NUTRIENT AND CONTAMINANT ANALYSIS REPORT

Nutrient and Contaminant Analysis Report

Submitted to:
California State Coastal Conservancy
U.S. Fish & Wildlife Service
California Department of Fish and Game

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ACRONYMS

AMP	Adaptive Management Plan
Conservancy	California Coastal Conservancy
Delta	San Joaquin River Delta
DFG	California Department of Fish and Game
DO	dissolved oxygen
EIS/R	Environmental Impacts Report / Environmental Impacts Statement
ER-M	Effective Range-Median
ESR	Environmental Setting Report
IHg	inorganic mercury
ISP	Initial Stewardship Plan
MeHg	methylmercury
MTM	Mercury Technical Memorandum
NCAR	Nutrient and Contaminant Analysis Report
PHg	particulate mercury
SBSP	South Bay Salt Ponds
SCE	shallow coastal ecosystem
SFRWQCB	San Francisco Bay Regional Water Quality Control Board
SOD	sediment oxygen demands
THg	total mercury
TMDL	total maximum daily load
TSS	total suspended solids
USFWS	United States Fish and Wildlife Service
WQO	water quality objective

1. INTRODUCTION AND OVERVIEW

This Nutrient and Contaminant Analysis Report (NCAR) presents an analysis of the processes affecting the potential for environmental impacts due to low dissolved oxygen (DO) and mercury bioaccumulation that could result from implementation of the South Bay Salt Ponds (SBSP) Restoration Project. It is a companion to the SBSP Environmental Impacts Report / Environmental Impacts Statement (EIS/R) that provides additional technical detail supporting the findings of the EIS/R. Because the NCAR is intended to stand alone as a report, some brief project background is presented, followed by a summary of the adaptive management approach for the project.

The NCAR explores certainties and uncertainties related to nutrient and contaminant cycling through empirical analyses and the refinement and application of existing conceptual models. Water and sediment quality within the project boundaries and the surrounding areas was evaluated and summarized in an Environmental Setting Report (ESR) and used in this analysis. Existing loading and constituent flux models were reviewed as part of this analysis to predict changes in water and sediment quality that could result from the project. While detailed numeric analysis of future water quality conditions is not feasible, the conceptual analysis helps estimate the range of water and sediment quality conditions that could be expected as a result of the implementation of restoration alternatives and the No Action alternative.

The report is based on existing data and the current state of knowledge, subject to the constraints of available scope. It should be used as a reference source that provides details that were considered in the assessment of environmental impacts due to the SBSP restoration program. It summarizes the thinking used to evaluate impacts for complex, staircase issues in order to highlight where the key uncertainties are. Therefore, the report does not attempt to resolve all uncertainties, but rather described the critical uncertainties so that work to resolve them can be scoped and funded. Some of the most important findings of the report are about what we need to know, rather than what we know.

1.1 South Bay Salt Ponds Restoration Project Background

The State of California and the Federal government have embarked on the restoration of 15,100 acres of Cargill’s former salt ponds in South San Francisco Bay (South Bay Salt Pond Restoration Project website 2006). Acquisition of the South Bay salt ponds provides an opportunity for landscape-level wetlands restoration, improving the physical, chemical, and biological health of the San Francisco Bay. The historic loss of approximately 85 to 90 percent of the tidal marsh in the San Francisco Bay has led to dramatic losses of fish and wildlife in tidal marsh habitat, decreased water quality and increased turbidity in the Bay. As the Estuary shrank, there were also changes to physical processes, increasing the need for dredging and the hazards of flooding.

The South Bay Salt Pond Restoration Project is integrating restoration with flood management, while also providing for public access, wildlife-oriented recreation, and education opportunities. The Project is restoring and enhancing a mosaic of wetlands, creating a vibrant ecosystem. Restored tidal marshes will provide critical habitat for the endangered California clapper rail and the salt marsh harvest mouse. Large marsh areas with extensive channel systems will also provide habitat for fish and other aquatic life and haul out areas for harbor seals. Many of the ponds will remain as managed ponds and be enhanced to maximize their use as feeding and resting habitat for migratory shorebirds and waterfowl traveling on the Pacific Flyway.

The South Bay Salt Pond Restoration Project provides the opportunity to improve the physical, chemical and biological health of the San Francisco Bay. The California Coastal Conservancy (Conservancy) will work closely with the United States Fish and Wildlife Service (USFWS) and the California Department of Fish and Game (DFG) to meet the goals of the project:

- Restore and enhance a mix of wetland habitats,
- Provide for flood management, and
- Provide wildlife-oriented public access and recreation opportunities.

To achieve these goals, former salt ponds within the project area will be converted to either managed ponds or tidal wetlands. The ratio of ponds to wetlands defines two bookends of the final project outcome. The first bookend, referred to as "Alternative B" in the planning process, would result in an approximately 50:50 ratio of tidal wetlands to managed ponds, which is very close to the current configuration of the project area. The second bookend, known as "Alternative C," would result in an approximately 90:10 wetland to pond ratio. The 50 year project will proceed in phases, with each phase moving another step from bookend "B" to bookend "C."

1.2 Uncertainties, Adaptive Management, and Staircase Issues

A key uncertainty is the sustainability of the managed pond habitat. By proceeding in phases, impacts can be avoided by evaluating sustainability as each phase is completed. If the higher-density managed ponds developed in the initial project phases are shown to be sustainable, then subsequent phases can proceed further from bookend "B" (pond emphasis) to bookend "C" (tidal wetland emphasis). If monitoring shows that high-density managed ponds are not sustainable, despite all available adaptive management actions, then the final project configuration would be closer to bookend "B," and subsequent phases may even need to reverse the actions of initial phases to avoid environmental impacts. This approach is the foundation of the determination that environmental impacts of the 50-year project will be avoided through adaptive management.

The EIS/R discussion at the programmatic level addresses many uncertainties about the sustainability of managed pond habitat. For example, some key questions include whether higher densities of birds would suffer higher mortality due to predation or disease, whether pond conversions lead to invasions of nuisance species, and whether sediment accretion within pond areas causes problems due to sediment deficits in adjacent sloughs (Science Team for the South Bay Salt Pond Restoration Project 2005). This report explains the technical basis of two key uncertainties related to water and sediment quality:

1) Can ponds be managed to avoid low DO that harms aquatic and benthic organisms (Section 2)?

2) Will project activities lead to harmful effects from the mobilization, transformation and bioaccumulation of mercury (Section 3)?

These uncertainties result from previous experience of the Initial Stewardship Plan (ISP) and knowledge of the regional setting. During the ISP, severe depressions of DO resulted in fish kills within a managed pond in the Alviso Area (U.S. Fish and Wildlife Service 2005b). The DO sags were associated with blooms of phytoplankton, which can be stimulated by increased nutrient and/or light availability.

The project area is also known to be significantly affected by mercury discharges from historic mining activities in the watershed. Mercury can be a problem for wildlife and people that eat fish if it is converted to methylmercury, the form which most readily bioaccumulates in the food web. Since wetland areas are known to be at risk for increased methylmercury production, this was identified as an important uncertainty that needs to be addressed through adaptive management. Low DO is also known to be a risk factor for increased methylmercury production, so the two issues are linked.

Technical uncertainties that will be adaptively managed have been defined as "staircase issues." At the top of the staircase is a pre-defined threshold for environmental impacts. The purpose of adaptive management is to avoid crossing the threshold. To achieve this, triggers for adaptive management, or steps along the staircase, will be defined that are well below the threshold for impacts. Exceedance of a trigger would lead to adaptive management actions that avoid or reverse additional progress up the staircase.

The thresholds for significant impacts are defined in the SBSP EIS/R. For water discharges from the project area, the threshold of significance is to avoid causing DO less than 5 mg/L in the Bay, which is established by the regional water quality regulations[1]. Within the project area, where lower DO levels are expected to occur more commonly, the threshold of significance is to avoid low DO associated with one or more of the following negative impacts:

1) Mortality of aquatic or benthic organisms;
2) Odors that cause nuisance;
3) Degraded habitat; or
4) Unacceptably high net methylmercury production rates.

For the purposes of this evaluation, the lower threshold for the project area can be approximated by a DO level of 2 mg/L.

For mercury, the key threshold of significant impacts is to avoid causing or contributing to mercury levels exceeding 0.2 ppm in large fish and 0.03 ppm in small fish, both in the project area and in the Bay. This threshold is driven by the recently adopted total maximum daily load (TMDL) plan for mercury in San

[1] For both mercury and dissolved oxygen, there is some existing uncertainty about the applicable water quality objectives based on current regulations. These uncertainties should be resolved through review of this report and the EIS/R with the SFRWQCB.

Francisco Bay (San Francisco Bay Regional Water Quality Control Board 2006). The Bay mercury TMDL also requires that activities avoid release of sediments into the bay that have a median mercury concentration greater than 0.2 ppm, and that existing water quality objectives (0.025 – 0.050 μg/L) for mercury be attained. These requirements set additional mercury thresholds, but the most sensitive threshold is considered to be the fish tissue concentrations, because fish tissue mercury concentrations are directly related to beneficial uses such as fishing and wildlife habitat.

Triggers to avoid these thresholds will be defined by the Science Team through the development of the Adaptive Management Plan (AMP). The purpose of this report is to develop a conceptual framework that defines the key processes that would move the project up the staircase towards exceedance of thresholds. This sets the stage for development of specific triggers and associated management actions through subsequent project planning phases.

2. PHYTOPLANKTON ABUNDANCE AND DISSOLVED OXYGEN

2.1 Conceptual Overview

2.1.1 Problem Statement

Phytoplankton are microscopic algae that live in the water column. There are 5,000 species of algae that are unicellular and diverse in cell size, morphology, physiology, and biochemical composition (Riley 1967). Phytoplankton population dynamics can change in response to alterations in the processes regulating the biomass quantity, species composition, and the spatial distribution of the phytoplankton population. These responses include localized changes in phytoplankton growth and regional changes as a result of horizontal transport (Cloern 1996). For this reason, restoration activities in the project region may influence phytoplankton population dynamics in the regional setting. Changes in phytoplankton populations can directly affect water quality factors, most notably DO.

During the ISP, low DO in discharges from managed ponds to the Bay were attributed to growth, accumulation and subsequent decay of phytoplankton. The DO sags appear to be localized to specific areas of the ponds. Deep areas of the ponds that can accumulate dead phytoplankton and inhibit re-aeration appear to be particularly problematic. Areas that accumulate wind-blown phytoplankton and other detritus also are at risk. Adaptive management responses focused on diverting low-DO water found within the ponds away from discharge points. This was initially done by simply closing tide gates. While this action protected the Bay, closing tide gates may have contributed to the fish mortalities observed in Alviso Pond A16. An alternative action to prevent discharge of low DO water to the Bay is the installation of baffles to redirect flow from low-DO areas of the ponds (U.S. Fish and Wildlife Service 2005b). Installation of “Solar Bees” (solar powered re-aeration devices) is another adaptive management action proposed by the managers of the wildlife refuge to protect the Bay while concurrently avoiding low DO within ponds (U.S. Fish and Wildlife Service 2005b).

The sequence of events and actions taken to address low DO conditions during the ISP follows the paradigm for adaptive management. A problem is observed, action is taken to correct the problem, the positive and negative responses are evaluated, and the subsequent actions are improved based on observed responses. What is needed is an organized framework for explaining observations and responses. It is unlikely that modeling can precisely forecast responses to actions, but the conceptual model developed through this analysis can help guide adaptive management strategies by highlighting the most important factors.

Presentation of the conceptual model begins with a discussion of factors affecting phytoplankton growth. Then, details are provided on the linkage between phytoplankton growth and low DO. The environmental setting as it affects phytoplankton blooms and DO is discussed next, followed by a summary of how restoration actions are predicted to affect phytoplankton blooms and DO.

A simple numeric model of DO was developed for Pond A16 in the Alviso Area. Pond A16 was selected because it will be reconfigured in Phase 1, and because it had DO problems during the ISP. The numeric model is used to evaluate the relative importance of source water quality, pond residence time, sediment oxygen demand, and re-aeration rates on DO concentrations.

With any simple model, one of the most important questions to ask is "where does the model fail?" The model evaluation concludes with a comparison of model predictions to real world observations. This provides a basis for recommendations on future monitoring needs, improved modeling, and adaptive management strategies.

2.1.2 Factors Affecting Phytoplankton Growth in Shallow Coastal Ecosystems

Phytoplankton growth is influenced by many physical and chemical factors in the aquatic environment. Nutrient availability is one factor that is known to stimulate phytoplankton growth in freshwater and deep ocean ecosystems. Nutrients such as nitrogen and phosphorous are critical components of aquatic ecosystems, fueling primary production at the base of the food chain, which in turn fuels higher trophic level production (Tchobanoglous 1985).

While nutrient loading is a critical factor for phytoplankton growth in lakes and deep oceans, shallow coastal ecosystems (SCEs) including estuaries and tidal rivers such as San Francisco Bay and the Bay Delta are different. SCE waters are typically high in nutrient concentrations due to land sources (including surface runoff and WWTP effluent) and geochemical and biological processes that act as "filters" to retain and recycle nutrients within estuaries (Sharp 1984). Instead, light availability and grazing rates are often found to be limiting factors for phytoplankton growth in SCEs (Cloern 2001).

Besides nutrients, phytoplankton need light to photosynthesize and grow. Attenuation of irradiance in the water column is a function of depth and turbidity. Suspended particles absorb and scatter light, limiting the light energy available for phytoplankton to photosynthesize. Turbidities in SCEs are typically much greater than in the open ocean due to riverine sediment inputs and resuspension from tidal and wind mixing. Shallow depths also promote a strong coupling between the pelagic and benthic environments, meaning that filter feeders living on the bottom have a relatively greater grazing impact on drifting phytoplankton (Cloern 1996). Tidal and wind-driven mixing limits phytoplankton growth by bringing phytoplankton down to deeper areas where light is limited and predation higher (Cloern 1991; Koseff and others 1993). This is part of the reason why the onset of the phytoplankton bloom in the South Bay is triggered by seasonal stratification of the water column, which focuses phytoplankton in upper layer where light is more plentiful and grazing pressure less intense.

Horizontal transport of phytoplankton results from tidal flow, wind-driven flow, and horizontal gradients of water density (Cloern 1996). It is important to note, however, that phytoplankton are not completely passive and subject to their environment. Species living in SCEs have evolved to better suit the physical and chemical variability of their environment. For instance, the photosynthetic ciliate *Mesodiniuin* *rubrum*, which forms visible red tides in San Francisco Bay (Cloern and others 1994), actively swims toward the surface on incoming flood tides and downwards to avoid seaward migration during ebb tides. Other swimming species migrate to the surface during the day to take advantage of sunlight for photosynthesis and return to nutrient rich bottom waters at night (Crawford and Purdie 1992).

Phytoplankton blooms occur when growth rates exceed mortality, predation and transport. Cloern (1996) breaks blooms down into three classes: recurrent seasonal events can occur in any season and usually last over a period of weeks, aperiodic events can occur at any time and usually last over a period of days and exceptional events are typically dominated by few species, sometimes noxious or toxic, and persist for months.

Phytoplankton blooms alter not only the population but also the composition of phytoplankton species. Species composition of blooms changes in response to differing resources and physical environments. Some species produce resting cells that sink to bottom sediments when growth is limited and then seed blooms when conditions favor growth (Cloern 1996). Blooms of certain algal species can be toxic to fish, shellfish, and marine mammals and may pose a direct threat to public health. The factors that cause harmful algal blooms, such as red tides, are poorly known (National Research Council 2000).

2.1.3 Phytoplankton Effects on Dissolved Oxygen

Phytoplankton photosynthesize during daylight hours and respire at night. This creates a diurnal cycle of higher DO concentrations during the day and lower DO concentrations at night and early morning hours.

Phytoplankton populations will continue to increase until one of the nutrients or light availability necessary to their growth is depleted. As phytoplankton die, their biomass settles to the bottom sediments where it is metabolized by bacteria. In breaking down the algal biomass, the nutrients and carbon that made up the algal cells are remineralized. The bacteria use oxygen as they metabolize the algal biomass. Significant loads of biomass to the bottom sediments create a high demand for oxygen transfer from the water column. Thus, significant phytoplankton growth and subsequent death can lead to low DO conditions in surface waters.

Low DO conditions have detrimental effects on the plant and animal communities in the ecosystem. Most aquatic organisms require oxygen in specific concentration ranges for respiration and efficient metabolism, DO concentrations above or below this range can have adverse physiological effects. These effects range from lowered immune system function to organism mortality (Mellergaard and Nielson 1987). Low DO conditions can result in decreased species variability.

Low flow rates and long residence times can also lead to low DO conditions. Transfer of oxygen from the atmosphere to the water column is the primary source of reaeration. Atmospheric transfer occurs at the water surface and is driven by the concentration gradient between the air and the water. Flow induced mixing brings low DO water to the surface and homogenizes DO concentration in the water column. Low flow rates produce stratified systems where the water at the surface has a higher DO concentration than the water at the bottom. Deep systems are almost always stratified. Bathymetric features can lead to low DO areas which are not well-mixed with the rest of the water column.

2.1.4 Regional Factors Important to Project Evaluations

The South Bay is a semi enclosed basin with salinity that is near oceanic during the dry season and is diluted by freshwater inputs during the winter-spring wet season. Phytoplankton primary production is the largest source of organic carbon to the South Bay (Jassby 1993). As noted above, phytoplankton growth is a function of nutrient availability and light energy. The South Bay is nutrient-limited only about 15% of the time (Cloern 1999). Summer phosphate concentrations often exceed 10 µM, much greater than the typical <0.5 µM found in the adjacent Pacific Ocean (Van Geen and Luoma 1993). Anthropogenic nutrient inputs to the South Bay are primarily from wastewater treatment plant effluent and riverine flows. Although high in nutrients, the South Bay experiences limited phytoplankton blooms because suspended sediments maintained by wind and tidal mixing are sufficient to produce a light limited environment (Cloern 1996).

Much of the phytoplankton population variability in San Francisco Bay is in response to changing physical forcing functions including freshwater flows, tides, wind stress at the water surface, and irradiance (Lucas and others 1999a; Lucas and others 1999b). Freshwater riverine flows vary seasonally and yearly. The tidal cycle is characterized by two unequal flood and ebb cycles each day and two unequal spring and neap tides every lunar month. Day to day fluctuations in phytoplankton biomass are commonly associated with meteorological events such as rain fall pulses, wind conditions, or periods of abrupt warming (Cloern 1996).

Wind speeds in the South Bay vary with location and reach maximums in the summer. The tidal amplitude in the far South Bay below the Dumbarton Bridge is about 2 meters. The tidal excursion (horizontal displacement of a water parcel during a tidal cycle) ranges between 7 km at neap tide and 13 km at spring tide in the main South Bay channel and between 3 km and 8 km in the subtidal shoals, with much smaller excursions occurring on expansive South Bay intertidal mudflats (Cheng and others 1993; Walters and others 1985). The maximum tidal current speed is about 0.75 m/s in the main South Bay channel, and less than 0.3 m/s on the shoals (Conomos 1979; Walters and others 1985). Background chlorophyll-a concentration is usually less than 5 µg/L. Chlorophyll-a concentrations peak at the two slack tides each day and increase with increasingly shallow water (Cloern 1996). A more detailed description of these processes can be found in the SBSP Hydrodynamic and Sediment Dynamics Existing Conditions Report (PWA and others 2005).

The South Bay's tidally-averaged circulation is slow with mean seaward flow along the eastern shallows and landward flow along the channel (Cheng and Gartner 1985). The hydraulic residence time is several months (Walters and others 1985).

The Bay has a large and somewhat predictable spring phytoplankton bloom. Blooms in the South Bay correspond with enhanced stratification when surface salinity is diluted by increased freshwater inflows (Cloern 1996). Phytoplankton losses in the South Bay are due to mainly benthic and some pelagic grazing (Cloern 1982). A hypothesis is that the absence of blooms in the late summer-autumn period is a response to seasonal cycles of benthic grazers whose biomass and grazing rate are highest in summer (Cloern 1996). The difference in the wind climate, and therefore light availability within the South Bay is another possible factor inhibiting regular bloom occurrence in the fall (May and others 2003).

In September of 2004 San Francisco Bay experienced the largest phytoplankton bloom in close to 30 years of observation. The bloom occurred when the water column stratified due to calm winds and high air temperatures. This suppressed mixing to allow increased dinoflagellate growth, the phytoplankton that causes red tide. The bloom dissipated within a week before any harmful effects resulted (Cloern and others 2005).

2.1.5 Potential Effects of Restoration Actions

The SBSP Restoration Project plans restoration of large pond areas to tidal action through levee breaching. The tidally-restored areas will rely on estuarine derived sediment deposits from the water column to build up bottom elevations where the ponds have subsided. On the incoming tide, waters with ambient Bay suspended sediment concentrations fill the breached ponds. At slack water (high tide in a closed system), suspended sediments deposit in the accreting ponds, lowering the concentration of suspended sediments in the outgoing water. The time averaged effect just outside the levee breach is decreased turbidity and increased light penetration that can stimulate algal growth.

Some ponds will be operated as muted tidal systems by opening the culverts and flap gates to allow the maximum possible tidal exchange. These muted-tidal systems may also experience increased phytoplankton growth as a result of weakened tidal mixing. Tidal mixing re-suspends bottom sediments promoting a turbid environment, and delivers phytoplankton biomass to benthic consumers. Weakened tidal mixing could lead to a deeper photic zone that promotes phytoplankton growth (Schoellhamer 1996), slower phytoplankton removal rates due to less diffusive transport (Koseff and others 1993) and decreased grazing loss from decreased exposure to benthic grazers (Cloern 1996).

In addition to weakened tidal mixing the muted-tidal systems will have diminished wind mixing in comparison to the Bay. Wind mixing at the water surface will be inhibited by pond levees left in place to decrease wind-wave erosion in the restoration area. The net effect of both decreased tidal- and wind-induced mixing may be increased phytoplankton growth and potentially low DO conditions.

Additional ponds will be reconfigured and managed to provide ideal shorebird habitat for both nesting and foraging. This includes many in-pond islands for bird nesting and typical water depths of 0.15 m (6 inches). Mixing within the reconfigured ponds will be low due to low flow rates and decreased wind speed as a result of pond levees, berms, and islands. Sunlight will penetrate the shallow depth providing the light energy necessary to promote phytoplankton growth. Increased water temperatures are also a likely result of the shallow conditions and could further enhance phytoplankton growth, especially in the summer.

In addition to increased phytoplankton growth, long hydraulic residence times may result in low DO conditions. Stagnant and near stagnant conditions may allow water column DO to be depleted by sediment oxygen demands (SOD) and low reaeration rates, especially in bottom waters.

2.2 Managed Pond DO Model

The reason to develop a model for DO in managed ponds is to evaluate specific questions about managed pond design and operations:

- Can flow (and therefore residence time) through managed ponds be adjusted to avoid low DO?
- If so, what is the target residence time for water in a managed pond?
- If not, what other alternatives can avoid low DO?

The residence time is defined as the volume of the pond divided by the flow rate. It reflects the average time that it takes to replace the volume of a cell with flow from the inlet to the outlet. The residence time of a cell directly affects the phytoplankton accumulation rate within the pond, which in turn affects the sediment oxygen demand.

A model of DO in a managed pond was developed to assess potential effects of various residence times. The model was based on a shallow cell in Pond A16, because Pond A16 will be one of the main areas for applied studies in the first phase of the project. In addition to assessing effects of various residence times, the model was used to characterize the relative importance of sediment oxygen demand versus wind speed.

Upon review of the limited SBSP data available to support model development and evaluation, it was determined that a simple spreadsheet box model would be most suitable. A box model means that the water body is modeled as a simple box of a defined volume, with either constant or time-variable inputs of important factors such as wind speed, phytoplankton growth and respiration rates, and sediment oxygen demand. More complex models may provide more realistic predictions, but they also require more detailed input data than is currently available.

2.2.1 Conceptual Approach

Figure 1 shows a conceptual illustration of the processes affecting oxygen concentration that were modeled. The model accounts for oxygen consumption through algal respiration and SOD and oxygen replenishment through photosynthesis and atmospheric reaeration. Atmospheric reaeration is modeled as it would be in a reservoir, as opposed to a river, and is therefore wind dependent, rather than flow dependent. The model simulates a typical Bay Area daily wind pattern with the wind speed very low at night, low in the morning, and higher in the afternoon and evening. SOD is assumed to be constant.

Phytoplankton population is quantified as chl-a concentration and undergoes growth, respiration, non-predatory death, and grazing. The model decreases the phytoplankton growth rate with increasing phytoplankton concentration to simulate the effects of self-shading. One grazing rate represents both benthic and pelagic grazing loss. The model simulates diurnal effects by turning on phytoplankton photosynthesis during the day only, and turning on phytoplankton respiration at night only. Day – night lengths were set for mid-summer conditions, when phytoplankton growth rates are the highest.

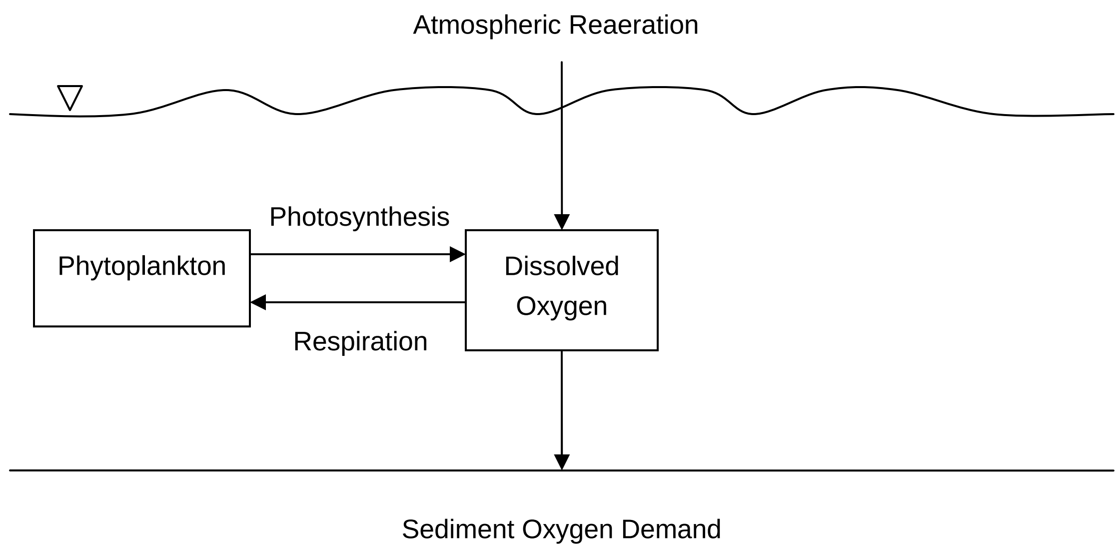

Figure 1 – Conceptual Model of Oxygen Dynamics in Shallow Ponds

2.2.2 Assumptions and Resulting Limitations

Nutrients and light energy are assumed to be unlimited such that phytoplankton grows at a maximum rate with regard to these two factors, a potential "worst-case" scenario. The ponds are assumed to be well-mixed. The model assumes that water depth is low and the pond is not stratified. Temperature effects are not included; each rate used is for a system at 20 º C. The model caps chl-a concentrations at 3000 mg/m^3, which is reasonable given the observed variations of chl-a in the project area, but capping chl-a rather than modeling growth limiting-factors is a simplifying assumption.

These assumptions lead to limitations on the model accuracy. The well-mixed assumption works for shallow ponds, but not for the deeper borrow areas and canals that proved problematic in the ISP. The temperature assumption may also be important. The fish kills occurred during one of the hottest times of year, when water temperatures reached 25-28 º C. Temperature increases can increase phytoplankton growth rates and decrease the saturated dissolved oxygen concentration. While the model may not always accurately predict outcomes because of these limiting assumptions, the general trends identified in this simple, “first-cut” analysis are worth considering before refining the model further.

2.2.3 Inputs and Computations

Table 1 defines the model parameters used in the simulations and sensitivity analyses. Values for the variables that vary by pond, including system dimensions and initial chl-a concentration, are given as ranges, the ranges are derived from a limited data set and do not represent all the possible values.

Table 1 – Box Model Inputs and Parameters

Model Parameters	Symbol	Value	Units	Reference or comment
Concentrations				
DO saturation	C_{sat}	8.5	mg/L	Based on 20 º C
Initial DO	C	5	mg/L	Assumes inlet water meets water quality objectives
Initial Chlorophyll-a	P	0 to 331	mg/m^3	(U.S. Geological Survey 2005)
Maximum Chlorophyll-a	P_{max}	3000	mg/m^3	(Zimba and Gitelson 2006)
Rates				
Maximum daily wind speed	v	0.4 to 3.6	m/sec	California Irrigation Management Information System, Union City Station
Sediment Oxygen Demand	SOD	10 to 64	mg/m^2hr	(Grenz and others 2000)
Maximum phytoplankton growth rate	μ_{max}	1.3	day^{-1}	1.3 to 2.5 per day at water temp 20 celsius (U.S. Environmental Protection Agency 1985);Note this does not model worst case, this could increase at higher temperatures
Phytoplankton respiration rate	ρ	0.1	day^{-1}	0.05 to 0.15 per day at water temp 20 C (U.S. Environmental Protection Agency 1985)
Phytoplankton non predatory mortality rate	m	0.1	day^{-1}	0.01 to 0.1 per day (U.S. Environmental Protection Agency 1985)
Grazing rate	G	0.1	day^{-1}	This is a "worst-case" assumption, i.e., that grazing rates are lowest
System Dimensions				
Water Surface Area	A_s	Less than 25 to nearly 500 acres	m^2	Assuming pond water surface area will be total surface area of current configuration minus 15% for levees, berms, and islands
Bottom Area	A_b	Assumed to be equal to water surface area	m^2	
Water depth	h	Averages 0.15	m	Pond A16 Phase 1 Project Description
Time step	Δt	1	hour	

The model computes the oxygen consumption and replenishment as follows:

Reaeration:

The flux of dissolved oxygen to the water column is computed as

$F_c = k_L\left(C_{sat} - C\right)$ (U.S. Environmental Protection Agency 1985)

where: F_c = flux of DO to the water column
k_L = wind dependant surface transfer coefficient
C_{sat} = DO saturation concentration
C = DO concentration

and

$k_L = 0.362v^{\frac{1}{2}}$ for $v \le 5$m/sec (Banks 1975)

where: v = wind speed

Respiration and Photosynthesis:

DO is replenished in the water column through photosynthesis during daylight hours. Daylight is set from 6:00AM to 9:00PM, a typical summer day. DO is then consumed at night through phytoplankton respiration. Oxygen depletion or enrichment is calculated by

$$\frac{dC}{dt} = \left(a_1\mu_t - a_2\rho\right)P$$ (U.S. Environmental Protection Agency 1985)

where: C = DO concentration
a_1 = mass of oxygen produced per mass chl-a
μ_t = phytoplankton growth rate at time t
a_2 = mass of oxygen consumed per mass chl-a
ρ= phytoplankton respiration rate
P = phytoplankton concentration as chl-a

P is computed by

$P_t = P_{t-1} + (\mu_t - \rho - m - G)P_{t-1}\Delta t$ (U.S. Environmental Protection Agency 1985)

where: P_t = chl-a concentration at time t
P_{t-1} = chl-a concentration one time step before time t
m = phytoplankton non-predatory death rate
G = grazing rate
Δt = one time step

for each time step. The algal growth rate, μ, is decreased with increasing phytoplankton concentration to simulate the effects of self-shading.

$$\mu_t = \mu_{\max}\left(\frac{P_{\max} - P_{t-1}}{P_{\max}}\right)$$

where: μ_{max} = maximum phytoplankton growth rate

P_{max} = maximum chl-a concentration, a specified constant

Mass Balance:

The model computes the overall change in oxygen concentration, ΔC, for each time step as

$$\Delta\,\mathrm{C} = \mathrm{Reaeration} - \mathrm{SOD} + \mathrm{Photosynthesis\ \text{-}\ Respiration}$$
$$\Delta C = F_c A_s V^{-1}\Delta t - SOD * A_b V^{-1}\Delta t + (a_1\mu - a_2\rho)P\Delta t$$

where: V = volume of water in the pond

A_s = water surface area of pond

A_b = bottom area of pond

and the new oxygen concentration as

$$C_t = C_{t-1} + \Delta C$$

Where: C_t = the DO concentration at time t

C_{t-1} = the DO concentration at time t-1

2.2.4 Model Implementation

The algal growth and respiration rates were normalized by the number of light and dark hours per day, respectively. The model assumes that there are 15 hours of daylight for each 24 hour day. The hourly oxygen replenishment through photosynthesis is the daily rate divided by 15. Similarly, the hourly oxygen consumption through respiration is the daily respiration rate divided by 9. All other rates were assumed to be constant over 24 hours. Zooplankton and benthic grazer populations are not modeled explicitly because community data are not available for the SBSP Restoration Project Area. Instead, a specified constant grazing rate equal to the non-predatory death rate is applied. This is a conservative assumption because benthic grazing is a known significant loss of phytoplankton biomass in a shallow, well-mixed system like the one modeled (Cloern 1982).

Daily average wind speed is measured at the Union City Station of the California Irrigation Management Information System, which is maintained by Alameda County Water District. Data from January 2001 to July 2006 were available. The daily average varies from 0.4 to 3.6 m/sec and averages 1.6 m/sec. No seasonal trends were evident in this data set, however there is a known seasonal trend in wind speed in the Bay Area, with the strongest winds occurring in the spring and summer (Conomos 1985).

2.2.5 Model Outputs and Initial Findings

The model provides two key outputs to answer management questions:

- **The time it takes for model to reach equilibrium.** If the time to reach equilibrium is longer than design residence time, then water will be replaced faster than phytoplankton accumulation can cause low DO, and the model conclusion is that low DO would not be a problem. This model output addresses the first two pond management questions: can flow be used to manage DO, and if so, what is the target residence time? The target residence time would be shorter than the model equilibration time, because keeping the system at disequilibrium means that phytoplankton are being flushed out faster than they can grow to their maximum density.

- **The DO and chlorophyll concentrations at equilibrium.** If the residence time is longer than the equilibration time, then the equilibrium DO concentration is evaluated to determine whether they are above or below desirable concentrations. This model output addresses the third management question: what other options are there to manage DO? Input parameters such as wind speed and SOD can be manipulated in the model to determine what physical parameters are most important to final DO concentrations. This helps managers prioritize actions – e.g., is it more important to configure ponds and levees to increase wind exposure, or is avoidance of algal accumulation in deep ponds a more effective action?

Some important, initial conclusions can be drawn from the outputs of this very crude model. By starting with some reasonable estimates for model parameters (Figure 2), and then varying the key parameters one at a time (Figure 3 – Figure 5), the response of the model compared to actual system responses observed in the ISP help clarify the management priorities.

The system evaluated reaches steady-state within 5 days (Figure 2 – Figure 5). For any residence time longer than this, the chl-a and DO concentrations at the inlet will not affect concentrations within the ponds. Also, if the modeled pond has a hydraulic residence time longer than about 5 days, it will reach maximum chl-a concentrations, which typically indicates eutrophic conditions that would lead to depressed DO. Note that this conclusion is based on the lower end of phytoplankton growth rates, i.e., not even the “worst case” scenario.

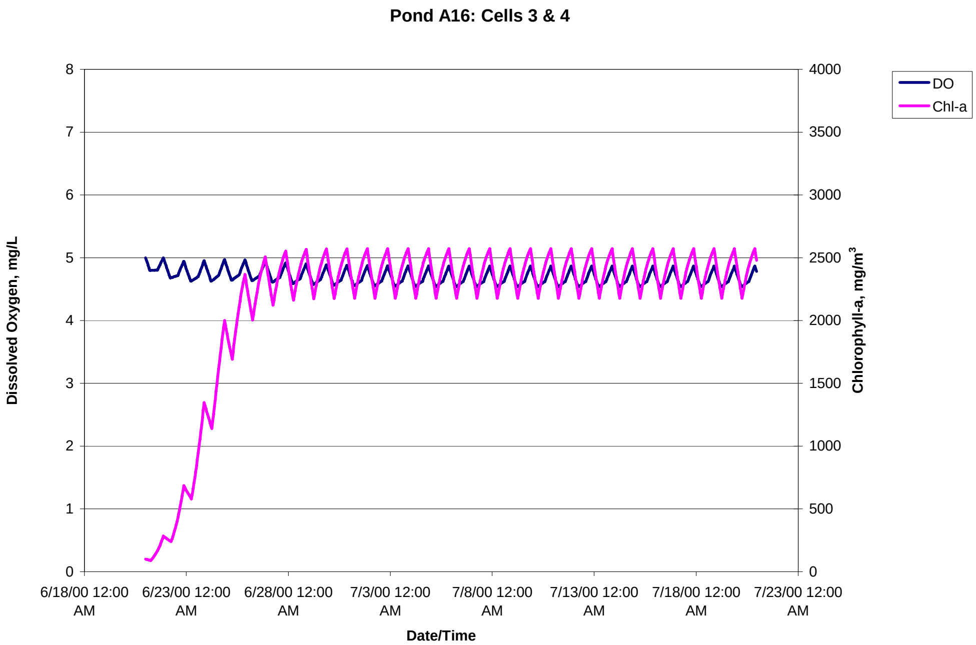

Figure 2 – Model outputs for initial DO = 5 mg/L, initial chl-a = 100 mg/m^3, SOD = 40 mg/m^{2}hr, max. wind speed = 1 m/sec.

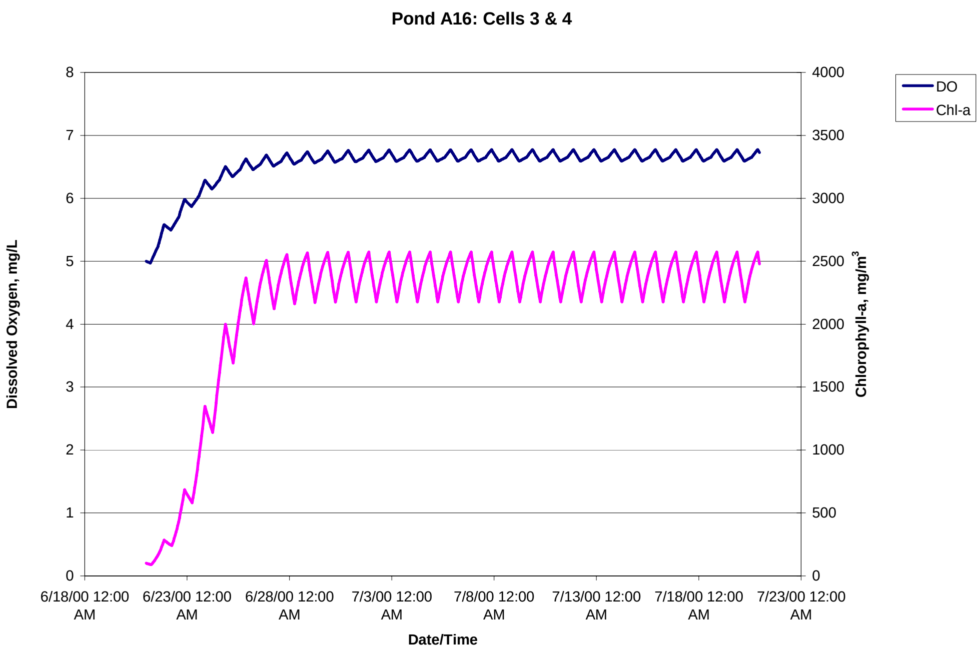

Figure 3 – Model outputs for initial DO = 5 mg/L, initial chl-a = 100 mg/m^3, SOD = 20 mg/m^{2}hr, max. wind speed = 1 m/sec.

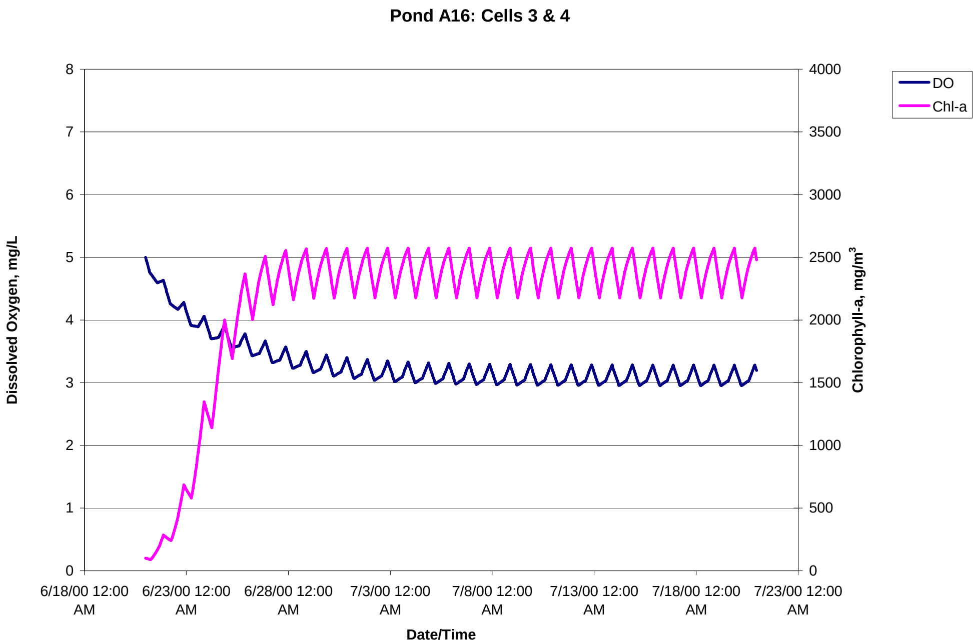

Figure 4 – Model outputs for initial DO = 5 mg/L, initial chl-a = 100 mg/m^3, SOD = 40 mg/m^{2}hr, max. wind speed = 0.5 m/sec.

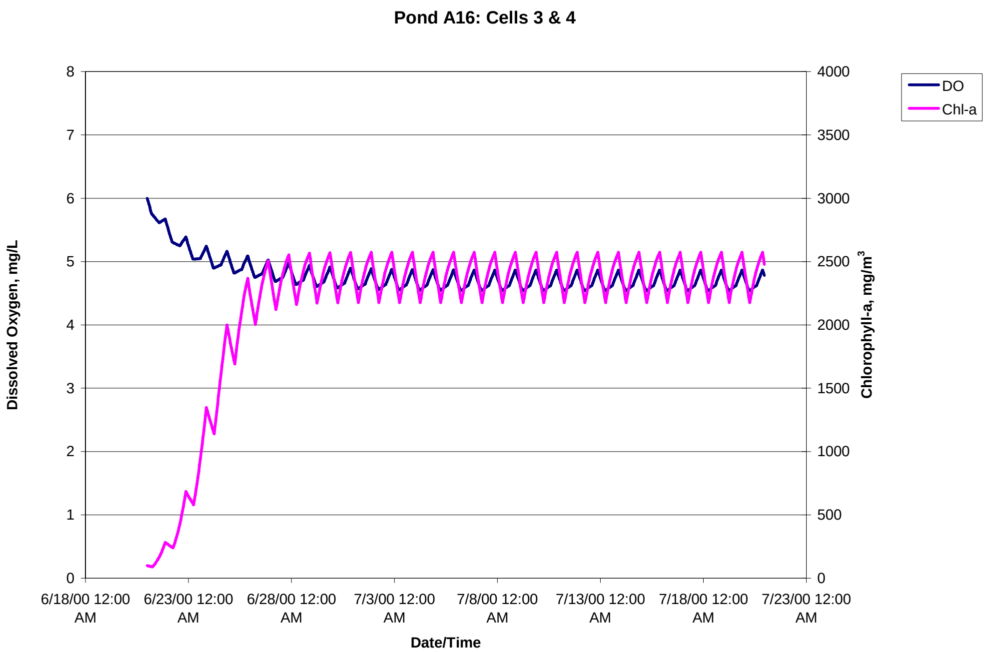

Figure 5 – Model outputs for initial DO = 6 mg/L, initial chl-a = 100 mg/m^3, SOD = 40 mg/m^{2}hr, max. wind speed = 1 m/sec.

The outcome that the modeled DO does not drop below 5 mg/L at high chl-a concentrations results from the simplifying assumption that sediment oxygen demand is constant. In reality, as observed in the ISP, SOD increases when phytoplankton bloom, die, and decay on the bottom. This highlights the importance of the second conclusion, which is that SOD has a significant effect on the final DO concentration. Comparison of Figure 2 to Figure 3 shows that attainment of the water quality objective of 5 mg/L is possible at an SOD of 20 mg/m^2hr, but not 40 mg/m^2hr. for the conditions modeled.

This "break-point" is within the SOD range of 10 to 64 mg/m^2hr found by Grenz and others (2000) during their study of the Bay spring phytoplankton bloom in 1996. Their study had two sample locations, one in the main South Bay channel and one in the shoal. Water depths at mean tide were 15 m at the channel site and 2 m at the shoal site. Chl-a concentrations were similar at each site and ranged from 2 to 3 mg/m^3 pre-bloom to a maximum of 55 mg/m^3. SOD varied between 0 and 35 mg/m^2hr before the bloom with no consistent inter-site difference. After the bloom peak, the SOD increased at the channel site reaching a maximum demand of 64 ± 26 mg/m^2hr. No clear increase in SOD was observed at the shoal site.

A significant difference in benthic chl-a concentration was observed between the channel and shoal sites and is the likely cause of increased SOD in the channel. Surface sediments at the shoal site had a mean chl-a concentration of 43 mg/m^2. Chl-a in the channel sediments was 150 mg/m^2 before the bloom and reached a maximum concentration of 335 mg/m^2. The increased SOD in the channel corresponds in time to the increased chl-a concentrations in the channel surficial sediments. Tidal mixing in the shoals is very weak due to the slower tidal currents in the shallow depths (Lucas and others 1999a). Wind-induced mixing and resuspension likely inhibited settling of algal biomass to shoal sediments. Shoal generated biomass was likely transported to the channel in the channel-shoal exchange that is known to occur in the South Bay (May and others 2003). Therefore, an increase in chl-a concentrations and SOD were not observed at the shoal site.

The existing and predicted water column chl-a concentrations in the ponds of interest are greater than that found by Grenz et al. (2000) during the peak of the bloom. The water depths in the ponds will average 0.15 meters, significantly shallower than the channel and shoal sites and wind-induced resuspension and mixing will likely be minimal. Therefore the phytoplankton biomass will settle to the pond bottom quickly, potentially resulting in high SOD. To relate this back to the third pond management question, the model is telling us that monitoring should focus on SOD and adaptive management actions should be directed at reducing accumulation of phytoplankton biomass in sediments.

Decreasing wind speeds cause steady-state DO concentrations to drop even more, from just below 5 mg/L at a wind speed of 1 m/sec (Figure 2) to around 3 mg/L when the wind speed drops to 0.5 m/sec (Figure 4). Increasing the inlet concentration from 5 mg/L (Figure 2) to 6 mg/L (Figure 5) does not affect the final DO concentration, though it does slightly extend the time to reach steady-state. Therefore, also addressing the third pond management question, where the water comes from is less important than how it is re-aerated within the pond, either by wind exposure or mechanical aeration.

2.2.6 Application to Pond A16

Modeling of Phase 1 managed pond restorations is focused on Pond A16. Pond A16 will be an applied study to test habitat configurations and management techniques on vegetation, predation, and water quality.

Under the ISP, Pond A16 is managed as a system with Pond A17. The intake is located at Pond A17 from Coyote Creek and the discharge is from Pond A16 to Artesian Slough. The flow direction is reversed in the winter to prevent entrapment of migrating salmonids coming in from Coyote Creek (U.S. Fish and Wildlife Service 2005a; U.S. Fish and Wildlife Service 2006).

Pond A16 experienced anoxic conditions in July and August 2005. Low DO conditions began in July and significantly declined at the end of the month. On August 1, 2005, managers temporarily closed the levee gates in order to protect Artesian Slough. This exacerbated the low DO problem in Pond A16 and led to a within-pond fish kill on August 10, 2005 (U.S. Fish and Wildlife Service 2005b). Managers then fully opened the pond intake gates which alleviated the low DO conditions in the pond, but may have had a short-term effect on the slough.

Wind speeds during this period averaged 1.5 m/sec at the Union City weather station, the closest known weather station to the Alviso project area. These conditions are consistent with the overall daily average of 1.6 m/sec over the five year period of record.

In 2005, the daily mean temperature in Pond A16 varied from approximately 15 to 28 degrees Celsius (U.S. Fish and Wildlife Service 2006). Peak temperatures were reached in mid-July. Water temperatures above 25 degrees C were maintained through August 10th after which they steadily declined. The DO depression could have been caused by increased algal growth and decay as a result of heightened water temperature, coupled with sufficient light intensity and other conditions favoring phytoplankton growth.

Water depths in the ponds are currently maintained at an average of 0.37 m in A17 and 0.52 m in A16. The shallower depths proposed for the A16 pond reconfiguration (0.15 m) will likely raise the average water temperature in the system. Light intensity is also greater over a shorter water depth. An increase in temperature and light intensity may result in increased phytoplankton growth and dissolved oxygen consumption.

The preliminary restoration plan for A16 will reconfigure the pond area into an inlet canal, four cells, and an outlet canal, as shown in Figure 6. The inlet canal and outlet canal utilize the pond's existing borrow ditches, with depths averaging 1.5 to 2 meters, with a maximum depth of 5.6 meters. As a result, the canals cannot be modeled accurately with the present spreadsheet box model because their depths are such that stratification will occur and the model does not account for stratification. With stratification DO concentrations in the canals would be considerably lower than concentrations in the cells. The spreadsheet model is appropriate for the cells which have a typical depth of 0.15 m. The water flow through the cells will be in parallel except for cells 3 and 4 which will be run in series.

The model of Pond A16 was developed with the physical parameters presented in Table 2.

Table 2 – A16 Restoration Design Physical Parameters (data from PWA, July 27, 2006)

Cell	Open water area (m^2)	Typical depth (m)	Design volume (m^3)	Evaluation Residence Time Range* (days)
1	109,654	0.15	35,918	1.6 - 3.5
2	224,241	0.15	119,448	5.3 - 11.5
3+4	323,561	0.15	186,919	12.5 - 25.8

** The residence time ranges are based on preliminary modeling and analyses and include assumptions regarding the size and number of culverts in each cell. The actual residence times will vary depending on the design.*

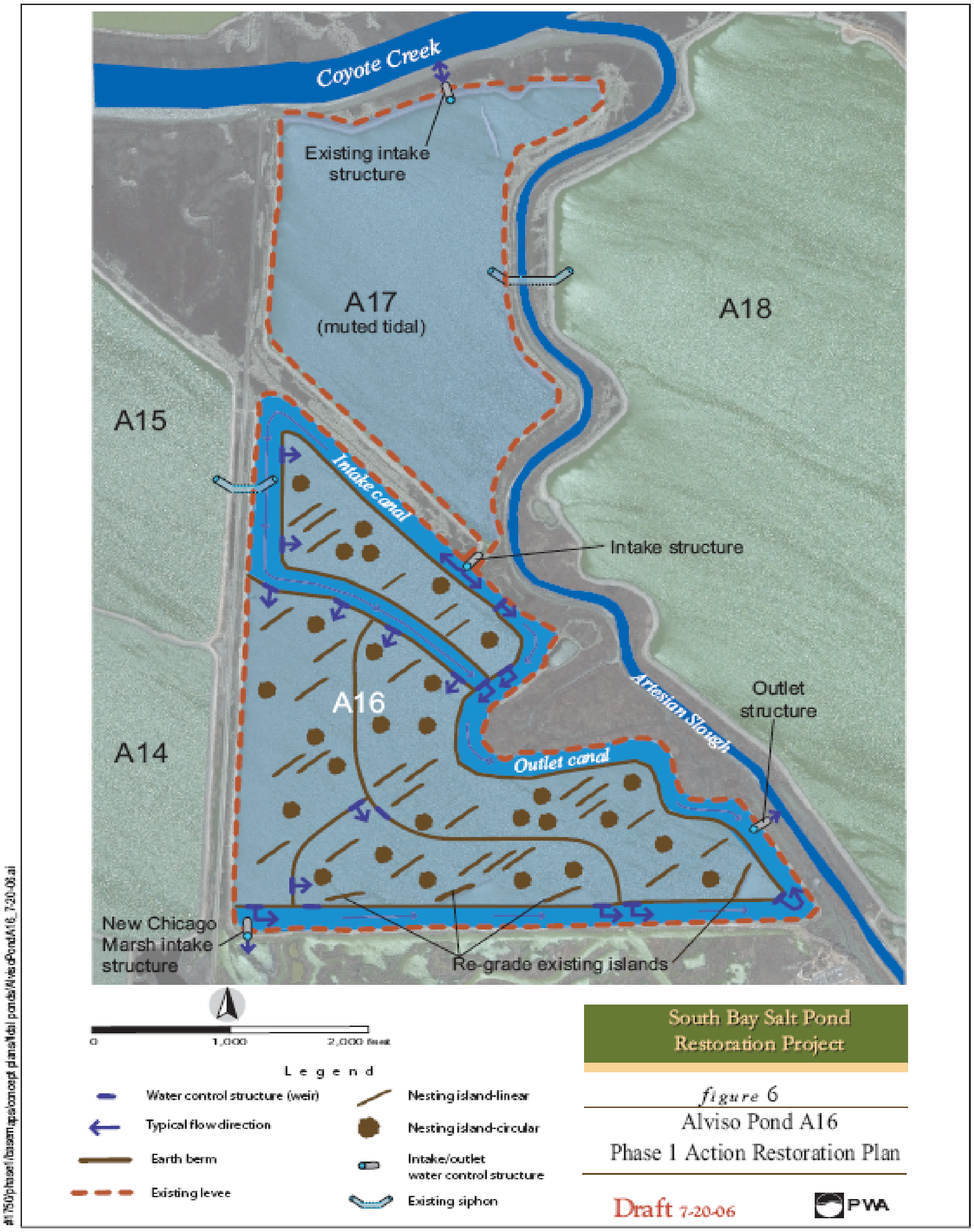

Figure 6 – Plan view of Pond A16 reconfiguration design

The flow path with the lowest surface area to volume ratio, and therefore the lowest reaeration, is through cells 3 and 4. This flow path will also have the longest hydraulic residence time. Cells 3 and 4 were modeled as the potential worst case of the three flow paths. Water quality inputs, such as initial DO and initial chl-a concentrations, are best approximated by USGS data collected in spring 2003. This data set includes the only chl-a data collected from the ponds to date. These data may not be representative of future worst case conditions for two reasons. Chl-a levels could be expected to be higher and DO levels could be lower later in the summer season when temperatures are warmer and light intensity is greater. Chl-a levels could increase and DO levels could decrease under the proposed shallower conditions as well. However, as noted above, the initial DO and chl-a concentrations do not drive the ultimate DO concentrations, but rather the time to reach those levels.

Pond A17 is the source of water to Pond A16. Chl-a concentrations in A17 averaged 240 mg/m^3 and chl-a in A16 averaged 330 mg/m^3 in spring 2003 (U.S. Geological Survey 2005). Average daily DO concentrations during this period ranged from approximately 4 to 7 mg/L (U.S. Fish and Wildlife Service 2006).

Table 3 shows model results for a range of inputs. Since the system doesn't exist yet, a range of conditions needed to be explored, though even the modeled range may not bracket actual conditions. For example, SOD was based on Bay sediment conditions, which are the only available SOD data, whereas in-pond SOD may likely be higher based on recent observations. All rate constants assume water temperature is 20 °C, whereas critical DO depletions have been observed in the ponds when water temperatures were up to 28 °C. If more predictive numeric models are desired, there are likely other parameters in need of refinement, such as heterotrophic respiration and salinity effects. The residence time implications are presented for the outlet water quality objective (WQO) of 5 mg/L DO and for an in-cell DO of 2 mg/L. Table 3 is not a prediction of expected future conditions, ponds cells may well drop below 2 mg/L DO based on recent observations. Table 3 provides a starting framework for the assessment of potential future DO responses based on existing knowledge and stated model assumptions. It helps understand how "turning the knobs" makes the system respond. Refining the model and obtaining adequate monitoring data will lead to greater predictive power.

Row one shows the results for initial conditions similar to those found in July and August 2005. At the time the gates were closed the daily average DO was about 5 mg/L. Assuming an initial chl-a concentration of 240 mg/m^3 and a continuous sediment oxygen demand of 60 mg/m^{2}hr, the daily average dissolved oxygen after 10 days in the pond was predicted to be about 3.8 mg/L. In comparison, the actual measured daily average in the pond was about 1.6 mg/L after 10 days.

The model's over approximation of the average DO may be an over approximation of reaeration and/or an under approximation of SOD. The model reaeration rate may be too high because the actual wind speed at the water surface is reduced due to the presence of levees and berms. The sediment oxygen demand used is the high end of the range found by Grenz and others (2000) as discussed above. It is possible that the low water depths and low flows within the ponds lead to higher sediment oxygen demands as decaying biomass quickly reaches the bottom sediments and may not be swept away with water flow. There is also generally more biomass in the ponds than there was in the Bay. Chl-a concentrations found in the ponds by USGS (240 and 330 mg/m^3) are significantly greater than the maximum concentration (55 mg/m^3) observed during the bloom in the South Bay when SODs were measured.

Table 3 – Pond A16 Model Results

Initial Dissolved Oxygen (mg/L)	Initial Chlorophyll-a (mg/m3)	SOD (mg/m2hr)	Maximum Daily wind speed (m/sec)	Residence Time to meet 5 mg/L DO	Residence Time to meet 2 mg/L DO
Maximum SOD					
5	240	60	1	Never met	Always met
5	100	60	1	Never met	Always met
6	100	60	1	Met first 24 hours	Always met
Minimum SOD					
5	100	10	1	Always met	Always met
5	240	10	1	Always met	Always met
Variation in initial chlorophyll-a					
5	50	30	1	Met after 12 hours*	Always met
5	100	30	1	Met after 13 hours*	Always met
5	240	30	1	Met after 11 hours*	Always met
Variation in initial dissolved oxygen					
2.5	100	30	1	Met after 88 hours	Always met
4	100	30	1	Met after 62 hours	Always met
8	100	30	1	Always met	Always met
Variation in maximum daily wind speed					
5	100	20	0.1	Never met	Always met
5	100	20	0.3	Consistently met after 60 hours*	Always met
5	100	20	0.5	Consistently met after 11 hours*	Always met
5	100	20	0.7	Consistently met after 7 hours*	Always met
5	100	20	1	Always met	Always met
At what point does SOD exceed reaeration?					
5	100	10	1	Always met	Always met
5	100	20	1	Always met	Always met
5	100	30	1	Met after 12 hours*	Always met
5	100	40	1	Never met	Always met

* DO drops below 5 mg/L at night

It should be noted when interpreting the results in Table 3 that model error has not been quantified and residence time implications are approximate. Actual DO concentrations may be higher or lower. The assessments included in Table 3 are purely hypothetical. The purpose of Table 3 is to indicate how assessments could be done once the appropriate data become available and incorporated into a model that includes all the key processes.

For all conditions simulated, the DO concentration in the cells never drops below 2 mg/L. Under high SOD and/or low wind speed conditions, the DO concentration does drop below 5 mg/L.

With an initial DO of 5 mg/L, if the SOD is at the high end of the range found by Grenz and others (2000) the DO will not meet the outlet WQO of 5 mg/L but will meet the in cell objective of 2 mg/L. If the SOD is at the low end of the range, both WQOs are always met, given a maximum daily wind speed of 1m/sec. Even with low SOD, if the maximum daily wind speed drops below 0.1 m/sec, the outlet WQO will not be met in the cells.

When the SOD is 30 to 40 mg/m^2hr, the high end of the range observed by Grenz and others (2000) before the spring 1996 phytoplankton bloom, the DO WQO of 5 mg/L may not be met in the cells, even with average wind conditions.

As can be seen in the table of model results, variations in initial chl-a have less significant effects on residence time results than variations in initial DO. Meeting DO objectives in Pond A17 will improve chances of meeting the objectives in Pond A16. DO conditions in Pond A17 can be improved, if necessary, by opening the pond to further tidal action.

The DO response to high SOD is illustrated in Figure 7. Here, the initial DO is 6 mg/L, initial chl-a is 100 mg/m^3, SOD is 60 mg/m^2hr, and maximum wind speed is 1 m/sec. DO decreases below 5 mg/L within 24 hours.

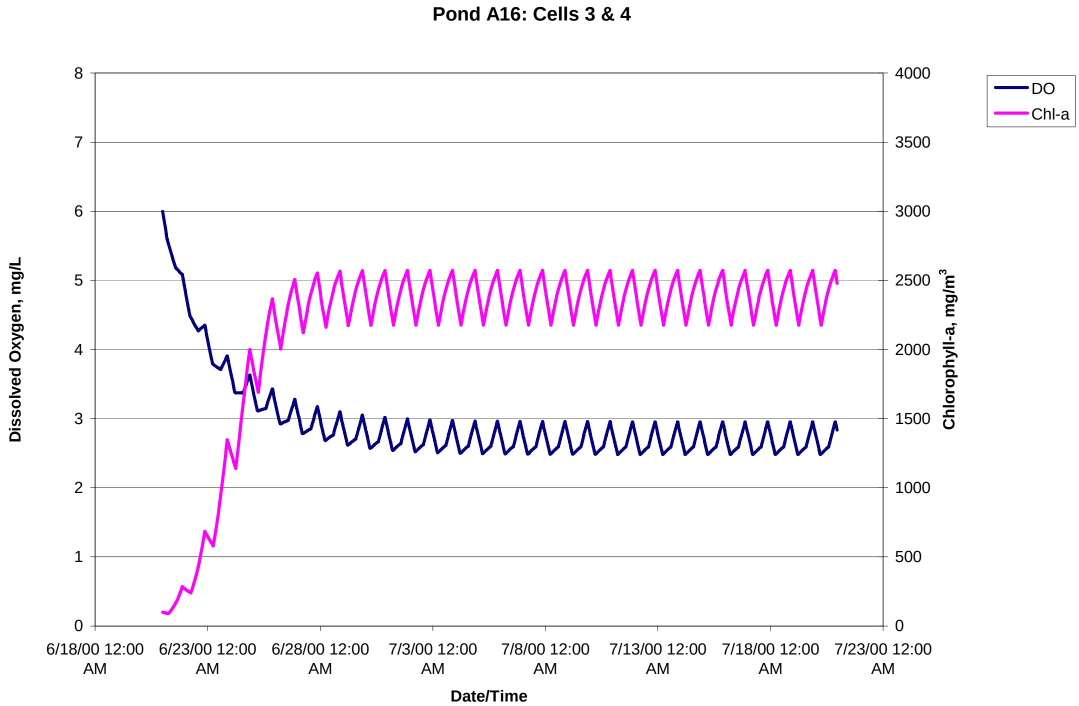

Figure 7 – Pond A16 Model Response to High SOD

Varying the daily maximum wind speed shows us that wind speeds at the water surface below 1 m/sec lead to nighttime periods of low DO even at low SODs, in this case 20 mg/m^2hr. With moderate SODs of 30 to 40 mg/m^2hr, low wind conditions will result in low DO conditions. Wind speeds at the pond's water surface will be less than those at the nearby open water of the Bay due to the presence of the existing pond levees and proposed additional berms and islands. The effects of the levees, berms and islands on wind speed are not included in this analysis.

For all simulations, changes in initial DO and initial chl-a concentrations do not alter the steady-state concentrations reached, but do alter the time it takes to get there. The approximate maximum time needed to reach steady-state is fourteen days. This was determined by starting the model at 0 mg/m^3 DO or chl-a and at maximum DO and chl-a values.

2.2.7 Uncertainties/Data Needs for the Pond A16 Model

The preliminary possible hydraulic residence times for each cell range from 1.6 to 25.8 days. For cells with residence times less than 14 days, modeled water quality depends on initial chl-a and DO concentrations. For cells with residence times greater than fourteen days, the model reaches steady-state concentrations independent of initial conditions. All water quality results are dependent on SOD and wind speed, regardless of residence time.

The accuracy of model results will be greatly improved with monitoring and studies focused on achieving a better understanding of SOD in the ponds. SOD a key monitoring need and is a very likely candidate for a trigger in the adaptive management plan.

The wind speed information currently employed is from a monitoring station in Union City maintained by the Alameda County Water District as part of the California Irrigation Management Information System. The daily average wind speed at 2 meters above the ground surface is recorded. The ground surface is at an elevation of 16 feet (datum unknown). The station is located approximately 5 miles east of the Bay.

Levees and islands within the restored ponds will block winds at the water surface and decrease wind driven waves. Placement of an anemometer at one of the ponds would improve model input accuracy and be a useful tool for adaptive management. Wind speed and re-aeration rates play another critical role in controlling DO, so having site-specific information on wind exposure at the pond surface will help determine whether mechanical re-aeration is necessary, or whether natural wind exposure is sufficient.

2.2.8 Model Limitations and Suggested Improvements

The present model is limited in function and applicability. It is only applicable to the shallow cells of the proposed restorations and not to the deeper inlet and outlet canals where the water column may be stratified.

The model does not presume to be an accurate depiction of phytoplankton dynamics, but rather attempts to quantify the effects of worst-case phytoplankton growth on pond DO concentrations. Additional oxygen demands are present in the ecosystem that are not accounted for in the model, these include oxygen depletion from chemical oxygen demands in the water column such as nitrification, as well as any non photosynthetic respiration by organisms including microbes, fish, and zooplankton. These oxygen demands are generally expected to be minor in comparison to the SOD, although the ISP updates did speculate that when the tide gates on A16 were closed, the fish inside may have consumed their own oxygen.

The limited data currently available to provide initial conditions and calibrate the model are collected from the ponds in their current conditions. Initial water quality conditions at the time of restoration are not known. Analysis of the effects of the restoration, including decreased flow, tidal influence, and shallow depths on initial water quality should be considered in analyzing model results.

The phytoplankton growth rates and respiration and photosynthesis rates are typical values found in a number of studies and summarized in Rates, Constants, and Kinetics Formulations in Surface Water Quality Modeling (Second Edition), a publication of U.S EPA (1985). These data are not site specific but provide the best approximation to actual processes at this time. A key area where the model departs from reality is the mechanism for limiting phytoplankton abundance. The spreadsheet model simply caps the chl-a at a maximum value, when in increased grazing pressure, self-shading, agglomeration and settling, and other factors limit maximum phytoplankton densities.

A potentially significant limitation of the model is its inability to model temperature effects on phytoplankton growth, reaeration, and SOD. This is largely due to the lack of quantitative knowledge of each processes response to temperature. Conceptually we know that increased water temperature promotes algal growth and SOD and inhibits reaeration. SOD is increased with warmer water temperature because of increased biological activity and because greater phytoplankton growth results in more biomass settling to the bottom, further increasing the sediment biological activity. Reaeration is decreased with increasing water temperature because the saturation DO concentration is decreased. The magnitude of the difference between the present DO concentration and the saturation concentration is the driving force behind reaeration. A lower saturation level results in a decreased driving force and diminished reaeration rates. Long hydraulic residence times and shallow water depths increase the potential for warm water conditions.

The model artificially "caps" phytoplankton abundance / chloryphyll-a. An improvement to the model would be to limit phytoplankton abundance by making death rates increase with increasing populations due to predation, and growth rates decrease with increasing populations due to competition for light / nutrients. Changes like this could be added iteratively in the future, subject to available data and information, until the model more appropriately matches observations. Another possible model improvement would be increasing SOD as phytoplankton density increases. This would reflect the observation from the ISP that low DO was associated with areas where algal blooms were settling out. The need is to come up with a defensible formula to relate SOD to chl-a in the model. These model limitations may explain why the model appears to underpredict the magnitude of daily oscillations in DO known to occur in the Ponds, and overpredict chlorophyll-a concentrations compared to limited initial observations.

If a more detailed model is desirable, the proposed next steps in model development are as follows:

> **1) Sensitivity Analysis.** A more formal sensitivity analysis is necessary to identify the most significant model parameters and corresponding primary data needs. Our initial findings, that SOD and wind speed primarily control DO, are consistent with Cloern's finding that blooms are largely dependent on the rate of vertical mixing from tides and wind (Cloern 1991). Results of the sensitivity analysis will provide valuable direction for monitoring efforts and scientific research in the project area in support of the AMP.
>
> **2) Development of a more sophisticated model.** As more data become available, a sophisticated DO model could be developed. Capabilities can be improved, including the ability to model stratified systems. Development of a multi-box model would allow systems with multiple cells to be modeled at once and would introduce flow rate as a model parameter. The multi-box model would account for flow induced reaeration in the oxygen balance.

Although these are useful ideas for model refinement, the PMT and the Science Team should also balance the value of additional model refinement against the value of information that could be gained from adaptive management. The next section describes the adaptive management approach that can lead to specific management techniques while avoiding exceeding thresholds of significant impacts for DO.

2.2.9 Adaptive Management Discussion

It is important in adaptively managing the ponds to recognize the interactive effects of phytoplankton abundance and low DO. Phytoplankton blooms can lead to low DO concentrations as a result of increased sediment oxygen demands and increased nightly respiration demands. Low dissolved oxygen can lead to increased phytoplankton abundance if the oxygen concentrations are too low to support the grazing community. Grazing loss is an important factor in the balance of phytoplankton growth and death/decay.

The Science Team will need to consider these narrative thresholds defined by water quality regulations in the development of numeric thresholds and triggers for the adaptive management plan. Another factor that should be considered is that the estuarine beneficial use designation, which presumably applies to the project area, recognizes that low DO does occur transiently in shallow ponds and wetlands. The consulting team recommends that thresholds for certain areas within the ponds allow DO levels as low as 2 mg/, as long as harm or nuisance is avoided.

Pond A16 Management Techniques

Experience and model results show that Pond A16 may experience low DO conditions as a result of high SODs and/or low reaeration rates. Furthermore, model results suggest that it is unlikely that increasing flow will be a viable management technique prevent low DO.

If low DO can be tolerated within the pond, one way to avoid discharging low DO water outside the pond is to use cell 4 as a mixing basin to increase DO concentrations with mechanical enhancement such as a Solar Bee. Another alternative identified is to divert some flow from Artesian Slough and mix it with the potentially low DO water in the Pond A16 outlet canal.

Low DO concentrations within the pond cells can be mitigated by lowering berms to increase wind exposure, or installing an active reaeration system. Active re-aeration may be particularly useful in deeper intake and outlet canals, where DO sags are more problematic.

Management techniques to reduce SOD are more challenging. One idea that could be considered is limiting phytoplankton growth rates by installing artificial shading. This could have ancillary temperature benefits as well, which could also improve DO.

2.2.10 Data Gaps and Monitoring Needs for Assessing Regional Impacts

The foregoing discussion has been developed with a very "pond-centered" focus, i.e. analysis of project-level as opposed to broader regional impacts. Making managed ponds ponds sustainable is essential if the program is to proceed from managed pond emphasis to tidal marsh emphasis. Phase 1 projects such as Pond A16 are high priorities for modeling and analysis because sustainability of individual ponds for water quality is a key adaptive management uncertainty for the program. The Consultant Team's analysis reflected in Sections 2.2.1 – 2.2.9 above focuses on pond-level analysis as a way of addressing the program-level question: "can ponds that sustain high bird densities also feasibly be managed to avoid low DO impacts?"

It is worth concluding Section 2 with a discussion of data gaps related to regional impacts, i.e. how does pond management affect the quality of water in the Bay? Connecting ponds to the Bay can affect water quality by discharging biochemical oxygen demand (BOD) to the Bay, and by introducing new and possibly harmful phytoplankton species to the Bay. Both of these impacts to the regional setting are discussed in the Environmental Impacts Report, including an analysis of thresholds related to Water Quality Objectives and a discussion of Adaptive Management triggers and actions.

Discharge of BOD is a concern because it could reverse decades of progress towards improved DO levels in the Bay through Clean Water Act implementation. To adequately plan to avoid impacts, decision makers will need to know the maximum tolerable loads of BOD that will prevent low DO in receiving waters. This data gap could be addressed in two steps: numeric modeling and monitoring to verify discharge rates and the actual effects on Water Quality. A simple model, building on existing models for BOD and DO in the South Bay, could be developed to characterize the maximum load from all sources that will attain DO water quality objectives in the Bay.

Shifting phytoplankton community structure is also a potential problem that can lead to direct impacts such as blooms of toxic algae, as well as more subtle changes, such as increases in mercury bioaccumulation in the food chain. At the South Bay Salt Ponds Science Sypmosium in 2006, it was noted that toxic species of phytoplankton (e.g. *Aureococcus* and *Karenia* species) were found at high abundances in some ponds and in the Bay after levees were breached. The development of the AMP could include guidance for identifying and monitoring for toxic species of concern, so that the decision criteria for sustainability of managed ponds includes avoiding harmful shifts in phytoplankton population and community structure in the Bay. Monitoring studies and pilot projects could also address the role of phytoplankton community structure on mercury bioaccumulation. This is discussed in more detail in Section 3.

3. MERCURY

3.1 Conceptual Overview

3.1.1 Problem Statement

Mercury is a major contaminant of concern in both the program-level and the project-level evaluation, because there are numerous legacy and contemporary mercury sources in the project area. Mercury is a concern because of its potential to be bio-transformed to methylmercury, which then bioaccumulates in the food web(Tetra Tech Inc. 2006). Methylmercury is a potent developmental neurotoxin. In this analysis, the main effect of concern is accumulation to harmful levels in the eggs of birds that forage in the project area. Mercury toxicity has been shown to cause bird eggs to fail to hatch at levels above 0.5 ppm (fresh wet weight) in the eggs (U.S. Fish and Wildlife Service 2003).

Another, related concern is the potential for harmful effects on humans. In the tragedy of Minimata Bay, Japan, infant birth defects and death resulted from pregnant mothers consuming fish with 5 – 50 ppm (wet weight) mercury (Davies 1991). It is important to be very clear that concentrations of mercury in San Francisco Bay fish are well below this concentration range, and that no such mercury-related effects have been observed in the Bay Area. However, more recent risk assessment information provided by the United States Environmental Protection Agency raises the concern that people who depend on fish for food need to limit their consumption if fish tissue concentrations exceed 0.3 ppm (wet weight). This guideline is to be adjusted for local consumption patterns (U.S. Environmental Protection Agency 2006).

In San Francisco Bay, the San Francisco Bay Regional Water Quality Control Board (SFRWQCB) has determined that 0.2 ppm (wet weight) mercury in large fish, such as striped bass and white croaker, is an appropriate level of protection for human consumers. The SFRWQCB has likewise determined that 0.03 ppm or less mercury (wet weight) in smaller fish that birds prey on is the appropriate level needed to protect wildlife. Because monitoring data indicate that mercury concentrations in fish exceed this level in the Bay, the SFRWQCB has developed a mercury TMDL for the Bay, which establishes numeric targets for mercury in water, sediments, and fish tissue, and implements those targets through a coordinated plan that includes monitoring, special studies, load reduction requirements from all sources, and fish consumption guidance for people who depend on the Bay for food (San Francisco Bay Regional Water Quality Control Board 2006). As discussed in the Regulatory setting, the Bay Mercury TMDL, and the Federal and State policies that require adoption and implementation of the TMDL form the basis for thresholds of significant impacts for mercury.

The mechanisms of mercury cycling, especially transformation to methylmercury and subsequent bioaccumulation, are key to predicting and managing mercury impacts from project activities. A conceptual model for mercury cycling in the different types of SBSP project areas was developed and presented in a Mercury Technical Memorandum (MTM) that is an appendix to this EIS/R (Brown and Caldwell 2004) Information on the geographic distribution of mercury in the Regional Setting and the

Project Setting is summarized in the ESR that is also an appendix to this EIS/R. Rather than repeating information summarized in the MTM and ESR, this NCAR uses the conceptual model as a framework to interpret key information from the ESR that is needed to support findings related to the impact analysis for mercury.

3.1.2 Mercury Sources

The geography and history of the Bay affects the distribution of mercury-contaminated sediments within and surrounding the project area. South San Francisco Bay has been subjected to discharges of mercury-contaminated sediments originating from the historic New Almaden mining district. The mining activities causing these discharges date back to the late 1800s and early 1900's, although the discharges persist as a legacy source in the Guadalupe River watershed. The land area around the New Almaden mines has been cleaned up and restored to beneficial use, and downstream remediation and stewardship is underway in the watershed.

However, a legacy of mercury contamination persists in the form of a north-south mercury concentration gradient in sediments. The average concentration of mercury in Bay sediments is 0.4 ppm, and the median concentration of mercury in suspended sediments is 0.3 ppm. This gradually increases to 0.5 - 0.8 ppm in the South Bay, and then sharply increases to 1 – 2 ppm in Alviso Slough, especially just after high-flow events (Tetra Tech Inc. 2005; Tetra Tech Inc. 2006).

In addition to the mining legacy sources that dominate the signature of contaminated sediments in South Bay, other sources bring mercury into regional setting and the project setting. As summarized by the SFRWQCB (2006), municipal treatment plants account for less than 1% of all mercury sources by mass, but receive a great deal of attention because of concerns over possibility that mercury in treated municipal effluent is more readily converted to methylmercury. Permitted dischargers of treated municipal wastewater are summarized in the Regulatory Setting for Mercury (Section 3.4.2 of the EIS/R). Over the lifetime of this project, the permitted dischargers will be required by the SFRWQCB to study the bioaccumulation impacts of their discharges and take appropriate management actions if needed (San Francisco Bay Regional Water Quality Control Board 2006). A number of those studies are already under way in this watershed and in other areas of California.

Atmospheric deposition of mercury is a known mercury source that in many other U.S. waters is the leading cause of excessive mercury concentrations in fish (Tetra Tech Inc. 2006). In the Bay Area, mercury from atmospheric deposition has been characterized in a series of studies (San Francisco Estuary Institute 2001.) (Steding and Flegal 2002). The deposition rates cited (3 – 20 $\mu g/m^2$) in those studies are typical of urban settings. Mercury from atmospheric deposition may be particularly susceptible to methylation, since ionic mercury is more readily methylated than mercury bound up by naturally occurring substances in water (Hintelmann and others 2002). Atmospheric deposition of mercury is important to the impact analysis because it is a constant source to the project area that will likely continue at present rates for the duration of the project. Therefore, the AMP will need to focus on actions that can reduce the net conversion of atmospherically deposited mercury to methylmercury.

Atmospheric deposition occurs not only over the project area, but on upstream watersheds that drain into the Bay and the project area. Mercury deposition over urban watersheds is one component of the urban runoff mercury load. Other contributions to urban runoff include improperly disposed mercury-containing electric devices such as fluorescent lights, and other, less well characterized sources. Urban runoff programs are described in the Regulatory Setting Section for Water Quality (Section 3.4.2). Similar to wastewater dischargers, urban runoff programs will be required by the SFRWQCB to conduct special studies, reduce loads, and take other management actions as appropriate (SFRWQCB, 2006). The impacts analysis focuses on areas where project activities are likely to introduce urban stormwater to methylating areas. If this is found to be a problem, the AMP should focus on actions that either divert urban stormwater away from highly methylating areas, or actions to reduce the net methylation of mercury in the project area.

3.1.3 Total mercury: Linking Contaminated Sediments to Water Quality Objectives

Re-mobilization of mercury-contaminated sediments into the water column can lead to exceedance of water quality objectives for mercury. This is because there is a direct relationship between the concentration of suspended sediments in the water column, the concentration of mercury on those suspended sediments, and the concentration of total mercury in the water column.

Project activities can impact attainment of water quality objectives by changing the ambient TSS or by changing the mercury concentration of suspended particles. Table 4 below illustrates how this works. For sediments with the current Baywide ambient concentration of 0.4 µg/g mercury, moderate TSS levels (e.g, 100 – 200 mg/L) will cause exceedance of the 0.025 and 0.051 water quality objectives. The Bay TMDL target for mercury in fine suspended sediments corresponds to a median value of 0.2 µg/g. This condition is expected to be attained over a long time frame (50-100 years). Suspended sediments having 0.2 µg/g mercury will cause exceedance of water quality objectives at somewhat higher TSS concentrations (200 – 300 mg/L). For relatively clean sediments (e.g, 0.1 µg/g Hg or less), TSS levels up to 500 mg/L or more could still be in attainment of water quality objectives. For context, 50 – 100 mg/L TSS is relatively common throughout the regional and project setting, whereas 500 – 1000 mg/L TSS is rare.

Table 4 – Relationship between total mercury in the water column (THg), mercury concentrations in suspended particles (PHg), and total suspended solids (TSS). Relationship makes simplifying assumption that dissolved mercury concentrations are negligible.

Total mercury in the water column (THg = PHg x TSS / 1000)	Conditions	
	PHg (µg/g)	TSS (mg/L)
0.080	0.4	200
0.040	0.4	100
0.060	0.2	300
0.040	0.2	200
0.050	0.1	500
0.030	0.1	300

Project activities can impact attainment of the Bay TMDL target for mercury in sediments, which calls for a 50% reduction in the mass of mercury in the actively resuspended layer. This target is implemented by requiring sediment sources to the Bay to be at or below the current ambient condition of 0.4 ppm. Therefore, project activities which discharge sediments to the Bay having greater than 0.4 ppm have the potential for impacts to the regional setting.

Activities that result in sediments in the project area having mercury concentrations exceeding the LTMS guideline (0.7 ppm) have the potential to cause impacts to the project setting. In this case, because the LTMS guideline is based on an Effective Range-Median (ER-M), the potential impact is toxic effects on benthic communities, not bioaccumulation. The potential for bioaccumulation impacts is discussed separately below.

3.1.4 Methylmercury: Linking Mercury Contamination to the Food Chain

A major concern with mercury pollution in the Bay is the accumulation of methylmercury (MeHg) in biota, particularly at the top of aquatic food webs. Mercury occurs in many forms, but MeHg is the form which poses the highest bioaccumulation risk. MeHg is converted from inorganic mercury (IHg) primarily by the metabolic activity of bacteria, especially sulfate reducing bacteria. Because microbial activity is generally increased in productive wetlands and marshes, restoration of tidal marshes has the potential to increase the net production of MeHg.

It is important to emphasize that the the bioaccumulation of MeHg is the impact of interest, and that net production of MeHg (as opposed to MeHg concentrations) is a key indicator for that impact. A recent discovery of the CALFED mercury studies is that the Sacramento San Joaquin River Delta (Delta) is a net MeHg sink, as evidenced by the MeHg mass balance across the Delta and by the lower concentrations of MeHg in organisms within the Delta compared to peripheral tributary rivers (Foe and others 2003).

Therefore, it is not certain that restoration of tidal marsh will cause a bioaccumulation impact. Rather, increased net MeHg production and bioaccumulation is a risk that will need to be adaptively managed.

Water quality regulators have been struggling for a number of years to develop standards that are based on MeHg in the food chain, rather than THg in the water column. As discussed in the thresholds section above, this analysis of MeHg impacts to the project and regional setting focuses on MeHg in the food chain. This recognizes the latest science supporting water quality standards and moves the evaluation closer to the actual beneficial uses of interest: making fish safe for wildlife and people to eat.

The linkage between IHg and MeHg is complex. Clearly, when no IHg is present, no MeHg can be formed. Increased IHg concentrations in sediments are known to drive increased MeHg production when considering order-of-magnitude increases. For example, comparing ambient Bay sediments to mercury-contaminated sediments in the Guadalupe River watershed, the latter sediments typically also have higher MeHg concentrations.

However, for the range of IHg concentrations in sediments found within the project setting (from 0.1 to 4 ppm) during the initial ISP monitoring, the concentration of IHg did not have a significant correlation with the concentration of MeHg. This impact analysis subsection focuses on bioaccumulation effects, and so considers movement and transport of THg along with other water quality factors that affect net MeHg production and bioaccumulation.

DO is a factor that can affect net MeHg production. Sulfate reducing bacteria that produce MeHg are known to thrive under low oxygen conditions. Low DO also promotes the breakup of oxide surfaces on particles, which can release MeHg into the water column. There are national studies showing the linkage between low DO and elevated MeHg in the water column. Regional studies have showed a similar linkage, and have led to a novel pilot project in the Guadalupe River Watershed that attempts to reduce methylmercury in reservoirs by oxygenating bottom waters. The previous section (Section 2) describes DO as a staircase water quality issue for the regional and project setting. One of the important points of that discussion is that low DO does occur in wetland and marsh habitats. If low DO is found to drive elevated net methylmercury production and bioaccumulation, the increased bioaccumulation would be deemed a significant impact.

Another key factor that affects net MeHg production is the chemical form of the raw material, IHg. Some forms of IHg are more readily acquired by methylating bacteria than other forms. Formation of neutrally charged soluble sulfide complexes is one mechanism that can enhance mercury availability. The amount of available sulfide, in turn, can be affected by iron redox chemistry, which is strongly affected by the nature of vegetative root matter and sediment characteristics. This sets up complex spatial variation in MeHg production rates, with unique pockets of localized enhanced net MeHg production rates. At least two examples are relevant to MeHg impact analysis for the SBSP Restoration Project: the peak of MeHg production rates at optimal sulfate concentrations, and the variation of MeHg production rates depending on vegetation type.

There appears to be an optimum window of sulfate concentrations that maximizes net MeHg production. Too little sulfate prevents sulfate reducing bacteria from thriving and producing sulfide, too much produces so much sulfide that the availability of IHg is diminished (Benoit and others 1998; Gilmour and others 1992; Gilmour and others 1998). When the sulfide concentration is "just right," MeHg production peaks. This is commonly referred by mercury scientists as the "Goldilocks effect" of sulfate stimulation. The Goldilocks effect is important enough to cause State Water Quality Regulators to propose that California's water management practices can have a significant impact on MeHg production in the Delta. For the SBSP Restoration Project, the Goldilocks effect may be significant to conversion of salt ponds to tidal marsh habitat. Creation of estuarine microzones in the window of sulfate concentrations causing the Goldilocks effect could cause enhanced MeHg production.

Net methylation rates are emphasized because the overall release of MeHg reflects the balance of production and destruction of MeHg. MeHg can be degraded by sunlight, with a half-life on the order of days. Microbial activity can greatly accelerate this breakdown. Dissolved oxygen and sulfide are examples of water quality factors that affect production of MeHg. In contrast, microbial community composition affects net MeHg production by influencing both production and degradation.

For this reason, vegetation also affects net MeHg production rates. The type of vegetation influences the microbial communities in the root zone, as well as oxygen transport into the root zone and the consequent sulfide concentrations. For example, pickleweed appears to enhance methylation rates when compared to cordgrass at the Hamilton Air Force Base wetland restoration site, and when compared to mudflats in the Stevens Creek Marsh in the South Bay. This variation in methylation rates with habitat types is the basis for a publicly-funded pilot study of mercury methylation in wetlands (San Francisco Estuary Institute 2006). Because this is an emerging area of research, the effect of tidal marsh elevation and vegetative cover remains a major uncertainty in forecasting the impacts of project alternatives on net MeHg production.

The ecological endpoint that needs to be evaluated is MeHg in the food web. Most of the foregoing discussion has been focused on net MeHg production rates, because net MeHg production is an important factor affecting MeHg bioaccumulation. But the structure of the food web is an also important control on MeHg bioaccumulation.

MeHg bioaccumulation increases at increasing trophic levels and with increasing food web complexity. This is driven by the biomagnification of MeHg. MeHg binds strongly to the sulfur atoms of protein residues. Large organisms eat smaller organisms for their protein, and so retain the associated MeHg. With every step up the food chain, mercury concentrations are found to increase, which is why large predators such as leopard sharks and striped bass have higher mercury concentrations than smaller fish like surf perch. Increasing food web complexity can also increase mercury concentrations at the top of the food web. Adding links to the food web increases the overall biomagnification of MeHg for top level predators. Therefore, project activities that alter ecosystem structure can have significant impacts on mercury uptake by phytoplankton and subsequent accumulation in the food web.

Most of the MeHg biomagnification in the food web occurs in the lower trophic levels (e.g, from direct MeHg uptake by phytoplankton to zooplankton). MeHg concentrations in lower organisms can strongly regulate MeHg concentrations at the top of the food web. Therefore, changes in the community structure or life cycle of lower organisms such as phytoplankton and zooplankton can play a significant role in MeHg bioaccumulation. For example, smaller phytoplankton that have not lived as long will tend to have smaller MeHg concentrations per unit mass, simply because they haven't had as much time to accumulate MeHg as larger organisms of the same species. So phytoplankton blooms which result in large standing stocks of relatively low-MeHg phytoplankton can reduce mercury concentrations at the top of the food web, a phenomenon known as "biodilution." Intense zooplankton grazing pressure which keeps phytoplankton communities "young" can also keep the average MeHg concentration per unit mass low, resulting in lower concentrations in top level predators. These ecosystem effects are complex and difficult to predict, which is why MeHg bioaccumulation impacts will need to be adaptively managed.

3.1.5 South Bay Salt Ponds Regional Concerns

Much of the project area has been separated from the Bay and from source tributaries, so ponds in the northerly project areas (Eden Landing and Ravenswood) are known or expected to have mercury concentrations below the Bay ambient condition. In these situations, it is reasonable to forecast that breaching levees will bring sediments within the pond up to bay ambient mercury concentrations in sediments. The impact analysis focuses on whether this moderate increase of total mercury concentrations (0.05 – 0.15 ppm up to 0.4 ppm) in sediments poses a risk of mercury bioaccumulation impacts for these project areas.

In contrast, ponds in the Alviso Project Area, notably ponds A8 and A12 along Alviso slough, have total mercury concentrations in sediments up to 4 ppm. The cause of this localized increase is deposition of mercury-laden sediments from the Guadalupe River watershed. In this situation, the impact that needs to be analyzed is whether project activities will release these mercury-contaminated sediments to the Bay. Movement of mercury-contaminated sediments from the Guadalupe River into the project area around Alviso already occurs under the ISP and as a result of the Lower Guadalupe River Flood Control Project, and so this should be considered part of the baseline condition.

It should be noted that the discharge of mercury-contaminated sediments to the regional setting would be most likely in the southerly portion of the project, around Alviso, because sediments in northerly project areas are below bay ambient mercury concentrations.

3.1.6 South Bay Salt Ponds Project Area Concerns

A potential benefit of deliberate rather than unintentional breaching is that levees will be maintained to maintain sheltered conditions that encourage evolution from mudflat to marsh habitat. With regard to total mercury concentrations in water, this will have the effect of maintaining lower TSS concentrations in the project area, which can ameliorate the effect of bringing mercury concentrations in sediments up to Bay ambient conditions.

Since restored tidal marshes are generally sediment traps, moving towards the 90:10 Tidal:Pond endpoint of Alternative C would tend to result in movement of mercury loads from the regional setting into the project setting.

While introduction of mercury-contaminated sediments due to levee breaching and flood control design may be of concern in the southerly project areas, it is important to emphasize that preliminary findings show no relationship between IHg and MeHg in the project area. This will continue to be investigated, starting with the baseline and follow-up monitoring initiated by the San Francisco Estuary Institute in Pond A8 of the Alviso Project Area, which is known to have mercury-contaminated sediments originating from the Guadalupe River.

Restoration activities that alter landscape morphology and vegetation will inevitably alter the microbial community composition. Whether this alteration is a positive or negative influence on net MeHg production is unknown. Likewise, restoration activities have the potential to cause low DO within the project setting due to changes in hydraulic residence times, phytoplankton abundance, and other factors. Low DO can enhance net MeHg production. Since the factors affecting DO concentrations are complex, the effect of Alternative B on DO-related increases in net MeHg production are also uncertain.

Conversion of high and medium salinity managed ponds to low salinity managed ponds and tidal marshes has the potential to increase MeHg bioaccumulation. As noted in the Environmental Setting Report, the food webs of medium salinity ponds are simpler than food webs found in low salinity ponds or tidal marshes food webs. As noted above, increasing the number of links in the food web tends to increase MeHg bioaccumulation in top level predators. However, foraging patterns will also change as the mosaic of the SBSP project area evolves, so the net effect on the dietary intake of MeHg by foraging birds and other wildlife is uncertain.

Managed ponds can potentially avoid the intermediate salinity and sulfate concentrations that are optimum for MeHg production by adjusting flow and depths. In contrast, some tidal marsh areas will be restored by breaching flood control levees, which can introduce freshwater that creates estuarine-type microzones of enhanced net MeHg production. This makes increased net MeHg production a potential impact of some of the levee breaching activities. As noted in the discussion above, the relationship between sulfate concentration and net MeHg production is complex, and so the potential for impacts is uncertain.

Restored tidal marshes open new connections to the Bay, and the upland areas could generally be expected to have higher net methylation rates compared to the open Bay, consistent with previous findings summarized in the EIS/R Physical Setting (Section 3.4.1).

3.2 Implications for the AMP

Once restoration activities commence, ongoing monitoring of water quality conditions would be used to detect changes in the transport of mercury-contaminated sediments into and out of the project area. "Triggers" would be established to signal project impacts that are approaching the threshold of significance.

In the project setting, the applicable objectives for total mercury in water (0.051 µg/L and 0.025 µg/L, depending on location) are reasonable triggers for actions. Exceeding these water quality objectives in a project area after a restoration action is completed would trigger the adaptive management actions described below. In the regional setting, the trigger should be if the average concentration of mercury in sediments discharged from the project area is 0.2 ppm or more, and this was not already occurring under baseline conditions.

Adaptive management actions to address exceedance of water column mercury objectives should either reduce TSS in the project setting, reduce the mercury concentrations of the actively re-suspended sediments, or both. For example, bringing in clean fill to cap tidal marsh areas would decrease mercury concentrations in re-suspended sediments. Increasing levee height to decrease wind-driven re-suspension would decrease TSS in the project setting.

In contrast, if exceedance of the water column mercury objectives is not associated with bioaccumulation impacts, the appropriate adaptive management action would be to implement emerging guidance for tissue-based mercury water quality objective in favor of the less protective water column objective for the project area in question. This would have the effect of making the water quality standards implementation for mercury in the project area consistent with that of the Bay, where water column objectives are expected to soon be replaced with tissue-based objectives.

If restoration activities discharge significant amounts of sediment having greater than 0.2 ppm mercury from the project area into the regional setting, the first step would be to determine if this has a significant effect on the average mercury concentration in Bay sediments, and whether the discharge causes a localized bioaccumulation effect. The effect on mercury concentrations in Bay sediments can be evaluated by considering the mass of sediment discharged and average concentration of mercury in that sediment, along with the mass of sediment in the impacted receiving waters. Significant impacts on the mass of mercury in the actively re-suspended layer of the Bay can be offset by removal of mercury-contaminated sediments from the impacted area or nearby areas of the Bay. Bioaccumulation effects can be evaluated using sentinel species monitoring.

It should be noted that the discharge of mercury-contaminated sediments to the regional setting would be most likely in the southerly portion of the project, around Alviso, because sediments in northerly project areas are below bay ambient mercury concentrations. The adaptive management strategy as described is equivalent to a "no net increase" in mercury loads discharged from the project area, in that monitoring and adaptive management actions are coordinated to ensure that loads from the mercury-contaminated southern interface of the Bay are reduced over time. This is consistent with the overall goals of both the

San Francisco Bay mercury TMDL and the Guadalupe River mercury TMDL, which are the overarching water quality regulatory drivers for mercury in the regional setting.

The project's AMP would address the uncertainties regarding the relationship between project activities and State water quality regulations based on total mercury loads and concentrations by monitoring loads, concentrations, and bioaccumulation in sentinel species and adaptively managing the project to ensure that adverse effects do not reach a significant level.

Monitoring sentinel species is the main trigger for adaptive management actions. AMP monitoring should include methylmercury concentrations in water and sediments, as well as special studies of methylmercury production, degradation, and transport, but management actions should be triggered by changes in food web indicators. Since thresholds are defined by tissue concentrations in predators (bird eggs, larger food fish for people, smaller prey fish for wildlife), the triggers should be concentrations in their prey (small fish, benthic invertebrates, zooplankton and phytoplankton). An early implementation action for the AMP should be to develop a suite of sentinel species and associated desirable mercury concentrations that are based on a food web model.

Adaptive Management Process

Following development of sentinel species and trigger levels, baseline levels in sentinel species will be monitored so that changes in response to project activities can be detected. It is important to note that San Francisco Bay is already impacted by mercury, so it would be expected that many sentinel species would exceed desirable levels of mercury for a healthy ecosystem under baseline conditions.

Therefore, adaptive management actions should be triggered when sentinel species mercury concentrations increase significantly, regardless of whether they are over or under desirable levels. The goal of the AMP for mercury is to ensure that over time project actions help, or at least do not hinder, progress towards less mercury in the food chain of both the project and the regional setting.

To attain that goal, monitoring in the individual project areas in the initial phases will make them pilot-scale studies that guide next steps in the AMP. The AMP studies should focus on management questions as outline in the NCAR and the Mercury Technical Memorandum, including:

- Does tidal marsh habitat produce and / or bioaccumulate more MeHg compared to pond habitat?
- Does tidal marsh habitat release significant amounts of MeHg compared to pond habitat?
- What are the design, operation, and management features in ponds and tidal marshes that minimize methylmercury production and bioaccumulation?

The project's AMP will comply with emerging regulations and guidance affecting methylmercury. An important new regulation the AMP should address is the United States Environmental Protection Agency's Draft Guidance for Implementing Methylmercury Criteria (U.S. Environmental Protection

Agency 2006), as well as the State's pending adoption of a fish tissue objective for methylmercury through implementing the Bay Mercury TMDL (San Francisco Bay Regional Water Quality Control Board 2006).

The project's AMP would address the uncertainties regarding the relationship between project activities and State water quality regulations based on total mercury loads and concentrations by monitoring loads, concentrations, and bioaccumulation in sentinel species and adaptively managing the project to ensure that adverse effects do not reach a significant level.

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