Appendix A: Year Three 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, sub-sampling 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 Giacominis continued to operate a full-scale dairy operation under a Reservation of Use Agreement. There were at least three dairy herds, and the ranch was actively maintained through manure spreading, haying, and flood and spray irrigation of certain pastures in the summer. This period is referred to in data analyses as Pre-Restoration as it pre-dates any restoration efforts.

Monitoring was continued on a more limited scale following discontinuation of the full-scale dairy operation in 2006. In 2006, the Giacominis sold the dairy string and instead grazed a much smaller herd of dairy heifers. Maintenance activities were also scaled back, with reduced haying, manure spreading, and irrigation of pastures during the summer. 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 if 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); five scheduled/storm sampling events conducted in Year 2 (2009-2010), and six (6) scheduled and storm sampling events conducted in Year 3 (2010-2011). 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. Sampling events in Year 3, which occurred in Water Year (WY) 2011, were conducted in early late October-early November 2010; November 2010; late January-early February 2011; late March 2011; late-April to early May-2011; and mid- to late-July 2011.

This technical memorandum summarizes changes in water quality conditions within the Project Area during the third year after restoration for WY 2011. Water quality conditions in the Project Area during the first and second years after restoration and prior to restoration were summarized in two previous reports (Parsons 2009; Parsons 2010; Parsons 2011). 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 two years after restoration, the speed with which conditions improved within the Project Area for variables such as dissolved oxygen and nitrate and fecal coliform concentrations far exceeded our expectations, and expected issues as discussed above with large temporary increases in turbidity and temporary decreases in dissolved oxygen and pH did not materialize. Some of this may have partially 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. Even in the second year, which was much wetter, there were no overbank flooding events. 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 two years after restoration, loading rates of pathogens and presumably nitrates actually increased in the Project Area relative to pre-restoration conditions, because, prior to levee removal, the pastures had either no direct connection to Lagunitas or other creeks (East Pasture) or only muted tidal connection (West Pasture) and, therefore, were only very infrequently in a position to contribute to downstream "loading." 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 and detection limits 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 from the surrounding urban watershed 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 later added to the analytes list, so there is no comparative pre-restoration data for this constituent: inclusion of this parameter 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, a statistical package such as Minitab (State College, PA) or Systat (Chicago, IL) were used to statistically analyze data using traditional either parametric or non-parametric techniques. For non-parametric procedures, the Mood's Median Test has been more employed in recent years rather than Kruskal-Wallis, because, while the Kruskall-Wallis test may be more powerful in terms of detecting change, the Mood Median test 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, total dissolved phosphates, and total phosphorous 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), Kaplan-Meier Survival Analysis, 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 Three: Results and Discussion

Changes in Climatic Patterns

As noted earlier, six (6) scheduled or storm sampling events occurred in Year 3 or WY 11. At least one of these sampling events actually coincided with a sizeable storm event (March), and, in general, WY 2011 (October 2010 – September 2011) was a much wetter year (49.09 inches) than even the previous WY, 2010 (October 2009 – September 2010; 47.95 inches; Table 1). Both years exceeded the 20- to 30-year rainfall averages for Bear Valley area of approximately 36.4 to 37.0 inches (Table 1). WY 2011 differed from WY 2010 not so much in volume of rainfall, but in the distribution of rainfall, with rainfall events in 2011 stretching well into June, which is typically a dry month in California (Table 1). The U.S. Geological Survey stream discharge data for the Point Reyes Station depicts this unusual pattern by showing above average stream discharge throughout the summer of 2011 (Figure 1).

The first year after restoration was actually slightly drier than average (31.36 inches), but the driest year during the monitoring period actually occurred in the early years of monitoring, while the dairy was still in operation (WY 2004: 22.29 inches; Table 1). While levee removal during restoration in 2008 resulted in lowering of creek "bank" elevations to that sufficient to allow overflow of a 2-year-flood event, no overbank flooding from storm flows occurred during either WY 2009 or 2010, even with higher rainfall levels in 2010. In WY 2011, at least one overbank flooding event occurred.

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 of the smaller channels would develop naturally over time. While levees were removed, the undiked marsh that had developed on the outboard of the levees was, in many cases, higher in elevation than the marshplains or former pastures. These marsh shelves, then, represent mini "levees" that can direct – or even constrain – flow within the former pastures.

Hydrologic changes were not notable after final removal of the West Pasture levees and completion of preliminary restoration activities in Olema Marsh in mid-October 2008 (KHE 2009a). However, very dramatic changes occurred almost immediately after final removal of the East Pasture levees on October 25, 2008 (KHE 2009a). Within days, much of the East Pasture -- and the very southern portion of the West Pasture -- was seemingly 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 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 restored and fully evolved conditions was not quite so great in the West Pasture, where subtidal extent predicted under fully evolved conditions (2.2 acres; KHE 2006) was only slightly lower than actual (7.4 acres; NPS, unpub. data). 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 only a 1- to at most 3- foot difference in tidal water elevation. These results suggested that drainage was being constrained by the larger volume of water that flowed into the newly restored wetland on a spring tide, when high tides are very high, than on a neap tide, when high tides are lower.

Circulation and drainage patterns are expected to be further altered in the future by changes in Lagunitas Creek (and interior tidal channel) geometry. Immediate post-project surveys had indicated a uniform increase (1.0 ft) 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). Since restoration, elevations within the Lagunitas Creek cross-sections have not changed appreciably, with the exceptions of shoals at channel outlets (KHE 2011a). In 2009, ebb shoals or gravel bars developed at the mouth or downstream of the mouth of the newly constructed channels draining the East Pasture, with accretion during the first year totaling more than 1 to 2 feet (KHE 2009b). These deltaic-type shoals had encroached into the mainstem

Lagunitas Creek channel, reducing the cross-sectional flow area, although they did not span the full width of the channel (KHE 2009b). While both of these shoals rapidly formed after restoration, their evolutionary paths have diverged somewhat. The horseshoe-shaped Tomasini Slough outlet shoal has remained relatively consistent in elevation between 2010 and 2011. It is comprised of an inner and outer shoal that range in elevation from 1- to 2 feet NAVD88 (KHE 2011a). Conversely, the shoal at the mouth of the new side channel off Lagunitas Creek has continued to accrete or build in elevation with estimated deposition rates of 0.7 feet in WY 2010 and 0.75 feet in WY 2011 (KHE 2011a).

While elevations may have increased at the mouth, both of the newly constructed channels had actually deepened since restoration was completed. By 2010, the downstream portions of Tomasini Slough had decreased in elevation relative to constructed elevations by as much as 1.8 feet, with an additional drop of 0.75 feet during the next year (KHE 2011a). The one upstream station with historic data showed little elevation chance since pre-restoration conditions (KHE 2011a). A similar pattern of channel incision occurred at the newly created side channel off Lagunitas Creek in the East Pasture. At least 1 foot of both channel deepening and widening took place in downstream portions of this small tidal creek, while upstream portions widened, but actually became more shallow through deposition of approximately 1 foot of sediment (KHE 2011a). Unconstructed channels are also becoming deeper: these are naturally developing channels on the marsh floodplain. Unfortunately, the lack of vegetation, particularly in the northern portion of the East Pasture, may slow down this process somewhat by encouraging overflow of tidal waters and floodwaters onto the marshplain rather than keeping them in channels (KHE 2010a).

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 from 13.5 mm in the West Pasture to 19 mm in the East Pasture and sediment deposition rates ranging from 4 mm to 4.5 mm in the East and West Pastures, respectively (Parsons and Ryan, *in prep*.). In 2011, the trends in total elevation gain relative to pre-restoration conditions reversed somewhat, with increases since 2008 higher in the West Pasture (11.3 mm) than in the East Pasture (9.0 mm), and, at least for the East Pasture, total elevation gain was appreciably lower in 2011 than 2010 (Parsons and Ryan, *in prep*.). Interestingly, some of the other sampling sites that had not appeared to gain in elevation since 2008 such as the southern end of the Undiked Marsh directly adjacent to the Giacomini Wetlands and the western end of Walker Creek Marsh in the northern end of Tomales Bay appeared to have positive elevation increases for the first time in 2011 (Parsons and Ryan, *in prep*.). Most of the elevation gains in the restored wetlands appear to result from changes in subsurface processes, with reintroduction of tides potentially increasing porewater volume in the soils and slowing down subsurface oxidation rates of organic matter (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 the first winter being a dry one, sediment inputs from the upper watershed were probably minimal, particularly as there were no overbank flooding events. While the 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. Some overtopping did occur during the winter of Year 3, when rainfall totals were even higher than Year 2, but there was only one very brief event. Even during large storms, most of the peak flood flow and sediment generated are trapped by upstream dams, reducing flood volume and sediment loading to downstream areas. Despite this lack of overbank flooding, sedimentation monitoring has shown that sediment was still deposited on Project Area marshplains during the last three years (Parsons and Ryan, *in prep*.). While much of this sediment may derive from re-working of Project

Area soils, deposition rates (5.9 – 7.6 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 in the downstream portions of the Tomasini Slough and the Lagunitas Creek side channel. Should flood flows continue to be reduced, shoals such as described above 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 continue to increase in size to accommodate flood flows, and new tidal channels will develop, increasing exchange between the restored wetland and Lagunitas Creek and creating 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.

This evolution appears to be already well underway. Hydrologic data suggested that the marsh was draining slightly faster during outgoing or ebb flows in 2009 than 2008 (KHE 2010a), and drainage improved slightly again between 2009 and 2011, at least during spring tides (KHE 2011b). Low tide elevations continue to be constrained as they were prior to restoration by the presence of gravel and sand bars at the mouths of creeks, which keep water levels at about 2.0 feet NAVD88 (KHE 2011b). This improvement in drainage efficiency can be seen in the dramatic declines over the last few years in the extent of subtidal areas during an extreme low tide. Acreage declined from 109.4 acres in the East Pasture immediately after restoration to 68.1 acres in summer 2010 under approximately equivalent tide conditions (-0.44 to -1.74 ft MLLW in 2008 vs. -1.54 to -1.67 ft MLLW in 2010, Figure 2). This represents a 38% decrease in extent of permanent inundation during extreme low tides within two years.

In 2011, this trend appears to continue (Figure 2): acreage of subtidal areas in the East Pasture dropped to 51.0 acres, even though water levels may have been influenced somewhat by the unusual rainfall pattern in WY 2011, where precipitation extended well into the summer. Summer stream discharge flows averaged 10 cfs compared to the median estimate of 6 cfs (Figure 1), which may have kept the marsh from fully draining during low tide events. This situation demonstrates that ecosystem evolution following restoration is not a linear process, but can occur in distinct stages or phases that involve triggering or exceeding thresholds before the wetland moves into the next evolutionary stage or phase.

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 parts per thousand (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. Median salinities climbed from 1.6 ppt during Pre-Restoration and 4.2 ppt during Passive Restoration to 11.1 ppt during Full Restoration (Mood Median Test, df=2, Chi-Square=60.5, P<0.0001). Average salinities were 6.9 (± 0.3; SE) ppt during Pre-Restoration, 8.5 (± 0.9; SE) ppt during Passive Restoration, and 12.1 (± 0.6; SE) ppt after restoration. Based on the fact that average salinities seemed higher than median ones, at least for Pre- and Passive-Restoration, these periods were characterized more by episodic instances or periods of higher salinity than was Full Restoration.

Median salinities jumped during the first year after restoration (16.6), but dropped during Year 2 (4.3 ppt), climbing slightly back up in Year 3 (8.7 ppt; Mood Median Test, df=4, Chi-Square = 73.9, P<0.0001). With the exception of Year 3, average salinities appeared to be slightly higher than medians, suggesting that either some higher salinity 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 and 9.43 (± 0.7; SE) ppt in Year 3. The unusual pattern of high and prolonged rainfall in 2011 may have skewed average salinities in the opposite direction from previous years. 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, 2008, although the West Pasture levee removal and Olema Marsh restoration components completed two weeks earlier appeared to have no immediate discernible 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 practical salinity units (psu; psu~=ppt) 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 psu (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 intra-annual and inter-annual 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) or in 2010-2011 (9.43 ppt), when rainfall was higher and more evenly distributed. 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 a similar pattern of salinity changes 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, df=4, Chi Square=35.0, P<0.0001). In Year 2, salinities in Reference Areas dropped back down to a median of 21.0 ppt and, in Year 3, dropped even further to a median of 18.7 ppt. Average salinities in Reference Areas during these sampling periods were 22.0 ± 0.7 (SE) ppt (Pre-Restoration); 20.4 ± 1.2 (SE) ppt (Passive Restoration); 29.3 ± 1.0 (SE) ppt (Year 1-Full Restoration); 19.1 ± 1.5 (SE) ppt (Year 2-Full Restoration), and 17.6 ± 1.2 (SE) ppt (Year 3-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 Areas sampled following restoration are either at the opposite end of the estuary near the estuary's mouth or in a completely different watershed and were unlikely to have been substantially affected by restoration activities.

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, 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 dropped by 6% within the Project Area Project Area from 15.1 degrees Centigrade Pre-Restoration to 14.2 degrees Centigrade during Passive Restoration and 14.1 degrees Centigrade during Full Restoration (Mood Median Test, df=2, Chi-Square =9.64, P=0.008; Figure 4). During Full Restoration, median temperatures varied widely from a high of 15.4 degrees Centigrade in Year 1 to a low of 12.9 degrees Centigrade in Year 2, with Year 3 falling in-between (14.2 degrees Centigrade; Mood Median Test, df=4, Chi-Square=20.68, $P<0.0001$). Average temperatures appeared slightly higher most years than median ones, ranging from 13.6 ± 0.3 (SE) degrees Centigrade in Year 2 to 16.6 ± 0.5 (SE) degrees Centigrade in Year 1. Temperatures in Year 3 within the restored wetland averaged 15.1 ± 0.5 (SE) degrees Centigrade.

Comparatively, mean temperatures in Reference Areas also decreased with time, dropping approximately 11% from 17.2 ± 0.3 (SE) degrees Centigrade Pre-Restoration to 15.7 ± 0.5 (SE) degrees Centigrade during Passive Restoration and 15.3 ± 0.4 (SE) degrees Centigrade during Full Restoration (GLM, df=2, F=8.82, $P<0.0001$). Temperatures averaged 16.1 ± 0.7 (SE) degrees Centigrade during Year 1, 14.4 ± 0.5 (SE) degrees Centigrade during Year 2, and 15.5 ± 0.7 (SE) degrees Centigrade during Year 3 of the Full Restoration period. Median temperatures differed significantly between treatment years, ranging from 17.3 degrees Centigrade during Pre-Restoration to 13.5 degrees Centigrade in Year 3 (Mood Median, df=4, Chi-Square=10.96, P=0.027).

In general, trends within the Reference Areas appeared to parallel those of the Project Area, even if the Project Area would be expected to have slightly different temperature regimes due to the greater influence of colder freshwater creeks and groundwater at the southern end of Tomales Bay. Temperatures declined during the monitoring period, although temperatures climbed slightly during Year 1 post-restoration. Higher rainfall – and more freshwater inflow from watershed sources – may have led to colder water temperatures in both the Project and Reference Areas during Years 2 and 3.

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. In Year 3, the restored wetlands exceeded the lethal limit during approximately 8% of the sampling events and the sub-optimal during approximately 17% of the sampling events, compared to 14% and 17%, respectively, in the 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 3 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, and substantial groundwater inflow. However, despite this, median temperatures after restoration during Years 1 through 3 did not differ significantly between the Project Area (median = 13.5 degrees Centigrade) and Reference Areas (median=13.7 degrees Centigrade; Mood Median, df=1, Chi-Square=0.04, P=0.84).

While these results would suggest that the Project Area is converging with conditions in the Reference Areas, 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, 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).

While introduction of full tidal flows to the Project Area might have been expected to boost pH, 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 7.22 during Full Restoration (Mood Median, df=2, Chi-Square=123.98, P<0.0001). Median pH during the first three years after restoration was 7.25 during Year 1; 7.14 during Year 2; and 7.26 during Year 3, with these three years and Passive Restoration appearing to differ primarily from Pre-Restoration (Mood Median, df=4, Chi-Square=128.17, P<0.0001).

The increase in tidal exchange and decrease in water residence time after levees were removed 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 or low-water periods, with soluble acids from oxidation then released into overlying waters when tidal exchange is reintroduced. The Project Area was deliberately dried out before and during construction to improve constructability conditions, resulting in even drier conditions than when the Project Area was ranched.

Interestingly, a similar seemingly slight, but significant, decrease was also observed in Reference Areas. In Reference Areas, pH dropped from a median of 7.62 during the Pre-Restoration and Passive Restoration sampling periods to 7.34 during the Full Restoration sampling period (Mood Median, df=2, Chi-Square=36.46, $P<0.0001$). Median pH following restoration in Reference Areas was 7.44 during Year 1; 7.28 during Year 2; and 7.32 during Year 3, with Year 2 differing significantly from Pre- and Passive Restoration periods (Mood Median, df=4, Chi-Square=38.63, $P<0.0001$). These changes have now led for pH to weakly differ between the Project Area and Reference Areas (Mood Median, df=1, Chi-Square=3.33, $P=0.068$).

While the overall median between Pre-Restoration and Passive Restoration periods for Reference Areas remained consistent at 7.62, slight variation did occur between years, ranging from 7.53 in WY 2005 to 7.8 in WY 2001, although the latter monitoring year did not include the Undiked Marsh. 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. Starting in WY 2007, prior to restoration, the median for Reference Areas dipped down to 7.50 and has dropped slightly every year since. The median pH appears to have dropped most 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. During Pre-Restoration and Year 3 periods, the median pH averaged 7.48 and 7.25, respectively, in the Undiked Marsh; 7.80 and 7.10, respectively, in Limantour Marsh, and 7.70 and 7.65, respectively, for Walker Creek Marsh. From a historical perspective, pHs in Limantour Marsh and the Undiked Marsh appeared relatively stable until WY 2007, when values began to seemingly decline. In contrast, Walker Creek Marsh has maintained relatively consistent pH levels except for Year 2 post-restoration: Year 2 results may represent more year-to-year variation than a potential trend indicator.

The seeming downward trend in median pHs in the Undiked Marsh and Limantour Marsh since WY 2007 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 have declined 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 (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 CO^2 from the atmosphere into estuarine waters (A. Russell, UCDavis, *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 of oceans even without the influence of climate change.

Dissolved Oxygen

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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 (ibid).

Prior to restoration, oxygen levels in the East Pasture averaged 4.98 ± 0.24 (SE) 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 mg/L; ibid). Median oxygen concentrations in other Project Area sampling locations – 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 to 8.39 mg/L during Passive Restoration and 8.27 mg/L during Full Restoration (Mood Median, df=2, Chi-Square=14.53, P=0.001). Median oxygen levels were 8.63 mg/L in Year 1; 8.13 mg/L in Year 2; and 8.14 mg/L in Year 3, with the largest differences occurring between Pre-Restoration and Year 1 (Mood Median Test, df=4, Chi-Square=15.19, P=0.004). Average oxygen levels were seemingly slightly different from median levels, ranging from 7.30 ± 0.13 (SE) mg/L during Pre-Restoration to 8.73 ± 0.39 (SE) mg/L in Year 1. Lower D.O. levels occurred in Year 2 (7.79 ± 0.24 (SE) mg/L) than either Year 1 or Year 3 (8.06 ± 0.24 (SE) mg/L): both Years 2 and 3 were quite wet, although Year 2 may have captured more storm events. Cold temperatures and strong flow conditions could suppress biological activity in waters relative to warmer, more quiescent periods.

Oxygen concentrations in the East Pasture jumped 93% from a median of 4.56 mg/L pre-restoration to 8.80 mg/L after restoration (Mood Median, df=2, Chi-Square=86.63, P<0.0001). After restoration, median oxygen levels varied somewhat from 9.85 mg/L during Year 1 to 8.83 mg/L during Year 2,and 7.96 during Year 3, with strong differences between Pre-Restoration and Passive and all post-restoration years (Mood Median Test, df=4, Chi-Square=87.89, P<0.0001; Figure 6). In terms of mean oxygen levels, concentrations in the East Pasture averaged 4.98 ± 0.24 (SE) mg/L Pre-Restoration to 9.89 ± 0.49 (SE) mg/L during Year 1, 8.27 ± 0.37 (SE) mg/L during Year 2, and 7.74 ± 0.38 (SE) during Year 3. While D.O. also appeared to increase almost 20% in Olema Marsh between pre- and post-restoration conditions, increasing from 5.83 ± 0.38 (SE) mg/L to 6.99 ± 0.52 (SE) mg/L during Full Restoration this change was not statistically significant, perhaps because of the relatively low number of replicates and associated low power (GLM, df=2, F=1.61, P=0.218; 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). After Full Restoration, the number of Basin Plan standard exceedances in the Project Area dropped 43% from 25% to 14.2% in Year 1 and 4% in Year 2, rising slightly to 13.5% again in Year 3. Year 3 exceedances were only slightly higher than Reference Areas (12.5%). Incidences of hypoxia (< 2 mg/L) and anoxia (<0.5 mg/L) in the Project Area totaled 3.2% and 0.1%, respectively, in the Project Area during Year 3, compared to 12.2% and 5.4%, respectively, prior to restoration. This represents a 74% decrease in hypoxic incidences within the Project Area, although numbers did appear slightly higher in Year 3 than Year 1 (2.7% and 0%, respectively) and Year 2 (1.6% and 0%, respectively). .

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, median oxygen levels between these two Study Areas after restoration were equivalent from a statistical perspective: 8.27 mg/L in the Project Area and 8.17 mg/L in Reference Areas (Mood Median Test, df=1, Chi Square=0.19, P=0.67).

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, 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, 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 have been 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, 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, df=3, Chi-Square=35.70, P=<0.0001; Figure 7). Means were also seemingly higher 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, 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).

In Year 3, turbidity levels dropped somewhat relative to Year 2, averaging 21.3 ± 3.6 (SE) NTU, which was seemingly equivalent to average turbidity levels during Pre-Restoration, but higher than Year 1. Median levels in Year 3 (13.6 NTU) did not differ significantly from either Pre-, Passive- or Year 1 (range 10.5-10.7 NTU, Mood Median, df=4, Chi-Square=39.2, P<0.0001), although all differed from Year 2. Overall, because of increases in Year 2, turbidity levels appeared to increase by 51% as a result of restoration (GLM, df=2, F=3.59, P=0.03), with significant differences occurring between Pre- (22.7 ± 2.3 (SE) NTU) and Full-Restoration (34.3 ± 4.5 (SE) NTU; P=0.02).

The very disparate trends in turbidity levels between Years 1 and 3 and Year 2 following restoration may be largely due to very different climatic conditions between these years. In WY 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 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 have better represented the short-term increase in turbidity levels immediately after restoration that was predicted in the environmental compliance analysis documents.

Interestingly, WY 2011 was also wet, but turbidity levels decreased during that year. This may represent an artifact of sampling effort – that is, more samples were taken during storm events in Year 2 (50% of sampling events) than Year 3 (33% of sampling events) – but storm sampling was conducted in Year 3, as well. Also, the rainy season was prolonged in Year 3, increasing the potential to capture turbidity-generating events. Regardless, median turbidity levels did not significantly differ between the Project Area (14.7 NTU) and Reference Areas (12.7 NTU) after restoration (Mood Median, df-1, Chi-Square=2.19, P=0.14), although some individual post-restoration years did show clear differences between Study Areas (see above).

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 (NO_3^-) for the East Pasture and then dropped to below 1.10 mg/L for the other sub-sampling areas (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 marshes 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). For some of the marshes closer to the ocean, higher summer concentrations could potentially also result from inflow of higher nutrient waters as a result of upwelling events in adjacent coastal waters.

As with some of the other parameters, a temporary increase in nitrate levels had been expected with restoration due to decomposition of pasture vegetation and biogeochemical processes in soils. 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 WY 2009, with January, May, and August 2009 sampling results differing significantly from November 2008, and the two February 2009 storm sampling events (MLE, 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, but they climbed again in Year 3 to 1.02 ± 0.12 (SE) mg/l (Wilcoxon, df=4, Chi-Square=39.2, $P<0.0001$; Figure 8). A slightly different trend appeared to occur with estimated medians. Estimated median nitrate values appeared to drop 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 to rise again to 0.38 mg/L in Year 2 and 0.28 mg/L in Year 3 (Figure 8). Overall, restoration has resulted in lower average nitrate levels in the Project Area, with concentrations averaging 0.87 ± 0.22 (SE) mg/L in the restored wetlands compared to 3.22 mg/L for the dairy ranch and 4.52 mg/L for the passively restored Project Area (Wilcoxon, df=2, Chi-Square=38.4, $P<0.0001$). Median concentrations also appeared to drop between restoration phases, ranging from 0.83 mg/L during Pre-Restoration to 0.38 mg/L and 0.26 mg/L during Passive and Full Restoration phases, respectively.

The slight disparity in trends between Year 1 and Years 2 and 3 probably results from the influence of very different climatic conditions and the immediate post-breach nitrate pulse in Year 1. Conditions were wetter in Years 2 and 3, with 50% of the sampling periods in Year 2 and 33% in Year 3 falling during moderate to large storm events, so estimated median nitrate concentrations in Years 2 and 3 were driven up relative to Year 1, which was drier in terms of total rainfall. 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, if not Year 3.

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) and Year 3 (0.34 mg/L and 0.29 mg/L; Wilcoxon, df=4, Chi-Square=67.7, $P<0.0001$; Figure 8). Even though rainfall totals were also high in Year 3, the decrease in storm sampling events between Year 2 (50%) and Year 3 (33%) may have had an effect on results. 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) and 0.49 mg/L and 0.21 mg/L during Full Restoration (Wilcoxon, df=2, Chi-Square=62.0, 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 Year 1, the estimated median nitrate concentration for both marshes was 0.07 mg/L. In Years 2 and 3, 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 NO3) 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 received 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 Years 2 and 3 and 75% of the events in Year 1. Nitrate concentrations at this site in Year 3 ranged from 10.03 mg/L (April 2011) to 42.18 (January 2011). The highest nitrate levels in the Project Area in Year 3 occurred in the Tomasini Triangle Pond, which is a created freshwater marsh that receives the non-point source run-off and septic influenced groundwater from the sampling site described above. Nitrate levels in this pond were 7.02 mg/L in July 2011 and 23.87 in mg/L in January 2011.

Despite these issues, nitrate concentrations between the Project Area and Reference Areas did not significantly differ by Year 3 of Full Restoration (Wilcoxon, df=1, Chi-Square=0.26, P=0.61), although this result may be partially due to high variability in the data. Nitrate concentrations averaged 0.87 mg/L in the Project Area and 0.49 mg/L in Reference Areas during the three years after restoration was implemented. Median concentrations appeared closer, with medians in the Project Area being 0.26 mg/L and those in Reference Areas being 0.21 mg/L.

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 nitrites were only rarely recorded prior to restoration, they 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-detects 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 sub-sampling areas that accounted for no more than 11 % of the detections (Contingency Table, Chi Square, 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, 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 (Contingency, df=4, Chi-Square=14.1, $P=0.01$). In Year 1, detections dropped 43% from 22.8% of the samples Pre-Restoration to 14.0% of the samples (Figure 9). The number of total ammonia detections decreased even more dramatically in Year 2 of Full Restoration, dropping 48% to 6.8% of the samples exceeding detection limits (Figure 9). In Year 3, detections climbed again slightly to 11.1% of the samples, but were still seemingly lower than Pre-Restoration and even potentially Year 1. Interestingly, the number of detections was lowest during Passive Restoration (4.6%).

Estimated mean total ammonia concentrations within the Project Area did appear to drop 73%-88% after restoration from 1.26 ± 0.58 (SE) mg/L Pre-Restoration to 0.34 ± 0.10 (SE) mg/L during Year 1, 0.15 ± 0.06 (SE) mg/L during Year 2, and 0.52 ± 0.24 (SE) mg/L during Year 3. High variability may have reduced power of analysis, as seeming differences between restoration phases were not statistically significant (Wilcoxon, df=4, Chi-Square=7.04, $P=0.13$). Dissolved ammonia samples were only collected during the Passive and Full Restoration phases, but they have much lower detection limits than total ammonia samples. There are some similarities with estimated total ammonia, although concentrations, overall, were much lower. As a point of comparison, median dissolved ammonia levels in the Project Area during Passive Restoration were 0.05 mg/L: they rose to 0.09 mg/L during Year 1 and then fell to 0.06 mg/L in Year 2 and 0.05 mg/L in Year 3. However, these differences were not statistically significant (Mood Median, df=3, Chi-Square=3.9, $P=0.28$).

Estimated East Pasture mean total 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 in Year 2, a decrease of 83% and 91%, respectively, from Pre-Restoration levels (Wilcoxon, df=4, Chi-Square=14.7, $P=0.005$). In Year 3, however, estimated concentrations in the East Pasture jumped back up, averaging 1.17 ± 0.66 (SE) mg/L. Concentrations in Year 3 in the Project Area and in the East Pasture appeared to have been largely skewed by two very high values in one of the newly created Tomasini Slough side channels – 8.70 mg/L in October 2010 and 6.60 mg/L in April 2011. This is reflected in the fact that estimated median total ammonia concentrations in the East Pasture in Year 3 were much lower than average ones (0.18 mg/L).

The increase in ammonia detections between Passive and Full Restoration periods – and betweens Years 1-2 and Year 3 -- 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 Year 1 or WY 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 rose again in Year 3 to 14.3%, although these differences were not statistically significant (Contingency, df=4, Chi-Square=5.10, P=0.28). Despite the potential increase in ammonia detections during Year 1, median dissolved ammonia concentrations in Reference Areas appeared roughly equivalent between Passive Restoration (0.10 mg/L) and Year 1 post-restoration (0.10 mg/L), dropped in Year 2 (0.05 mg/l), and rose again in and Year 3 (0.09 mg/L), but those differences were not statistically significant (Mood Median, df=3, Chi-Square=4.75, P=0.19).

The overall increase in Years 1 and 3 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 after the Giacomini Wetlands were restored do not entirely result from 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 WY 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 Years 2 and 3, wetter conditions drove down salinities below Pre-Restoration median levels by as much as 9-11 ppt, so higher ammonia detection frequencies in Years 2 and 3 relative to Passive Restoration periods are harder to explain.

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 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 promote dissociation of ammonia into its unionized ion.

In general, ammonia detection frequencies between the Project Area (10.6%) and Reference Areas (10.0%) in Years 1 -3 of Full Restoration showed no statistically significant differences (Contingency, df=1, Chi-Square=0.021, P=0.88). Dissolved ammonia concentrations also did not differ significantly between these Study Areas after restoration (Mood Median, df=1, Chi-Square=0.08, P=0.78), with medians being 0.07 for both the Project Area and Reference Areas. 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 and Phosphorous

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-Square=9.29, p=0.010), with the number of detections disproportionately higher in the Project Area than in the other areas (ibid). Total dissolved 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 total dissolved phosphates were detected Pre-Restoration (26%; Chi-Square Test, n=51, df=4, Chi-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 sub-sampling 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 dropped significantly, decreasing from 0.99 ± 0.16 (SE) mg/L during Pre-Restoration to 0.68 ± 0.37 (SE) mg/L during Passive Restoration and 0.12 ± 0.16 (SE) mg/L during Full Restoration (Wilcoxon, df=2, Chi-Square=59.0, P<0.0001). Estimated mean phosphate concentrations in the Project Area ranged only slightly after restoration from 0.10 ± 0.03 (SE) mg/L during Year 3 to 0.12 ±

0.02 (SE) mg/L during Year 1 of Full Restoration. Estimated levels after levee removal were even lower than those during Passive Restoration (average=0.68 ± 0.37 (SE) mg/L).

Estimated phosphate concentrations in the East Pasture dropped from 2.40 ± 0.33 (SE) mg/L during Pre-Restoration to 1.69 ± 0.98 (SE) mg/L during Passive Restoration and 0.11 ± 0.16 (SE) mg/L during Full Restoration (Wilcoxon, df=2, Chi-Square=80.0, P<0.0001). Concentrations averaged 0.11 ± 0.02 (SE) mg/L in Year 1; 0.10 ± 0.02 (SE) mg/L in Year 2; and 0.13 ± 0.06 (SE) mg/L in Year 3. 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.

Total phosphorous levels were also assessed within Study Areas. Total phosphorous incorporates both free and bound forms of phosphorous, unlike phosphates, which are only in dissolved form. In general, total detections of phosphorous in Project Area waters decreased from 62.9% during Passive-Restoration to 39.5 – 41.0% in Years 1 and 2, climbing back up to 68.9% in Year 3 (Contingency, df=3, Chi-Square=11.8, P=0.008; Figure 10). Interestingly, a similar pattern occurred in Reference Areas (Figure 10). Total phosphorus was detected during approximately 75.6% of the sampling events during Passive Restoration, with detections falling to between 37.9- 48.3% in Years 1 and 2 and then climbing back up to 75.0% in Year 3 (Contingency, df=3, Chi-Square=14.6, P=0.002).

Estimated total phosphorous concentrations in the Project Area dropped from 0.99 ± 0.13 (SE) mg/L during Passive Restoration to between 0.14 ± 0.01 (SE) mg/L in Year 2 and 0.22 ± 0.06 (SE) mg/L in Year 1 (Wilcoxon, df=3, Chi-Square=11.9, P=0.008). In Reference Areas, estimated phosphorous concentrations also dropped between Passive Restoration (0.18 ± 0.03 (SE) mg/L) and Years 1 (0.05 ± 0.04 (SE) mg/L) and 2 (0.08 ± 0.03 (SE) mg/L), but climbed back up in Year 3 (0.18 ± 0.06 (SE) mg/L; MLE, df=3, Chi-Square=15.1, 0.001<P<0.01). While total phosphorous might be expected to increase during storm events, when sediment loads are greatest, this pattern was not necessarily reflected in the data, which showed pulses in total phosphorous during low flow, as well as high flow, sampling events. A comparison of total phosphorous and orthophosphate data for Year 3 alone showed very low correlation between these two phosphorous parameters (Pearson Correlation=0.016, P=0.88).

For the first year since restoration, median phosphate concentrations did not differ between the Project Area (0.08 mg/L) and the Reference Areas (0.07 mg/L; Mood Median, df=1, Chi-Square=0.80, P=0.37). In addition, total phosphorous detections during post-restoration between the Project Area (50.0%) and Reference Areas (53.5%) did not differ significantly (Contingency, df=1, Chi-Square = 0.254, P=0.614). Estimated total phosphorous concentrations also appeared similar between the Study Areas, averaging 0.18 ± 0.02 (SE) mg/L for the Project Area and 0.17 ± 0.02 (SE) mg/L for Reference Areas (Wilcoxon, df=1, Chi-Square=0.14, P=0.71). Therefore, the Project Area appears to converging further with conditions in with Reference Areas, at least in terms of phosphate and total phosphorous levels, however, more data will be needed to make any definitive conclusions, particularly as the restored marsh is still actively evolving. 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 continuous 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, 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 sub-sampling 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 93% from 1,600.9 mpn/100 ml during Pre-Restoration to 120.11 mpn/ml during Full Restoration (MLE, df=2, Chi-Square=76.4, P<<0.0001; Figure 11). Estimated median concentrations during Passive Restoration fell in-between those of Pre- and Full Restoration (919.9 mpn/100 ml) and varied significantly from Full Restoration (P<0.0001; Figure 11). With the exception of Pre-Restoration (P=0.17), all treatment phases or years differed significantly from Passive Restoration (MLE, df=4, Chi-Square=77.5, P<0.0001; all P<0.0001). Coliform levels, which had already declined somewhat during Passive Restoration, dropped sharply again immediately after restoration in Year 1 (median=90.4 mpn/ml). Median concentrations appeared to climb slightly in both Years 2 and 3 to 141.3 and 134.1 mpn/100 ml, respectively, but this increase was not statistically significant relative to Year 1 (P>0.10).

Not surprisingly, large decreases were recorded in the once heavily managed East Pasture, with estimated median levels dropping more than 78-98% from 6,298.8 mpn/100 ml during Pre-Restoration to 1,385.3 mpn/100 ml during Passive Restoration and to 100.6 mpn/100 ml during Full Restoration (MLE, df=2, Chi-Square=49.9, P<0.0001). After restoration, levels in the East Pasture varied from 64.1 mpn/100 ml in Year 1 to 119.7 mpn/100 ml in Year 2 and 135.1 mpn/100 ml in Year 3. Estimated mean concentrations in the West Pasture also dropped by more than 95% from 1,131.7 mpn/100 ml Pre-Restoration to 655.4 mpn/100 ml during Passive Restoration and to 83.1 mpn/100 ml during Full Restoration (MLE, df=2, Chi-Square=30.5, P<0.0001). After restoration, levels in the West Pasture have averaged 48.7 mpn/100 ml in Year 1; 88.2 mpn/100 ml in Year 2; and 95.2 mpn/100 ml in Year 3.

Decreases in Olema Marsh also appeared substantial after restoration. Concentrations in Bear Valley Creek, which flows through Olema Marsh, at the downstream boundary of the Project Area appeared to rise from 1,821.4 mpn/100 ml during Pre-Restoration to 2,048.52 mpn/100 ml during Passive Restoration and then fall to 413.9 during Full Restoration, although these differences were not strongly significant (MLE, df=2, Chi-Square=5.34, 0.10>P>0.05), No actions were taken in Olema Marsh during Passive Restoration, so this phase represents more pre-restoration for this particular study area, and restoration actions overall were not quite as extensive in this area as in Giacomini.

Concentrations in downstream Lagunitas Creek after restoration were seemingly lower (196.8 mpn/100 ml) than those recorded before restoration (median=356.9 mpn/100 ml) and considerably lower than levels during restoration (769.3 mpn/100 ml), however, these differences were not statistically significant (MLE, df=2, Chi-Square=1.94, P>0.20). In Year 1, coliform levels in downstream areas of Lagunitas Creek dropped to 141.5 mpn/100 ml, but rose in Year 2 to 341.8 mpn/100 ml, only to drop again in Year 3 to 157.7 mpn/100 ml.

Estimated median levels in Reference Areas were lower in WY 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, df=3, Chi-Square=7.56, 0.10>p>0.05). This may suggest that lower concentrations in all Study Areas in WY 2009 might 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 2010, coliform levels generally increased in both the Project Area and in the Reference Areas, with Reference Areas reaching almost pre-restoration levels of 70.6 mpn/100 ml. WY 2010 was much wetter than WY 2009, and almost 50% of the sampling events just happened to occur during storms. While Year 3 was wet, too, only 33% of the sampling events captured storm events. During Year 3, coliform levels in Reference Areas dropped down again to 46.1 mpn/100 ml, while those in the Project Area, as noted earlier, remained similar to Year 2.

The dramatic declines in fecal coliform concentrations in the Project Area following even passive restoration and high rainfall 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 slightly from 95% of all samples collected in the Project Area Pre-Restoration to between 80% and 89% of all samples collected during Years 1 – 3 of Full Restoration, respectively. Approximately 39% of samples from Year 1; 44% of samples from Year 2; and 40% of samples from Year 3 exceeded the contact water recreation standards of 200 mpn/100 ml, compared to approximately 78% Pre-Restoration, at least a 43% decrease. In Year 3, only 2% to 4% 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. In comparison, the frequency of exceedance of contact water recreation standards (200 mpn/100 ml) in Reference Areas appeared to decrease from 34% during Pre-Restoration to 9% in Year 1, 28% in Year 2, and 17% in Year 3.

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, 71% of the time in

Year 2, and 65% of the time in Year 3, 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, 30% of the sampling periods in Year 2, and 56% of the sampling periods in Year 3 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; 98% during Year 2; and 77% during Year 3. 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. In Year 3, these levels were estimated at 446.9 mpn/100 ml and 6,924.7 mpn/100 ml. With the exception of Year 3's 90th percentile, all of these estimated concentrations appear 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.

Given the somewhat similar patterns in fecal coliform levels observed between the Project Area 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 Years 2 and 3 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 2009 Water Year. Conversely, the fact that more than 50 % of the sampling events in the WY 2010 coincided with storm events undoubtedly influenced Year 2 results. Year 3 was also wet, but, based on some of the fecal coliform results, the distribution of precipitation across the rainy season might have created different mobilization patterns than what occurred in Year 2. In addition, only 33% of the storm events were captured in Year 3, which can strongly influence results.

However, despite some similarity in trends between the Project Area and Reference Areas, estimated median coliform levels during Full Restoration differed between these two (MLE, df=2, Chi-Square=23.6, P<0.0001), with medians being 120.1 mpn/100 ml for the Project Area and 43.6 mpn/100 ml for Reference Areas. Again, as the Project Area receives more surface water and groundwater (influenced by septic) than Reference Areas, these two Study Areas may never totally converge for this particular parameter.

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 very little 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 Giacominis 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, nitrate 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*u*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 increased from 57.5 mpn/s during Pre-Restoration to 98.3 mpn/s in Year 1 and even higher to 291.2 mpn/s in Year 2 and 306.5 mpn/s in Year 3, although differences between these restoration phases or years were not strongly significant (MLE, df=4, Chi Square = 8.81, 0.10>p>0.05; Figure 13). Conversely, estimated arithmetic mean loading rates, which were higher than median values, seemingly decreased from 249,389 ± 369,023 (SE) mpn/s during Pre-Restoration to 52,716.9 ± 74,464.5 (SE) mpn/s during Year 1, 33,657.5 ± 37,936.7(SE) mpn/s during Year 2, and 34,036 ± 41,753.4 (SE) mpn/s during Year 3 (Figure 13). This suggests that peak loading during high flow events may have dropped relative to Pre-Restoration conditions, but the average loading rate for fecal coliform during non-storm flows has apparently increased.

Trends were slightly different in Reference Areas, except for perhaps Years 2 and 3. For Reference Areas, estimated median fecal coliform loading rates dropped from 98.3 mpn/s during Pre-Restoration to 23.3 mpn/s during Passive Restoration before climbing to 52.5 mpn/s during Full Restoration (MLE, df=2, Chi-Square=6.02, 0.05>P>0.02), During the three years after restoration, median loading rates in Reference Areas were17.7 mpn/s in Year 1; 86.1 mpn/s in Year 2; and 75.1 mpn/s in Year 3, although only differences between Pre- and Passive Restoration phases were significant (Z-test: P=0.015, MLE, df=4, Chi-Square=11.3, 0.05>P>0.02)

For nitrates, loading increased between Pre- and Full Restoration, with estimated means climbing from 0.54 ± 0.35 (SE) mg/s prior to restoration to 0.82 ± 0.28 (SE) mg/s in Year 1, 0.91 ± 0.28 (SE) mg/s in Year 2, and 0.57 ± 0.23 (SE) mg/s in Year 3 (Wilcoxon, df=4, Chi-Square=26.6, P<0.0001; Figure 12). Conversely, nitrate loading within Reference Areas climbed from 0.51 ± 0.11 (SE) mg/s Pre-Restoration to 0.84 ± 0.70 (SE) mg/s during Passive Restoration and 0.70 ± 0.51 (SE) during Full Restoration (Wilcoxon, df=2, Chi-Square=20.5, P<0.001). Nitrate loading in Reference Areas was low during Year 1 (0.09 ± 0.04 (SE) mg/s), but then climbed sharply in Year 2 (1.63 ± 1.30 (SE) mg/s) and fell again in Year 3 (0.14 ± 0.06 (SE) mg/s; Wilcoxon, df=4, Chi-Square=28.7, p<0.0001; Figure 12).

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 and then fell again in Year 3, which also wet, but which only captured 33% of the sampling events.

Ultimately, as discussed in earlier sections, 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, although it also has its pollution issues. However, despite these factors, conditions appear to becoming more similar between Study Areas. During the first three years after restoration, median fecal coliform loading rates in the Project Area (201 mpn/s) were actually lower than those in Reference Areas (596.2 mpn/s; MLE, df=1, Chi Square=11.6, 0.001>P>0.0001). Nitrate loading rates did not differ significantly between the restored Project Area (0.78 ± 0.21 (SE) mg/s) and Reference Areas (0.70 ± 0.51 (SE) mg/s; Wilcoxon, df=1, Chi-Square=1.79, P=0.182).

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.

While this type of analysis is important to understanding wetland health and restoration success, obtaining an accurate understanding is confounded by a number of factors and may not be possible using our sampling approach. Water quality data is highly variable, and this high variability, coupled with low sample size, may largely obscure any upstream-downstream patterns. Also, almost none of the creeks or water bodies, including Lagunitas, Fish Hatchery, Bear Valley, and even those in the East Pasture, are not what would be considered "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. This type of analysis requires that all sources of surface water inflow be accurately accounted for to reliably estimate both inputs and outputs.

Lastly, one grab sample may not accurately capture the cross-sectional loading profile, particularly for wider creeks.

Some preliminary analysis of downstream reductions in pollutants was conducted for nitrate and fecal coliform loading rates prior to restoration (Parsons 2009). Fecal coliform concentrations and loading showed no statistically significant pattern of downstream reductions for any of the Project Area creeks, although there was high variability in the data (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 prior to 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).

Following restoration in 2008, the Giacomini Wetlands began a new process 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 after 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 strongly significant (MLE, df=1, 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 appeared lower However, as with nitrate loading, none of these differences were statistically significant (Paired t-tests and Mann-Whitney, all p>0.116).

Drawing definitive conclusions from these data are difficult, given the issues discussed above. For those reasons, further analyses of upstream-downstream loading were not conducted in Year 3. A truly valid understanding of the newly restored wetland's role in improving downstream water quality will require a more intensive, research-type approach with sampling of all input sources at multiple locations in the channel during storm events of varying magnitudes. What conclusion can be drawn from the preliminary data analysis is that the Giacomini Wetlands is a very complicated hydrologic system, and it would be difficult even with a research-type approach to tease out how much pollutant reduction the restored wetlands are responsible for given the numerous surface water, groundwater, and non-point source discharges into the marsh.

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 have not materialized, although turbidity levels did increase during Year 2.

In Year 3, many parameters actually showed strong similarity with Year 1, despite the fact that climatic conditions in Year 3 more closely resembled those in Year 2. Levels of turbidity, dissolved oxygen, pH, nitrates, and nitrate loading were all more similar to those in Year 1 than in Year 2. For some parameters such as salinity, temperature, and ammonia detections, conditions in Year 3 appeared to be intermediate between Year 1 and Year 2. Levels of total dissolved phosphates and fecal coliform appeared roughly equivalent between all post-restoration years, while related parameters such as total phosphorous and fecal coliform loading were actually higher in Year 3 than other previous restoration years.

Initially, some of these lower-than-expected temporary impacts may have resulted from the fact that Year 1 or WY 2009 was a dry year, and few large storms occurred that would have contributed to reworking of this evolving landscape, even though we captured some of the few larger storm events that did occur. However, Year 3 had higher-than-average rainfall totals, and, as noted, levels of many parameters such as salinity, temperature, turbidity, nitrates, and nitrate loading, as well as ammonia detections, were lower in Year 3 than Year 2, which was also a wet year. Some of the differences between Year 2 and Year 3 may relate to the fact that fewer storm events were sampled in Year 3 (33% of events occurred during storm event) compared to Year 2 (50%). Also, differences in the pattern of rainfall may influence water quality results: the rainy season was prolonged in Year 3 relative to most years, stretching into June.

Some of these analyses are undoubtedly affected by the fact that these systems are not closed and receive inputs of nitrates and fecal coliform 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 Years 1-3. 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.

With a few exceptions, water quality parameters have shown significant, positive improvements in conditions between Pre- and Full-Restoration phases. Dissolved oxygen levels increased, while nitrate, ammonia, phosphate, phosphorous, and fecal coliform levels decreased, with some of these parameters falling quite substantially. As might be expected, salinity in the Project Area climbed relative to the diked dairy ranch conditions, while temperatures dropped, probably because residence time of waters decreased, and exchange with other water bodies increased, even if these waters are potentially warmer. Unexpectedly, overall pH within the restored wetland also declined, despite the fact that higher pH tidal waters are now entering the Project Area. A few parameters did display negative trends. Turbidity, as discussed above, was expected to increase as part of the restoration process, and it did, but perhaps not to the degree anticipated. As would be expected, loading rates of pathogens and 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).

In addition to a significant change relative to Pre-Restoration conditions, the other way in which the restored wetland can be evaluated in terms of water quality improvement is the degree of similarity in water quality conditions between the Project Area and Reference Areas. After only a short time (three years), most of the parameters, including temperature, dissolved oxygen, ammonia, phosphates, phosphorous, and seemingly nitrates were actually statistically equivalent to Reference Areas (Table 2). Even turbidity and nitrate loading levels in the Project Area showed no statistical difference with those in Reference Areas, despite the fact that they had increased relative to Pre-Restoration conditions in the Project Area. Fecal coliform loading rates post-restoration were actually lower than Reference Areas, even though they had increased in the Project Area relative to Pre-Restoration rates. The only parameter that continued to differ negatively from Reference Areas was fecal coliform, even though levels had dropped substantially from Pre-Restoration conditions. The pH levels in the restored wetland also continue to be lower than tidally influenced reference marshes.

Table 2. Similarity between the Project Area and Reference Areas (natural marshes) in water quality variables before and after restoration and changes in parameters within the Project Area before and after restoration. CV refers to coefficient of variation, with the best comparison variables having a comparatively low CV (0.2 or less) in natural systems.

Parameter	Similarity to natural marshes before restoration	CV of natural marshes	Change in Project Area since restoration	Similarity to natural marshes after restoration
Salinity (ppt)	≠	0.54	↑	≠**
Dissolved oxygen (mg/L)	≠	0.37	↑	=
p.H.	=	0.06	↓	≠
Temperature (° C)	≠	0.31	↓	=
Nitrates	≈	0.78	↓	=
Ammonia detections	=	0.54	↓	=
Phosphates	≠	0.61	↓	=
Fecal Coliform	≠	>>1.0	↓	≠
Turbidity	=	>>1.0	↑	=

** Not evaluated statistically

In addition to the effects of restoration on the Project Area, our monitoring has also revealed some other interesting trends that may pertain to 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.

One possible explanation for the increase in ammonia detections in Year 1 may have been 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 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. However, in Year 3, detection frequencies increased again in both the Project Area and Reference Areas, even though Year 3 was also a wet year, and salinities were still lower than Year 1, although higher than Year 2.

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 decreased every year in the Project Area from 7.60 to 7.30 during Passive Restoration and 7.22 during Full Restoration 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 7.34 after restoration. Median pH declined more sharply in the Undiked Marsh and Limantour Marsh than in Walker Creek Marsh.

The dramatic decreases in median pH in the Undiked Marsh and Limantour Marsh since WY 2007 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 UCDavis 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, UCDavis, *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 CO^2 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 success fullness 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 Area 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 fourth 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 have improved 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 – NO_3 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 11). This sampling site continues to show elevated nitrates even after restoration and exceeded 10 mg/L during every sampling event in Years 2 and 3 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).

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

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

References

Chambers, R. M., S. V. Smith, et al. (1994b). An ecosystem-level context for tidal exchange studies in salt marshes of Tomales Bay, California, USA. Global Wetlands: Old World and New**:** 265-276.

Craft, C. (2009). Effects of Climate Change on Water Quality Improvement Functions of Tidal Forests and Marshes. Oral Presentation. Coastal and Estuarine Research Federation. Portland, OR.

Day, J. W., C. A. S. Hall, et al. (1989). Estuarine Ecology. New York, Wiley-Interscience.

Fischer, D. T., S. V. Smith, et al. (1996). "Simulation of a century of runoff across the Tomales watershed. Marin County, California." Journal of Hydrology 186: 253-273.

Helsel, D. (2005). Nondetects and Data Analysis: Statistics for Censored Environmental Data (Statistics in Practice). New York, New York, Wiley-Interscience.

Hollibaugh, J. T., B. E. Cole, et al. (1988). "Tomales Bay, California: a "macrocosm" for examining biogeochemical coupling at the land-sea interface." Eos 36: 843-845.

Joye, S. et al. (undated). Salinization of Coastal Freshwater Wetlands. University of Georgia, Athens. Web site.

KHE (2006). Hydrologic Feasibility Assessment Report: Giacomini Wetland Restoration Project. Point Reyes National Seashore. Point Reyes Station, California, Prepared for Point Reyes National Seashore.

KHE (2009a). Olema Marsh Restoration: Salinity Impact Assessment. Prepared for Point Reyes National Seashore, Point Reyes Station, California.

KHE (2009b). Evaluation of recent topographic profiles at selected locations in Lagunitas Creek: 2002-2009. Prepared for Point Reyes National Seashore, Point Reyes Station, California.

KHE (2010a). Giacomini Wetlands 2009 Monitoring Data Summary. Preparedfor Point Reyes National Seashore, Point Reyes Station, California.

KHE (2010b). Survey of Cross-Sectional Profiles in the Giacomini Wetland Restoration Project Area. Prepared for Point Reyes National Seashore, Point Reyes Station, California.

KHE (2011a). Giacomini Wetlands 2009 Monitoring Data Summary. Preparedfor Point Reyes National Seashore, Point Reyes Station, California.

KHE (2011b). Survey of Cross-Sectional Profiles in the Giacomini Wetland Restoration Project Area. Prepared for Point Reyes National Seashore, Point Reyes Station, California.

Largier, J.L., C.A. Lawrence, M. Roughan, D.M. Kaplan, E.P. Dever, C.E. Dorman, R.M. Kudela, S.M. Bollens, F.P. Wilkerson, R.C. Dugdale, L.W. Botsford, N. Garfield, B. Kuebel Cervantes, and D. Koraĉin. (2006). WEST: A northern California study of the role of wind-driven transport in the productivity of coastal plankton communities. Deep-Sea Research II. 53: 2833-2849.

Lewis, D. J., E.R. Atwill, et al. (2001). Linking on-farm dairy management practices to storm-flow fecal coliform loading for California coastal watersheds. Santa Rosa, California, University of California Cooperative Extension.

Mitsch, W. J. and J. G. Gosselink (2000). Wetlands. New York, New York, John Wiley & Sons.

Moyle, P. B. (2002). Inland Fishes of California, Univ. of California Press, Berkeley, CA.

Murphy, S. (2001). General information on phosphorous. BASIN. USGS. City of Boulder. Web site.

NPS (2007). Final Giacomini Wetland Restoration Project Environmental Impact Statement/Environmental Impact Report. Point Reyes National Seashore. June 2007.

Parsons, L. (2005). Long Term Monitoring Program: Part 1. Monitoring Framework. Point Reyes Station, California, Golden Gate National Recreation Area / Point Reyes National Seashore.

Parsons, L. (2009). Improving Wetland and Ecosystem Health through Restoration: Pre-Restoration Hydrologic and Water Quality Conditions. Giacomini Wetland Restoration Project. Point Reyes National Seashore in collaboration with Kamman Hydrology and Engineering. March 2009.

Parsons, L. (2010). Year One of the Giacomini Wetland Restoration Project: Analysis of Changes in Water Quality Conditions in the Project Area and Downstream. Appendix A: Tomales Bay Watershed Council Annual Report to State Water Resources Control Board.

Parsons, L. and A. Ryan. *in prep*. Keeping Pace with the Tides: Results of SET and Marker Horizon Monitoring – Giacomini Wetland Restoration Project, Tomales Bay. Point Reyes National Seashore.

Regional Water Quality Control Board (RWQCB) (2005). Pathogens in Tomales Bay Watershed. Total Maximum Daily Limit (TMDL). Staff Report, San Francisco Bay Regional Water Quality Control Board.

Russell, A.D., T.M. Hill, B. Gaylord, E. Sandford, L.M. Jacobs, and J. Hostelt. (2010). Marine and terrestrial carbon supplying Tomales Bay: Modern versus historical estimates. Poster presentation to American Geophysical Union (AGU).

Vanni, M. J., W. H. Renwick, et al. (2001). "Dissolved and Particulate Nutrient Flux from Three Adjacent Agricultural Watersheds: A Five-Year Study." Biogeochemistry **54**(1): 85-114.

Wilkerson, F.P., A.M. Lassiter, R.C. Dugdale, A. Marchi, and V.E. Hogue. (2006). The phytoplankton bloom response to wind events and upwelled nutrients during the CoOP WEST study. Deep-Sea Research II. 53: 3023-3048.

Table 1. Bear Valley rainfall totals during the Giacomini Wetland Restoration Project water quality monitoring study period.

Month	2002-03	2003-04	2004-05	2005-06	2006-07	2007-08	2008-09	2009-10	2010-11	Month	Ave
Jul						0.02	0	0.01	0	Jul	0.01
Aug	0	0	0.07	0.07	0	0	0	0	0	Aug	0.02
Sep	0.1	0	0.37	0	0	0.13	0.02	0.43	0.11	Sep	0.13
Oct	0.9	0.17	4.97	1.08	0.82	3.35	1.66	6.82	5.76	Oct	2.84
Nov	6.02	2.71	1.61	3.27	5.54	1.1	3.47	1.52	5.21	Nov	3.38
Dec	17.33	12.14	10.13	19.99	7.64	6.76	3.17	5.21	11.97	Dec	10.48
Jan	6.45	6.03	4.85	7.94	1.09	14.93	1.22	12.34	2.11	Jan	6.33
Feb	7.2	1.24	5.33	4.76	9.57	4.54	14.49	7.23	6.97	Feb	6.81
Mar	2.1	0	7.69	11.75	0.72	0.59	3.41	5.53	10.89	Mar	4.74
Apr	1.09	0	1.93	6.8	2.67	0.34	0.73	6.02	1.54	Apr	2.35
May	0	0	5.21	0.35	1.18	0.13	3.19	2.84	2.75	May	1.74
Jun	1.02	0	1.86	0	0	0	0	0	1.78	Jun	0.52
Ave	42.21	22.29	44.02	56.01	29.23	31.89	31.36	47.95	49.09		

20 Yr Mean	36.38
30 Yr Mean	36.91

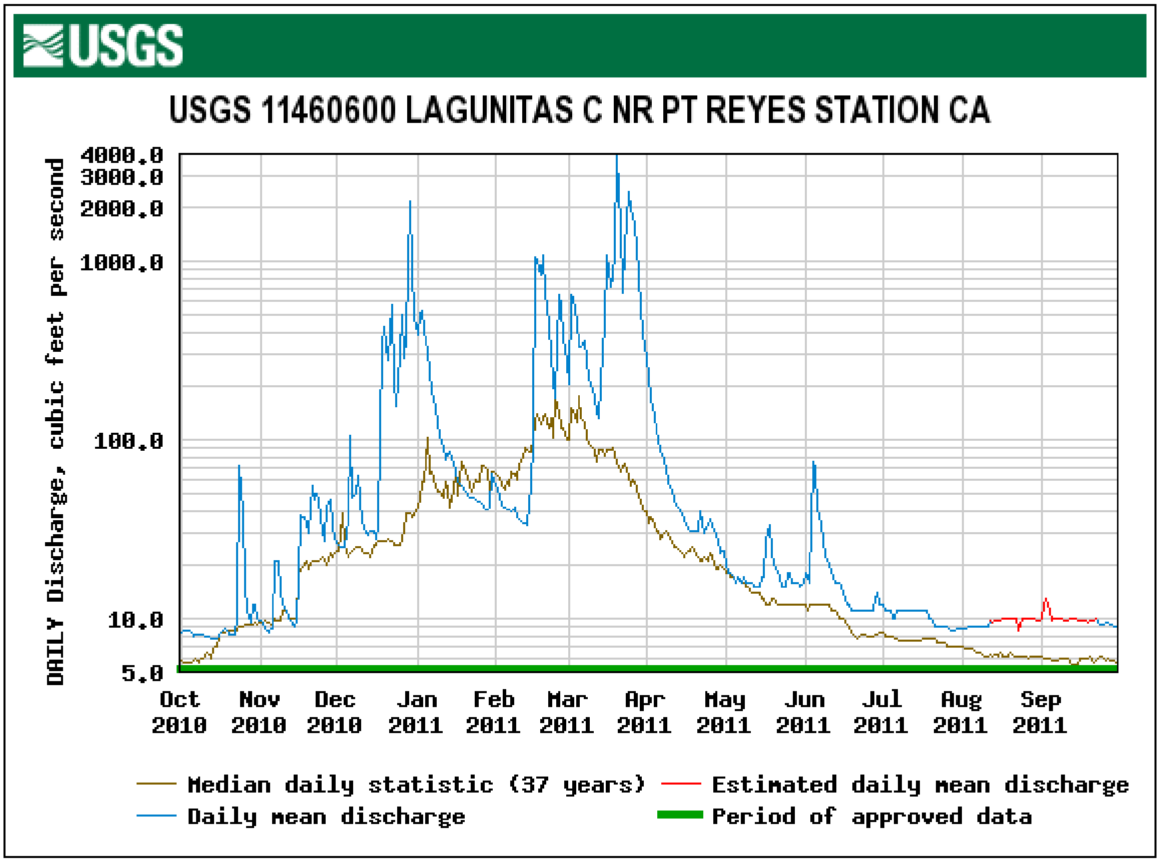

Figure 1. Lagunitas Creek discharge in Year 3 or WY2010/2011. Graph courtesy of USGS.

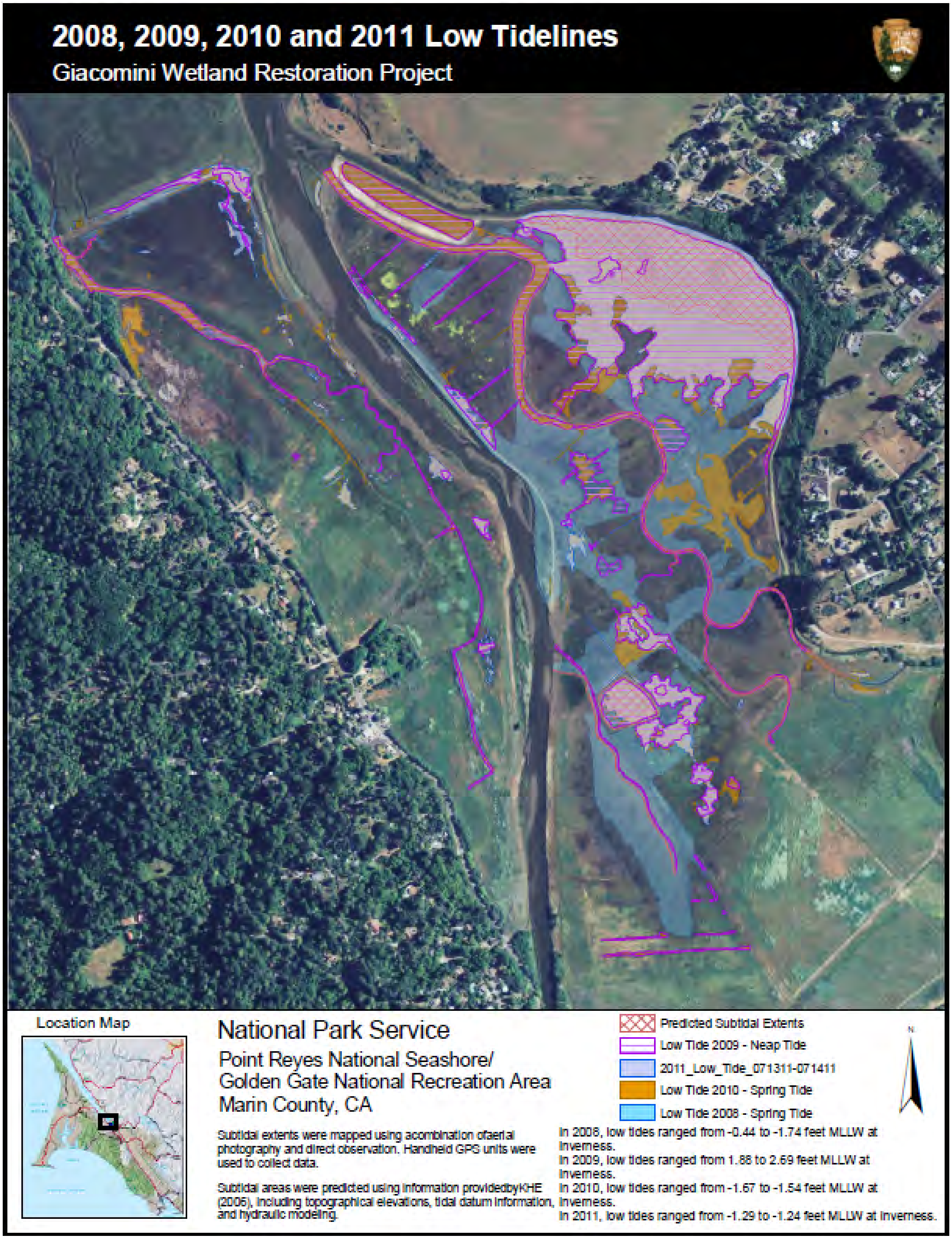

Figure 2. Low tidelines under spring and neap tide conditions since restoration was implemented. Colored areas represent inundated areas under extreme low tide conditions. Predicted refers to areas that were predicted by hydrologic modeling to remain subtidal or inundated under fully evolved marsh conditions.

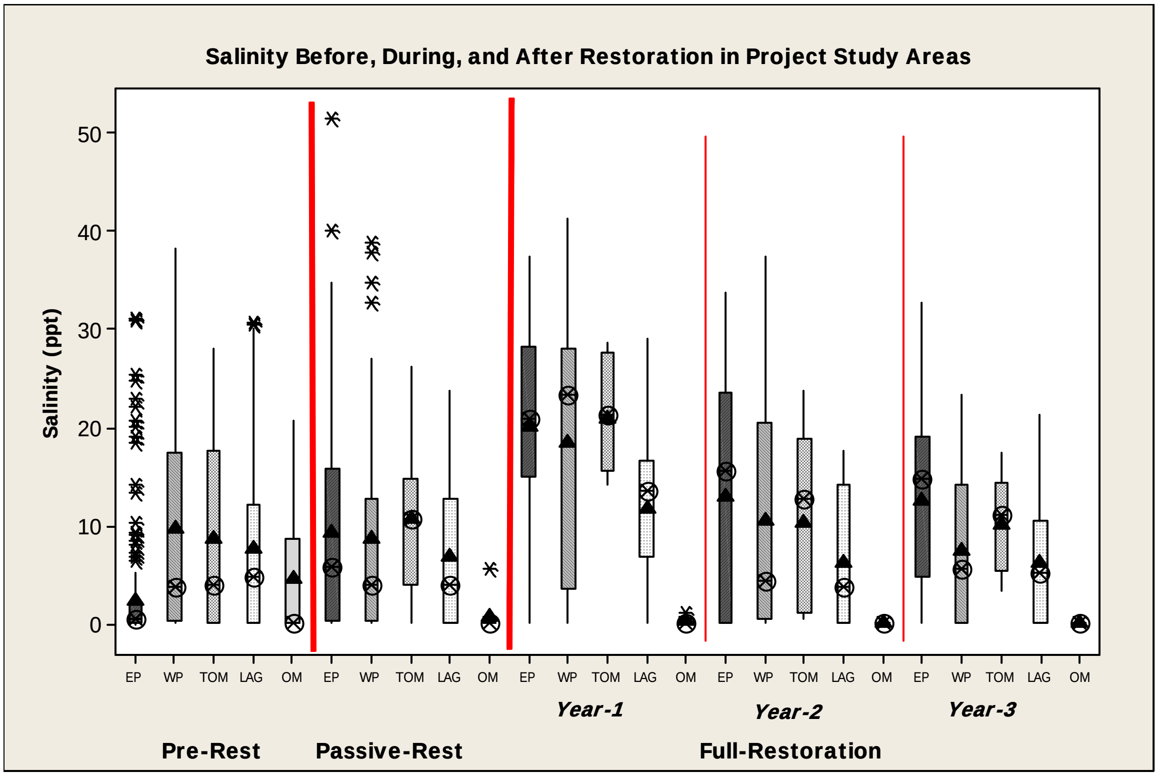

Figure 3. Average, median, and other summary statistics for salinity in the Project Area subsampling units Pre-Restoration, during Passive Restoration, and in the first three years of Full Restoration.

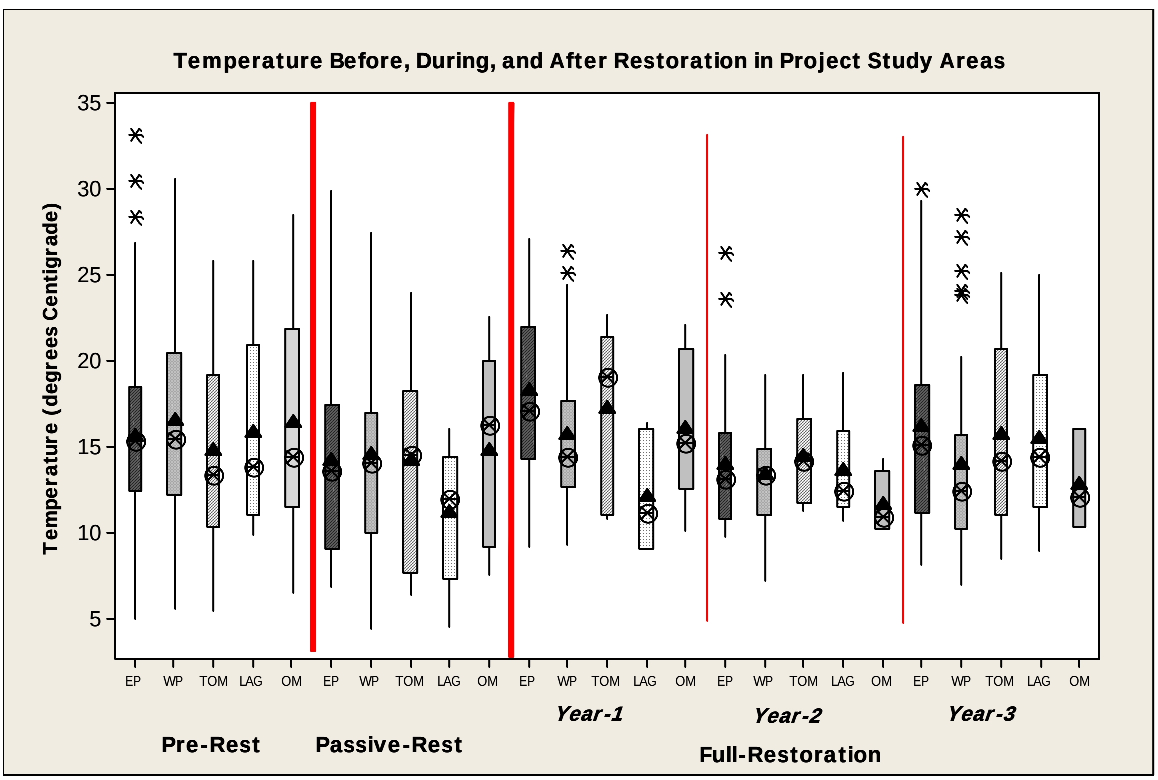

Figure 4. Average, median, and other summary statistics for temperature for Project Area subsampling units Pre-Restoration, during Passive Restoration, and in the first three years of Full Restoration.

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.

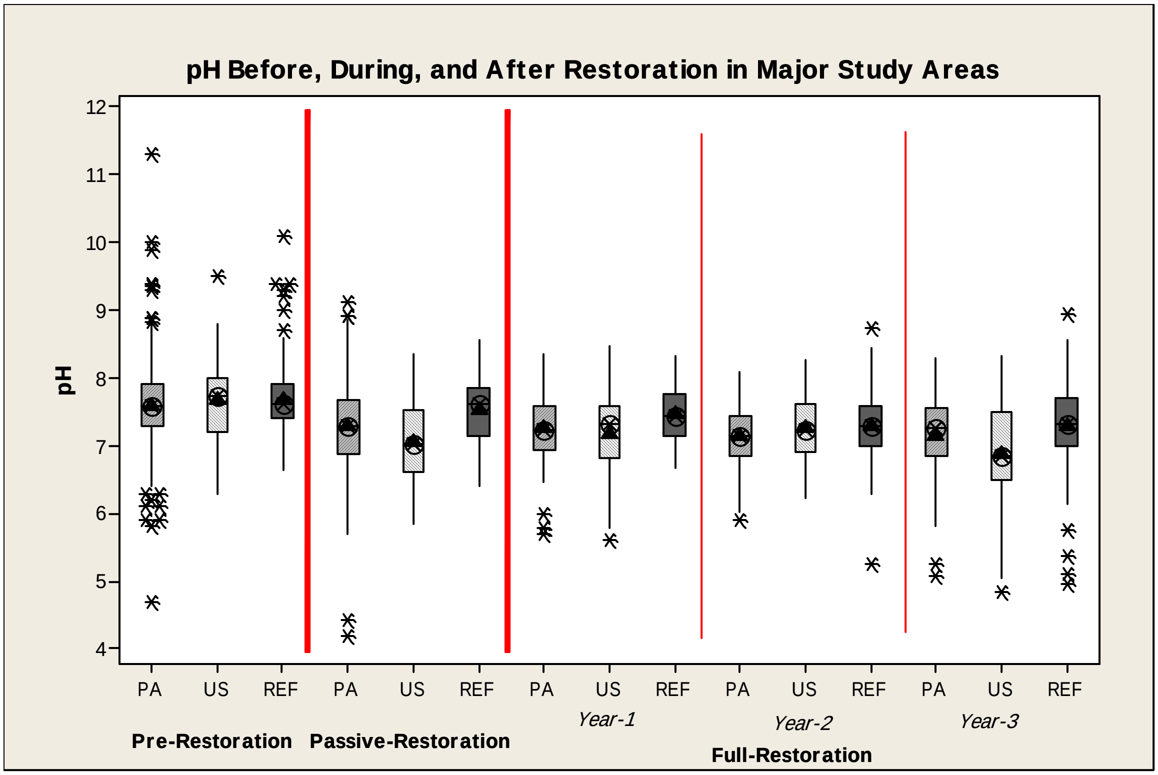

Figure 5. Average, median, and other summary statistics for pH for Study Areas Pre-Restoration, during Passive Restoration, and in the first three years of Full Restoration.

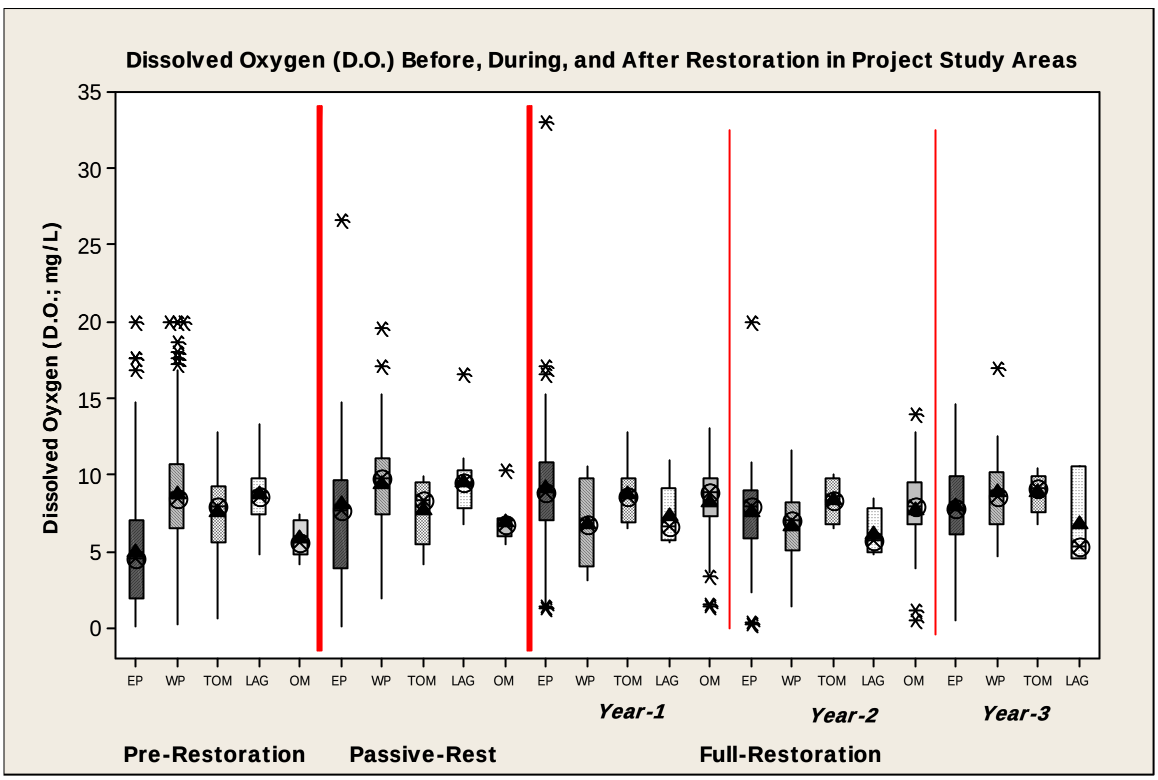

Figure 6. Average, median, and other summary statistics for dissolved oxygen for sub-sampling areas in the Project Area Pre-Restoration, during Passive Restoration, and in the first three years of Full Restoration.

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.

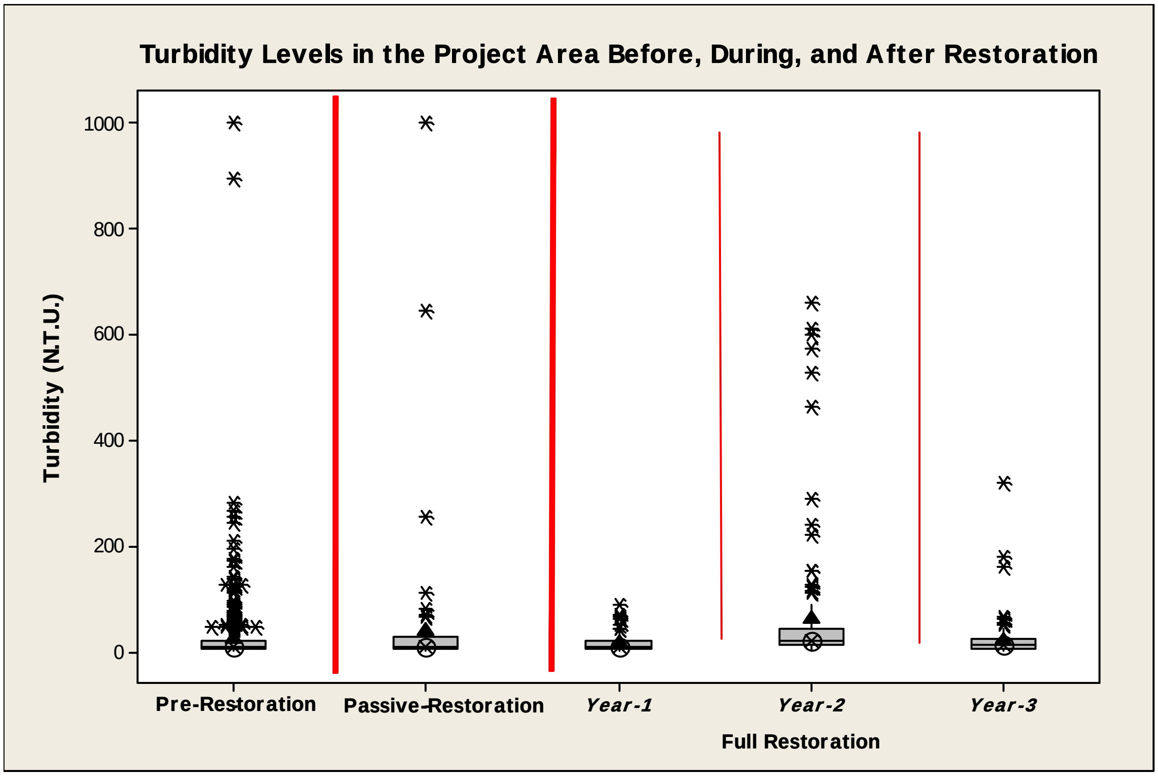

Figure 7. Average, median, and other summary statistics for turbidity in the Project Area Pre-Restoration, during Passive Restoration, and in the first three years of Full Restoration.

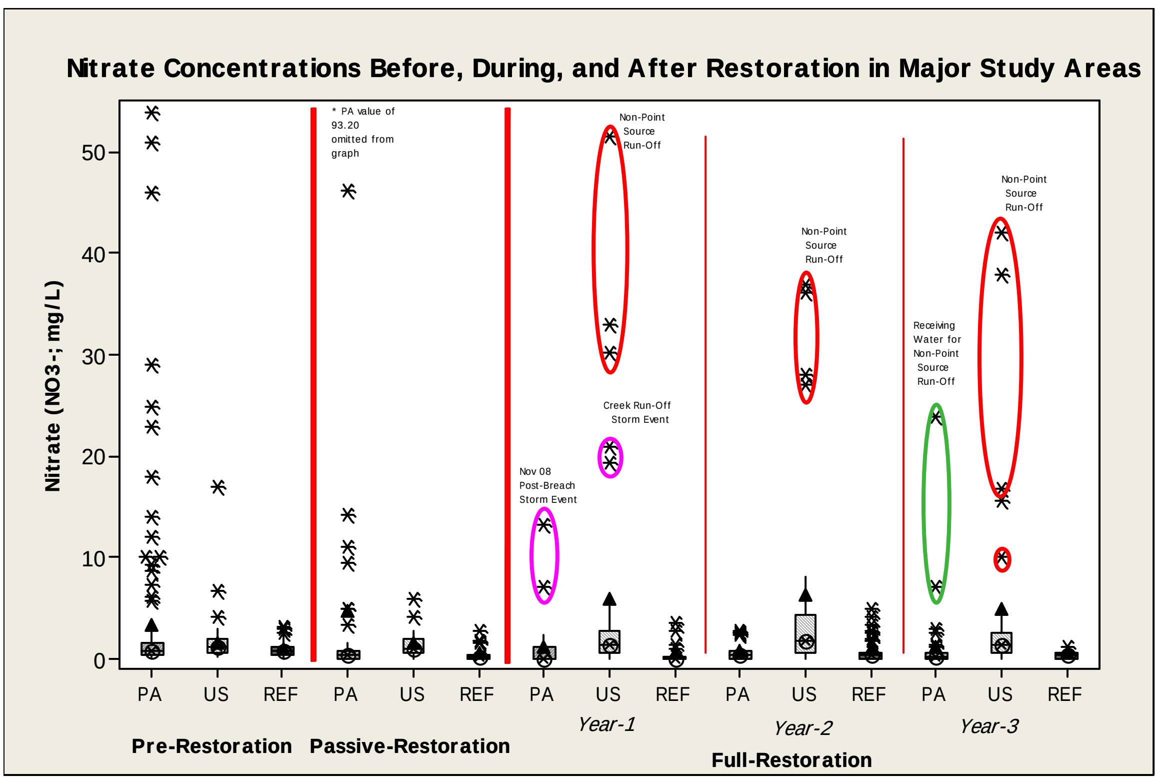

Figure 8. Average, median, and other summary statistics for nitrates (NO3-) for Study Areas Pre-Restoration, during Passive Restoration, and in the first three years of Full Restoration.

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.

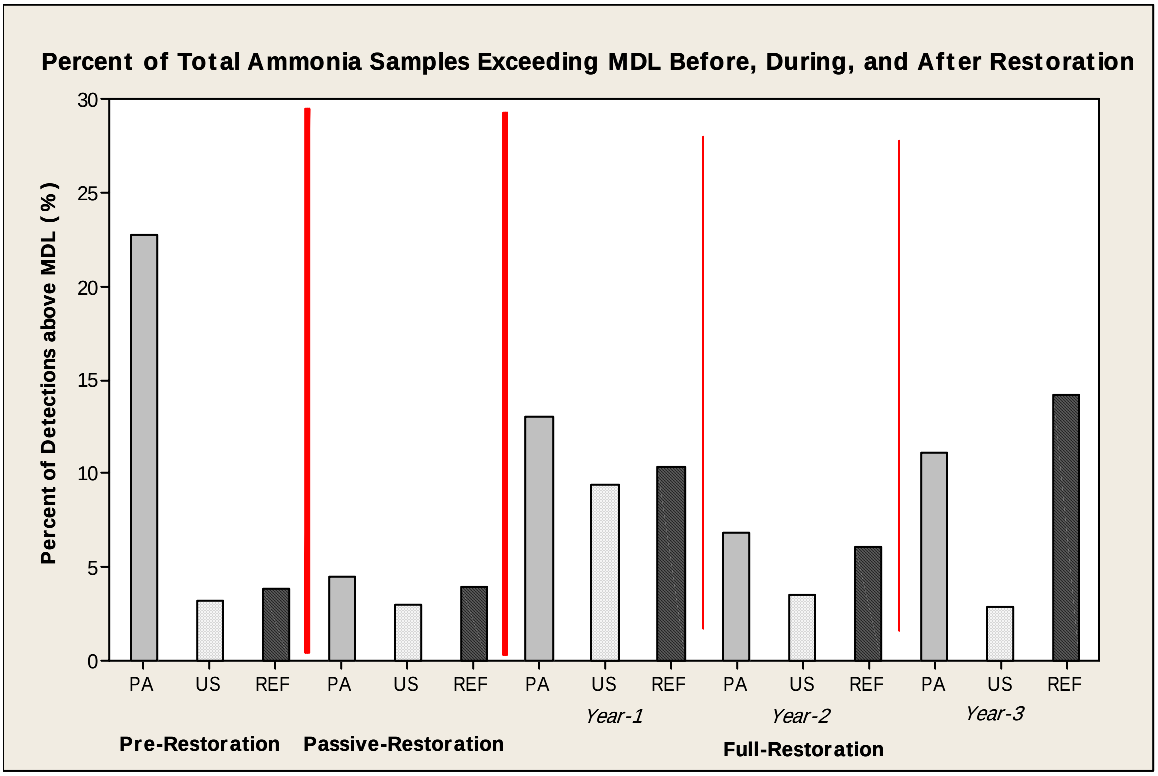

Figure 9. Percent of samples above Total Ammonia concentration detection limits for Study Areas Pre-Restoration, during Passive Restoration, and in the first three years of Full Restoration.

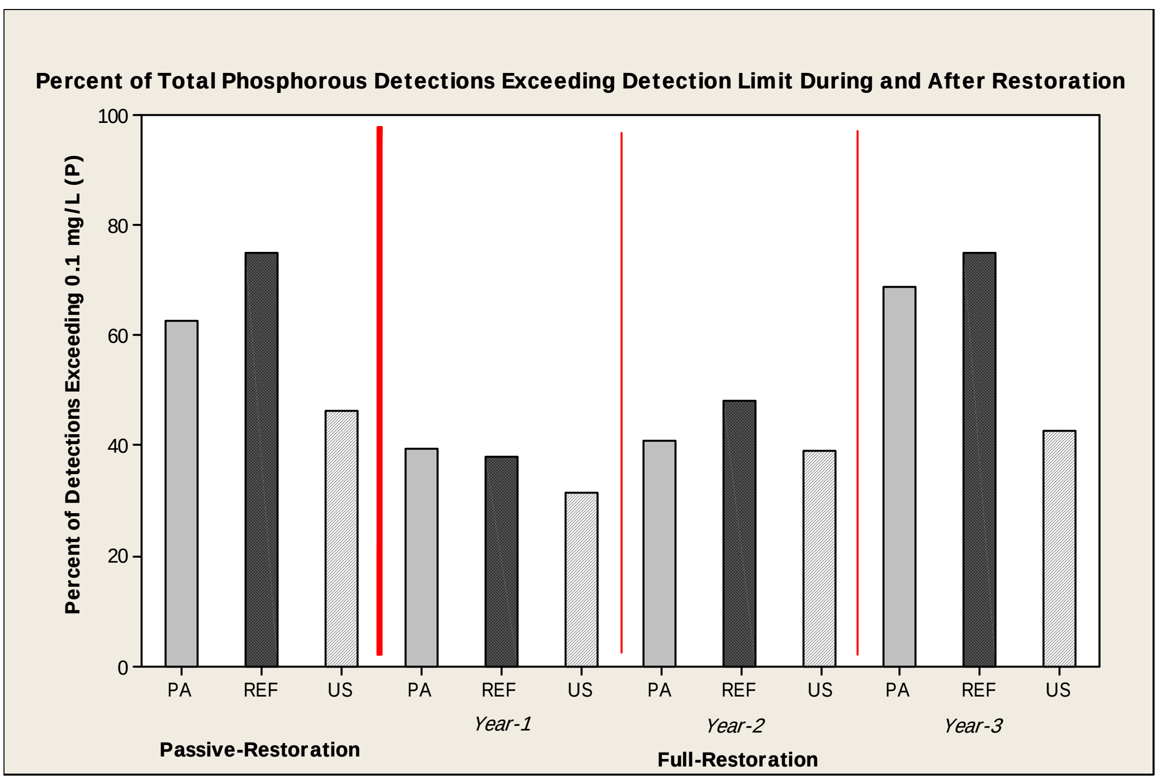

Figure 10. Percent of samples above Total Phosphorus concentration detection limits for Study Areas Pre-Restoration, during Passive Restoration, and in the first three years of Full Restoration.

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

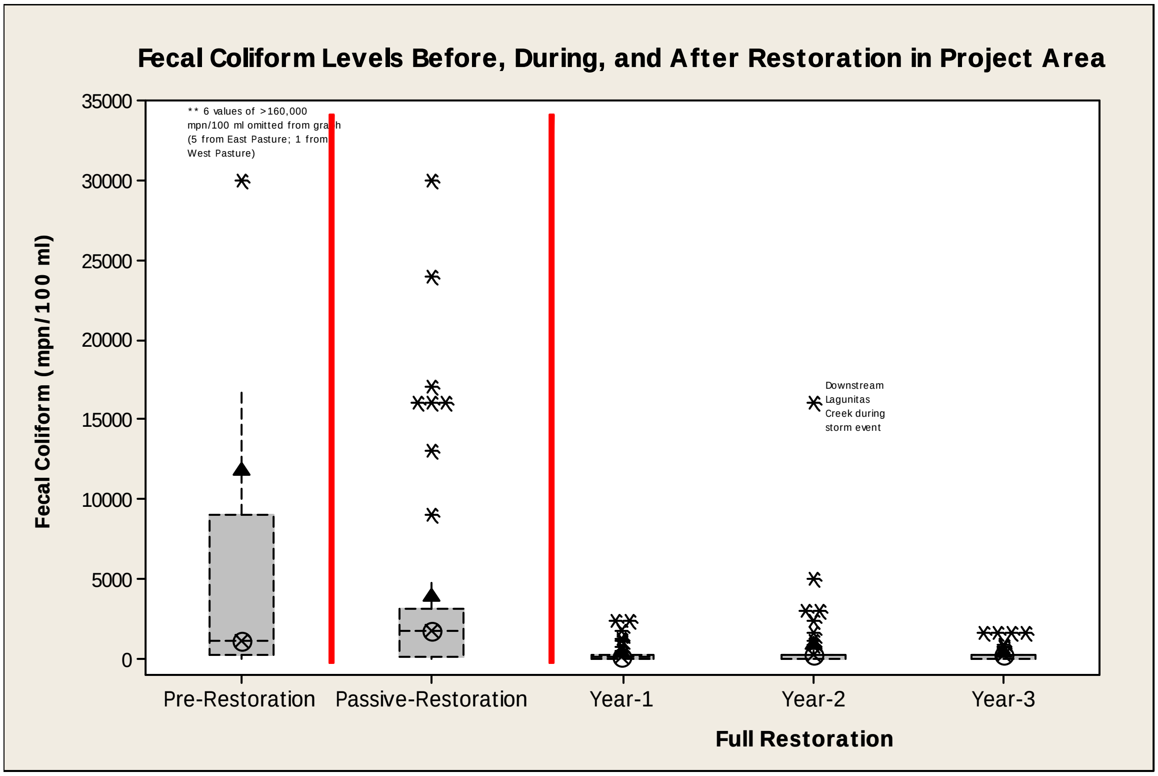

Figure 11. Average, median, and other summary statistics for fecal coliform concentrations for the Project Area Pre-Restoration, during Passive Restoration, and in the first three years of Full Restoration. Note: dashed lines for boxplots indicates both right- and left-censored data.

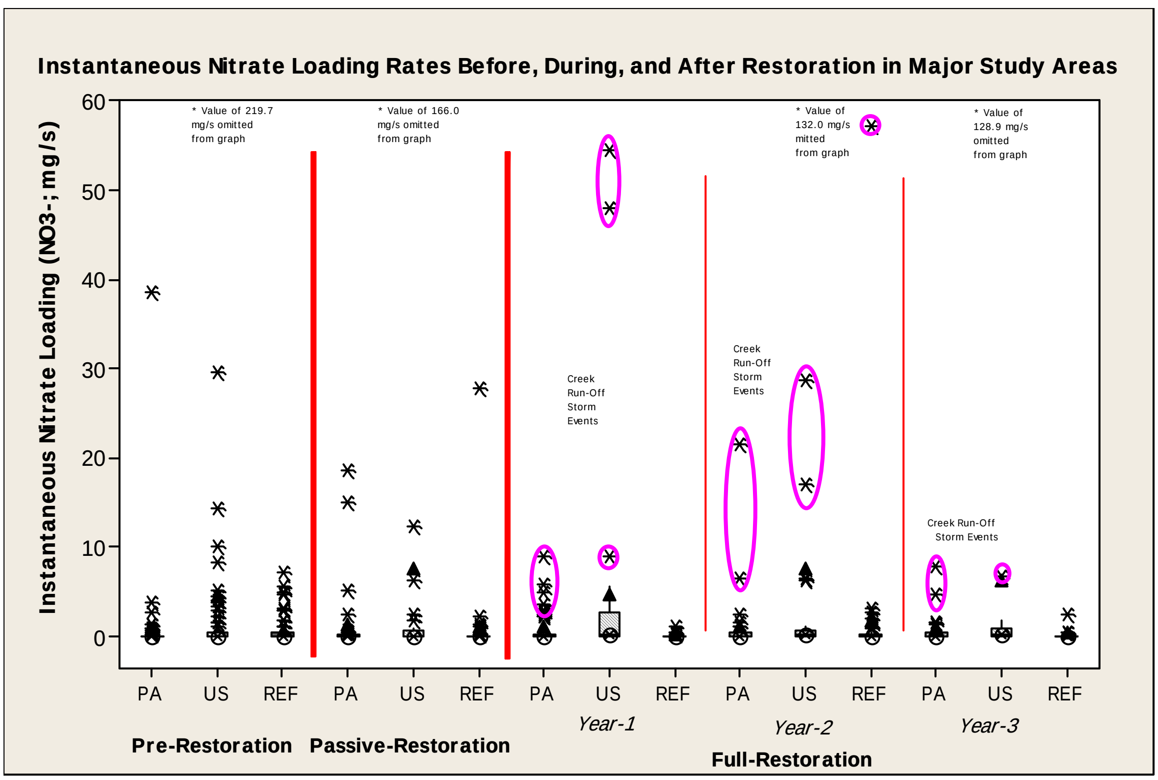

Figure 12. Average, median, and other summary statistics for nitrate instantaneous loading for Study Areas Pre-Restoration, during Passive Restoration, and in the first three years of Full Restoration.

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.

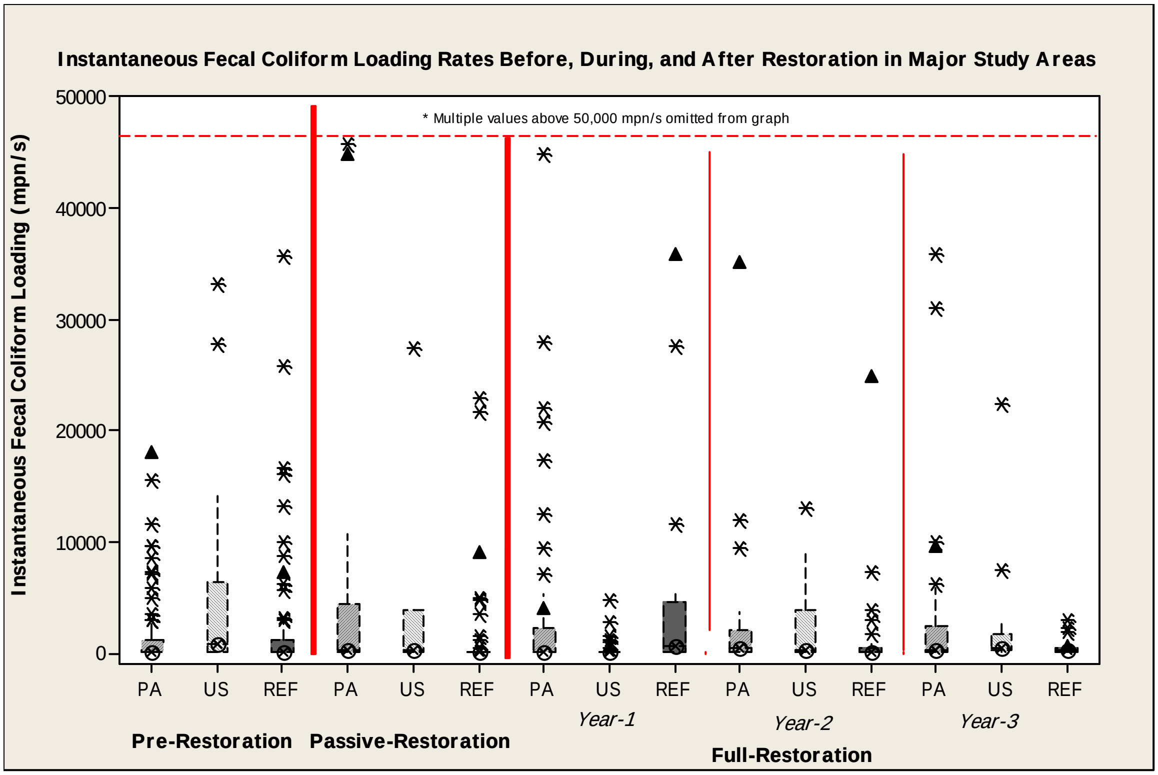

Figure 13. Average, median, and other summary statistics for fecal coliform loading for Study Areas Pre-Restoration, during Passive Restoration, and in the first three years of Full Restoration.

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.