/
Golden Gate National Recreation Area
Marin County, CA

Subtidal extents were mapped using a combination of aerial photography and direct observation. Handheld GPS units were used to collect data.

Subtidal areas were predicted using information provided by KHE (2006), including topographical elevations, tidal datum information, and hydraulic modeling.

- Low Tide 2008 - Spring Tide
- Low Tide 2009 - Neap Tide
- Low Tide 2010 - Spring Tide
- Predicted Subtidal Extents



In 2008, low tides ranged from -0.44 to -1.74 feet MLLW at Inverness.

In 2009, low tides ranged from 1.88 to 2.69 feet MLLW at Inverness.

In 2010, low tides ranged from -1.67 to -1.54 feet MLLW at Inverness.

Figure 2. 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.

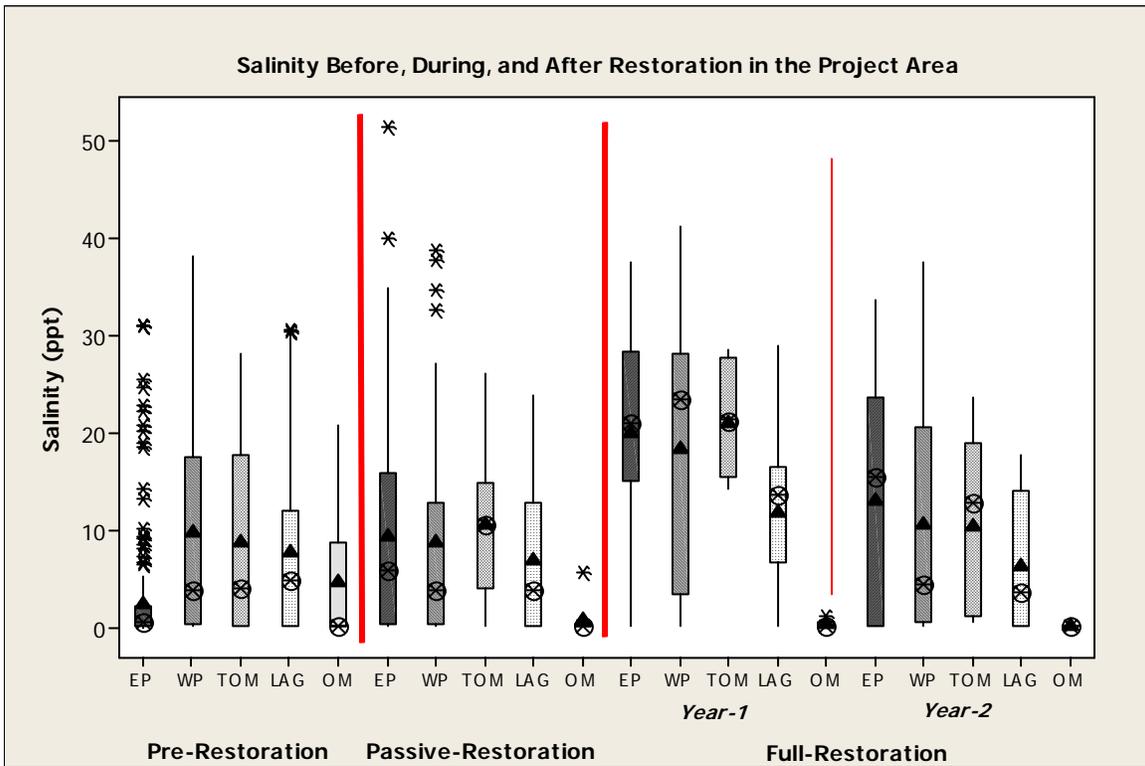


Figure 3. Average, median, and other summary statistics for salinity in the Project Area subsampling units Pre-Restoration, during Passive Restoration, and in the first two years of Full Restoration.

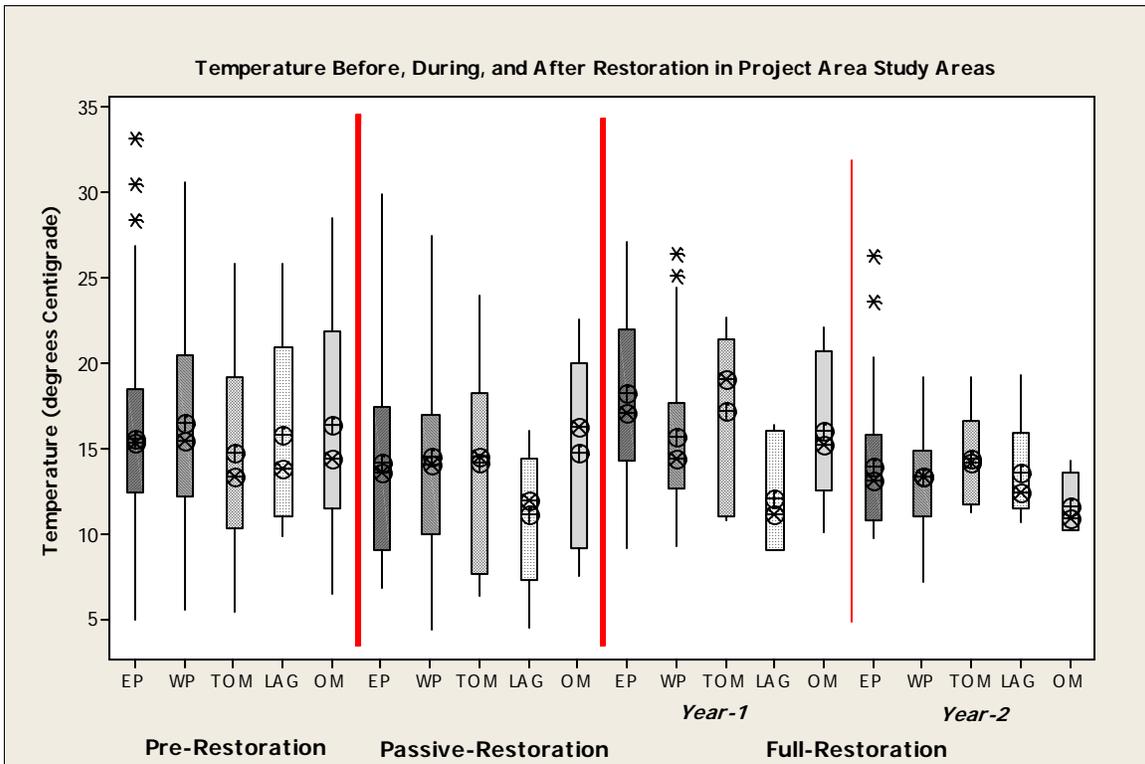


Figure 4. Average, median, and other summary statistics for temperature for Project Area subsampling units Pre-Restoration, during Passive Restoration, and in the first two years of Full Restoration.

Boxplots indicate first and third quartiles (25%, 75%). Lines indicate 10th and 90th percentiles. Medians are indicated by diagonal-hatched circle, with means designate by black triangles, except in Figure 4, where means are designated by vertical/horizontal-hatched circle.

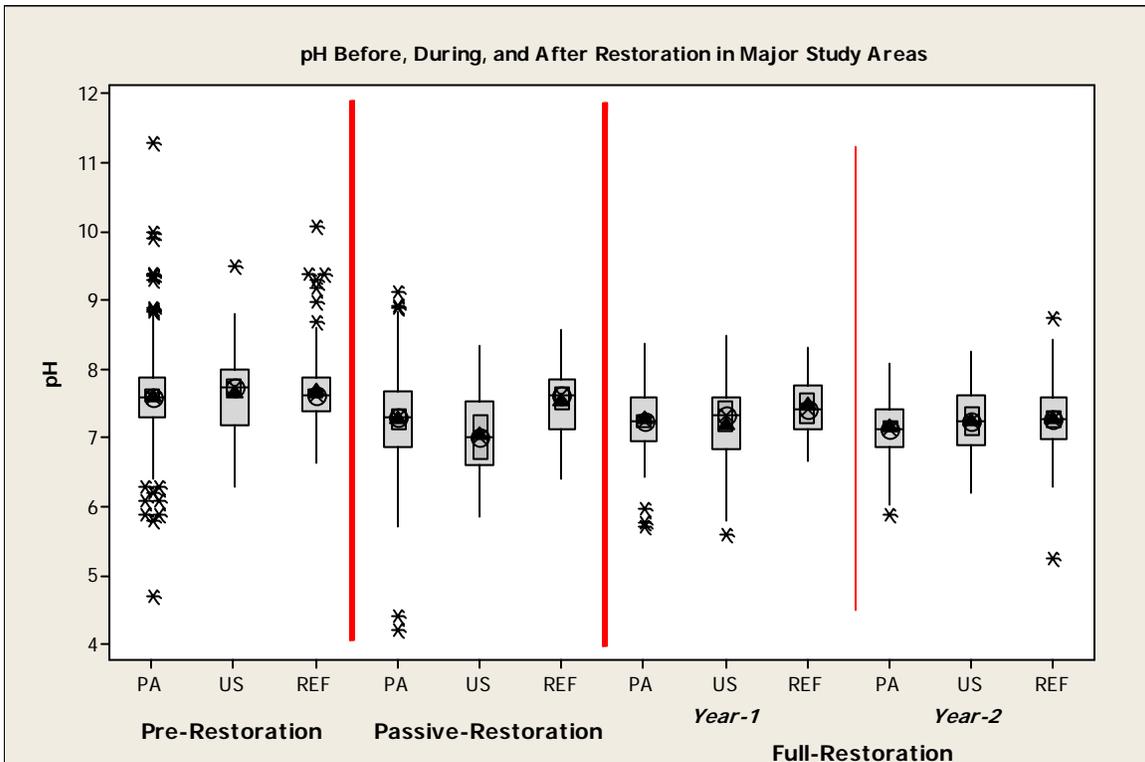


Figure 5. Average, median, and other summary statistics for pH for Study Areas Pre-Restoration, during Passive Restoration, and in the first two years of Full Restoration.

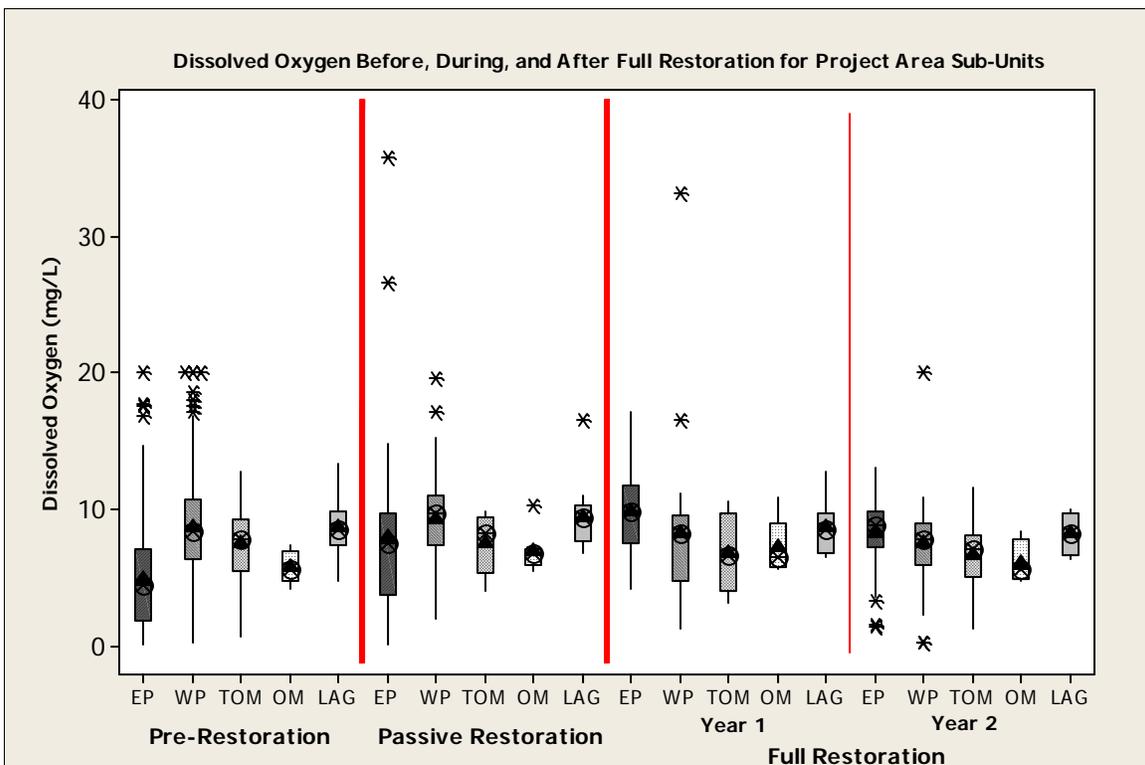


Figure 6. Average, median, and other summary statistics for dissolved oxygen for sub-sampling areas in the Project Area Pre-Restoration, during Passive Restoration, and in the first two years of Full Restoration.

Boxplots indicate first and third quartiles (25%, 75%). Lines indicate 10th and 90th percentiles. Medians are indicated by diagonal-hatched circle, with means designate by black triangles.

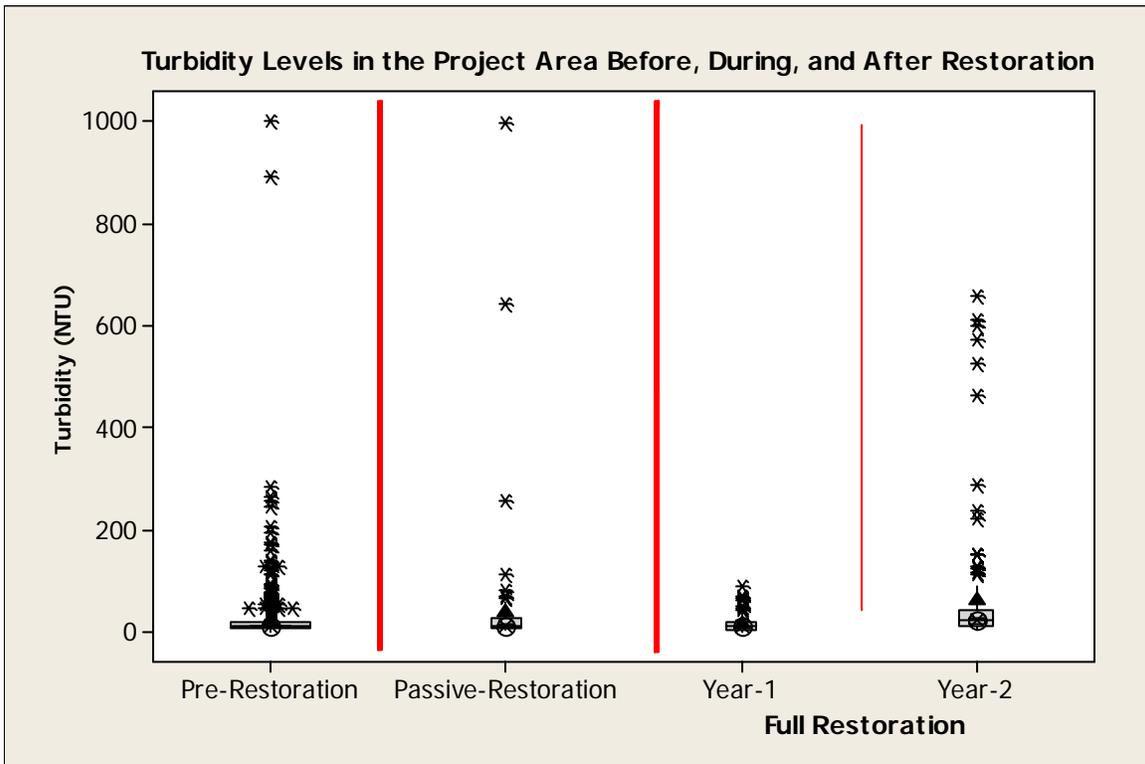


Figure 7. Average, median, and other summary statistics for turbidity in the Project Area Pre-Restoration, during Passive Restoration, and in the first two years of Full Restoration.

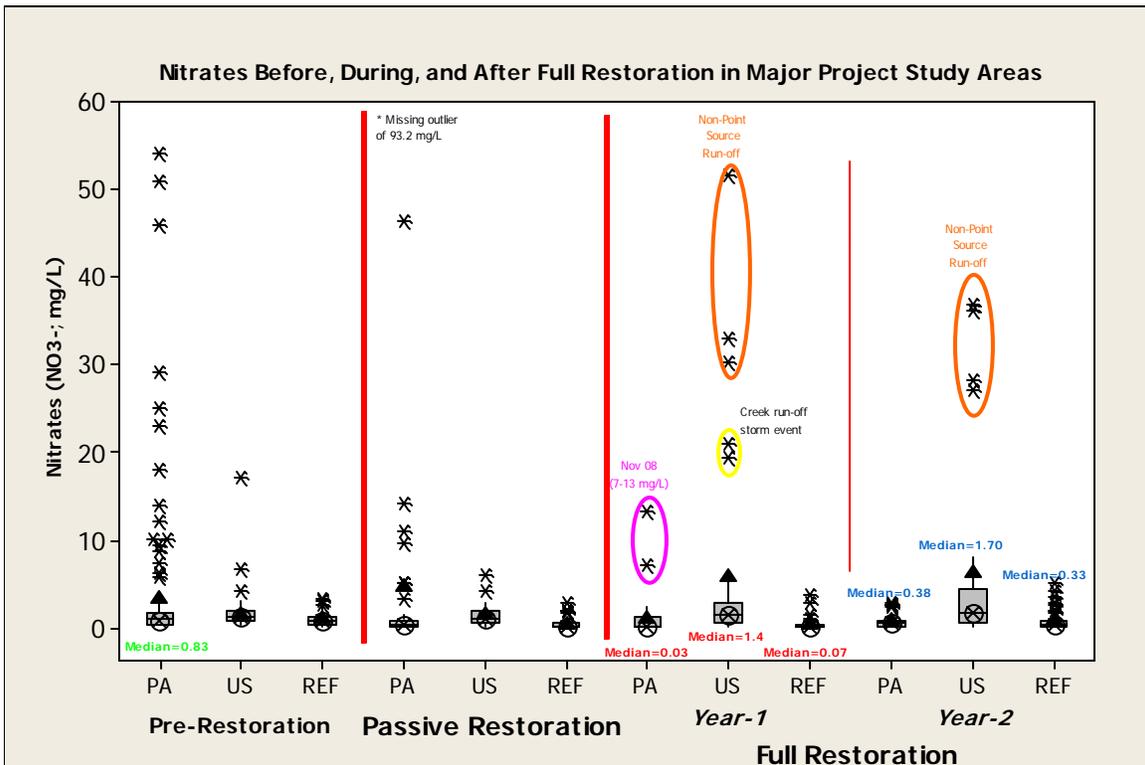


Figure 8. Average, median, and other summary statistics for nitrates (NO₃⁻) for Study Areas Pre-Restoration, during Passive Restoration, and in the first two years of Full Restoration.

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

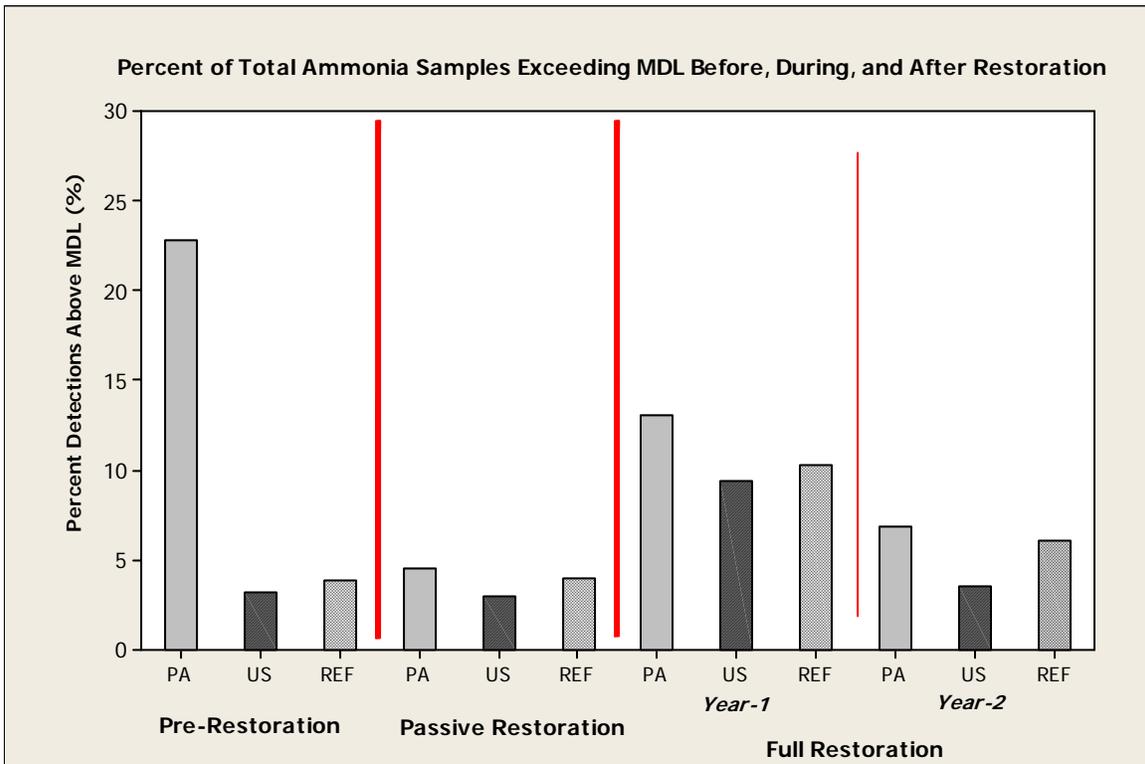


Figure 9. Percent of samples above Total Ammonia concentration detection limits for Study Areas Pre-Restoration, during Passive Restoration, and in the first two years of Full Restoration.

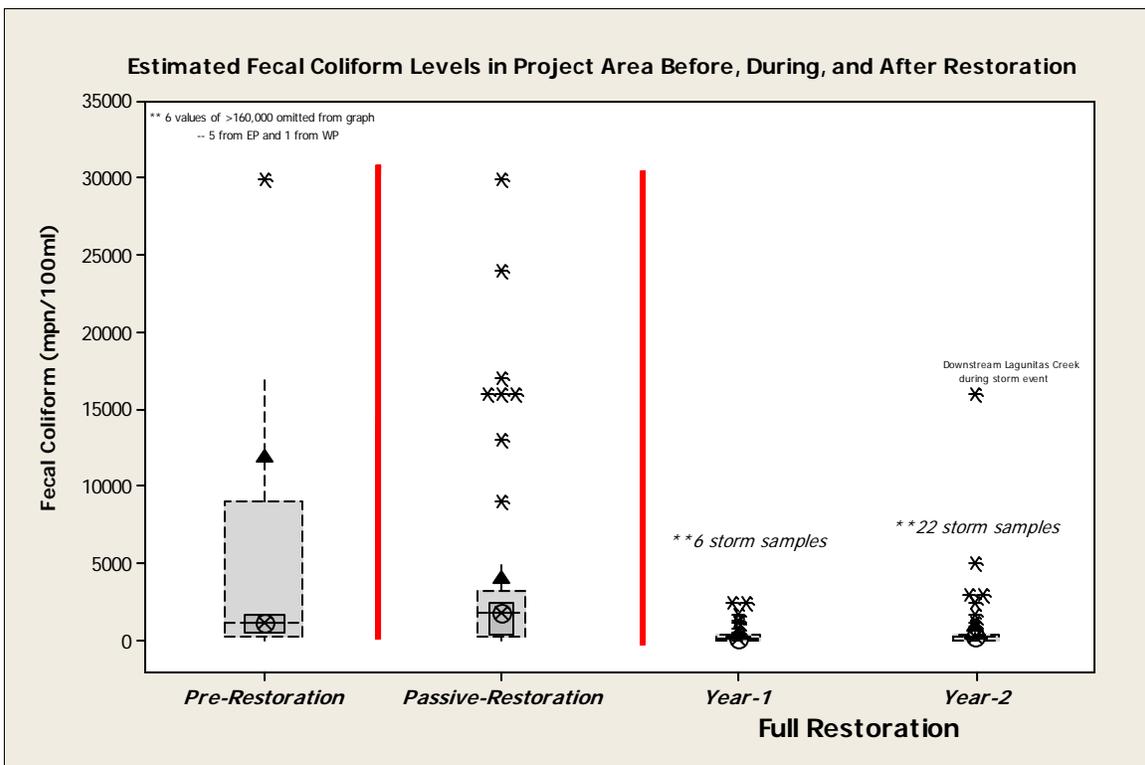


Figure 10. Average, median, and other summary statistics for fecal coliform concentrations for the Project Area Pre-Restoration, during Passive Restoration, and in the first year of Full Restoration. Note: dashed lines for boxplots indicates both right- and left-censored data. Light-shaded grey boxplots indicate first and third quartiles (25%, 75%), while dark-shaded grey boxplots indicate median confidence intervals. Lines indicate 10th and 90th percentiles. Medians are indicated by diagonal-hatched circle, with means designate by black triangles.

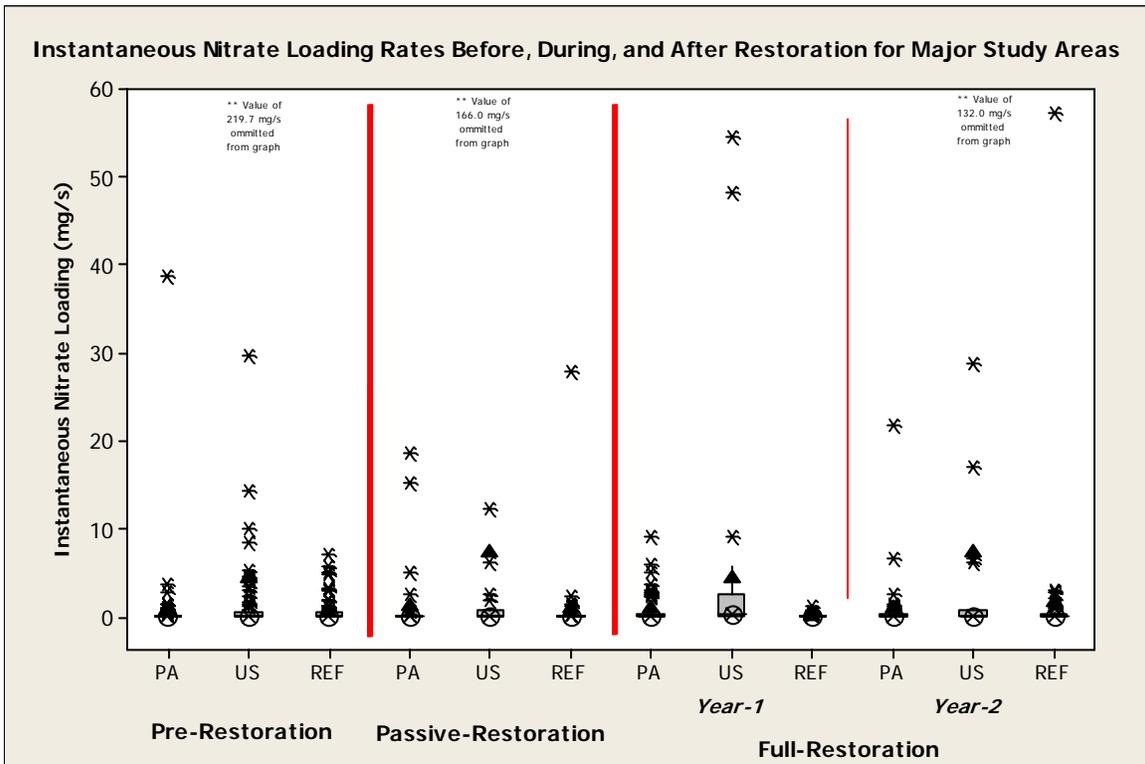


Figure 11. Average, median, and other summary statistics for nitrate instantaneous loading for Study Areas Pre-Restoration, during Passive Restoration, and in the first two years of Full Restoration.

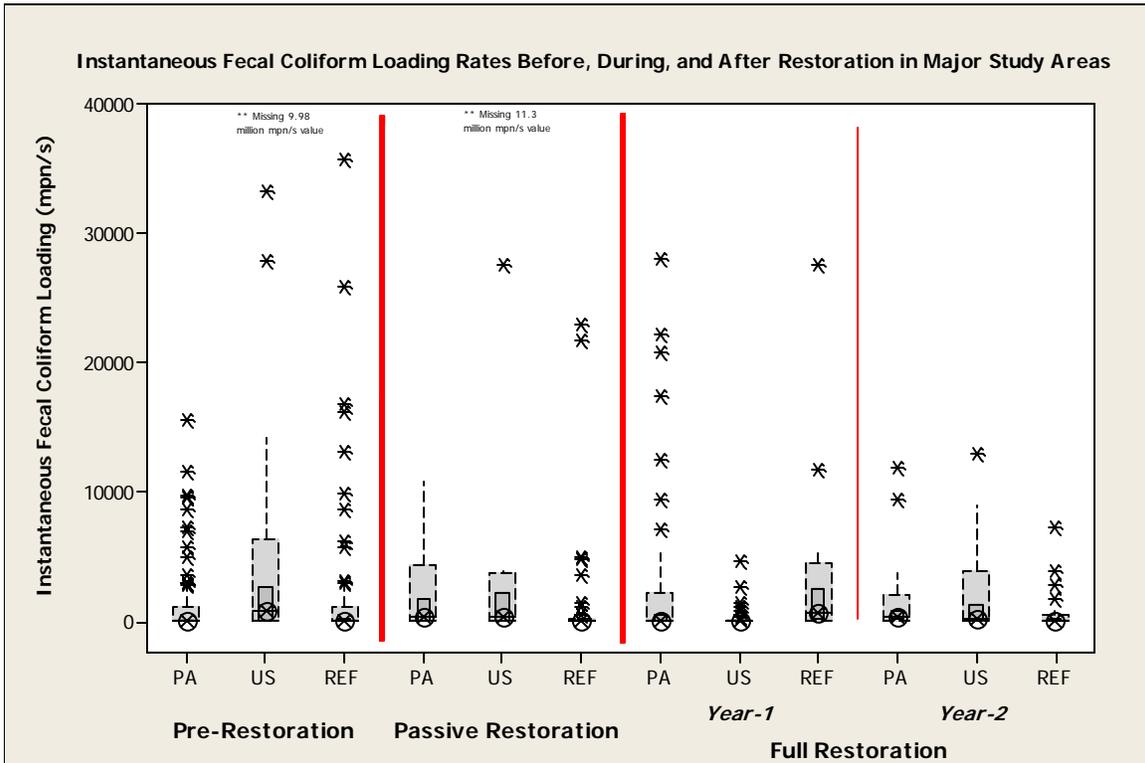


Figure 12. Average, median, and other summary statistics for fecal coliform loading for Study Areas Pre-Restoration, during Passive Restoration, and in the first two years of Full Restoration.

Light-shaded grey boxplots indicate first and third quartiles (25%, 75%), while dark-shaded grey boxplots indicate median confidence intervals. Lines indicate 10th and 90th percentiles. Medians are indicated by diagonal-hatched circle, with means designate by black triangles. Note: dashed lines for boxplots indicates both right- and left-censored data.