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Ecological investigations of a fish kill in Pescadero Lagoon, California

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ECOLOGICAL INVESTIGATIONS OF A FISH KILL IN
PESCADERO LAGOON, CALIFORNIA

A Thesis

Presented to

The Faculty of the Department of Environmental Studies

San Jose State University

In Partial Fulfillment

of the Requirements for the Degree

Master of Science

by

Rebecca M. Sloan

December 2006

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ABSTRACT

ECOLOGICAL INVESTIGATIONS OF A FISH KILL IN PESCADERO LAGOON, CALIFORNIA

By Rebecca M. Sloan

This study investigated the ecological conditions and possible causes of a fish kill in Pescadero Lagoon, Northern California. The kill occurs upon the breaching of a sandbar and is affecting federally threatened steelhead (*Oncorhynchus mykiss*). Between September 2004 and January 2006, eight sites were monitored for ambient water quality, nutrient concentrations, and sediment and vegetation characteristics. Surface dissolved oxygen concentrations during lagoon draining were low enough to cause acute mortality in salmonids. Toxic hydrogen sulfide concentrations existed in the bottom waters of the lagoon and the lower arm of Butano Creek prior to sandbar breach. Salinity explained 39% of the observed variability in water column dissolved oxygen during sandbar closure. Macrophytic coverage, microphytic concentrations, and sediment organic carbon content suggest Pescadero Marsh suffers medium to high eutrophication. Although the eutrophic condition in Pescadero should be considered in marsh management, alleviating stratification will provide the best opportunity for stopping fish kills.

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This work is dedicated to the community of Pescadero and those who work tirelessly to preserve the natural and cultural heritage of the Central California Coast.

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INTRODUCTION

Estuaries form where the land meets the sea. Carved out of land by the physical forces of water, geomorphic landforms such as bays, sloughs, marshes, inlets, sounds, and lagoons are often associated with brackish water environments. Estuaries have become an integral part of the global economy. In 1995, 39% of the world's population lived within 100 km of the coast (Burke et al. 2001). The bulk of the world's marine fishery harvest, up to 95%, is caught or reared in coastal waters (Sherman et al. 1993). In the United States, three quarters of the economy is generated in coastal states and half of that revenue originates in coastal counties (Colgan 2004). Tourism is the number two contributor to the US Gross Domestic Product (GDP) and 85% of these dollars are generated in coastal states (Houston 1995). Over 180 million US citizens visit the US coast every year, spending \$54 billion in goods and services and providing over 28.3 million jobs (Houston 1995).

Based on 2005 gross state product (GSP) estimates, California is the tenth largest economy in the world (CIA 2006). In the year 2000, 86% of California's GSP, approximately \$1.15 trillion, was generated in coastal counties (Kildow and Colgan 2005), and 77% of California's population lived in coastal counties (NOEP 2000). The largest overall GSP growth occurred in the central coast region of California (Monterey, Santa Cruz and San Mateo Counties) and was driven primarily by tourism and recreation (Kildow and Colgan 2005).

In the United States, coastal estuaries are valued for the economic, health, and recreational roles they play in our society. Commercial fisheries experts state that

estuaries play an integral role in the life cycle of over 75% of the commercially important marine species in the US (NOAA 1998). Estuaries protect human development from river flooding and high elevation storm surges by “storing” large volumes of water thereby protecting human life and property. By slowing down water flow, estuaries allow pollutants to settle into sediments or to be taken up by microbes, thereby removing them from the water column.

The greatest threats to the ecological health of estuaries around the world, in order of impact significance and trend direction, are pollution, habitat loss, overexploitation, invasive species introduction, and climate change (Millennium Ecosystem Assessment 2005). In the United States, nutrients, specifically nitrogen and phosphorus, are thought to pose the largest pollution threat to coastal waters (NRC 2000; Howarth et al. 2000), with an estimated two-thirds of coastal rivers and bays moderately to severely degraded from nutrient pollution (Bricker et al. 1999).

One of the most dramatic and obvious results of coastal degradation are fish kills. Fish kills are both natural and anthropogenic in nature, however, the line between the two can sometimes be difficult to determine. Chemical or sewage spills are easily identified as human-induced as they are often large, from a known origin, and detectable by laboratory analysis. The cause of fish kills related to anoxia (0 mg/L dissolved oxygen), harmful algal blooms, or pathological distress can be more difficult to determine, as they are often the result of a combination of natural and anthropogenic processes.

Anoxia has been widely studied, especially on the Eastern Coast of the United States, and is beginning to be recognized as a great threat to estuarine waters all over the

world (Nixon 1985, 1995; NRC 2000; Cloern 2001). Hydrogen sulfide and ammonia are both toxic chemicals produced by natural processes in anoxic environments.

Fish kills have been associated with sediment re-suspension into the water column. Sediments can be disturbed during dredging or a storm event. Toxic chemicals stored in the sediment can be re-released into the water column and thus re-exposed to fish and invertebrates. Sediments can also clog or damage gill tissue and cause fish and invertebrate mortality.

Anoxia

Fish kills, affecting thousands to millions of fish, have occurred on each of the world's continents, except Antarctica (Breitburg 2002). These kills are often associated with hypoxic (defined here as in Diaz and Rosenberg 1995 to be < 2 mg/L dissolved oxygen) and anoxic (defined here as in Diaz and Rosenberg 1995 to be 0 mg/L dissolved oxygen) events related to anthropogenic nutrient enrichment (Diaz and Rosenberg 1995; Breitburg 2002).

Systems most susceptible to low dissolved oxygen (DO) events are typically characterized by strong density stratification during the season in which low oxygen is most severe, long residence times, low tidal energy, and a high watershed area to estuary volume ratio (Nixon 1988; Monbet 1992; Caddy 1993; Cloern 2000). With a dense, saline hypolimnion (bottom layer) and a freshwater epilimnion (surface layer), restricted backwaters of estuarine systems are not well mixed (Beck and Bruland 2000; Luther III et al. 2004). This creates a barrier to nighttime oxygen replenishment of bottom waters and leads to a hypoxic/anoxic hypolimnion.

Fish die-off events are most often associated with a rapid change in the DO concentration (Breitburg 2002). D'Avanzo and Kremer (1994) found that bottom-water DO concentrations crashed as a result of low irradiance and killed fish when mixed into the water column by a storm event. Breitburg (1992) reported a fish kill in Chesapeake Bay when hypoxic bottom water was forced near-shore by winds and lateral tides. Fallsen et al. (2000) chronicled a fish kill caused by anoxia and high sulfidic concentrations when a large dinoflagellate bloom crashed in Mariager Fjord, Denmark.

Anoxic events associated with sandbar breaches have been documented in both Southeastern Australia (Becker personal communication) and Southeastern Africa (Whitfield 1995). In July 2005, a fish kill involving tens-of-thousands of fish occurred in the Surry River estuary in Southwest Victoria, Australia. Estuary monitoring revealed depleted oxygen levels throughout the entire water column in back marsh areas and throughout half of the water column in the fore marsh areas. It is believed that upon sandbar breaching, the top, oxic layer of the fore-marsh drained off, leaving behind only the hypoxic/anoxic water from both the fore marsh and the back marsh areas (Becker personal communication).

In a review of DO requirements for salmonids, the US EPA (1986) synthesized 96-hour LC50 results and found mortality or loss of equilibrium to occur between 1 mg/L and 3 mg/L. LC50 or lethal concentration tests are based upon 50% of the organisms experiencing a similar effect, e.g. mortality, endocrine disruption, or behavioral changes.

In the US EPA (1986) synthesis of findings from Sheppard (1955), mortality of brook trout occurred in one hour at 10 °C with DO concentrations of 1.2 mg/L and in ten

hours at concentrations of 1.5 mg/L. A 3.5 hour exposure to 1.1 and 1.6 mg/L found mortality of trout at 10 °C and 20 °C respectively (Downing and Merkens 1957 from US EPA 1986). The EPA (1986) concluded that dissolved oxygen levels must remain > 3 mg/L to avoid acute mortality of salmonid and non-salmonid species in freshwater. Results from one to four day LC50 tests for blue, green, and flat mud crab species were < 1.0, 0.54 and 0.57 mg/L DO respectively (Carpenter and Cargo 1957; Poucher and Coilo 1997; Stickle 1988 from US EPA 2000).

Besides highly visible fish and invertebrate kill events due to anoxia, short- and long- term hypoxia have been shown to have less dramatic, but nonetheless very serious, ecosystem effects (Diaz and Rosenberg 1995; Breitburg 1997). Breitburg et al. (1997) investigated the effects of low oxygen on predator-prey relationships among estuarine species through laboratory experiments and field sampling. The study found that low oxygen concentrations had the potential to greatly alter the trophic structure of the ecosystem and had varied positive and negative effects on predator-prey relationships. In another study, Breitburg (1992) found that direct exposure to hypoxic water in Chesapeake Bay caused substantial egg mortality in benthic reef fishes such as blennies, clingfish, and gobies. Diaz and Rosenberg (1995) found prolonged hypoxia and anoxia to cause significant structural changes in benthic communities and to affect benthic-pelagic coupling.

Ammonia and hydrogen sulfide toxicity

Ammonia (NH₃) and hydrogen sulfide (H₂S) are produced in anaerobic sediments by heterotrophic bacteria. Under suboxic conditions, ammonia is formed when nitrate

and iron are used as terminal electron acceptors in the oxidation of organic matter (Figure 1). Sulfate (SO_4^{2-}) reduction occurs strictly under anoxic conditions (Beck and Bruland 2000) and results in the formation of both ammonia and hydrogen sulfide (Figure 1).

Figure 1. Predicted thermodynamic order of major reduction reactions under oxic, suboxic and anoxic conditions in marine sediments (Schulz 2000 from Froelich 1979; Rue et al. 1997).

Ammonia in aqueous solution exists in two forms, un-ionized ammonia (NH_3) and ionized ammonium (NH_4^+). Un-ionized ammonia is the more toxic of the two fractions as it is a neutral molecule and therefore can diffuse across epithelial membranes of organisms much more readily (US EPA 1999). Hydrogen sulfide, like ammonia, exists in an ionized and unionized form. HS^- is the ionized fraction and H_2S is the

unionized fraction. Again, like ammonia, the unionized form of hydrogen sulfide, H₂S, is the more toxic of the two (US EPA 1976).

The fraction of un-ionized ammonia (NH₃) and the ammonium ion (NH₄⁺) has been shown to vary with pH (Chipman 1934 from US EPA 1999; Wuhrmann and Woker 1948 from US EPA 1999) and temperature (Erickson 1985). The US EPA (1999) has created two sets of criteria for ammonia, acute criteria (CMC) and chronic criteria (CCC). Acute ammonia nitrogen (mg/L) levels are calculated using the equations in Figure 2. US EPA (1999) water quality criteria for salmonids and early-life-stage fish require one-hour average concentrations of ammonia nitrogen (mg/L) do not exceed the acute criterion calculated using the below equations more than once every three years on average.

$$\text{CMC} = \frac{0.275}{1 + 10^{7.204 - \text{pH}}} + \frac{39.0}{1 + 10^{\text{pH} - 7.204}}$$

$$\text{CCC} = \left(\frac{0.0577}{1 + 10^{7.688 - \text{pH}}} + \frac{2.487}{1 + 10^{\text{pH} - 7.688}} \right) \cdot \text{MIN}(2.85, 1.45 \cdot 10^{0.028 \cdot (25 - T)})$$

Figure 2. Acute criteria (CMC) and chronic criteria (CCC) for ammonia (US EPA 1999).

The proportion of ionized and unionized hydrogen sulfide fractions is dependent upon pH (US EPA 1976b). For example, at pH 9.0, 99% of hydrogen sulfide exists in the nontoxic, ionized form (HS⁻), at pH 7 approximately 50% is ionized and 50% is unionized and at pH 5 approximately 99% exists as the toxic form H₂S (US EPA 1976).

The criterion for freshwater and marine concentrations of undissociated H₂S is 2.0 µg/L (US EPA 1976).

Hydrogen sulfide has been documented as contributing to fish kills in estuaries but ammonia toxicity kills in the literature appear to be limited to aquaculture facilities and sewage treatment plant spills. Diaz and Rosenberg summarize the findings of Theede (1969) in which mortality was approximately 20% greater when hydrogen sulfide concentrations were added to hypoxia experiments. Luther et al. (2004) concluded that hydrogen sulfide, and not ammonia, was most likely the primary cause of fish kills occurring in the “dead-end” canals of suburban Rehoboth Bay, Delaware.

Sediment toxicity and fine particle suspension

Estuarine sediment re-suspension can be harmful to aquatic organisms through the direct damage to the gill structure (Sigler et al. 1984; Lloyd 1987) or through the toxicity of a chemical that has become bound to the sediment particle (Hameedi et al. 1999). Sediment can be mobilized from the estuary bed by extreme weather events, dredging, or high flows. In 96-hour LC50 experiments with juvenile coho salmon (*Onchorhynchus kisutch*), stress responses were found to occur at concentrations > 40 g/L while mortality occurred at concentrations > 100 g/L, which is a value that is about an order of magnitude greater than the high concentrations found in salmonid streams (Lake and Hinch 1999). Anderson et al. (1996) found that duration of exposure had a greater effect on mortality than concentration. Greater particle size was found by Servizi and Martens (1987) to have greater effects on mortality.

Agricultural runoff is a non-point source of pollution in rivers, lakes, and estuaries across the US (Gilliom et al. 2006). Toxic chemicals released from non-point sources can sink to the bottom in low-energy, estuarine environments and become part of the sediment floor (Long 2000). In California, it has been shown that rain and irrigation water draining from urban and agricultural surfaces transport pesticides and other contaminants into surface waters and aquatic habitats (De Vlaming et al. 2000).

Water and sediment samples from drainage channels, rivers, and estuaries in numerous areas of California have been found to have levels of contaminants high enough to induce mortality in test organisms (Norberg-King et al. 1991; De Vlaming et al. 2000; Hunt et al. 2001). Toxicity has been found in drainage waters from both urban and agricultural runoff (De Vlaming et al. 2000; Bailey et al. 2000). Toxicity identification evaluations (TIEs) performed in many of these studies identified that toxicity to a commonly used test invertebrate was primarily caused by two organophosphate pesticides (OP): diazinon and chlorpyrifos (De Vlaming et al. 2000).

On the Central Coast of California, toxicity to resident mysid crustaceans was found in most agricultural drainages, 25% of wetland samples, and 11% of the samples from the main branch of the Pajaro River (Hunt et al. 1999). In that study, organochlorine and OP pesticide levels were potentially high enough to cause toxicity to test invertebrates. A Salinas River watershed assessment by Hunt et al. (2003) identified a contaminated agricultural drainage creek with levels of diazinon and chlorpyrifos that consistently caused toxicity to laboratory invertebrates.

A second phase of the Salinas River assessment was performed in order to determine the effects of the contaminated drainage creek on the river ecosystem by investigating water and sediment toxicity in situ and in the laboratory (Phillips et al. 2004). Researchers concluded that toxicity due to pyrethroids or organochlorines (Phillips et al. 2004) may explain the discrepancies between in situ and laboratory experiments and recommended further studies to investigate this possibility.

Pyrethroids, an alternative to historic OP pesticides, have been widely used in the last ten years (Werner personal communication). This group of chemicals is known to be sediment-associated, especially with smaller particles such as silts and clays, and is commonly used in nurseries (Werner personal communication).

Stratification vs. eutrophication

Density stratification in estuaries is generally controlled by temperature or salinity differences between surface and bottom waters and has been found to predict bottom water oxygen levels. In Western Long Island Sound, Welsh and Eller (1991) found oxygen depletion corresponded with thermal stratification. D'Avanzo and Kremer (1994) found salinity stratification caused bottom water hypoxia and anoxia in Waquoit Bay, Massachusetts. In the Neuse River Estuary, Buzzelli et al. (2002) found that the change in water column salinity and temperature were the two most important variables contributing to bottom-water DO concentrations, explaining 30% and 23% of variability respectively.

Eutrophication has been defined as an increase in the rate of supply of organic matter to an ecosystem (Nixon 1995). The most common cause of eutrophication is

anthropogenically-derived nutrient enrichment, particularly in the form of nitrogen and phosphorus (Nixon 1995). Vitousek et al. (1997) describes how nitrogen applications, particularly in agriculture, have accelerated the loss of biodiversity and altered the form and function of estuarine and near-shore ecosystems.

Several studies have created models that predict DO based on factors associated with nutrient enrichment. Kauppila et al. (2003) found that 55% of the variability in bottom water oxygen levels were predicted both as a function of the percentage of watershed under agriculture and of mean depth. In the creation of a two-dimensional hydrodynamic and water quality model, Park et al. (1996) concluded that hypoxia could be relieved more by decreasing carbonaceous biological oxygen demand than by increasing the DO concentrations in incoming bay water.

Bricker et al. (1999) created a method for categorizing the eutrophic status for estuaries. Below (Table 1) is a list of some of the parameters used by Bricker et al. (1999) to estimate the severity of eutrophication.

Table 1. Eutrophication criteria from Bricker et al. (1999).

Parameter	Hyper-eutrophic	High eutrophication	Medium eutrophication	Low eutrophication
Chlorophyll a	$\geq 60 \mu\text{g/L}$	$> 20, \leq 60 \mu\text{g/L}$	$> 5, \leq 20 \mu\text{g/L}$	$> 0, \leq 5 \mu\text{g/L}$
Secchi depth		$< 1 \text{ m}$	$> 1, \leq 3 \text{ m}$	$> 3 \text{ m}$
Surface nitrogen		$> 1 \text{ mg/L}$	$> 0.1, \leq 1 \text{ mg/L}$	$> 0, \leq 0.1 \text{ mg/L}$
Surface phosphorus		$> 0.1 \text{ mg/L}$	$> 0.01, \leq 0.1 \text{ mg/L}$	$> 0, \leq 0.01 \text{ mg/L}$

PROBLEM STATEMENT

The Central Coast region of California is one of the 10 fastest growing coastal areas in the US (Crossett et al. 2004). Rapid urbanization coupled with a well-established agricultural industry increases the pressure on coastal watersheds. Hunt et al. (2003) found pesticide concentrations at ecologically unhealthy levels in residential and agricultural runoff furrows of the Salinas River. Los Huertos et al. (2003) found that agricultural practices on the central coast are contributing nutrient concentrations that pose health hazards to humans and aquatic ecosystems. Zedler et al. (1996) identified nutrient enrichment and sedimentation as the two greatest threats to California coastal lagoon habitats.

Populations of aquatic species on the Central Coast continue to decline. A 2005 NOAA status review of West Coast salmon found that federally threatened steelhead (*Onchorynchus mykiss*) populations on the Central Coast of California were “likely to become endangered in the foreseeable future” and that coho populations on the Central Coast were “likely to become extinct” (Good et al. 2005). The federal listing for Coho salmon (*Onchorynchus kisutch*) was changed from threatened to endangered on June 28, 2005 for the Central California Coast ESU (NOAA 2005). The status of federally endangered tidewater goby (*Eucyclogobius newberryi*), a small coastal lagoon fish species, has been described in a 2005 Draft Recovery Plan (US Fish and Wildlife Service) as extirpated from 17% of its formal range and another 41 – 52% of its previous known range has been listed as either naturally too small or too degraded to maintain viable populations over the long term. The federally threatened red-legged frog (*Rana*

aurora draytonii) has been extirpated from over 70% of its former range and is now primarily isolated to central and southern California coastal drainages (US Fish and Wildlife Service 2002). The federally endangered San Francisco garter snake (*Thamnophis sirtalis tetrataenia*) is the most notable of these rare species as its range is limited to San Mateo and extreme northern Santa Cruz counties and its population is believed to be continuing to decline (Laurelsen personal communication).

The decline of anadromous species, especially steelhead, are of special concern on the Central Coast as they are recreationally and, therefore, economically important. Although there are no specific numbers on the economic contribution of steelhead to angler revenues, it was estimated in 1991 that a doubling of steelhead populations would increase state revenues by \$37.5 million a year (McEwan and Jackson 1996). The loss of steelhead is not only an economic concern, but one of biodiversity. The decline of salmonid species on the West Coast of the US is primarily due to the “destruction and modification of habitat, over-utilization for recreational purposes, and natural and human-made factors” (Busby et al. 1996).

The 1996 Steelhead Management and Restoration Plan for California (McEwan and Jackson) identifies three primary factors affecting steelhead habitat loss: over-appropriation of stream flow, blocked access to historic spawning and rearing areas, and chronic watershed perturbations that discharge sediment and debris into watercourses. In California, data do not suggest that other factors, such as marine ecological conditions, over-harvesting, or high seas gill nets, are having a great impact on steelhead populations (McEwan and Jackson 1996).

Research on the Central California coast is beginning to show that lagoon habitats play a much larger role than previously understood in the life history of steelhead (Smith 1990; Bond 2006). A study by Bond (2006) on Scott Creek, north of Santa Cruz on the central coast of California, found that lagoons provide essential rearing habitat for juvenile steelhead. The highly productive lagoon complex allows fish to reach larger sizes than those that rear in the upper watershed. Larger smolt size greatly increases ocean survival rates (Bond 2006).

In a study of the Pescadero Lagoon system, Smith (1990) collected scale samples from 27 returning adults. Scale analysis showed that at least 70% of those fish had reared in the lagoon. This suggests that lagoon-reared individuals have a far greater chance of returning to spawn. The benefit of the lagoon/estuary system to smolt growth and subsequent adult survival is disproportionate to the 1% of watershed habitat occupied by the lagoon (Bond 2006), suggesting that the lagoon systems that dot the California Coast may be providing critical habitat for steelhead.

Pescadero Marsh Natural Preserve fish kill

Pescadero Marsh Natural Preserve, a seasonal lagoon system on the Central Coast of California owned and managed by California State Parks (CSP) (Figure 3), has experienced a fish and invertebrate kill involving juvenile steelhead for seven of the last eleven years (Appendix I). The fish kill occurs during a dramatic draining event of the lagoon that takes place when the sandbar that separates the lagoon from the ocean is breached.

The lagoon at Pescadero Marsh forms when the estuary at the terminus of Pescadero and Butano Creeks is separated from tidal waters by the formation of a sandbar. The sandbar forms naturally during the summer or fall when freshwater flows are low, and usually remains intact until winter rain and wave events breach it. During sandbar closure, the lagoon and marsh plain become flooded as freshwater input from Pescadero and Butano Creeks continues (Figure 4). Eventually pressure from the engorged lagoon system, in combination with ocean wave action at the mouth, erodes the sandbar and reestablishes tidal connection with the Pacific Ocean. Figure 5 illustrates the force with which the lagoon drains.

Figure 3. Aerial photo of Pescadero Marsh Natural Preserve (Aerial photo courtesy of CSP).

Figure 4. Pescadero Marsh water level during sandbar open (pink) and sandbar closed (blue) conditions (Map produced by author with permission of CSP).

Figure 5. Mouth of Pescadero Lagoon during the 2005 breaching event (photo courtesy of Eric Abma).

Carcass surveys conducted after the 1995 (Figure 6) and 2003 (Figure 7) events revealed an uneven distribution of fish carcasses. Staghorn sculpin (*Leptocottus armatus*), topsmelt (*Atherinops affinis*), jacksmelt (*Atherinopsis californiensis*), three-spined stickleback (*Gasterosteus aculeatus aculeatus*), starry flounder (*Platichthys stellatus*), dunghness crab (*Cancer magister*), lined shore crab (*Pachygrapsus crassipes*), and most notably federally threatened juvenile steelhead trout (*Oncorynchus mykiss*) carcasses were found at the scene (ESA 2004; Corbett personal communication; Arnold personal communication; Augustine 1995). To date, this is the only known fish kill on

Figure 6. 1995 fish kill carcass distribution (Augustine 1995 CSP safety report; reprinted with permission from CSP).

Figure 7. 2003 fish kill carcass distribution (CSP undated; reprinted with permission).

the Central Coast of California to be associated with a sandbar breaching event.

Appendix I describes the fish kill history at Pescadero Marsh.

Although the exact cause of the kill is unknown, past monitoring has revealed hypoxic (< 2 mg/L) and anoxic (0 mg/L) conditions in marsh bottom waters before sandbar breaching (Smith 2004) (ESA 2004) (SH&G 2003). In 2003, several carcasses were taken to the University of California at Davis Fisheries Pathology Lab to be analyzed for parasites and disease. There was no evidence of pathological related stress and full stomachs indicated the fish had been eating prior to the event. Pathologist Ron Hedrick concluded that the cause is most likely environmental in nature (ESA 2004). Local scientists agree that the cause of the kills is most likely a water quality related event, however the re-suspension of sediment-bound toxic chemicals or gill siltation by fine marsh sediments have not been ruled out (Smith 2004; ESA 2004; SH&G 2001; Atkinson personal communication; O'Doherty personal communication).

Competing hypotheses regarding cause of poor water quality in Pescadero Lagoon

Experts familiar with the Pescadero lagoon system have several hypotheses regarding the cause of poor water quality. Smith (2004) speculates that salinity stratification is the primary cause of anoxia in bottom waters while SH&G (2002) implicates nutrient enrichment. Figure 8 is a conceptual model that illustrates these two concepts.

Figure 8. Conceptual model of stratification and eutrophication in Pescadero Lagoon during sandbar closure.

Density stratification in Pescadero creeks and lagoon has been well documented after sandbar formation (Smith 1990; ESA 2004). Without vertical mixing, bottom water temperatures rise and DO concentrations decrease (Smith 1990; ESA 2004). In time, bottom water dissolved oxygen levels reach hypoxic and then anoxic levels. Smith (2004) suggests that the breaching event mixes the anoxic, saline bottom layer with the fresh, oxic surface layer resulting in an overall oxygen deficit that kills fish and invertebrates.

SH&G (2002) implicates watershed-born nutrients, and the subsequent eutrophication of marsh waters, as the source of bottom water hypoxia and anoxia in

Pescadero lagoon and creeks. Nitrates and phosphates from septic systems, urban activities and agricultural practices (i.e. greenhouses) are thought to fuel the excess production of vegetation, and contribute to unnaturally high levels of organic matter deposition into bottom waters (SH&G 2002). This excess organic matter deposition fuels heterotrophic bacteria respiration and leads to bottom water anoxia. Under anoxic conditions, bacteria begin to use hydrogen sulfate as an alternative electron acceptor, creating a build-up of hydrogen sulfide (H_2S). It is the mixing of hydrogen sulfide into the water column upon sandbar breaching that SH&G (2002) believes is killing fish and invertebrates.

Smith (2004) believes that a recent change in the timing of sandbar formation is a critical factor in the occurrence of fish kills. Prior to 1995 the sandbar formed in late spring/early summer and broke in late fall. Today formation does not occur until early to mid fall. Smith (2004) hypothesizes the replacement of the Highway One bridge at the mouth of Pescadero Lagoon by California Department of Transportation (Figures 9a and 9b) and/or a restoration project implemented by CSP (Figure 10) is responsible for the change in sandbar dynamics. Smith (2004) believes a longer period of sandbar closure would result in a freshwater conversion of the water column, an alleviation of bottom water anoxia and a reduced chance for fish and invertebrate kills. SH&G (2002) has stated that the break up of salinity stratification by freshwater conversion, provided there is sufficient photosynthetic activity in surface waters, may not prohibit the formation of hydrogen sulfide and prevent fish and invertebrate kills.

Figure 9. Highway One Bridge over Pescadero Lagoon in 1985 (left) (reproduced from Curry et al. 1985 with permission from CSP) and new Highway One Bridge that was completed in 1991 (right) (photo by author).

Figure 10. Synopsis of California State Parks 1993 and 1997 restoration in Pescadero Marsh (Unpublished map reproduced with permission of CSP).

Understanding the nature of poor water quality and how it contributes to fish kills is vitally important for steelhead and lagoon management along the entire Coast of California. If stratification-driven anoxia due to shorter sandbar closures is indeed responsible for the fish kills, local and state agencies will need to take this in consideration when manipulating sandbar regimes. If nutrient enrichment can be linked to fish kill events, watershed managers will be required to reconsider land-use practices.

OBJECTIVES

There are two primary objectives of this exploratory study. The first objective is to identify the most likely cause of aquatic species mortality among the following possibilities: anoxia, hydrogen sulfide and/or ammonia poisoning, sediment toxicity, or gill siltation. The second objective is to determine the respective roles of stratification and eutrophication in the production of bottom water anoxia and hypoxia.

The investigations in this study stem from the following research questions:

- I. What is most likely responsible for the fish kill in Pescadero Marsh?
 - Q₁: Are dissolved oxygen levels low enough to cause mortality during sandbar breach?
 - Q₂: Are ammonia and hydrogen sulfide levels high enough in bottom waters prior to breach to contribute to a fish kill?
 - Q₃: Are there any significant differences between sites for dissolved oxygen, ammonia, or hydrogen sulfide that may explain the uneven carcass distribution?
 - Q₄: Are there toxins present in the sediment that may be contributing to mortality during the fish kill?
 - Q₅: Is the particle size distribution of sediment mobilized during the breach and the duration of the event likely to cause steelhead mortality?
- II. What is driving poor water quality in Pescadero Marsh, density stratification or eutrophication?
 - Q₆: How widespread spatially and temporally is stratification?
 - Q₇: What do indicators of eutrophication tell us about the trophic status of the system?

STUDY SITE

Pescadero Marsh Natural Preserve is located approximately 15 miles south of Half Moon Bay on the San Mateo County Coast (Figure 11). The 340-acre preserve serves as a catchment basin for the 210-square kilometer watershed of Pescadero and Butano Creeks (Figure 12). The lagoon complex includes two main creek arms, several channels, an inactive levee system, and a labyrinth of backwater ditches and depressions (Figure 13).

The Pescadero-Butano watershed is approximately one-third timber land, one third agricultural preserve land, and one-third protected lands (Figure 14). Rainfall during the 2004-2005 water year was 30.39 inches (California Department of Forestry personal communication), which was above the yearly average of 28.7 inches (based on 41-year average between 1954 and 1995 at neighboring watershed of San Gregorio) (NCDC 2006). Mean stream flow for the 2005 water year was 51.3 ft³/s, slightly above the 53-year average of 43 ft³/s (USGS 2006).

As sea level began to rise approximately 11,000 years ago, the historic river canyon that was Pescadero Creek filled with sediment (Williams 1990). Sea level stabilized to its current level approximately 5,000 years ago and Pescadero Marsh began to assume its current form. The first map of the marsh was drawn in 1854 by the United States Coast Survey (Figure 15). At that time Spanish settlers had been using the area as rangeland for well over a hundred years, but the effects on the marsh were probably insignificant (Viollis 1979).

Beginning in the 1920's, European settlers created an impressive network of levees, gates, and channels for the purposes of farming the fertile marsh plain (Viollis 1979).

Levees isolated portions of the marsh plain from flooding, tide gates controlled seawater inflow, and a system of channels provided freshwater for irrigation. Cultivation of the marsh plain peaked in the 1950's but was well on the decline by the 1960's (Williams 1990). Attempts by the California Department of State Parks and Recreation to restore natural tidal flux resulted in the breaching of several levees and the creation of several new channels (Figure 10). For the purposes of this study, areas of the marsh complex will be referred to as the seven hydrologically and morphologically unique bodies of water and three isolated marsh plains listed in Figure 16.

Figure 11. Location of Pescadero Marsh Natural Preserve on the San Mateo County Coast of California (Map reprinted with permission from Microsoft Corporation).

Figure 12. The Pescadero-Butano watershed boundary (Modified from ESA 2004).

Figure 13. Contour map of Pescadero Marsh Natural Preserve (Map reprinted with permission from MAPTECH).

Figure 14. Pescadero-Butano watershed land-use map (Unpublished map reproduced with permission of CSP).

Figure 15. US Coast Survey 1854 Pescadero Marsh map (Reproduced with permission from William Cook, *The Flooding of Butano Creek*, 2003 Edition).

Figure 16. Seven general water bodies (orange) and three isolated marsh plains (yellow) in Pescadero Marsh Natural Preserve (Unpublished map reproduced with permission of CSP).

MATERIALS AND METHODS

Between September 2004 and December 2004, seven representative marsh sites (Figure 17) (BC1, BCh, HO, NM, NP, PC, and PCh) were monitored weekly. Vertical profiles were conducted at 0.25 m intervals for dissolved oxygen, salinity, and temperature. Bottom water tests for hydrogen sulfide concentrations were also conducted. Six continuous monitoring devices recorded bottom water dissolved oxygen, salinity, temperature, depth, and pH between November 15, 2004 and December 10, 2004, at the following sites: BC1, BCh, BCB, BC2, CONF, and PCh (Figure 17). A survey of fish carcasses after the December 5, 2004, breach event was conducted throughout the marsh complex during the first low tide after lagoon draining.

Beginning several days after the December 5, 2004, sandbar breaching, an expanded one-year project was begun. While the sandbar was open (12/5/04 – 10/10/05), the following data was collected at each of the seven sites the first full week of each month, on an outgoing tide: dissolved oxygen, salinity, and temperature at 0.25 m intervals; duplicate concentrations of nitrate, nitrite, ammonia, phosphate, and chlorophyll a approximately 30 cm below the surface; bottom water hydrogen sulfide concentrations; maximum and average surface wind speed; air temperature and depth. Beginning in March 2005, site BC2 was added to make a total of eight marsh sampling sites. After sandbar closure on October 10, 2005, sampling frequency was increased to once a week for all water quality parameters except nutrients and chlorophyll a which were sampled bi-weekly. Sondes for the monitoring of bottom water dissolved oxygen, salinity, temperature, pH, and depth were in place at sites HO and BCh between

Figure 17. Pescadero Marsh and creek monitoring stations (Unpublished map reproduced with permission of CSP).

September 23, 2005 and December 10, 2005. An additional 14 sondes with oxidation reduction potential (ORP) probes were put in at sites BC1, BC2, BCB, BCh, CONF, HO, PC, and PCh (Figure 17) to make for a total 8 sites with surface and bottom sondes.

Duplicate nitrate, nitrite, ammonia, phosphate, and chlorophyll a samples were also collected monthly beginning on December 7, 2005, from Pescadero Creek at Stage Road and Butano Creek at Pescadero Creek Road (Figure 17). Pescadero Creek stream flow was downloaded for the time of nutrient sample collection from the USGS monitoring station number 11162500 real-time website (USGS 2006). Samples were collected bi-weekly alongside marsh samples during sandbar closure (October 10, 2005 – December 4, 2005), however two additional creek samplings took place on November 21, 2005 and November 28, 2005 for the characterization of first flush events.

Sediment and vegetation data were collected approximately monthly between March and October 2005 while the mouth of the lagoon remained open. After sandbar formation on October 8, 2005, sediment samples were collected twice, once two days prior to breach on December 2, 2005, and once just after breach on December 10, 2005. Vegetation percent cover was not surveyed during sandbar closure but was sampled just prior to lagoon closure on September 9, 2005, and immediately after sandbar breach on December 10, 2005. The depth and hydrogen sulfide concentrations during sandbar closure made sediment data collection dangerous and vegetation percent cover sampling impossible.

Weather, wave, and rain data were collected from September 2004 to January 2006. Weather and wave data were downloaded from real-time data sets for Half Moon Bay buoy number 46012 (NDBC 2006). Daily rainfall totals were obtained from California Department of Forestry's station on Pescadero Creek Road in Pescadero, California. Figure 18 synthesizes the entire sampling schedule for this project.

Daily

Monthly

Bi-weekly

Weekly

Continuously

[illegible]

Figure 18. 2004/2005 Pescadero Marsh sampling schedule.

Water quality

On the first Tuesday of each month, at approximately 2 hours after high tide, each of the eight monitoring sites were sampled for oxygen, salinity, temperature, bottom water hydrogen sulfide concentration, surface wind speed, secchi depth, and nutrient concentrations. Profiles of the vertical distribution of oxygen, salinity, and temperature were recorded using an YSI 85 handheld probe. Readings were taken at 0.25 meter intervals from the surface to the bottom. The probe was calibrated to atmospheric oxygen concentrations prior to each sampling day. If calibration was not within 95% accuracy, the sensor fluid and membrane were replaced.

Water samples for hydrogen sulfide concentrations were collected in a polyethylene, two-liter Van Dorn bottle. The unit was lowered to the bottom and moved quickly up and down for the purposes of including bottom sediment. The sample was quickly analyzed for hydrogen sulfide concentrations using a HACH HS-C field test kit. The colorimetric test kit detects concentrations between 0.1 mg/L and 5.0 mg/L.

Secchi depth was measured with a standard, weighted, polyethylene black and white secchi disc. Wind speed was measured at the water's surface perpendicular to predominant wind direction using a handheld, Kestral 1000 anemometer. Prevailing wind direction and speed were obtained from historical records of Half Moon Bay buoy 46012 (NDBC 2006). Air temperature was measured with a handheld digital thermometer. Depth was recorded with a handheld stadia rod graduated in increments of hundredths of a foot.

Duplicate nutrient samples were collected in two 250-ml, dark polyethylene bottles at each of the eight marsh monitoring sites (Figure 17). Bottles were triple rinsed with sample water and filled by hand at approximately 30 centimeters below the surface. During bar closure, a duplicate set of samples were collected from bottom waters with a Van Dorn bottle. Samples were immediately placed on ice and kept in the dark until they were either frozen or filtered in the lab. Due to the timing of the tides, some sampling events allowed us to filter samples the same day while other days required samples to be frozen for several days before they could be filtered.

Sixty milliliters of sample were filtered by vacuum through a 0.7 micron glass fiber filter (GF/F) at approximately 2 drops/second. Filters were digested for 24-hours in a 90% acetone solution and analyzed for chlorophyll a concentrations following the procedures of Strickland and Parsons (1972). The 60-ml filtrate was analyzed for nitrite (NO_2^-) and ammonia (NH_4^+) using Strickland and Parsons (1972) spectrophotometric methods. Nitrate (NO_3^-) + Nitrite (NO_2^-) and soluble reactive phosphate (SRP) were analyzed using the Alpkem rapid flow method of Sakamoto *et al.* (1990). Analysis usually occurred within 24 hours after filtration, however, on occasion, samples were refrozen after filtering and stored for analysis which was always within 1 month. Detection limits were 1 $\mu\text{g/L}$ for all analytes.

Water quality dynamics of the sandbar breaching event were captured by continuous monitoring sondes. Surface and bottom continuous monitoring probes (HACH Minisonde 4a's) were placed at eight sites (Figure 17) within the marsh complex on November 23, 2005, and removed on December 9, 2005. Sondes took a reading every

15-minutes for temperature, pH, dissolved oxygen, salinity, oxidation reduction potential (ORP), and depth. Sondes were pre-calibrated in HACH Laboratories in Loveland, CO for every parameter except DO and depth.

DO was calibrated using atmospheric oxygen concentrations in the Half Moon Bay field office one day prior to deployment. Depth was calibrated *in situ* just prior to deployment. QA/QC procedures for DO probes included post-deployment recordings for atmospheric oxygen saturation. If readings were outside the acceptable margin of error ($\pm 5\%$) data was not used in final analyses. No attempts were made to correct sonde data for any detected drift as there is no standard method for this procedure.

Bottom water sondes were placed inside three-inch PVC tubes approximately 10-15 centimeters above the sediment bed (Figure 20). The PVC tube was anchored to a half-inch pipe that was buried approximately 1.5 m into the sediment. Surface sondes were placed into a PVC tube that floated just below the surface. Both Minisonde units were attached to a third anchor via a coated stainless steel cable.

Figure 19. Continuous monitoring sonde *in situ* housing (illustration by author).

Benthic sediment cores

Sediment cores were taken at each of the eight water quality sites (Figure 18) monthly between March and December 2005. At each site, a 2 in diameter benthic push core was used to extract an 8 cm long core. The top 2 cm of sediment was measured and placed into Ziploc baggies. Samples were homogenized and broken into two sub-samples, one for freezer storage and one for refrigeration. Frozen samples were analyzed for organic carbon content and the refrigerated samples were analyzed for particle size. Organic carbon content was obtained through loss on ignition methods. A dried weight was obtained, the sample was then brought to 480 °C, a temperature which will burn off any organic carbon particles and then a post-burn weight was obtained. The difference

between the pre- and post-burn weight represents the weight of carbon that was stored in the sediment.

Particle size was determined with a Beckman-Coulter LS230 laser diffraction particle analyzer. Sediment samples predominately made up of clays and silts were processed using a wet module. Samples were homogenized inside the Ziploc baggie, sub-sampled with a spatula and suspended in the wet module holding tank. Sediment was not pre-treated. Sandy samples were dried in an oven for 48 hours at 40 °C, separated with a mortar and pestle and added, whole sample, to the dry module.

Vegetation production

Percent cover of vegetation was estimated with a 0.25 m² quadrat divided into 30 equal squares by fishing line to make 25 intersections. Each intersection that came into contact with a plant or alga was recorded as one point-of-contact for a given species. A 30-meter transect was laid on the right bank of each water quality monitoring site (Figure 17) and quadrats were laid at five, randomly chosen locations on the transect line. Percent cover was measured approximately once every two weeks from mid-May until sandbar formation and then once again just after sandbar breaching.

Toxicity testing and chemical analysis

Sediments were collected from the lagoon and from Butano Channel (Figure 17) in six, 250-ml, amber glass, titanium capped bottles on 11/19/05. One liter of bottom water was also collected from each site in a titanium capped glass bottle. The jars were taken to depth, opened, filled with sediments by scraping the jar along the surface, and then recapped so as to avoid contact with air. The 1-liter bottle of bottom water was also

filled and capped at depth. Five gallons of surface water from the lagoon was also collected. Samples were kept on ice, but not frozen, and transported to the State of California Department of Fish and Game (CDFG) Laboratory the next day.

Sediments and bottom waters were analyzed by the CDFG Water Pollution Control Laboratory for organochlorines, organophosphates, carbamates, herbicides, and pyrethroids using methods outlined in Appendix III. A static definitive acute sediment toxicity test was performed on 8-month old rainbow trout (*Onchorynchus mykiss*) (fork length 7-10 cm) using the modified methods of Finlayson et al. (2000) and US EPA (1993). Fish were submerged in a 4:1 water:sediment elutriate for 96-hours.

Analysis

For the purposes of determining the toxic status of bottom waters two days prior to breach, bottom water DO, hydrogen sulfide and ammonia concentrations from December 2, 2005 were compared to EPA acute criteria for freshwater. In an attempt to identify areas of the marsh with potentially greater concentrations of toxic bottom waters during sandbar closure, a one-way factorial ANOVA was performed by site for DO, hydrogen sulfide, and ammonia. Particle size distribution during lagoon draining was compared to known lethal concentrations found in the literature.

To determine the relative roles of salinity and temperature in density stratification, a linear regression analysis was performed between density and temperature and density and salinity. Density was determined using the equation of state in Appendix IV. To evaluate the relative contributions of stratification and nutrient enrichment to bottom water anoxia during sandbar closure a multiple, stepwise, backward regression was

performed. The dependent variable DO was compared to the following independent variables: mean surface wind speed, nitrate concentrations, SRP concentrations, and salinity.

ANOVA was used to determine spatial and temporal patterns for the followings variables: DO, salinity, temperature, nitrate, nitrite, phosphate, ammonia, chlorophyll a, and surface wind speed. These variables were compared by site, season, sandbar status, and the two-way interaction between site*sandbar status. Two *a priori* comparisons were made for the site variable, Butano Channel was compared to all other sites and then Butano Creek 1 and Butano Creek 2 were compared to all other sites.

Continuous surface water dissolved oxygen data from the first 24-hours after sandbar breach was compared for significant differences among sites using a one-way factorial ANOVA. Continuous DO data from the entire five days after breaching was compared using a General Linear Model as data sets were not the same size. Santa Cruz County, CA creek nutrient loading rates were compared to those of Pescadero and Butano with a one-way factorial ANOVA analysis. All analyses seek to avoid type I statistical error so α is equal to 0.025.

RESULTS

In the fall of 2004, the sandbar formed on October 10 and broke on the evening of December 5 at approximately 5:00 pm. Figure 20 reveals the carcass distribution observed during a survey of the lagoon complex on Monday December 6, 2004 between 9:00 am and 4:00 pm. The tidal information for December 6, 2004 was as follows: 5:52 +5.1 ft, 12:24 +1.8 ft, and 17:49 +3.7 ft. In Butano Creek, in the area of the Butano Channel inlet, one starry flounder (*Platichthys stellatus*) and one staghorn sculpin (*Leptocottus armatus*) were observed sitting moribund on the creek bank. Also in this area one medium steelhead (*Oncorhynchus mykiss*), approximately 30 cm in length, was observed gulping for air at the surface, however, upon being approached, the fish swam off vigorously. In Pescadero Creek, approximately half way between monitoring site PC and the confluence with the lagoon, a school of several hundred topsmelt (*Atherinops affinis*) and jacksmelt (*Atherinops californiensis*) were observed swimming erratically up and downstream. As the school swam back and forth individual fish were observed floating upside down on the surface and lying dead on the bottom.

In 2005, the sandbar formed on October 8 and broke on December 4 at approximately 5:00 pm. Carcass surveys were conducted on Monday December 5. The tidal information for December 5 is as follows: 2:12 +4.5 ft, 6:39 +3.4 ft, 12:22 +5.8 ft, and 19:40 – 1.0 ft. At approximately 9:00 am on December 5 a preliminary search of the left bank of Butano Creek, downstream of the Butano Channel/Butano Creek confluence, was conducted by local fisherman John Fowler (Frahm personal communication). Fowler counted 50 steelhead (*Oncorhynchus mykiss*) carcasses in his 35-meter search. A

search of the entire marsh area during the second low tide revealed the carcass distribution in Figure 21.

Figure 20. 2004 fish kill carcass distribution (Unpublished map reproduced with permission of CSP).

Figure 21. 2005 fish kill carcass distribution (Unpublished map reproduced with permission of CSP).

Cause of mortality

Anoxia. Continuous data sets for surface and bottom water DO, salinity, temperature, ORP, pH, and depth (Figures 22-29) were recorded between November 23, 2005, and December 9, 2005, for the eight sonde sites shown in Figure 17. Surface sonde data sets from monitoring locations PC, BC2, and CONF were incomplete or lost due to

power failure (see Figure 17 for sonde locations). Portions of data sets were removed if there was significant evidence that the sonde was out of the water and directly exposed to the atmosphere. The bottom sonde at the Butano Channel Bridge site (BCB) may have been stuck in the sediment as the dissolved oxygen levels never rose above 1.0 mg/L during the breach event or in the days that followed. The bottom water sonde at site BCh did not pass post-deployment QA/QC standards as it produced DO readings that were 1.5 times greater than the atmospheric standard. Although not used in analyses, these incomplete or questionable data sets are still presented here.

Prior to breaching, bottom water DO levels were < 0.5 mg/L while surface water DO levels were generally > 5.0 mg/L. Salinity levels were brackish in bottom waters and generally fresh in surface waters. Bottom water temperatures were relatively stable around 15 °C while surface temperatures were lower and tended to fluctuate over time. ORP levels in bottom waters hovered around -50 mV while surface water levels hovered around 400 mV. Surface water pH was generally between 7.5 and 8.0 pH units. Bottom water levels reached as low as 4.5 at site BC2 (Figure 23) and as high as 9.5 at the confluence of Pescadero and Butano Creeks (site CONF) (Figure 26). Surface water elevations were very similar between sites with the sandbar breach occurring at approximately 2.0 m above mean sea level.

Upon sandbar breaching, sites with complete data sets illustrated a complete mixing of the water column with a convergence of surface and bottom water DO, temperature, pH, ORP, and salinity (Figures 23-30). However, there were several anomalies in the data sets. During the initial breach, salinities at site HO were inverted

with the saltier water on the surface, possibly due to ocean water rushing in (Figure 27). ORP and pH at the Butano Channel (BCh) site never converged (Figure 24). DO, temperature, salinity, and ORP at the Butano Channel Bridge site (BCB) indicated limited mixing (Figure 23). Although bottom water DO and salinity data sets from the Butano Channel sites BCh and BCB indicate an initial mixing, they quickly return to pre-breach levels (Figures 23 and 24).

Surface DO readings from both Butano Channel sondes, during the first 24 hours after the sandbar breach, were < 3 mg/L 79% of the time and < 2 mg/L 53% of the time (Table 2). The lowest surface water DO concentration in Butano Channel (BCh and BCB) was 1.7 mg/L at 20:00 hours on 12/4, approximately 8-hours before fish carcasses were exposed the next morning (Table 2). In Pescadero Channel, the minimum DO reading of 0.40 mg/L was recorded at 9:45 am on 12/5 (Table 2). Excluding questionable data sets from the two Butano Channel sites, bottom water DO levels were lowest in Butano Creek (BC1), downstream of the Butano Channel inlet with over 65% of the readings < 1 mg/L (Table 3). In the post-breach, five-day data set, Pescadero Channel had the lowest surface water DO concentrations with 48% of the readings < 1 mg/L (Table 4).

Figure 22. Continuous DO, salinity, temperature, pH and ORP data for site BC1.

Figure 23. Continuous DO, salinity, temperature, pH and ORP data for site BC2.

Figure 24. Continuous DO, salinity, temperature, pH and ORP data for site BCh.

Figure 25. Continuous DO, salinity, temperature, pH and ORP data for site BCB.

Figure 26. Continuous DO, salinity, temperature, pH and ORP data for site CONF.

Figure 27. Continuous DO, salinity, temperature, pH and ORP data for site HO.

Figure 28. Continuous DO, salinity, temperature, pH and ORP data for site PC.

Figure 29. Continuous DO, salinity, temperature, pH and ORP data for site PCh.

After the initial dynamic breaching event, DO, salinity, and temperature oscillations were indicative of tidal action, with the exception of the Butano Channel sites (BCh and BCB) which had consistently low bottom water DO concentrations (Figures 22-29). Dissolved oxygen levels at all other sites were low during low tides, sometimes almost to 0.0 mg/L, and elevated during high tide events. Temperature, salinity, pH, and ORP increased during high tide and decreased during low tide.

Surface and bottom sonde readings from the first 24 hours after sandbar breach were broken down into the percentage of readings that were below 3 mg/L, 2 mg/L and 1 mg/L (Tables 2 and 3). Butano Channel sites (BCh and BCB) had significantly lower surface DO levels ($df = 3$, $p < 0.001$) than sites HO and PCh, the only two sites with large enough data sets for comparison purposes (Figure 30).

Table 2. Minimum DO, mean temp and percent of total surface sonde readings (N) that are < 3 mg/L, < 2 mg/L, and < 1 mg/L during the first 24 hours after sandbar breaching.

Site	N	% readings < 3 mg/L	% readings < 2 mg/L	% readings < 1 mg/L	Minimum DO (mg/L)	Mean temp
BC1	17	41.18	29.41	0.00	1.03 (12/4 @16:53)	11.23
BCh	97	79.38	53.47	0.00	1.16 (12/4 @ 22:12)	8.72
BCB	97	80.41	59.79	0.00	1.26 (12/4 @ 21:42)	8.93
PCh	97	19.59	9.28	7.22	0.40 (12/5 @ 9:45)	10.24
HO	97	26.80	7.21	0.00	1.7 (12/4 @ 19:03)	9.63

Figure 30. Box plot describing median, 10th, 25th, 75th, 90th percentiles, and outliers of 24-hour surface sonde data.

Table 3. Minimum DO, mean temperature and percent of total bottom sonde readings (N) that are < 3 mg/L, < 2 mg/L, and < 1 mg/L during the first 24 hours after sandbar breaching.

Site	N	% readings < 3 mg/L	% readings < 2 mg/L	% readings < 1 mg/L	Minimum DO (mg/L)	Mean temp
BC1	97	46.39	25.77	5.15	1.13 (12/4 @ 17:45)	8.91
BC2	97	81.48	71.13	65.01	0.32 (12/5 @ 3:06)	9.42
CONF	97	39.13	34.02	23.71	0.09 (12/4 @ 10:46)	9.29
HO	97	32.99	9.28	2.06	1.35 (12/4 @ 18:24)	9.88
PC	23	57.73	13.04	0.00	1.92 (12/4 @ 14:10)	9.7
PCh	97	35.05	10.31	4.12	0.40 (12/5 @ 9:45)	10.59

Table 4 shows the percentage of surface sonde readings that were < 3 mg/L, < 2 mg/L and < 1 mg/L for six sites (BC1, BCB, BCh, HO, PC, and PCh) for the duration of

deployment (approximately five days). For this time period, Pescadero Channel had significantly lower DO levels than all other sites ($df=5$, $p<0.001$) (Figure 31).

Table 4. Minimum surface DO, mean temperature and percent of total sonde readings (N) that were < 3 mg/L, < 2 mg/L, and < 1 mg/L during the five days after sandbar breach.

Site	N	% readings < 3 mg/L	% readings < 2 mg/L	% readings < 1 mg/L	Minimum DO (mg/L)	Mean temp
BC1	385	11.43	10.13	3.90	0.66 (12/9 @ 2:38)	8.21
BCh	327	49.54	35.78	9.79	0.21 (12/7 @ 14:57)	8.66
BCB	470	40.00	22.13	0.00	1.21 (12/8 @ 00:42)	8.71
HO	475	13.05	7.16	3.58	0.23 (12/9 @ 00:33)	10.00
PC	161	18.63	18.63	18.63	0.35 (12/5 @ 23:55)	8.15
PCh	664	74.85	68.98	48.64	0.13 (12/6 @ 11:00)	11.98

Figure 31. Box plot describing median, 10th, 25th, 75th, 90th percentiles, and outliers of continuous surface DO data for the five days following sandbar breach.

Ammonia and hydrogen sulfide toxicity. Prior to the sandbar breach, bottom water ammonia concentrations did not exceed acute US EPA criteria except for at the Highway One site (Table 5). Bottom water hydrogen sulfide levels exceeded US EPA (1976) acute toxicity concentrations at several locations prior to sandbar breaching (Table 6). Discrete bottom, middle, and surface water samples taken approximately three hours after sandbar breach were tested for hydrogen sulfide concentrations at three locations: the outlet of Pescadero Channel (south of the culverts), Butano Channel bridge (BCB) and downstream of the Pescadero/Butano confluence (CONF). Although the olfactory detection of hydrogen sulfide was observed at all three sites, all tests returned negative results (Table 7). However, the detection limit (0.1 mg/L) of the test used in this study was not sufficient to detect sulfide down to the US EPA acute toxicity criteria of 2.0 µg/L.

Table 5. US EPA (1999) acute ammonia criteria calculation results of bottom water samples two days prior to sandbar breach.

Date	Site	24-hour pH mean	Mean ammonia (µM)	Mean ammonia (mg/L)	EPA acute criteria (mg/L)
12/2/05	BC1	7.34	74.89	1.05	16.63
12/2/05	BCh	6.51	63.73	0.89	32.48
12/2/05	BC2	5.39	93.53	1.31	38.41
12/2/05	HO	8.33	513.14	7.18	2.97
12/2/05	PCh	7.05	9.11	0.13	23.03

Table 6. Percent of unionized (toxic) hydrogen sulfide calculation results (US EPA 1976b) from bottom water concentrations two days prior to sandbar breach.

Date	Site	HS	24-hour mean pH	Estimated % unionized (toxic) HS
12/2/2005	BC1	-	7.34	-
12/2/2005	BCh	1	6.51	40
12/2/2005	BC2	5	5.39	90
12/2/2005	HO	5	8.33	15
12/2/2005	LA1	4.5	8.33	15
12/2/2005	PCh	-	7.05	-

Table 7. Hydrogen sulfide concentrations during 2005 lagoon draining event.

Date	Time	Site	Bottom HS mg/L	Middle HS mg/L	Bottom HS mg/L
12/4/2005	5:30	PCh	< 0.01	< 0.01	< 0.01
12/4/2005	6:00	CONF	< 0.01	< 0.01	< 0.01
12/4/2005	6:20	BCB	< 0.01	< 0.01	< 0.01

Sediment toxicity. California Department of Fish and Game (CDFG) Water Pollution Control Laboratory found negative results for sediment and pore water concentrations of pyrethroids, organochlorines, organophosphates, carbamates, and herbicides at the two sites samples (CONF and BCB). High levels of sediment sulfur were found in both test locations. Butano Channel (BCB) sediments had 205 µg/L and the lagoon, just downstream of the confluence of Pescadero and Butano Creeks (CONF), had 595 µg/L (Appendix III). CDFG's Aquatic Toxicology Laboratory found zero mortality of juvenile rainbow trout in a 96-hour definitive acute test (Appendix IV) on Pescadero Marsh sediment and bottom water.

Particle size of suspended sediments. Results from discrete sampling of surface, middle, and bottom waters for particle size are below (Figures 32 and 33). Mobilized

sediment particle size distribution in Butano Channel and the lagoon, downstream of the Butano Creek and Pescadero Creek confluence, were significantly smaller than those in Butano Creek (BC1) upstream of the Butano Channel inlet.

Figure 32. Suspended particle size distribution during sandbar breach.

Figure 33. Size of the sediment particles that comprised 90% of the total sample.

Significant spatial differences for naturally occurring toxics. The concentrations of DO, ammonium, and hydrogen sulfide were compared by site for the season of fall when the fish kill is known to occur (Figures 34 and 35). The results are listed in Table 8. No significant differences were found between sites.

Figure 34. Dissolved oxygen (a) and ammonia (b) box plot for fall data distribution.

Figure 35. Hydrogen sulfide fall means +/- SE.

Table 8. One-way ANOVA results for fall DO, ammonia, and HS by site.

Dependent variable	df	P-value
DO	7	0.687
Ammonia	7	0.782
Hydrogen sulfide	7	0.498

Evidence of eutrophication

Summer and fall hypoxia and anoxia. Surface and bottom water DO levels diverged in the fall (September 22 to December 22) during sandbar closures in both 2004 and 2005 (Figures 36-43). The sandbar was in place between October 10th and December 5th in 2004 and between October 10th and December 4th in 2005. Bottom water oxygen levels remained hypoxic to anoxic throughout sandbar closure. Surface levels fluctuated but were generally much higher than bottom water DO, except in North Marsh (Figure 41). The radical change in bottom water DO levels just after sandbar breaching can be seen at each of the eight sites after the 2004 and 2005 breaching.

Figure 36. Temporal distribution of surface and bottom DO data for site BC1.

Figure 37. Temporal distribution of surface and bottom DO data for site BC2.

Figure 38. Temporal distribution of surface and bottom DO data for site BCh.

Figure 39. Temporal distribution of surface and bottom DO data for site HO.

Figure 40. Temporal distribution of surface and bottom DO data for site NM.

Figure 41. Temporal distribution of surface and bottom DO data for site NP.

Figure 42. Temporal distribution of surface and bottom DO data for site PC.

Figure 43. Temporal distribution of surface and bottom DO data for site PCh.

Dissolved oxygen concentrations were also very low on the flooded marsh plain during sandbar closure (Figure 44). Pescadero and Delta Marsh had the highest levels of bottom water hydrogen sulfide (Figure 44).

Figure 44. November 14, 2005 flooded marsh plain DO and hydrogen sulfide bar chart.

Fall diel oxygen fluctuation. Pre-sandbar data sets were obtained for Butano Channel (BCh) and Highway One (HO) sites. DO concentrations in Butano Channel were high when depth increased with high tide and low when depth decreased at low tide (Figure 45). After sandbar closure, DO concentrations in Butano Channel (BCh) continued to fluctuate up and down with tidal activity (Figure 45).

Figure 45. Continuous DO and depth data from Butano Channel (BCh) three days prior to bar formation on 10/10/05 PM.

The site under the Highway One Bridge (HO) experienced hypoxia during major low tide events two weeks prior to sandbar closure (Figure 46) and complete bottom water anoxia in the week leading up to sandbar closure (Figure 47). On the days between 10/7/05 and 10/11/05, continuous depth data from Butano Channel (Figure 45) and Highway One (Figure 47) showed only one high and one low tidal event per day.

Figure 46. Continuous DO and depth data from Highway One site fifteen days prior to 10/10/05 PM sandbar closure.

Figure 47. Continuous DO and depth data from Highway One site from 10/4/05 through sandbar closure on 10/10/05.

Nutrient enrichment. Nutrient concentrations sampled in Butano Creek (Figure 48) and Pescadero Creek (Figure 49) just above the marsh plain showed seasonal patterns. Concentrations of dissolved inorganic nitrogen (DIN) coming into the system

appeared to fluctuate while soluble reactive phosphate (SRP) levels remained stable. In general DIN concentrations decreased from winter to spring and then began to increase through summer with the highest concentrations in the fall. Butano Creek especially had high levels of DIN during sandbar closure.

Figure 48. Butano Creek DIN and SRP (μM) +/- SE.

Figure 49. Pescadero Creek DIN and SRP (μM) $\pm$ SE.

If concentrations are converted into loadings, which take into account stream flow and express the DIN and SRP values as a mass per unit time (Figures 50 and 51), fall is no longer the time of greatest nutrient input, but is replaced by winter and spring when stream flows were highest. Loading rates indicate that Pescadero Creek was an equal contributor of nutrients to the marsh system.

Figure 50. Pescadero Creek DIN and SRP loadings (kg/hour) +/- SE.

Figure 51. Butano Creek DIN and SRP loadings (kg/hour) +/- SE.

DIN concentrations for Pescadero and Butano Creeks consisted primarily of the nitrogen species NO_3^- , with very small fractions of ammonia or nitrite (Figures 52 and 53). Although nitrite concentrations remain negligible within the marsh complex, ammonia fraction percentages vary with season and site (Figures 54 and 55). Marsh ammonia levels were found to be high in the summer and again in the fall during sandbar closure (Figure 54). North Pond (NP) is a source of ammonia in the summer months while Butano Channel (BCh) and Highway One (HO) are sources of ammonia during sandbar closure (Figure 55).

Figure 52. Mean nitrogen species distribution for Butano Creek.

Figure 53. Mean nitrogen species distribution for Pescadero Creek.

Figure 54. Ammonia temporal distribution $\pm$ SE in the marsh complex.

Figure 55. Box plot showing median, 25th, 75th, 90th, 10th percentiles, and outliers of ammonia concentrations at eight marsh sampling locations.

Nitrate levels in the marsh complex were elevated on 7/28/05 and low during sandbar closure (Figure 56). The box plot showing the distribution of nitrate levels at each of the eight marsh monitoring sites indicates low median levels in North Marsh (NM) and North Pond (NP) and elevated median levels in Butano Creek (BC1) (Figure 57).

Figure 56. Nitrate temporal distribution +/- SE in the marsh complex.

Figure 57. Box plot describing median, 25th, 75th, 90th, 10th percentiles, and outliers of nitrate concentrations at each of the eight marsh monitoring sites.

Soluble reactive phosphate (SRP) levels within the marsh complex became elevated during the summer months and in the fall during sandbar closure (Figure 58). North Pond (NP) had very high summer SRP concentrations, although the highest concentrations found in North Pond (NP) were during sandbar closed conditions (Figure 59). Highway One (HO), Pescadero Channel (PCh), and North Marsh (NM) sites also had high SRP concentrations during sandbar closure (Figure 60).

Figure 58. Temporal distribution of SRP +/- SE in the marsh complex.

Figure 59. Temporal distribution of North Pond SRP +/- SE.

Figure 60. Box plot describing median, 10th, 25th, 75th, 90th percentiles and outliers of SRP for eight marsh sampling locations.

Nitrogen to phosphate ratios (N:P) in the marsh complex were above the 16:1 Redfield ratio in the summer and fall (Figure 61). The N:P loading ratio from Pescadero and Butano Creeks was also above the 16:1 Redfield ratio in late summer and fall.

Figure 61. Marsh complex mean N:P +/- SE.

Figure 62. Mean tributary creek N:P loading +/- SE.

Macrophytic and microphytic blooms. Chlorophyll a concentrations within the marsh complex peaked during the spring and again in the fall during sandbar closure (Figure 63). Spring peaks were mainly associated with a large phytoplankton bloom in North Marsh (NM) while winter peaks occurred at 6 of the 8 sites (Figure 64).

Figure 63. Temporal distribution scatter plot of chlorophyll a +/- SE.

Figure 64. Spatial distribution box plot describing median, 10th, 25th, 75th, 90th percentiles and outliers of chlorophyll a concentrations.

Mean percent cover of submerged aquatic vegetation (SAV), *Potamogeton pectinatus*, remained generally constant throughout the spring and summer while percent cover of the macroalgae *Ulva spp.* increased slightly through the summer (Figure 65). Vegetation transects for *P. pectinatus* and *Ulva spp.* were not performed during sandbar closure but were found to be near zero after sandbar breach (Figure 65). Macrophytic coverage was dominated by *P. pectinatus* with Butano Channel, Butnao Creek 2, North Marsh, and North Pond dominating as the major producers (Figure 66).

Figure 65. Temporal distribution of % macrophytic cover +/- SE.

Figure 66. Macrophytic species spatial distribution bar chart.

Hydrogen sulfide build-up. Bottom water hydrogen sulfide levels were virtually non-existent until sandbar closure when concentrations became elevated (Figure 67). The site under the Highway One Bridge (HO) exhibited the highest concentrations of hydrogen sulfide within the marsh complex (Figure 68). High hydrogen sulfide concentrations were also found on the flooded marsh plain (Figure 69).

Figure 67. Hydrogen sulfide temporal distribution scatter plot +/- SE.

Figure 68. Hydrogen sulfide box plot describing median, 10th, 25th, 75th, 90th percentiles and outliers.

Figure 69. 11/14/05 Marsh plain hydrogen sulfide scatter plot +/- SE.

Secchi depth. After sandbar formation, mean secchi depth in the marsh complex was ≤ 1.0 m (Figure 70). The site at Highway One was the only site with a mean secchi depth > 1.0 m (Figure 71).

Figure 70. Secchi depth temporal distribution scatter plot +/- SE during sandbar closure.

Figure 71. Secchi depth spatial distribution scatter plot +/- SE.

Fine sediment and organic carbon characterization. The percentage of fine material ($< 62.5 \mu\text{m}$) in the top 2-cm of sediment declined but did not change significantly over time (Figure 72). Only one sample was taken during sandbar closure and it was sampled just three days prior to breaching on 12/2/05. Percentages of fine material were much higher before the breach than after (Figure 72). Butano Channel (BCh) and North Marsh (NM) had the highest levels of fine material in the surface sediments with means near 60% of the total (Figure 73).

Figure 72. Temporal distribution scatter plot $\pm$ SE of percent fines ($< 62.5 \mu\text{m}$) in the top 2-cm of sediment.

Figure 73. Spatial distribution bar chart $\pm$ SE of percent fines ($< 62.5 \mu\text{m}$) in the top 2-cm of sediment.

The percentage of organic carbon content in the surface sediments did not change during summer and appeared to only increase slightly during sandbar closure (Figure 74). North Marsh was the only site with significantly higher levels of organic carbon in the top 2-cm of sediment (Figure 75).

Figure 74. Temporal distribution scatter plot $\pm$ SE of percent organic carbon content in top 2-cm of sediment.

Figure 75. Spatial distribution bar chart $\pm$ SE of percent organic carbon content in the top 2-cm of sediment.

Significant spatial and temporal water quality results. Results from each of the four ANOVAs and the two planned *a priori* comparisons are listed below in Tables 9 and 10. Significant differences by site were found for SRP and wind (Table 9). North pond had the highest SRP levels among marsh sites (Figure 76a) while sites HO and BC2 had

the highest surface wind speeds (Figure 76b). Significant differences among season were found for DO and wind (Table 9). Dissolved oxygen levels were high in winter and spring, lower in summer, and, on average, much lower in fall (Figure 77a). Surface wind speed was highest in spring and lowest in fall (Figure 77b).

Table 9. Significant ANOVA results for water quality trends.

Significant effects	df	F-statistic	P-value
<u>Site</u>			
SRP	7	4.039	0.009
Wind	7	8.291	<0.001
<u>Season</u>			
DO	3	12.86	<0.001
Wind	3	14.638	< 0.001
<u>Bar status</u>			
Ammonia	1	25.73	< 0.001
Wind	1	13.249	0.002
<u>Site*Bar status</u>			
Chlorophyll a	7	3.845	0.011
Hydrogen sulfide	7	129.547	<0.001

Table 10. Significant results for planned *a priori* comparisons by site.

A priori Comparisons	Significant effects	df	F-statistic	P-value
BC1 vs. BC2, BC3, BC4, BC5, BC6, BC7, BC8, BC9, BC10, BC11, BC12, BC13, BC14, BC15, BC16, BC17, BC18, BC19, BC20, BC21, BC22, BC23, BC24, BC25, BC26, BC27, BC28, BC29, BC30, BC31, BC32, BC33, BC34, BC35, BC36, BC37, BC38, BC39, BC40, BC41, BC42, BC43, BC44, BC45, BC46, BC47, BC48, BC49, BC50, BC51, BC52, BC53, BC54, BC55, BC56, BC57, BC58, BC59, BC60, BC61, BC62, BC63, BC64, BC65, BC66, BC67, BC68, BC69, BC70, BC71, BC72, BC73, BC74, BC75, BC76, BC77, BC78, BC79, BC80, BC81, BC82, BC83, BC84, BC85, BC86, BC87, BC88, BC89, BC90, BC91, BC92, BC93, BC94, BC95, BC96, BC97, BC98, BC99, BC100	Hydrogen sulfide	1	25.228	< 0.001
BC1, BC2 vs. BC3, BC4, BC5, BC6, BC7, BC8, BC9, BC10, BC11, BC12, BC13, BC14, BC15, BC16, BC17, BC18, BC19, BC20, BC21, BC22, BC23, BC24, BC25, BC26, BC27, BC28, BC29, BC30, BC31, BC32, BC33, BC34, BC35, BC36, BC37, BC38, BC39, BC40, BC41, BC42, BC43, BC44, BC45, BC46, BC47, BC48, BC49, BC50, BC51, BC52, BC53, BC54, BC55, BC56, BC57, BC58, BC59, BC60, BC61, BC62, BC63, BC64, BC65, BC66, BC67, BC68, BC69, BC70, BC71, BC72, BC73, BC74, BC75, BC76, BC77, BC78, BC79, BC80, BC81, BC82, BC83, BC84, BC85, BC86, BC87, BC88, BC89, BC90, BC91, BC92, BC93, BC94, BC95, BC96, BC97, BC98, BC99, BC100	Soluble reactive phosphate	1	12.745	0.002
BC1, BC2 vs. BC3, BC4, BC5, BC6, BC7, BC8, BC9, BC10, BC11, BC12, BC13, BC14, BC15, BC16, BC17, BC18, BC19, BC20, BC21, BC22, BC23, BC24, BC25, BC26, BC27, BC28, BC29, BC30, BC31, BC32, BC33, BC34, BC35, BC36, BC37, BC38, BC39, BC40, BC41, BC42, BC43, BC44, BC45, BC46, BC47, BC48, BC49, BC50, BC51, BC52, BC53, BC54, BC55, BC56, BC57, BC58, BC59, BC60, BC61, BC62, BC63, BC64, BC65, BC66, BC67, BC68, BC69, BC70, BC71, BC72, BC73, BC74, BC75, BC76, BC77, BC78, BC79, BC80, BC81, BC82, BC83, BC84, BC85, BC86, BC87, BC88, BC89, BC90, BC91, BC92, BC93, BC94, BC95, BC96, BC97, BC98, BC99, BC100	Hydrogen sulfide	1	60.559	< 0.001

Figure 76. Spatial distribution bar chart $\pm$ SE of (a) SRP and (b) surface wind speed.

Figure 77. Seasonal distribution bar chart $\pm$ SE of (a) DO and (b) surface wind speed.

Significant differences for sandbar status were found for ammonia and surface wind speed (Table 9). Ammonia levels were significantly greater during sandbar closure (Figure 78a) while surface wind speeds were significantly greater when the sandbar was open (Figure 78b).

Figure 78. Sandbar status bar chart $\pm$ SE of (a) ammonia and (b) surface wind speed.

Significant two-way interactions between site and sandbar status were found for hydrogen sulfide and chlorophyll a (Table 9). The typical chlorophyll a pattern, with lower concentrations when the bar is open and greater concentrations when the sandbar is closed, was broken by the pattern seen in North Marsh (NM) where chlorophyll a levels were high all year round (Figure 79a). Hydrogen sulfide levels were typically low when the bar was open and high when the bar was closed, however, North Marsh (NM), Butano Creek 1 (BC1), and Pescadero Creek (PC) had low hydrogen sulfide levels all year long (Figure 79b).

Figure 79. Two-way interaction bar chart $\pm$ SE between site and sandbar status for (a) chlorophyll a and (b) hydrogen sulfide.

A priori comparisons between site BCh and all other sites and between BC1 and BC2 and all other sites found sites BCh, BC1, and BC2 all had significantly lower levels of hydrogen sulfide than site HO (Table 10) (Figure 80a). North Pond was found to have had significantly higher levels of SRP than site BCh (Table 10) (Figure 80b).

Figure 80. Spatial distribution bar chart $\pm$ SE for (a) hydrogen sulfide and (b) SRP.

Evidence of stratification

Salinity predicts density stratification during sandbar closure. During the 2004 and 2005 sandbar closure, the difference between surface and bottom salinity (ΔS) significantly predicted $\Delta \sigma_t$ (Δ density) ($R^2 = 0.996$) (Figure 81). Temperature differences between surface and bottom waters (ΔT) predicted much less of the variability seen between surface and bottom water density ($R^2 = 0.221$) (Figure 81).

Figure 81. Relationship between $\Delta \sigma\text{-}t$ (density) and ΔT and $\Delta \sigma\text{-}t$ and ΔS .

Salinity predicts dissolved oxygen during sandbar closure. Salinity significantly predicted DO concentrations ($R^2 = 0.387$, $P < 0.001$) during the 2005 sandbar closure (Table 11). Salinity and dissolved oxygen have a negative linear relationship (Figure 82), as salinity increases DO concentrations decrease. The greatest variability in salinity occurred at DO concentrations of 0 mg/L (Figure 82).

Table 11. Summary table of multiple, stepwise, backward regression model results for DO prediction during sandbar closure.

Model	Variable	B	SE	β	p-value
1	Salinity	-0.260	0.044	-0.653	0.000
	Nitrate	-0.386	0.538	-0.087	0.476
	Phosphate	-0.481	0.692	-0.073	0.490
	Chl. A	0.013	0.032	0.046	0.690
2	Salinity	-0.260	0.043	-0.654	0.000
	Nitrate	-0.475	0.487	-0.107	0.334
	Phosphate	-0.480	0.687	-0.073	0.487
3	Salinity	-0.262	0.043	-0.659	0.000
	Nitrate	-0.530	0.479	-0.119	0.273
4	Salinity	-0.248	0.041	-0.622	0.000

Figure 82. Negative relationship between salinity and dissolved oxygen during sandbar closure ($R^2 = 0.387$, $P < 0.001$).

Regional comparison

As part of the Comparative Lagoon Ecological Assessment Plan (CLEAP) in Santa Cruz, CA, 2nd Nature Incorporated estimated creek loadings for seven tributary creeks on the Santa Cruz County coast from May 2005 to October 2005 (2nd Nature Incorporated 2006). Creeks were sampled monthly for stream flow (ft³/s) and DIN and SRP concentrations. From these data, nutrient loadings (mass of a nutrient per unit time) were calculated (Appendix V).

A one-way factorial ANOVA analysis of creek loadings (kg/day) revealed Pescadero Creek had significantly higher loadings of SRP (df = 8, $P < 0.001$) (Figure 83a) and Butano Creek and Pescadero Creek both had significantly higher loadings of DIN (df = 8, $P < 0.001$) (Figure 83b).

Figure 83. SRP loading (a) and DIN loading (b) comparison between nine Central Coast creeks: Aptos Creek (AC), Valencia Creek (VC), Laguna Creek (LC), Scott Creek (SC), Branciforte Creek (BC), San Lorenzo Creek (SLC), Soquel Creek (SoC), Pescadero Creek (PC) and Butano Creek (BuC).

When loading calculations were normalized to take flooded marsh plain area into consideration (mass per unit time per unit area), Soquel Creek had significantly higher

levels of SRP ($df = 8$, $P < 0.001$) (Figure 84a). Aptos Creek and Laguna Creek had significantly lower DIN concentrations than San Lorenzo and Soquel Creeks ($df = 8$, $p < 0.001$) (Figure 84b).

Figure 84. Normalized loading rates for SRP (a) and DIN (b) for nine Central Coast creeks: Aptos Creek (AC), Valencia Creek (VC), Laguna Creek (LC), Scott Creek (SC), Branciforte Creek (BC), San Lorenzo Creek (SLC), Soquel Creek (SoC), Pescadero Creek (PC) and Butano Creek (BuC).

DISCUSSION

Causes of mortality

There is strong evidence to support hypoxia as the main cause of mortality of steelhead just downstream of the Butano Channel outlet. Out of the 97 readings taken from the two sondes in Butano Channel during those first 24 hours, 79% were less than 3.0 mg/L, the US EPA (1986) acute criteria for DO. In addition, surface waters in Butano Channel had less than 2.0 mg/L for more than 50% of the readings. These conditions are very close to the experimental conditions of Sheppard (1955) where brook trout mortality was found to occur after a ten hour exposure to DO concentrations of 1.5 mg/L at 10 °C.

There is also some evidence to suggest hypoxia may have been responsible for *C. magister* mortality in 2005. Excluding the two questionable bottom-water data sets from Butano Channel, bottom water DO concentrations were lowest in Butano Creek (BC2), downstream of the Butano Channel outlet, and at the confluence site (CONF), downstream of the merging of Pescadero and Butano Creeks. During the first 24 hours of lagoon draining, 65% of the bottom water readings in Butano Creek (BC2) were < 1 mg/L and 23% of the DO readings at site (CONF) were < 1 mg/L (Table 3). LC50 results for DO presented by Thursby et al. (2000) found mortality of blue crab (*Callinectes sapidus*), green crab (*Carcinus maenus*), and flat mud crab (*Eurypanopeus depressus*) to occur at < 1.0, < 0.54 and 0.57 mg/L DO respectively in tests ranging from one to four days in duration.

Prior to sandbar breaching, there were high and most likely toxic concentrations of hydrogen sulfide detected within the marsh complex. Hydrogen sulfide concentrations were greatest in the bottom waters of the lagoon (sites HO and LA1) and Butano Creek (BC2) on December 2, 2005, two days prior to the sandbar breaching event. The pH levels in Butano Creek (BC2) were low enough to suggest that possibly 90% of the hydrogen sulfide at that location was in the toxic, unionized form. The pH at the Highway One (HO) site was much higher and so it was estimated that only 15% of detected hydrogen sulfide was in the toxic, unionized form.

Despite the olfactory detection of hydrogen sulfide in the marsh complex, HACH HS-C field tests conducted on bottom and surface waters during the 2005 sandbar breach found no detectable concentrations of hydrogen sulfide. However, the minimum detection limit of the HACH test, 0.1 mg/L, is well above the US EPA (1976b) 0.002 mg/L acute criteria for hydrogen sulfide in aquatic environments. It is for this reason that hydrogen sulfide toxicity can be considered a possible contributor to fish and invertebrate mortality, particularly in the main embayment and lower arm of Butano Creek. More sensitive monitoring equipment would be needed for future investigations into the relative contribution of hydrogen sulfide toxicity to the fish kill.

Chemical analyses of bottom sediments by CDFG did not reveal the presence of any known agricultural associated toxins. The two sites where sediments were collected from had relatively sandy substrates and further investigations into sites with high silt/clay fractions is advised as known toxins, such as pyrethroids, are known to be associated with fine sediments (Werner personal communication).

There is little evidence in the literature to suggest that suspended sediments in the lagoon during sandbar breaching caused fish and invertebrate mortality. Although the data suggests Butano Channel is a source of fine particles, Servizi and Martens (1987) found that large particle sizes had greater effects on salmonid mortality. Anderson et al. (1996) also found that duration of exposure to suspended sediments had greater effects on mortality than did particle size. As the breaching event occurs in less than 12 hours and typical exposures of salmonids to suspended sediments is generally on the order of days in stream habitats, it is unlikely that sediment siltation is major contributor to the kill in Pescadero. Future investigations of the fish kill should focus on collecting total suspended solid (TSS) data for comparisons to US EPA criteria.

A situation similar to that at Pescadero Marsh occurs at the Surry River lagoon in South East Australia, which also experiences fish kills after the breaching of a sandbar (Becker personal communication). From carcass counts on exposed banks, researchers at Deakin University estimate over one million fish and invertebrates, including tens of thousands of the economically important bream (*Sparidae*), were killed during a draining event on July 11, 2005 (Becker personal communication). The Surry River is a long, ribbon-like estuary that floods to approximately 35 hectares (Becker personal communication) (Appendix VI) while Pescadero Marsh floods to cover an area that is triple that size, approximately 118 hectares.

Despite the immense size disparity between the two systems, the kill in Pescadero, based on stranded carcass counts, appears to be several orders of magnitude smaller. In addition, fish were observed gulping for air at the surface of the Surry River

lagoon during the entirety of the Surry River draining event (Becker personal communication), while this behavior was only observed in Butano Creek, just downstream of the Butano Channel inflow, during the first low tide after the 2004 breach in Pescadero.

The small number of carcasses found after the kill in Pescadero Marsh can be interpreted in several different ways: 1) the number of fish in the lagoon during a bar breaching could be small, 2) the majority of carcasses could have been washed out to sea, leaving little evidence of the numbers affected or 3) the kill could have been very small and geographically isolated in nature, creating a build up of carcasses in one specific location at one specific time.

Past and present studies in the Pescadero watershed do not suggest a major loss or catastrophic decline in the number of juvenile steelhead. Smith (1990) estimated the number of juvenile steelhead in the entire Pescadero lagoon complex to exceed 17,000 individuals. In this same report, Smith estimated that over 70% of the returning adults in Pescadero spent some time rearing in the lagoon. If a large portion of juvenile steelhead were being lost in the lagoon fish kill it would stand to reason that this would lead to a large decrease in the number of returning adults and therefore a drastic change in the number of steelhead fry and young-of-the-year rearing in the upper watershed. Although Department of Fish and Game staff have commented on the under-utilization of habitat by juvenile steelhead in the Pescadero watershed (Nelson personal communication), the population has not changed to such a degree as to suggest a major decline in the return of spawning adults.

Carcass survey and water quality data from this study suggest different modes of mortality are likely operating within different areas of the marsh. Steelhead carcasses found primarily between the Butano Channel outlet and the confluence of Pescadero and Butano Creeks are likely explained by the hypoxic surface water exiting Butano Channel. *C. magister* massmortality in the lagoon and lower creek arms in 2005 may be explained by hypoxia and elevated hydrogen sulfide concentrations in the main embayment before breaching. Observed topsmelt and jacksmelt behavior and mortality in Pescadero Creek during the 2004 draining event suggests the fish were trapped, perhaps between the freshwater of Pescadero Creek and the hypoxic water being pushed back up into the creek during high tide.

Future studies concerning the fish kill should focus on characterizing the size and distribution of lagoon and creek species during open and closed conditions. Understanding the size of the fish population in the lagoon prior to the kill would aid managers in understanding the overall ecosystem effects of the kill. Connecting species distribution with water quality and carcass distribution data would provide insight into the temporal and spatial nature of the kill. Lastly, fish sampling during sandbar closure will allow the detection of any population declines or pre-breach mortality that may be associated with bottom water anoxia.

Water quality, sediment, and vegetative evidence of eutrophication and stratification

During summer and fall, Pescadero lagoon exhibits many characteristics that have become familiar in eutrophied systems. Tidal-associated oxygen fluctuation, bottom water anoxia, macrophytic and planktonic blooms, low secchi depth, nutrient loading,

and fish and invertebrate kills are evidence of a eutrophic system (Cloern 2001; Breitburg 2002; Bricker et al. 1999; Diaz and Rosenberg 1995). Salinity stratification is another common characteristic of coastal systems, and although studies have found significant relationships between salinity and bottom-water DO, (Buzzelli et al. 2002; D'Avanzo and Kremer 1994; Welsh and Eller 1991), these studies do not attribute anoxic bottom waters solely to stratification.

In Pescadero, complete bottom water isolation by salinity stratification does not exist until after sandbar closure. By the time closure occurs in early fall, areas in Pescadero Marsh are already exhibiting many of the symptoms of eutrophication in the form macroalgal and SAV blooms, diel oxygen fluctuations, and increased levels of sediment organic carbon. Sandbar-associated stratification isolates this large reserve of macrophytic organic matter from light penetration and nighttime oxygen recharge. The result of this is a shift from a macrophytic-dominated system to a microphytic-dominated system. SAV and macroalgal species eventually senesce during this period and fuel the populations of anaerobic bacteria, the evidence of which is seen in the increase of bottom-water hydrogen sulfide concentrations.

A multiple-step regression analysis on the effects of salinity, wind, nitrate and phosphate on marsh DO during sandbar closure found that salinity alone explained 38% of oxygen variability. Nitrate and phosphate were chosen in this regression as a proxy for measuring site-specific nutrient enrichment. However, in a closed system, nitrate and phosphate are assimilated or re-mineralized by benthic and pelagic heterotrophs and may not accurately capture the eutrophic state of a particular site.

Common means of assessing the state of eutrophication in a system include: total organic carbon content in surface sediments, percent cover by macroalgal and SAV species, and chlorophyll a concentrations (Bricker et al. 1999). Butano Creek (BC2) and Butano Channel (BCh) were the areas with the highest mean percent cover of macrophytes, > 90 % and > 60% respectively. This compares with the > 90% macroalgal coverage rates found in Upper Newport Bay, a highly eutrophic Southern California estuary (Kamer et al. 2001).

Mean chlorophyll a concentrations were > 20 µg/L during the sandbar closure in Pescadero Marsh. Boynton et al. (1996) found severe eutrophication in the upper bays of coastal systems in Maryland with mean chlorophyll a concentrations > 20 µg/L and < 50 µg/L. In the eutrophic dead end canals of Delaware Bay where anoxic events have been known to occur, Maxted et al. (1997) found mean chlorophyll a concentrations to be 25.5 µg/L. Bricker et al. (1999) defines systems with > 20 µg/L and ≤ 60 µg/L chlorophyll a as highly eutrophic. Bricker et al. (1999) also listed secchi depths ≤ 1 m as an indication of high eutrophication. During sandbar closure, mean secchi depth in Pescadero was < 1 m.

Mean surface sediment organic carbon content from two sites, Butano Creek and Butano Channel, were approximately 1.5 % of total dried weight, while in North Pond it was approximately 3.5%. These numbers are comparable to those found in eutrophic systems around the world. In the top 2-cm of sediment, the percent of estuarine organic carbon has been documented to be 4.4% and 2.1% in Parker River, MA (Hopkinson Jr. et al. 1999), between 2.75% and 1.1% in Palmones River Estuary, Spain (Clavero et al.

2000) and between 4.1% and 0.97% in the River Thames estuary, UK (Trimmer et al. 2000). Like Pescadero, organic carbon percentages were also found to be greatest in areas furthest from tidal flushing (Trimmer et. al 2000) (Hopkinson Jr. et al. 2000).

Elevated organic carbon content in anaerobic sediments has been shown to lead to the release of ammonium and phosphate into the water column, enhancing primary production (Hopkinson Jr. et al. 1999; Clavero et al. 2000; Beck and Bruland 2000; Trimmer et al. 2000). Caffrey et al. found ammonia flux from sediments was greatest (2002), and nitrogen mineralization by sediments was lowest (2003), at poorly flushed agricultural sites in Elkhorn Slough, California. Sutula et al (2006) concluded that sediment-born nutrients contributed 20% of the 2002 TMDL (total maximum daily load) allocation for nitrogen and phosphorus in Upper Newport Bay Estuary, California.

Diel oxygen fluctuations within the marsh appear to be tidally controlled before sandbar closure, where suboxic (< 5 mg/L) bottom water conditions only improve with the high tide. After sandbar formation, bottom water anoxia is prevalent throughout the marsh. Summertime hypoxia has been well documented in coastal areas and, in recent decades, has been associated with anthropogenic eutrophication and density stratification (Nixon 1995; NRC 2000; Cloern 2001). Shallow (< 2 m), eutrophic coastal systems often exhibit seasonal hypoxia in the form of diel oxygen fluctuations that are generally controlled by daytime macro-algal and SAV photosynthesis and nighttime microbial respiration (Welsh and Eller 1991; D'Avanzo and Kremer 1994; Boynton et al. 1996). Deeper systems (> 2 m) are more often characterized by prolonged summertime bottom water hypoxia and/or anoxia associated with temperature or salinity stratification (Parker

and O'Reilly 1991; Buzzelli et al. 2002; Luther III et al. 2004). Bottom water hypoxia/anoxia in these deeper systems is generally the result of microbial consumption of vegetative material deposited from the photic zone (Rosenberg 1985).

Long-term bottom water anoxia during sandbar closure poses serious problems to aquatic communities. In an extensive review of the effects of hypoxia and anoxia on benthic communities, Diaz and Rosenberg (1995) chronicled diverse examples of structural shifts in benthic communities that suffer from short- and long-term hypoxia. Areas with prolonged seasonal hypoxia were shown to suffer almost complete elimination of macrofauna with a large proportion of secondary production shifting from epifaunal predators to microbes. Decreases in nitrification and denitrification rates in hypoxic sediments was shown to shift the balance of benthic-pelagic coupling by increasing nitrogen and phosphorus flux rates into the overlying water column. The associated increase in primary production and therefore organic matter deposition was found to create a positive feedback loop that perpetuated continued hypoxia.

This study found Pescadero Lagoon to exhibit many of the signs of a eutrophied system during sandbar open and sandbar closed conditions. Macroalgal and SAV blooms in Butano Creek, Butano Channel, and North Marsh suggest site-specific eutrophication, most likely associated with poor flushing. Elevated chlorophyll a concentrations and low mean secchi depths throughout the marsh complex during sandbar closure suggest nutrient enrichment, with sediments likely supplying a portion of the total nutrient budget. Stratification during sandbar closure exacerbates already poor bottom water

conditions by prohibiting oxygen renewal in bottom waters and causing mortality of SAV and macroalgal species.

Regional perspective on nutrient loading rates

Monitoring the loading rates of nutrients (particularly nitrate and phosphate) to a system is a common means of assessing the level of anthropogenic nutrient enrichment. Compared to the summer and fall tributary loading rates to lagoon systems in Santa Cruz County (2nd Nature 2006), Pescadero and Butano Creeks had greater mean loading rates of nitrogen (> 5 kg/day) and Pescadero Creek had greater mean phosphate loading rates (2 kg/day). However, when these loading rates are normalized for lagoon surface area, Pescadero and Butano Creek loading rates are no longer high due to the size of the flooded marsh plain in comparison with those of Santa Cruz County. Because Pescadero lagoon is the only system that experiences a fish kill upon sandbar breach it is necessary to take a closer look at the differences between these systems.

Soquel lagoon has a one-way gated culvert underneath the sandbar, allowing saline bottom water to slowly drain out of the lagoon as freshwater fills in, quickening the freshwater conversion process. The sandbar at San Lorenzo is periodically breached by unknown persons which temporarily re-establishes tidal flow and replenishes bottom waters and sediment with oxygenated water. These measures may be masking the effects of nutrient enrichment and prohibiting fish kill events as seen in Pescadero.

Lagoon size and summer/fall mean stream flow must also be considered when analyzing the effects of nutrient enrichment on central coast lagoons. For instance, Scott Creek and Soquel Creek lagoons are relatively small in area (< .03 km²) (2nd Nature

2005) while summer and fall 2005 creek flows were moderate ($>13 \text{ ft}^3/\text{s}$) (2nd Nature 2006). Pescadero Marsh is twice as large as any of the other lagoon complexes (1.2 km^2) with creek inflows slightly less than those of Scott and Soquel in the summer and fall 2005 ($12 \text{ ft}^3/\text{s}$) (2nd Nature 2006). San Lorenzo river, although less than half the size of Pescadero lagoon (0.16 km^2) (2nd Nature 2005), had more than twice the mean creek inflow in summer and fall ($45 \text{ ft}^3/\text{s}$) (2nd Nature 2006). Greater freshwater inflow contributes to the freshwater conversion, the subsequent breakup of salinity stratification, and therefore the improvement of bottom water quality of these lagoon systems (Smith 1990).

Hydrological modification to marsh areas is another possible explanation for differences among central coast systems. Although Pescadero, San Lorenzo, Soquel, and Aptos lagoons have all experienced large reductions in lagoon surface area since 1850 (2nd Nature 2004), Pescadero, because of its size, is the only marsh plain with significant remnant ditches and levees from past agriculture attempts on the marsh plain. These levees restrict tidal and freshwater flushing, creating a labyrinth of backwater areas with low dissolved oxygen. Anisfeld and Benoit (1995) found lowered alkalinity and elevated nutrient levels of sediments in ditches restricted from tidal flow in Long Island Sound. Caffrey et al. (2002, 2003) found that sediments, in poorly flushed areas of Elkhorn Slough estuary, California, had high pore water NH_4^+ , SRP and dissolved S^2 and suffered less efficient nitrogen removal by denitrification.

Finally, recent changes in the timing of sandbar formation at Pescadero Lagoon are unique among systems on the San Mateo and Santa Cruz coast (Appendix I) (Smith

1990; Swanson Hydrology and Geomorphology 2001, 2002, 2003; 2nd Nature 2004, 2005, 2006). In the last thirty years, the sandbar at the mouth Pescadero Lagoon typically formed in late spring or early summer, depending upon winter rains (Viollis 1979; Curry et al. 1985; Smith 2004). Freshwater conversion of the lagoon and creek arms, from watershed inflow, would ameliorate any water quality problems associated with stratification (Smith 1990). However, beginning in 1995 the sandbar began to close much later, sometimes not until early fall (Smith 2004). In 2005, all Santa Cruz and San Mateo lagoons had closed during the summer months while Pescadero did not close until early October (2nd Nature 2006). Late closure may not provide necessary time for the freshwater conversion documented by Smith (1990) in the second half of the 1980's. In addition, because the sandbar is open all summer, the lagoon, creeks, and channels remain shallow and warm, a perfect environment for SAV and macroalgal production.

CONCLUSIONS

Surface oxygen concentrations were below EPA minimum criteria for salmonids (3.0 mg/L) during the first and subsequent low tides after sandbar breach. Hydrogen sulfide and pH levels were such that several areas in the marsh were likely to have some toxic bottom waters prior to the sandbar breach. Suspended sediment data were insufficient to make a conclusion regarding their relative contribution to fish mortality. Ammonia concentrations were not above US EPA acute criteria and toxic pollutants were not detected in creek sediments.

The effects of stratification and eutrophication appear to be feeding back onto one another. Macrophytic growth proliferates in backwater areas with fine, organic-rich sediments during sandbar open conditions. Stratification during sandbar closure leads to the die-off of macrophytic species, the deposition of organic matter onto sediments and the creation of an anaerobic environment in bottom waters. Anaerobic sediments flux nutrients upward into the water column and work together with watershed-derived nutrients to fuel microphytic blooms.

The nutrient loading rates of Pescadero and Butano Creeks, when considered relative to flooded marsh surface area, do not exceed those of Santa Cruz County creeks. This suggests the eutrophication witnessed in particular areas in the marsh may be fueled in part by sediment-derived nutrients. The hydrological complexity, low stream flow and late sandbar closure set Pescadero Marsh apart from Santa Cruz County systems and no doubt compromise marsh function. Furthermore, Pescadero Lagoon has a bedrock sill at

the mouth that may limit scour elevation and decrease tidal flushing within the marsh, particularly during the summer and fall months when freshwater inflow is low.

MANAGEMENT RECOMMENDATIONS

Based on this study, managers, agency personnel and the community may do well to think of the Pescadero problem as one of chronic, bottom-water hypoxia/anoxia rather than as a singular fish kill event. Stratification and eutrophication are likely the result of natural and artificial hydrodynamic constraints, low stream flows, and possibly elevation limits at the mouth. Implementing measures that address only one of these constraints may alleviate the fish kill, but may fall short of promoting sustainable wetland function.

Pescadero Marsh is by far the most complex, intact, seasonally-closed lagoon system on the coast of San Mateo, Santa Cruz and Monterey Counties. It provides habitat for five federally threatened and endangered aquatic species, myriad rare and special status species, and is a very important stop on the Pacific flyway. Evidence from this study suggests the natural function of marsh sediment to remove nitrogen by way of nitrification and denitrification is being compromised. Managing for a single species based upon the evidence of one event has the potential to create successful niche environments, but may allow overall marsh function to continue to degrade. Complete, full-scale restoration is the only option if ecosystem-wide health is the ultimate goal.

The first step to restoration should be a complete analysis of the physical and biological changes that have occurred since the restoration actions of the mid 1990's. When pre-restoration data is available, post-restoration comparisons should be made for bird, fish, frog, snake, and aquatic invertebrate populations as well as physical changes to water quality, vegetation, and creek/channel/lagoon morphology. In combination with a

simple one-dimensional hydrodynamic model, this data can be used to plan future restoration actions.

Small-scale experiments are another way to analyze marsh function and develop restoration alternatives. Agencies can evaluate the biological impact of water quality and the effect of hypothetical restoration actions and use this information to inform management decisions. This also provides the public with a tangible product of agency cooperation.

Restoration should be a careful and contemplative process, using adaptive management techniques to analyze project alternatives. If it is to have any chance of prolonged success, there must be long-term buy-in from the agencies and the community involved. Otherwise, only small steps will continue to be made forward while the marsh ecosystem slowly becomes irretrievable. It is not an over exaggeration to suggest that the loss of wetland function in Pescadero would be an absolute tragedy for the Central Coast of California. If managers, agencies and the community can find a way to create mutually agreed-upon goals and objectives in Pescadero Marsh, there will be no better place to spend restoration and research dollars on the Central Coast.

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APPENDIX I

Fish kill and sandbar closure history

FISH KILL HISTORY						
SANDBAR FORMATION DATES AND OBSERVED FISH KILLS						
Year	Date bar formed	Bar Breach	Sandbar Citation	Fish Kill	Species Observed	Numbers
1985	Late April	Oct 31 & Nov 28 (artificial)	Smith 1990	NO		
1986	Mid July	Oct 31 (artificial)	Smith 1990	NO		
1987	Late March	Unknown	Smith 1990	NO		
1988	Early April & late May	Mid. Dec	Smith 1990	NO		
1989	May 28 & Mid Sept	> Oct	Smith 1990	NO		
1990	Late Sept.	After Jan 19	Smith 1991	NO		
1991	Unknown	Unknown	None	NO		
1992	Unknown	Unknown	None	NO		
1993	Dec (art. Open Aug.)	Jan	Smith 1997	NO		
1994	Mid May	Nov	Smith 1997	NO		
1995	Oct	6-Dec	Parks Report	YES	SH, SB, SF, C*	Hundreds

*SH = steelhead, SF = Starry flounder, C = Crab, SB = stickleback, SS = staghorn sculpin, S = jacksmelt or topsmelt, DC=dungeness crab

***SHG = Swanson Hydrology and Geomorphology

FISH KILL HISTORY						
SANDBAR FORMATION DATES AND OBSERVED FISH KILLS						
Year	Date bar formed	Bar Breach	Sandbar Citation	Fish Kill	Species Observed	Numbers
1996	Late Aug	Early Nov	Smith 1997	UNCONFIRMED		
1997	Unknown	Unknown	None	YES		
1998	Unknown	Unknown	None	UNCONFIRMED		
1999	No closure	No closure	N. Beck pers comm	UNCONFIRMED		
2000	Unknown	Unknown	None	UNCONFIRMED		
2001	Mid June	14-Nov	SHG 01 & 02	YES	SH* (young and adult)	>30
2002	18-Sep	2-Dec	SHG 02	YES	C, SF*	DC>5/m2, SF >10
2003	14-Sep	Night of Dec 1	Smith 2004	YES	SH* smolts	several hundred
2003					SH, C*	approx. 300
2004	30-Apr	15-May	Frahm pers comm	NONE KNOWN		
2004	5-Oct	Dec 5 (noon)	Frahm pers comm	YES	SH,C,SB,SF,SS,S	351
2005	8-Oct	4-Dec	Frahm pers comm	YES	SH, DC	50 SH & 474 DC

*SH = steelhead, SF = Starry flounder, C = Crab, SB = stickleback, SS = staghorn sculpin, S = jacksmelt or topsmelt, DC=dungeness crab

***SHG = Swanson Hydrology and Geomorphology

APPENDIX II

Equation of State (density equation)

(Reprinted with permission from the lecture notes of
Barry Klinger
Associate Professor
George Mason University)

Density of Seawater

Equation of State: $\rho = \rho(T, S, p)$

T = Temperature

units: °C

ocean range: -2°C to 30°C

Potential Temperature

if we raise water without changing heat content

==> pressure decreases

==> temperature decreases

convenient to use *potential temperature* θ

$\theta = T$ water would have at surface

$\theta < T$

difference pretty small....

$\theta \approx T - .5^\circ\text{C}$ for 5 km

BUT important effect for deep water

Accurate formula for θ can be found in Gill

My approx formula: $\theta = T - Ap - Bp^2$

$A = .04(1 + .185T + .035[S - 35])$

$B = .0075(1 - T/30)$

T in deg C, p in “dynamic km”

For $30 \leq S \leq 40$, $-2 \leq T \leq 30$, $p \leq 6$ km,

$\theta - T$ good to about 6%

(except for some shallow values w/ tiny $\theta - T$)

S = Salinity = mass of salt (gm) dissolved in 1 kg seawater

units: ppt or psu

kind of salt the same everywhere in sea

(ocean mixes up in time it takes to dissolve)

55% chlorine

31% sodium

8% sulphate

4% magnesium

1% potassium

S can go from 0 (coast) to about 40psu (Red Sea)

BUT 90% is between 34 and 35 psu

P = pressure = force/area

for ocean, use *hydrostatic* approximation:

pressure = weight of fluid above

MKS units: Newton/m² = Pascal

atmospheric weight $\approx 10^5$ N/m² $\equiv$ 1 bar

ocean weight $\approx$ 1 bar every 10 m depth of water

$\Rightarrow$ ocean depth in dbar $\approx$ depth in meters

ocean pressure measured relative to surface (ignore atm)

Rules About Density

- 1) Density hardly changes

$$\rho \approx 1000 \text{ kg/m}^3 = 1 \text{ gm/cm}^3$$

- 2) Density increases when pressure increases

z	ρ	$(S = 35 \text{ psu}, T = 0^\circ \text{C})$
0	1028.1	
1000	1032.8	
4000	1046.4	

- 3) Density increases when salinity increases

S	ρ	$(T = 0, z = 0)$
0	999.8	
35	1028.1	

- 4) Density decreases as T increases

- 5) BUT the *amount* it changes depends on T, p

colder water: *less* sensitive to T

deeper water: *more* sensitive to T

- 6) Variations due to T and S are small but **very important**

Accurate formula for $\rho(T, S, p)$ can be found in Gill appendix

My Approximate Density Formula:

$$\rho = C(p) + \beta(p)S - \alpha(T, p)T - \gamma(T, p)(35 - S)T$$

units: p in "km", S in psu, T in $^\circ \text{C}$

$$C = 999.83 + 5.053p - .048p^2$$

$$\beta = .808 - .0085p$$

$$\alpha = .0708(1 + .351p + .068(1 - .0683p)T)$$

$$\gamma = .003(1 - .059p - .012(1 - .064p)T)$$

For $30 \leq S \leq 40$, $-2 \leq T \leq 30$, $p \leq 6 \text{ km}$:

good to $.16 \text{ kg/m}^3$

For $0 \leq S \leq 40$, good to $.3 \text{ kg/m}^3$

cabelling

nonlinear T dependence, so mixing changes density

thermobaric effects

α depends on p , so

density ordering depends on depth

Potential Density

sigma-t: $\sigma_t = \rho(T, S, 0) - 1000$

potential density: $\rho(\theta, S, 0)$

$\sigma_\theta = \rho(\theta, S, 0) - 1000$

$\sigma_1 = \rho(\theta, S, 1\text{km}) - 1000$

APPENDIX III

**Department of Fish and Game
water and sediment pollution report**
(Reprinted with permission from CSP)

DEPARTMENT OF FISH AND GAME
FISH AND WILDLIFE
WATER POLLUTION CONTROL LABORATORY

2005 NIMBUS ROAD
RANCHO CORDOVA, CA 95670
PHONE (916) 358-2858 ATSS 8-434-2858 FAX (916) 985-4301

LABORATORY REPORT

Name: Rebecca Sloan
Agency: California State Parks
Address: 95 Kelly Ave
City: Half Moon Bay, CA 94019-1627

Lab Number: L-596-05
Other Number:
Date Sampled:
Date Received: 11/30/05
Date Completed: 01/22/06
Index-PCA Code:

RE: Pesticide Analysis

RESULTS OF CHEMICAL ANALYSIS:

Two water and two sediment samples were extracted and screened for pesticides, which include organochlorines, organophosphates, carbamates, herbicides and pyrethroids. No pesticides, herbicides or carbamates were found in water or sediment. Results of the analyses are attached.

High levels of sulfur were found in the water samples. **Sample #1 (P. Marsh) had 295 µg/L and Sample #2 (P. Marsh confluence) had 570 µg/L.** These samples were quantified and confirmed by GC/MS-ITD.

NA = Not Applicable
ND = Not Detected
MBLK = Method Blank
LCS = Laboratory Control Spike
LCSD = Laboratory Control Spike Duplicate

Cost of Analysis: \$ 2688.00

Pesticide Section Leader

01.23.2006
Date

Laboratory Director

1/23/2006
Date

California State Parks
L-596-05 Water

WPCL Lab#	Estimated MDL	Reporting Limit	L-596-05-1	L-596-05-2	QA/QC Samples	L-596-06-BLK	LCS
Sample Identification			P.MARSH	P.MARSH CONFLUENCE			
Date Received			30/Nov/2005	30/Nov/2005			
Date Extracted			05/Dec/2005	05/Dec/2005		05/Dec/2005	05/Dec/2005
Date Analyzed			20/Dec/2005	20/Dec/2005		20/Dec/2005	20/Dec/2005
Sample Type	water	water	water	water		water	water
Organochlorine Pesticides	ppb (ug/L)	ppb (ug/L)	ppb (ug/L)	ppb (ug/L)		ppb (ug/L)	% Recovery
Aldrin	0.001	0.002	ND	ND		ND	64.2
Chlordane, cis-	0.001	0.002	ND	ND		ND	83.5
Chlordane, trans-	0.001	0.002	ND	ND		ND	88.8
Chlordane, alpha-	0.001	0.002	ND	ND		ND	76.2
Chlordane, gamma-	0.001	0.002	ND	ND		ND	88.1
Dacthal	0.001	0.002	ND	ND		ND	98.3
DDD(o,p')	0.001	0.002	ND	ND		ND	88.7
DDD(p,p')	0.001	0.002	ND	ND		ND	102
DDE(o,p')	0.001	0.002	ND	ND		ND	79.0
DDE(p,p')	0.001	0.002	ND	ND		ND	103
DDMU(p,p')	0.001	0.002	ND	ND		ND	99.6
DDT(o,p')	0.001	0.002	ND	ND		ND	120
DDT(p,p')	0.002	0.005	ND	ND		ND	101
Dieldrin	0.001	0.002	ND	ND		ND	73.4
Endosulfan I	0.001	0.002	ND	ND		ND	88.5
Endosulfan II	0.001	0.002	ND	ND		ND	99.6
Endosulfan sulfate	0.001	0.002	ND	ND		ND	75.8
Endrin	0.001	0.002	ND	ND		ND	109
Endrin Aldehyde	0.002	0.005	ND	ND		ND	84.1
Endrin Ketone	0.002	0.005	ND	ND		ND	82.9
HCH, alpha	0.001	0.002	ND	ND		ND	88.1
HCH, beta	0.001	0.002	ND	ND		ND	84.3
HCH, gamma	0.001	0.002	ND	ND		ND	72.5
HCH, delta	0.001	0.002	ND	ND		ND	33.7
Heptachlor	0.001	0.002	ND	ND		ND	77.0
Heptachlor epoxide	0.001	0.002	ND	ND		ND	87.5
Hexachlorobenzene	0.0005	0.001	ND	ND		ND	59.9
Methoxychlor	0.001	0.002	ND	ND		ND	109
Mirex	0.001	0.002	ND	ND		ND	90.3
Nonachlor, cis-	0.001	0.002	ND	ND		ND	90.8
Nonachlor, trans-	0.001	0.002	ND	ND		ND	88.6
Oxadiazon	0.001	0.002	ND	ND		ND	112
Oxychlorodane	0.001	0.002	ND	ND		ND	79.9
Tedion	0.001	0.002	ND	ND		ND	110
Surrogate (% Recovery)							
Dibromooctatetrafluorobiphenyl	0.001	0.002	92.9	84.3		87.2	79.0

California State Parks
L-596-05 Water

WPCL Lab#	Estimated MDL	Reporting Limit	L-596-05-1	L-596-05-1		L-596-05-BLK	L-596-05-LCB	L-596-05-LCSO
Sample Ident/Reaction			P MARSH	P MARSH				
Date Received			30/Nov/2005	30/Nov/2005				
Date Extracted			06/Dec/2005	06/Dec/2005		06/Dec/2005	06/Dec/2005	06/Dec/2005
Date Analyzed			07/Dec/2005	07/Dec/2005		07/Dec/2005	07/Dec/2005	07/Dec/2005
Sample Type	water	water	water	water		water	water	water
Organophosphate Pesticides	ppb (ug/L)	ppb (ug/L)	ppb (ug/L)	ppb (ug/L)		ppb (ug/L)	% Recovery	% Recovery
Azinphos ethyl	0.030	0.050	ND	ND		ND	NA	NA
Azinphos ethyl	0.030	0.050	ND	ND		ND	NA	NA
Carbophenothion	0.030	0.050	ND	ND		ND	NA	NA
Chlorfenvinphos	0.030	0.050	ND	ND		ND	NA	NA
Chlorpyrifos	0.020	0.050	ND	ND		ND	83.9	88.4
Chlorpyrifos methyl	0.020	0.050	ND	ND		ND	NA	NA
Codrin	0.030	0.050	ND	ND		ND	NA	NA
Coumaphos	0.040	0.050	ND	ND		ND	NA	NA
Demeton-s	0.040	0.050	ND	ND		ND	NA	NA
Diazinon	0.005	0.020	ND	ND		ND	110	88.2
Malathion	0.030	0.050	ND	ND		ND	NA	NA
Dichlorofenathion	0.030	0.050	ND	ND		ND	NA	NA
Dichlorvos	0.030	0.050	ND	ND		ND	NA	NA
Dicofthos	0.030	0.050	ND	ND		ND	NA	NA
Dimethoate	0.030	0.030	ND	ND		ND	84.8	79.8
Dioxathion	0.030	0.030	ND	ND		ND	NA	NA
Disulfoton	0.010	0.050	ND	ND		ND	NA	NA
Ethion	0.020	0.050	ND	ND		ND	NA	NA
Fenphur	0.030	0.050	ND	ND		ND	NA	NA
Fenchlorphos	0.030	0.050	ND	ND		ND	NA	NA
Fenitrothion	0.030	0.050	ND	ND		ND	NA	NA
Fenathion	0.030	0.050	ND	ND		ND	NA	NA
Fenoxon	0.020	0.050	ND	ND		ND	NA	NA
Azinphos methyl	0.030	0.050	ND	ND		ND	NA	NA
Leptophos	0.030	0.050	ND	ND		ND	NA	NA
Malathion	0.030	0.050	ND	ND		ND	86.7	77.8
Methidathion	0.030	0.050	ND	ND		ND	72.8	87.2
Parathion, Methyl	0.010	0.050	ND	ND		ND	79.4	88.4
Molinate	0.020	0.050	ND	ND		ND	NA	NA
Parathion, Ethyl	0.030	0.050	ND	ND		ND	NA	NA
Phorate	0.050	0.100	ND	ND		ND	NA	NA
Mevinphos	0.030	0.050	ND	ND		ND	NA	NA
Phosmet	0.050	0.100	ND	ND		ND	73.7	88.2
Phosphamidon	0.030	0.050	ND	ND		ND	NA	NA
Ethion	0.030	0.050	ND	ND		ND	NA	NA
Sulfotep	0.030	0.050	ND	ND		ND	85.3	77.3
Boater	0.030	0.050	ND	ND		ND	NA	NA
Terbufos	0.030	0.050	ND	ND		ND	NA	NA
Tetrahydrovinphos	0.030	0.050	ND	ND		ND	NA	NA
Thiobenzoth	0.020	0.050	ND	ND		ND	NA	NA
Thionazin	0.040	0.050	ND	ND		ND	NA	NA
Toluthion	0.030	0.050	ND	ND		ND	NA	NA
Mepphos	0.030	0.050	ND	ND		ND	NA	NA
Trichloron	0.030	0.050	ND	ND		ND	NA	NA
Trichloronate	0.030	0.050	ND	ND		ND	NA	NA
Surrogate (% Recovery)								
Triphenyl phosphate	0.030	0.050	94.5	86.0		98.8	99.9	108

QA/QC Samples

California State Parks
L-596-05 Water

WPCL Lab#	Analytical Method	Estimated MDL	Reporting Limit	L-596-05-1	L-596-05-2	QADC Samples	L-596-05-BLK	L-596-05-LCS	L-596-05-LCSB
Sample Identification				P.MARSH	P.MARSH CONFLUENCE				
Date Received				30/Nov/2005	30/Nov/2005				
Date Extracted				07/Dec/2005	07/Dec/2005		07/Dec/2005	07/Dec/2005	07/Dec/2005
Date Analyzed				15/Dec/2005	15/Dec/2005		15/Dec/2005	15/Dec/2005	15/Dec/2005
Sample Type	water	water	water	water	water		water	water	water
Carbamate Pesticides		ppb (µg/L)	ppb (µg/L)	ppb (µg/L)	ppb (µg/L)		ppb (µg/L)	% Recovery	% Recovery
Aldicarb	EPA 632M	0.010	0.020	ND	ND		ND	72.3	88.4
Captaf	EPA 632M	0.050	0.100	ND	ND		ND	74.5	77.2
Carbaryl	EPA 632M	0.010	0.020	ND	ND		ND	85.1	79.2
Carbofuran	EPA 632M	0.010	0.020	ND	ND		ND	92.0	88.4
Diuron	EPA 632M	0.002	0.005	ND	ND		ND	93.5	81.4
Linuron	EPA 632M	0.002	0.005	ND	ND		ND	99.1	85.5
Methiocarb	EPA 632M	0.150	0.250	ND	ND		ND	88.7	88.8
Methomyl	EPA 632M	0.010	0.020	ND	ND		ND	102	98.8

California State Parks
L-596-05 water

WPCL Lab#	Analytical Method	Estimated MDL	Reporting Limit	L-596-05-1	L-596-05-2	QADCC Samples	L-596-05-BLK
Sample Identification				P.MARSH	P.MARSH CONFLUENCE		
Date Received				30/Nov/2005	30/Nov/2005		
Date Extracted				05/Dec/2005	05/Dec/2005		05/Dec/2005
Date Analyzed				20/Dec/2005	20/Dec/2005		20/Dec/2005
Sample Type				water	water		water
Herbicides		ppb (µg/L)	ppb (µg/L)	ppb (µg/L)	ppb (µg/L)		ppb (µg/L)
Atrazine	EPA 619	0.020	0.050	ND	ND		ND
Cyanazine	EPA 619	0.020	0.050	ND	ND		ND
Simazine	EPA 619	0.020	0.050	ND	ND		ND
Molinate	DFG-WPCL	0.020	0.050	ND	ND		ND
Thiobencarb	DFG-WPCL	0.020	0.050	ND	ND		ND
Trifluralin	DFG-WPCL	0.010	0.020	ND	ND		ND
Atraton	DFG-WPCL	0.010	0.020	ND	ND		ND
Prometon	DFG-WPCL	0.010	0.020	ND	ND		ND
Propazine	DFG-WPCL	0.010	0.020	ND	ND		ND
Terbutylazine	DFG-WPCL	0.020	0.050	ND	ND		ND
Sebumeton	DFG-WPCL	0.020	0.050	ND	ND		ND
Propanil	DFG-WPCL	0.020	0.050	ND	ND		ND
Alachlor	DFG-WPCL	0.010	0.020	ND	ND		ND
Simetryn	DFG-WPCL	0.020	0.050	ND	ND		ND
Ametryn	DFG-WPCL	0.020	0.050	ND	ND		ND
Prometryn	DFG-WPCL	0.020	0.050	ND	ND		ND
Terbutryn	DFG-WPCL	0.020	0.050	ND	ND		ND
Metolachlor	DFG-WPCL	0.010	0.020	ND	ND		ND
Pendimethalin	DFG-WPCL	0.020	0.050	ND	ND		ND
Oxyflufen	DFG-WPCL	0.020	0.050	ND	ND		ND
Norfurazon	DFG-WPCL	0.020	0.050	ND	ND		ND
Propagme	DFG-WPCL	0.020	0.050	ND	ND		ND

California State Parks
L-596-05 Water

WPCL Label	Analytical Method	Estimated MDL	Reporting Limit	L-596-05-1	L-596-05-2	QAVOC Samples	L-596-05-BLK	L-596-05-LCS	L-596-05-LC
Sample Identification				P.MARSH	P.MARSH CONFLUENCE				
Date Received				30/Nov/2005	30/Nov/2005				
Date Extracted				05/Dec/2005	05/Dec/2005		05/Dec/2005	05/Dec/2005	05/Dec/2005
Date Analyzed				20/Dec/2005	20/Dec/2005		20/Dec/2005	20/Dec/2005	20/Dec/2005
Sample Type		water	water	water	water		water	water	water
Pyrethroid Pesticides by GC/ECD		ppb (ug/L)	ppb (ug/L)	ppb (ug/L)	ppb (ug/L)		ppb (ug/L)	% Recovery	% Recovery
Bifenthrin	EPA 1660M	0.0005	0.001	ND	ND		ND	98.8	92.7
Cyfluthrin-1	EPA 1660M	0.002	0.004	ND	ND		ND	104	98.0
Cyfluthrin-2	EPA 1660M	0.002	0.004	ND	ND		ND	100	101
Cyfluthrin-3	EPA 1660M	0.002	0.004	ND	ND		ND	105	92.8
Cyfluthrin-4	EPA 1660M	0.002	0.004	ND	ND		ND	102	97.0
Cypermethrin-1	EPA 1660M	0.002	0.004	ND	ND		ND	98.8	98.0
Cypermethrin-2	EPA 1660M	0.002	0.004	ND	ND		ND	105	95.5
Cypermethrin-3	EPA 1660M	0.002	0.004	ND	ND		ND	101	92.0
Cypermethrin-4	EPA 1660M	0.002	0.004	ND	ND		ND	104	102
Deltamethrin	EPA 1660M	0.002	0.004	ND	ND		ND	99.9	105
Esfenvalerate/Fenvalerate-1	EPA 1660M	0.001	0.002	ND	ND		ND	110	99.9
Esfenvalerate/Fenvalerate-2	EPA 1660M	0.001	0.002	ND	ND		ND	104	91.6
Lambda-Cyhalothrin-1	EPA 1660M	0.0005	0.001	ND	ND		ND	96.5	98.3
Lambda-Cyhalothrin-2	EPA 1660M	0.0005	0.001	ND	ND		ND	96.9	98.7
Permethrin-1	EPA 1660M	0.003	0.005	ND	ND		ND	97.2	94.0
Permethrin-2	EPA 1660M	0.003	0.005	ND	ND		ND	102	100
Surrogate (% Recovery)									
Dibromooctatetrafluorobiphenyl	NA	NA	NA	92.8	94.3		97.2	91.3	87.8

APPENDIX IV

Department of Fish and Game
aquatic toxicology report
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STATE OF CALIFORNIA
DEPARTMENT OF FISH AND GAME

AQUATIC TOXICOLOGY LABORATORY REPORT

9300 Elk Grove-Florida Road
Elk Grove, CA 95624

Lab No: P-2425
E.P. No: _____
Index: PCA:

Report Date: 12/08/2005
ATL: S120105-1

To: Joanne Kerbavaz
Address: California State Parks
95 Kelly Avenue
Half Moon Bay, CA 94019

Remarks

Annual fish kills have occurred at the Pescadero Creek/Marsh area coinciding to the break of the sandbar separating the marsh from the ocean allowing a mixing of salt and freshwater as well as a mobilization of sediments contained within the marsh. Sediment and water samples were collected to attempt to determine the cause of fish mortality. Sediment and water were collected by California State Parks and Recreation Department Personnel on November 29, and delivered to the Aquatic Toxicology Laboratory on November 30, 2005. Toxicity was determined by Department of Fish and Game's Aquatic Toxicology Laboratory staff. A static definitive acute sediment toxicity test (96-h) was performed on sediment elutriate using rainbow trout (*Oncorhynchus mykiss*) approximately 8 months old (7-10 cms fork length) following procedures described in Environmental Toxicology and Chemistry (19:2 pp 485-494) and EPA/600/4-90/027F, with the following changes: age and size of trout were larger as described above; sediment separation by centrifugation was done at 4000 rpm and waters were oxygenated by bubbling at approximately 50 to 80 bubbles/minute.

RESULTS OF EXAMINATION

Conclusion

S120105-1 (T421-05, Pescadero Creek Sediment Sample)
The 4:1 (water : sediment) elutriate was not toxic to rainbow trout.

Pesticide Investigations Laboratory Report P-2425

Table 1: Mean characteristics of water samples and survival of *Oncorhynchus mykiss* in 96-h definitive acute test.

ATL Sample Number	Percent of Sample Control	Specific Cond ($\mu\text{mho/cm}$)	Temp $^{\circ}\text{C}$	pH	D.O. (mg/L)	Alkalinity (mg/L)	Hardness (mg/L)	Percent Survival %
ATL	Control	343	12.7	7.8	9.88	106.5	118.5	100
Receiving Water	Control	4475	12.6	8.0	10.1	231	612	100
T421-05	6.25	10180	12.5	7.8	10.1	304	456.2	100

APPENDIX V

Nutrient loading calculations for Pescadero and Santa Cruz County streams

SITE	mean summer/fall ft ³ /s	Closed lagoon area (km ²)	PO ₄	+/- SE	N	DIN	+/- SE	N
Aptos: Aptos Creek	3.75	0.01	1.82363	0.11	9	1.167927	0.09	9
Aptos: Valencia Creek	1.20	0.01	1.01074	0.15	6	8.432065	0.40	6
Butano Creek	6.08	1.20	0.27825	0.04	14	6.18981	0.14	14
Laguna Creek	1.19	0.03	0.1349	0.05	8	1.01779	0.15	8
Pescadero Creek	12.16	1.20	1.68123	0.08	14	4.905998	0.13	14
San Lorenzo: Branciforte Creek	8.25	0.16	0.14826	0.06	7	1.992634	0.22	7
San Lorenzo: San Lorenzo River	44.88	0.16	1.40538	0.13	8	11.49611	0.31	8
Scott Creek	16.64	0.03	1.49284	0.18	6	6.934122	0.39	6
Soquel Creek	13.41	0.01	5.56839	0.27	8	10.85251	0.46	8

APPENDIX VI

Surry River, Australia aerial photograph

(Photograph reproduced with permission from
Alistair Becker BSc. (Hons)
Deakin University
Victoria, Australia)

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☆ ● Rebecca Sloan show details Jan 15 (1 day ago) Reply

Hello Genan,

I hope you had a nice holiday. I am following up to a phone message I left with you on Friday evening. I am finishing up my thesis and am seeking permission to reproduce several GIS maps, a picture from a report and a map from a ranger safety report.

All GIS map were created by me or under my management using the State Parks Aerial layer on citrix, with the exception of two maps. These two maps were created by the ESI prior to me for the purposes of internal use.

The photo that I have scanned and am seeking permission to reproduce is of the bridge over Pescadero Marsh Natural Preserve. It was submitted as part of a 1985 report from a UC professor. The document was paid for by State Parks through a contract. I am also seeking permission from the author of the report, although technically I do not think it is required, as the report is Park's.

The map is a hand drawn product illustrating the location of fish carcasses found after a 1995 fish kill event. It was submitted as part of a safety report. I have talked with Ranger Paul Olson in HMB and he has stated that use of a safety report is fine.

I have shown all documents intended to be used with Joanne Kerbavaz, the senior ecologist in our office, and she has given her permission to use them. But of course, we need to seek permission from Sacramento.

The format I used to give credit to state Parks is as follows:

Unpublished map reproduced with permission of CSP. For the photo (photo from Curry et al. 1985; reprinted with permission of CSP). For the ranger map (Augustine 1995 CSP safety report; reprinted with permission from CSP)

I hope this works for you as the document is already at the printer. Please call me ASAP if there is a problem. I only have until 5:00 pm today to correct. Sorry about the late notice, it is my fault for not truly understanding the nature of copyright protection. Thank you so much and have a good day.

--
 Rebecca Sloan
 650.464.0215
 Fax 650.726.8816

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No worries on our end, Rebecca. The credit lines are fine and we can do the documentation. But I am copying Joanne with this email because copyright license agreements do not go through our office unless they're for material that we have in our possession (we would be horribly bogged down in work if they had to). I am only required to get involved if someone wants to use the California State Parks logo. I'm attaching the form that Joanne should use to give you permission to use these items. I filled it out with the information you gave me and made bold the more specific information that still needs to be filled in. So now it just needs to be verified that the information is correct, signed by you, and signed by a supervisor in the district/sector. Then it should be kept on file in the sector office (ideally with copies of it filed with any files they have on the information that was licensed to you). I hope this helps.

Jenan

From: Rebecca Sloan [mailto:rsloan.parks@gmail.com]**Sent:** Monday, January 15, 2007 11:54 AM**To:** Saunders, Jenan**Subject:** PERMISSION TO REPRODUCE

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PERMISSION TO USE Pescadero photo

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rsloan.parks@gmail.com

I am a graduate student at SJSU and am finishing up my masters thesis on an annual fish kill event at Pescadero Marsh. In my thesis I would like to include a photo of the old Highway One Bridge in your 1985 Pescadero Marsh Management report to California State Parks. Written permission given in this e-mail is all that is required. I am also soliciting permission from State Parks for use.

Thank you so much and have great day.

Rebecca Sloan
650.464.0215
Fax 650.726.8816

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Rebecca Sloan <rsloan.parks@gmail.com>**Pescadero Watershed GIS layer**

6 messages

Rebecca Sloan <rsloan.parks@gmail.com>

Thu, Mar 30, 2006 at 11:23 AM

To: info@mbnmsf.org, "Sicular, Dan" <dsicular@esassoc.com>

Hello Dennis,

I am contacting you seeking permission to share GIS layers that the consulting firm ESA produced for a project in the Pescadero watershed. I am an intern at California State Parks interested in putting together a land use map for the Pescadero Watershed. ESA created their own watershed layer that we were hoping to retain for continuity sake. We are also interested in the salmonid habitat layers that ESA used. Dan Sicular from ESA is also a recipient of this e-mail so a reply from you stating yes or no will suffice. Thanks for your time.

Have a good day.

--

Rebecca Sloan
650.464.0215
Fax 650.726.8816

Sicular, Dan <DSicular@esassoc.com>

Thu, Mar 30, 2006 at 11:41 AM

To: Rebecca Sloan <rsloan.parks@gmail.com>

Rebecca--

here is the e-mail address I have for Dennis:

lighthousegroup@earthlink.net

-Dan

Dan Sicular
Senior Project Manager
Environmental Science Associates
436 14th Street, Suite 600
Oakland, CA 94612
510-839-5066 fax 510-839-5825
Direct: 510-740-1724
dsicular@esassoc.com

Hi Rebecca,
We approve of the request.
Please send us a copy of the land use map, or a link to it, once you have it completed.
Best regards,
Dennis Long

----- Original Message -----

From: Rebecca Sloan
To: info@mbnmsf.org; Sicular, Dan
Sent: 3/30/2006 11:23:54 AM
Subject: Pescadero Watershed GIS layer

Hello Dennis,

I am contacting you seeking permission to share GIS layers that the consulting firm ESA produced for a project in the Pescadero watershed. I am an intern at California State Parks interested in putting together a land use map for the Pescadero Watershed. ESA created their own watershed layer that we were hoping to retain for continuity sake. We are also interested in the salmonid habitat layers that ESA used. Dan Sicular from ESA is also a recipient of this e-mail so a reply from you stating yes or no will suffice. Thanks for your time.

Have a good day.

Rebecca Sloan
650.464.0215
Fax 650.726.8816

Sicular, Dan <DSicular@esassoc.com>
To: ighthousegroup@earthlink.net
Cc: Rebecca Sloan <rsloan.parks@gmail.com>

Fri, Mar 31, 2006 at 12:25 PM

Thanks for the word, Dennis. Hope you are doing well.

-Dan

Dan Sicular
Senior Project Manager
Environmental Science Associates
436 14th Street, Suite 600
Oakland, CA 94612
510-839-5066 fax 510-839-5825
Direct: 510-740-1724
dsicular@esassoc.com

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Rebecca Sloan <rebecca.sloan@gmail.com>

Pescadero Map of 1854

2 messages

Bill Cook <Bill@teamcook.net>

Fri, Dec 15, 2006 at 2:31 PM

To: Rebecca Sloan <rebecca.sloan@gmail.com>

Hello Rebecca,

You have my permission to use the Pescadero Marsh Map of 1854, that I used in my booklet *The Restoration of Butano Creek, Its Lower Channel and Floodplains & The Flooding of Pescadero Road*. I found the map in a report from California State Parks and Recreation, and enhanced it with color, captions and a legend. Attached is a digital version of the map so that you will be able to print a clearer copy.

Congratulations on getting your thesis published! I would like to see a copy if you wouldn't mind sending me one.

It's a good life!

Bill Cook
RE/MAX Dolphin Real Estate
790 Main Street
Half Moon Bay, CA 94019
650-879-0824 office
650-879-9202 fax
Bill@TeamCook.net

Oh, By The Way... if you or your friends and neighbors are interested in buying or selling a home, please let us know and we will be happy to follow up and take good care of them!

Pescadero Marsh Map of 1854.doc
3426K

Rebecca Sloan <rebecca.sloan@gmail.com>
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