# AN EXAMINATION OF WATER QUALITY IN THE SOUTH BAY SALT POND RESTORATION PROJECT

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by

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CERTIFICATION OF APPROVAL

I certify that I have read An Examination of Water Quality in the South Bay Salt Pond Restoration Project by Samuel Benjamin Stein, and that in my opinion this work meets the criteria for approving a thesis submitted in partial fulfillment of the requirement for the degree Master of Arts in Geography: Resource Management and Environmental Planning at San Francisco State University.

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AN EXAMINATION OF WATER QUALITY IN THE SOUTH BAY SALT POND RESTORATION PROJECT

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The South Bay Salt Pond Restoration Project (SBSPRP) was established in 2003 from 15,100 acres of former Cargill salt harvesting ponds in the San Francisco Bay Area. Since then, the SBSPRP has utilized an adaptive management framework to restore the ponds with the goal of habitat restoration, public access, and flood protection as its guiding principles. The SBSPRP is the largest wetland restoration project on the West Coast and the complexity of the project is compounded by nearby land use, including wastewater treatment plants (WWTPs) and urban development. The majority of previous water quality studies in the area have primarily focused on legacy pollutants, such as methylated mercury. In the SBSPRP and the South SF Bay as a whole, eutrophication and hypoxia are issues of ongoing concern. During the summer of 2016, a selection of Alviso ponds with diverse management histories were sampled for water quality parameters including dissolved oxygen (DO), nitrate ($NO_3^-$), and ammonium ($NH_4^+$) during spring and neap tides. Nutrient concentrations were positively correlated with DO values; distance to WWTPs was negatively correlated with DO. When examining change in DO from pond inlet to outlet, volume was negatively correlated. Pond management regime and tidal action also influenced the change in DO, suggesting that residence time may be a control on DO levels in the SBSPRP. These results suggest that greater considerations for DO and other water quality parameters may be of use in future adaptive management strategies in the SBSPRP, both in making new management decisions and anticipating effects of selected actions.

I certify that the Abstract is a correct representation of the content of this thesis.

Chair, Thesis Committee | Date

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## 0. List of Abbreviations and Terms

AMP – Adaptive Management Plan

DO – Dissolved Oxygen

RMP – Regional Management Plan

SBSPRP – South Bay Salt Pond Restoration Project

SFEI – San Francisco Estuary Institute

SJSCRWF – San José-Santa Clara Regional Wastewater Facility

SWPCP – Sunnyvale Water Pollution Control Plant

USFWS – United States Fish and Wildlife Service

USGS – United States Geological Survey

WWTPs – Wastewater Treatment Plants

# 1. <u>Introduction</u>

## 1.1 Tidal Wetlands in the South San Francisco Bay

Tidal wetlands, although historically undervalued, represent some of the most productive and important ecosystems on earth (Anderson-Wilk, 2007; Ramsar Convention Secretariat, 2010). Benefits of functional tidal wetlands include flood protection, transitional habitat for key species, water pollution filtration and zones of cultural and recreational significance (Barbier, 2012). Two thirds (64-71%) of all global wetlands have been lost, mostly due to development or concerns over vector control (Ambrose, 1999). In the SF Bay Area, wetland loss is even more severe: 95% of all estuarine wetlands have been diked and/or filled since the mid-1800s (Cloern & Jassby, 2012).

This loss of tidal wetlands is especially concerning with the prospect of environmental changes facing the San Francisco Bay Area (hereafter, SF Bay Area). Wetlands are of particular value to regional resilience given their ecosystem services value - the benefits ecosystems provide to humans, such as flood protection, pollution filtration, and habitat for species of concern (Barbier, 2012). Sea level rise, compounded by the increasing effects of climate change, has placed natural ecosystems and coastal development near the SF Bay at risk of flooding, especially during high tides or storm events. The California State Coastal Conservancy estimates that the majority of natural baylands will be destroyed by sea level rise by 2100 in the absence of additional mitigation efforts (San Francisco Bay Area Wetlands Ecosystem Goals

Project, 2015). The Conservancy further recommends creating and safeguarding 100,000 acres of tidal wetland to prevent this from occurring (San Francisco Bay Area Wetlands Ecosystem Goals Project, 2015). Other issues, including the decline in habitat for protected species such as the salt harvest mouse and California clapper rail and increased nutrient loading, are also symptoms of historic wetland loss, increasing urbanization, and population growth (Finlayson, 2012).

### 1.2 The South Bay Salt Pond Restoration Project

The South Bay Salt Pond Restoration Project (SBSPRP) was established in 2004 to restore portions of the South Bay used for commercial salt harvesting operations to a more natural, tidal marsh morphology and to support the functions described above. The SBSPRP is enormous in size and scope, requires significant financial resources, involves stakeholders from a variety of levels, and is the largest wetland restoration project on the west coast of the Americas (EDAW, Philip Williams and Associates LTD, H.T. Harvey and Associates, Brown and Caldwell, & Geomatrix, 2007). The SBSPRP has a high degree of complexity and associated uncertainty, which managers are attempting to mitigate through the use of an Adaptive Management Plan (AMP).

A large component of the SBSPRP's AMP involves scientific study and monitoring of the restored ponds and surrounding area. Studies include measurements of mercury in sediment and water, bird counts and sediment deposition (Bourgeois, 2015b). Less is known about how other water quality parameters, including nutrients and dissolved oxygen, change during the restoration process at the SBSPRP, and how these changes might impact surrounding aquatic ecosystems such as the naturally occurring sloughs (tributaries with freshwater influence) that run throughout the ponds. The influence of controlling factors, such as pond volume and management, on these water quality parameters during restoration has not been examined.

### 1.3 Estuarine Water Quality

The quality of estuarine waters plays a large role in the overall health of the regional ecosystem. Dissolved Oxygen (DO), the amount of oxygen freely available in a system for organisms to use, supports the productivity of an ecosystem, and lack of sufficient DO can lead to fish kills and strains on ecosystem services (Mclaughlin & Choen, 2013; National Estuarine Experts Workgroup, 2010; Peña, Katsev, Oguz, & Gilbert, 2010). Excess nutrient loading in an ecosystem can lead to eutrophication, resulting in hypoxic (low DO) conditions. This occurs when nutrient loading leads to algal blooms and subsequent drops in DO when algal matter decomposes (Peña et al., 2010). The most common sources of nutrient loading in SF Bay is discharge from sewage treatment plants and other wastewater facilities (Glibert et al., 2010; Yigzaw, 2014). In the South SF Bay, 92% of all sewage is discharged into the Bay post-treatment, representing a nutrient loading of 28 g Nitrogen (N) per meter squared per year, among the highest nutrient loading of all US estuaries (Cloern & Jassby, 2012).

High nutrient loading from wastewater treatment plants (WWTPs) is compounded by long residence times in the South Bay, up to four times longer than the residence time in the easternmost portions of SF Bay (Cloern & Jassby, 2012). Residence time is defined as the length of time that a parcel of water remains in a larger water body (Defne & Ganju, 2014). Longer residence times mean that many compounds such as nutrients are retained in the system, making it more vulnerable to issues such as eutrophication (Defne & Ganju, 2014). Due to tidal hydrology, estuarine water quality is also sometimes discussed in terms of exposure time, or the time it takes for a water parcel to permanently exit the extent of tidal influence (Wolanski & Elliot, 2016). Exposure times are far longer than residence times, given that many particles that exit a system during an ebb tide can reenter during the following flood tide.

The SBSPRP is located nearby several WWTPs, and nutrient loading from treated sewage discharge has been noted in previous water quality studies for some restored ponds (Senn, Novick, Downing-Kunz, Bresnahan, & Hollerman, 2015; Topping et al., 2013). The influence of WWTP discharges on water quality across different pond locations, morphologies, and management types- the controls of water flow through the pond system- is still poorly understood. In some ponds in exceptionally close proximity to WWTPs, the influence of sewage discharge may also mask the role of other drivers of water quality change, such as surrounding land use. Tidal patterns also influence water quality; Helton, Brand, Piotter, and Takekawa

(2010) found that DO is related to tidal elevation, and that DO can dip during low tides.

### 1.4 Research Questions

To further address these gaps in knowledge, this study aimed to answer the following research questions:

1. How do dissolved oxygen and nutrient levels vary spatially in restored ponds?
2. How do these water quality parameters vary during different tidal conditions?
3. What other pond characteristics, such as morphology, management type, distance to wastewater treatment plants (WWTPs), and surrounding land use, are correlated with variation in water quality?
4. How can this information be used to support future management decisions in the area?

To address these research questions, this study examined water quality in multiple ponds in the Alviso section of the SBSPRP for parameters such as DO and nutrients. Supplemental data regarding pond characteristics such as volume and land use were extracted in GIS for further analysis in conjunction with field measurements. DO, nutrients, and other parameters were examined in conjunction with the supplemental data sets to determine potential drivers of pond water quality changes, and potential recommended changes to future restoration strategy.

## 2. Background and Literature

### 2.1 Historical Conditions of the South Bay

Until the late 19th century, the South SF Bay Area was dominated by tidal wetlands and salt marshes (Figure 1) and supported robust and diverse estuarine ecosystems (Cloern & Jassby, 2012). During the 1800s, Spanish settlers began expediting the Ohlone's minimally invasive salt harvesting process by building clay dikes around naturally occurring salt ponds (Johnck, 2008). The creation of levees and dikes within the bay accelerated from the mid-1850s to the late 1900s to create a commercial salt harvesting system that remained in place until 2003 (Johnck, 2008). This system included ponds with a variety of different morphologies, salinity concentrations, and functions to allow for the movement and concentration of brine over time and space. SF Bay water would enter into the furthest ponds (closest to the bay) where the water would evaporate over time. As salt concentration increased, the brine would move from pond to pond until it reached a processing pond next to shore, where the crystallized brine could be dried and shipped (California Research Bureau & California State Library, 2002; Johnck, 2008).

The transition to commercial salt harvesting by large corporations such as Cargill, Inc. resulted in huge yields of commercial salt – by the early 2000s, over a million tons were harvested annually in the South SF Bay (California Research Bureau & California State Library, 2002) – but with devastating environmental consequences. Conversion from marsh to segmented salt pond systems interrupted the natural hydrologic flow patterns of tidal wetlands and limited the number and diversity of species which resided there (EDAW et al., 2007).


**Figure 1**: 1928 imagery of the Alviso section of salt marshes. Images overlaid with current salt pond outlines. Historical imagery courtesy of USGS, pond outlines courtesy of SFEI. Some portions of the South Bay were already diked at the time of these photographs, while other sections (such as those in the center of the figure) retained a more natural morphology.

The salt harvesting process also led to a buildup of salt byproducts, which generally do not support vegetation (EDAW et al., 2007). Although some species of birds prefer open water areas such as the salt ponds for breeding, the conversion of tidal marsh to salt pond led to a loss of habitat, biodiversity, and flood protections in the region (Bourgeois, 2014). Unfortunately, minimal data are available regarding the conditions of the South Bay prior to salt pond construction, particularly in regards to water quality.

### *2.1.1 Environmental Complexities in the South Bay*

Given its high degree of development and high population, the SF Bay Area is a region of ever-increasing environmental complexity (Cloern & Jassby, 2012; Grenier & Davis, 2010). The South Bay Salt Pond Restoration Project (SBSPRP) acknowledges the potential for human influences on water quality from WWTP outfalls, railway lines, landfills, construction projects, urban runoff, golf courses, airplane runways and hangars, storm water collections facilities, and other forms of development (Figure 2). Separating the individual influences of these systems is challenging, although WWTP discharge is generally considered to be one of the largest influences on water quality in the South SF Bay (Yigzaw, 2014).

Among the 50 WWTPs in the Bay Area (Cloern & Jassby, 2012), two feed into the Alviso section of the SBSPRP (Yigzaw, 2014). The largest is the San José-Santa Clara Regional Wastewater facility, which releases an average of 78.7 million gallons a day (MGD) of effluent into Coyote Creek, near A17 (San José-Santa Clara Regional Wastewater Facility, 2015). The smaller of the WWTPs is the Sunnyvale Water Pollution Control Plant (SWPCP), which has a max capacity of 29.5 MGD and discharges an average flow of 16 MGD (City of Sunnyvale, 2001) into Guadalupe Slough near A3W.

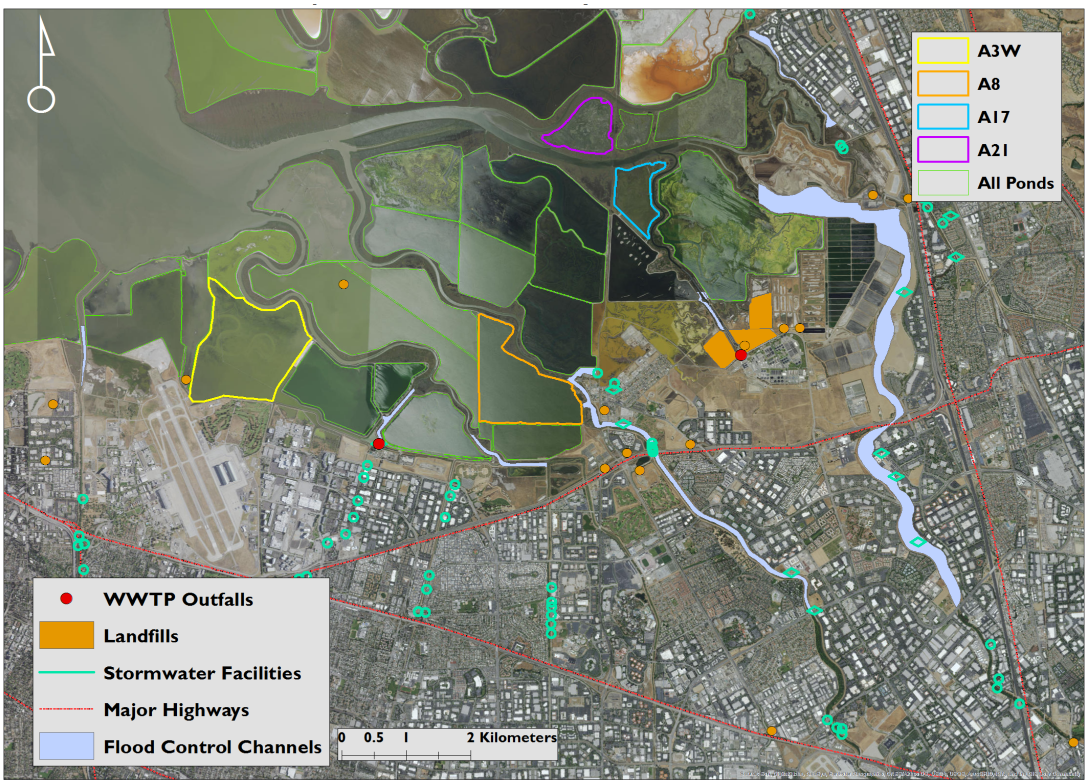



**Figure 2**: Environmental complexities in the South SF Bay. Red dots are WWTP discharge points, blue dots are storm water system infrastructure, and orange shapes indicate landfill areas. The green outline notes the existing pond structure.

## 2.2 South Bay Salt Pond Restoration Project

In 2003, Cargill, Inc. sold 15,100 acres of salt ponds to various federal, state, and local agencies, including the U.S. Fish and Wildlife Service (USFWS), the California Department of Fish and Wildlife (CAFWS), and the City of San Jose (H.T. Harvey & Associates, 2008). This purchase established the SBSPRP (Bourgeois, 2015b). The SBSPRP project was established with 6 main objectives in mind (from EDAW et al., 2007):

- "Create, restore, or enhance habitats of sufficient size, function, and appropriate structure to:
    - Promote Restoration of native special-status plants and animals that depend on South San Francisco habitat for all or part of their life cycles.
    - Maintain current migratory bird species that utilize existing salt ponds and associated structures such as levees.
    - Support increased abundance and diversity of native species in various South San Francisco Bay aquatic and terrestrial ecosystem components, including plants, invertebrates, fish, mammals, birds, reptiles and amphibians.
- Maintain or improve existing levels of flood protection in the South Bay Area.
- Provide public access and recreational opportunities compatible with wildlife and habitat goals.
- Protect or improve existing levels of water and sediment quality in the South Bay, and take into account ecological risks caused by restoration.
- Implement design and management measures to maintain or improve current levels of vector management, control predation on special status species, and manage the spread of non-native invasive species.
- Protect services provided by existing infrastructure."

The first three goals are more commonly cited within SBSPRP planning documents and other project literature (Bourgeois, 2014; EDAW et al., 2007; South Bay Salt Pond Restoration Project, 2016a). In discussion of water quality related objectives in the SBSPRP, a greater focus has been placed on legacy pollutants such as mercury compared to other parameters such as DO or nutrients (Ackerman, Marvin-DiPasquale, Slotton, Horzog & Eagles-Smith, 2010; Bourgeois, 2015b; Helton et al, 2011; H.T. Harvey & Associates, 2008;).

In meeting the 6 objectives above, the SBSPRP managers hope to achieve two different goals: restoring the entire project to 50% tidal ponds and 50% managed ponds by 2025, and, if possible, restoring the project to 90% tidal ponds and 10% managed ponds by 2050 . Working towards the 90%:10% will only be adopted as a management goal if it can be achieved without significant degradation of other AMP objectives such as habitat for migratory birds (Bourgeois, 2015b; South Bay Salt Pond Restoration Project, 2016a).

Tidal ponds exchange water freely with the South Bay and corresponding sloughs. These systems allow for greater sediment deposition and expedite vegetation growth (South Bay Salt Pond Restoration Project, 2016b). SBSPRP manages water flow through the use of pond control structures and/or culverts that can be opened and closed. These ponds have less sediment deposition and retain more of the open water habitat preferred by many species of migratory birds, although such open water ponds did not exist prior to the advent of salt harvesting in the area. Pond managers hope to restore as much of the system as possible to the more natural tidal pond morphology without significantly reducing bird populations (Bourgeois, 2015b).

### *2.2.1 Restoration Actions to Date*

The primary mode of restoration in the SBSPRP is to implement a more natural hydrologic flow pattern to the ponds. Allowing more flow into the ponds decreases the salinity and leads to sediment deposition from bay and slough waters. When salinity decreases and sediment is deposited, vegetation and ecosystem health can be restored in the ponds. Allowing flow to return to the ponds requires breaching the existing levees, either through the complete removal of a section of levee to allow free exchange of water or by creating a pond control structure that allows for greater management of the dynamic flow patterns (EDAW et al., 2007).

Other forms of restoration include depositing sediment pulled from slough dredging or nearby construction sites; installing buffers or grates to control the flow of algal mats or fish (including protected species of fish that may have difficulty surviving in certain pond environments); installing monitoring stations and/or implementing of regular monitoring plans; and creating new signs, trails, and resources to inform visitors of the importance of restoration activities (Bourgeois, 2014).

## 2.3 Estuarine Water Quality

### *2.3.1 Water Quality in West Coast Estuaries*

Compared to large estuaries on the Atlantic (such as Chesapeake Bay), west coast estuaries are not as extensively studied or as well understood (Nezlin, Kramer, Hyde, & Stein, 2009). Previous research indicates that coastal zones along the eastern Pacific are susceptible to hypoxic events when upwelling along the continental shelf bring low-DO, nutrient rich waters from the sea floor to surface waters (Peña et al., 2010). Shallow estuaries or those with greater degrees of restriction on water exchange can also depress DO levels and make systems susceptible to hypoxia (Diaz & Rosenberg, 2008). Estuaries with large seasonal freshwater flows also exhibit changes in DO and other parameters as seasons change, most notably decreases in DO as freshwater inputs are at a minimum (Smith & Hollibaugh, 2006).

Many studies west coast estuaries have focused on the impact of anthropogenic drivers such as dredging of estuary sediments or discharge of sewage effluent. Haushild & Stoner (1973) examined the impact of dredging in the Duwamish River Estuary and predicted that changes to estuarine morphology increased residence time and would subsequently decrease DO levels by as much as 1.4 mg/L (Smith & Hollibaugh, 2006). Nezlin et al. (2009) examined physical drivers behind hypoxic conditions in the Upper Newport Bay and found that photosynthetic rate (controlled by factors such as ambient cloud conditions and algal mat shading) was a driver of low DO; stratification was also found to drive low DO conditions. This included stratification due to temperature (controlled by solar inputs) and salinity (largely a byproduct of freshwater inputs) (Nezlin et al., 2009). Finally, Cloern and Jassby (2012) noted in their examination that although the SF Bay has been more resistant to accelerated eutrophication than other bays (such as Chesapeake Bay), this resilience is beginning to erode (Cloern & Jassby, 2012; Yigzaw, 2014).

### *2.3.2 Water Quality in the South Bay*

The South SF Bay is adjacent to the heavily developed Silicon Valley, with major influences from wastewater treatment plants, urban runoff, landfills, construction projects, and increasing population (Cloern and Jassby, 2012). The South Bay has a long history of environmental crises, including legacy pollution loading from mercury mines in the New Almaden region (Grenier & Davis, 2010; US EPA, 2010) and sea-water intrusion as a result of groundwater overdraft (Department of Water

Resources, 1975; Parsons, Kulongoski, & Belitz, 2013). As such, the health of the South Bay and the quality of the bay waters are largely driven by anthropogenic factors.

Cloern & Jassby (2012) reviewed 40 years' worth of monitoring data in the SF Bay, and observed six major anthropogenic drivers of change: water diversion, modification to sediment supply, introduction of invasive species, wastewater inputs, environmental policy, and shifts in climate. Of those, wastewater inputs are of special concern in the South Bay, and the Alviso section of the SBSPRP in particular. Even after treatment to federal standards, wastewater contains high levels of nitrogen (nitrate and ammonium) and phosphorus (US EPA, 2016). Nutrient enrichment can lead to eutrophication, in which algal biomass blooms and decomposes, depleting dissolved oxygen and leading to hypoxic conditions (Cloern & Jassby, 2012). Severe hypoxic conditions (<2.0 mg/L DO) can lead to fish kills and other impacts to aquatic ecosystems (Diaz & Rosenberg, 2008).

In the South Bay and its tributary sloughs, water quality parameters including DO and nutrient levels exhibit high degrees of variability due to multiple compounding factors. Tidal influences cause large variations in DO, nutrients, and chlorophyll based on flood or ebb tide conditions (Cloern, Cole, Edmunds, Schraga & Arnsberg, 2000; Senn et al., 2015). There is also some evidence of seasonal impacts whereby low freshwater input during summer months decreases DO, although lack of precipitation in recent years has muted seasonal influence on DO (Novick, Bresnahan, Dowing-Kunz, & Senn, 2015). As discussed above, the WWTPs in the area release treated sewage effluent continually, although the magnitude of discharges varies throughout the day. This causes significant spikes in nutrient concentrations at the point of discharge, decreasing with distance traveled from the discharge point (Yigzaw, 2014). Sewage effluent can lead to corresponding shifts in chlorophyll and DO, as nutrients are utilized for algal growth (Senn et al., 2015).

During a high-resolution nutrient mapping study in July of 2016, the San Francisco Estuary Institute (SFEI) found that nitrate levels in Coyote Creek, the receiving water body for WWTP effluent from the San José-Santa Clara Regional Wastewater Facility, jumped to over 600 mg/L during ebb tides (Senn et al., 2015). In contrast, ambient nitrate levels in the SF Bay near the same time period typically ranged from 2-5 mg/L (Novick et al., 2015). SFEI's high-resolution mapping data also suggest that pond water quality can have a detectable impact on slough water (Senn et al., 2015). Although this and other anecdotal evidence of pond influence on slough water quality exist, the exact relationship between pond and slough water quality is not well understood.

### *2.3.1 Regional Water Quality Thresholds*

The variability and DO decreases in the South Bay are of concern partly due to the goal of 5.0 mg/L DO (or no less than 80% oxygen saturation for a three month median) set by the Regional Management Plan (RMP) to ensure adequate DO levels for fish spawning, and to prevent conditions that are optimal for botulism organisms (California Regional Water Quality Control Board San Francisco Bay Region, 2007).

Botulism has historically been an issue in other wildlife refuges with shallow pond habitats (Wilson, 2016). Although there is little historical evidence to indicate whether this threshold is consistent with natural South SF Bay conditions, the DO levels currently exhibited by some of the sloughs and ponds drop far below 5.0 mg/L threshold (SFEI, 2016). Given the lack of long term DO data for the South Bay, it's difficult to determine if decreases in DO are a natural part of the ecosystem or a product of human influence in the area.

In recognition of the DO levels' variability and frequent dips below the RMP threshold, SBSPRP managers operate under a second DO threshold of 3.3 mg/L, referred to as a "trigger" (Helton et al., 2011). If the $10^{th}$ percentile DO levels drop below this level on a weekly basis, managers will examine potential management actions to remedy low DO conditions and act accordingly. In managed ponds where water circulation is reduced, at least one additional negative impact must occur to activate the "trigger": mortality of organisms, odors that cause nuisance, destruction of habitat, or unacceptably high methylmercury rates (South Bay Salt Pond Restoration Project, 2016b). Methylmercury is the form of mercury that is readily absorbed by organisms, and is formed by anaerobic bacteria (primarily in sediment) (Compeau & Bartha, 1985).

### *2.3.2 Previous Water Quality Studies in the SBSPRP*

Pond A3W has available water quality data from 2004-2007, including nutrient data (Mruz, 2008; Shellenberger et al., 2008, and Topping et al., 2013). Previous research in A3W occurred in 2004-2007 (Mruz, 2008), 2007 (Shellenbarger et al., 2008) and 2010/2012 (Topping et al., 2013); Topping et al. (2013) served as a model for sampling point selection during this study.

Mruz (2008) monitored pond A3W from the start of restoration in 2004 through its first four years of restoration. Mruz (2008) noted exceptionally low levels of DO, with 10% percentile levels of DO rarely exceeding the 3.3 mg/L DO management threshold and only 8% of sampling weeks in A3W during 2007 reaching the threshold. Efforts to reduce algae or increase flow into the pond did not produce consistent increases to DO; Mruz (2008) concluded that uncontrollable factors such as precipitation and climate had the greatest impact on DO.

Shellenbarger et al. (2008) examined the quality of water discharged from pond A3W and its influence on Guadalupe Slough. Shellenbarger et al. determined that DO was lowest during low tide and early mornings, and had the greatest influence on sloughs when tides were receding (2008). DO in A3W averaged significantly lower than slough water during the course of the study. Pond water discharge also had significant impact on the conductivity and pH of slough water immediately surrounding the pond outlet, although this impact dissipated with distance from the outlet (Shellenbarger et al., 2008).

The more recent research by Topping et al. (2013) provides additional understanding of pond water quality. By studying nutrient data at multiple points within the pond, Topping et al. found that lowest DO readings occurred during low tides, and that DO was significantly higher in water entering the ponds than in the mid-pond points or in water exiting the ponds. The deepest sampling point suffered the worst hypoxic conditions. Surface water nutrient levels were found to be relatively low (<1 μM/L) during the duration of the study (Topping et al., 2013).

The most recent water quality data collected in pond A8 was from Amato and Valoppi (2015), who found 5 out of 48 sampling days in summer 2014 exhibited hypoxic conditions of less than 2.0 mg/L DO. However, patterns of low DO were more closely related to wind and diurnal patterns than to tidal influence (which is the dominant influence on DO in tidal ponds such as A21).

Pond A17 has had minimal previous direct water quality sampling. In 2003-2005, at the beginning of restoration actions, Takekawa et al. (2015) monitored water quality at several points in pond A16; this included placing a minisonde adjacent to the outlet where pond water was exchanged between A16 and A17. Although the degree of influence by A17 on this data is unclear, Takekawa et al. noted periods of extremely low DO (<1.0 mg/L) and large fish kills, although DO was >4.2 mg/L for most of the study period. This study represented water quality immediately after restoration began. In addition to more than a decade of restoration efforts, pond A17 has also undergone a shift in management style since the removal of the former pond control structure and creation of a more tidal flow regime since Takekawa et al. (2015)'s study.

Previous water quality research focused on pond A21 indicates that it exhibits some of the lowest DO levels amongst all ponds. Previous studies of water quality grab samples from 2010-2012 indicated average DO of 5.7 mg/L, with peaks of 2.6 mg/L and 10.5 mg/L (Hobbs, 2012). However, continuous monitoring in A21 during Summer of 2013 indicated that A21 is vulnerable to periods of extremely low DO (Amato & Valoppi, 2015). Amato and Valoppi (2015) found that the pond experienced hypoxic conditions (less than 2 mg/L of DO) during 18 out of 32 total days of monitoring. They also noted that A21's low volume, minimal exchange with the open bay and close proximity to the San José-Santa Clara Regional Wastewater Facility (SJSCRWF) probably caused these consistent hypoxic conditions. DO was also correlated with tide height, with lower tides producing lower DO values (Amato & Valoppi, 2015).

## 2.4 Adaptive Management

Adaptive Management of natural resources was first developed in the 1970s as a more effective approach to managing systems with high degrees of uncertainty. Definitions of adaptive management can vary, but Doremus et al. (2011) argue that adaptive management plans should contain the following elements:

- Overarching goal of reducing uncertainty over time
- Explicit goals and measurable indicators of progress
- Iterative approaches to decision making and the opportunity to adjust as more information becomes available
- Systematic monitoring
- Feedback loops, continual assessment, and systematic learning that can be included in future decision making.

- Acknowledgement and characterizing of risks and uncertainties

In addition to the above points, others have urged greater involvement from a variety of stakeholders within the entire adaptive management process. McFadden, Hiller, and Tyre, (2011) found that stakeholder involvement was an element of the majority of successful adaptive management plans. Gunderson (1999) noted that adaptive management has the power to open up new means of communication between stakeholders, although extant tensions or institutional inflexibility among some stakeholder groups can impede management success.

Adaptive management can be an effective tool for managing complex systems, where the precise outcomes of different management actions are rarely certain. However, as Doremus et al. (2011) noted, adaptive management can also fail when plans are applied to poor candidates, such as areas where uncertainty is unlikely to be resolved, risks of long term damage are high, or plans lack sufficient enforceability. Eberhard et al. (2009) noted that there are additional complexities involved in applying adaptive management practices to water quality given that many benefits to water quality are realized more slowly than benefits to other systems. They also noted that while adaptive management models have proven successful in previous applications, it's uncertain whether those models were effective at targeting improvements to water quality (Eberhard et al., 2009).

### *2.4.1 Adaptive Management in the SBSPRP*

The SBSPRP has a strong adaptive management component with a high degree of stakeholder involvement, informed decisions made in the absence of scientific certainty, and a built-in timeline for assessment and adjustment in management planning (Truilio, Clark, Ritchie, Hutzel, & The South Bay Salt Pond Restoration Project Science Team, 2015). The SBSPRP's managers have utilized adaptive management since the outset of project planning, which is evident in the infrastructure of the restoration planning itself. The SBSPRP is organized in phases, with organizational infrastructure in place to make minor changes in day-to-day operations during each restoration phase, and larger changes with input from various stakeholders when planning the next phase (EDAW et al., 2007). The SBSPRP also takes public comments on all of its management plans and arranges meetings every two years to present its progress and solicit additional feedback from various members of the public and the scientific community (Bourgeois, 2015b).

The SBSPRP's Initial Stewardship Plan (ISP), in which some initial actions were taken to determine potential impacts of restoration, ran from 2004-2007. At the conclusion of the ISP, the SBSPRP managers determined that there was sufficient certainty and minimal risk to continue with additional restoration action (Truilio et al., 2015). In 2007, Phase 1 of the AMP was finalized and restoration actions were expanded (Bourgeois, 2015b). Plans for Phase 2 are being finalized as of 2016 (South Bay Salt Pond Restoration Project, 2016a).

## 2.5 Knowledge Gaps

Despite the existence of long term monitoring in the South Bay and a major focus on scientific research within the Adaptive Management Plan, major gaps in water quality knowledge remain. Much of the water quality research within the ponds is either 10+ years old, focused on legacy pollutants such as mercury, or both (Peterson, 2006). Past studies often focus on a single pond or a cluster of adjacent ponds with similar management types. A more current study evaluating the spatial and temporal variability in water quality of restored ponds across a larger region may be of greater value in supporting management decisions, especially given that Phase 2 of the AMP is in the process of being finalized (Bourgeois, 2015b).

There is also a high degree of uncertainty in identifying the major drivers of pond water quality, including DO and nutrients, across the SBSPRP. Some factors, such as WWTP discharge, tidal elevation, and diurnal cycle are documented as major influences on pond water quality parameters such as DO and nutrient loading. However, the degree of influence, and how it changes across ponds, is less well understood. The influence on pond water quality of other factors such as surrounding land use, management type, and pond morphology have even greater uncertainty. The SBSPRP management team and the SFEI nutrient team expressed a desire to identify the major influences on pond water quality and pond-slough interactions for management purposes (Bourgeois, 2015a). A greater understanding of these drivers is of immediate concern due to the impending finalization of phase 2 and the aging infrastructure of the ponds, some of which the management team will need to replace or remove within the next few years.

### *2.5.1 Research Objectives*

In order to address some of the gaps listed above, this study was undertaken with the following objectives:

- Determine the spatial variability of DO, nutrients, and other water quality parameters across ponds.
- Determine how the above parameters vary with different tidal conditions, especially spring and neap tides.
- Determine what factors may be driving changes tin water quality, such as pond morphology, management regime, and surrounding land use.
- Determine how this information could be applied to future AMP decisions and other estuarine restoration work.

## 3. Methods

In order to address research questions posited above, this study was broken up into three phases: collection of *in situ* water quality data including DO and nutrients, GIS analysis and computation of supplemental parameters including land use and pond morphology, and analysis of the water quality data in conjunction with the supplemental data, including comparison to historic data.

### 3.1 Study Area

The study area is a sample of ponds within the Alviso section of the South Bay Salt Pond Restoration Project (SBSPRP) (Figure 2). Given the size of the study area and the total number of ponds, including 27 distinct ponds in the Alviso complex, it was necessary to select a small subset of ponds to act as case studies. Ideally, these ponds would represent the diversity of management regimes, pond morphologies, surrounding land use, and slough interactions present within the Alviso section of the SBSPRP.

In addition to the above considerations, ponds were selected based on the following criteria:

- Existence of previous water quality data related to that pond, including DO
- Logistical access by vehicle (for kayak launching) and boat (for data collection)
- Sufficient time since restoration efforts began (5+ years).
- Management by the Don Edwards Wildlife Refuge, to ensure access via special use permit

Ultimately, four case study ponds were selected: ponds A3W, A8, A17, and A21, as described below (Figure 3).


**Figure 3:** Partial map of the Alviso section of the SBSPRP with case study ponds A3W, A8, A17, and A21 noted. Subset shows location within the South SF Bay.

### *3.1.1 Pond A3W*

Pond A3W is classified as a managed pond in the eastern portion of the Alviso pond complex. It is connected to Guadalupe Slough on its westernmost side, and is adjacent to the Sunnyvale Water Pollution Control Plant, which discharges treated wastewater to Guadalupe slough via a holding pond with multiple aeration devices. Pond A3W is also connected to a small pond strip in its southeastern corner; this strip runs parallel to both the wastewater holding pond and the Moffett Channel (which receives runoff from the nearby Golf Club at Moffett Field). Restoration efforts in A3W began in 2004. Pond A3W has two inlets and an outlet with three separate culvert-style breaches, although one was defunct at the time of this study (Cheryl Strong, personal communication, June 2nd, 2016).

***3.1.2 Pond A8***

Pond A8 is classified as a managed pond near the center of the Alviso pond complex, with high degrees of connectivity to nearby ponds A8S, A7, and A5. Pond A8 has one semi-managed breach that opens onto Alviso Slough, which acts as an inlet or an outlet depending on tidal conditions. However, A8 is a "muted tidal", managed pond, meaning flow doesn't directly follow tidal changes (South Bay Salt Pond Restoration Project, 2016a). Pond A8 is the only tidal pond in this study not immediately influenced by wastewater treatment plant discharges, although it may be influenced by pollutants connected to the Alviso Marina or other urban sources.

Restoration efforts in pond A8 began in 2010 with the installation of a pond "notch" control structure in the southeastern corner (Helton et al., 2011). The other water exchange in the pond occurs via two smaller breaches along the western edge of the pond and via diffuse exchange along its western and southern edges, where previous levees and hunting roads are deteriorating and allowing free water exchange between ponds. After the sampling occurred, it was determined that restoration work had been undertaken to fill portions of A8S (the small pond immediately below A8) with loads of dirt from a nearby construction site (South Bay Salt Pond Restoration

Project, 2016b). The filling didn't occur at the same time as water sampling, but the effect of this restoration work on the data collected is difficult to determine.

### *3.1.3 Pond A17*

Pond A17 was previously classified as a managed pond, although it has undergone significant changes in its management regime. Pond A17 was originally a managed pond, with a pond control structure installed to regulate flow between Coyote Creek and the pond; water was also exchanged between A16 and A17 (EDAW et al., 2007). In 2013, the pond control structure that used to regulate flow between A17 and Coyote Creek was removed, allowing much greater tidal influence (Wolfe, 2012). A new control structure was built to regulate flow between A16 and A17. Pond A17 is characterized by a deep outer trench and a very shallow center, attracting significant numbers of waterfowl to rest when the tide lowers and exposes the mudflat.

### *3.1.4 Pond A21*

Pond A21 was restored via two open breaches connecting the pond to Coyote Creek in 2006. These breaches have increased in size over time, but the eastern breach is still disconnected during low tide (as observed on June 26$^{th}$ and July 10$^{th}$). During Phase 2 planning, the possibility of creating additional breaches in the northern portion of A21 is being discussed.

Pond A21 was one of the first ponds restored to a full tidal flow, and is considered by project managers to be one of the successes thus far of the SBSPRP

(Krieger, 2015). Pond A21 has a large amount of sediment deposition and vegetation growth, and was the site of the first endangered Clapper Rail sighting in the SBSPRP since salt pond restoration began (Krieger, 2015).

## 3.2 Pond Sampling

### *3.2.1 Permit for Special Use*

Prior to any pond sampling, a Special Use Permit for Research and Monitoring was obtained from the USFWS. The permit was approved by Refuge Biologist Cheryl Strong on April 20$^{th}$, 2016. All conditions and requirements listed within the permit were followed during the course of this study.

### *3.2.2 Sampling Period*

The study was conducted during early Summer 2016 on June 18$^{th}$-19$^{th}$, June 25$^{th}$-26$^{th}$, July 2$^{nd}$-3$^{rd}$, and July 10$^{th}$-11$^{th}$. These dates represented (approximately) two spring tides, when tides are stronger with higher highs and lower lows, and two neap tides, when tides tend to be weaker with less difference in elevation (NOAA, 2016). Each pond was sampled on one day of each sampling "week" (e.g. pond A3W was sampled on June 18$^{th}$ but not June 19$^{th}$). The sampling period was similar to previous water quality studies in the area, which all took place during summer. See Table 1 below for ponds sampled and tidal conditions during each 2016 sampling date.

### *3.2.3 Pond Water Quality Parameters*

The following water quality parameters were examined for ponds A3W, A8, A17 and A21: DO, temperature, pH, conductivity, clarity, nitrate ($NO_3^-$), and ammonium ($NH_4^+$). DO, temperature, pH, and conductivity were measured using a YSI 556 Multi-Parameter System (YSI Incorporated, 2009) at every sampling point. The longitude and latitude of each sampling point was captured using a Juno SC GPS (Trimble, 2008). The depth of the pond water at each sampling point was measured using an anchor rope marked every two decimeters with electrical tape listing the distance from the bottom of the anchor.

During the first two sampling periods (June 18$^{th}$, 19$^{th}$, 26$^{th}$, and 27$^{th}$) grab samples of pond water were collected for nutrient analysis. These samples were analyzed for nitrate and ammonium at the UC Davis Analytical Laboratory (UC Davis Analytical Lab, 2010) on July 8$^{th}$. Samples were collected, filtered, stored, and shipped in accordance with the Analytical Lab's Sampling and Preparations Instructions (UC Davis Analytical Lab, 2014).

During the final two sets of sampling dates, clarity for each sampling point was measured using a Secchi disk. The turbidity measurement was skipped in cases where the pond bottom was clearly visible, or the flow was too rapid for the Secchi disk to sink downwards.

**Table 1**: Sampling dates and tidal conditions

| Date | Ponds Sampled | Tide Type | High Tide during sampling | Low Tide During Sampling |
|---|---|---|---|---|
| June 18th | A17; A21; CC | Spring | 1:08 pm | - |
| June 19th | A3W; A8 | Spring | 1:55 pm | - |
| June 26th | A17; A21; CC | Neap | - | 11:28 am |
| June 27th | A3W; A8 | Neap | - | 12:16 pm |
| July 2nd | A17; A21; CC | Spring | 12:31 pm | - |
| July 3rd | A3W; A8 | Spring | 1:28 pm | - |
| July 10th | A3W; A8 | Neap | - | 12:07 pm |
| July 11th | A17; A21; CC | Neap | - | 12:51 pm |

### *3.2.4 Sampling Regime*

Water was sampled at both the inlet and the outlet of each pond during each sampling period. For the purposes of this study, an inlet is defined as either a pond control structure or levee breach where the dominant direction of flow was into the case study pond; conversely, an outlet is defined as a point where the dominant direction of flow is out of the pond. Some breaches (most notably at ponds A17 and A21) can act as either an inlet or an outlet depending on the tidal conditions (Topping et al., 2013). During this study, any sampling point labeled as an inlet has a dominant direction of flow (meaning the majority of the water volume) into the pond *at the time of sampling*, and vice versa for points labeled as an outlet.

For ponds with multiple inlet and/or outlet points, the inlet and/or outlet selected appeared to have the greatest volume of water exchanged in or out of the pond at the time of sampling. This was especially difficult to assess for A8, which has an increasing amount of free pond water exchange as older internal levees continue to break down. For A3W, multiple inlets were sampled for DO and other *in situ* data, but only the largest inlet was sampled for nutrients due to limitations in fiscal resources.

Some supplemental sampling points in addition to pond inlet and outlet were examined. *In situ* water quality data and several nutrient samples were recorded for Coyote Creek, which has free tidal exchange with ponds A17 and A21 in addition to receiving the discharge from the neighboring San José-Santa Clara Regional Wastewater Facility (SJSCRWF). This data was collected to further examine the interplay between pond water and slough water, and the role of wastewater treatment plant discharges in those interactions.

Some additional samples were collected within pond A3W for comparison with Topping et al. (2013). Samples were collected at a deep and a shallow point within the pond at approximately the same points where Topping et al. collected their deep and shallow samples. During weeks when an algal mat of significant size was present and easily accessible from another sampling point within the pond, *in situ* water quality data was collected near the algal mat as well (although never at the same area where Topping et al. collected their algal mat data). See Table 2 and Figures 4A-4C below for sampling locations.

**Table 2**: Sampling points by pond/slough

| **A3W** | **A8** | **A17** | **A21** | ***Coyote Creek** |
|---|---|---|---|---|
| *1- Inlet (North)* | *1 - Inlet* | *1 - Inlet* | *2 – Inlet* | 3 - West of A21 |
| 2- Inlet (South) | *2 - Outlet* | *1 - Outlet* | *2 - Outlet* | *4 - East of A17* |
| *3 - Outlet* | | | | |
| 4 - Shallow | | | | |
| *5 - Deep* | | | | |
| Algal** | | | | |

*Sampling points in italics also had water samples collected for nutrient analysis. Numbers correspond with sampling point locations in Figures 4A-4C.*

*Some sampling dates have repeat samplings for Coyote Creek at different tidal conditions

**Point was only sampled when algal mat was present and easily accessible; algal mat drifted between sampling dates and was not necessarily the same algal mat as in previous weeks.

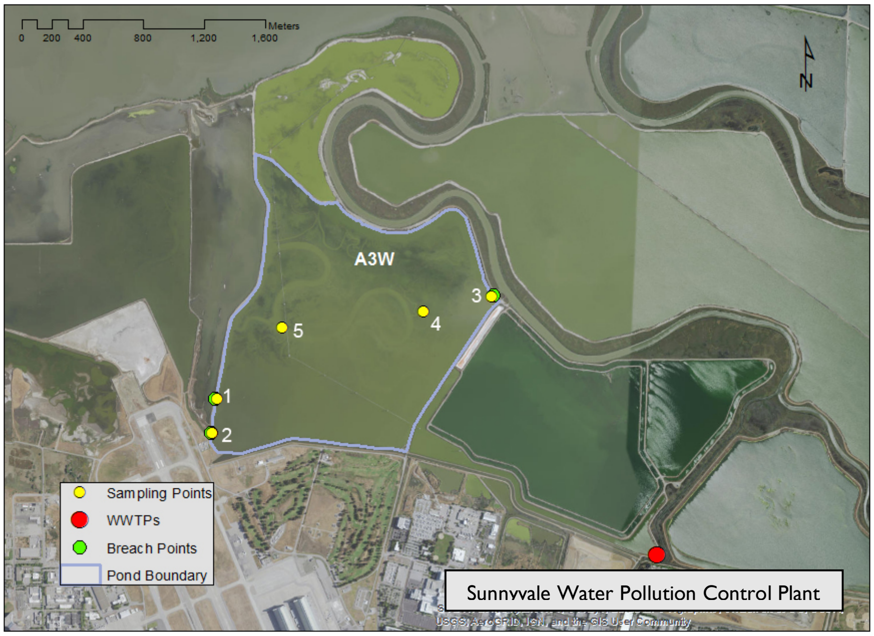



**Figure 4A**: Map of sampling points in pond A3W. Sampling points include: 1) north inlet, 2) south inlet, 3) outlet, 4) shallow, and 5) deep.


**Figure 4B**: Map of sampling points in pond A8. Sampling points include: 1) inlet, and 2) outlet.

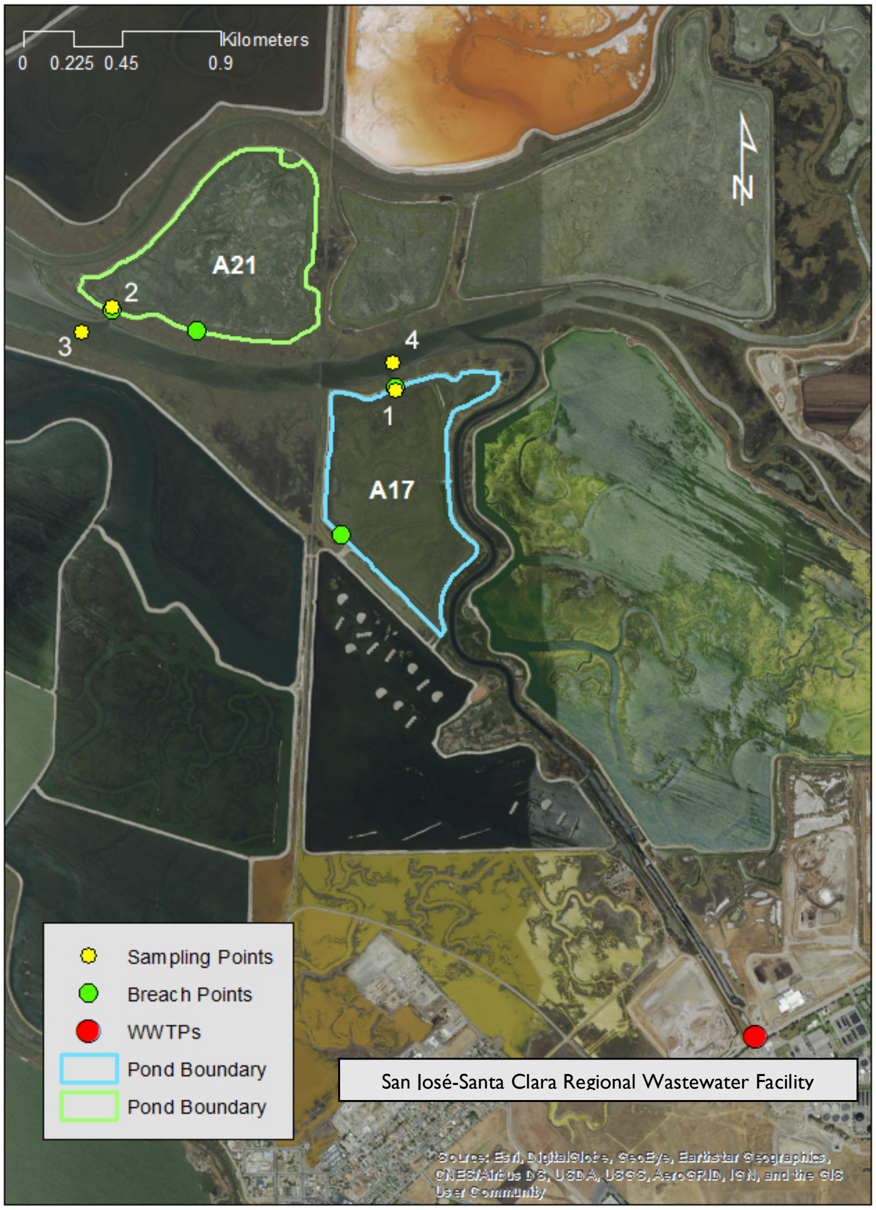



**Figure 4C:** Map of sampling points in ponds A17, A21, and Coyote Creek. Sampling points include: 1) A17 inlet/outlet, 2) A21 inlet/outlet, 3) Coyote Creek west of A21, and 4) Coyote Creek east of A17.

*3.2.4.1 Sampling Regime in Tidal Ponds*

Ponds A17, A21, and Coyote Creek were always sampled on the same dates for purposes of consistency and logistical feasibility. Because the breaches for both A17 and A21 act as both inlets and outlets, sampling was done both before and after maximum tide. Samples were collected in the morning for each breach and in Coyote Creek when the dominant direction of flow for both pond breaches and the slough was consistent; sampling was halted during the midday slack tide when the dominant direction of flow changed. Once the flow was consistent in the new direction in both breaches and Coyote Creek, sampling resumed at each sampling point.

*3.2.4.2 Sampling Regime in Managed Ponds*

Non-tidal ponds A3W and A8 were always sampled on the same days for consistency and ease of access. Pond A8's outlet and inlet were sampled early in the morning before the dominant direction of flow within the pond changed. Because A8 is a managed pond with significant influence from the concrete breach to the east and other neighboring ponds to the west and south, it does not directly follow the tidal patterns in neighboring Alviso Slough. During every sampling event during this study, the concrete pond "notch" acted as an outlet and the eastern breach acted as an inlet, although the outlet became an inlet soon after sampling was completed during Spring tides. Pond A8's inlet exchanged a much smaller volume of water at a much slower rate compared to inlets in the other case study ponds; this is likely due to the deterioration of A8's internal levees allowing for diffuse water flow at points other than the breaches.

After sampling at pond A8, the two inlets near the southwestern portion of pond A3W were sampled; these inlets always had a strong dominant direction of flow into the pond. The northernmost inlet (connected to pond AB2) was also sampled for nutrients. The internal sampling points (deep, shallow, and sometimes algal) and Pond A3W outlet were sampled last; the dominant direction of flow at the outlet sampling point was always strongly away from pond A3W into Guadalupe Slough.

### *3.2.5 Sampling Methodology*

All *in situ* sampling and sample collection was undertaken by a single researcher in one-person inflatable kayak following safety protocol. The kayak was selected to allow for access to portions of the ponds not accessible by roads or on foot, easier maneuvering in shallow tidal mudflats, and reduced chances of startling local wildlife compared with a motorized vessel.

The kayak was loaded with the following equipment during sampling:

- Basket containing an anchor with pre-marked rope for depth readings
- YSI multimeter, stored in a dry-bag when not in use
- GPS Unit, stored in a dry-bag when not in use
- Field Notebook
- Map of study area
- Water-resistant radio
- Emergency tie rope
- Empty sample bottles (during first two weeks)
- Secchi Disk (during final two weeks)

Upon arrival at each sampling point, water depth was determined by dropping anchor and noting the depth at the closest tape mark to the water surface. The YSI multimeter instrument (YSI Incorporated, 2009) was unpacked and the sensor situated approximately 1 foot below the water surface. While the readings were stabilizing, GPS location and time of sampling were noted. The *in situ* water quality data for DO, pH, SC, and temperature were then recorded, with 3-4 different measurements for each parameter recorded (to be averaged later). On nutrient sampling dates, the sampling bottle was triple-rinsed with pond water and then filled with water from that sampling point. Samples were stored in a cooler or refrigerator until filtration via a 45-micron filter within 24 hours, and were subsequently frozen until shipping and analysis. For turbidity sampling dates, the Secchi disk was used to determine turbidity, although many sampling points were either two shallow or had flow too vigorous to determine a Secchi depth.

## 3.3 Computation of Supplemental Data

Information regarding supplemental parameters was extracted from a variety of datasets using GIS. These supplemental datasets are described in Table 3 below.

### *3.3.1 Pond Volume*

Pond volume was calculated in ArcMap 10.2 (ESRI, 2013) with a 25m USGS bathymetry raster (US Geological Survey, 2004). The bathymetry layer was inputted into Surface Volume tool to determine both surface area and surface volume (ESRI, 2013). Surface area calculations were compared to other estimates of pond surface area (Grossinger & Askevold, 2004) to validate volume calculations.

**Table 3**: Summary of supplemental data

| Data Files | Publisher | Publication Date | Accessed From | Date Accessed |
|---|---|---|---|---|
| ***Salt Pond + Surface Area Shapefiles*** | SFEI | 2004 | SFEI | 11/5/15 |
| ***Pond Bathymetry*** | USGS | 2004 | SFEI | 11/5/15 |
| ***WWTP Outlets*** | SFEI | 2004 | SFEI | 11/5/15 |
| ***Storm Water Facilities*** | SFEI | 2005 | SFEI | 11/5/15 |
| ***Flood Control Structures*** | Janet Sowers William Lettis and Associates | 2005 | SFEI | 11/5/15 |
| ***Major Highways*** | City of San Jose Environmental Services Dept. | 1998 | SFEI | 11/5/15 |
| ***Landfills*** | City of San Jose Environmental Services Dept. | 2005 | SFEI | 11/5/15 |
| ***Pond Breach Points*** | Cargill (updated with aerial imagery by author) | 2004; revised 2016 by author | SFEI | 11/5/15 |
| ***Land Use/Land Cover (NLCD 2011)*** | USGS | 2011 | USGS | 10/26/16 |

### *3.3.2 Land Use*

Surrounding land use was calculated for each individual pond based on the National Land Cover Database (NLCD) 2011, a 30m land use/land cover raster (Homer, Dewitz, Yang, & Jin, 2015). A 1km buffer was run around each pond outline polygon (Grossinger & Askevold, 2004), and then the pond surface area was erased from the buffer (ESRI, 2013). Data from the NLCD for that 1km buffer zone was determined using the Extract By Mask and Frequency tools (ESRI, 2013). The land use data for that buffer was then consolidated into 5 categories based on similar land use: Developed Surfaces, Non-Vegetated Surfaces, Vegetated Surfaces, Wetlands, and Open Water.

#### *3.3.3 Distance to Wastewater Treatment Plant Outfalls*

Each sampled pond inlet and outlet was projected as a feature class, in addition to the WWTP outfalls for the SJSCWRF and the SWPCP. In order to measure the distance potential pollutants would need to travel by slough to arrive at an inlet/outlet, new polylines were digitized with the assistance of a base map to connect the breach to the closest wastewater treatment plan via the bay or a slough. The length of each polyline was then calculated (ESRI, 2013).

#### *3.3.4 Other Supplemental Data*

Pond management regime was ascertained from the SBSPRP's annual reports (Bourgeois, 2014) and environmental impact statements (EIS) (EDAW et al., 2007; South Bay Salt Pond Restoration Project, 2016a), in additional to personal communication with Refuge staff. Management type was partially based on the two restoration goal types (tidal and managed) in the Adaptive Management Plan and refined for the case study ponds at hand. Management types were coded as:

1. Tidal (A21; completely free exchange of water based on tidal flow)
2. Converted Tidal (A17; where previous pond management structures have been removed and water flow is now dominated by tidal conditions)
3. Diffuse Managed (A8; where water exchanges partially through pond control structures and some flow occurs outside of pond control structures)
4. Managed (A3W; where all water flow moves through pond control structures)

Additionally, the following data sets were used for data visualization purposes: landfill locations (City of San Jose Environmental Services Department, 2005), storm water facilities (Sowers & William Lettis and Associates Inc, 2005b), flood control infrastructure (Sowers & William Lettis and Associates Inc, 2005a), and major highways (San Jose Environmental Services Department, 1998).

### 3.4 Analysis of Collected and Supplemental Data

Variation in DO within each pond was examined using linear regression to determine possible correlations with the following:

- Nutrient levels (Nitrate and Ammonium)
- Temperature
- Clarity
- Depth at sampling point
- Pond Volume
- Distance to wastewater treatment plant (WWTP) outfalls
- Surrounding land use
- Management regime

DO and other water quality parameters were also compared to previous research for each pond, as well as to water quality data from the surrounding sloughs when available.

## 4. Results

### 4.1 Dissolved Oxygen

DO results at all sampling locations for each of the sampling weeks -- June 18$^{th}$-19$^{th}$ and July 2$^{nd}$-3$^{rd}$ (both spring tides) and June 26$^{th}$-27$^{th}$ and July 9$^{th}$-10$^{th}$ (both neap tides) -- are shown in figures 5A-5B. Symbol colors correspond to DO management thresholds: significantly above the Regional Management Plan (RMP) threshold, >7.5 mg/L; at or above RMP threshold (California Regional Water Quality Control Board San Francisco Bay Region, 2007), 5.0-7.5 mg/L; below RMP threshold but at or above Adaptive Management Plan (AMP) threshold (Helton et al., 2011), 3.3-5.0 mg/L; below the AMP threshold, 2.5-3.3 mg/L; and significantly below AMP threshold, <2.5mg/L (Figure 5A-5B).

For each sampling week, the managed ponds A3W and A8 had DO concentrations that ranged from 3.45 mg/L to 9.25 mg/L; this was higher than DO in samples collected from tidal ponds A17 and A21, where DO ranged from 2.48 mg/L to 5.54 mg/L. Pond A8 had the highest DO measurements, with all sites exceeding the RMP DO threshold of 5.0 mg/L. Pond A3W exhibited more spatial variability in DO levels, with lower DO values at the outlet and deep (water depth range = 1.4 to 2.0 m) sampling points, and higher DO values in the inlet and shallow (water depth range = 0.4 to 0.6 m) sampling points. The DO concentrations in A3W were frequently below the RMP threshold, but no sample fell below the AMP threshold.

DO concentrations for ponds A17 and A21 ranged from 2.48 mg/L to 5.54 mg/L, lower DO values than samples from the managed ponds, especially during neap tides. With the exception of the first week of sampling, June 18th-19th, DO levels even in the managed ponds were seldom above the RMP threshold (5.0 mg/L), and often fell below the AMP threshold (3.3 mg/L) as well. The largest decreases in outlet DO levels occurred during neap tides; during spring tides, the DO levels in A17 and A21 generally increased from inlet to outlet. DO concentrations in Coyote Creek varied from week to week, but were similar to each week's tidal pond DO concentrations and ranged from 1.97 mg/L to 5.34 mg/L.

The highest DO concentrations across all four ponds were measured during June 18th-19th (also a spring tide), the first week of sampling. Elevated DO levels (from 3.51 mg/L to 9.25 mg/L) were especially apparent in the managed ponds A3W and A8. Neap tide samples generally showed less variability in DO compared to samples collected during spring tides. However, the lowest DO concentrations during the study period (2.48 mg/L to 3.51 mg/L) were measured during a neap tide on June 26th in the tidal ponds.

Each pond's influence on water quality was estimated by comparing the difference in outlet DO levels to inlet DO levels with the following equation:

$$\text{Change in DO} = \text{Outlet DO} - \text{Inlet DO}.$$

Change in DO was utilized to examine the potential influence of individual pond characteristics (such as management type or volume) on water quality. The results of the regression analyses are reported in Table 8. The data are reported in Table 4.

When averaging all sampling weeks, tidal ponds showed a slight increase in DO from inlet to outlet (ranging from +0.29 to +0.30 mg/L) and managed points showed a decrease in DO from inlet to outlet (ranging from -0.79 to -2.23 mg/L) (see Table 5). In the case of A3W, there was a significant average decrease of 2.23 mg/L, enough to push DO levels to near the 5.0 mg/L RMP threshold for DO every sampling week (see Table 4).

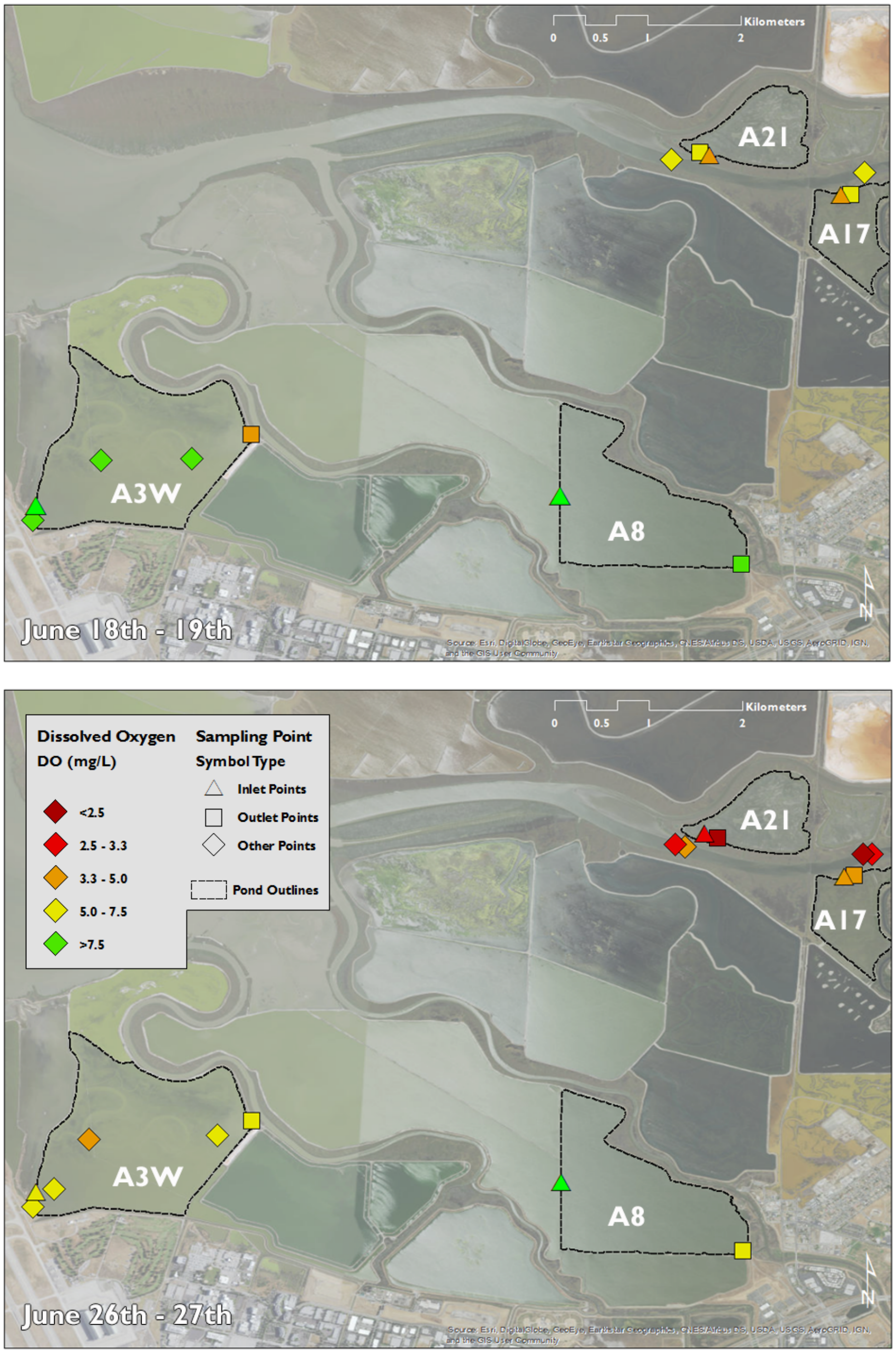



**Figure 5A**: Dissolved Oxygen across all sampling points during June 18th-19th and June 26th-27th. See Table 4 for all Dissolved Oxygen values.

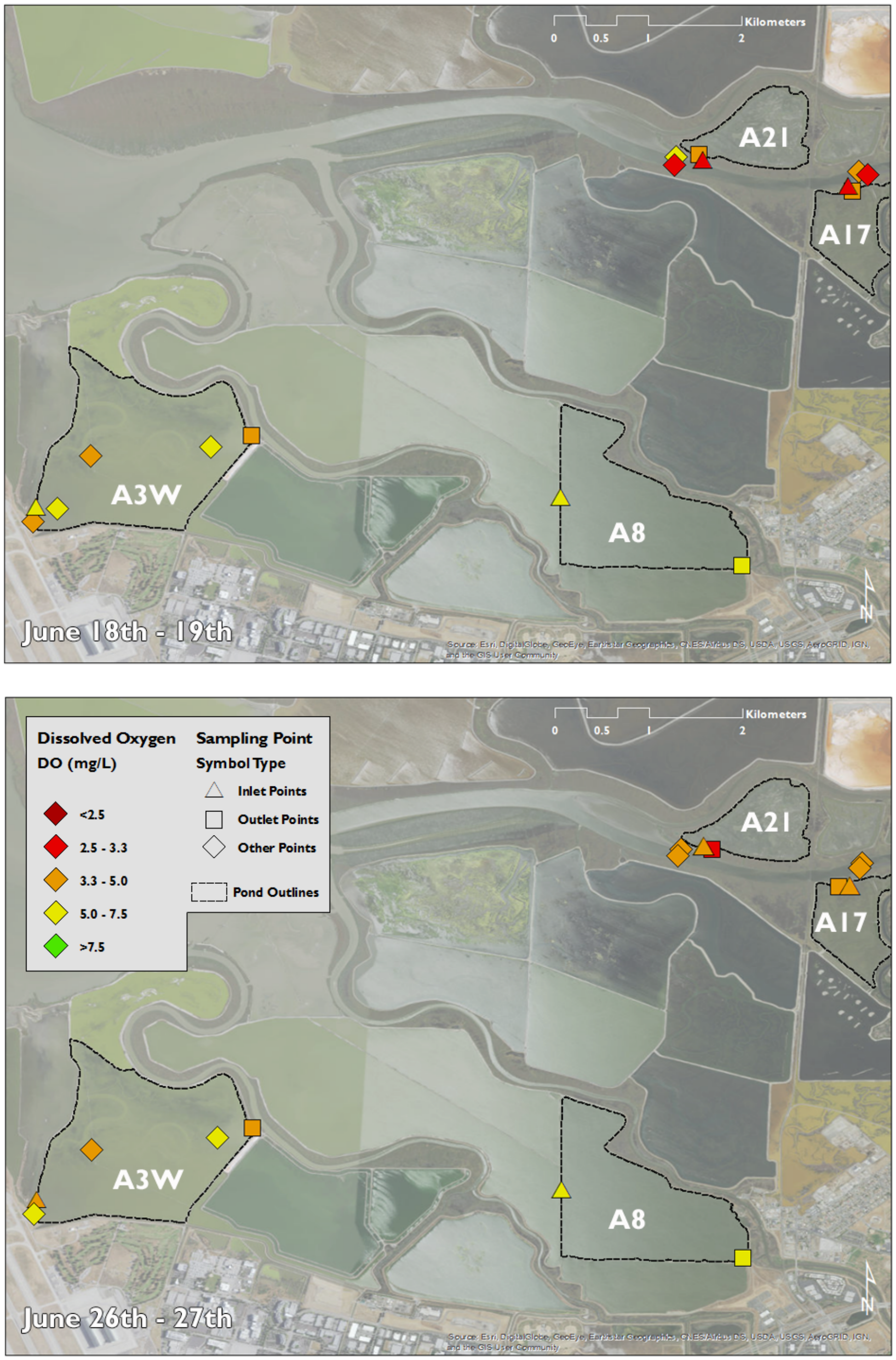



**Figure 5B**: Dissolved Oxygen across all sampling points during July 2nd-3rd and July 9th-10th. See Table 4 for all Dissolved Oxygen values.

**Table 4:** Changes in DO from inlet to outlet sampling points

| Sampling Period | Pond | Inlet DO | Outlet DO | Change in DO from Inlet to Outlet |
|---|---|---|---|---|
| June 18-19 Spring Tide | A21 | 4.32 | 5.54 | 1.22 |
| | A17 | 4.68 | 5.01 | 0.33 |
| | A8 | 9.08 | 9.25 | 0.17 |
| | A3W | 8.95 | 3.51 | -5.44 |
| June 26-27 Neap Tide | A21 | 2.89 | 2.48 | -0.41 |
| | A17 | 3.51 | 4.19 | 0.68 |
| | A8 | 8.94 | 6.61 | -2.33 |
| | A3W | 7.41 | 5.42 | -1.99 |
| July 2-3 Spring Tide | A21 | 3.10 | 4.47 | 1.37 |
| | A17 | 3.22 | 4.19 | 0.79 |
| | A8 | 7.38 | 6.42 | -0.96 |
| | A3W | 5.01 | 3.45 | -1.56 |
| July 9-10 Neap Tide | A21 | 3.88 | 2.87 | -1.01 |
| | A17 | 4.49 | 3.88 | -0.61 |
| | A8 | 6.85 | 7.08 | 0.23 |
| | A3W | 4.72 | 4.80 | 0.08 |
| *Average for all Spring Tides* | *A21* | *3.71* | *5.01* | *1.30* |
| | *A17* | *3.95* | *4.60* | *0.56* |
| | *A8* | *8.23* | *7.83* | *-0.40* |
| | *A3W* | *6.98* | *3.48* | *-3.50* |
| *Average for all Neap Tides* | *A21* | *3.39* | *2.68* | *-0.71* |
| | *A17* | *4.00* | *4.04* | *0.04* |
| | *A8* | *7.90* | *6.85* | *-1.05* |
| | *A3W* | *6.07* | *5.11* | *-0.96* |
| ***Average for All Sampling Weeks*** | ***A21*** | ***3.55*** | ***3.85*** | ***0.29*** |
| | ***A17*** | ***3.98*** | ***4.32*** | ***0.30*** |
| | ***A8*** | ***8.07*** | ***7.34*** | ***-0.72*** |
| | ***A3W*** | ***6.53*** | ***4.30*** | ***-2.23*** |

: Below RMP DO Threshold of 5.0 mg/L
: Below AMP DO Threshold of 3.3 mg/L

## 4.2 Nutrients

Samples collected during the first two weeks of the study period were analyzed for nitrate ($NO_3$-N) and ammonium ($NH_4$-N) to compare nutrient concentrations during a spring tide (June 18$^{th}$-19$^{th}$) and a neap tide (June 26$^{th}$-27$^{th}$). Spatial variability in nutrient concentrations across the four ponds was also evaluated and was compared to Coyote Creek.

### *4.2.1 Nitrate*

Nitrate concentrations are shown in Table 5 and Figure 6. Nitrate concentrations were highest in tidal ponds and Coyote Creek (ranging from 2.95 mg/L $NO_3$-N to 8.32 mg/L $NO_3$-N), and mostly fell below the detection limit (0.05 mg/L $NO_3$-N) in managed ponds. Nitrate concentrations ranging from 0.08 to 0.37 mg/L $NO_3$-N were observed at A3W outlet and during the neap tide at A8 outlet, but none of these levels hit the 1.1 mg/L $NO_3$-N concentration threshold at which fish eggs and fingerlings have been observed to be impacted (Kincheloe, Wedemeyer, & Koch, 1979; Hickey, Martin, & NIWA, 2009). In the tidal ponds, nitrate concentrations at the majority of sampling locations was greater during the neap tide than during the spring tide; average nitrate concentration increased from 3.38 mg/L $NO_3$-N during the spring tide to 5.83 mg/L $NO_3$-N during the neap tide. The highest nitrate concentration was 8.32 mg/L $NO_3$-N at Coyote Creek east of A17 breach, which was the closest sampling point to the SJSCWPCP discharge point (distance = 4512 m).

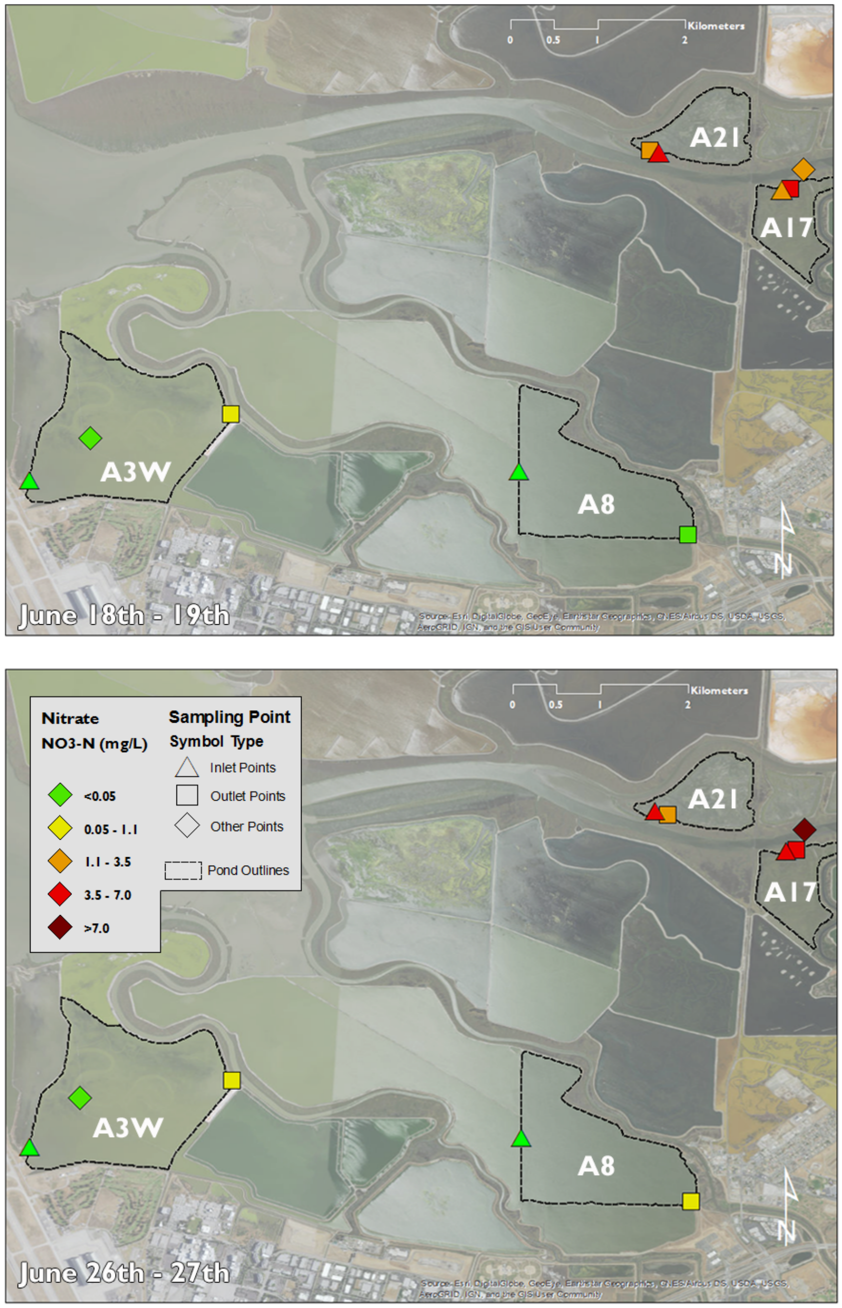



**Figure 6:** Values for nitrate (mg/L $NO_3$-N) across all sampling points. Data is symbolized in the following groups: Not Sampled; below the 0.05 mg/L $NO_3$-N detection limit; above the detection limit but below the nitrate level in which fish eggs and fingerlings can be impacted, 0.05 – 1.1 mg/L $NO_3$-N; from 1.1 mg/L $NO_3$-N to 3.5 mg/L $NO_3$-N; from 3.5 mg/L $NO_3$-N to 7.0 mg/L $NO_3$-N; and above 7.0 mg/L $NO_3$-N.

### *4.2.2 Ammonium*

Pond and Coyote Creek ammonium concentrations are shown in Table 5 and Figure 7. Ammonium concentrations were highest in tidal ponds and Coyote Creek, and below detection limit (<0.05 mg/L $NH_4$-N) in managed ponds. The only managed pond samples with detectable ammonium were from the A3W outlet, and values were relatively low (0.17 mg/L $NH_4$-N during the spring tide and 0.12 mg/L $NH_4$-N during the neap tide). In tidal ponds, ammonium levels were greater during the neap tide than during the spring tide at each sampling point. Across all tidal pond sampling points, the mean ammonium level increased from 0.21 mg/L $NH_4$-N during the spring tide to 0.36 mg/L $NH_4$-N during the neap tide. The highest ammonium concentration was 0.45 mg/L $NH_4$-N at the A17 outlet during the neap tide.

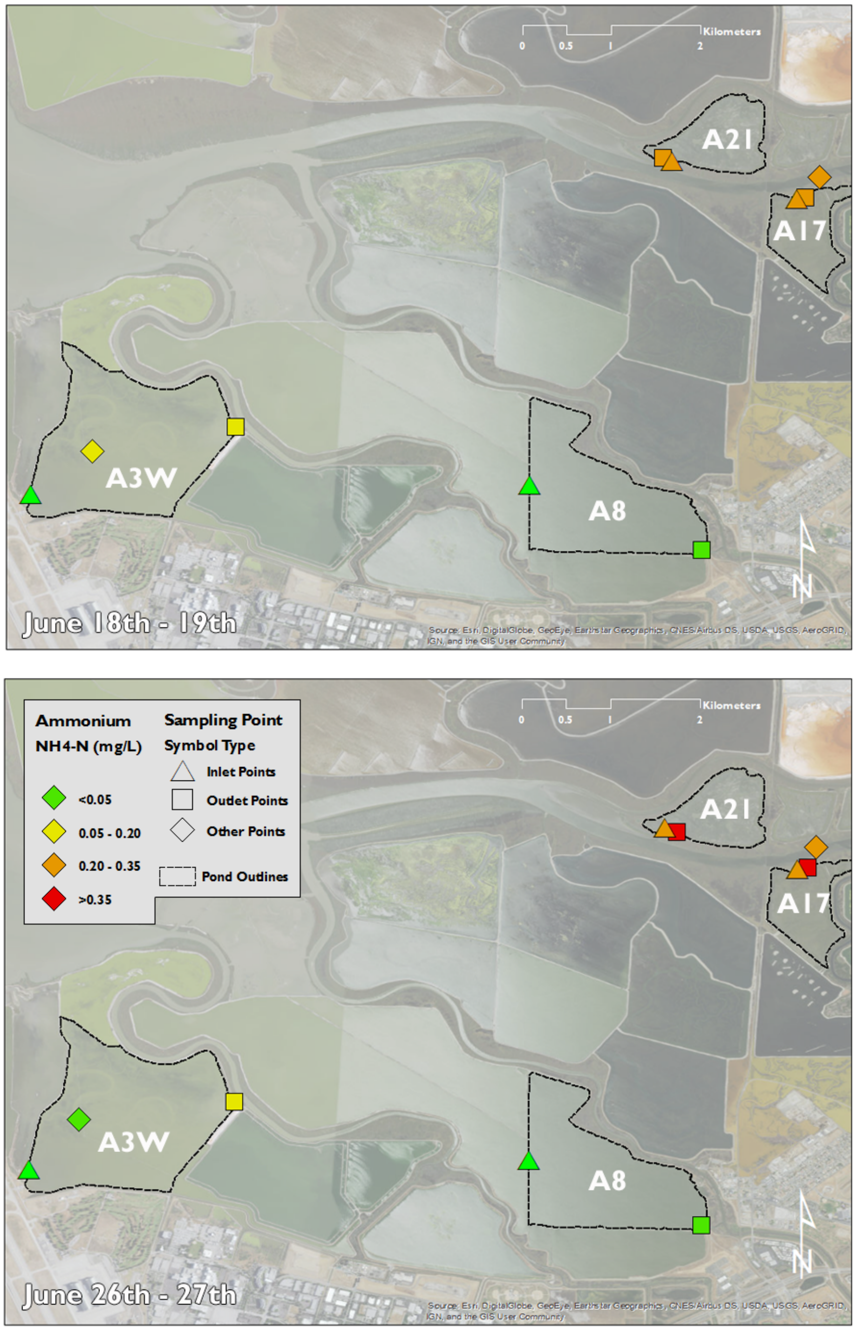



**Figure 7**: Values for ammonium (mg/L $NH_4$-N) across all sampling points. Green sampling points were below the 0.05 mg/L detection limit. Data is symbolized in the following groups: Not Sampled; below the 0.05 mg/L $NH_4$-N detection limit; from 0.05 to 0.20 mg/L $NH_4$-N; from 0.20 to 0.35 mg/L $NH_4$-N; and above 0.35 mg/L $NH_4$-N.

**Table 5**: Nutrient concentrations for all sampling points for all sampling weeks

| Sampling Period | Pond Outlet | Nitrate (mg/L $NO_3$-N) | Ammonium (mg/L $NH_4$-N) |
|---|---|---|---|
| June 18th – 19th Spring Tide | A3W Inlet | 0.05 | 0.05 |
| | A3W Outlet | 0.11 | 0.17 |
| | A3W Deep | 0.05 | 0.05 |
| | A8 Inlet | 0.05 | 0.05 |
| | A8 Outlet | 0.05 | 0.05 |
| | A17 Inlet | 3.28 | 0.21 |
| | A17 Outlet | 3.62 | 0.21 |
| | A21 Inlet | 3.95 | 0.22 |
| | A21 Outlet | 2.95 | 0.20 |
| | Coyote Creek | 3.08 | 0.21 |
| June 26th-27th Neap Tide | A3W Inlet | 0.05 | 0.05 |
| | A3W Outlet | 0.08 | 0.12 |
| | A3W Deep | 0.05 | 0.05 |
| | A8 Inlet | 0.05 | 0.05 |
| | A8 Outlet | 0.37 | 0.05 |
| | A17 Inlet | 6.93 | 0.29 |
| | A17 Outlet | 3.79 | 0.45 |
| | A21 Inlet | 6.66 | 0.34 |
| | A21 Outlet | 3.46 | 0.38 |
| | Coyote Creek | 8.32 | 0.33 |
| ***Average for All Sampling Weeks*** | ***A3W Inlet*** | ***0.05*** | ***0.05*** |
| | ***A3W Outlet*** | ***0.10*** | ***0.15*** |
| | ***A3W Deep*** | ***0.05*** | ***0.05*** |
| | ***A8 Inlet*** | ***0.05*** | ***0.05*** |
| | ***A8 Outlet*** | ***0.21*** | ***0.05*** |
| | ***A17 Inlet*** | ***5.11*** | ***0.25*** |
| | ***A17 Outlet*** | ***3.71*** | ***0.33*** |
| | ***A21 Inlet*** | ***5.31*** | ***0.28*** |
| | ***A21 Outlet*** | ***3.21*** | ***0.29*** |
| | ***Coyote Creek*** | ***5.70*** | ***0.27*** |

### 4.3 Other Water Chemistry Parameters

In addition to DO and nutrients, other water quality parameters including temperature, pH, and specific conductance were measured at each sampling location. Data from outlet points (representing the quality of pond water immediately before its discharge into sloughs and the bay) are presented in Table 6. The table identifies readings that exceeded RMP thresholds for temperature (+2.8 °C greater than the receiving water per California Regional Water Quality Control Board San Francisco Bay Region, 2007.)

Temperature ranged from 21.09-26.38 ºC during the sampling period, with pond A3W having the highest temperatures on average, and A21 having the lowest. Pond A3W experienced exceedances of the SWRQB's Thermal Plan for Enclosed Bays and Estuaries, of >2.8 ºC above receiving waters (US Geological Survey, 2001) at its outlet and shallow points during spring and neap tides. Pond A17 had greater temperatures than A21 at nearly all sampling points, and A8 had lower temperatures than A3W at all sampling points (Table 6).

Pond pH ranged from 7.43 (A21 outlet) to 9.05 (A3W south inlet) across all sampling weeks, with a clear difference between tidal pond pH and managed pond pH (Table 6). Tidal pond pH was between 7.4-7.8 for all samples, well within the RMP's pH range of 6.5-8.5 (California Regional Water Quality Control Board San Francisco Bay Region, 2007). The managed pond water was more basic and experienced exceedances of >8.5 pH during every week of sampling (although never at an Outlet point). The greatest exceedance during the sampling period occurred at the south inlet of pond A3W. Neap tides were slightly more basic on average compared to spring tides across all four ponds.

Conductivity ranged from 17.77- 45.54 millisiemens/cm (mS/cm) among tidal ponds, slightly lower for pond A8, and very high for pond A3W. Tidal ponds experienced 7-10 mS/cm increases in conductivity during the 3rd sampling week (a spring tide) but otherwise readings were relatively consistent per pond. Conductivity in the South SF Bay during late June and early July 2016 ranged from 32-45 mS/cm (US Geological Survey, 2001).

### 4.4 Depth and Clarity

Depth was recorded at every sampling location (Table 7) and varied across sampling locations and between weeks as tide levels varied. Depths varied considerably within the same general sampling area, e.g. depth at the A21 breach ranged from 1.6 – 4.2 m. This is likely due to a combination of high variability in bathymetry in the natural tidal marsh structures (Figure 1) and changes in tidal conditions. Clarity data were measured during the final two weeks of the sampling period, and were only recorded for sampling locations in which water velocity allowed for a reliable reading of the Secchi disk.

**Table 6**: Temperature, pH, and conductivity at all outlet sampling points across all sampling weeks.

| Sampling Period | Pond Outlet | Temperature (°C) | pH | Conductivity (mS/cm) |
|---|---|---|---|---|
| June 18-19 Spring Tide | A21 | 22.69 | 7.72 | 28.46 |
| | A17 | 22.26 | 7.70 | 27.65 |
| | A8 | 22.53 | 8.18 | 19.26 |
| | A3W | 25.12 | 8.39 | 40.08 |
| June 26-27 Neap Tide | A21 | 21.95 | 7.43 | 26.23 |
| | A17 | 24.23 | 7.61 | 24.94 |
| | A8 | 23.71 | 8.14 | 18.53 |
| | A3W | 26.14 | 8.01 | 43.51 |
| July 2-3 Spring Tide | A21 | 22.62 | 7.52 | 38.62 |
| | A17 | 23.92 | 7.60 | 33.33 |
| | A8 | 22.58 | 8.06 | 22.56 |
| | A3W | 23.91 | 8.24 | 43.77 |
| July 9-10 Neap Tide | A21 | 21.80 | 7.46 | 25.55 |
| | A17 | 22.26 | 7.57 | 25.02 |
| | A8 | 22.37 | 8.00 | 22.79 |
| | A3W | 23.85 | 7.95 | 45.54 |
| *Average for all Spring Tides* | *A21* | *22.66* | *7.62* | *33.54* |
| | *A17* | *23.09* | *7.65* | *30.49* |
| | *A8* | *22.56* | *8.12* | *20.91* |
| | *A3W* | *24.51* | *8.32* | *41.93* |
| *Average for all Neap Tides* | *A21* | *21.88* | *7.45* | *25.89* |
| | *A17* | *23.25* | *7.59* | *24.98* |
| | *A8* | *23.04* | *8.07* | *20.66* |
| | *A3W* | *25.00* | *7.98* | *44.53* |
| ***Average for All Sampling Weeks*** | ***A21*** | ***22.27*** | ***7.54*** | ***29.72*** |
| | ***A17*** | ***23.17*** | ***7.62*** | ***27.74*** |
| | ***A8*** | ***22.80*** | ***8.10*** | ***20.79*** |
| | ***A3W*** | ***24.76*** | ***8.15*** | ***43.23*** |

: Exceeds Thermal Plan standards by being in excess of 2.8 °C higher than receiving waters as measured at Dumbarton Bridge in the South SF Bay (US Geological Survey, 2001).

**Table 7**: Water depth and Secchi disk depth for all sampling weeks

| Sampling Period | Pond | Depth at Inlet (m) | Depth at Outlet (m) | Secchi at Inlet (in) | Secchi at Outlet (in) |
|---|---|---|---|---|---|
| June 18-19 Spring Tide | A21 | 3.2 | 2.6 | | |
| | A17 | 3.4 | 3.0 | | |
| | A8 | 1.6 | 1.2 | | |
| | A3W | 2.2 | 2.4 | | |
| June 26-27 Neap Tide | A21 | 2 | 2.2 | | |
| | A17 | 2 | 1 | | |
| | A8 | 1 | 1.8 | | |
| | A3W | 1.8 | 1.8 | | |
| July 2-3 Spring Tide | A21 | 1.6 | 4.2 | 6 | N/A |
| | A17 | 3.4 | 3.6 | N/A | N/A |
| | A8 | 1 | 1.6 | 52* | 18 |
| | A3W | 2 | 2.6 | 44 | 36 |
| July 9-10 Neap Tide | A21 | 1.6 | 1.6 | 7 | 9 |
| | A17 | 2.2 | 1.6 | 10 | 7 |
| | A8 | 1.2 | 1.8 | 30* | 26 |
| | A3W | 2.2 | 3.2 | 25 | 26 |

* indicates that Secchi disk depth extended to pond bottom.
N/A indicates that water velocity was too high to record an accurate Secchi disk depth reading.

## 4.5 Statistical Analysis

Water quality data were analyzed using linear regression for all sampling points, and by isolating spring and neap tide sampling points. This information is summarized in Table 8 and discussed in the sections below.

**Table 8:** Summary of linear regression analyses

| Parameter | All Samples | | Spring Tide Samples | | Neap Tide Samples | |
|---|---|---|---|---|---|---|
| ***vs DO*** | ***Correlation*** | ***$R^2$*** | ***Correlation*** | ***$R^2$*** | ***Correlation*** | ***$R^2$*** |
| Nitrate | Negative | 0.55 | Negative | 0.42 | Negative | 0.66 |
| Ammonium | Negative | 0.65 | Negative | 0.86 | Negative | 0.75 |
| Temperature | None | 0.00 | Negative | 0.38 | Positive | 0.16 |
| Clarity (all points) | Positive | 0.09 | Positive | 0.07 | Positive | 0.17 |
| s). Clarity (<=30 inches) | Positive | 0.60 | Positive | 0.62 | Positive | 0.63 |
| Volume | Positive | 0.09 | None | 0.00 | Positive | 0.47 |
| Surface Area | Positive | 0.09 | None | 0.00 | Positive | 0.46 |
| Depth | Negative | 0.05 | Negative | 0.36 | None | 0.00 |
| Management Type | Positive | 0.25 | Positive | 0.12 | Positive | 0.46 |
| Wetland Cover | Negative | 0.11 | None | 0.00 | Negative | 0.45 |
| Developed Cover | Positive | 0.29 | Positive | 0.07 | Positive | 0.76 |
| Distance to WWTP | Positive | 0.52 | Positive | 0.59 | Positive | 0.46 |
| ***vs Change in DO*** | ***Correlation*** | ***$R^2$*** | ***Correlation*** | ***$R^2$*** | ***Correlation*** | ***$R^2$*** |
| Change in Nitrate | Negative | 0.17 | NA | NA | NA | NA |
| Change in Ammonium | Negative | 0.13 | NA | NA | NA | NA |
| Volume | Negative | 0.38 | Negative | 0.66 | Negative | 0.10 |
| Management Type | Negative | 0.34 | Negative | 0.67 | Negative | 0.04 |
| Wetland Cover | Positive | 0.35 | Positive | 0.57 | Positive | 0.13 |
| Developed Cover | Negative | 0.38 | Negative | 0.70 | Negative | 0.08 |

NA : These analyses would have produced an n<5, and thus were excluded

### *4.5.1. DO and Nutrient Analysis*

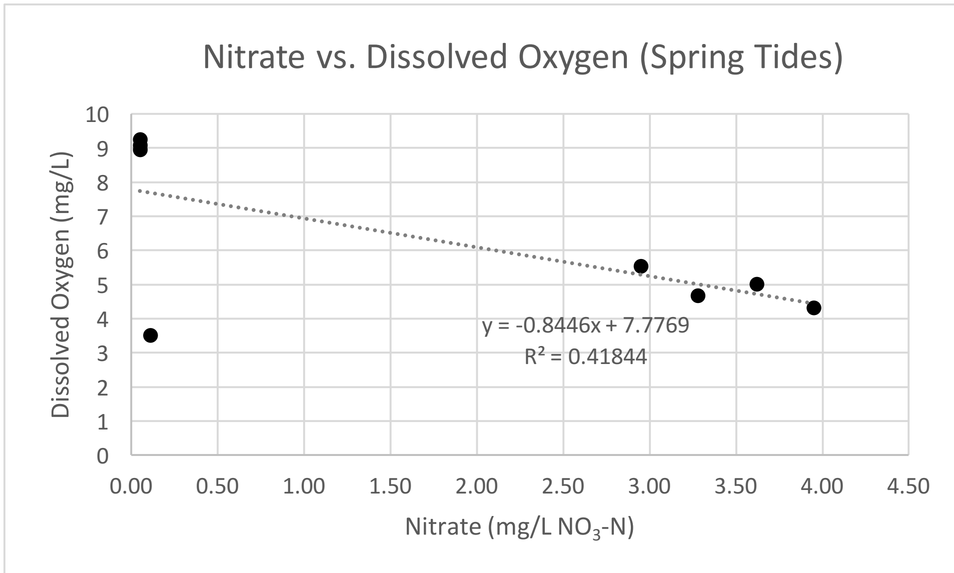
Nutrient loading can lead to eutrophication and potentially create hypoxic conditions (Cloern & Jassby, 2012). In this study, DO concentrations were negatively correlated with nitrate ($R^2$ = 0.55) (Figure 8A and 8B) and ammonium ($R^2$ = 0.65)

(Figure 9A and 9B) across all ponds. Ammonium had greater coefficients of determination than nitrate for both spring and neap sampling weeks (Table 8). Ammonium had a greater coefficient of determination during spring tide than neap tide; nitrate showed the opposite trend, with greater correlation during neap than spring tide (Table 8). During spring tide, the furthest outlier for both nitrate and ammonium compared with DO was collected at the A3W outlet, the closest sampling location among managed ponds to a wastewater treatment plant (WWTP). This was the only managed pond sampling location collected during spring tide to have nutrient levels above the detection limit (nitrate concentration = 0.11 mg/L $NO_3$-N).

Change in nitrate and ammonium concentration from inlet to outlet was calculated via the following equations:

$$\text{Change in } NO_3^- = \text{Inlet } NO_3^- - \text{Outlet } NO_3^-$$
$$\text{Change in } NH_4^+ = \text{Inlet } NH_4^+ - \text{Outlet } NH_4^+$$

Change in nutrients was calculated to examine the impact of each pond on water quality. These data are presented in Table 9 below. When examining the average change of all nitrate concentrations, ponds with a net decrease of nitrate (A17 and A21) exhibited a net increase in DO, suggesting that nitrate may have stimulated primary production. Ponds with a net increase in nitrate (A3W and A8) exhibited a net decrease in DO.



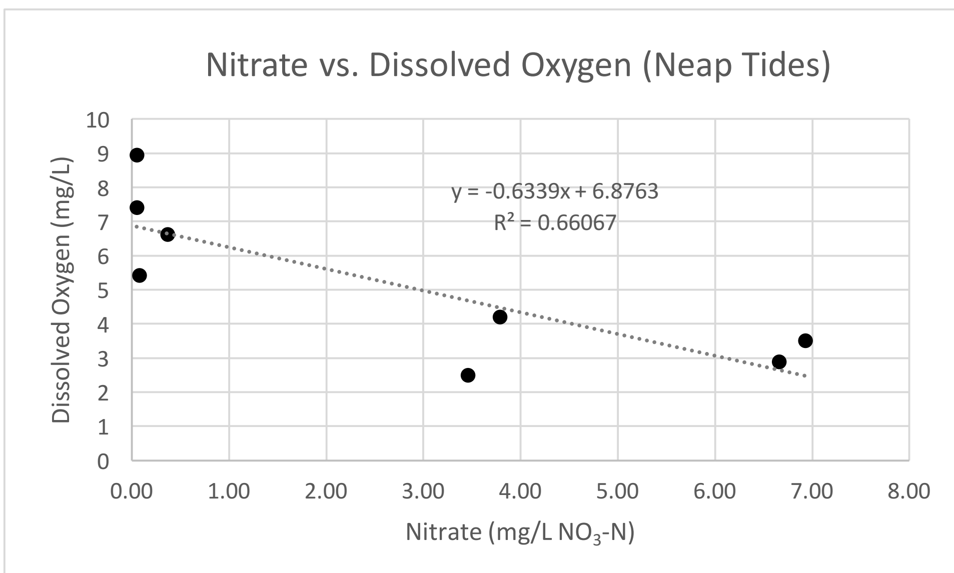
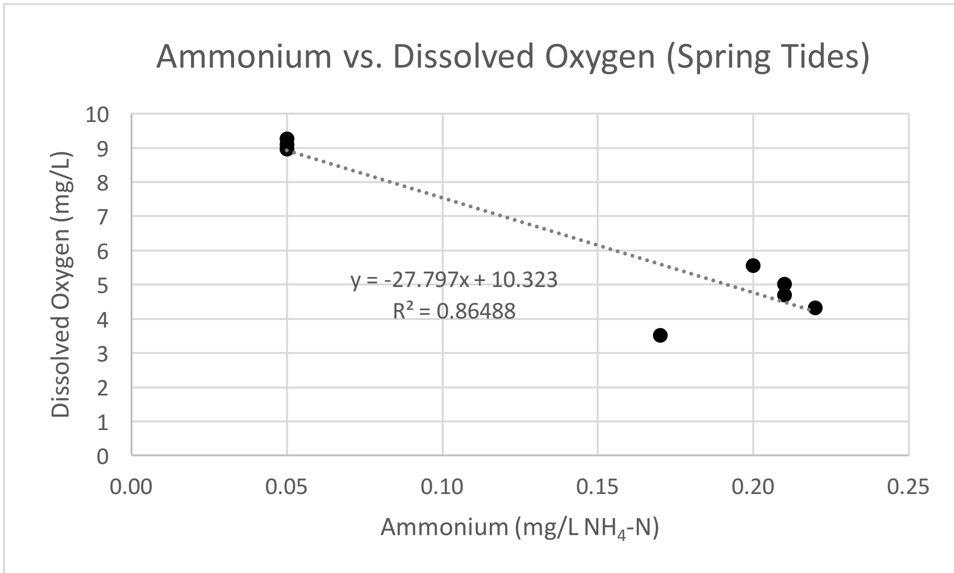




**Figure 8A**: (Top) Nitrate compared with DO for all sampling locations from June 18$^{th}$-19$^{th}$, a spring tide.
**Figure 8B**: (Bottom) Nitrate compared with DO for all sampling locations from June 26$^{th}$-27$^{th}$, a neap tide.



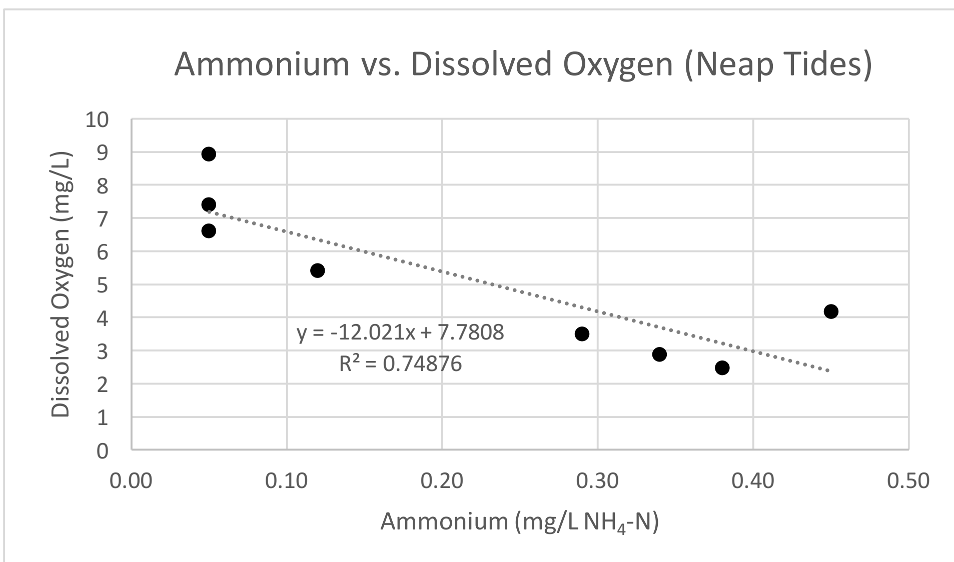




**Figure 9A**: (Left) Ammonium compared with DO for all sampling locations from June 18th-19th, a spring tide.

**Figure 9B:** (Right) Ammonium compared with DO for all sampling locations from June 26th-27th, a neap tide.

**Table 9**: Change in nutrients across all nutrient sampling points

| Sampling Period | Pond | Change in Nitrate (mg/L $NO_3$-N) | Change in Ammonium (mg/L $NH_4$-N) | Change in DO (mg/L) |
|---|---|---|---|---|
| June 18-19 Spring Tide | A21 | -1 | -0.02 | 1.22 |
| | A17 | 0.34 | 0 | 0.33 |
| | A8 | 0 | 0 | 0.17 |
| | A3W | 0.06 | 0.12 | -5.44 |
| June 26-27 Neap Tide | A21 | -3.20 | 0.04 | -0.41 |
| | A17 | -3.14 | 0.16 | 0.68 |
| | A8 | 0.32 | 0 | -2.33 |
| | A3W | 0.03 | 0.07 | -1.99 |
| ***Average for All Sampling Weeks*** | ***A21*** | ***-2.10*** | ***0.01*** | ***0.41*** |
| | ***A17*** | ***-1.40*** | ***0.08*** | ***0.51*** |
| | ***A8*** | ***0.16*** | ***0*** | ***-1.08*** |
| | ***A3W*** | ***0.05*** | ***0.10*** | ***-3.72*** |

### *4.5.2 DO and Parameters*

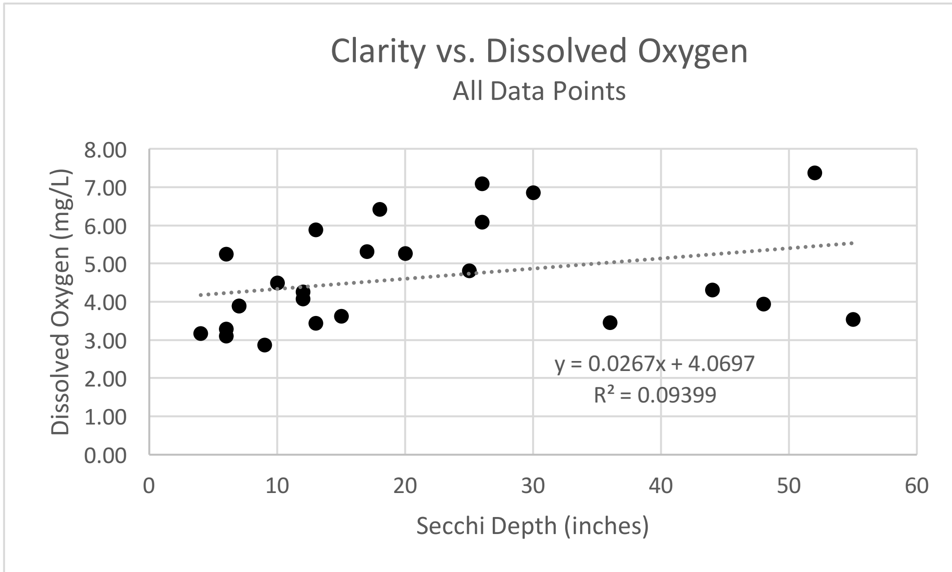
Water temperature limits the amount of oxygen that can be dissolved in a given body of water; if DO is not being limited by other factors, it will decrease with increasing water temperature (Debelius, Gómez-Parra, & Forja, 2009). When compared to DO values in this study, temperature had different patterns in differing tidal conditions: DO and temperature showed a slight negative correlation during spring tides, and a much weaker positive correlation during neap tides (see Table 8).

In west coast estuaries, there is some evidence that water depth may impact DO concentrations and water mixing (Nezlin et al., 2009). When compared to DO, depth at sampling point exhibited very weak negative correlation. When isolating spring and neap tides, correlation improved during spring tides and was virtually non-existent during neap tides (see Table 8).

Light penetration into the water column is a major control on primary productivity, which can impact DO levels during the daytime (Cloern & Jassby, 2012). Clarity has also been negatively correlated with concentrations of treated sewage effluent and chlorophyll-*a* in some estuarine systems (Glibert et al., 2010). When compared to DO levels, water clarity had a high degree of positive correlation up to 30 inches of depth. After that point, DO no longer increased with increasing Secchi disk depth (see Figures 10A-10B). When isolating spring and neap tide weeks for Secchi disk depths of 30 inches or less, the degree of correlation was similar (see Table 8).

### *4.5.3 Land Use*

Land use in the areas surrounding the ponds was analyzed via two parameters: land cover within 1km of each pond and estimated distance along a water body (slough or bay) from each pond breach point to a WWTP discharge point. Urbanization has been linked to various water quality degradations, most notably high nutrient loading (Glibert et al., 2010). Wetlands are used in some developed areas to improve water quality (Vymazal, 2010), although wetlands can cause low DO and other water quality issues (Verhoeven, Arheimer, Yin, & Hefting, 2006).



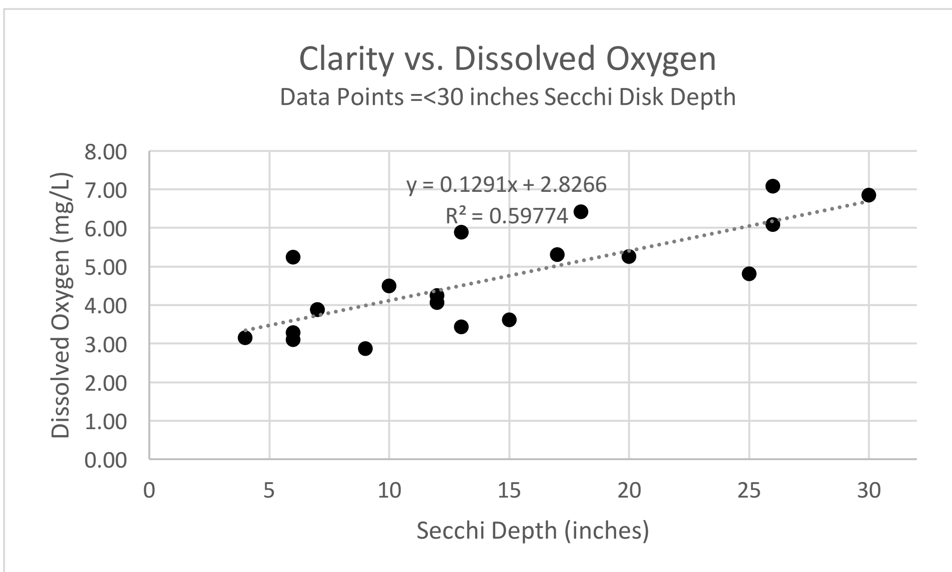




**Figure 10A** (Top): DO compared to Secchi disk depths for all sampling points. **Figure 10B** (Bottom): DO compared to Secchi disk depths that were equal to or less than 30 inches.

*4.5.3.1 Land Cover*

Figures 11A-11D below show DO concentrations for spring and neap tides in ponds surrounded by wetland or developed land. Spring tides showed very weak or no correlation for both land cover types, and neap tides had a much stronger agreement ($R^2$ = 0.45 and 0.76, respectively). Wetlands were positively correlated with DO values and developed land was negatively correlated (Table 8).

*4.5.3.2 Distance to Wastewater Treatment Plant Outfall*

DO measurements showed a strong positive correlation with distance to WWTP outfalls (Figure 12). The degree of correlation was similar during both spring and neap tides (Table 8).

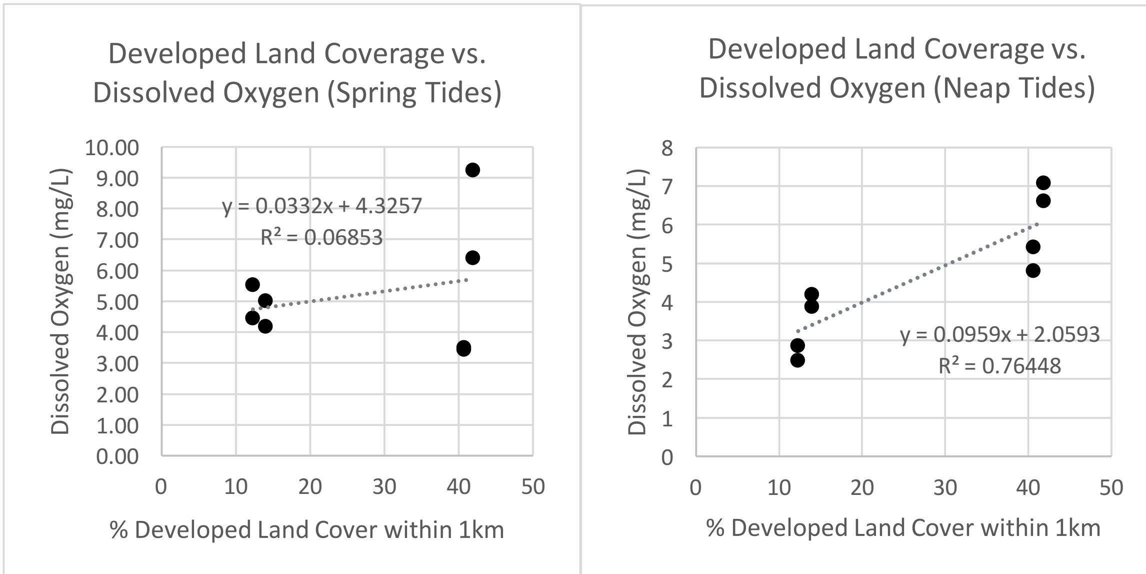



Figure **11A** (Left): DO concentrations for all spring tides by percent of developed land cover within 1km.
Figure **11B** (Right): DO concentrations for all neap tides by percent of developed land cover within 1km.

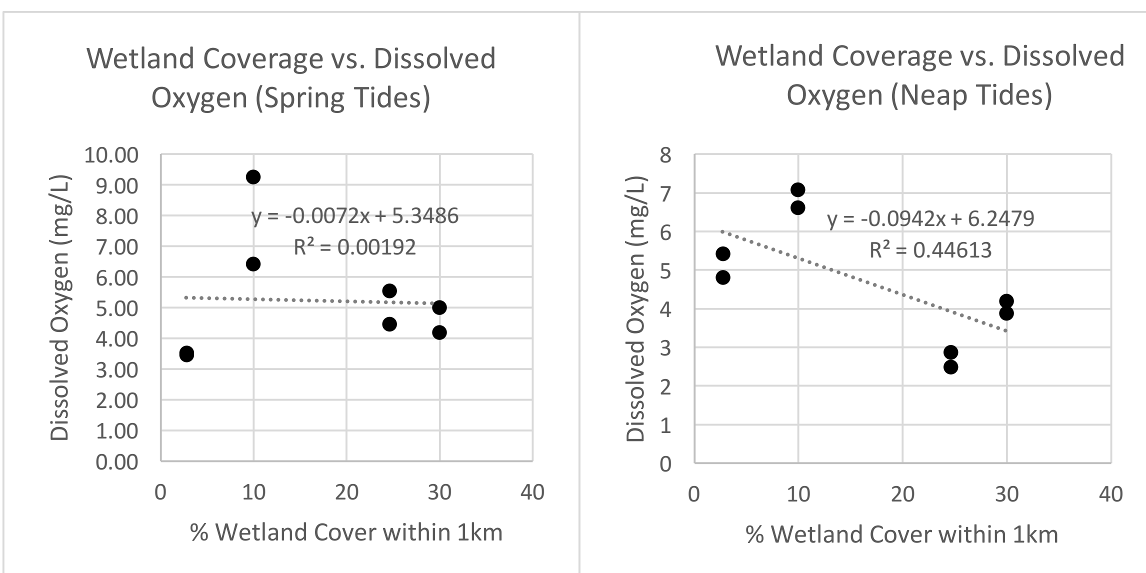



Figure **11C** (Left): DO concentrations for all spring tides by percent of wetland cover within 1km.
Figure **11D** (Right): DO concentrations for all neap tides by percent of wetland cover within 1km.

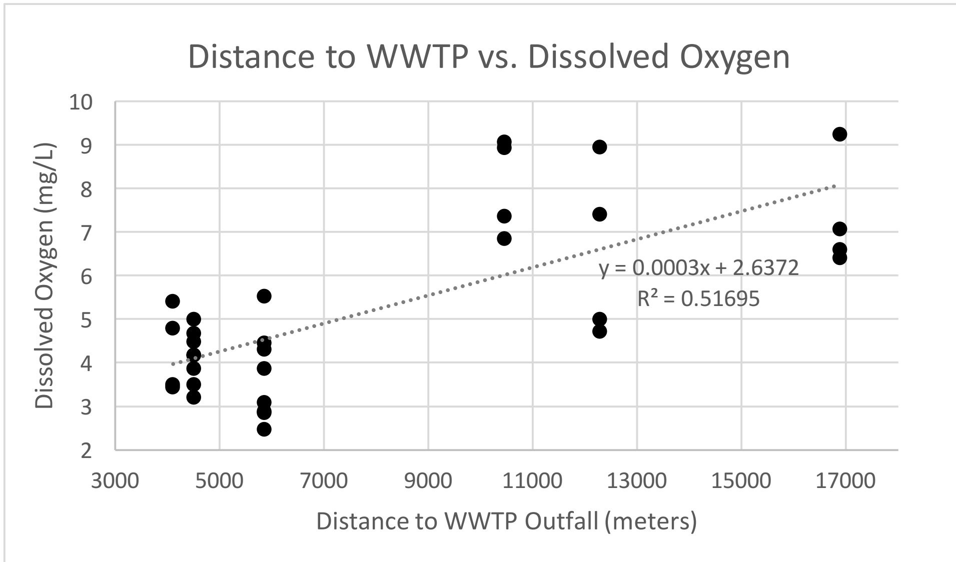
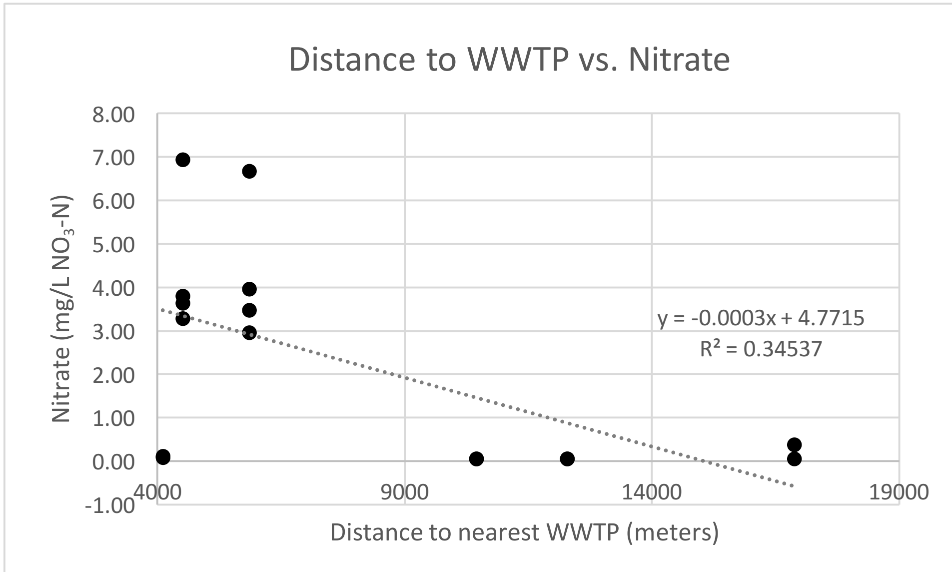



**Figure 12**: DO concentrations compared with distance from nearest WWTP outfall for all sampling weeks.

As anticipated, both nitrate concentrations and ammonium concentrations decreased with increasing distance from WWTP outfalls ($R^2$ = 0.35 and $R^2$ = 0.49, respectively; Figures 13A-13B). The vast majority of nutrient samples above the detection limit occurred at sampling points within 6,000m of a WWTP outfall (pond A17, A21, Coyote Creek and the Outlet of A3W). Only one sampling location further than 6,000m from a WWTP outfall was above the detection limit (A8 Outlet on June 27th) and that sample contained far lower nutrient concentrations than the other samples within 6,000m of an outfall.



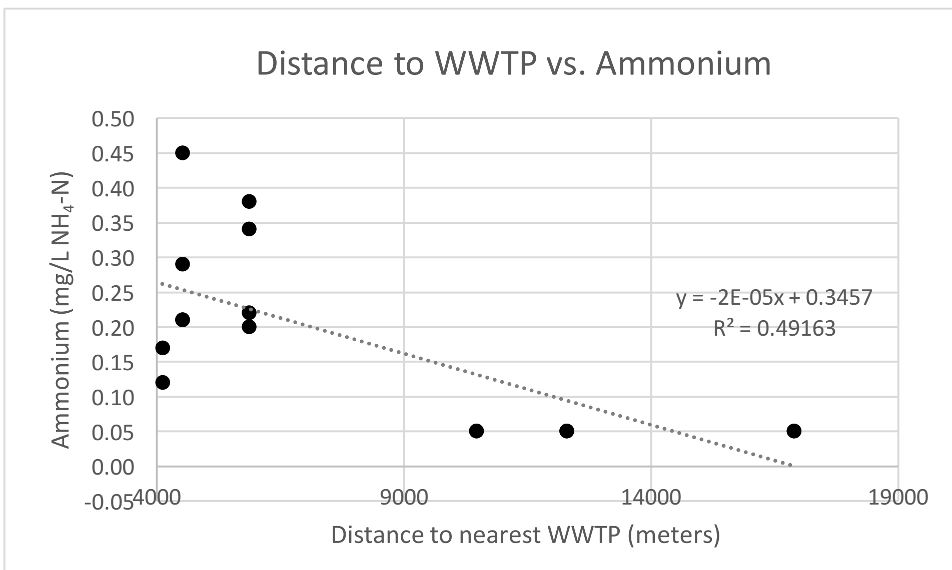




**Fig 13A:** Nitrate plotted against distance to the nearest WWTP outfall.
**Fig 13B**: Ammonium plotted against distance to nearest WWTP outfall.

### *4.5.4 Management Type and Pond Morphology*

Although minimal data exists to directly compare the impacts of different pond management regimes on water quality, the impact of pond management on water quality has been of great concern to project managers (EDAW, et al., 2007; Helton et al., 2011; South Bay Salt Pond Restoration Project, 2016b). In this study, both DO and change in DO for all sampling weeks were plotted against a qualitative scale wherein lower numbers represented ponds with the lowest degree of management intervention governing water flow and higher numbers represented ponds with the greatest degree of management intervention. DO readings showed a positive correlation with management type (Figure 14), meaning that ponds with the greatest degree of management involvement (the managed pond complex) tended to have higher DO concentrations than less-managed ponds of the tidal pond complex. This pattern was stronger during neap tides than spring tides (Table 8).

In contrast, change in DO showed a negative correlation with management type, meaning that DO tended to decrease more from inlet to outlet in managed ponds than in tidal ponds (Figure 15A). This pattern was stronger during spring tides than during neap tides (Figures 15B-15C). The tidal ponds also showed less variability in changes in DO from inlet to outlet compared to the managed ponds during the sampling period.

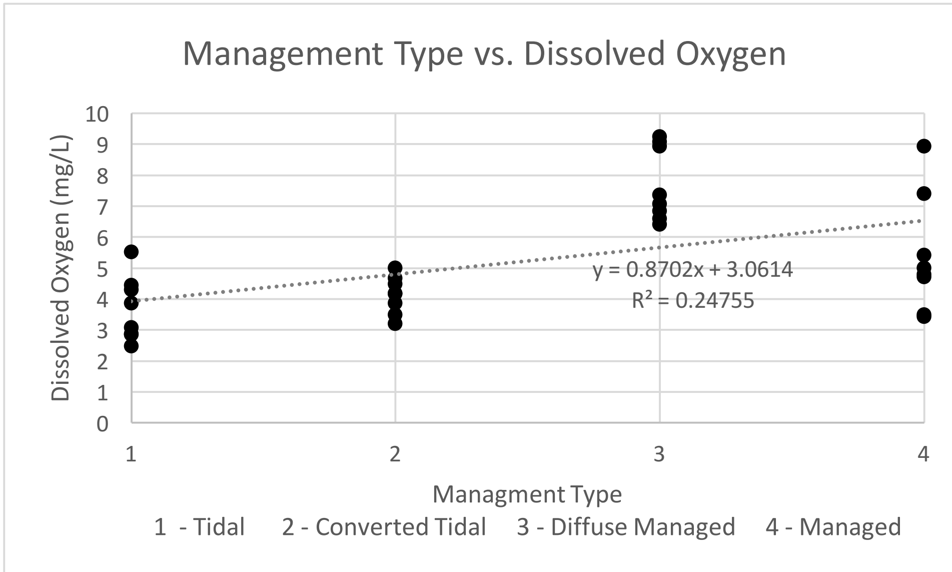



**Figure 14**: DO readings for all sampling locations plotted against a qualitative management type scale.

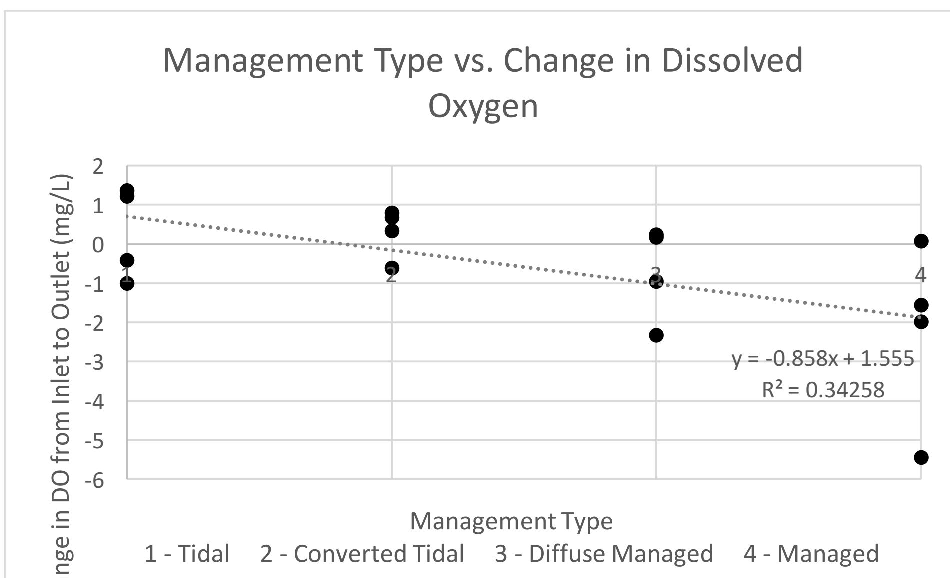
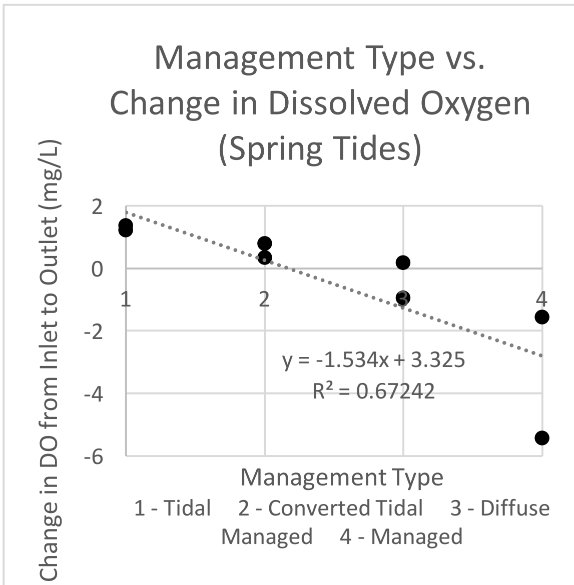



**Figure 15A:** Change in DO concentrations from inlet to outlet sampling point by pond management types.



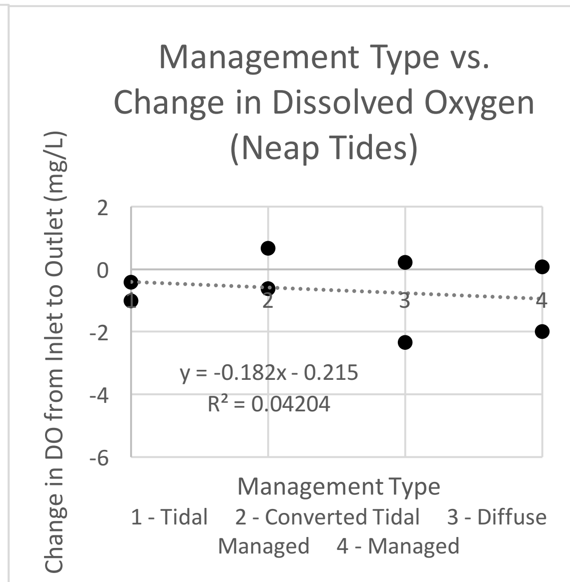




**Figure 15B** (Left): Change in DO from inlet to outlet sampling location by pond management type across spring tides.
**Figure 15C** (Right): Change in DO from inlet to outlet sampling point by pond management type across neap tides.

The tidal ponds, on average, exhibited an increase in DO from inlet to outlet samples; the managed ponds showed on average a decrease in DO from inlet to outlet, especially in pond A3W. However, this trend was not reflected in the final week of sampling, when the managed ponds showed a slight increase in DO concentrations and the tidal ponds showed a slight decrease. The tidal ponds showed more moderate changes in DO compared to the managed ponds. Changes in DO were greater in magnitude during spring tides than neap tides.

#### *4.5.5 Pond Morphology*

Pond morphology, most notably volume, may influence the residence time of the pond; residence time has been shown to influence estuarine water quality (Defne & Ganju, 2014). Pond surface area measurements and volume estimations are presented in Table 10. Pond volume had very weak positive correlation ($R^2 = 0.09$) with DO during all sampling weeks. When isolating spring and neap tides, agreement improved for neap tides ($R^2 = 0.47$) and was nonexistent for spring tides (Figure 16A-16B). A similar pattern was nearly identical when comparing pond surface area to DO: a weak overall positive correlation ($R^2 = 0.09$), no spring tide correlation and stronger neap tide agreement ($R^2 = 0.46$).

However, when compared to change in DO from inlet to outlet, pond volume had a weak negative correlation ($R^2 = 0.38$). Ponds with greater volumes tended to have greater decreases in DO, and greater variability in those decreases. When isolating for spring and neap tides, agreement was much stronger for spring tides and far weaker for neap tides (Figures 16C-16D).

**Table 10**: Measured surface area and calculated pond volume

| Pond | Surface Area ($km^2$) | Volume (acre-feet) |
|---|---|---|
| A3W | 2.23 | 15,164 |
| A8 | 1.6 | 13,029 |
| A17 | 0.5 | 5,884 |
| A21 | 0.54 | 3,116 |

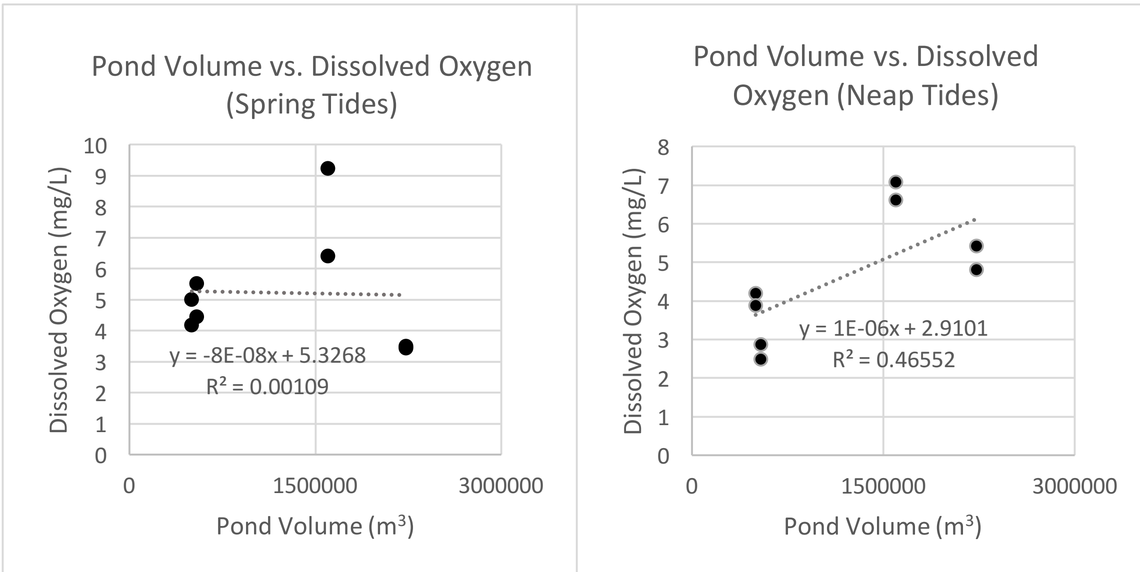



**Figure 16A** (Left): Dissolved Oxygen compared to pond volume for all spring tides.
**Figure 16B** (Right): Dissolved Oxygen compared to pond volume for all neap tides.

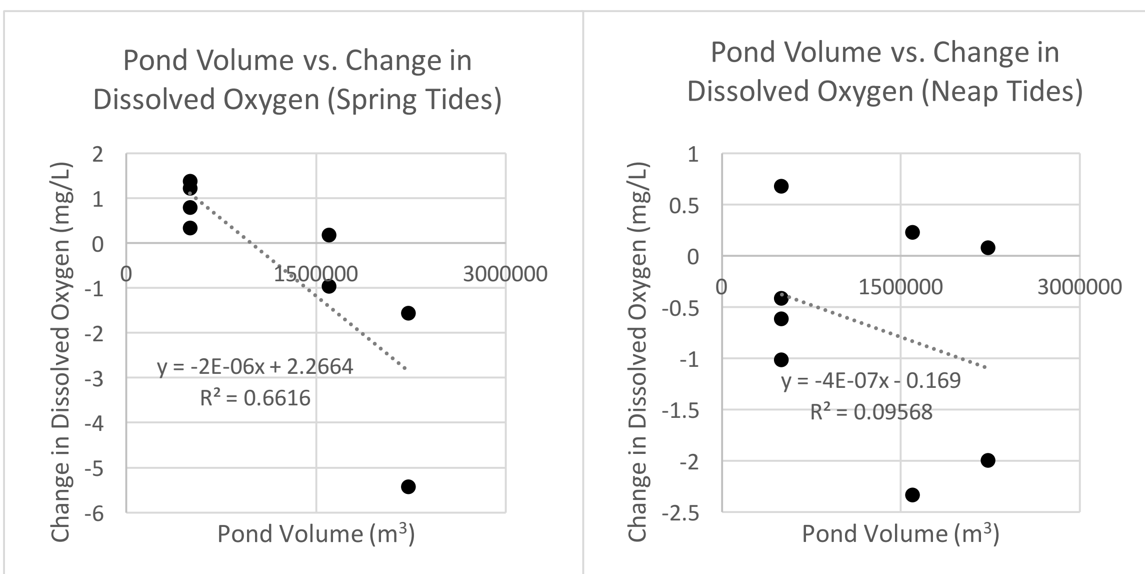



**Figure 16C** (Left): Change in DO (from inlet to outlet) compared to pond volume for all spring tides.
**Figure 16D** (Right): Change in DO (from inlet to outlet) compared to pond volume for all neap tides.

## 4.6 Inner-pond Comparison

To examine intra-pond spatial variability, additional points were examined within A3W: south inlet, deep, shallow, and algal mat (when present). Supplemental nutrient data was collected at A3W deep (in addition to A3W inlet and outlet). Sampling locations A3W inlet, outlet, and deep (see Table 2 and Figure 3A) were chosen to approximate the sampling locations in Topping et al. (2013). During this study, depth at the A3W deep point ranged from shallow ranged from 1.4 - 2 m; depth at the A3W shallow point ranged from 0.4 – 0.6 m.

### *4.6.1 Dissolved Oxygen and other parameters*

For the study period, DO was lowest at the A3W outlet and deep, and highest at the A3W inlet and south inlet (see Table 11). DO most often fell below the RMP threshold at the A3W outlet and deep points, with the exception of one inlet and one south inlet reading. DO at pond A3W's shallow point was greater during spring tides, and more moderate during neap tides. When pond A3W algal mat data were available, DO readings exceeded both the outlet value and the RMP threshold. DO values varied from week to week, with the greatest range occurring at A3W deep (3.54 - 8.76 mg/L).

Temperature was greatest at A3W shallow and A3W's outlet, which is located near a large region of shallow pond. With the exception of pond A3W's algal mat, temperature readings were consistent. pH was low across all pond sampling points, and fell below the RMP threshold (California Regional Water Quality Control Board

San Francisco Bay Region, 2007) at A3W's south inlet, A3W's inlet, and A3W deep during the majority of sampling weeks; however, pH never fell below the threshold at pond A3W's outlet. Specific conductance was similar to ambient South SF Bay levels during the study period ranging from 30 mS/cm to 46 mS/cm (US Geological Survey, 2001). Specific conductance was generally lower at the A3W inlet and south inlet, and greater at the A3W outlet, deep, and shallow sampling locations.

### *4.6.2 Nutrients*

Samples collected from the A3W inlet, outlet, and deep were analyzed for nitrate and ammonium (Table 12). Throughout both nutrient sampling weeks, the outlet was the only sampling point where nutrient loads were detected. In comparison to nitrate levels found in ponds A17, A21, and in Coyote Creek, pond A3W's outlet had less nitrate detected (ranging from 0.08 to 0.11 mg/L $NO_3$-N). Ammonium values were also low, ranging from 0.12-0.17 mg/L $NH_4$-N) but were higher than nitrate concentrations in the same sample, and were comparable to ammonium concentrations found in the tidal ponds.

**Table 11**: Average water quality data for pond A3W for spring, neap, and all sampling weeks.

| Sampling Period | Pond Outlet | DO (mg/L) | Temp. (°C) | pH | Conductivity (mS/cm) |
|---|---|---|---|---|---|
| *Average for all Spring Tides* | Inlet (N) | *6.98* | *23.75* | *8.62* | *39.44* |
| | Inlet (S) | *6.59* | *23.95* | *8.86* | *40.47* |
| | Outlet | *3.48* | *24.52* | *8.32* | *41.93* |
| | Deep | *6.16* | *23.86* | *8.63* | *41.70* |
| | Shallow | *7.01* | *25.08* | *8.41* | *42.23* |
| | *Algal Mat | *5.25* | *22.74* | *8.60* | *41.26* |
| *Average for all Neap Tides* | Inlet (N) | *6.07* | *23.28* | *8.50* | *40.61* |
| | Inlet (S) | *6.10* | *23.38* | *8.76* | *41.47* |
| | Outlet | *5.11* | *25.00* | *7.98* | *44.53* |
| | Deep | *4.13* | *23.25* | *8.50* | *43.42* |
| | Shallow | *5.58* | *25.63* | *8.19* | *44.13* |
| | *Algal Mat | *6.35* | *24.07* | *8.58* | *39.90* |
| ***Average for All Sampling Weeks*** | ***Inlet (N)*** | ***6.53*** | ***23.52*** | ***8.56*** | ***39.80*** |
| | ***Inlet (S)*** | ***6.35*** | ***23.67*** | ***8.81*** | ***40.97*** |
| | ***Outlet*** | ***4.30*** | ***24.76*** | ***8.15*** | ***43.23*** |
| | ***Deep*** | ***5.15*** | ***23.55*** | ***8.57*** | ***42.56*** |
| | ***Shallow*** | ***6.30*** | ***25.36*** | ***8.30*** | ***43.18*** |
| | ***Algal Mat*** | ***5.8*** | ***23.41*** | ***8.59*** | ***40.58*** |

*Algal mat data only available for one spring and one neap sampling week.

: Below RMP DO Threshold of 5.0 mg/L

: Exceeds Thermal Plan standards by being in excess of 2.8 °C higher than receiving waters [South SF Bay as measured at Dumbarton Bridge, (USGS, station ID 373015122071000)].

: Outside of RMP Acceptable pH range of 6.5-8.5.

**Table 12:** Nutrient samples for pond A3W for nutrient sampling weeks.

| Sampling Period | Pond Outlet | Nitrate (mg/L $NO_3$-N) | Ammonium (mg/L $NH_4$-N) |
|---|---|---|---|
| June 18th – 19th Spring Tide | Inlet (N) | 0.05 | 0.05 |
| | Outlet | 0.11 | 0.17 |
| | Deep | 0.05 | 0.05 |
| June 26th-27th Neap Tide | Inlet (N) | 0.05 | 0.05 |
| | Outlet | 0.08 | 0.12 |
| | Deep | 0.05 | 0.05 |
| ***Average for All Sampling Weeks*** | ***Inlet (N)*** | ***0.05*** | ***0.05*** |
| | ***Outlet*** | ***0.10*** | ***0.15*** |
| | ***Deep*** | ***0.05*** | ***0.05*** |

### 4.7 Slough Water Quality Data

The following water quality measurements were recorded for every sampling week in Coyote Creek: DO, temperature, pH, and Conductivity. During the first sampling week, Coyote Creek was sampled twice, west of A21 during a flood tide and east of A17 during an ebb tide. For the other three sampling weeks, Coyote Creek was sampled west of A21 and east of A17 during both a flood and an ebb tides. These data are presented below in Table 13.

**Table 13**: Water quality data for Coyote Creek

| Sampling Period | Pond Outlet | DO (mg/L) | Temp. (°C) | pH | Condct. (mS/cm) |
|---|---|---|---|---|---|
| June 18th – 19th | West of A21 (Flood) | 4.83 | 21.43 | 7.68 | 27.01 |
| | East of A17 (Ebb) | 5.34 | 22.31 | 7.71 | 28.03 |
| June 26th – 27th | West of A21 (Ebb) | 3.36 | 23.64 | 7.65 | 19.73 |
| | East of A17 (Ebb) | 1.97 | 23.45 | 7.56 | 20.1 |
| | West of A21 (Flood) | 3.21 | 24.04 | 7.63 | 20.91 |
| | East of A17 (Flood) | 3.21 | 24.8 | 7.63 | 19.14 |
| July 2nd – 3rd | West of A21 (Ebb) | 5.24 | 22.43 | 7.55 | 39.87 |
| | East of A17 (Ebb) | 4.01 | 24.20 | 7.58 | 33.34 |
| | West of A21 (Flood) | 3.29 | 22.77 | 7.55 | 32.38 |
| | East of A17 (Flood) | 3.16 | 22.66 | 7.55 | 32.52 |
| July 9th – 10th | West of A21 (Ebb) | 3.62 | 22.77 | 7.62 | 21.45 |
| | East of A17 (Ebb) | 3.43 | 23.42 | 7.70 | 17.77 |
| | West of A21 (Flood) | 4.06 | 23.35 | 7.63 | 23.11 |
| | East of A17 (Flood) | 4.25 | 23.72 | 7.64 | 21.19 |
| ***Average for All Sampling Weeks*** | ***West of A21 (Ebb)*** | **4.07** | **22.95** | **7.61** | **27.02** |
| | ***East of A17 (Ebb)*** | **3.69** | **23.34** | **7.64** | **24.81** |
| | ***West of A21 (Flood)*** | **3.85** | **22.90** | **7.62** | **25.85** |
| | ***East of A17 (Flood)*** | **3.54** | **23.72** | **7.61** | **24.28** |

: Below RMP DO Threshold of 5.0 mg/L

: Below AMP DO Threshold of 3.3 mg/L

## 5. Discussion

### 5.1. Water Quality

#### *5.1.1 Dissolved Oxygen*

In this study, DO often fell below both the Regional Management Plan threshold (5.0 mg/L) and the Adaptive Management Plan threshold (3.3 mg/L) (Tables 4, 11, and 13). The managed ponds tended to have higher DO (Figures 5A-5B) and the tidal ponds tended to have positive changes in DO from inlet to outlet (Table 4). The results of this study were similar to many of the major findings in previous water quality research in the ponds. Wastewater treatment plant (WWTP) discharges played a significant role in DO (Figure 12), as noted by Senn et al. (2015) and others (Cloern & Jassby, 2012; Grenier & Davis, 2010; Yigzaw, 2014).

In A3W, the most studied pond, the observed trends were similar to those noted by Topping et al. (2013). For example, DO values in pond A3W decreased from inlet to outlet, and were lowest in the deep parts of the pond and near the outlet, similar to findings of Topping et al. (2013). pH and conductivity were significantly different than other ponds in the system, although conductivity was similar to ocean levels across the same time frame. Some of the results of earlier studies, such as findings from Mraz et al. (2007) of frequent hypoxia and DO concentrations consistently below the 3.3 mg/L Adaptive Management Plan (AMP) threshold, were not reflected in this study. It could be that restoration efforts since the publication of Mraz at al. (2007), Shellenbarger et al. (2008), and Helton et al. (2011) have changed the characteristics of the SBSPRP enough to prevent some of the previous low-DO conditions from forming. It's also possible that the sampling methodology of this study, which did not examine diurnal or water column variability, missed periods of low-DO levels during the sampling period.

In A8, no sampling locations were found to have DO concentrations below the 3.3 mg/L AMP threshold. Previous studies of a 48-day period found 5 days with hypoxic conditions, mostly driven by wind and diurnal conditions (Amato & Valoppi, 2015). The lack of hypoxia noted in this study could be a result of changed conditions since 2014 or the limited sampling schedule which simply missed the hypoxic events, especially given that no samples were taken at night, when DO is depressed due to lack of photosynthesis (Cloern et al., 2000; Helton et al., 2011).

In the tidal ponds, DO concentrations in pond A17 ranged from 3.22 to 5.01 mg/L and differed dramatically from the extensive hypoxic events and fish kills noted in previous examinations of water discharging from A17 (Takekawa at al., 2015). This is mostly likely due to the 10-year gap between Takekawa et al. (2015) and to extensive changes in management within that decade. Removing the former pond control structure and allowing a more tidal regime likely had significant impacts on residence time and other factors such as sedimentation and water clarity. Conditions in pond A21 for this study, where DO ranged from 2.48 to 5.54 mg/L were similar to grab samples from 2012 where DO ranged from 2.6-10.5 mg/L (Hobbs, 2012), although DO concentrations were more moderate in this study. Extended hypoxic conditions such as those found by La Luz at al. (2015) were not reflected in the results of this study, but again, this may be a result of limited water column samples and no nighttime or daybreak sampling.

### *5.1.2. Nutrients*

Nutrient data in this study showed that proximity to a WWTP outfall was positively correlated with increased nitrate and ammonium (see Figure 13). The greatest nitrate concentrations were found in Coyote Creek, the slough that receives treated sewage effluent from the San José-Santa Clara Regional Wastewater Facility (SJSCRWF). The greatest ammonium concentrations were found in pond A17 (the closest pond to SJSCRWF), with elevated ammonium levels also found in pond A21, Coyote Creek, and pond A3W - the closest pond to the Sunnyvale Water Pollution Control Plant (SWPCP).

Nitrate data measured during ebb tides in Coyote Creek (nitrate = 3.08–8.32 mg/L $NO_3$-N) were similar to magnitudes reported during ebb tides by Senn et al. (2015) during high-resolution nutrient mapping of the slough in July 2015 which reported an ebb tide nitrate concentration of ~8.40 mg/L $NO_3$-N. Although no nutrient data had previously been collected in A17 or A21, the high nutrient concentrations found there varied in relation to the nutrient loading input from Coyote Creek, although in-pond nitrate levels were generally lower than in Coyote Creek.

Pond A3W had low levels of nitrate (0.05 – 0.11 mg/L $NO_3$-N) and ammonium (0.05 – 0.17 mg/L $NH_4$-N) in surface water, similar to Topping et al. (2013). The highest nutrient readings in any of the managed pond sampling points were at pond A3W's outlet. This is likely a result of nutrient loading from the SWPCP, which has reported greater TN and ammonia concentrations in its discharges compared to the SJSCRWF in recent years (San Francisco Baykeeper, 2013).

### 5.2 Role of Wastewater Treatment Plants

Wastewater treatment plants had a significant impact on water quality, as anticipated based on results of previous studies in the South Bay Salt Pond Restoration

Project (SBSPRP) and other urban estuaries. Distance to WWTP outfalls was negatively correlated with increased nutrient loading and positively correlated with DO levels (Figures 12-13). High nutrient loading from the WWTPs can lead to DO depletion due to either phytoplankton growth and decomposition (Yigzaw, 2014), marcoalgal shading (Nezlin et al., 2009), or some combination of the two.

While the influence of WWTP nutrient loading on DO concentrations was observed consistently across all ponds, the impact of WWTP loading on change in DO from pond inlet to pond outlet was less clear. Change in DO from inlet to outlet for each pond was used to analyze the influence of individual pond characteristics on water quality. The two closest ponds to a WWTP outfall, A17 and A21, showed more moderate changes in DO than at A8, the pond located furthest from a WWTP outfall. Ponds A17 and A21 also exhibited more instances of positive changes in DO from inlet to outlet. In contrast, A3W outlet (which is a similar distance to a WWTP outfall compared to A17 and A21) showed mostly decreases in DO from inlet to outlet (range = +0.08 to -5.44 mg/L); the greatest change in DO across all ponds was in A3W (-5.44 mg/L). Pond A3W also had lower levels of nitrate (0.05-0.11 mg/L $NO_3$-N) than A17 (3.28-6.93 mg/L $NO_3$-N) and A21 (2.95-6.66 mg/L $NO_3$-N) and had lower levels of ammonium (0.05-0.17 $NH_4$-N) compared with A17 (0.21-0.45 $NH_4$-N) and A21 (0.22-0.38 $NH_4$-N). This indicates that factors other than distance to WWTP and associated nutrient loading may influence DO and other water quality parameters.

Pond A3W's outlet is the closest sampling location to the SWPCP. During a study of all the WWTP plants in the SF Bay, SWPCP reported nutrient loadings of 27 mg/L TN in its effluent, higher than the SJSCRWF's reported discharges of 15 mg/L TN (San Francisco Baykeeper, 2013). However, the SWPCP also produced far less effluent during that time frame, about 10 million gallons per day (MGD) compared to SJSCRWF's 92 MGD (San Francisco Baykeeper, 2013). It's possible that although nutrient loadings from SWPCP can have a greater potential impact on pond water, this impact is more localized to the discharge site due to smaller discharge volumes with relatively higher TN concentrations.

### 5.3 Pond Morphology

The role of pond morphology, and how any effects mask or are compounded by other factors, are unclear. DO concentrations had differing correlations with pond volume and surface area depending on tidal dynamics. During spring tides, no correlation was observed between DO and both volume and surface area ($R^2$ = 0.00), and during neap tides a positive correlation was observed between DO and both volume ($R^2$ = 0.47) and surface area ($R^2$ = 0.46). This may be caused by lower DO inputs to the two smaller ponds, A17 and A21, suppressing the outlet DO levels. When examining change in DO, pond volume was negatively correlated, although less so during neap tides. This supports the theory that longer residence times (such as those found in larger volume ponds) may lead to a decrease in DO from inlet to outlet points. Residence time can significantly impact DO levels in estuarine systems (Haushild & Stoner, 1973; Nezlin et al., 2009).

Depth of the water column at the sampling point also had unclear impacts on DO concentration and other variables, although this may be due in part to only sampling water quality near the top of the water column. During spring tides, DO and depth were weakly negatively correlated ($R^2$ = 0.36), but no correlation was found during neap tides ($R^2$ = 0.00). It may be that during spring tides, water is mixed vertically, allowing lower DO water from near the pond bottom to impact the near-surface waters in ways that neap tides (with minimal water mixing) cannot. In a previous study of pond A3W, near-bottom DO conditions were consistently lower than in samples from other parts of the water column (Helton et al., 2011; Topping et al., 2013). Stratification in DO has been found in the sloughs surrounding the ponds (Senn et al., 2015) as well as in addition some of the ponds (Mruz, 2008).

In addition to issues with stratification, pond depth may also influence water temperature due to absorption of solar radiation or input from sensible heat (Peña et al., 2010). At shallow sampling points, the entire water column may be impacted by solar radiation and sensible heat, in contrast to deeper points where the near-bottom water may not be impacted. This could influence water temperatures near the surface if vertical water mixing were to occur. However, no correlation ($R^2$ = 0.00) was noted between water depth at sampling point and temperature during this study, even when isolating spring and neap sampling weeks.

Current patterns of stratification and vertical water mixing in the ponds may change as a result of restoration efforts or ongoing environmental changes. Along the west coast of the US, upwelling is generally believed to cause low-DO conditions as near-bottom waters are mixed and brought to the surface (Cloern & Jassby, 2012). These low-DO waters can enter estuaries through tidal mixing. However, climate change is projected to alter upwelling patterns and create stronger seasonal differences in the thermal gradient across the SF Bay (San Francisco Bay Area Wetlands Ecosystem Goals Project, 2015). The impact of these changes on the SBSPRP and the South SF Bay remains to be seen.

The impact of fluid interaction where winds meet the water surface is less clear, based on the results of this study. In many estuaries, winds create vertical water mixing, decreasing stratification and exchanging nutrients and DO between water layers (Defne & Ganju, 2014; Peña et al., 2010). Variation in wind patterns for each pond may also be a factor; the wind at larger ponds like A3W was much stronger than at other ponds, particularly in the early afternoon. Unfortunately, it is not possible to determine the impact of wind was at each sampling location or pond based on regional wind velocity measurements.

### 5.4 Role of Management

Pond management had a noticeable impact on both DO concentration and change in DO from inlet to outlet within ponds, likely due to the influence of hydrologic flow regime on residence time. When managers control water flux, less water can enter or exit the ponds at any given time, leading to (presumably) longer residence time. The longer water is retained in an estuarine water body with minimal water exchange, the more susceptible it is to hypoxic conditions and stratification (Defne & Ganju, 2014).

Ponds A17 and A21 had the least management control of water exchange, and were thus ranked lower on the qualitative management scale. These ponds had lower DO overall, which could be attributed either to lack of management control or to the shallower depth and lower volume compared with ponds A3W and A8. Ponds A17 and A21 had more moderate changes in DO compared to A3W and A8, and were more likely to see modest increases in DO from inlet to outlet. However, the impact of management type is difficult to isolate from other factors that might impact residence time, such as pond volume.

Additionally, the managed ponds A8 and A3W had more instances of algal mat development. Pond A8 had small floating pods of macroalgae during the sampling period, and on several weeks pond A3W developed large mats of macroalgae, some estimated to be 20 meters or more in length. Macroalgae has been linked to low DO due to shading of the underlying water by algal mats preventing photosynthesis (Nezlin et al., 2009). It may be that the relatively low velocities of water in the managed ponds allows algal mats to form.

Although ponds with less management control exhibited lower temperature and pH readings and were more likely to have increases in DO from inlet to outlet, management of ponds may still be necessary to control other degradations of water and pond habitat quality. For instance, the management of ponds may help to reduce the amount of mercury reintroduced when older sediment is moved by tidal action (Helton et al., 2011). Managers may also need to retain tighter control over water level fluctuations in ponds that are susceptible to very shallow conditions during low tide, as shallow ponds are more likely to have lower DO and warmer temperatures (South Bay Salt Pond Restoration Project, 2016b). Although shifting from a managed to a more tidal regime may improve select water quality parameters such as DO, the timing and location of management changes may need to be selected carefully.

**5.5 Land Use/Land Cover**

Surrounding land cover, defined as the percentage of wetland habitat within 1km of each pond, had differing impacts on pond water quality depending on the tidal action during the sampling period. During spring tides, no correlation was observed between DO and wetland coverage; during neap tides, wetland coverage was negatively correlated with DO ($R^2 = 0.45$). Wetlands often have low DO, especially in coastal systems (Day et al., 2005). It's possible that when neap tides occur and water remains in or near ponds, longer, decay of wetland vegetation has a greater opportunity to depress DO levels compared to waters that move further into the open bay more quickly.

Change in DO from inlet to outlet across all ponds was positively correlated with surrounding wetland cover ($R^2 = 0.35$), with a much stronger agreement during spring tides ($R^2 = 0.57$) than neap tides ($R^2 = 0.13$). This could indicate that reduced residence time during spring tide doesn't allow sufficient time for wetland vegetation to impact water quality, or it could be that more complex dynamics are in play that the current study was unable to capture.

Developed land coverage was also positively correlated with DO levels during neap tides, although not during spring tides. This may indicate that runoff from urban areas in the South Bay is naturally higher in DO or is of sufficient quality to avoid low DO conditions (such as reduced nutrient loading in comparison to WWTP effluent). Alternatively, positive correlation between developed land coverage and DO may be a symptom of decreased wetland coverage. Ponds with less wetland coverage by definition have a greater portion of non-wetland coverage surrounding them. This non-wetland coverage may include greater portions of developed land coverage, creating a positive correlation with DO.

The ponds with the greatest proportion of wetlands surrounding them also had the greatest nitrate concentrations in their inlet and outlet samples. Previous studies have shown that wetlands can reduce nitrogen loading of a system through denitrifying bacteria (Verhoeven et al., 2006). Constructed wetlands are being employed in regions around the globe (including at some wastewater treatment plants) specifically to reduce the concentrations of select pollutants (Vymazal, 2010). However, wetlands with persistently high nutrient loading are also vulnerable to changes in species composition, and wetlands inundated with high-nitrate waters can emit high levels of nitrous oxide ($N_2O$), a greenhouse gas (GHG) (Verhoeven et al., 2006). Ponds such as A17 and A21 that exhibit high nutrient loading and have significant surrounding wetland vegetation could undergo ecological changes and increased GHG emissions as restoration efforts continue. Monitoring of the vegetation structure and micrometeorological observations of carbon cycling may further elucidate these issues.

### 5.6 Tidal Action

The impact of tidal action was clearly significant throughout the sampling period, as demonstrated by the positive correlation between changes in DO in tidal ponds (A17 and A21) and difference in elevation between the first tides of each sampling day ($R^2 = 0.63$). Greater difference in tide elevation indicates that more water is being flushed in and out of each pond, and that water is moving with a greater velocity. Changes in water velocity were apparent when high water velocity precluded Secchi disk measurements at several pond sampling locations during a spring tide; Secchi disk measurements were possible at the same approximate pond locations during the next neap tide. The increase in water exchange reduced residence time (and possibly exposure time) in the system, and led to more moderate changes in DO from inlet to outlet. Other estuarine studies have found that residence time can significantly impact DO levels, and leave narrow estuaries more vulnerable to hypoxic events (Haushild & Stoner, 1973; Nezlin et al., 2009).

Tidal action and its impact on residence time also lead to distinctions in how different factors influence the ponds. This was most apparent in the relationship between temperature and pond DO. If all other factors remain equal, increasing the temperature of a water body will decrease its ability to retain oxygen and decrease DO concentrations (Debelius et al., 2009). However, neap tides actually exhibited a weak positive correlation with DO ($R^2$ = 0.16), meaning that DO increased with increasing temperature due to other drivers in the system.

Isolating spring and neap samples for other potential drivers of variations in water quality often produced opposing patterns as well (Table 4). This suggests that differences in hydrologic conditions create opportunities for different drivers to impact pond systems. Most notably, spring tides are assumed to have shorter residence times and increased water velocity, leading to less opportunity for nutrient loading or wetland vegetation to impact water quality within the ponds. If the moderate tidal differences of neap tides are assumed to increase residence time and minimize water mixing, stratification of DO in the water column may occur during those conditions.

The lowest-low tides may also affect DO levels. Amato and Valoppi (2015) found that in pond A21, the periods of lowest DO coincided with low tides, when the depth decreased in A21. Given that the lowest of low tides occur in spring tides, spring tides could have the potential to trigger periods of hypoxia even though they may produce an increased flushing effect and shorter residence time. Depth had no correlation with neap tides ($R^2 = 0.00$) and weak negative correlation during spring tides ($R^2 = 0.36$), meaning that shallow tides had greater concentrations of DO. No evidence of hypoxic events or adverse impacts of DO levels on ecosystems such as fish kills were noted during the course of this study.

## 5.7 Implications for Adaptive Management

This study has implications for existing and future AMP actions in the SBSPRP, although some results have greater relevance than others. Since the SBSPRP resource managers may have little control over the quality of the water as it enters the ponds, this section will focus on how pond controls will impact water within ponds and at outlets, and how that may change in the future.

Many factors that impact the ponds (most notably WWTP discharges) do not fall under the regulatory purview of the SBSPRP managers. Additionally, a number of factors -- the logistical and financial limitations of adjusting water management systems in the South Bay, WWTP discharges, ongoing urbanization near the restoration site, and urban and storm water runoff issues -- mean that the infrastructure surrounding the SBSPRP is inherently inflexible. WWTPs in the South SF Bay have also undergone significant renovation since they become operational, and are now among the most advanced WWTP systems in the US (Yigzaw, 2014). However, if equipped with greater knowledge of how factors such as nutrient loading from WWTPs impact the SBSPRP, managers may be able to lobby for changes to specific management practices. This could include investments in improved WWTP infrastructure to reduce the chance of sewage spills (Rogers, 2016) or expanded pollution prevention programs in South Bay municipalities.

### *5.7.1 Trajectory of Water Quality Changes Based on Existing AMP Goals*

Based on the results of this study and the stated objectives of the AMP, some changes to water quality can be anticipated. As sedimentation increases, ponds will become shallower. Shallow water may increase in temperature, which could push temperature out of compliance with Regional Management Plan (RMP) thresholds (California Regional Water Quality Control Board San Francisco Bay Region, 2007) and potentially impact capacity to hold DO (Debelius et al., 2009). Increasing wetland vegetation (a goal for the AMP to improve habitat and flood protection) will also likely decrease DO concentrations, as DO in wetlands is naturally low (Verhoeven et al., 2006). This was observed during the study, where ponds with surrounding wetland cover had lower DO than ponds in more developed areas. However, tidal ponds with more wetland vegetation than managed ponds exhibited increases in DO concentrations from inlet to outlet compared to managed ponds. This phenomenon may be due to decreased residence time or increased turbidity of water in tidal ponds (Defne & Ganju, 2014).

### *5.7.2 Considerations for Future AMP Decisions*

The results of this study may be of value when making future AMP decisions, especially given the flexibility of the decision making process and the limited financial resources available to the project.

In this study, ponds that were less managed tended to have less variation in DO and were more likely to have an increase in DO concentration from inlet to outlet. As existing pond control structures age over time, managers will need to divert critical funds towards replacing pond control structures or move towards tidal control of ponds (South Bay Salt Pond Restoration Project, 2016a). If nutrient loading from a nearby WWTP outfall is not an issue, then allowing for a more natural flow regime may actually increase DO concentrations in pond water, or at least lead to less DO variability.

For ponds that retain pond control structures, managers can potentially anticipate incoming spring/neap impacts in conjunction with effective monitoring strategies. Once water quality patterns for different tidal elevations are established, managers may be able to adjust the water exchange in advance. For example, managed pond A3W was found to be more susceptible to decreases in DO during spring tides. Opening up the maximum number of breaches available during a spring tide to allow a more natural tidal flow could potentially minimize these decreases in DO.

In fully tidal systems, there is very little opportunity to anticipate spring/neap changes. Systems with strong differences between spring and neap tides or where spring/neap changes are unpredictable may be poorly suited for change in tidal control, or may need to be stabilized with additional restoration work (such as increased sediment deposition or more pond control structures) before being switched to tidal control. Pond A17 was one such pond that was converted from managed to a mostly-tidal system (Wolfe, 2012), and this study found that A17 had the most consistent inlet to outlet changes in DO concentration, ranging from -0.61 to +0.79 mg/L.

The results of this study may be of value when revising the end management goal for each individual pond. The eventual goal of the SBSPRP is to establish a 9:1 ratio of tidal to managed ponds (EDAW et al., 2007). The process of designating managed ponds is currently focused on providing optimal bird habitat (South Bay Salt Pond Restoration Project, 2016a) and/or reducing methylmercury levels (Helton et al., 2011). In the future, additional considerations could include proximity to WWTP outfalls. For example, if one location (such as A3W) is likely to be strongly impacted by a WWTP, it could be selected as one of the remaining managed ponds; tidal ponds could be selected based on increased distance from nutrient loading areas or shorter residence times.

## 5.8 Limitations of Study

Results and trends analyzed in the study face a variety of limitations, including sampling equipment inaccuracies or calibration challenges and human error in data sampling and analysis. Three additional limitations are discussed below including impacts from limited sampling locations and ponds, uncertainties in GIS analysis, and data analysis.

### *5.8.1 Sampling Locations and Case Study Ponds*

Selecting discrete samples for water quality analysis meant some patterns were lost in the gaps between sampling locations. Most notably absent are diurnal patterns;

in all previous pond studies, lower DO concentrations were found at night or at daybreak when algae were unable to photosynthesize and produce DO (Amato & Valoppi, 2015; Helton et al., 2011; Topping et al., 2013). This pattern was likely at play in all case study ponds during the study period, however, the magnitude of decreases in DO concentrations and how they varied from pond to pond or during different tidal conditions is unclear.

Sampling only near the surface of the ponds and sloughs also produces an incomplete representation of the water column. Estuaries, especially west coast estuaries, can have a high degree of stratification, which can increase by thermohaline conditions (Nezlin et al., 2009; Smith & Hollibaugh, 2006). Most of the sampling locations in this study were shallow (<2m depth), but near-bottom DO concentrations may differ dramatically from near-surface DO due to benthic activity or low light penetration (Nezlin et al., 2009; Peña et al., 2010). In addition, nutrient sampling was only conducted during the first two weeks of sampling.

The small number of case study ponds also potentially impacted the results of this study, especially when examining spatial variability across ponds and more subjective parameters such as management type. Limiting the project to four case study ponds made the project more feasible, but may be too limiting to see larger patterns at play for management type or pond morphology. Expanding the study to multiple ponds for each management type and additional ponds along Guadalupe and Alviso slough may produce more clear or nuanced patterns in water quality data.

### *5.8.2 Uncertainties in GIS Analysis*

Additional uncertainty remains in the extraction of data using GIS analysis, most notably in relating volume estimations to diurnal and seasonal variabilities in pond volume. As the tidal changes push different volumes of water into the bay and sloughs, pond volumes also change. It's unclear how calculated pond volumes compare to actual pond volumes at very low or very high tides. Restoration efforts since bathymetry data were collected (US Geological Survey, 2004) may also have created error in volume estimations. The bathymetry data were also collected at a coarse resolution (30m) that masks minute changes in pond bottoms which naturally have a high degree of variation (Figure 1).

There are few previous estimations of pond volume to validate the Surface Volume tool calculations (ESRI, 2013), and significant margins of uncertainty remain. The USGS published estimations of volume for a selection of ponds in the Alviso complex (Lionberger, Schoellhamer, Shellenbarger, Orlando, & Ganju, 2007). Although these pond volumes did not correspond to any of the ponds studied in this study, volumes for several reference ponds (A6 and A10) were determined from the same bathymetry data set and compared to volumes from Lionberger et al. (2007). Pond volumes were found to be within 6-19% agreement; given the variability of tidal elevation and differences in pond morphology since the 2007 study, the pond volumes calculated from the Surface Volume tool were deemed acceptable for this study.

Other uncertainties relate to land use calculations. The distances to WWTPs were calculated by connecting WWTP outfalls to pond breach points along a slough or portion of the bay, trying to follow the thalweg (deepest portion of the channel.) There is no guarantee that WWTP effluent directly followed these paths, nor that the paths were consistently drawn within the thalweg of the sloughs. Surrounding land cover was arbitrarily calculated within a 1km buffer of each pond to provide consistent land cover data; determining the catchment area for each individual pond would be challenging given the hydrologic complexities of each system and the relative flatness of the surrounding land. Additionally, the land cover data were captured in 2011, five years before this study was undertaken (Homer et al., 2015) so any restoration work completed between 2011 and 2016 is not reflected in land cover estimations.

***5.8.3 Data analysis***

Partially as a symptom of the limited number of case study ponds, there is a high potential for some variables to be masking or over-emphasizing the impact of other variables. For example: the ponds with the lowest volumes (A17 and A21) also had the lowest degree of manager control over water flow (2 and 1 on the qualitative management type scale respectively). This makes is difficult to separate the true impact of either volume or management type on overall water quality. Other examples of potentially obfuscating variables include pond volume and pond surface area; nitrate and ammonium levels; nutrient loading and distance to WWTP; and land cover and distance to WWTP. A greater variety of sample ponds including tidal ponds without such intense nutrient loading from a WWTP and managed ponds of a smaller volume would assist in insolating some of these factors.

**5.9 Future Research**

To reduce the impact of the limitations discussed above, this research could act as a pilot study for a longer, more expansive examination of water quality in salt pond restoration. This future study could expand the temporal coverage of sampling through the use of continuous water quality monitoring for parameters such as DO, temperature, pH, and specific conductance in conjunction with increased nutrient sampling. Increased sampling across diurnal and tidal cycle would reduce the likelihood of missing important water quality changes such as a hypoxic episode. This would also allow for comparison of diurnal and tidal trends across multiple ponds, which could indicate which types of ponds are more likely to be influenced by different drivers.

Sampling should also be expanded to analyze DO throughout the water column. This could be done at one sampling location per pond to provide more insight into how management regime impacts mixing and stratification. Water column sampling should be collected in conjunction with wind data to better understand the role that wind plays in water mixing at that sampling point (Glibert et al., 2010). Examining additional parameters such as mercury and other legacy pollutants may also provide a more holistic view of water quality, although those parameters are currently studied through other research efforts.

To contextualize the data collection above, future studies would be improved by greater understanding of hydrology in each specific pond. This should include a more refined calculation of pond volume and estimations of residence time at different tidal conditions. Calculating exposure time (which includes residence time plus the length of time that a parcel of water may re-enter a system due to tidal hydrology) of each pond may also elucidate the impact of tidal action on certain parameters such as nutrients (Wolanski & Elliot, 2016). Gaining a greater understanding of how water moves through each pond can assist in determining which factors have the greatest influence on the system, and when each pond is most vulnerable to impacts from those drivers.

Future studies would also be improved by increasing the number of case study ponds, including multiple ponds of similar management types, as well as ponds with different flow regimes, inlet water sources, and pond morphologies. This could help remove some of the obfuscating variables discussed above, and may reveal more nuances in patterns that are unavailable in the current data pool. In addition to more pond samples, including some analysis of less-disturbed marsh habitat in the same area (such as the New Chicago Marsh, located south of pond A17 and east of A8) may provide better insights into how native marsh responds to the same environmental changes, and how a pond restored to a more natural ecology and tidal flow regime would function.

A broader question remains: even if resource managers better understand the range and drivers of water quality in the SBSPRP, how does that information fit in with other factors that need to be considered in adaptive management? Managers are placed in the precarious position of balancing their stated objectives (EDAW et al., 2007) with the environmental complexities of the South SF Bay and all the human stakeholders and ecosystems that depend on it (Grenier & Davis, 2010; Yigzaw, 2014). Actions that benefit one objective or group, such as converting a given pond to tidal marsh for field mouse habitat and reducing residence time for more moderate DO, may come at the direct detriment of another, removing migratory bird habitat and releasing additional legacy pollutants into the water column. A greater understanding of water quality issues does not guarantee that water quality challenges can be avoided, but it can reduce the uncertainty involved in some management decisions during the upcoming phases of restoration. It may also allow for better prediction of future impacts of different management scenarios, to allow better targeting of mitigation efforts in advance of restoration. Refinement in mitigation efforts can reduce waste of funding, labor, and time, three resources that may be limited given the magnitude and urgency of the SBSPRP objectives.

## 6. Conclusion

In this four-pond case study, proximity to wastewater treatment plant (WWTP) discharge points, tidal conditions, and pond management regimes were found to most directly influence water quality. The influence of other potential drivers (such as surrounding land cover) differed between spring and neap tide conditions. Ponds with water exchange that was primarily driven by tides exhibited more moderate changes in DO from inlet to outlet compared to larger volume ponds where South Bay Salt Pond Restoration Project (SBSPRP) managers retained greater control over water exchange. This may indicate that restored tidal ponds behave more like a shallow, west-coast estuary (Nezlin et al., 2009) than a pond; this information could be of particular value when predicting future changes to water quality as restoration continues.

Samples collected in this study were limited temporally, in the water column, and across sampling ponds. However, expanding this study to include more ponds, collecting water quality data across different tidal and diurnal cycles, and gathering information about other potential drivers of water quality such as wind would resolve those issues. Expanding the study in this manner may allow for confirmation or greater nuance to the patterns of water quality changes observed in this study. This may provide a stronger linkage between water quality knowledge of the South SF Bay and the adaptive management practices in play.

Gaining greater understanding of the water quality issues involved in the SBSPRP has the potential to inform not just the current AMP, but potential estuarine restoration projects around the globe. As sea level rise is predicted to increase and climate change continues to threaten ecosystem health, the need to restore the earth's coastal wetlands will continue to increase (Ramsar Convention Secretariat, 2010). It's likely that a portion of these restoration sites will be areas with significant degradation, nutrient loading, and anthropogenic influence, as was the case in the SBSPRP (EDAW et al., 2007). Understanding how natural drivers and human management decisions converge to impact water quality in estuarine restoration could be valuable in other salt pond restoration projects, such as the South San Diego Bay Coastal Wetlands Restoration and Enhancement Project (Yu, 2008), and other forms of wetland restoration. Given the role of water quality in determining the ecosystem services of wetlands (Ramsar Convention Secretariat, 2010), reducing the uncertainty surrounding water quality degradation in coastal wetland restoration projects like the SBSPRP and learning how to best apply that knowledge towards effective resource management will be crucial.

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