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9-14-0731 (POSEIDON WATER)

MAY 9, 2019

APPENDIX

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APPENDIX B

STATE OF CALIFORNIA—NATURAL RESOURCES AGENCY EDMUND G. BROWN, JR., *GOVERNOR*

CALIFORNIA COASTAL COMMISSION

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May 28, 2014

Stan Williams
Poseidon Water
5780 Fleet Street, Suite 140
Carlsbad, CA 92008

Re: Functional Lift Analysis for Determining Mitigation Credit at the South San Diego Bay Salt Ponds

Dear Mr. Williams:

I am writing on behalf of Coastal Commission staff and the Science Advisory Panel (SAP). We have reviewed Poseidon's Technical Memorandum titled "Poseidon Mitigation Credit Analysis, Marine Life Mitigation Plan – Integrated Restoration Plan dated August 26, 2013." The analysis presented in the Technical Memo follows the approach recommended by the SAP to calculate the total functional lift provided by restoration of Pond 15 as the average of the lift to vegetation, macro-invertebrates, fish and birds. Each category is equally weighted with a maximum value of 1. Staff and SAP agree with the Technical Memo's conclusions that the functional lift to vegetation, macro-invertebrates and fish is 100% or a value of 1 for each category. However, we have some concerns about the analysis for birds. To address these concerns, the SAP conducted a supplementary analysis that calculated the functional lift for birds for five different cases:

Case 1: Uses the values proposed by Poseidon, including Pond 15 bird values (not including birds on the berms) and reference values based on the mean values from the three reference sites.
Case 2: Uses Pond 15 bird abundance values that include birds on the berms
Case 3: Uses Case 1 values for Pond 15 and values from Carpinteria Salt Marsh for the reference values
Case 4: Uses Case 2 values for Pond 15 and values from Carpinteria Salt Marsh for the reference values
Case 5: Uses 90 percent confidence values for Pond 15 and values from Carpinteria Salt Marsh for the reference values. Confidence values were calculated using the data presented in Appendix B of the 8/26/13 memo

Cases 3-5 were included because they represent the assessment that will occur post-construction to determine compliance with the restoration permit's requirements. Once the mitigation project is constructed, performance will be determined by comparing the mitigation wetland to reference wetlands. The mitigation project will be judged similar if the performance (e.g. abundance and diversity) of the mitigation wetland is no worse than the average value for the lowest performing reference wetland.

In this case study, data from Carpinteria Salt Marsh is used as a proxy for the lowest performing reference site. These data, along with data for Mugu Lagoon, were collected as part of the SONGS mitigation program and were used in this analysis to represent the post-construction condition. Of the two reference sites, Carpinteria Salt Marsh had the lowest values for bird abundance and diversity, and thus, these values are used in the functional lift analysis.

Results of this analysis show that the "lift" for birds ranges from +0.43 to -0.18 (see Table 1 below). The range of results, including both positive and negative lift values, illustrates the significant level of variability and uncertainty associated with this analysis for birds. Based on these values, staff and the SAP have concluded that there is no basis for distinguishing between a positive and a negative lift contribution for birds. Thus, we assign a lift of 0 for birds, leading to an overall functional lift of 0.75. Assigning a lift of 0 for birds means that Poseidon will not get credit for improving habitat at the salt ponds for birds. Although this analysis does indicate that a negative lift with respect to birds is a possible outcome, this analysis is limited to the immediate restoration site. When viewed in a larger spatial context, such as the South San Diego Bay Refuge, we would not anticipate a negative effect on birds from this proposed mitigation project.

Thus, based on the SAP's analysis, Poseidon can receive 0.75 acres of mitigation credit for every acre of Pond 15 that is fully restored to tidal salt marsh, corresponding to a functional lift of 0.75. If you have any questions or would like to discuss this determination further, please call me at 415-396-9708.

Sincerely,

Kate Huckelbridge
Environmental Scientist
Energy, Ocean Resources and Federal Consistency Division

Table 1

Case #	Pond 15		reference site		Diversity	Abundance	subtotal	Overall Lift
	Species metric	abundance metric	Species metric	abundance metric				
case 1	15.23	0.0016	31.79	0.0024	0.52	0.33	0.43	0.86
case 2	15.23	0.0021	31.79	0.0024	0.52	0.13	0.32	0.83
case 3	15.23	0.0016	25.11	0.0008	0.39	-0.50	-0.05	0.74
case 4	15.23	0.0021	25.11	0.0008	0.39	-0.62	-0.11	0.72
case 5	16.41	0.002647	25.11	0.0008	0.35	-0.70	-0.18	0.71

TECHNICAL MEMORANDUM

POSEIDON MITIGATION CREDIT ANALYSIS
MARINE LIFE MITIGATION PLAN—INTEGRATED RESTORATION PLAN

August 26, 2013

PURPOSE

The purpose of this memorandum is to provide documentation on the calculation of the mitigation credit associated with the restoration of solar evaporator Pond 15 to intertidal wetland habitat as part of the Integrated Restoration Plan (IRP) proposed by Poseidon Resources to comply with the Marine Life Mitigation Plan (MLMP).

BACKGROUND

An element under consideration by the California Coastal Commission (Commission) is the amount of mitigation credit that will be allotted to Poseidon Resources from the restoration of the salt evaporator Pond 15 on the US Fish and Wildlife San Diego Bay National Wildlife Refuge. The salt evaporator ponds are non-tidal basins containing brines of varying levels of salinity and are used as part of the solar salt production system operated by the South Bay Salt Works. The Salt Works takes in bay water to supply the source of the salt, and through a process of sequential evaporation, produces crystalline salt at the plant site. The salt evaporator ponds do not support tidal wetland vegetation and since salinities in the ponds quickly exceed those tolerable to marine life, do not support fish or invertebrates typical or similar to that found in San Diego Bay. The restoration of these basins to intertidal habitats will likely improve the diversity and productivity of these ponds and provide increased fish production to San Diego Bay.

The success of the MLMP is tied to a number of performance criteria that have been set by the Commission's Coastal Development Permit. Following construction, the IRP will be evaluated by its ability to meet these performance criteria, some of which are related to reference wetlands that will be sampled in accordance with the Mitigation Monitoring Plan prepared by the Commission. Among those criteria, one relates to biological communities and one relates to vegetation:

> Biological communities: The total densities and species densities of fish, macro-invertebrates (> 3 mm), and birds shall be similar to the densities and species densities in similar habitats in the reference wetlands.

Vegetation: The proportion of total vegetation cover and open space in the marsh shall be similar to those proportions found in the reference sites.

One way to establish the change that will be expected from the implementation of the IRP is to compare the conditions within the salt evaporator ponds in the before condition to the required outcome under the MLMP performance standards. Because the monitoring effort by the Commission is already underway for the San Dieguito Restoration Project, an initial sampling of the reference sites has been conducted that can provide an initial dataset for the after condition.

The methodology for the final calculation is based on the following table:

	Vegetation	Macro-invertebrates		Birds		Fish	
	Percent cover	Species (# spp/m^2)	Abundance (#/m^2)	Species (# spp)	Abundance (#/m^2)	Species (# spp/m^2)	Abundance (#/m^2)
Before Pond 12	VA	MA	MC	BA	BC	FA	FC
After (Reference)	VB	MB	MD	BB	BD	FB	FD

And the calculation of the change from the before condition to a project in compliance with the reference wetlands is described by the Functional Lift Index (FLI):

$$FLI = \frac{FLI_V + FLI_M + FLI_B + FLI_F}{4}$$

Where:

$$FLI_V = [(VB-VA)/VB]$$

$$FLI_M = \frac{[(MB-MA)/MB] + [(MD-MC)/MD]}{2}$$

$$FLI_B = \frac{[(BB-BA)/BB] + [(BD-BC)/BD]}{2}$$

$$FLI_F = \frac{[(FB-FA)/FB] + [(FD-FC)/FD]}{2}$$

Each of the four component FLI's will be between 0 and 1 with 0 representing no improvement and 1 representing 100% improvement. The value of the composite FLI equally weighted between the four components will also be between 0 and 1 with 0 representing no improvement and 1 representing 100% improvement.

STUDY AREA

The salt ponds consist of shallow, open water cells with different salinity levels. As the water flows through the pond system, the ponds become more saline until near the end of the process, sodium chloride and other salts precipitate out and either form crystals or stay in a heavy brine solution. These products are then harvested, processed, and sold for industrial, commercial, and residential uses. Pond 15 is located in the northeast quadrant of the South Bay Salt Works complex, northeast of Ponds 13 and 14, west of Pond 28 and immediately southeast of San Diego Bay. Pond 15 is approximately 90 acres in size. San Diego Bay water flows through Ponds 12, 13 and 14 before entering Pond 15.

Briefly, the ponds are divided into four categories based on their salinity levels with the lower salinity ponds referred to as primary ponds, followed by secondary ponds, pickling ponds, and crystallizer ponds, which have the highest salinity levels.

US Fish and Wildlife Service salinity readings within salt ponds based on measurements taken between 1996 and 2002.

Pond Number	Salinity Range	Acreage
12	44.0 – 96.0	101
13	57 – 102.5	67
14	67.8 - 122	45
15	71.3 – 128.5	90

Since the restoration of the Western Salt Ponds, the solar salt production operator may have changed the sequence of how the ponds transfer brines and therefore the salinities may have changed. Pond 12 is currently serving as an intake pond with brines transferred to higher numbered ponds in the system. Recent measurements taken by Chris Nordby confirm the higher salinities of these evaporators.

At present, it is anticipated that Pond 15 will be the primary pond converted to tidal action that will provide mitigation credits to Poseidon and will be subject to the performance standards listed above.

To address concerns raised regarding Pond 15 may have a reduced salinity after bay water intake, measurements of salinity were taken on March 28, 2013 just after the South Bay Salt Works took in San Diego bay water at Pond 12 on March 25, 26 and 27, 2013. The measurements show that three days of intake resulted in a slight lowering of the salinity in Pond 12 (see Sample #3 and #6 in the table below), the salinity of Ponds 14 and 15 did not decrease (see Samples #7-10).

Salinity and Baume at Approximately 1-foot Depth Evaporators 12 - 15, on January 22 and March28, 2013.

Sample/Location	Salinity (ppt)	
	January 22, 2013 Before Bay Water Intake	March 28, 2013 After Bay Water Intake
#1. Pond 12 at Siphon Lock	98	No Sample
#2. Otay River at Tide Gate	30	36
#3. Pond 12 at Tide Gate	100	74
#4. Pond 13 SW Corner	100	No Sample
#5. Pond 13 Mid South Levee	120	No Sample
#6. Pond 12 North Levee	98	80; 85
#7. Pond 14 NW Corner	100	120
#8. Pond 14 at Pond 15 Levee	110	112; 124
#9. Pond 15 NW Levee	108	130
#10. Pond 15 SW Levee	120	130
#11. Pond 15 Mid NW Levee	No Sample	110

Vegetation

Before condition

In general, the salt evaporator ponds have been constructed to support a water depth of 3-5 feet and water levels are maintained within the evaporators such that intertidal plants are not able to survive. Salinities are too high to support submerged vascular plants such as eelgrass and no eelgrass has been noted within the evaporators. In addition, the salt pond levees themselves are barren for the most part. The lack of vegetation has been attributed to the result of on-going maintenance as well as the high salinities that exist in the vicinity of the levees. Therefore, the before condition for the percent cover of vegetation is zero.

After condition

Those portions of the reference wetlands that are within the intertidal range suitable for tidal vegetation would support a variety of halophytic vascular plants with a cover of over 90%.

Vegetation calculation

$$FLI_V = (90-0)/90 = 1$$

No intertidal habitat currently exists in Pond 15 and the portions of the restoration to be created as marsh habitat will be of suitable elevation to support tidal vegetation. With the requirement that the portions to be created as tidal marsh meet the cover percentage of the reference wetlands, the vegetation calculation results in a 100% increase, or value of 1.

Macro-Invertebrates

Before condition

The salt evaporators vary in salinity and temperature throughout the year and salinities in the upper 80-120 ppt range are expected with lower salinities and more freshwater input occurring in the winter. Recent winter sampling within Pond 15 shows salinities from 108 to 120 ppt, marking the lower of the salinities expected in the Pond due to winter precipitation. Based on data from San Francisco Bay solar evaporators within this salinity range, most of the macro-invertebrates associated with tidal wetland ecosystems will be absent and overall densities and species diversity will be close to zero. In addition, biomass density and productivity will be extremely reduced compared to natural tidal wetland ecosystems.

Brief macro-invertebrate sampling surveys were conducted on March 12, 2013 by Chris Nordby in Pond 15 with Brian Collins, USFWS Refuge Manager, in attendance. Benthic cores (15cm diameter by 20cm depth, and sieved through 3mm mesh) were taken to represent Pond 15 macro-invertebrate community: six cores were taken near the junction of Pond 15 and 14; six cores at the northeastern corner; and four cores in the southeastern corner. Although brine flies and brine shrimp were present, there were no marine or estuarine invertebrate infauna nor epifauna present. The performance criteria that have been set by the Coastal Development Permit do not include insects and planktonic invertebrates. Therefore, the before condition for the macro-invertebrates is zero macro-invertebrates per m^2 and zero species per m^2.

Brine flies and brine shrimp may provide forage for animals such as fishes and birds; however, a separate suite of planktonic and shoreline species more diverse and abundant will replace those high-salt-tolerant smaller invertebrate species and result in a higher functioning ecosystem in the restored condition. Further consideration of brine shrimp and flies is discussed under uncertainties described below.

After condition

We used the reference wetland surveys being collected by the Commission as part of their monitoring of the San Dieguito Restoration Project to provide data for the post-restoration condition for Pond 15. The benthic cores (10cm diameter by 50cm deep) sieved through 3mm mesh and collected both in the main channels and tidal creeks were used. The reference wetlands are densely populated by a large diversity and high abundance of macro-invertebrates indicative of highly functioning intertidal marshes.

The average sample abundance and average species values among all samples from the two reference marshes, Carpinteria Salt Marsh and Mugu Lagoon were calculated. The overall species density was 0.0088 species per cm^2. The overall average abundance was 0.0197 individuals per m^2. Refer to Appendix A for reference marsh data.

Macro-Invertebrate calculation

$$FLI_M = \frac{(0.0088-0)/0.0088 + (0.0197-0)/0.01970}{2} = 1.0$$

Birds

Before condition

Pond 15 is composed of a shallow pond with an artificially maintained water level. As noted above, vegetation within the pond is near absent and provides little habitat for birds. The ponds do provide foraging and loafing habitat for a number of shorebirds and wading birds.

For bird abundances and species diversity of Pond 15, we used the SWIA Surveys (SDNHM and ARA 2011), which were conducted monthly from March 2010 to September 2010 and SWIA Surveys (SDNHM and ARA, in prep.) monthly from September 2010 through January 2013. Surveys included the shallow water habitat up to the high tide line. Both wetland and aerial counts were used. The berms and uplands of the ponds were surveyed as well, but are not included here because the restoration will be designed to preserve these habitats (and include appropriate buffers). Surveys were conducted using the methods employed in the multi-year bay-wide survey of avian species (Tierra Data Incorporated 2009). The timing of surveys was planned to maximize avian use set during high tide for the interior ponds during which shorebird roost flocks would be present. Abundance values were calculated by dividing the total number of birds observed by the 90 acres of Pond 15. Species values used were calculated as the average number of species observed per survey at Pond 15.

The average number of species within Pond 15 was 15.23. The average abundance within Pond 15 was 0.0016 individuals per m^2. Refer to Appendix B for Pond 15 before condition marsh data. Birds that were roosting on levees were included in bird species totals as they also were observed flying Pond 15. However, since levee habitat were not included in the after condition observations at reference sites and roosting birds can vary considerably from year to year, they were not included in the density numbers. This is discussed further within the uncertainties section of this report below.

After condition

We used the San Dieguito Restoration Project reference data to provide a reference for the post-restoration condition for Pond 15. The average abundance and species values among the three reference sites, Carpinteria Salt Marsh, Mugu Lagoon, and Tijuana Estuary were used. The reference conditions average species per survey was 31.79 species. The average abundance among the three reference sites was 0.0024 individuals per m^2. Refer to Appendix A for reference marsh data.

Bird FLI calculation

$$FLI_B = \frac{(31.79-15.23)/31.79 + (0.0024-0.0016)/0.0024}{2} = 0.43$$

Fish

Before condition

The salt evaporators vary in salinity and temperature throughout the year and salinities above 100 ppt are expected with lower salinities and more freshwater input occurring in the winter. Recent winter sampling within Pond 15 shows salinities from 108-120 ppt (2.5 to 3 times ocean salinity), marking the lower of the salinities expected in the Pond. Although some fishes may survive temporarily in these high salinities, none are expected to be significantly productive (grow or reproduce) and most will die. Because the fishes indicative of a natural tidally-influenced wetland ecosystem will be absent, the fish species abundance and species values in Pond 15 are zero.

The before condition is zero fish per m^2 and zero species per m^2.

After condition

The reference wetlands are densely populated by a large diversity and high abundance of fishes indicative of highly functioning intertidal marshes that eventually export productivity to the bay-wide system rather than act as a sink. Reference data collected in 2012 from the seine net sampling reveal an average species value of 1.95 fish species per m^2 between the two reference sites (Carpinteria Salt Marsh and Mugu Lagoon). The average abundance among the two reference sites was 7.59 fishes per m^2. Refer to Appendix A for reference marsh data.

Fish FLI calculation

$$FLI_F = \frac{(1.95 - 0)/1.95 + (7.59 - 0)/7.59}{2} = 1.0$$

Combined Total FLI Calculation

As previously stated, as a combined and equally weighted average among the four different components (i.e., vegetation, invertebrates, birds, and fish) the following equation can be used (also see Appendix C).
:

$$FLI = \frac{FLI_V + FLI_M + FLI_B + FLI_F}{4}$$

$$FLI = \frac{1.0 + 1.0 + 0.43 + 1.0}{4} = \mathbf{0.86}$$

UNCERTAINTIES

In discussion with the Scientific Advisory Panel to the Poseidon MLMP, three areas of uncertainty related to functional lift may affect the final outcome of the functional lift calculation:

<u>Variation in the bird usage from year to year</u>: Currently, the reference marsh data that were used for the bird evaluation are based on an average of total bird species for all reference marshes and is based on the observations made over one year. The number for average total species as provided by the Commission contract scientists was 31.79 as compared to 15.23 in Pond 15. Bird species numbers may fluctuate between years due to large scale effects related to migratory patterns and climatic factors; however, this is expected to be reflected across all the reference wetlands. On the other hand, there will be variation between reference wetlands that are unaccounted for by regional variation. For the purposes of evaluating the effect of this variation, the change in bird functional lift was tested by using the highest and lowest bird species number from the reference wetlands as collected in 2012:

Condition	Average Total Bird Species Observed/Year	Bird Functional Lift	Overall Functional Lift
Average of all Ref marsh	31.79	0.43	0.86
Lowest	25.11	0.36	0.84
Highest	38.16	0.47	0.87

The change in the functional lift for birds alone changed by 0.11 units; but because it is only one component of the overall functional lift, it only has an influence of 0.03 units when combined with the other values. This represents an approximate uncertainty of 3.4 percent.

<u>The presence of brine flies and brine shrimp within Pond 15 is unaccounted for by the analysis.</u> These two species of invertebrates (one an insect and the other a planktonic crustacean) are found in high salinity ponds. Brine flies lay eggs which hatch into larvae that are found on the pond bottom and edges. Brine shrimp produce cysts that result in planktonic organisms in the water column, with adults having a life span of 50-70 days. Both occur in very high numbers, at least seasonally, when temperature and salinity conditions are optimal (60 to 100 ppt). Both species forage on algae and the abundance of food also affects their population sizes. Brine shrimp are consumed by birds that are capable of accessing them such as eared grebes in the water column. Brine flies larvae are the most accessible life history stages for consumption by birds if in shallow enough water where they can be accessed, and at the adult stage where they are consumed off the surface of the water or from the air.

Neither occur within reference marshes and the reference marsh sampling protocols do not evaluate insects or planktonic organisms. Nonetheless, brine shrimp and brine flies can be important food sources for birds, particularly shorebirds and some water bird specialists. Studies summarized by Warnock (undated) found that brine flies had higher nutritional value than brine shrimp and were a significant food source for some bird species during migration or periods when raising young. While the total bird density may be reflective of the occurrence of these two invertebrates in the salt ponds, this relationship tends to be poor due to other factors that independently affect birds and brine shrimp and flies such as seasonality and migration patterns, interannual variation in factors affecting growth, and pond salinity as controlled by the salt producer (Warnock, undated).

There is no information on the relative density of either brine flies or brine shrimp within saline salt ponds of San Diego Bay. Primary production of their primary food source, single celled algae, influences adult brine shrimp densities in various hypersaline lakes. The lower production in the Great Salt Lake supports peak summer adult densities of brine shrimp near 3/l; whereas the more productive Mono Lake has densities of 6-8/l in the pelagic region (Conte et al., 1988). Density of brine shrimp tend to be highest in deeper water (Verkuil et al, 1993). The density levels of 6-8/l were hypothesized by Belovsky et al (2009) as needed to provide sufficient food energy to migrating eared grebes in the Great Salt Lake. Data on brine fly density, which do tend to be concentrated along pond edges, is not readily available. The few studies that are available provide estimates in dry bio/mass, not in density of individuals.

The US Fish and Wildlife Service (Service) prepared a Comprehensive Conservation Plan and Environmental Impact Statement for the proposed restoration of the South San Diego Bay Units, including Pond 15. The Service established an objective for the restoration of salt ponds to assure that there would be stable source of brine invertebrates for migratory birds (Objective 3.2 for the South San Diego Bay Unit). The EIS for the IRP proposed by Poseidon and the Service will evaluate the impact, if any, due to the change from salt pond to tidal marsh and provide for mitigation should any significant impact be determined. According to Brian Collins, Refuge Manager, the goal of the overall restoration is to assure that the important functions that are currently present within the salt works are not reduced to the point of causing impacts to those species which may depend upon some elements of the salt production process. On the other hand, the Refuge does intend to replace and restore habitats lost to historic regional development that will have a positive supporting impact to endangered species such as light-footed clapper rail and Belding's savannah sparrow.

The presence of birds on the levees surrounding Pond 15 were not included in the analysis of bird use in the before condition. The vast majority of the birds that use the berms are migratory sea birds. These migratory sea birds, mostly various tern species, nest on the berms and forage in the bay and open ocean. No one is quite sure why they use the salt pond berms, but the theory is that they "feel safe" since they have an unobstructed view of potential predators and the salt works keeps people out of the area. These species prefer the interior levees of Pond 13 and a few other isolated spots on other levees which is why the project has proposed Pond 15 as there is no history of sea birds nesting on any Pond 15 levees. Based on 15 years of annual monitoring data, the levees around pond 15 are not a preferred nesting area for most of the breeding avian community of the salt works. The USFWS Refuge will be concerned that restoration of the salt ponds does not impact or affect these nesting sea birds and the IRP expects to preserve nesting habitat for these species on these berms.

Nesting by migratory sea birds at the salt works is highly variable and dependent upon factors that may not be linked to restoration or other changes in physical habitat. Ocean water temperatures and prey availability are certainly factors. For example, in 2010 there were approximately 18,000 nesting elegant terns at the salt works with perhaps up to 50,000 adults and fledglings at one point in time. The following year in 2011 and prior to any work on the western salt ponds, there were less than 150 nesting elegant terns. They all apparently nested elsewhere such as LA/Long Beach harbor areas, the Bolsa Chica Ecological Reserve or Isla Rasa in the Gulf of California where 90% of the nesting population can be found in most years. Had the western salt ponds project been constructed between the 2010 and 2011 nesting seasons it could have been argued, incorrectly, that the restoration project was the cause, or a causal factor, of the drastic drop in elegant tern nesting. Therefore, the functional lift should be calculated based on the changes in wetland birds which actually use the wetland habitats, either open water or intertidal, and not on upland birds whose habitat won't be directly affected and who are subjected to other environmental drivers. This is consistent with the reference bird data that is being taken within wetland habitats of the reference wetlands rather than in upland areas.

Because berms will be retained, and additional nesting habitat provided within the project design, it is anticipated that the project will not have a significant impact on nesting birds or the use of the berms around the restored wetland. It is possible that with less open water near the levees, that they will become less desirable to roosting species. However, in the San Francisco Bay tidal wetland restoration project, there has been substantial use of the former salt pond levees and created island habitats by shorebirds though often the consistency of that use varies with size, shape, location, and predator pressure on the nesting species (Moskal *et. al*. 2013).

As a measure of the uncertainty associated with bird use associated with the levees, the birds using the levees were included in the analysis. The density of birds associated with Pond 15 changes from 0.0016 to 0.0021 individuals per m^2 resulting in a decrease in the functional lift associated with birds from 0.43 to 0.32. This reduces the overall functional lift by 3 percentage points; or 3.4%.

The restored tidal wetlands within the salt ponds are expected to have wide ranging benefits to invertebrates, fish, birds, and overall productivity within the southern portion of San Diego Bay. In San Francisco Bay, for example, monthly abundance of most groups and species of birds was higher within the restoration project in ponds that were subject to interim restoration than in commercial salt ponds and was most pronounced for ducks, shorebirds, and their representative species (Athearn *et. al*. 2012). The project will need to meet requirements as set forth by the Commission and to meet both absolute and relative standards. However, given the uncertainties discussed above as it relates to the functional lift calculation, it is expected that the functional lift may vary between the predicted 0.86 by 6-7% or 0.80 to 0.92.

CONCLUSION

Based on the data available for existing conditions, the mitigation credit that is appropriate to the IRP is 0.86 acres of credit for every 1 acre of salt evaporator restored to intertidal marsh habitat.

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Appendix A – Reference Marsh Data

Reference Macro-Invertebrate Species Density							
		Main Channel		Tidal Creek		Main Channel & Tidal Creek	
Lagoon	Month	Mean (# spp/m^2)	n	Mean (# spp/m^2)	n	Mean (# spp/m^2)	n
CSM	June	0.0135881	6	0.0096249	6	0.0116065	12
	August	0.0120311	6	0.0106157	6	0.0113234	12
CSM mean		0.0128096	12	0.0101203	12	0.011465	24
PML	June	0.0042463	6	0.008351	6	0.0062987	12
	August	0.0038217	6	0.0084926	6	0.0061571	12
PML mean		0.004034	12	0.0084218	12	0.0062279	24
Grand mean		0.0084218	24	0.0092711	24	**0.0088464**	48

Reference Macro-Invertebrate Abundance							
		Main Channel		Tidal Creek		Main Channel & Tidal Creek	
Lagoon	Month	Mean (# ind/m^2)	n	Mean (# ind/m^2)	n	Mean (# ind/m^2)	n
CSM	June	0.0271762	6	0.0225053	6	0.0248408	12
	August	0.019816	6	0.0222222	6	0.0210191	12
CSM mean		0.0234961	12	0.0223638	12	0.0229299	24
PML	June	0.0058033	6	0.0127388	6	0.009271	12
	August	0.0268931	6	0.0203822	6	0.0236376	12
PML mean		0.0163482	12	0.0165605	12	0.0164543	24
Grand Mean		0.0199221	24	0.0194621	24	**0.0196921**	48

Reference Bird Data		
Lagoon	Abundance (# ind /m2)	Number Species per Survey
CSM	0.0008	
PML	0.0052	
TJE	0.0012	
Average	**0.0024**	**31.79**

Reference Fish Data (Seine Only)		
lagoon	Abundance (# ind /m2)	Average Species per Survey
CSM	8.38	2.57
PML	6.79	1.33
TJE	no data	no data
Average	**7.59**	**1.95**

Appendix B – Before Condition Marsh Data (Pond 15)

Before Condition Bird Data - Pond 15				
Survey Date	**Number of Species per Survey**	**Number Wetland and Aerial Birds**	**Abundance (# ind / acre)**	**Abundance (# ind/m^2)**
3/27/2010	18	114	1.27	0.0003
4/24/2010	19	465	5.17	0.0013
4/25/2010	21	306	3.40	0.0008
5/24/2010	17	184	2.04	0.0005
6/21/2010	14	72	0.80	0.0002
7/14/2010	18	2328	25.87	0.0064
8/11/2010	17	1297	14.41	0.0036
8/14/2010	13	253	2.81	0.0007
9/8/2010	23	1204	13.38	0.0033
10/11/2011	15	507	5.63	0.0014
11/8/2011	17	1628	18.09	0.0045
11/10/2011	19	2468	27.42	0.0068
12/10/2011	13	941	10.46	0.0026
1/10/2012	10	147	1.63	0.0004
2/6/2012	11	256	2.84	0.0007
2/6/2012	15	233	2.59	0.0006
3/21/2012	10	90	1.00	0.0002
4/4/2012	11	82	0.91	0.0002
4/6/2012	14	128	1.42	0.0004
5/4/2012	14	117	1.30	0.0003
6/4/2012	12	79	0.88	0.0002
7/8/2012	18	435	4.83	0.0012
8/18/2012	12	107	1.19	0.0003
8/19/2012	11	191	2.12	0.0005
9/16/2012	16	529	5.88	0.0015
10/15/2012	14	717	7.97	0.0020
11/14/2012	8	418	4.64	0.0011
11/15/2012	21	591	6.57	0.0016
12/14/2012	15	409	4.54	0.0011
1/12/2013	21	1153	12.81	0.0032
Average	**15.23**	581.63	6.46	**0.0016**

Appendix C – Functional Index Calculation

Functional Index Calculation

	Pond 15 % cover				Ref % cover				RATIO
	A				*B*				
Vegetation	0				90				**1**
	Species				**Abundance**				
	Before - Pond 15		**Reference**		**Before - Pond 15**		**Reference**		
Component	***A***	***Units***	***B***	***Units***	***C***	***Units***	***D***	***Units***	
Macro-Invertebrates	0	#spp/cm^2	0.0088	#spp/cm^2	0	#ind/cm^2	0.020	#ind/cm^2	**1.00**
Birds	15.23	# spp	31.79	# spp	0.0016	#ind/m^2	0.0024	#ind/m^2	**0.43**
Fish	0	#spp/m^2	1.95	#spp/m^2	0	#ind/m^2	7.59	#ind/m^2	**1.00**
Total									**0.86**

APPENDIX C

Sensitivity Analysis of Potential DDT Deposition in the Otay River Estuary Restoration Plan (ORERP) Post-100 Year and 50-Year Floods

by:

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ABSTRACT: This analysis focuses on an assessment of potential impacts on the ORERP from erosion of soils containing DDT by the 100 yr. flood, with additional analysis of the 50-year flood impacts. Scour potential associated with the 100-year flood on the ORERP have been evaluated in a companion study (Everest 2014). The present analysis evaluates the effects associated with erosion of soils containing DDT from the floodplain under the 100-year flood event that may release DDT to downstream portions of the project. Because the duration of the 100-yr flood is only 24 hours, it was assumed that tidal exchange will quickly re-establish flow dominance post-flood; and that the transport and settling dynamics of potentially contaminated silts and clays will be driven and limited by the tidal hydraulics and tidal residence times.

A sensitivity analysis was developed based on a parameter sweep of the amounts of soils containing DDT that might be eroded by the 100-year flood. Sediment coring data indicates that the depth of erosion in the area of soils containing DDT might vary between 1 ft. and 3 ft.; and the concentrations of DDT in the eroded soils could vary between 790 µg/kg and 310 µg/kg, depending on the depth of erosion. These eroded soils containing DDT could mix with as much as 438,000 cubic yards (cy) of "clean" (*i.e.*, assumed to be free of DDT) fine-grained sediments from the Otay River watershed below the Savage Dam; but that estimate was based on a surrogate watershed (Buena Vista Creek) for which more complete sediment yield data was available. Based on the uncertainties of applying that surrogate analysis to the Otay River watershed, it is sensible to consider the sensitivity of the final outcome to omitting consideration of that flux of what is believed to be "clean" sediments from upstream sources by eliminating the dilution effects that blending with clean fines exerts on DDT concentrations during the post-flood deposition. From this assessment of the possible sediment erosion input assumptions, a sensitivity analysis **is provided** for the post 100-year flood DDT deposition that is based on

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erosion fluxes from three erosion depths (1 ft., 2 ft. and 3 ft.) in the floodplain that are each combined with two possible fluxes of "clean" fines (0 cy and 438,000 cy) from the watershed below the Savage Dam. In addition, the biological risk assessment of these six possible deposition scenarios also considers bioturbation exposures occurring post-flood within the top 20 mm, 40 mm and top 80 mm of the muddy sediments in the tidal basins of the ORERP. This range of parameters yields a sensitivity analysis with 18 possible outcomes including worst-case scenarios.

It was found that the post 100-year flood will result in the deposition of less than 1 mm to as much as 8 mm of partially consolidated mud in the tidal basins of either restoration alternative that will have an average dry bulk DDT concentration of 42 μ g/kg to 790 μ g/kg, depending on the particular scenario. The DDT concentrations in the muds deposited in the ORERP can range as high as 310 μ g/kg to 790 μ g/kg, but the deposition thicknesses of these scenarios reduce to only fractions of a millimeter once these muds become consolidated. Using a depth-proportional exposure approach, and assuming all exposure occurs within the top 20 mm under worst-case, we calculated that the DDT concentration experienced by the benthic biota would range from approximately 13 μ g/kg to 29 μ g/kg initially, and would decrease with compaction and consolidation to a final 20 mm-based dry bulk concentration of 4.2 μ g/kg to 7.9 μ g/kg. The controlling variable in worst-case exposure determination is the total mass of DDT in the post-flood sediment deposition, which is maximized by the scenarios in which the largest volumes of DDT-contaminated sediments that were eroded, (*i.e.*, the 3ft. erosion depth scenarios) in the absence of mixing with additional sediments from upstream sources. Worst case exposures were found to be relatively insensitive to the dilution provided by sediments from upstream sources, while the mixing depth of bioturbation has a much stronger influence. DDT concentrations experienced by the benthic biota under worst-case are reduced 2 to 4 fold when bioturbation extends over the top 40 mm or top 80 mm of the muddy sediments in the tidal basins of the ORERP. The depth of bioturbation will be determined by the species that ultimately colonize the tidal basins, but we would not expect that to be less then approximately 20 mm.

Upon advice from the California Coastal Commission Science Advisory Panel, the above analysis was repeated for the 50-yr flood, to assure the most extreme potential DDT exposure outcomes have been modeled. The DDT deposition results for the 50-yr flood were found to be within the range of those for the 100-yr flood. The DDT concentrations in the muds deposited in the ORERP post 50-year flood can range as high as 111 μ g/kg to 790 μ g/kg, and again, deposition thicknesses of these scenarios reduce to only fractions of a millimeter once these muds become consolidated. Using a depth-proportional exposure approach within the top 20 mm under worst-case, we calculated that the DDT concentration experienced by the benthic biota would range from approximately 12 μ g/kg to 26 μ g/kg initially after the 50-year flood, and would decrease with compaction and consolidation to a final 20 mm-based dry bulk concentration of 4.0 μ g/kg to 7.1 μ g/kg.

Relative to impacts on the benthic organisms as the prey base, the maximum short-term DDT concentrations in the post-flood deposition fall between the ER-L and ER-M values. Thus, we would expect that impacts on benthic organisms could occur occasionally during the short-term. Given the likelihood of effects combined with the short-term nature of this condition, population level impacts are expected to be limited in nature and extent. Once these post-flood muddy deposits have compacted and consolidated, the DDT concentrations in the top 20 mm of muddy sediment are very close to the ER-L, and even lower for the top 40 mm and top 80 mm of sediment; so that negative effects are expected to be rare. This condition is not likely to have a measurable effect on the prey base for aquatic-dependent species.

In regards to the aquatic-dependent birds' exposures to DDT in prey , comparison of the 20 mm-based DDT concentrations to screening levels indicates that these concentrations fall within the range of highest and lowest NOAELs. Given the species known to be the most sensitive are pelicans and cormorants, (which are very closely related, and our target species are not members of groups believed to be particularly sensitive), impacts on aquatic-dependent birds are unlikely to result from the anticipated deposition of sediments in the ORERP following either a 100-year or 50-year flood event.

Upon advice from the California Coastal Commission Science Advisory Panel, the above analysis was repeated for the 100-yr flood in the absence of the ORERP (*i.e.,* No Project Alternative). Those results are given in APPENDIX-B. The DDT deposition results in Ponds 10 & 11 of the No Project Alternative were found to be within the range of those for the ORERP tidal basins post 100-yr flood, so that the above conclusions on potential flood-induced DDT impacts to the existing wetlands ecology are upheld; and it can be concluded that the ORERP does not increase the risk of exposure of wetland ecology to DDT, a risk that exists with or without the project.

Sensitivity Analysis of Potential DDT Deposition in the Otay River Estuary Restoration Plan (ORERP) Post-100 Year Flood

Scott A. Jenkins, Ph. D., Ying Poon, D.Sc., Catherine Zeeman, Ph.D., and Carol Roberts

1.0) Introduction:

In this study we estimate rates of fine-grained sediment deposition in the tidal basins of the ORERP *Intertidal and Subtidal Alternatives* for a model problem in which the wash load source is defined by the sediment yield of the Otay River during the 100 year flood. Because of the nearby Savage Dam, the sediment yield is assumed to be derived from scour and erosion of the Otay River floodplain, downstream from the dam. Scour impacts from the 100 year flood on the ORERP have been evaluated in a companion study Everest, (2014). The primary concern of the present analysis is that a portion of the floodplain that could be scoured and eroded by the 100 year flood has surficial layers of soil comprised of a high percentage of silts and clays that contain various concentrations of DDT; and that some of those fine-grained sediments might re-settle in the tidal basins of the ORERP post-flood. Because the duration of the 100-yr flood is only 24 hours, we assume that tidal exchange will quickly re-establish flow dominance post-flood; and that the transport and settling dynamics of potentially contaminated silts and clays will be driven and limited by the tidal hydraulics and tidal residence times detailed in Sections 4 and 5.

This study is a multi-disciplinary effort of four scientists. The study begins with a soil characterization and erosion analysis of the 100 year flood in Sections 2 and 3, respectively, which was conducted by Ying Poon, D.Sc. of Everest International. Section 2 provides the essential sediment flux initial conditions for a post-flood suspended sediment tidal transport and deposition analysis in Sections 4 and 5 that was performed by Scott Jenkins, Ph.D. of Michael Baker International. The post flood deposition thicknesses and DDT concentrations in the tidal basins that were calculated in Sections 5 were throughput to a biological impact assessment presented in Section 6 that was conducted by Catherine Zeeman, Ph.D. and Carol Roberts of the Environmental Contaminants Division, Carlsbad Fish and Wildlife Office. Deposition results for the 50-year flood appear in APPENDIX-B and were found to remain within the range of variability of scenarios for the 100-year flood.

2.0) Erosion Analysis for the 100-Year Flood in the Otay River Basin:

Everest International Consultants (Everest) conducted an analysis on the potential for the DDT containing soils in the Otay River Floodplain (ORF) to be eroded and transported to the proposed wetland during a 100-year flood event. The analysis was based on numerical simulations conducted with the two-dimensional hydrodynamic model – TUFLOW, which simulated the velocities over the ORF during a 100-year flood event. The analysis was also based on soil property data from the soil sampled in the ORF to evaluate the potential for soil erosion. Details of the TUFLOW model setup can be found in Everest (2014).

The 100-year flood hydrographs for the Otay River, Poggi Canyon Creek and Nestor Creek are shown in Figure 1. The total flow volume during a 100-year flood for the Otay River is 35,200,000 cubic yards (cy), or 26,911,315 cubic meters (m^3). The corresponding flow volumes for Poggi Canyon Creek and Nestor Creek are respectively 2,240,000 cy (1,712,254 m^3) and 1,748,800 cy (1,337,003 m^3), so that the combined flow through the floodplain is $\overline{Q} =$ 39,188,800 cy (29,960,856 m^3), or 24,290 acre ft. The flow for the Otay River is an order of magnitude higher than those for Poggi Canyon Creek and Nestor Creek. The percent of Nestor flow that would pass through the wetland was not analyzed, but since Nestor Creek directly flows into the proposed wetland area, it was assumed that all of the Nestor Creek flow would enter the wetland. Figures 2 & 3 give the distributions of maximum stream flow velocities for the 100-year flood velocity throughout the Otay River floodplain and adjacent pond complexes for the ORERP Intertidal and Subtidal Alternatives, respectively.

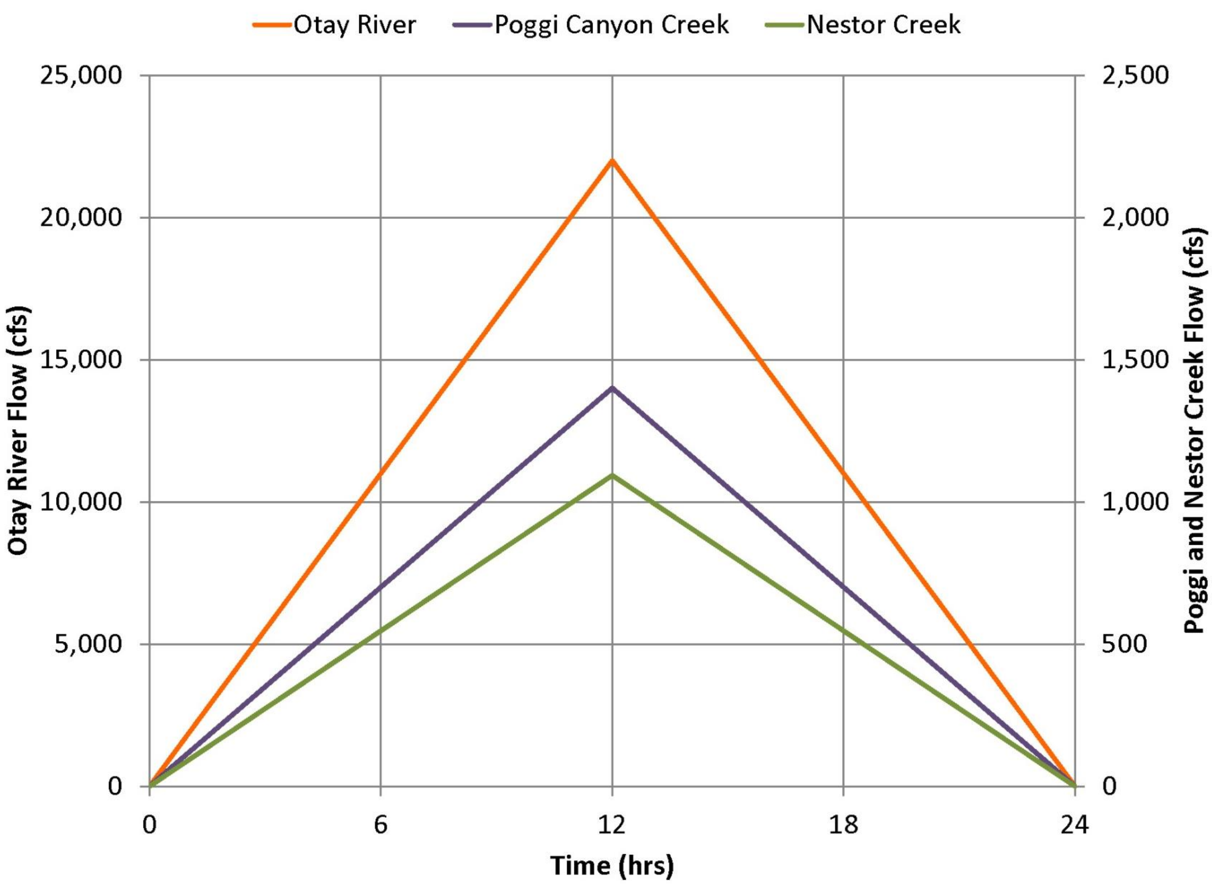

Figure 1. The 100-Year Return Period Flood Hydrographs**.**

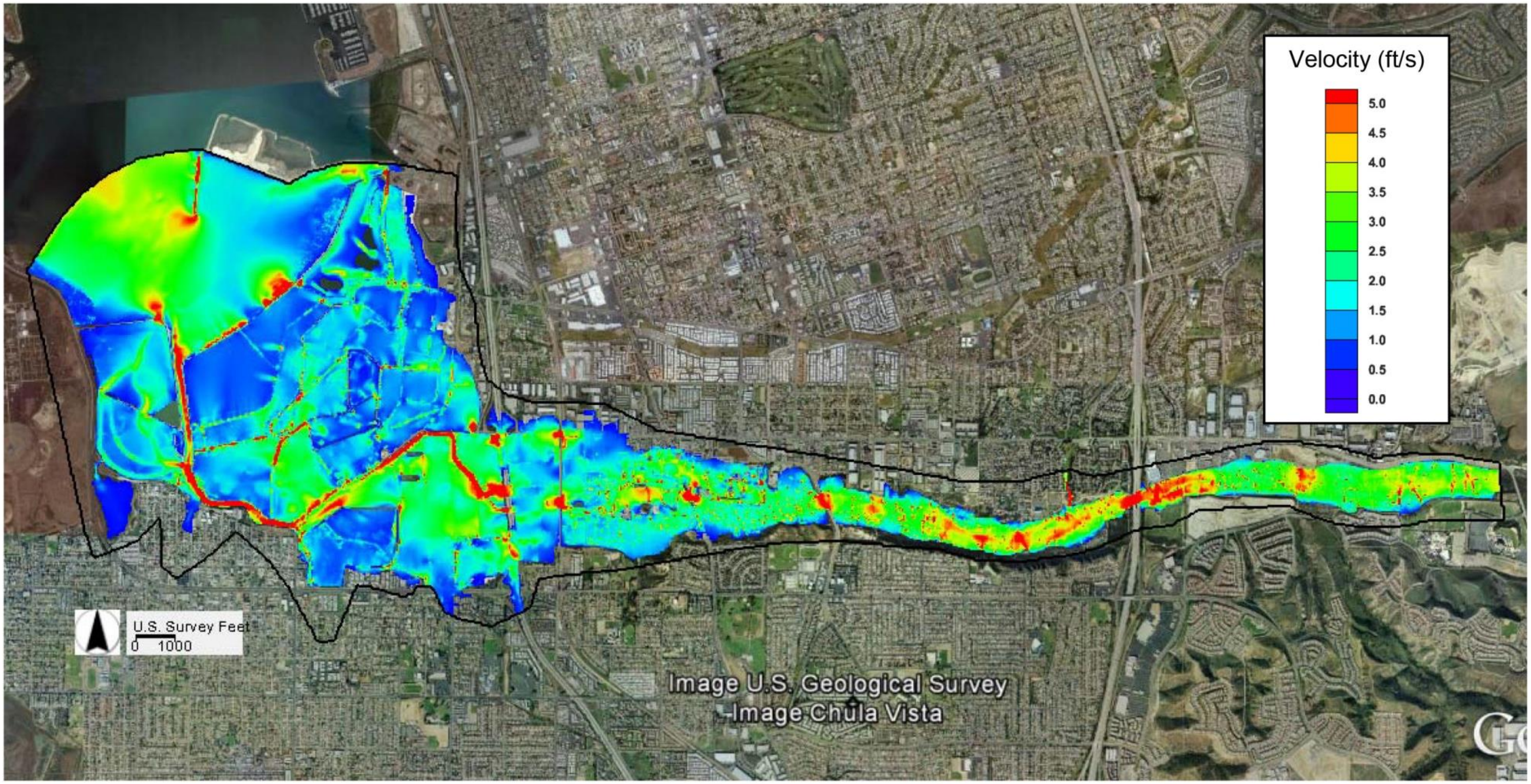

Figure 2: Distribution of maximum stream flow velocities for the 100-year flood in the lower Otay River flood plain with the fully implemented Intertidal Alternative, (after Everest, 2014)

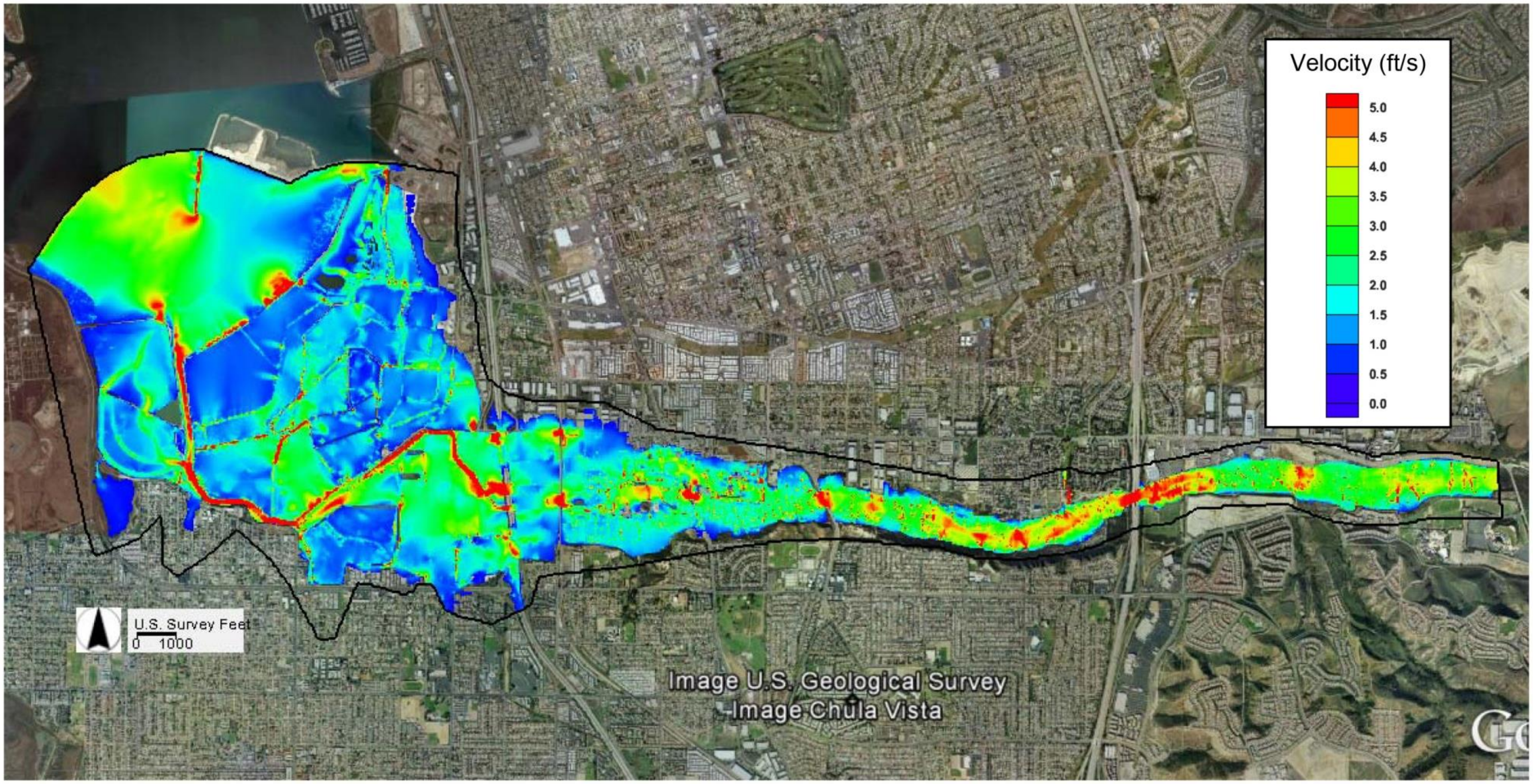

Figure 3: Distribution of maximum stream flow velocities for the 100-year flood in the lower Otay River flood plain with the fully implemented Subtidal Alternative, (after Everest, 2014).

Data were not available for the sediment discharge from the Otay River Watershed during a 100-year flood event; hence, it was estimated based on sediment discharge from the Buena Vista Creek (BV) Watershed for which sediment discharge during a 100-year study was available. In an earlier fluvial hydraulic and sediment transport study, Everest (2008) estimated that the sediment discharge during a 100-year flood event for the BV Watershed would be about 603,000 cy. Characteristics of the Otay River Watershed and BV Watershed are compared in Table 1. The area for the BV Watershed is approximately 19 square miles, while the Otay River Watershed (portion below the dam) is about 46 square miles with the entire Otay Watershed covering 143 square miles. Compared with the BV Watershed, the Otay River Watershed (below dam) is less urbanized with more open space land use, potentially more susceptible to soil erosion during a flood event. Nevertheless, simply based on scaling by the watershed size, sediment discharge from the Otay River Watershed is about 1,460,000 cy during a 100-year event. Based on Taylor (1981), about 50% of the sediment delivered from the Otay River Watershed is fine grain size (d < 0.065 mm), and the other 50% is sand. Hence, the fine portion is about 730,000 cy. It is estimated that during a 100-year flood, approximately 60% of the fine grain sediment discharge, i.e. 438,000 cy (334,880 m^3) would pass through the proposed wetland.

Table 1: Comparison between Otay River and Buena Vista Creek Watersheds

COMPARISON	OTAY RIVER	BUENA VISTA CREEK
Watershed Area	46 mi^2 *	19 mi^2
Urban Land Uses	39.1%	75.2%
Agricultural Land Uses	0.6%	22.4%
Open Space Land Uses	60.3%	2.4%

**Watershed area below dam*

2.1) Soil Erosion from Otay River Floodplain: The potential for soil erosion from the ORF during a 100-year flood event is evaluated based on the flood velocities and soil properties. In general, silt and clay are less susceptible to erosion while sand is relatively easier to be eroded. Based on the sediment characterization study conducted by Anchor QEA (2013), the top three feet of sediment consists of fine to coarse sand (i.e., easy to be eroded), and below three feet, based on data for samples taken between 3 to 5 ft below ground, sediments are cohesive, consisting mainly of silt and clay (less susceptible to erosion). As illustrated by the Hjulstrom Curve shown in Figure 4, sediments consisting mainly of fine sand to coarse sand (the blue shaded area in Figure 4) are likely to be scoured (eroded) when the flood flow velocity is higher than approximately 0.6 ft/sec. The TUFLOW model simulated maximum flood velocities over the ORF area during a 100-year flood event is shown in Figure 5. The color scale of the figure is selected such that the lowest velocity shown is 0.6 ft/sec (threshold for scouring). As can be seen in the figure, the maximum flood velocities over the entire ORF are higher than 0.6 ft/sec; hence likely to be eroded based on the Hjulstrom curve. In addition, based on the TUFLOW model results, (Figures 2 & 3) the bed shear stress over the ORF ranges from about 0.2 N/m^2 to 0.9 N/m^2 during a 100-year flood event. Based on empirical data relating sediment erosion to bed shear stress, these bed shear stresses are high enough to result in sediment erosion (Roberts et al, 1998).

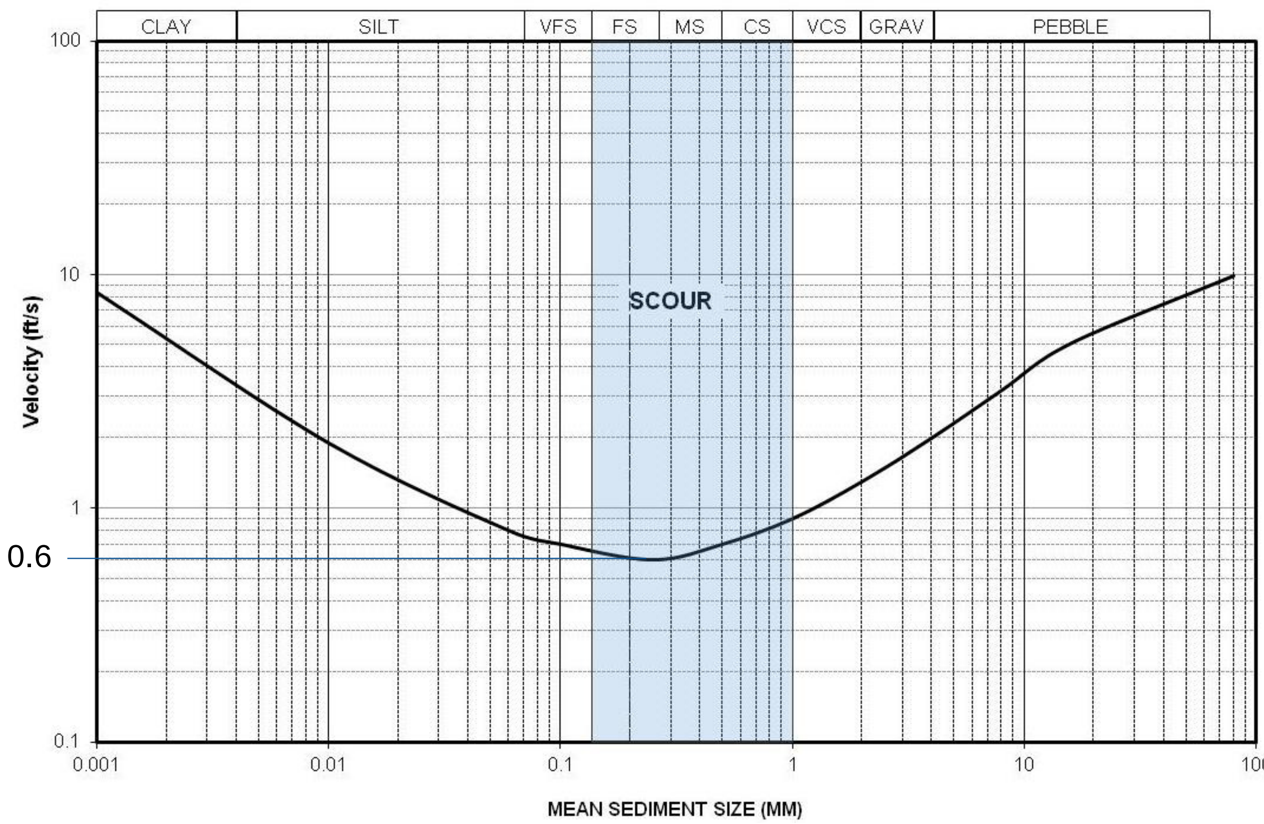

Figure 4. Hjulstrom Curve

Not all the sediment being eroded from the ORF would be transported and delivered to the proposed wetland. Sediment being eroded may simply move along the bed from one location to another, or remain suspended in the water column (portion that are likely to be transported). Some of the suspended sediment may be re-deposited in another area (not entering the wetland). In lieu of conducting a sediment transport modeling study, it is not easy to quantify the amount of eroded soil from the ORF that would be transported to the proposed wetland. Hence, for this study, three erosion scenarios for erosion depths of 1 ft, 2 ft, and 3 ft over the entire ORF were considered for the evaluation of potential transport and deposition of DDT contaminated soils from the ORF to the proposed wetland. The volume of eroded soil, percent fines (d < than 0.065 mm), and volume of fines for these three scenarios are summarized in Table 2.

Table 2: Volume and Properties of Eroded Soils of ORF

EROSION DEPTH (ft.)	VOLUME OF ERODED SOIL (cy)	PERCENT FINES	VOLUME OF FINES (cy)
1	114,890	21.1%	24,260
2	229,780	33.2%	76,350
3	344,700	37.2%	128,300

2.2) DDT Concentrations of the Eroded Soils: Two soil sampling and analysis datasets were utilized to evaluate the DDT concentrations of the ORF soils under the three erosion scenarios described above. The two datasets include data from an earlier Anchor QEA study (2013) along with newer data from a U.S. Fish and Wildlife Service study (Zeeman 2014). The two datasets consist of data for different sampling locations and boring depths. The Anchor data consist of 11 borings over the ORF, with DDT concentrations for depth layers of 0 to 1, 1 to 3, and 3 to 5 feet below ground. Soil data from the U.S. Fish and Wildlife Service consists of 14 sampling locations, and data were collected from the top 0.5 feet below ground.

Based on discussion with the project team, it was decided to assume that the DDT concentrations for the U.S. Fish and Wildlife samples would apply to the top one foot of soils; hence can be combined with the Anchor data for the top 1 ft to evaluate the DDT concentrations for the top 1 ft. of soil over the ORF. From these data, the DDT concentrations of the ORF were estimated using Voronoi diagrams, in which each cell area is partitioned based on the sampling locations. Figure 6 shows the resulting Voronoi diagram for the top 1 ft of the soil over the ORF

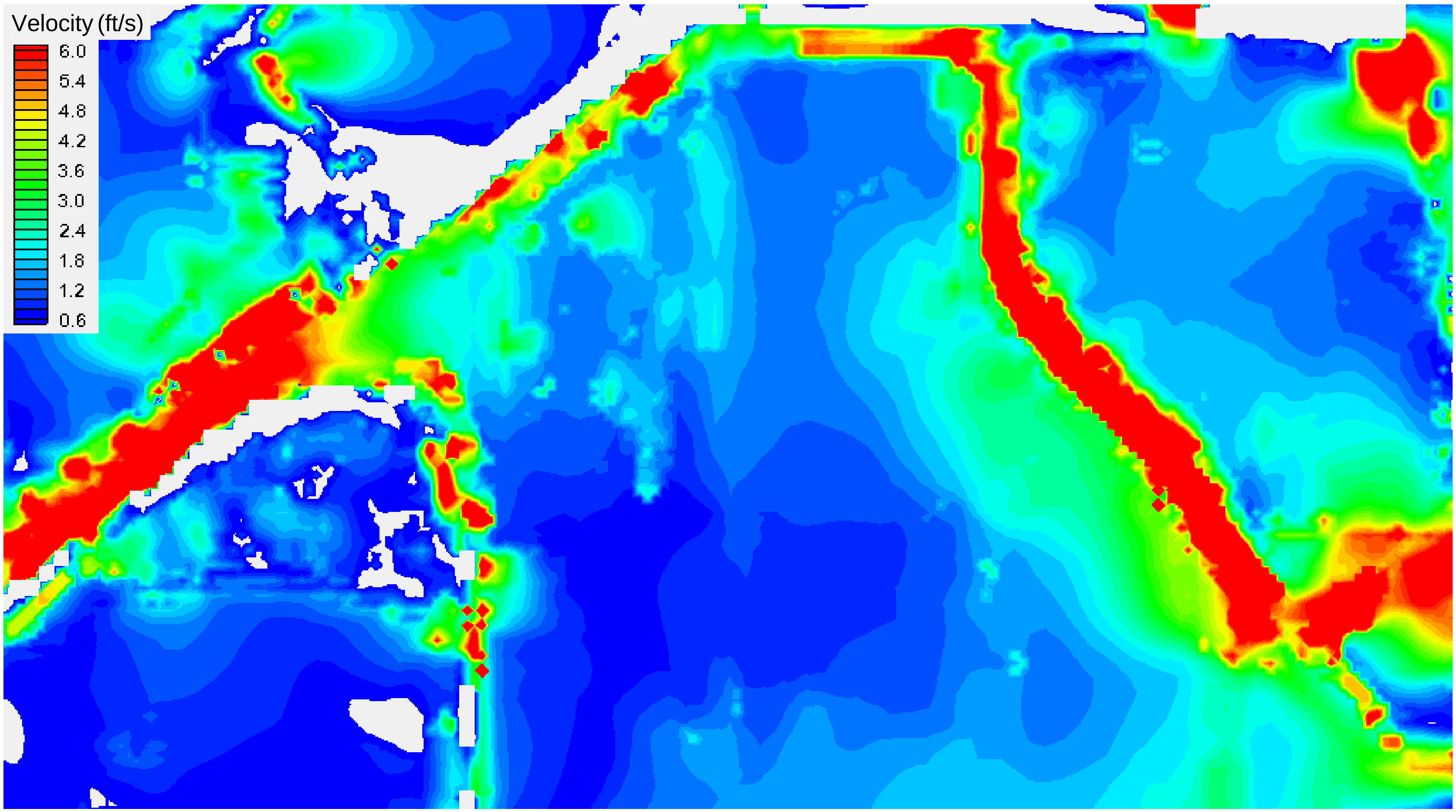

Figure 5. Maximum Velocity during a 100-year Flood under Existing Conditions. Note: white color indicates maximum velocity less than 0.6 ft/s

using both datasets (a total of 25 locations—11 from Anchor QEA and 14 from the U.S. Fish and Wildlife Service). The numbers shown in the figure are the Voronoi cell size in square feet and associated DDT concentrations in µg/kg. Similar Voronoi diagram for soil layer from 1 to 3 feet below ground is provided in Figure 7. This diagram is developed using only the Anchor QEA data. The number of cells in Figure 7 is fewer than those shown in Figure 6 since there are only 11 relevant boring locations for this layer. From these Voronoi diagrams, the average DDT concentration (weighted by soil volume) under the three erosion scenarios over the ORF were calculated and summarized in Table 3.

Table 3: Average DDT Concentrations for Three Erosion Scenarios

EROSION DEPTH (ft.)	AVERAGE DDT CONCENTRATION (µg/kg)
1	790
2	430
3	310

3.0) Specifying the Sensitivity Analysis for the Post 100-Year Flood DDT Deposition

The sensitivity analysis is based on a parameter sweep of the amounts of DDT containing sediments that might be eroded by the 100-year flood; and Section 2.2 has provided coring analysis that indicates the concentration of DDT in the eroded fine sediments could vary between 790 µg/kg to 310 µg/kg, depending on the depth of erosion. Section 2.0 indicates that these eroded contaminated sediments could mix with as much as 438,000 cy of fines (assumed to be uncontaminated) from the upper watershed below the Savage Dam. This estimate was based on a surrogate watershed (Buena Vista Creek) for which more complete sediment yield data was available. Based on the uncertainties of applying that surrogate analysis to the Otay River watershed, it is sensible to consider the sensitivity of the final outcome to omitting the flux of supposedly “clean” sediments from upstream sources altogether. In the absence of any new information revealing additional upstream sources of DDT, that omission will eliminate the dilution effects that blending with “clean” fines exerts on DDT concentrations during the post-flood deposition. From this assessment of the possible sediment erosion input assumptions, a sensitivity analysis **is posed** for the post 100-year flood DDT deposition that is based on erosion fluxes from three possible erosion depths (1 ft, 2 ft, and 3 ft) in the DDT contaminated area of the floodplain that are each combined with two possible fluxes of clean fines (0 cy and 438,000 cy) from the watershed below the Savage Dam; yielding a sensitivity analysis comprised of 6 separate deposition scenarios. The ensembles of input parameters for this sensitivity analysis are summarized in Table 4.

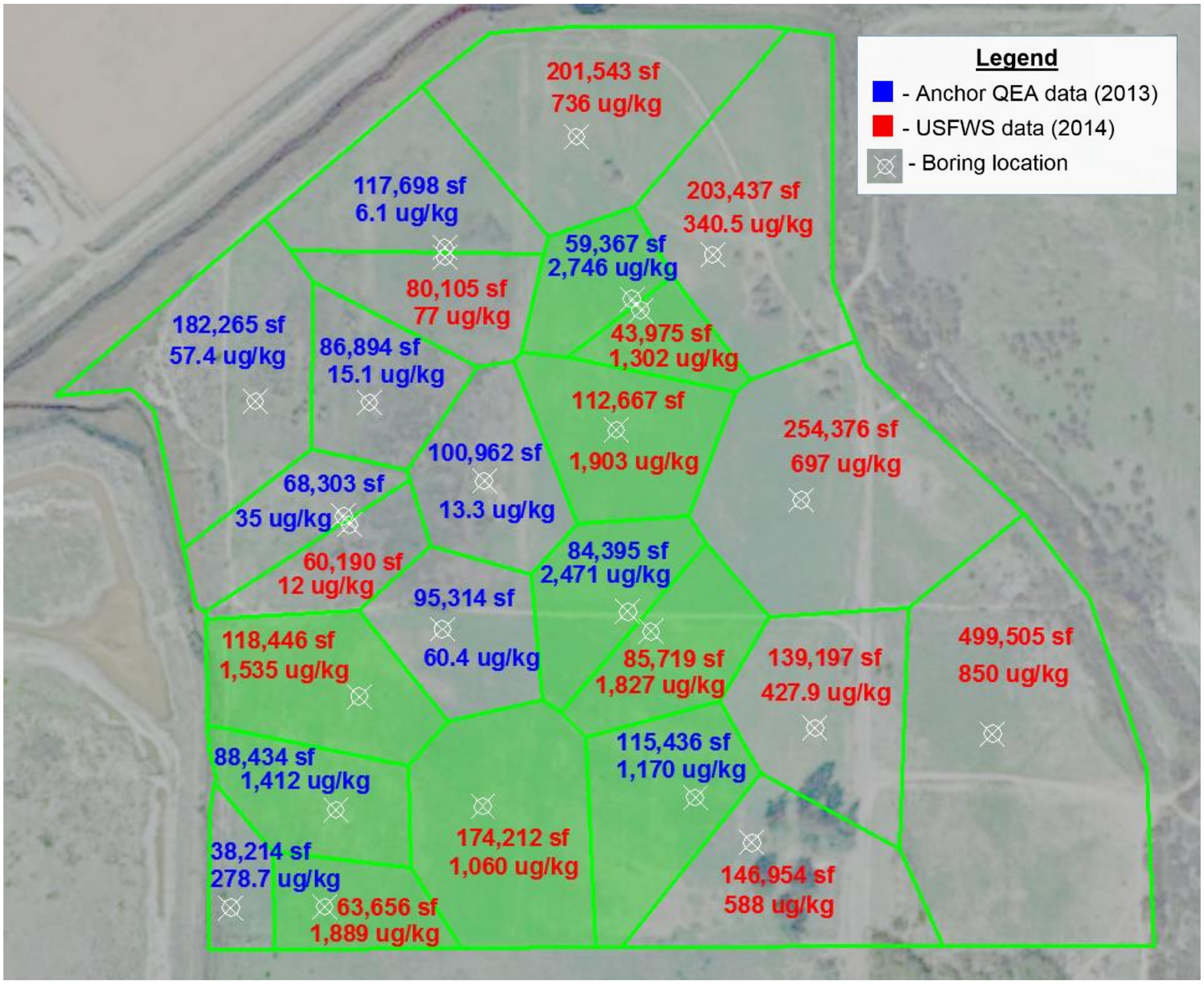

Figure 6. Voronoi Diagram for Soils 0 ft. to 1 ft. below ground surface - Cell Areas and DDT Concentrations

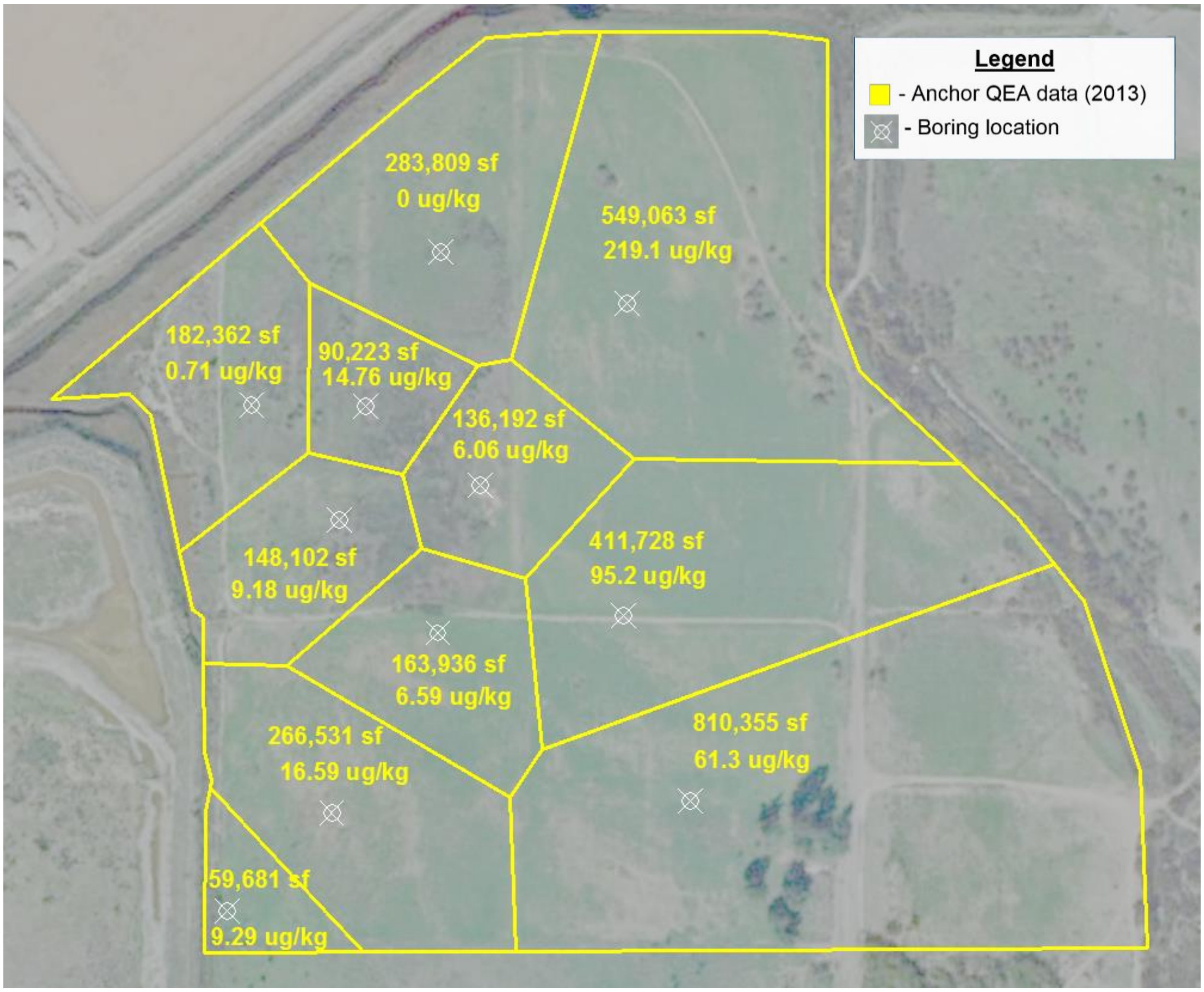

Figure 7. Voronoi Diagram for Soils 1 ft - 3 ft below ground surface - Cell Areas and DDT Concentrations

The suspended sediment concentrations in Table 4 are based on a dry bulk density for eroded soil of 2700 lb per cy, or 1.225 metric tons per cy; where a metric ton is 1000 kg. This conversion factor is applied to the sum of the volume of eroded DDT- bearing fines (column_2) and the volume of eroded fines from the upper Otay watershed (column_4) to obtain the total flux of suspended fine grained sediment in tons/day during the 24-hour flood period of the 100-year flood (cf. Figure 1). The sand and gravel sized fractions eroded from the floodplain by the 100-year flood are assumed to be transported as bed load. The suspended sediment flux component (column_2 + column_4) is divided by the flow volume of $\overline{Q}$ = 29,960,856 m^3 during the 24-hour flood period to give the average suspended sediment concentration in column_6 upon conversion of metric tons to grams and cubic meters to liters.

Table 4: Input Parameters for Sensitivity Analysis of Post 100-Year Flood DDT Deposition

Scenario	Volume of Eroded DDT-Bearing Fines	Average DDT Conc. in DDT-Bearing Fines	Volume of Eroded Upper Watershed Fines	Flood Flow Volume	Suspended Sediment Conc.
Erode top 3 ft. of Contaminated Area + Upper Watershed*	128,300 cubic yards	310 μ g/kg	438,000 cubic yards	24,290 acre ft	23.15 g/l.
Erode top 1 ft. of Contaminated Area + Upper Watershed	24,260 cubic yards	790 μ g/kg	438,000 cubic yards	24,290 acre ft	18.90 g/l
Erode top 2 ft. of Contaminated Area + Upper Watershed	76,350 cubic yards	430 μ g/kg	438,000 cubic yards	24,290 acre ft	21.03 g/l
Erode top 3 ft. of Contaminated Area Only*	128,300 cubic yards	310 μ g/kg	0 cubic yards	24,290 acre ft	5.25 g/l
Erode top 1 ft. of Contaminated Area Only*	24,260 cubic yards	790 μ g/kg	0 cubic yards	24,290 acre ft	0.99 g/l
Erode top 2 ft. of Contaminated Area Only*	76,350 cubic yards	430 μ g/kg	0 cubic yards	24,290 acre ft	3.12 g/l

4.0) Suspended Sediment Transport and Deposition:

Because DDT is hydrophobic, it can only be adsorbed and transported by the silt and clay fractions of floodplain soils eroded by the 100 year flood. These fine-grained fractions are transported as suspended load (commonly referred to as wash load), and capable of becoming re-distributed into the tidal basins of the restoration project; while the remaining coarser erodible fractions (primarily sands and gravels) are transported as bedload and remain confined to the streambeds of the Otay River, Poggi Canyon Creek and Nestor Creek, (Everest, 2014). For this reason, we focus on the tidally influenced suspended sediment transport dynamics of fine-grained silts and clays in the post-flood period.

While the duration of the 100 year flood is relatively brief (24 hr), the transport, redistribution and settling of the washload sediments can linger on for days, even weeks under the influence of tidal exchange. Typically in calm water, silt particles will require 4.3 hr. to settle to the bottom in 1 meter of water depth, while clay-sized particles can take as long as 18 days. The residence time of water in South San Diego Bay can be as long as 40 days (Largier, 1995); consequently, washload discharged into South San Diego Bay from the 100-year flood can potentially recirculate back into the tidal basins of the restoration project for many tide cycles before the fine-grained washload sediments completely settle out of the South Bay water mass.

From Anchor (2013), the average grain size of the silts and clays that make up the 37.2% of the sediments found in the top three feet of erodible sediments in the black outlined area of Figures 6 & 7 is only 25 microns ($\bar{d}_{fines} \cong 0.025$ mm). The settling velocity is only w_s = 0.030 cm/sec based upon 25 micron median aggregate size of silts and clays (Figure 8). Because of these very low settling rates, (*Stokes settling regime*), subsequent deposition of the silts and clays that contain DDT will be a slow process, which will extend for many tide cycles depending on the local water depth. In posing the problem of tidal flushing of these fine-grained sediments from the tidal basins of the restoration project, we shall neglect any hydrodynamic effect on the tidal hydraulics due to the river flow. This assumption is supported by the short duration of the flood hydrograph relative to the duration of settling and deposition processes. By this assumption, we are basically saying that the hydrodynamics are dominated by the fluvial processes during the first 24 hours, since the flow volume of the 100-year flood is 56 times larger than the combined tidal prisms of the restoration project tidal basins. Thereafter, tidal processes ensue; so that fluvial and tidal processes occur sequentially without interaction. In addition, we shall assume that the sediment yield of the 100-year flood is uniformly dispersed at the end of the flood period, with an initial suspended sediment concentration $\bar{C}_0$ given by column_6 in Table 4 that is uniform throughout the floodplain and adjacent South San Diego Bay as far north as the nodal points at the Chula Vista Wildlife Reserve (Figures 2 & 3). This initial uniform suspended concentration is subsequently modified by the action of tidal advection and diffusion and by gravity-induced settling that we shall represent by the following form of the sediment continuity equation:

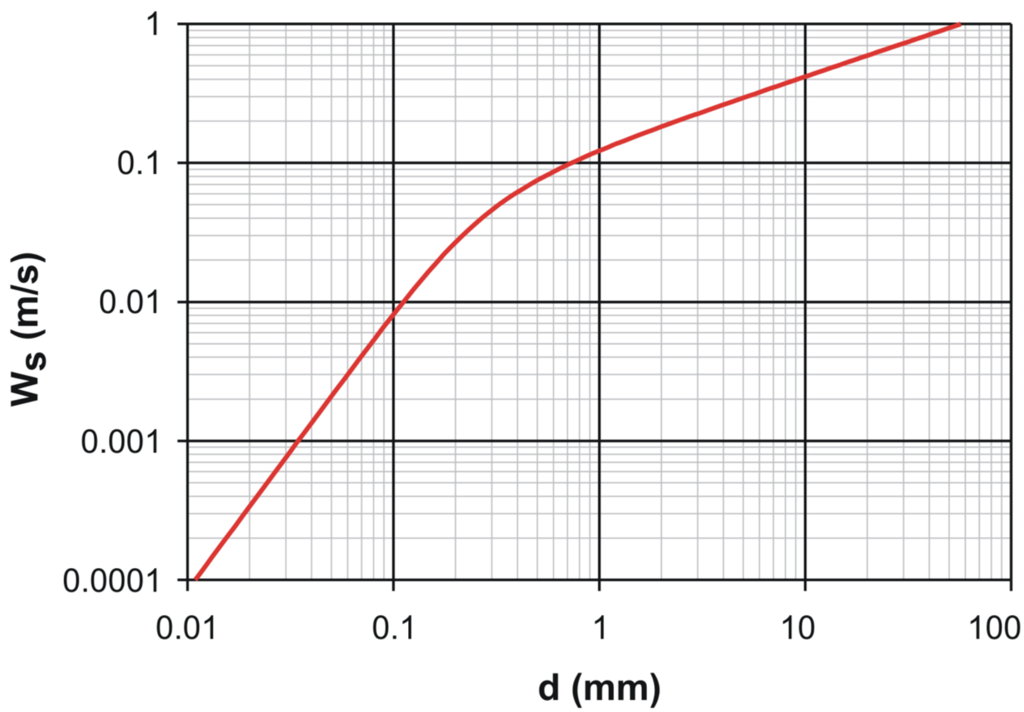

Figure 8: Settling velocity of quartz grains as a function of median grain size.

$$\frac{\partial}{\partial t} cH + \frac{\partial}{\partial x} J_x + \frac{\partial}{\partial y} J_y = \varepsilon_m \left(\frac{\partial^2 c}{\partial x^2} + \frac{\partial^2 c}{\partial y^2} \right) - Sc \qquad (1)$$

where $c = c(x, y, t)$ is the local suspended sediment concentration; S is the settling (sink) coefficient, $S \sim f(w_0)$; w_s is the settling velocity of the sediment that is independent of (x, y, t) and is a single valued function of grain size only according to Figure 8; the water depth at any finite element node is $H = \eta + h$; h is the local bottom elevation in NAVD 88; η is the tidal amplitude in NAVD 88; ε_m is the mass diffusivity, and J_x , J_y are local sediment flux components due to the local depth averaged tidal velocities, $(\bar{u}, \bar{v})$:

$$J_x = c \int_{-h}^{\eta} \bar{u} dz \qquad (2)$$

$$J_y = c \int_{-h}^{\eta} \bar{v} dz$$

Equation (1) is forced by the solutions for the water surface elevations, η , and tidal velocities, $(\bar{u},\bar{v})$ generated by the TIDE_FEM finite element tidal hydraulics model applied to the grading designs of the Intertidal and Subtidal Alternatives for the ORERP. These TIDE_FEM tidal hydraulics solutions are documented in Jenkins and Wasyl (2014).

The term Sc in Equation (1) represents a sink for suspended sediment, often referred to as the deposition flux, $D(x,y,t)$, that is the net of settling and re-suspension:

$$Sc = D(x,y,t) = c(x,y,t)\,w_s - \frac{\alpha\left[\tau(x,y,t)-\tau_c\right]}{\tau_c} \tag{3}$$

Where the term $c\,w_s$ is the downward-directed settling flux, while the upward flux of sediment re-suspended by bottom shear stress is $E(x,y,t)=\alpha(\tau-\tau_c)/\tau_c$. Here, α is an empirical coefficient, α = 2.356 X 10^{-4} g/cm^2/sec after Mehta (1981); τ_c = 0.5 dynes/cm^2 is the cohesive yield stress for unconsolidated mud after Mehta, et al. (1982); and $\tau = (\tau_x^2+\tau_y^2)^{1/2}$ is the tidally induced bottom shear stress from the that is quasi-linearized by Chezy-based friction using Manning's roughness factor, n_o:

$$\begin{aligned}\tau_x &= -\frac{g}{\rho H^2 C_z^{\,2}}\,q_x\left(q_x^{\,2}+q_y^{\,2}\right)^{1/2}\\ \tau_y &= -\frac{g}{\rho H^2 C_z^{\,2}}\,q_y\left(q_x^{\,2}+q_y^{\,2}\right)^{1/2}\end{aligned} \tag{4}$$

$$q_x = \rho\int_{-h}^{\eta}\bar{u}\,dz \quad ; \quad q_y = \rho\int_{-h}^{\eta}\bar{v}\,dz$$

Here, C_z is the Chezy coefficient calculated as:

$$C_z = \frac{1.49}{n_0}H^{1/6} \tag{5}$$

By Equations (1) – (5), the post flood deposition processes of settling and re-suspension are posed as a time-dependent, two-dimensional boundary value problem in which the forcing is provided by the depth averaged tidal velocities, $(\bar{u},\bar{v})$ resolved by the TIDE_FEM tidal hydraulics model detailed in Sections 2 and 3 of Jenkins and Wasyl, (2014). Boundary conditions and initial conditions on Equation (1) are imposed at the land-water and open water boundaries and open-water boundaries and nodes. Flux quantities normal to these boundary contours are denoted with "n" subscripts and tangential fluxes are given "s" subscripts. At any point along a boundary contour, the normal and tangential suspended sediment fluxes are:

$$J_n = \int_{-h}^{\eta} c u_n dz = \alpha_{nx} J_x + \alpha_{ny} J_y$$
$$J_s = \int_{-h}^{\eta} c u_s dz = -\alpha_{nx} J_x + \alpha_{ny} J_y \qquad (6)$$
$$\alpha_{nx} = \cos(n, x)$$
$$\alpha_{ny} = \cos(n, y)$$

On land-boundary contours, the suspended sediment flux components are prescribed as:

$$J_n = J_s = 0 \qquad \text{on land-water boundaries} \qquad (7)$$

On the open-water boundaries and nodes of the computational mesh, an initial post-flood condition is imposed requiring that the suspended sediment concentration is a constant, $\overline{C}_0$, given by the eroded area values from column_6 in Table 4, or $c = \overline{C}_0$ in Equation (6) at $t = 24$ hr.

Equation (1) is solved over the same finite element mesh as the ORERP tidal hydraulics simulations using the Galerkin weighted residual method detailed in Gallagher,(1981), Weiyan (1992). By this approach, the sediment continuity equation (1) reduces to a simple oscillator equation forced by the collection of algebraic terms which is easily integrated over time. The time integration scheme used over each time step of the post-flood tidal forcing period is based upon the *trapezoidal rule*. This scheme was chosen because it is known to be unconditionally stable, and in tidal propagation problems has not been known to introduce spurious phase differences or damping. It replaces time derivatives between two successive times, $\Delta t = t_{n+1} - t_n$, with a truncated Taylor series.

Solutions to Equation (1) for the post-flood suspended sediment concentration $c = c(x, y, t)$ are combined with solutions for the tidally induced bottom shear stress, $\tau(x, y, t)$, from the TIDE_FEM model to compute the deposition flux, $D(x, y, t)$ using Equation (3). As these solutions continue forward in time post-flood, $D(x, y, t) \to 0$ as the suspended sediments progressively fall out of suspension and $c(x, y, t) \to 0$. The deposition flux is integrated over this post-flood deposition period to compute the deposition thickness, but initially this deposition represents unconsolidated of fluid mud. With this initial deposition to consolidate and compact from an initial fluid-mud layer whose bulk concentration is C_f; to some partially consolidated mud layer whose bulk density is C_s, after Krone (1978), Mehta (1989). The deposition thickness at time $t = j\,\Delta t$ for any given nodal point is calculated [Krone, 1962]:

$$\Delta Z(x,y,t) = \int_0^{j\Delta t} \frac{D(x,y,t) - K_s C_s\, g}{1 - C_f\ / C_s}\, dt \tag{8}$$

where K_s = 4 x 10^{-13} sec is the sedimentation coefficient after the work of Fujita (1962). The fluid mud layer bulk concentration shall be set at C_f = 100g/l and the partially consolidated mud concentration shall be set at a rather low value of C_s = 200 g/l to allow for the effects of bioturbation. These are conservative values which will tend to overestimate deposition thickness. The mass diffusivity shall be set at ε_m = 4.9 cm^2/sec based upon work conducted in tidal basins in the San Francisco Bay Estuary, Jenkins and Wasyl (1980, 1983, and 1990).

5.0) Post-Flood Tidal Deposition Simulations for the 100-Year Flood:

The TIDE_FEM model was run for 276 hours immediately following the 100-year flood using tidal forcing with Δt = 2 sec time step intervals at the mouth of the Otay River, derived from a spectral correction applied to the NOAA tide gage #941-0170 located at the Navy Pier, as detailed in Section 3.4 of Jenkins and Wasyl (2014). The post-flood tidal deposition simulations were run on the same finite element grid using the TIDE_FEM outputs for depth averaged tidal velocities, $(\bar{u}, \bar{v})$, tidally induced bottom shear stress $\tau(x, y, t)$, and local water surface elevations, η as forcing functions to Equation (1).

Initial conditions post flood were a uniform dispersion throughout the model grid of a suspension of silt and clay sized sediment characterized by a 20 micron median grain size with a settling rate of w_s = 0.030 cm/sec to account for some degree of flocculation. The initial conditions were specified as a uniform suspended sediment concentration, $\overline{C}_0$, and companion DDT concentration for each scenario of the sensitivity analysis according column_6 and column_3 respectively in Table 4. The finite element model grid included the lower Otay River channel beginning at the presently contaminated area shown in Figures 6 & 7; the tidal basins of the restoration with all of the salt pond complexes; and extended out into south San Diego Bay as far as the Chula Vista Wildlife Reserve as shown in Figures 2 & 3. Boundary conditions on this grid consisted of no normal fluxes of suspended sediment through the land-water boundaries, and continuity of normal and tangential fluxes of suspended sediment across the open water boundaries where a constant suspended sediment concentration $c = \overline{C}_0$ from column_6 in Table 4 prevailed at time $t = 24$ hr at the start of the deposition simulation. For each time step, the TIDE_FEM model solves equations (3) and (8) for deposition flux and deposition thickness at each finite element node in the grid mesh. The deposition of partially consolidated mud in the tidal basins of the restoration was characterized by averaging deposition flux and deposition thickness at 6 (ea.) nodes distributed across the Floodplain Tidal Basin and 9 (ea.) nodes distributed across the Pond 15 Tidal Basin of the Intertidal and Subtidal Alternatives.

Figure 9 gives the time evolution of the post-flood deposition flux and deposition thickness for the first scenario (row_2 of Table 4) in the Floodplain Tidal Basin of the Intertidal and Subtidal Alternatives; and Figure 10 gives results for those same quantities in the Pond 15 Tidal Basin of the Intertidal and Subtidal Alternatives. This scenario is based on maximum flood-induced erosion depths of 3 ft. in the contaminated area adjacent the Floodplain Tidal Basin mixed with 438,000 cubic yards of fine-grained sediments from upstream erosion of the portion of the watershed below the Savage Dam. Results are similar for both tidal basins and restoration alternatives with dry bulk DDT concentrations of 70.2 μ g/kg everywhere in the post-flood deposition, because the initial post-flood suspended sediment concentration is the same in all areas in and around the restoration as a consequence of the 100 year flood over topping and flowing through these areas with its washload (cf. Figures 2 & 3). The general depositional features are that deposition flux peaks within one diurnal tide cycle after cessation of the flood in both basins of both restoration alternatives, with an initial deceleration in flux during the first semidiurnal ebb tide. After the first post-flood diurnal tidal cycle, the deposition flux declines as progressive settling depletes the suspended sediment concentration, and tidal residence times in the tidal basins limits the amount of time for settling and deposition to occur. Meanwhile, deposition thickness, which results from the cumulative sum of deposition flux over time, rapidly builds during the peak deposition flux period, and then gradually approaches a constant limit for partially consolidated mud at 200 g/l bulk density as the deposition flux vanishes after 120 to 150 hours post-flood. The minor differences in deposition flux and deposition thickness among tidal basins and restoration alternatives in Figures 9 & 10 is due to differences in residence times and grading elevations (i.e. water depth).

The Floodplain Tidal Basin, which has the shortest residence time (2 days for the Intertidal Alternative and 2.5 days for the Subtidal Alternative), has the lowest peak deposition flux (16.5 – 18.3 ton/acre/day) and the shortest deposition period (~120 hours); and accumulates only 3.3 to 3.4 mm of partially consolidated mud after 276 hours post-flood (Figure 9). Because of the sub-tidal channel graded into the Floodplain Tidal Basin design for the Subtidal Alternative, the residence time and consequently the deposition fluxes and thickness are slightly greater than for the Intertidal Alternative.

On the other hand, tidal residence times are nearly a day longer for the Pond 15 Tidal Basin of both alternatives (where residence times are 3.0-3.2 days), and consequently deposition fluxes and thickness are notably greater in Figure 10 than for the Floodplain Tidal Basin in Figure 9. In Pond 15, the deposition flux peaks at 18.9-19.9 ton/acre/day, and the deposition period is longer, about 150 hours post-flood. Consequently the deposition thickness is nearly double in Pond 15, with 7.6 to 8.0 mm of partially consolidated mud laid down after 276 hours post-flood. Because more dredge fill from the Floodplain Tidal Basin construction is deposited in Pond 15 of the Subtidal Alternative, its storage volume and residence times are less than for the Intertidal Alternative, whence the deposition fluxes and thickness are slightly less for the Subtidal Alternative in Figure 10 than for the Intertidal Alternative.

The initial post 100-year flood accumulations of partially consolidated mud computed in Figures 9 & 10 will, over time, dewater and compact under its own immersed weight. If

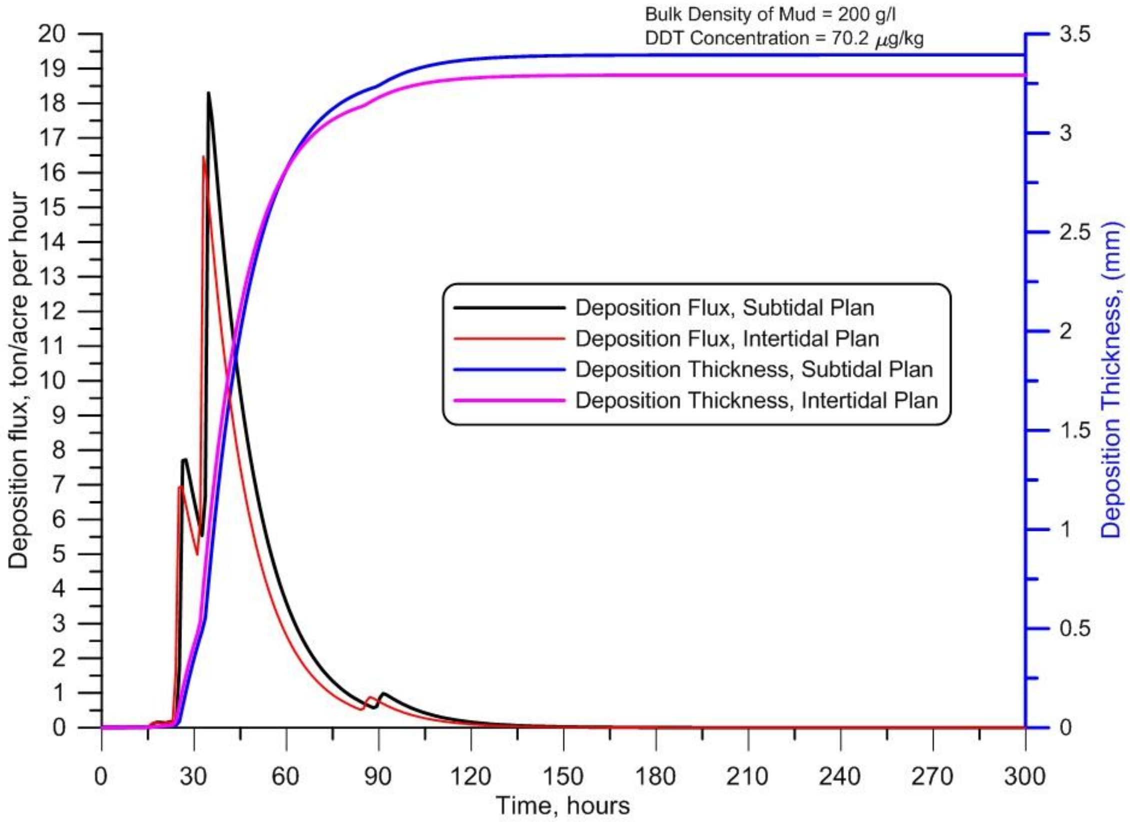

Figure 9. Sensitivity analysis of deposition of fine-grained sediment (mud) in the Otay River Floodplain Tidal Basin following the 100-year flood. Deposition flux (black & red); deposition thickness (blue & magenta). Results for 128,300 cubic yards of contaminated fines and 438,000 cubic yards of clean sediment from the upper Otay River watershed. DDT given as dry bulk concentration.

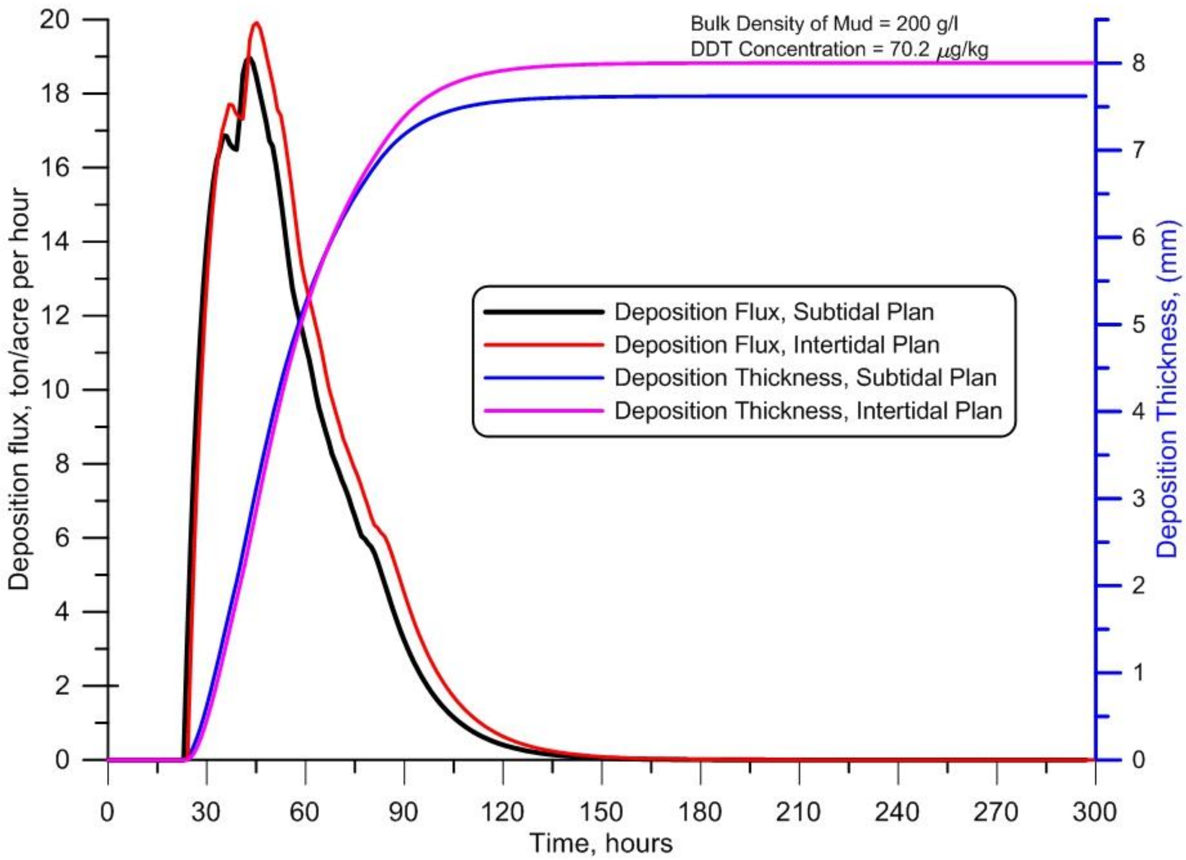

Figure 10. Sensitivity analysis of deposition of fine-grained sediment (mud) in the Pond-15 Tidal Basin following the 100-year flood. Deposition flux (black & red); deposition thickness (blue & magenta). Results for 128,300 cubic yards of contaminated fines and 438,000 cubic yards of clean sediment from the upper Otay River watershed. DDT given as dry bulk concentration.

we assume that the 3mm to 7mm of initial deposition would consolidate and compact to a maximum saturated density for fully consolidated mud, 1200 g/l, then the 100-year flood deposition for the first scenario in Table 4 (row_2) would eventually become a layer of consolidated mud only 0.5 mm to 1.2 mm thick; or:

$$\text{Floodplain Basin: } 3.3\,mm\ @\ 200g/l \ \Rightarrow \ \left\{\frac{dewatering}{consolidation}\right\} \ \Rightarrow \ 0.55\,mm\,@\,1{,}200g/l$$

$$\text{Pond 15 Basin: } 8\,mm\ @\ 200g/l \ \Rightarrow \ \left\{\frac{dewatering}{consolidation}\right\} \ \Rightarrow \ 1.4\,mm\,@\,1{,}200g/l$$

Consolidation only involves a reduction in the water content of the post-flood deposition, and therefore does not alter the DDT dry bulk concentration, which remains 70.2 μ g/kg once the muds have consolidated to a density of 1,200 g/l. The amount of time required for this degree of consolidation is uncertain, but experience with dredge material disposal ponds at Mare Island, CA and Charleston, SC [Jenkins, 1980; Jenkins et al., 1981; Jenkins and Skelly, 1983] suggests that consolidation to 600 g/l could occur within three months while full consolidation to saturation could take several years.

Next, consider how such results may be affected if we assume no erosion of soils occurs in the portion of the watershed upstream of the floodplain and below the Savage Dam. This scenario is specified by the fifth row in Table 4 and is based on maximum erosion depths of 3 ft. in the contaminated area only; and is considered *worst-case.* Here, runoff from the 100 year flood consists of a uniform suspended load of silts and clays with concentration of $\overline{C}$ = 5.25 g/l. Figure 11 gives the time evolution post-flood for deposition flux and deposition thickness in the Floodplain Tidal Basin of the Intertidal and Subtidal Alternatives; and Figure 12 gives results for those same quantities in the Pond 15 Tidal Basin of the Intertidal and Subtidal Alternatives. Again, results are similar for both tidal basins and restoration, but the dry bulk concentration of DDT in the post-flood deposition has increased to 310 μ g/kg, while the deposition thicknesses are greatly diminished. Again, the Floodplain Tidal Basin, with the shortest residence time (Figure 11), has the lowest peak deposition flux (3.7 – 4.1 ton/acre/day) and the shortest deposition period (~120 hours); and accumulates only 0.75 to 0.77 mm of partially consolidated mud after 276 hours post-flood. Deposition fluxes and thickness are slightly greater for the Floodplain Subtidal Alternative than for the Intertidal Alternative, due to its deeper sub-tidal channel and longer residence time. With tidal residence times being nearly a day longer for the Pond 15 Tidal Basin of both alternatives, deposition fluxes and thickness are notably greater in Figure 12 than for the Floodplain Tidal Basin in Figure 11. In Pond 15, the deposition flux peaks at 4.3 - 4.5 ton/acre/day, and the deposition period is longer, about 150 hours post-flood. Consequently the deposition thickness is nearly double in Pond 15, with 1.7 to 1.8 mm of partially consolidated mud laid down after 276 hours post-flood. Because more dredge fill is deposited in Pond 15 under the Subtidal Alternative, its storage volume and residence times are less than for the Intertidal Alternative, whence the deposition fluxes and thickness are slightly less in Figure 12 for the Subtidal Alternative. After dewatering and compaction to a density of 1200 g/l, the post-flood deposition for this worst case eventually become a layer of consolidated mud on the order of

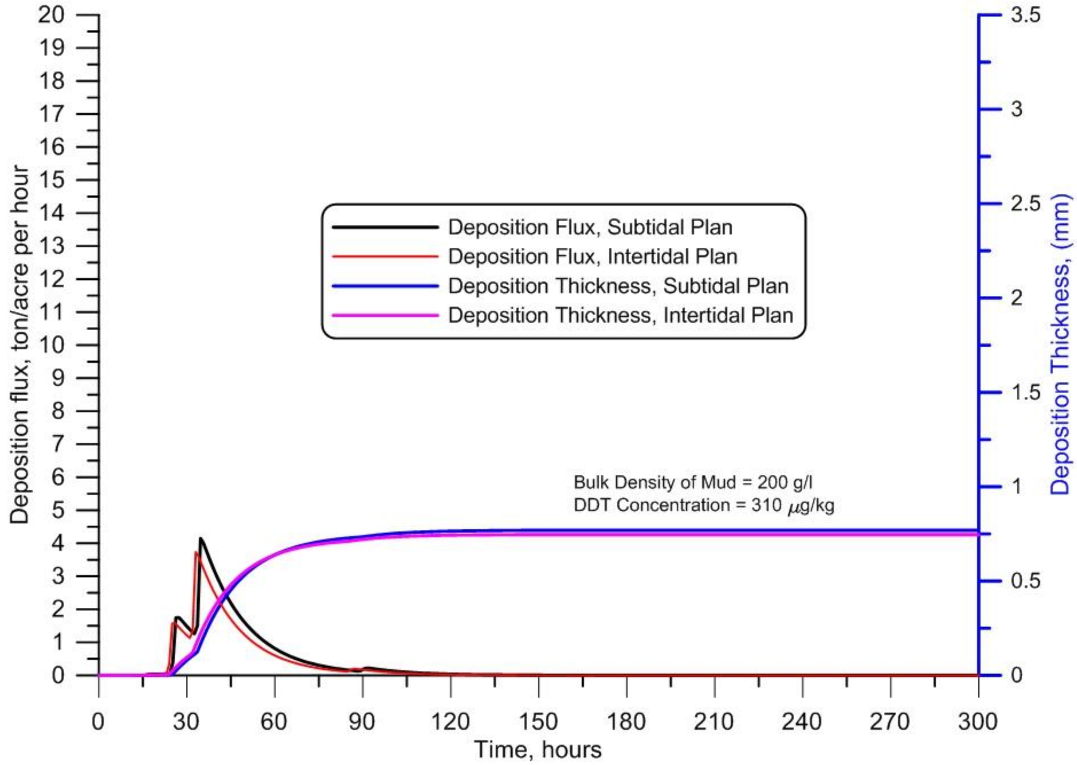

Figure 11. Sensitivity analysis of deposition of fine-grained sediment (mud) in the Otay River Floodplain Tidal Basin following the 100-year flood. Deposition flux (black & red); deposition thickness (blue & magenta). Results for 128,300 cubic yards of contaminated fines and no additional clean sediment from the upper Otay River watershed. DDT given as dry bulk concentration

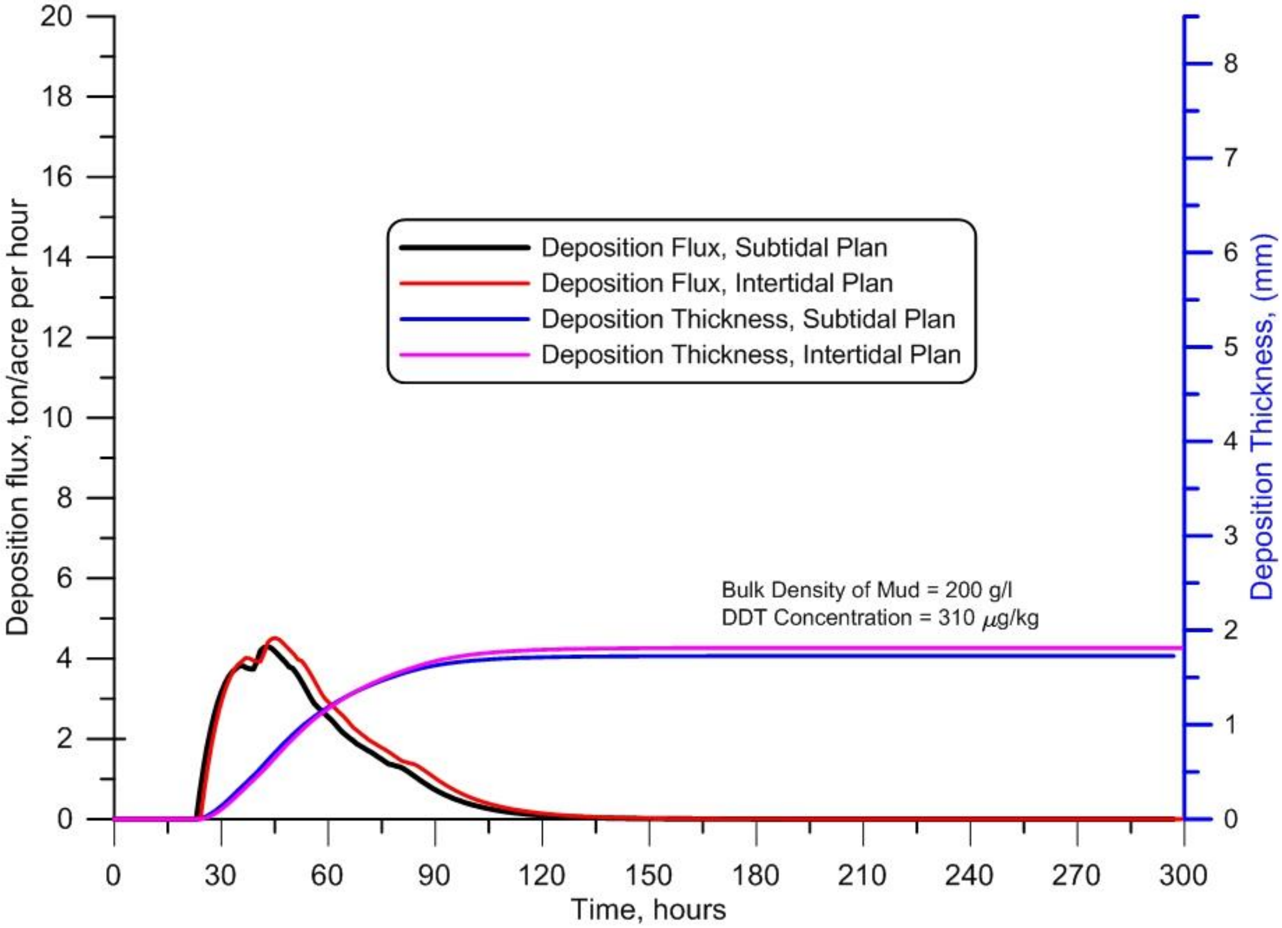

Figure 12. Sensitivity analysis of deposition of fine-grained sediment (mud) in Pond-15 Tidal Basin following the 100-year flood. Deposition flux (black & red); deposition thickness (blue & magenta). Results for 128,300 cubic yards of contaminated fines and no additional clean sediment from the upper Otay River watershed. DDT given as dry bulk concentration

only 0.2 mm to 0.4 mm thick; or:

Floodplain Basin: $0.75\,mm\ @\ 200g/l \;\Rightarrow\; \left\{\frac{dewatering}{consolidation}\right\} \;\Rightarrow\; 0.17\,mm\,@\,1{,}200g/l$

Pond 15 Basin: $1.8\,mm\ @\ 200g/l \;\Rightarrow\; \left\{\frac{dewatering}{consolidation}\right\} \;\Rightarrow\; 0.41\,mm\,@\,1{,}200g/l$

Again, dewatering and consolidation does not alter the dry bulk DDT concentrations in the post-flood muddy deposits, which will remain at 310 μ g/kg even if these muds consolidate to full saturation.

Plots of the deposition flux and deposition thickness time series for the other scenarios of the sensitivity analysis are found in APPENDIX-A. The complete ensemble of deposition scenarios from this sensitivity analysis are summarized in Table 5 below. Entries in the last three rows are based on the assumption of no erodible fine-grained sediments anywhere else in the Otay River watershed outside of the contaminated area adjacent the ORERP Floodplain Tidal Basin. While the DDT concentrations in the muds deposited under these scenarios of no upstream sources can range as high as 310 μ g/kg to 790 μ g/kg, the deposition thicknesses reduce to only fractions of a millimeter once these muds become consolidated (cf. column_8, Table 5). To assess the potential biological impacts of these simulation results, a risk assessment analysis based on screening levels of keystone wetland species is presented in Section 6 below.

6) Post-Flood Tidal Deposition Simulations for the 50-Year Flood:

6.1 Input Assumptions: The 50-year flood hydrographs for the Otay River, Poggi Canyon Creek and Nestor Creek are triangular with 24-hour durations, similar to those shown in Figure 1 for the 100-year flood, but involving significantly less flow volumes. The total flow volume during a 50-year flood for the Otay River is 19,200,000 cubic yards (cy), or 14,679,545 cubic meters (m^3). The corresponding flow volumes for Poggi Canyon Creek and Nestor Creek are respectively 1,488,000 cy (1,137,664 m^3) and 1,584,000 cy (1,211,062 m^3), so that the combined flow through the floodplain is $\overline{Q}$ = 22,272,000 cy (17,028,272 m^3), or 13,805 acre ft. The flow for the Otay River is an order of magnitude higher than those for Poggi Canyon Creek and Nestor Creek. It was estimated that during the 50-year flood, only about 60% of the Otay and Poggi flow would pass through the proposed wetland restoration areas, while the remainder would flow through the adjacent salt ponds and into South San Diego Bay. The percent of Nestor flow that would pass through the wetland was not analyzed, but since Nestor Creek directly flows into the proposed wetland area, it was assumed that all of the Nestor Creek flow would enter the wetland. Based on these flow volumes and the sediment stratigraphy revealed by the borings taken by Anchor 201, it was estimated that the 50-year flood would erode the top 1 ft of soil over the entire ORF. The eroded volume of soil in the ORF due to the 50-year flood was estimated to be 114,900 cy (87,848 m^3), of which 21.1% (24,260 cy or 18,545 m^3) are DDT bearing fin grained sediments. The average dry bulk DDT concentration in these fine grained sediments is 790 µg/kg.

Scenario	Volume of Eroded DDT-Bearing Fines	Average DDT Conc. in DDT-Bearing Fines	Volume of Eroded Upper Watershed Fines	Flood Flow Volume	Suspended Sediment Conc.	Initial Post-Flood Deposition Thickness (200 g/l Mud)	Final Post-Flood Deposition Thickness (1,200 g/l Mud)	DDT Conc. in Post-Flood Mud Deposition (dry bulk)
Erode top 3 ft. of Contaminated Area + Upper Watershed*	128,300 cubic yards	310 μ g/kg	438,000 cubic yards	24,290 acre ft	23.15 g/l.	3.3 mm to 8.0 mm	0.5 mm to 1.4 mm	70.2 μ g/kg
Erode top 1 ft. of Contaminated Area + Upper Watershed	24,260 cubic yards	790 μ g/kg	438,000 cubic yards	24,290 acre ft	18.90 g/l	2.7 mm to 6.5 mm	0.4 mm to 1.1 mm	41.5 μ g/kg
Erode top 2 ft. of Contaminated Area + Upper Watershed	76,350 cubic yards	430 μ g/kg	438,000 cubic yards	24,290 acre ft	21.03 g/l	3.0 mm to 7.3 mm	0.45 mm to 1.3 mm	63.8 μ g/kg
Erode top 3 ft. of Contaminated Area Only**	128,300 cubic yards	310 μ g/kg	0 cubic yards	24,290 acre ft	5.25 g/l	0.75 mm to 1.8 mm	0.17 mm to 0.41 mm	310 μ g/kg
Erode top 1 ft. of Contaminated Area Only	24,260 cubic yards	790 μ g/kg	0 cubic yards	24,290 acre ft	0.99 g/l	0.14 mm to 0.34 mm	0.02 mm to 0.06 mm	790 μ g/kg
Erode top 2 ft. of Contaminated Area Only	76,350 cubic yards	430 μ g/kg	0 cubic yards	24,290 acre ft	3.12 g/l	0.44 mm to 1.1 mm	0.07 mm to 0.12 mm	430 μ g/kg

Table 5: Matrix of Sensitivity Analysis of Potential DDT Deposition in the ORERP post-100 year flood.

The 50-year flood will cause additional soil erosion from the watershed below the Savage Dam. Based on scaling by the watershed size relative to the Buena Vista watershed, it was estimated that sediment discharge from the Otay River watershed below Savage Dam during the 50-yr flood is about 501,000 cy of which 50% is fine, or 250,500 cy. Because only 60% of flow from the upper Otay River watershed would pass through ORF, the eroded contaminated sediments from the ORF could mix with as much as 150,300 cy (114,913 m^3) of fines not known to contain DDT from the upper watershed below the Savage Dam.

From this assessment of possible sediment erosion input assumptions, we pose a sensitivity analysis for the post 50-Year flood DDT deposition that is based on erosion fluxes from one possible erosion depth (1 ft.) in the DDT contaminated area of the floodplain that is each combined with two possible fluxes of clean fines (0 cy and 150,300 cy) from the upper watershed below the Savage Dam; yielding a sensitivity analysis comprised of two separate deposition scenarios. The ensembles of input parameters for this sensitivity analysis are summarized in Table 6.

Table 6: Input Parameters for Sensitivity Analysis of Post 50-Year Flood DDT Deposition

Scenario	Volume of Eroded DDT-Bearing Fines	DDT Conc. in DDT-Bearing Fines	Volume of Eroded Upper Watershed Fines	Flood Flow Volume	Suspended Sediment Conc.
Erode top 1 ft. of Contaminated Area + Upper Watershed	24,260 cubic yards	790 μ g/kg	150,300 cubic yards	13,805 acre ft	12.60 g/l
Erode top 1 ft. of Contaminated Area Only*	24,260 cubic yards	790 μ g/kg	0 cubic yards	13,805 acre ft	1.8 g/l

The suspended sediment concentrations in Table 6 are based on a dry bulk density for eroded soil of 2700 lb per cy, or 1.225 metric tons per cy; where a metric ton is 1000 kg. This conversion factor is applied to the sum of the volume of eroded DDT- bearing fines (column_2) and the volume of eroded fines from the upper Otay watershed (column_4) to obtain the total flux of suspended fine grained sediment in tons/day during the 24 hour flood period of the 50-year flood. The sand and gravel sized fractions eroded from the floodplain by the 50 year flood are assumed to be transported as bed load. The suspended sediment flux component (column_2 + column_4) is divided by the flow volume of $\overline{Q}$ = 17,028,272 m^3 during the 24 hour flood period to give the average suspended sediment concentration in column_6 upon conversion of metric tons to grams and cubic meters to liters.

6.2 Deposition Results: Plots of the deposition flux and deposition thicknesses in the ORERP tidal basins for the 50-year flood scenarios are found in Figures 13 through Figure 16. The complete ensemble of 50-year flood deposition scenarios from this sensitivity analysis are summarized in Table 7 below. With initial dilution from mixing with the clean sediments from upstream sources, DDT concentrations post-50 year flood in the tidal basins of the ORERP are on the order of 110 μ g/kg. This concentration is higher than the companion result for the 100-year flood in row_1, column_9, Table 5. This is due to the fact that the 50-year flood causes proportional less erosion in the upper water shed of the Otay River than the 100 year flood. Entries in the last row of Table 7 are based on the assumption of no erodible fine-grained sediments anywhere else in the Otay River watershed outside of the contaminated area adjacent the ORERP Floodplain Tidal Basin and represent *worst case* for the 50-year flood. While the DDT concentrations in the muds deposited under worst case scenarios of no upstream sources can range as high as 790 μ g/kg, the deposition thicknesses are initially only 0.62 mm to 0.26 mm reduce to only fractions of a millimeter (0.06 mm to 0.14 mm) once these muds become consolidated (cf. column_8, Table 7). However, the DDT deposition results for 50-yr were found to be within the range of those for the 100-yr flood, so that the conclusions put forth previously in Section 6 on potential flood-induced DDT impacts to the ORERP wetlands ecology are upheld.

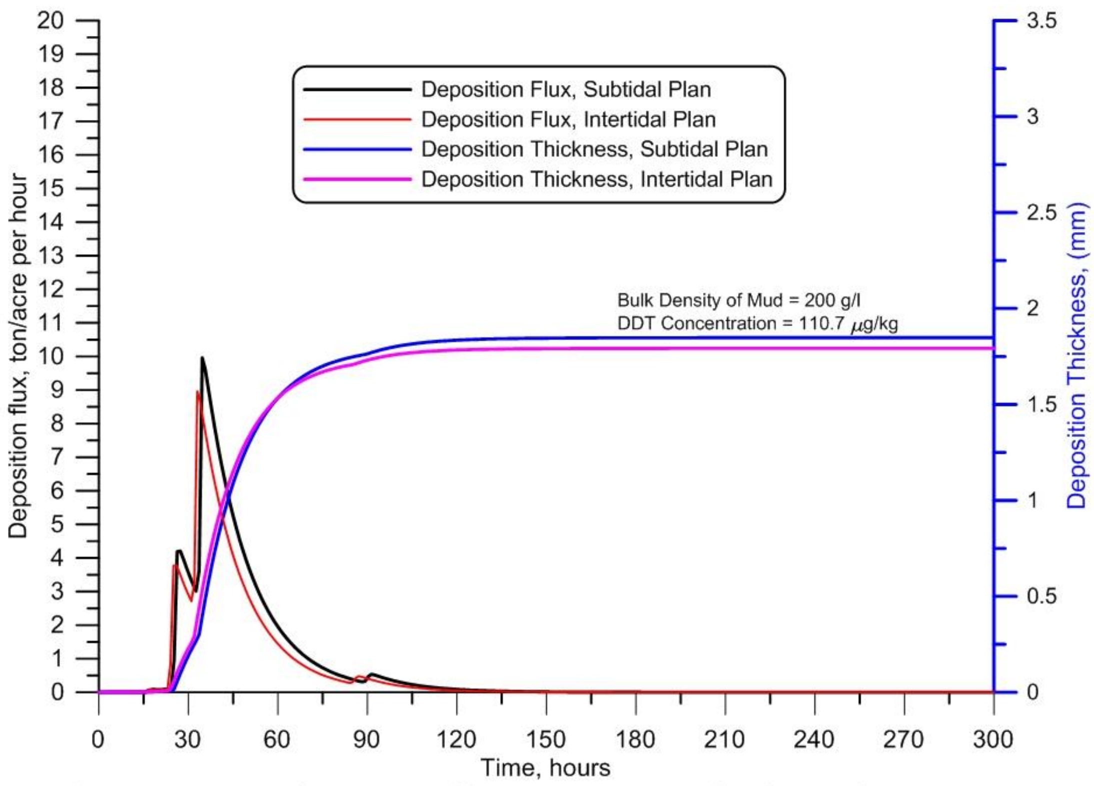

Figure 13. Sensitivity analysis of deposition of fine-grained sediment (mud) in the Otay River Floodplain Tidal Basin following the 50-year flood. Deposition flux (black & red); deposition thickness (blue & magenta). Results for 24,500 cubic yards of contaminated fines from erosion of the top 1 ft. and 150,300 cubic yards of clean sediment from the upper Otay River watershed. DDT given as dry bulk concentration.

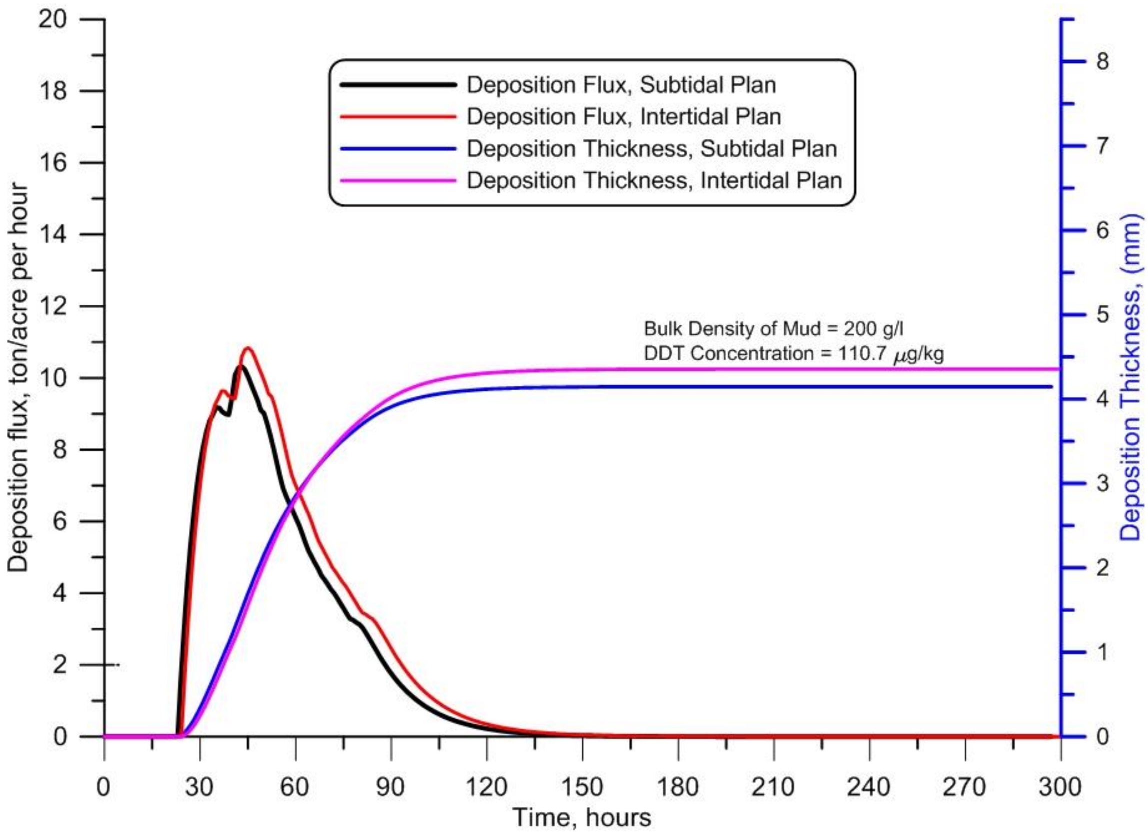

Figure 14. Sensitivity analysis of deposition of fine-grained sediment (mud) in the Pond-15 Tidal Basin following the 50-year flood. Deposition flux (black & red); deposition thickness (blue & magenta). Results for 24,500 cubic yards of contaminated fines from erosion of the top 1 ft. and 150,300 cubic yards of clean sediment from the upper Otay River watershed. DDT given as dry bulk concentration.

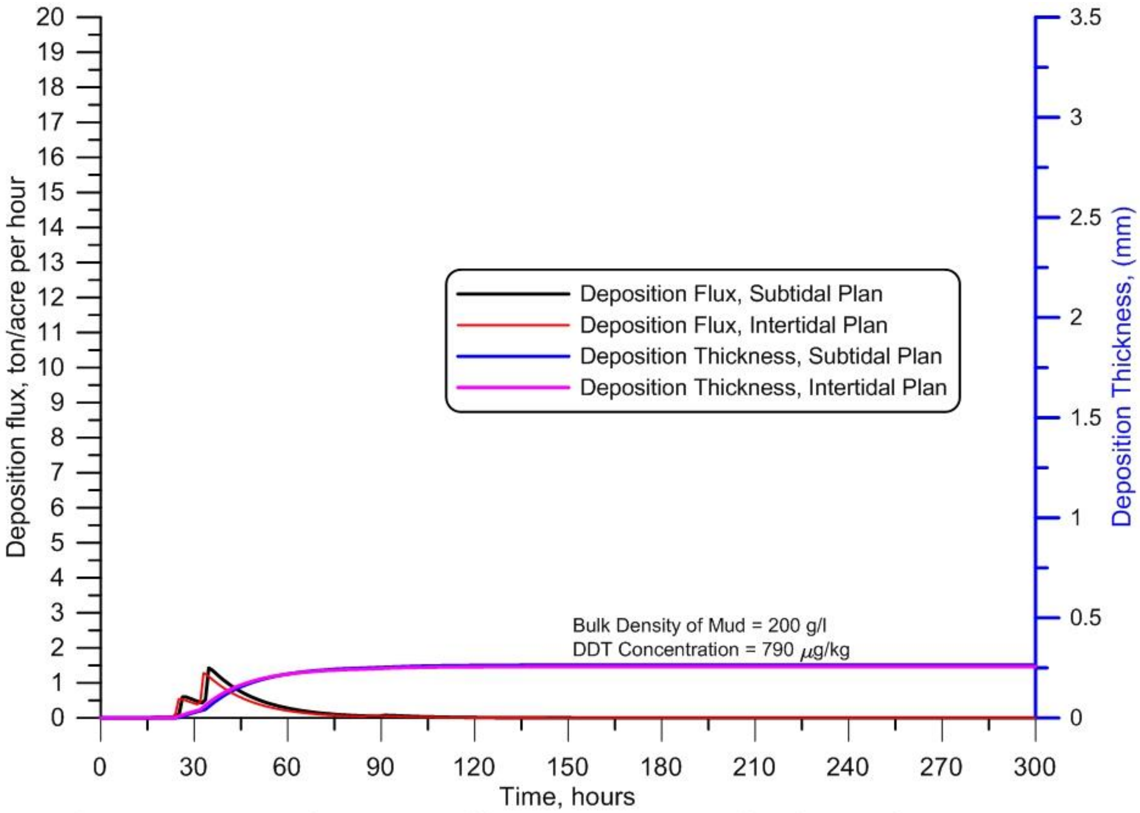

Figure 15. Sensitivity analysis of deposition of fine-grained sediment (mud) in the Otay River Floodplain Tidal Basin following the 50-year flood. Deposition flux (black & red); deposition thickness (blue & magenta). Results for 24,260 cubic yards of contaminated fines from erosion of the top 1 ft. and no additional clean sediment from the upper Otay River watershed. DDT given as dry bulk concentration.

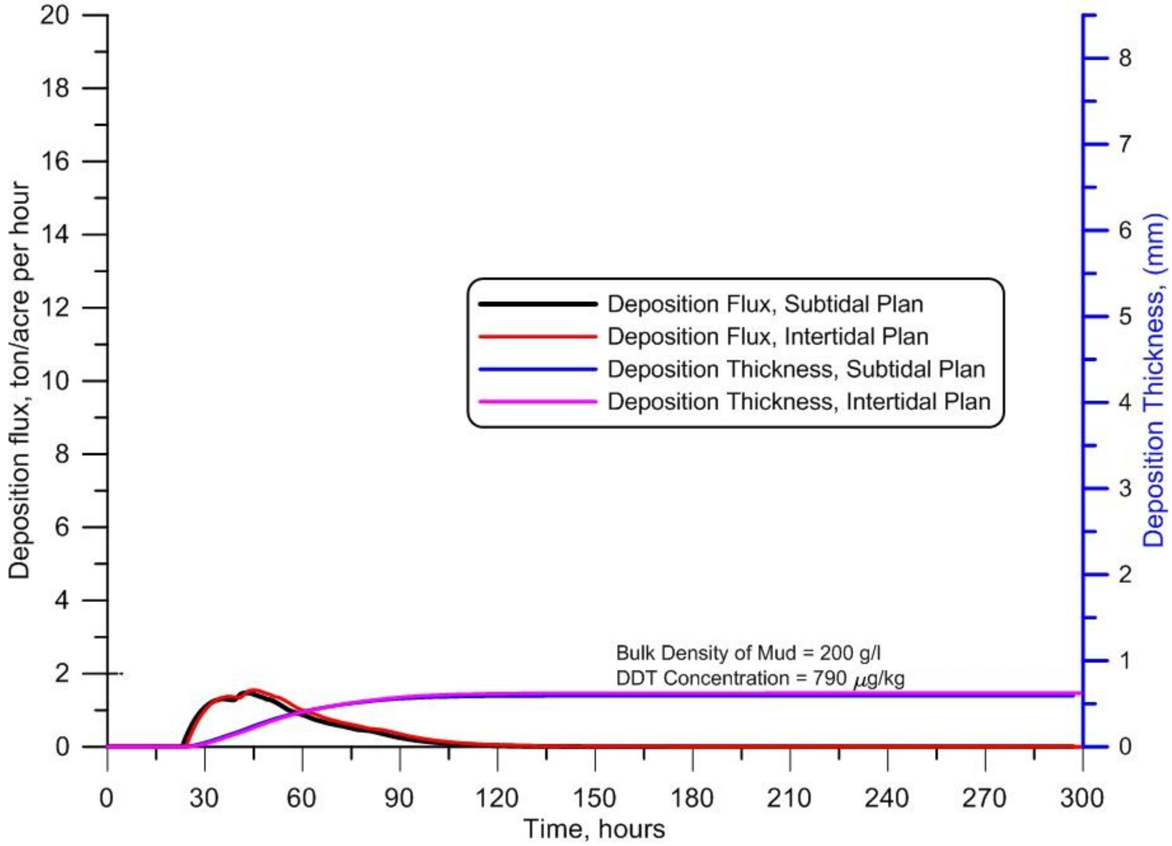

Figure 16. Sensitivity analysis of deposition of fine-grained sediment (mud) in the Pond-15 Tidal Basin following the 50-year flood. Deposition flux (black & red); deposition thickness (blue & magenta). Results for 24,500 cubic yards of contaminated fines from erosion of the top 1 ft. and no additional clean sediment from the upper Otay River watershed. DDT given as dry bulk concentration.

Scenario	Volume of Eroded DDT-Bearing Fines	Average DDT Conc. in DDT-Bearing Fines	Volume of Eroded Upper Watershed Fines	Flood Flow Volume	Suspended Sediment Conc.	Initial Post-Flood Deposition Thickness (200 g/l Mud)	Final Post-Flood Deposition Thickness (1,200 g/l Mud)	DDT Conc. in Post-Flood Mud Deposition (dry bulk)
Erode top 1 ft. of Contaminated Area + Upper Watershed	24,260 cubic yards	790 μ g/kg	150,300 cubic yards	13,805 acre ft	12.60 g/l	1.8 mm to 4.3 mm	0.30 mm to 0.72 mm	110.7 μ g/kg
Erode top 1 ft. of Contaminated Area Only	24,260 cubic yards	790 μ g/kg	0 cubic yards	13,805 acre ft	1.8 g/l	0.26 mm to 0.62 mm	0.06 mm to 0.14 mm	790 μ g/kg

Table 7: Matrix of Sensitivity Analysis of Potential DDT Deposition in the ORERP post-50 year flood.

*Entries in RED are based on the assumption of NO erodible fine-grained sediments anywhere else in the Otay River watershed below Savage Dam outside of the contaminated area adjacent the ORERP Floodplain Tidal Basin.**,** and represent *Worst-Case* for the 50-year flood

7.0) Biological Implications of the Post-Flood Deposition Simulations

The approach used for this analysis was to focus on critical and applicable information. The focus was on sensitive and potentially most exposed species, and data that would be applicable to the specific area (i.e., salt marshes in San Diego Bay). A risk assessment approach was used to identify wildlife risk-based screening levels for DDT in salt marsh sediment. A screening level approach was used, in that estimates were based on most exposed and/or sensitive species, and conservative assumptions used when there was uncertainty. This analysis entails the identification of no-effects based screening levels (doses and dietary concentrations) for birds, and factors that can be used to relate DDT concentrations in the bird's diet (specifically marsh invertebrates and forage fish) to concentrations in sediment. The availability of applicable data is greatest for effect levels in birds, while data on biota/sediment relationships are limited, especially for forage fish. Consequently, while it is possible to identify conservative dietary screening levels for avian receptors, whether factors used to relate DDT concentrations in biota to DDT concentrations in sediment are particularly conservative is not possible to tell at this time. In other words, this is not necessarily a worst case relative to this element of the analysis.

7.1 Screening Levels For DDT and Metabolites In Salt Marsh Sediment Relative to Proposed ORERP Activities: DDT in environmental media usually occurs as a mixture of parent compound (p,p'-DDT), and impurities and metabolites (i.e., o,p'-DDT, o,p'-DDD, o,p'-DDE, p,p'-DDD and p,p'-DDE). The metabolite, p,p'-DDE is the most persistent and the dominant of the six isomers (forms) in biological samples and in environmental media where there have been no recent DDT applications. The p,p'-DDE isomer is also the one associated with the most sensitive adverse effects in avian species. Consequently, some studies focus on p,p'-DDE only, while others consider the sum of the six isomers (total DDT). Data from studies on p,p'-DDE were considered in the development of the sediment screening levels. However, because of concerns about ongoing conversion of DDT to DDE, and because isomers other than p,p'-DDE are associated with adverse effects, sediment screening levels are used for comparison with total DDTs even though they are derived based on data for the most sensitive effects that are associated with p,p'-DDE.

Sediment-borne DDT and its metabolites (especially p,p'-DDE) can be toxic to directly exposed benthic organisms, and to indirectly exposed aquatic-dependent wildlife. Sediment-borne DDT and metabolites are known to enter and accumulate in the tissues of aquatic food web organisms. Through bioaccumulation and biomagnification (with trophic transfer), concentrations of DDT and metabolites can reach levels in tissues of aquatic food chain organisms that are unsafe for wildlife that rely on the aquatic biota for food. Sediment screening levels for DDT and metabolites must consider; 1) potential for toxicity to benthic invertebrates, and 2) potential for uptake and food chain transfer and therefore adverse effects via dietary exposure among aquatic-dependent wildlife.

The focus of this exercise is on avian species because marsh habitats on San Diego Bay NWR: 1) are specifically managed for federally listed species (birds and one plant) and migratory birds[1], and 2) do not support mammalian or reptile species of concern nor other species that (based on feeding habits) are likely to experience significant exposure to sediment-borne DDT. Avian species that are present during the nesting season are of particular concern because DDT (specifically p,p'-DDE) impairs eggshell production by adult females (thin shells) and, because it is readily transferred to eggs, may adversely affect developing embryos. Eggshell thinning is a well-documented effect in many species of birds, and it may be one of the most sensitive of sub-lethal effects leading to population-level impairments. Sensitivity to the thinning effects of p.p'-DDE varies among species. Species that are less sensitive to eggshell thinning may be at risk of endocrine disrupting effects of o,p'-DDT on developing embryos (e.g., developmental feminization) (Fry and Toone 1981). It is assumed that screening levels based on the toxicity of p,p-DDE but applied to total DDTs will protect against adverse effects associated with any of the isomers.

7.2: Wildlife Receptors: Two species of birds were considered as representatives of potentially most exposed aquatic-dependent wildlife to DDT in marsh sediments: One is the light-footed Ridgeway's rail (*Rallus obsoletus longirostrus* or LFRR; formerly light-footed clapper rail), and the other is the snowy egret (*Egretta thula*).

1. The LFRR is a federally endangered bird that is a year-round resident of salt marshes of coastal southern California, including at the San Diego Bay NWR. LFRR forage for food in vegetated marsh and tidal creek channels by gleaning and probing for benthic organisms. Their primary foods are snails and crabs, but they are opportunistic and will eat bivalves, shrimps, worms and fish (Zembal and Fancher 1988). LFRR exposure to sediment-borne DDT is almost completely via diet, but there may be some exposure via incidental ingestion of sediment while foraging as well. The LFRR is larger than two other rallid species with similar feeding habitats that might occur in the restored salt marsh (i.e. the Sora and the Virginia rail), but only infrequently and generally not during the nesting season (SDSU San Diego Bird Atlas). However, it is the same size or smaller, therefore has equal or greater nutritional needs, than most species with similar feeding habits that commonly forage in San Diego Bay salt marshes (e.g., willet, long-billed curlew and whimbrel). Given the estimated nutritional needs, and year-round residency, the LFRR is considered a reasonably conservative representative of marsh birds that rely on resident mid-trophic level invertebrates for food, and will be exposed to site-specific DDT during the nesting season.

2. The snowy egret is a wading bird that can be found foraging for fish in San Diego Bay marshes while nesting in colonies at nearby locations. The snowy egret mainly eats fish, but may opportunistically consume invertebrates and small terrestrial vertebrates. Because most of their diet is fish, snowy egrets are considered upper trophic level aquatic-dependent predators that may encounter even higher DDT concentrations in their diet than will species such as the LFRR. The snowy egret is one of the smaller wading bird species, which include egrets, herons and bitterns, and as such has proportionally

[1] Note: the highlights are provided to bring the reader's attention to the specific steps in this analysis.

greater nutritional needs than other larger species. Because of its diet, food requirements and foraging habits, snowy egrets are considered a conservative representative of piscivorous birds given they rely on upper trophic level salt marsh biota (fish) for food and are relatively small among wading birds.

7.3 General Approach: A couple approaches were used to derive wildlife risk-based sediment screening levels, determined largely by the kinds of data available for assessing effects thresholds in birds and relating thresholds for eggs and diet to concentrations in sediment. This is provided because each approach will give somewhat different but valid results relative to the question of risk posed by the sediments.

1. Tissue targets for p,p'-DDE and/or total DDTs in avian eggs: recommend 1.5 mg $_{DDT}$/kg wet weight (ww).

Total DDT and p,p'-DDE concentrations in eggs have been related to eggshell thinning and reduced nesting success of numerous avian species. Eggshell thinning appears to be one of the most sensitive of the adverse effects in birds (i.e., occurs at lower dose levels than other adverse effects such as neurotoxicity).

1a) There are species differences in sensitivity, reflected by DDE concentrations associated with eggshells that are 20% thinner than shells collected before DDT was in heavy use (e.g., pre 1940s). This is a convenient benchmark for comparison, but this extent of thinning (15–20%), when it is persistent over several years, is associated with population level impacts in many species. DDE concentrations in eggs associated with 20% shell thinning (as mg /kg ww; from Blus 2011) include:

- 5 - 10 mg $_{DDT}$/kg ww (pelican, condor, prairie falcon, osprey, sparrowhawk, ibis)
- 10 - 20 mg $_{DDT}$/kg ww (loon, great blue heron, peregrine falcon, and merlin)
- >50 mg $_{DDT}$/kg ww (black crowned night heron and bald eagle)

1b) DDE concentrations associated with adverse effects at <20% shell thinning – pelicans (Blus 1984)

- 3.0 mg $_{DDT}$/kg ww is associated with colony collapse (= effect concentration for productivity)
- 2.0 mg $_{DDT}$/kg ww is associated with productivity that is indistinguishable from productivity observed with non-detectable DDE levels in eggs (= potential no effect concentration for productivity). This concentration may affect eggshell thickness, but not to the extent that productivity is affected.

1c) Estimated no effect threshold for eggshell thinning in sensitive species - Using regression equations from Fry (1994) relating p,p'-DDE concentration to percent of pre-DDT eggshell thickness, one can estimate DDE concentration for an eggshell with no thinning (equal to 100% of pre-DDT eggshell thickness). This would be a true no-effect level for pp'-DDE relative to all endpoints, given it applies to the most sensitive endpoint.

- 1.5 mg $_{DDT}$/kg ww for brown pelican, and 1.2 mg/kg ww for double-crested cormorant.

1d) Data specific to rails and/or snowy egrets.

- 1.0 – 2.0 mg $_{DDT}$/kg ww in CA clapper rail eggs; no effect for shell thinning (Lonzarich et al. 1992)
- 0.45 & 1.02 mg $_{DDT}$/kg ww (means; range 0.197 – 1.78) in light footed clapper rail from Tijuana Slough and Seal Beach NWRs; no effect on shell thinning (Goodbred et al 1996).
- 2.13 mg $_{DDT}$/kg ww (mean; range 0.63-5.60) in light footed clapper rail eggs from Mugu Lagoon; no effect on shell thickness relative to pre-DDT, but shells thinner than for eggs from Seal Beach and Tijuana Slough NWRs (Goodbred et al 1996). Although we have a difference between sites in terms of measured eggshell thickness, this is not likely to have been manifested in adverse effects in productivity given the eggshell measurements were, for the most part, similar to pre-DDT era eggshells.
- 0.41, 0.97 and 1.3 mg $_{DDT}$/kg ww (means; overall range 0.1 – 6.4 mg/kg ww) in clapper rail eggs collected in 1972-73 from 3 Atlantic coast locations. No effect levels for eggshell thickness compared with pre-1947, but there were location-specific variations for both pre- and post- 1947 eggs (Klaas et al 1980).
- 1.05 mg $_{DDT}$/kg ww in single light-footed clapper rail egg; eggshell thickness within range for pre-DDT use, but thinner than eggs collected at the same time and location, but with lower concentrations (~0.45 – 0.70 mg/kg ww; Sutula et al 2005).
- 1.0 – 5.0 mg $_{DDT}$/kg ww in snowy egret egret eggs; no effect on productivity (Henny et al 1985)
- 5.0 – 10 mg $_{DDT}$/kg ww in snowy egret eggs; effect level for productivity (Henny et al 1985)

These species-specific concentrations give us confidence that our proposed target in avian eggs of 1.5 mg DDT/kg ww is appropriately protective for the rail and the egret.

Based on concentrations of DDE/total DDT in eggs, pelicans are among, if not the most sensitive species for eggshell thinning (compared with pre-DDT use) and productivity effects of DDT (primarily DDE). For pelicans, productivity in the field is impacted @ 3.0 mg/kg ww, but DDE-related impacts are not detectable @ 2.0 mg/kg ww, and an estimated no effect level for eggshell thinning is 1.5 mg/kg ww.

For rails, no shell thinning (compared with pre-DDT eggs) has been detected with mean concentrations of 1.02 mg/kg ww (Goodbred et al. 1996), and 1.3 mg/kg ww (Klaas et al. 1980). There is limited information to suggest that subtle thinning (but not different from pre-DDT eggs) may occur with concentrations as low as ~1.0 mg/kg ww. But the effect may be due to population-related variation in shell thickness or statistical artifact. The available data suggest that light-footed Ridgeway's rails are no more sensitive than pelicans to the eggshell thinning and productivity effects of DDT. Data are insufficient to determine if rails are less sensitive than pelicans. In comparison, data on snowy egrets indicate that they are less sensitive to DDT than pelicans. The recommended screening level is based on no effects in pelicans and as such will be protective of other species as well.

A screening level of 1.5 mg $_{DDT}$/kg egg ww is recommended for total DDT concentration in eggs. This value is based on data for pelicans. It is considered protective of rails and is within the range of no effect levels for snowy egrets.

2. Tissue targets for p,p'-DDE and/or total DDTs in avian diets (mg $_{DDT}$/kg fish or invertebrate ww).

2a) Combining the screening level for eggs (1.5 mg $_{DDT}$/kg egg ww), with egg-to-diet concentration ratios. Wet weight-based ratios were used, consistent with concentrations and ratios reported in the literature.

- Egg/invertebrate ratios in clapper rail studies - rail egg/crab ratios ~25 (Goodbred et al. 1996 & Foehrenrich et al. 1972), and rail egg/snail ratio ~73 (Foehrenrich et al. 1972). Given the target concentration of 1.5 mg/kg in rail eggs, corresponding target concentrations in crabs is 0.06 mg/kg ww and in snails, it would be 0.021 mg/kg ww, or an overall average of 0.03 mg/kg ww, assuming a 50:50 mix.
- Egg/forage fish ratios in studies of piscivorous birds - egg/fish ratios are generally between 20 and 60 (Davis et al 2007). Values of 32 to 45 have been reported for herring gulls on Lake Ontario (Braune and Norstrom 1989), and values between 15 and 32 are indicated by data for California brown pelican (Anderson et al. 1975). In one study by Zeeman et al (2008), the average concentration of total DDT in forage fish from South San Diego Bay was 0.042 mg/kg ww. Corresponding bird egg/fish concentration ratios were 43 using black skimmer and Caspian tern eggs and approximately 10 using elegant and California least tern eggs. If using the geometric mean concentration for all seabird egg samples (1.08 mg $_{DDT}$/kg egg ww), the average ratio is 25. With a target DDT concentration of 1.5 mg $_{DDE}$/kg egg ww, the target DDT concentration in forage fish consumed by egrets (or other piscivorous birds) based on the ratios of 10-43 identified above would be between 0.150 mg/kg ww and 0.034 mg/kg ww, or an overall average (based on the mean of 25) of 0.060 mg $_{DDT}$/kg fish ww.

Ratios used to estimate dietary screening levels from the avian egg screening level, are averages. For rails, geometric mean concentrations for snails and crabs were used. Similarly, for piscivorous birds, geometric mean concentrations of multiple species of forage fish were used. This was done because (1) data are limited, and (2) birds generally consume a variety of species. Also, data from four species of piscivorous birds were combined to produce a geometric mean concentration of DDT in bird eggs. This was done to simplify the analysis (using an average rather than a range), and we deemed it appropriate given we know that the snowy egret is not among the most sensitive species.. The outcome (estimated dietary concentration) is less conservative than what the worst case value would be, but the difference is less than 2-fold. If you assume the worst case at every step, it is possible to end up with a totally protective, yet totally unrealistic, result. We were trying to strike a balance between these two. Overall, the egg/diet ratios used for estimates in this analysis are: Rail eggs/invertebrates = 50 and piscivorous bird egg/fish = 25.

2b) Reference dose (TRV)-based (combined with food ingestion rates estimated from Nagy 2001)

- TRV @ 0.014 mg/kg-d (a hybrid approach using field data, and therefore some uncertainty about actual concentrations in diet): This TRV is a chronic value for California brown pelican, a species known to be sensitive to these effects (USEPA 1995), adjusted downward by a Lowest Observed Adverse Effect Level (LOAEL) to No Observed Adverse Effect Level (NOAEL) uncertainty factor of 2.0 (based on observed low effect- and estimated no effect concentration in egg for eggshell thinning), combined with an egg/diet concentrations ratio of 32X (from Anderson et al. 1975). Using ingestion rates from Nagy (2001), combined with a TRV of 0.014 mg/kg-d, the estimated dietary screening level for LFRR (concentrations in invertebrates) is 0.027 mg $_{DDT}$/kg ww and the screening level for snowy egret (concentrations in fish) is 0.029 mg $_{DDT}$/kg ww.

- TRV @ 0.227 mg/kg-d (from lab studies with known concentrations in diet): Highest bounded NOAEL lower than the lowest bounded LOAEL for effects on growth, reproduction and survival in multiple avian species including waterfowl and double-crested cormorants (a sensitive species; EPA ECO-SSL). It is equal to or less than bounded and unbounded NOAELs for biochemical effects, pathology, survival and growth in sub-chronically (9 week) exposed double-crested cormorants. Other than cormorants and kestrels, most of the species represented by the TRV are not among the most sensitive (Item 1a above). Consequently, this TRV is considered an upper bound of no effects-based TRVs. This approach is a reasonable one to use in assessing risk more broadly among species, as it is not based on the most sensitive endpoints nor on the most sensitive species. Using ingestion rates from Nagy (2001), combined with a TRV of 0.227 mg/kg-d, the estimated upper bound dietary screening level for LFRR (concentrations in invertebrates) is 0.432 mg $_{DDT}$/kg ww and the screening level for snowy egret (concentrations in fish) is 0.465 mg $_{DDT}$/kg ww.

2c) Literature values: 3.0 mg/kg ww: Concentration in avian diet which could cause adverse impacts (Goodbred et al. 1996)

Table 8: Screening levels for total DDT in marsh bird diets ($mg_{total\ DDT}$/kg diet ww)

Approach	Rails – Concentration in invertebrates
Egg SL/invertebrate ratio+*	0.030
Dose rate (hybrid)*	0.027
Dose rate (lab based)**	0.432
Approach	**Egrets – Concentration in forage fish**
Egg SL/fish ratio+*	0.060
Dose rate (hybrid)*	0.029
Dose rate (lab based)**	0.465

+ Based on field collections from southern California
* Based on No Observed Adverse Effects Levels in most sensitive species
** Based on No Observed Adverse Effects Levels in a few studies on most sensitive, but primarily in studies on less sensitive species; considered here as an upper bound no observed adverse effect level for avian species that forage in salt marsh habitats.

3. Sediment targets for total DDTs

3a) Benthic community:

ER-L = 0.00158 mg/kg dry weight (dw) and ER-M = 0.0461 mg/kg dw, (Long et al 1995). These two guidelines delineate three concentration ranges: concentrations below the ER-L represent "minimal-effects range" (adverse effects rarely observed), concentrations between the ER-L and ER-M represent a "possible effects range" (adverse effects may occur occasionally), and concentrations equal to or greater than the ER-M represent the "probable effects range" and at which effects to benthic invertebrates would frequently occur. (Note: the effect levels are considered to apply to an “active zone” that is 20 mm deep. These benchmarks would not be applicable to a thin layer such as that associated with our modeled sediment deposition as that thin layer is not biologically meaningful to the species and circumstances evaluated in this compellation.)

3b) Reference concentrations for San Diego Bay, with the term "reference" representing DDT concentrations measured in sediments from San Diego Bay, and not in the immediate vicinity of known contaminated sites

0.001 mg/kg ww, or between 0.0013 and 0.0016 mg/kg dw. These are geometric mean concentrations from the USFWS south San Diego Bay mudflats study (unpublished) and the F&G Street Marsh study (Zeeman et al. 2008a)

3c) Wildlife risk-based sediment screening levels using target concentrations in invertebrates and forage fish, combined with biota/sediment ratios (data are very limited)

Ratios are wet weight-based using geometric mean concentrations. USFWS south San Diego Bay mudflats study (unpublished) California horn snail/sediment = 2.5, fiddler crab/sediment = 6.8, and forage fish/sediment = 27. The ratio for invertebrates in general (fiddler crabs and snails combined) = 4.1. Goodbred et al. (1996) report shore crab/sediment ratios of 1.3 and 2.2, for two southern California salt marshes. (dry weight-based ratios available in Sutula et al. 2005; wet weight-based ratios would be lower than reported). These are the actual relationships derived from the field data collected by the Carlsbad Fish and Wildlife Office.

Overall, the biota/sediment ratios used for estimates in this analysis are: 3.0 for invertebrates /sediment and 27 for forage fish/sediment (all wet weight). The former ratio is another case where we avoided pursuing the worst case scenario into what would be an unrealistic result. We know that rails do eat more than one prey type.

Table 9: Inputs and estimates of wildlife risk-based screening levels for DDT in marsh sediment

Dietary screening levels (mg DDT/kg diet ww)	diet/sediment ratio (ww / ww)	Sediment screening level (mg $_{DDT}$/kg sediment ww)	Sediment screening level (mg $_{DDT}$/kg sediment dw)*
Rails (invertebrates)			
0.030	3	0.010	0.017
0.027	3	0.009	0.015
0.432	3	0.144	0.240
Snowy egrets (fish)			
0.060	27	0.002	0.003
0.029	27	0.001	0.002
0.465	27	0.017	0.028

* wet weight-dry weight conversion based on geometric mean moisture contents for sediment samples from the south San Diego Bay mudflats study (=35) and in F&G street marsh study (=43%);

Table 10: Summary of estimated wildlife risk-based screening levels for DDT in salt marsh sediments, San Diego Bay NWR

Dietary screening levels (mg/kg ww)		Sediment screening levels (ug/kg dw)[#]
Rails - Concentration	Approach	Concentration
0.030	Egg SL/invertebrates ratio*	17
0.027	Dose rate (hybrid)*	15
0.432	Dose rate (lab based)**	240
Egrets - Concentration	Approach	Concentration
0.060	Egg SL/fish ratio*	3
0.029	Dose rate (hybrid)*	2
0.465	Dose rate (lab based)**	28

For comparison: more broadly, surficial sediments in San Diego Bay have concentrations of 1.3-1.6 ug/kg dw, ER-L = 1.58 ug/kg dw, and ER-M = 46.1 ug/kg dw .
* Based on No Observed Adverse Effects Levels in most sensitive species
** Based on No Observed Adverse Effects Levels in a few studies on most sensitive, but primarily in studies on less sensitive species; considered here as an upper bound no observed adverse effect level for avian species that forage in salt marsh habitats.

7.4) Risk Assessment of DDT Deposition in the ORERP for the 100-Year Flood: The results of the first deposition scenario, (cf. row_2 of Table 5; Figures 9 & 10) were used as the starting point for the risk evaluation. This evaluation considers the potential for sediment concentrations of DDTs to impact the benthic organisms and thus the prey base for aquatic dependent wildlife and the potential for bioaccumulation of these compounds to result in impacts on the aquatic-dependent birds that are expected to use the restored areas. In evaluating these concerns, we needed to take into consideration not only the concentration of DDTs in the deposited materials, but how those deposited materials would result in exposure by the benthic organisms. For this element of the evaluation, we calculated exposure concentrations in the context of a vertical sediment layer. We assumed that sediments exposed by the restoration, but before deposition of flood-associated particles, have low levels of DDT equal to what has been observed in sediments from mudflats and marshes of south San Diego Bay (see notes in Table 8 above).

The vertical layer that was used was 20 mm (2 cm), as that thickness is used as the “active layer” for a variety of studies related to evaluation of sediment toxicity, including laboratory bioassays and in-situ mussel data (Long et al. 1995), and was deemed reasonable to represent the potential trophic relationships for the species evaluated here. The model outputs included the estimated depths of deposition of the contaminated materials in addition to the estimated concentration (70.2 ug/kg dw). In consideration of the range of particle sizes and the locations in which deposition would occur, the model results in Figures 9 & 10 indicated that a 3.3 to 8.0 mm layer of contaminated material would be deposited in restored areas over clean sediments (as based on soil and sediment sampling at depth). Over time, this would become fully consolidated into a layer 0.55 to 1.4 mm thick (Table 5, row-2, column_8). Using a depth-proportional exposure approach, assuming all exposure occurs within the top 20 mm, we calculated that the contamination experience by the benthic biota would range from approximately 13 to 29 ug/kg (dw) initially and would decrease with settlement to a final 20 mm-based concentration of 3.5 to 6.4 ug/kg (dw), see Table 9, row_2, column_10. While this approach does not take into consideration the potential effects of sediment density on the foraging behaviors of benthic organisms (and any resultant changes in exposure), we see this as a reasonable way to incorporate the thickness of the deposited material into our consideration of near-term and long-term potential effects in the restored areas (note that colonizing benthic organisms are not likely to be present in the early stages of settling). Given many benthic species burrow and forage to considerably deeper depths within the sediments, thus averaging the exposure over much thicker layers of clean sediment, we considered this to be a conservative approach.

Results for the 20 mm-based concentrations of the worst-case sediment deposition scenario appear in red font in column_10 of Table 9. The estimated post-flood DDT concentration for this worst case scenario is based on the assumption that DDT- contaminated soils from the former agricultural fields are the only source of sediment settled in restored marsh following a 100-year flood (column_10, row_5 through row_7 in Table 9). These results are considered worst case because higher concentrations could only occur if sediment from other (upstream) sources, and with higher DDT concentrations than those from the former agricultural fields, were added to the mix entering the restored marshes of the ORERP. Given the mixed but predominantly urban land uses in the Lower Otay River watershed (Aspen Environmental Group 2005), suspended fine-grained sediment entering the Otay River floodplain from upstream sources are expected to have lower DDT concentrations than fines from the former agricultural fields (e.g., Mahler et al. 2006). Consequently, the estimated DDT concentration in post-flood sediments under worst the case scenario (i.e., all from the former agricultural fields) forms the upper limit on what may occur in the marsh, and actual concentrations, which include contributions from less contaminated upstream sources, will be lower. Other, lower impact cases for the worst case scenario have also been considered in column_11 and in column_12 of Table 9 where depth-proportional exposure approach of the sensitivity analysis includes bioturbation exposures occurring within the top 40 mm and top 80 mm of the muddy sediments in the tidal basins of the ORERP for comparison. The depth of bioturbation will be determined by the species that ultimately colonize the tidal basins, but we would not expect that to be less then approximately 20 mm, and it could be more than 80 mm.

The final step in this evaluation was comparing our 20 mm, 40 mm and 80 mm-based DDT concentrations to our screening values. Relative to impacts on the benthic organisms as the prey base, the maximum short-term concentrations in Table 11 (initial concentrations) of 13-29 ug_{DDT}/kg dw fall between the ER-L and ER-M values (1.58 ug/kg dw and 46.1 ug/kg dw, respectively). Thus, we would expect that impacts on benthic organisms could occur occasionally during the short-term. Given the likelihood of effects combined with the short-term nature of this condition, population level impacts are expected to be limited in nature and extent. Once post-flood muddy deposits in the ORERP have compacted and consolidated, the DDT concentrations in the top 20 mm of muddy sediment, at 4.2-7.9 ug_{DDT}/kg dw are very close to the ER-L, and even lower for the top 40 mm and top 80mm of sediment; so that negative effects are expected to be rare. This condition is not likely to have a measurable effect on the prey base for aquatic-dependent species.

In regards to the aquatic-dependent birds' exposures to contaminated prey resulting in impacts, comparison of the 20 mm-based concentrations to our screening levels indicates that these concentrations fall within the range of our highest and lowest NOAELs. Given the species known to be the most sensitive are pelicans and cormorants, which are very closely related, and our target species are not members of groups believed to be particularly sensitive, impacts on aquatic-dependent birds are unlikely to result from the anticipated deposition of DDT-contaminated sediments following a 100-year flood event.

Scenario	Volume of Eroded DDT-Bearing Fines	Average DDT Conc. in DDT-Bearing Fines	Volume of Eroded Clean Fines	Flood Flow Volume	Suspended Sediment Conc.	Initial Post-Flood Deposition Thickness (200 g/l Mud)	Final Post-Flood Deposition Thickness (1,200 g/l Mud)	DDT Conc. in Post-Flood Mud Deposition (dry bulk)	Average DDT Concentration in top 20 mm of Sediment Post-Flood		Average DDT Concentration in top 40 mm of Sediment Post-Flood *		Average Concentration in top 80 mm of Sediment Post-Flood*	
									Initial	Final	Initial	Final	Initial	Final
Erode top 3 ft. of Contaminated Area + Upper Watershed	128,300 cubic yards	310 µg/kg	438,000 cubic yards	24,290 acre ft	23.15 g/l.	3.3 mm to 8.0 mm	0.5 mm to 1.4 mm	70.2 µg/kg	13 – 29 µg/kg	3.5 – 6.4 µg/kg	7.3 – 15 µg/kg	2.5 – 4.0 µg/kg	4.4 – 8.5µg/kg	2.1 – 2.8 µg/kg
Erode top 1 ft. of Contaminated Area + Upper Watershed	24,260 cubic yards	790 µg/kg	438,000 cubic yards	24,290 acre ft	18.90 g/l	2.7 mm to 6.5 mm	0.4 mm to 1.1 mm	41.5 µg/kg	7.0 – 15 µg/kg	2.4 – 3.8 µg/kg	4.3 – 8.1µg/kg	2.0 – 2.7 µg/kg	2.9 – 4.8 µg/kg	1.8 – 2.1 µg/kg
Erode top 2 ft. of Contaminated Area + Upper Watershed	76,350 cubic yards	430 µg/kg	438,000 cubic yards	24,290 acre ft	21.03 g/l	3.0 mm to 7.3 mm	0.45 mm to 1.3 mm	63.8 µg/kg	11 – 24 µg/kg	3.0 – 5.6 µg/kg	6.3 - 13µg/kg	2.3 – 3.6µg/kg	3.9 – 7.3µg/kg	1.9 – 2.6 µg/kg
Erode top 3 ft. of Contaminated Area Only**	128,300 cubic yards	310 µg/kg	0 cubic yards	24,290 acre ft	5.25 g/l	0.75 mm to 1.8 mm	0.17 mm to 0.41 mm	310 µg/kg	13 – 29 µg/kg	4.2 – 7.9 µg/kg	7.4 – 15 µg/kg	2.9 – 4.8 µg/kg	4.5 – 8.5 µg/kg	2.3 – 3.2 µg/kg
Erode top 1 ft. of Contaminated Area Only	24,260 cubic yards	790 µg/kg	0 cubic yards	24,290 acre ft	0.99 g/l	0.14 mm to 0.34 mm	0.02 mm to 0.06 mm	790 µg/kg	7.1 – 15µg/kg	2.4 – 4.0 µg/kg	4.4 – 8.3 µg/kg	2.0 – 2.8 µg/kg	3.0 – 5.0 µg/kg	1.8 – 2.2 µg/kg
Erode top 2 ft. of Contaminated Area Only	76,350 cubic yards	430 µg/kg	0 cubic yards	24,290 acre ft	3.12 g/l	0.44 mm to 1.1 mm	0.07 mm to 0.12 mm	430 µg/kg	11 – 25 µg/kg	3.1 – 4.2 µg/kg	6.3 – 13 µg/kg	2.3 – 2.9 µg/kg	4.0 – 7.5 µg/kg	2.0 – 2.2 µg/kg
Erode top 1 ft. of Contaminated Area + Upper Watershed: 50-year event	24,260 cubic yards	790 µg/kg	150,300 cubic yards	13,805 acre ft.	12.60 g/l	1.8 mm to 4.3 mm	0.30 mm to 0.72 mm	110.7 µg/kg	11 - 25µg/kg	3.3 – 5.5µg/kg	6.5 - 13 µg/kg	2.4 – 3.6µg/kg	4.1 – 7.5 µg/kg	2.0 – 2.6 µg/kg
Erode top 1 ft. of Contaminated Area Only: 50-year event**	24,260 cubic yards	790 µg/kg	0 cubic yards	13,805 acre ft.	1.8 g/l	0.26 mm to 0.62 mm	0.06 mm to 0.14 mm	790 µg/kg	12 – 26 µg/kg	4.0 – 7.1 µg/kg	6.7 - 14µg/kg	2.8 – 4.4µg/kg	4.2 – 7.7µg/kg	2.2 – 3.0µg/kg

Table 11: Matrix of Sensitivity Analysis of Potential DDT Deposition in the ORERP post-100 and post-50 year flood events.

***** Values initially calculated for these columns were calculated incorrectly; these are the revised values (please see comparison below).

**Entries in blue are based on 50-year floods.

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APPENDIX-A: Additional Deposition Flux and Deposition Thickness Simulations Supporting Tables 5 and 11

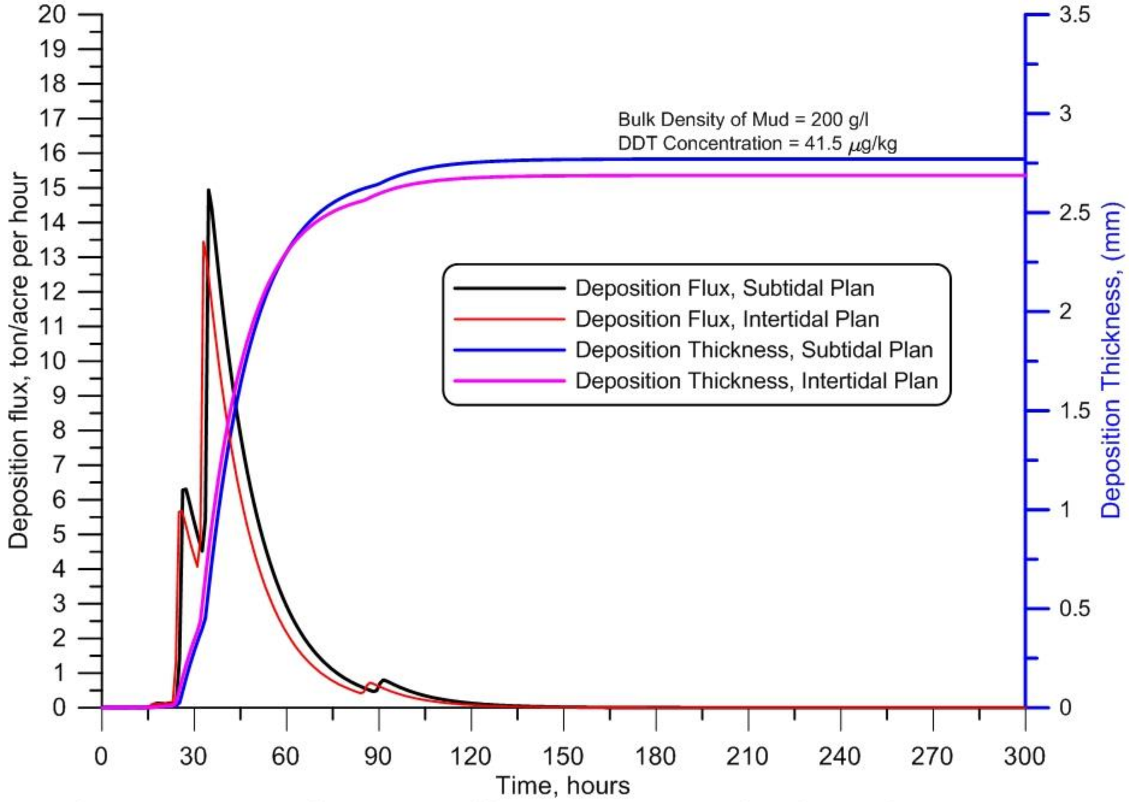

Figure A1. Sensitivity analysis of deposition of fine-grained sediment (mud) in the Otay River Floodplain Tidal Basin following the 100-year flood. Deposition flux (black & red); deposition thickness (blue & magenta). Results for 24,260 cubic yards of contaminated fines from erosion of the top 1 ft. and 438,000 cubic yards of clean sediment from the upper Otay River watershed. DDT given as dry bulk concentration.

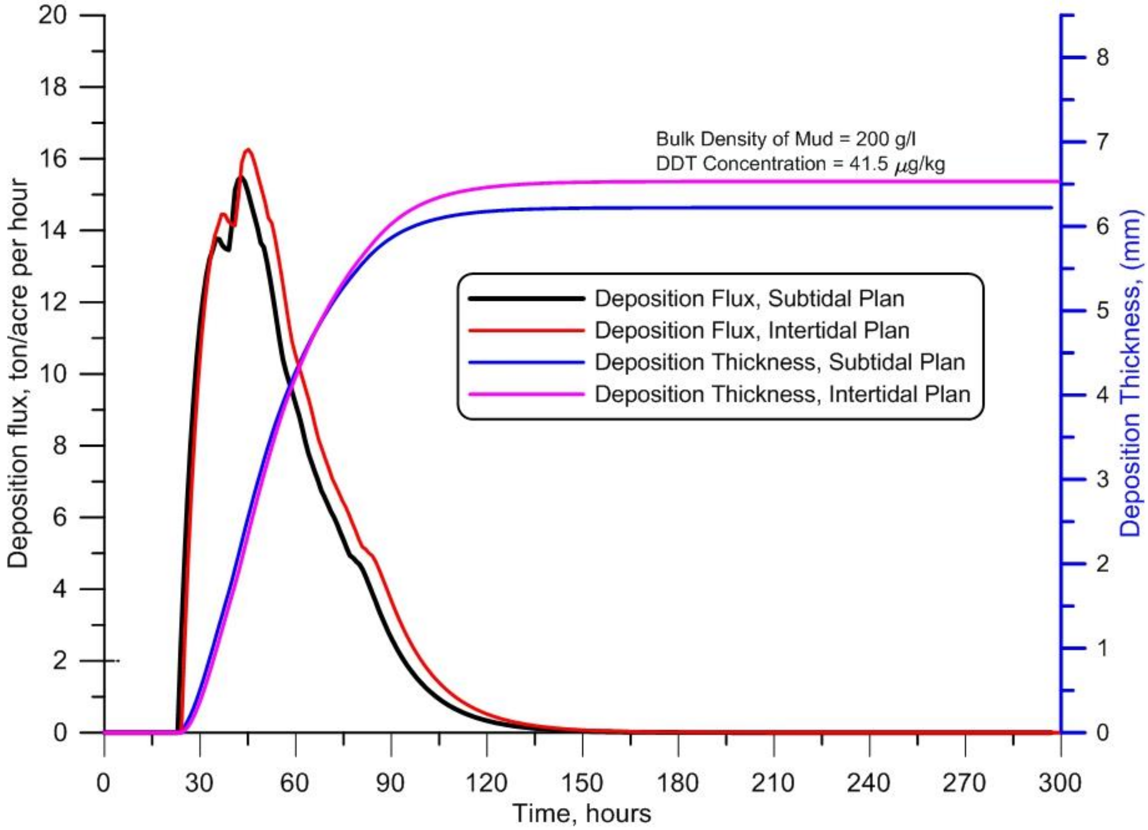

Figure A2. Sensitivity analysis of deposition of fine-grained sediment (mud) in the Pond-15 Tidal Basin following the 100-year flood. Deposition flux (black & red); deposition thickness (blue & magenta). Results for 24,260 cubic yards of contaminated fines from erosion of the top 1 ft. and 438,000 cubic yards of clean sediment from the upper Otay River watershed. DDT given as dry bulk concentration.

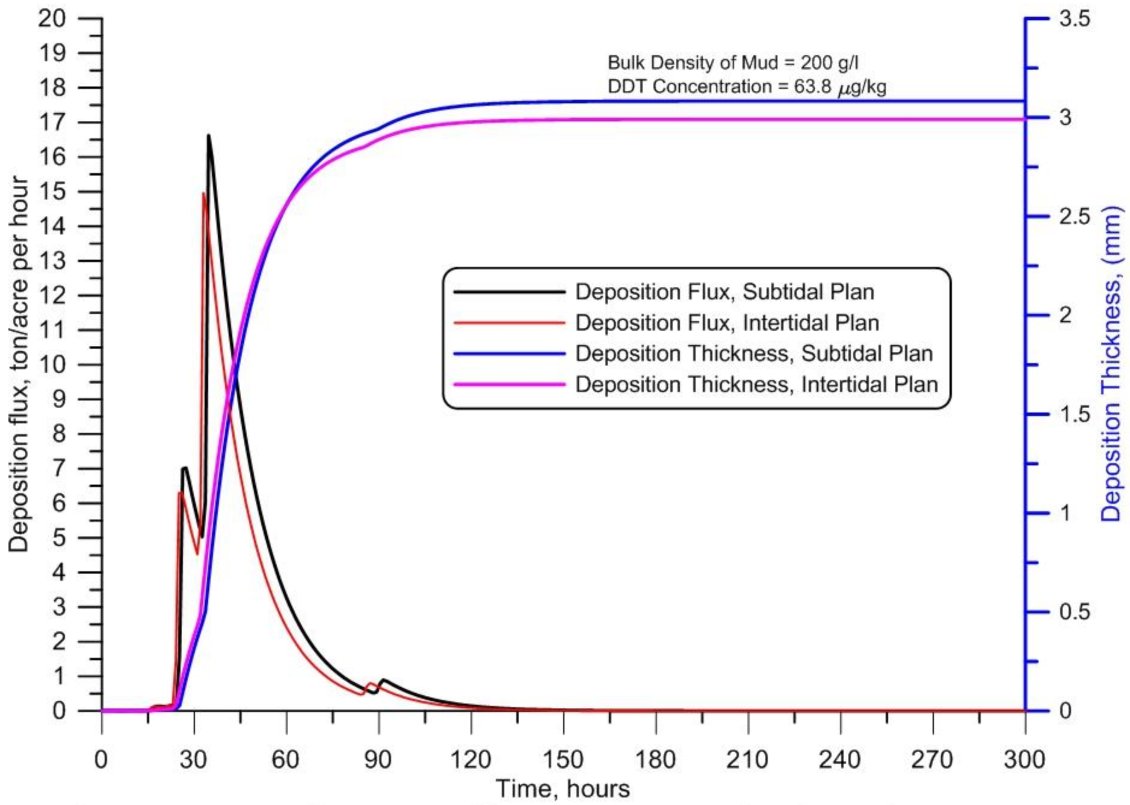

Figure A3. Sensitivity analysis of deposition of fine-grained sediment (mud) in the Otay River Floodplain Tidal Basin following the 100-year flood. Deposition flux (black & red); deposition thickness (blue & magenta). Results for 76,350 cubic yards of contaminated fines from erosion of the top 2 ft. and 438,000 cubic yards of clean sediment from the upper Otay River watershed. DDT given as dry bulk concentration.

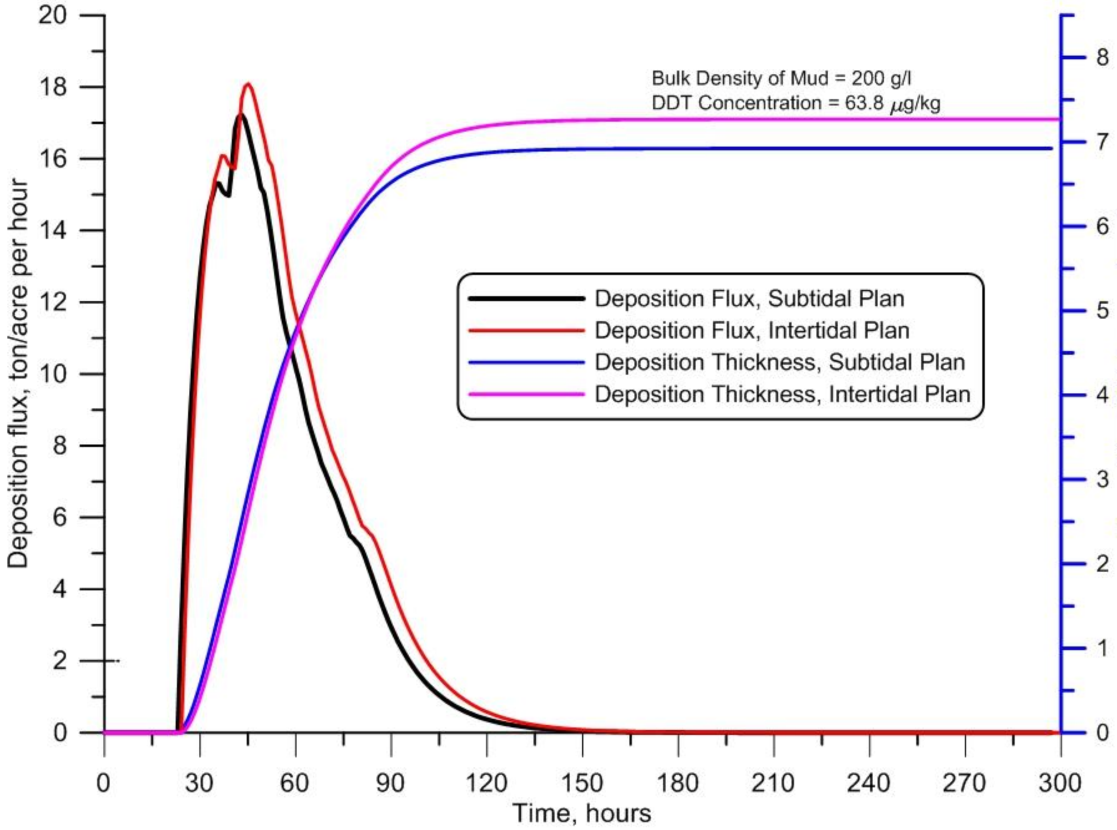

Figure A4. Sensitivity analysis of deposition of fine-grained sediment (mud) in the Pond-15 Tidal Basin following the 100-year flood. Deposition flux (black & red); deposition thickness (blue & magenta). Results for 76,350 cubic yards of contaminated fines from erosion of the top 2 ft. and 438,000 cubic yards of clean sediment from the upper Otay River watershed. DDT given as dry bulk concentration.

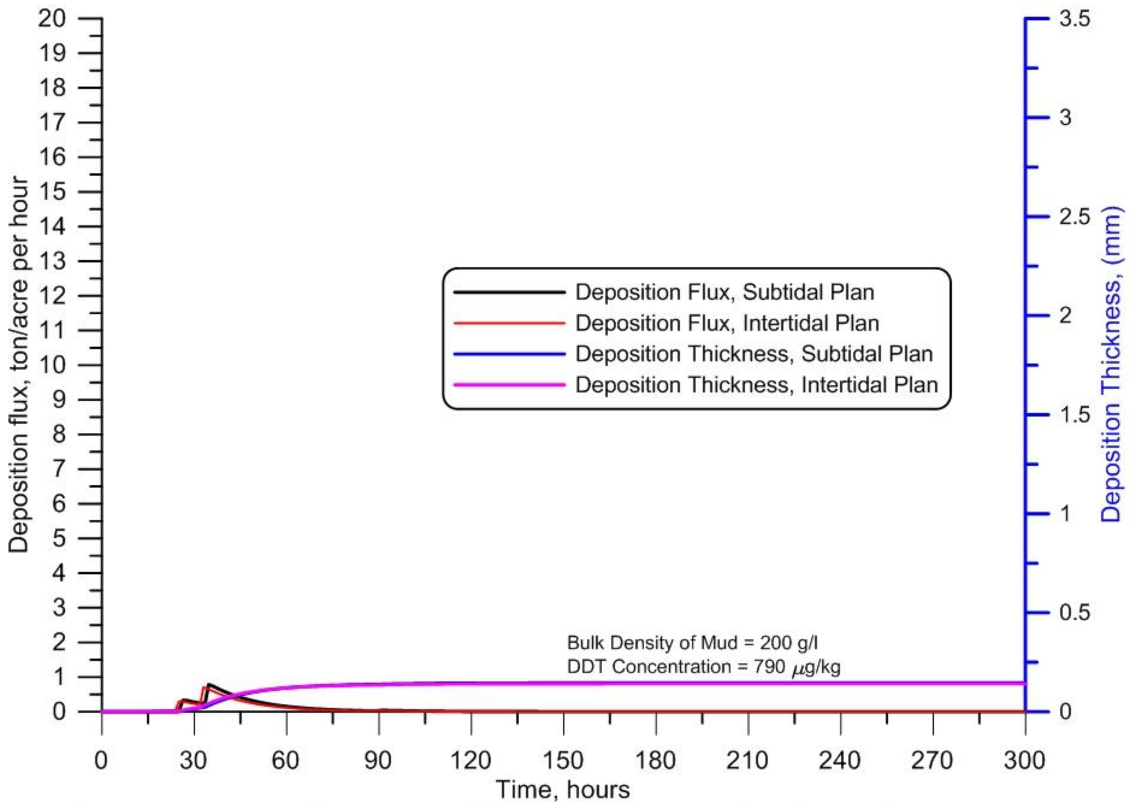

Figure A5. Sensitivity analysis of deposition of fine-grained sediment (mud) in the Otay River Floodplain Tidal Basin following the 100-year flood. Deposition flux (black & red); deposition thickness (blue & magenta). Results for 24,260 cubic yards of contaminated fines from erosion of the top 1 ft. and no additional clean sediment from the upper Otay River watershed. DDT given as dry bulk concentration.

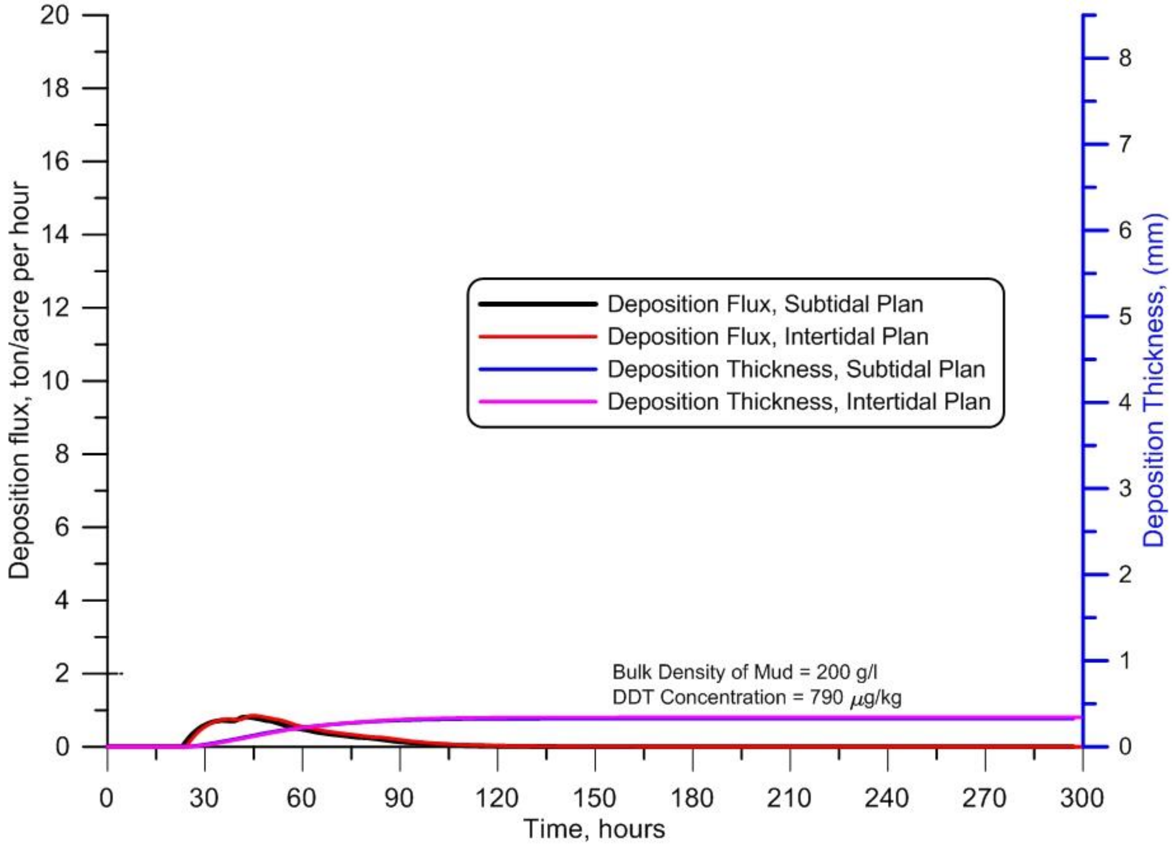

Figure A6. Sensitivity analysis of deposition of fine-grained sediment (mud) in the Pond-15 Tidal Basin following the 100-year flood. Deposition flux (black & red); deposition thickness (blue & magenta). Results for 24,260 cubic yards of contaminated fines from erosion of the top 1 ft. and no additional clean sediment from the upper Otay River watershed. DDT given as dry bulk concentration.

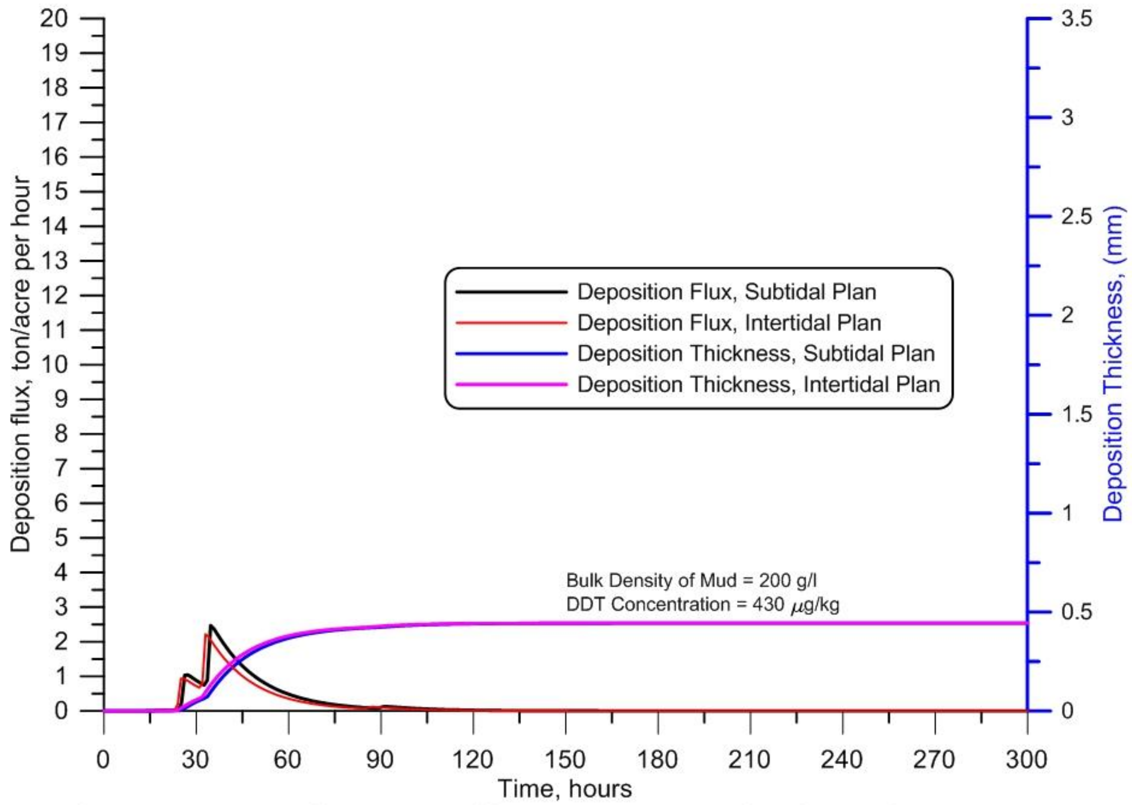

Figure A7. Sensitivity analysis of deposition of fine-grained sediment (mud) in the Otay River Floodplain Tidal Basin following the 100-year flood. Deposition flux (black & red); deposition thickness (blue & magenta). Results for 76,350 cubic yards of contaminated fines from erosion of the top 2 ft. and no additional clean sediment from the upper Otay River watershed. DDT given as dry bulk concentration.

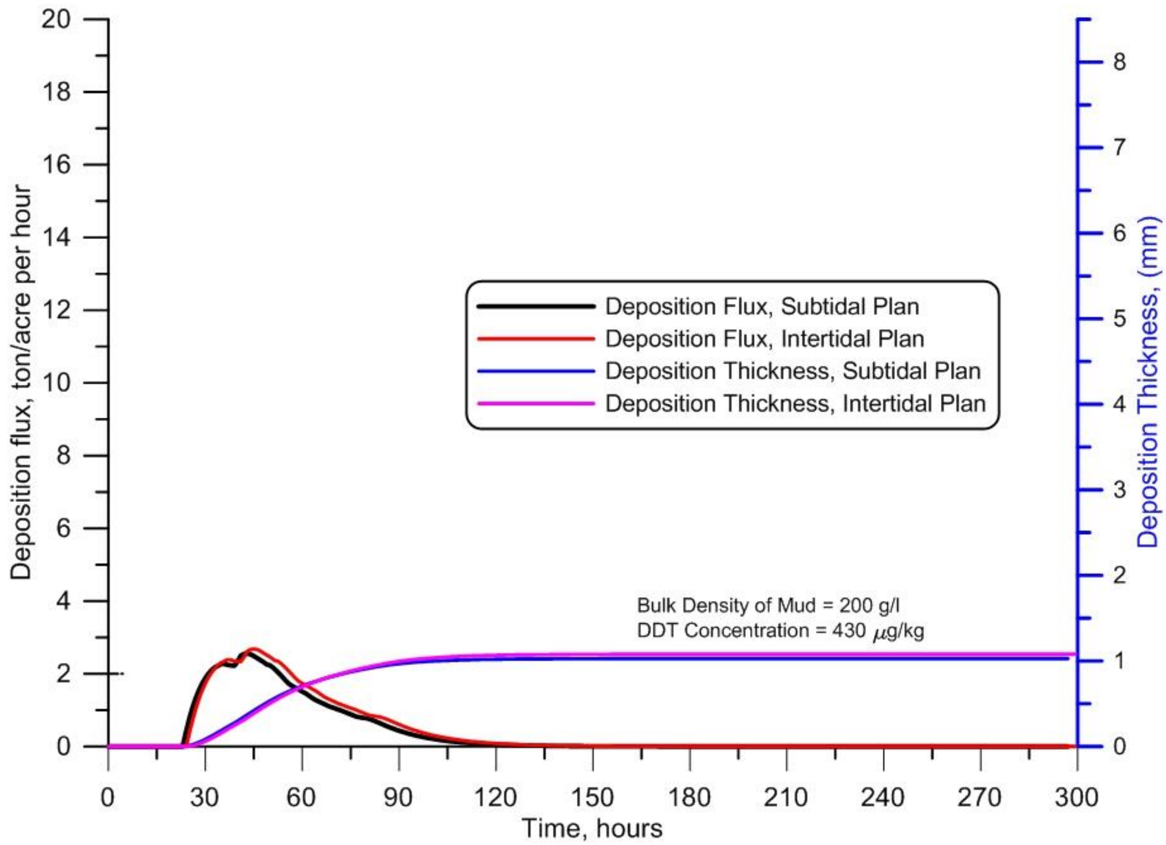

Figure A8. Sensitivity analysis of deposition of fine-grained sediment (mud) in the Pond-15 Tidal Basin following the 100-year flood. Deposition flux (black & red); deposition thickness (blue & magenta). Results for 76,350 cubic yards of contaminated fines from erosion of the top 2 ft. and no additional clean sediment from the upper Otay River watershed. DDT given as dry bulk concentration.

APPENDIX-B: Additional Deposition Flux and Deposition Thickness Simulations for the *No-Project Alternative* Post 100-Year Flood.

Input Assumptions: The 100-year flood hydrographs for the Otay River, Poggi Canyon Creek and Nestor Creek are unchanged by the presence of the ORERP. The total flow volume during a 100-year flood for the no-project alternative is 35,200,000 cubic yards (cy), or 26,911,315 cubic meters (m^3) for the Otay River. The corresponding flow volumes for Poggi Canyon Creek and Nestor Creek are respectively 2,240,000 cy (1,712,254 m^3) and 1,748,800 cy (1,337,003 m^3), so that the combined flow through the floodplain is $\overline{Q}$ = 39,188,800 cy (29,960,856 m^3), or 24,290 acre ft. Figure B1 give the distribution of maximum stream flow velocities for the 100-year flood in the lower Otay River flood plain and salt pond complex for the no-project alternative, (after Everest, 2014); while Figure B2 gives the velocity distribution for the ORERP Intertidal Alternative and Figure B3 gives the Subtidal Alternative. In each of these figures, the DDT contaminated area is bounded by a yellow polygon in the lower left hand corner.

Comparing the velocities in the DDT contaminated area among Figures B1 – B3, we find the maximum flood velocities for the 100-yr flood are about 0.5 ft/s to 1.0 ft./s greater for the Intertidal and Subtidal Alternatives relative to the no-project alternative. At first impression, this would suggest that the ORERP might cause more soil erosion in the DDT contaminated area than the no-project alternative. However, the sediment stratigraphy in this area indicates this is not the case, as revealed by sediment coring conducted by Anchor QEA (2013). In the DDT contaminated area of the floodplain, the top 3 ft of soils are comprised of 27 % silt and clay ($d < 0.0625$ mm) and 63 % fine sands to coarse sand ($d > 0.0625$ mm). However, from 3 ft to 5 ft below existing grade, 74.1 % of the soils are comprised of silt and clay, and 25.9 % are fine sand to coarse sand. Hence, there is an abrupt transition from more sandy, erodible, material in the top 3 ft, to more cohesive erosion-resistant soil below 3 ft. It was this difference in grain sizes that the original assumption set forth in Section 2.1 was based, whereby the top 3-ft of soil could be completely eroded during a 100-year flood. It is also this abrupt transition in grain sizes at 3 ft below existing grade in the DDT contaminated area that creates a hard enough basement on the depth of erodible soil so that erosion below 3 ft will not occur, with or without the project during a 100-year flood, (given the maximum flood velocities shown in Figures B1 – B3). Therefore we can assume the same amounts of DDT contaminated soils will be eroded from the floodplain for the no-project alternative as for the ORERP alternatives. From that assumption we formulate the model inputs for the post 100-yr floor flood analysis of the no-project alternative as listed in Table B1. The inputs for the no-project alternative are based on erosion fluxes from one possible erosion depth (3 ft.) in the DDT contaminated area of the floodplain, and is combined with two possible fluxes of clean fines (0 cy and 438,000 cy) from the upper watershed below the Savage Dam; yielding a sensitivity analysis comprised of 2 separate deposition scenarios. Thus, the ensemble of input parameters for the no-project sensitivity analysis are comparable to inputs used for the ORERP in Table 4, rows 2 & 5.

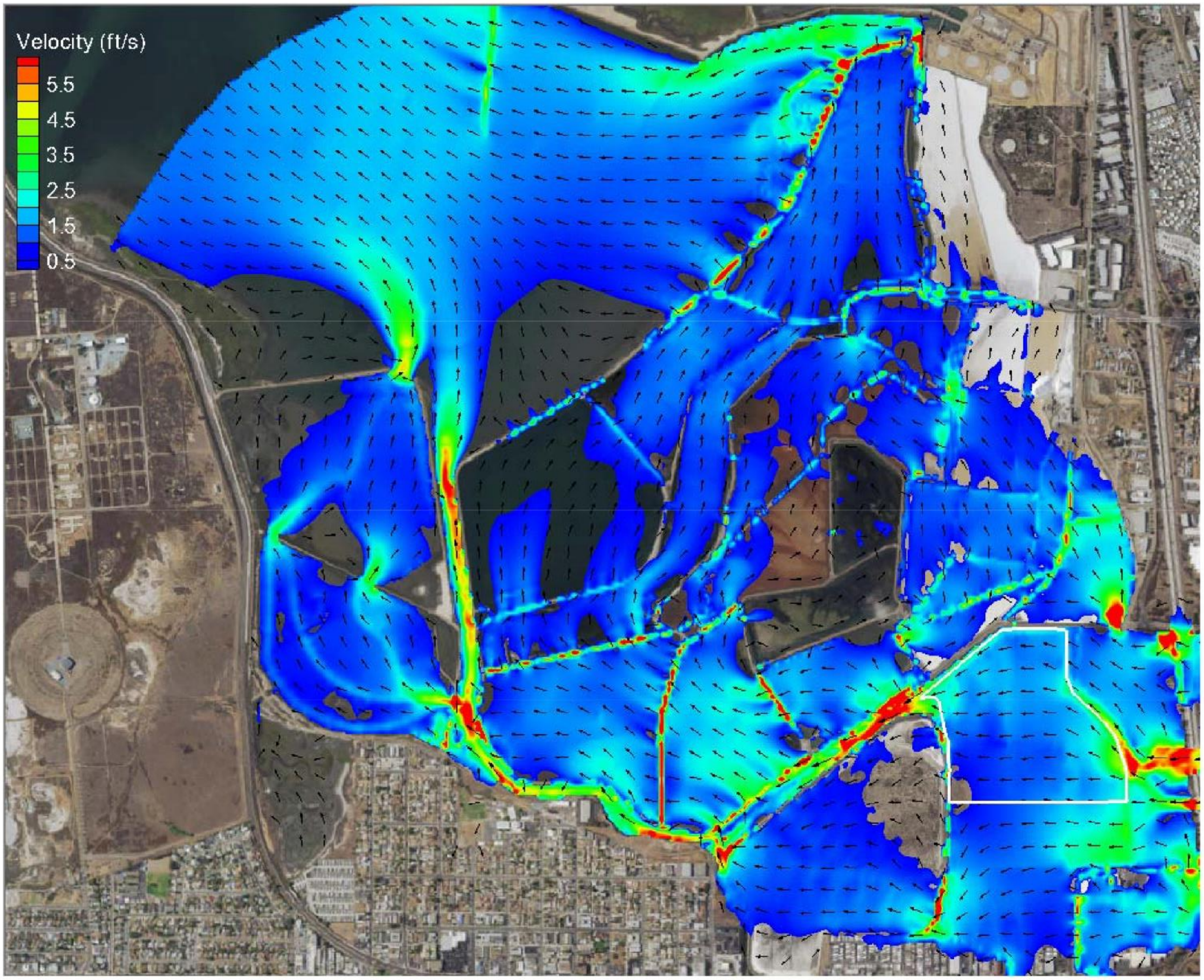

Figure B1: Distribution of maximum stream flow velocities for the 100-year flood in the lower Otay River flood plain and salt pond complex for the no-project alternative, (after Everest, 2014). DDT contaminated area bounded by yellow polygon in the lower right hand corner. Ponds 10 & 11 shown in the lower left corner.

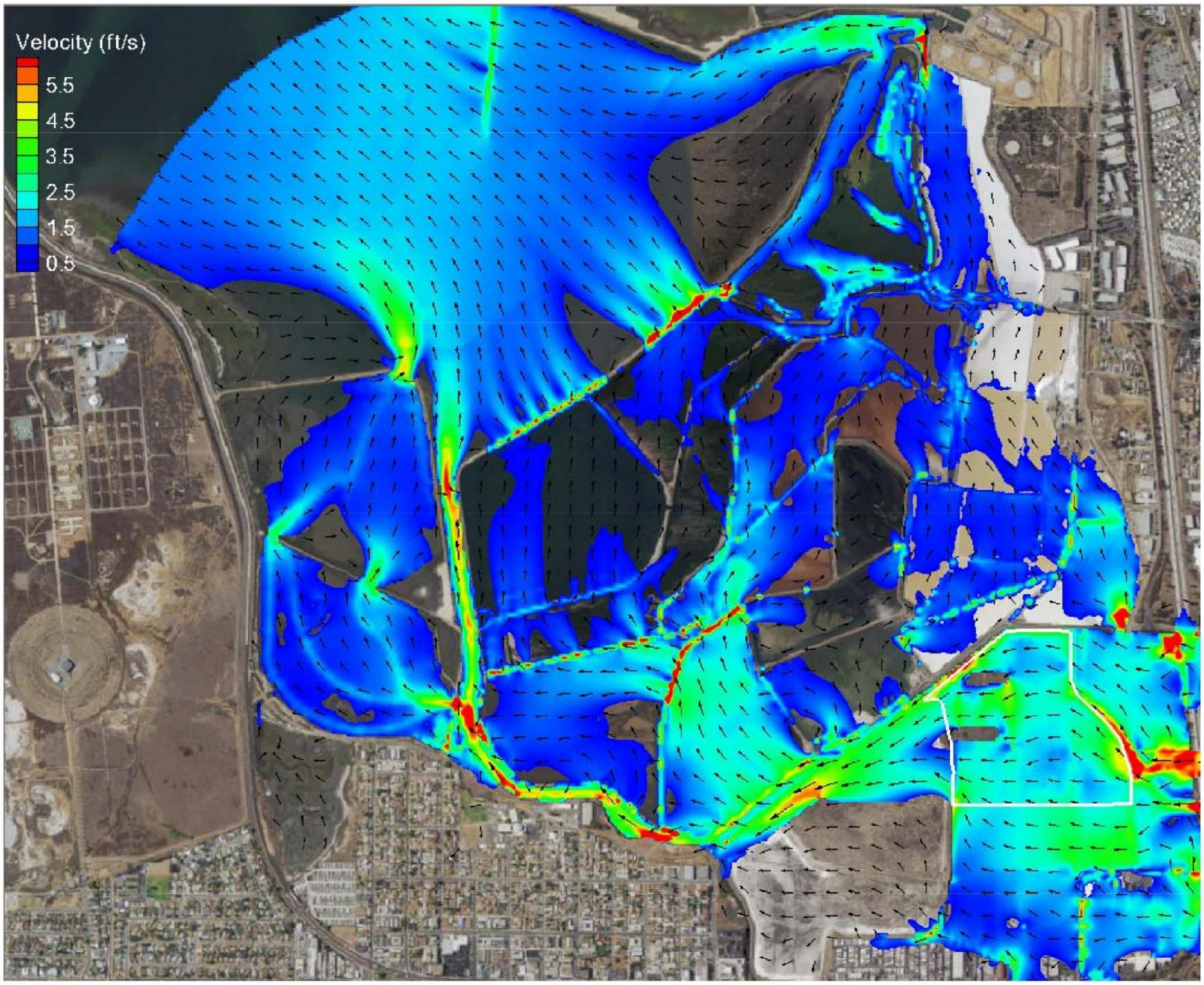

Figure B2: Distribution of maximum stream flow velocities for the 100-year flood in the lower Otay River flood plain and salt pond complex for the fully implemented Intertidal Alternative, (after Everest, 2014). DDT contaminated area bounded by yellow polygon in the lower right hand corner. Ponds 10 & 11 shown in the lower left corner.

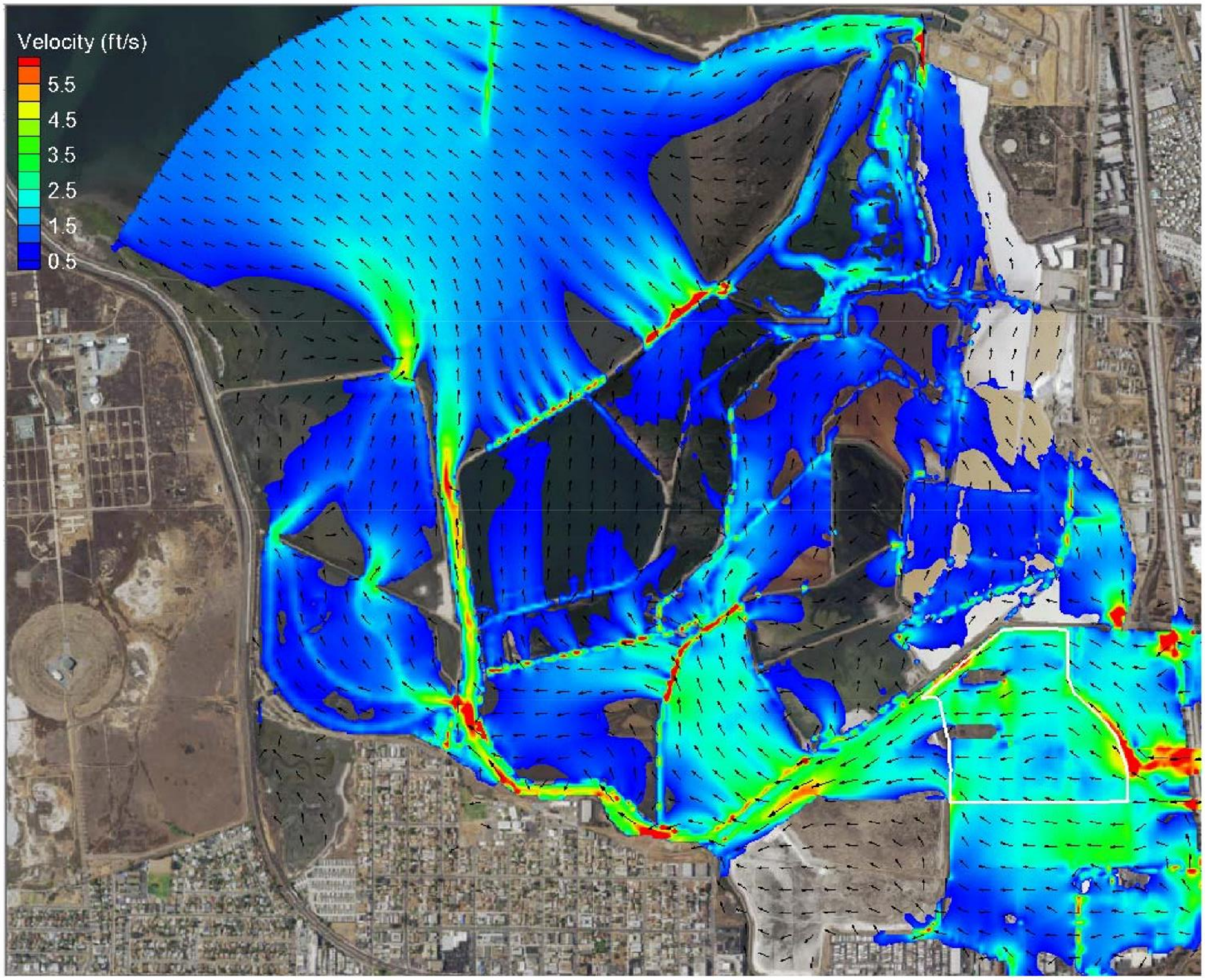

Figure B3: Distribution of maximum stream flow velocities for the 100-year flood in the lower Otay River flood plain and salt pond complex for the fully implemented Subtidal Alternative, (after Everest, 2014). DDT contaminated area bounded by yellow polygon in the lower right hand corner. Ponds 10 & 11 shown in the lower left corner.

Table B1: Input Parameters for Sensitivity Analysis of Post 100-Year Flood DDT Deposition for the No-Project Alternative

Scenario	Volume of Eroded DDT-Bearing Fines	DDT Conc. in DDT-Bearing Fines	Volume of Eroded Upper Watershed Fines	Flood Flow Volume	Suspended Sediment Conc.
Erode top 3 ft. of Contaminated Area + Upper Watershed*	128,300 cubic yards	310 μ g/kg	438,000 cubic yards	24,290 acre ft	23.15 g/l.
Erode top 3 ft. of Contaminated Area Only*	128,300 cubic yards	310 μ g/kg	0 cubic yards	24,290 acre ft	5.25 g/l

The suspended sediment concentrations in Table B-1 are based on a dry bulk density for eroded soil of 2700 lb per cy, or 1.225 metric tons per cy; where a metric ton is 1000 kg. This conversion factor is applied to the sum of the volume of eroded DDT- bearing fines (column_2) and the volume of eroded fines from the upper Otay watershed (column_4) to obtain the total flux of suspended fine grained sediment in tons/day during the 24 hour flood period of the 100-year flood for the no-project alternative. The sand and gravel sized fractions eroded from the floodplain by the 100 year flood (292,000 cy) are assumed to be transported as bed load and remain in the Otay River channel. The suspended sediment flux component (column_2 + column_4) is divided by the flow volume of $\overline{Q}$ = 29,960,856 m^3 during the 24 hour flood period to give the average suspended sediment concentration in column_6 upon conversion of metric tons to grams and cubic meters to liters.

Deposition Results: We use the 2011 bathymetric survey conducted by WRA for modeling post-flood deposition in no-project alternative (Figure B4), and use the deposition results from Ponds 10 and 11 as a proxy for evaluating potential wetlands impacts from the 100-yr flood. In Section 5, it was shown that the deposition thickness in the tidal basins is proportional to the water depths and tidal residence times in those basins, where greater deposition thickness was observed in Pond 15 where water depths are greater and residence times longer than in the Otay River Floodplain Basin of the ORERP, (cf. Figures 9 vs. 10). Figures B4 reveals that water depths in Ponds 10 and 11 are comparable to water depths in the Subtidal Alternative of the ORERP, and the TIDE_FEM solutions indicate that the residence times are also comparable (on the order of 2.5 days). Thus it is not surprising to find that the plots of the deposition flux and deposition thicknesses in the Ponds 10 and 11 in Figures B5 and B6 for the 100-year flood are very similar to those in Figures 9 and 11 for the Subtidal Alternative; although the exact time response (shape) of the two sets of curves are different than for the ORERP simulations.

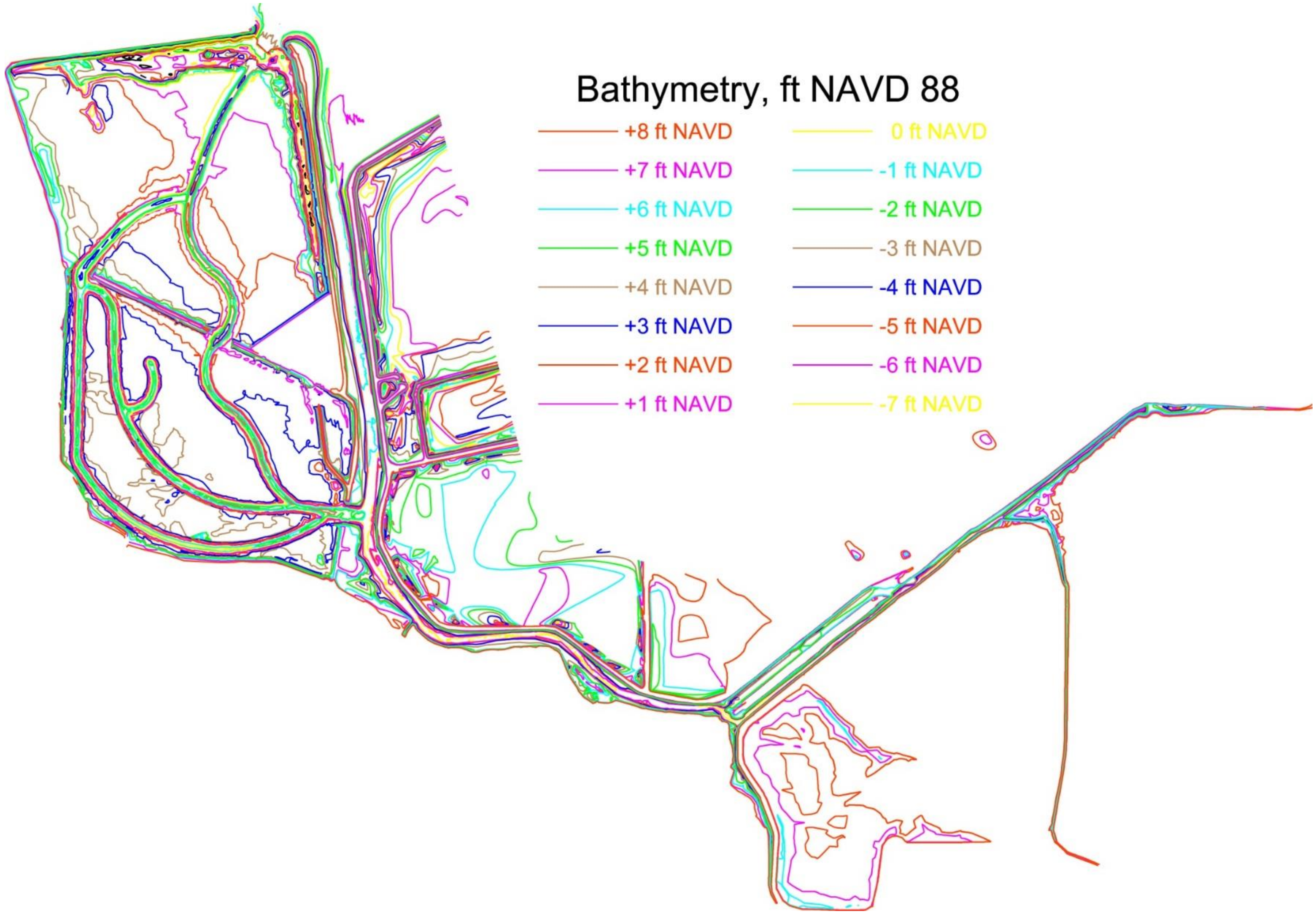

Figure B4: Bathymetry for the *No-Project Alternative* with Ponds 10 & 11 shown on the left hand side (west bank) of the Otay River at the river mouth. Bathymetry shown in ft. NAVD based on bathymetric survey by WRA (2011).

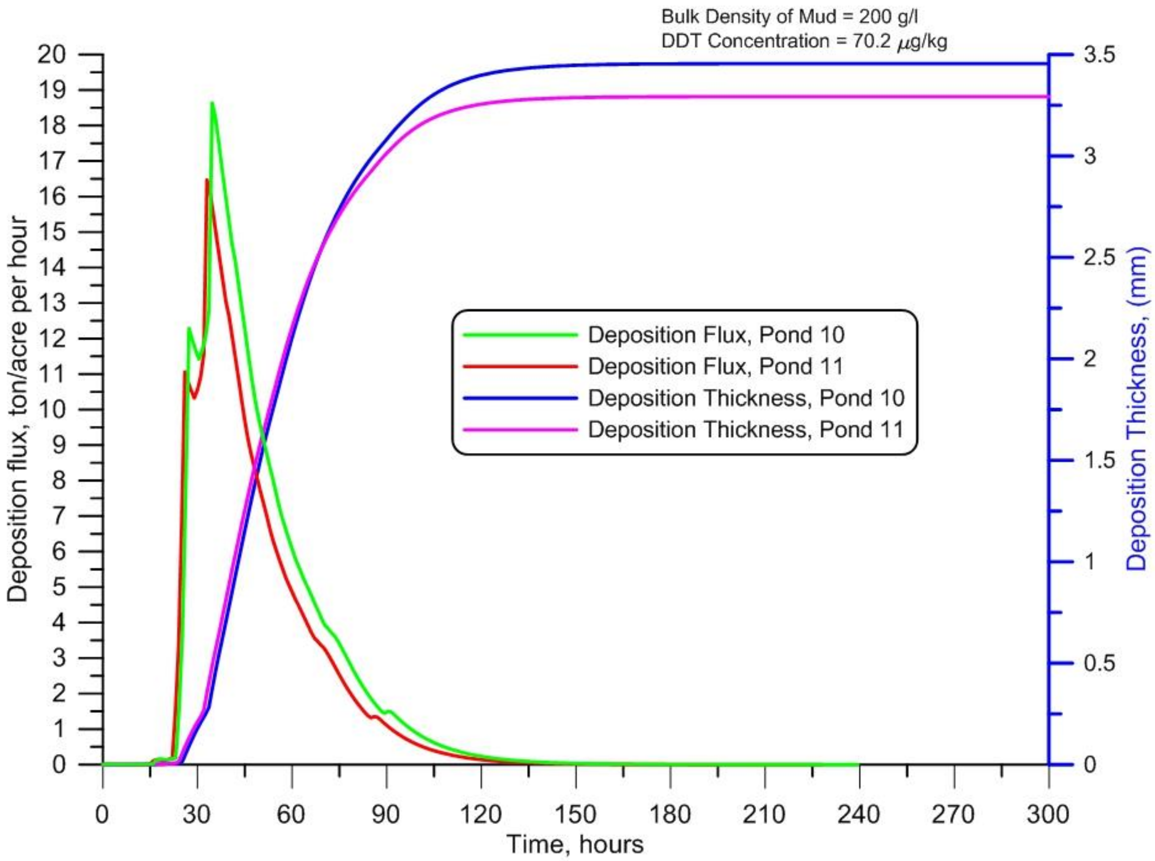

Figure B5. Sensitivity analysis of deposition of fine-grained sediment (mud) in the Ponds 10 and 11 in the Otay River Floodplain (*No-Project Alternative*) following the 100-year flood. Deposition flux (green & red); deposition thickness (blue & magenta). Results for 128,300 cubic yards of contaminated fines and 438,000 cubic yards of clean sediment from the upper Otay River watershed. DDT given as dry bulk concentration.

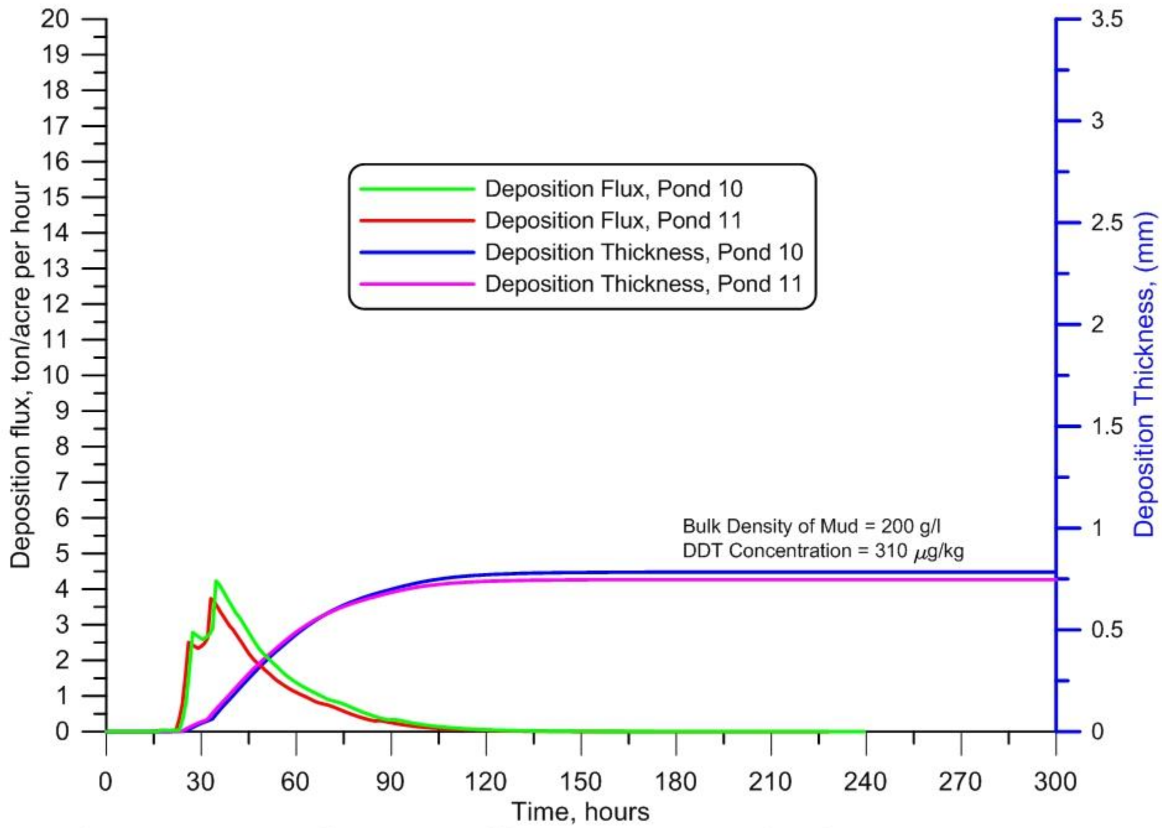

Figure B6. Sensitivity analysis of deposition of fine-grained sediment (mud) in the Ponds 10 and 11 in the Otay River Floodplain (*No-Project Alternative*) following the 100-year flood. Deposition flux (green & red); deposition thickness (blue & magenta). Results for 128,300 cubic yards of contaminated fines and no additional clean sediment from the upper Otay River watershed. DDT given as dry bulk concentration.

Figure B5 gives the time evolution of the post-flood deposition flux and deposition thickness for the first scenario (row_2 of Table B1) in Ponds 10 and 11 of the no-project alternative. This scenario is based on maximum flood-induced erosion depths of 3 ft. in the contaminated area adjacent the Floodplain Tidal Basin mixed with 438,000 cubic yards of fine-grained sediments from upstream erosion of the portion of the watershed below the Savage Dam. Results are similar for both Ponds 10 and 11 showing that accumulations range from 3.4 to 3.7 mm of partially consolidated mud after 276 hours post-flood, with dry bulk DDT concentrations of 70.2 μ g/kg everywhere in the post-flood deposition. The initial post-flood suspended sediment concentration is the same in all areas of the floodplain and salt pond complex because the 100 year flood overtops and flows through these areas with its washload, (cf. Figure B1). The general depositional features are that deposition flux peaks within one diurnal tide cycle after cessation of the flood in both basins of both restoration alternatives, with an initial deceleration in flux during the first semidiurnal ebb tide. After the first post-flood diurnal tidal cycle, the deposition flux declines as progressive settling depletes the suspended sediment concentration, and tidal residence times in the ponds limits the amount of time for settling and deposition to occur. Meanwhile, deposition thickness, which results from the cumulative sum of deposition flux over time, rapidly builds during the peak deposition flux period, and then gradually approaches a constant limit for partially consolidated mud at 200 g/l bulk density as the deposition flux vanishes after 120 to 150 hours post-flood. The minor differences in deposition flux and deposition thickness among the ponds and restoration alternatives are due to differences in residence times as a consequence of proximity of outlets to The Bay and river.

Next, consider in Figure B6 how such results may be affected if we assume no erosion of soils occurs in the portion of the watershed upstream of the floodplain and below the Savage Dam. This scenario is specified by the third row in Table B1 and is based on maximum erosion depths of 3 ft. in the contaminated area only. Here, runoff from the 100 year flood consists of a uniform suspended load of silts and clays with concentration of $\overline{C}$ = 5.25 g/l. Figure B6 gives the time evolution post-flood for deposition flux and deposition thickness in Ponds 10 and 11 of the no-project alternative. Again, results are similar for both ponds, but the dry bulk concentration of DDT in the post-flood deposition has increased to 310 μ g/kg, while the deposition thicknesses are greatly diminished. Ponds 10 & 11 accumulate only 0.74 to 0.78 mm of partially consolidated mud after 276 hours post-flood.

The initial post 100-year flood accumulations of partially consolidated mud computed in Figures B5 & B6 will, over time, dewater and compact under its own immersed weight. The initial deposition will consolidate and compact to a maximum saturated density for fully consolidated mud, 1200 g/l, so that the 100-year flood deposition for the two scenarios in Table B1 would eventually become a very thin layer of consolidated mud on the order of a fraction of a millimeter thick; or:

Deposition with upper watershed sediments:

$$3.4 - 3.7 \ mm \ @ \ 200 g/l \quad \Rightarrow \quad \left\{ \frac{dewatering}{consolidation} \right\} \quad \Rightarrow \quad 0.5 - 0.6 \ mm \ @ \ 1{,}200 g/l$$

Deposition without upper watershed sediments:

$$0.74-0.78\ mm\ @\ 200g/l \quad \Rightarrow \quad \left\{\frac{dewatering}{consolidation}\right\} \quad \Rightarrow \quad 0.17-0.18\ mm\ @\ 1{,}200g/l$$

Consolidation only involves a reduction in the water content of the post-flood deposition, and therefore does not alter the DDT dry bulk concentration, which remains 70.2 μ g/kg when there is dilution from upper watershed sediments and 310 μ g/kg when there is no deposition of upper watershed sediments. The amount of time required for this degree of consolidation is uncertain, but experience with dredge material disposal ponds at Mare Island, CA and Charleston, SC [Jenkins, 1980; Jenkins et al., 1981; Jenkins and Skelly, 1983] suggests that consolidation to 600 g/l could occur within three months while full consolidation to saturation (1,200 g/l) could take several years.

The DDT deposition results in Ponds 10 and 11 for the 100-yr flood under the no-project alternative are summarized in Table B2 below. These results are found to be within the range of those for the ORERP post 100-yr flood as detailed in Section 5. From that finding, we submit that the conclusions on potential flood-induced DDT impacts to the existing wetlands ecology, as detailed in Section 7 are upheld; and it can be concluded that the ORERP does not increase the risk of exposure of wetland ecology to DDT, (a risk that exists with or without the project).

Scenario	Volume of Eroded DDT-Bearing Fines	Average DDT Conc. in DDT-Bearing Fines	Volume of Eroded Upper Watershed Fines	Flood Flow Volume	Suspended Sediment Conc.	Initial Post-Flood Deposition Thickness (200 g/l Mud)	Final Post-Flood Deposition Thickness (1,200 g/l Mud)	DDT Conc. in Post-Flood Mud Deposition (dry bulk)
Erode top 3 ft. of Contaminated Area + Upper Watershed*	128,300 cubic yards	310 μ g/kg	438,000 cubic yards	24,290 acre ft	23.15 g/l.	3.4 mm to 3.7 mm	0.5 mm to 0.6 mm	70.2 μ g/kg
Erode top 3 ft. of Contaminated Area Only**	128,300 cubic yards	310 μ g/kg	0 cubic yards	24,290 acre ft	5.25 g/l	0.74 mm to 0.78 mm	0.17 mm to 0.18 mm	310 μ g/kg

Table B2: Matrix of Sensitivity Analysis of Potential DDT Deposition in Ponds 10 and 11 for the *No-Project Alternative* post-100 year flood.

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December 30, 2016

Brian Collins
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Re: **Comments on Draft Environmental Impact Statement for the Otay River Estuary Restoration Project**

Dear Mr. Collins:

Thank you for the opportunity to comment on the United State Fish and Wildlife Service's ("USFWS") Draft Environmental Impact Statement ("EIS") for the Otay River Estuary Restoration Project ("ORERP") in the South San Diego Bay Unit of the San Diego Bay National Wildlife Refuge in San Diego County. The proposed project is a partnership between Poseidon Resource's ("Poseidon") and the USFWS to create 33.5 acres of tidal wetlands in the Otay River Floodplain and to convert 90.9 acres of existing solar salt pond to tidal wetlands and associated upland habitat. ORERP is intended to fulfill Poseidon's obligation to create or substantially restore 66.8 acres of tidal habitat as mitigation for impacts to marine life from the operation of Poseidon's Carlsbad Desalination Plant, as required by Coastal Development Permit E-06-013.

This project will require a coastal development permit ("CDP") from the Coastal Commission. In addition, as noted in section 1.5.3 of the EIS, the Commission is the California Environmental Quality Act (CEQA) lead for the state. The Commission's CDP process is considered CEQA-equivalent and the San Diego Regional Water Quality Control Board will rely on the findings included in the CDP to analyze the proposed action for consistency with applicable policies. The Commission will therefore rely in part on the information contained in the EIS in assessing the project's conformity with the Chapter 3 coastal resource protection policies.

Thank you for your consideration of the following comments:

1. Alternatives
   a. It is surprising to me that the planned design for Alternative B does not have any subtidal habitat, even after 24 inches of sea level rise. Was this intentional? Would there be advantages (or disadvantages), biologically, to creating a slightly deeper channel so that some subtidal habitat is included in the longer term?
   b. Would the conveyor belt option use a covered conveyor or would it be open to the air?

2. Sea level rise. The sea level rise analysis for Alternatives B and C assesses the effect of 24 inches of sea level rise on the planned mix of habitats at both mitigation sites. I could not find where in the NEPA document it explained why the analysis stopped at 24 inches of sea level rise (i.e., projected for 2050). As part of the mitigation components of the project, Poseidon will be required to monitor and meet performance standards at the proposed wetlands for 30 years, or the estimated life of the Carlsbad Desalination Plant. However, depending on the wetland's performance, this may or may not occur in the 30 consecutive years between 2020 and 2050. In addition, these wetlands will remain part of the Refuge and the South San Diego Bay ecosystem in perpetuity. Thus, it is important to assess how sea level rise might affect these wetlands in the longer term and also how the anticipated mix of habitats compares between the two Alternatives. Thus, we recommend examining the effects of sea level rise up to 66 inches.

3. Contaminants - DDT Analysis. Sections 3.2.10 and 4.2.10 of the draft EIS included a discussion and analysis of the known contaminants on the Otay and Pond 15 sites as well as potential effects from contamination related to the proposed project. The EIS specifically discusses the results of a sensitivity analysis of potential DDT deposition at the project site in the case of a 50-year and 100-year flood (Appendix I). As you know, when the results of this analysis were first presented to the SAP, we requested that independent reviewers with expertise in hydraulic modeling and sediment transport and ecotoxicology be retained to provide an independent review of the DDT study. In the past few months, I was able to engage Dr. Keith Stolzenbach, Professor of Civil and Environmental Engineering at UCLA and Dr. Steve Schwarzbach, former Center Director of the USGS Western Ecological Research Center, to review the study. Their detailed comments are included as Attachment 1 to this letter. To summarize their review findings:
   a. DDT dispersion modeling: Overall, Dr. Stolzenbach found the modeling approaches used for the dispersion modeling were appropriate. He also found that many of the assumptions used in the DDT dispersion modeling work greatly simplify the actual sediment transport processes but are generally consistent with the overall approach used in the DDT dispersion calculations. However, uncertainty in the model results is not well described. Some key findings of Dr, Stolzenbach's review include (see Attachment 1 for full comments):
      i. A key factor in understanding the potential error in the DDT dispersion calculations is the fraction of the total mass of fine sediments and DDT that are deposited rather than flushed away. If this fraction is large, uncertainties in the DDT concentrations could be low by no more than a factor of two. But if the fraction is small (i.e., 1% or less) the modeling results could underestimate the final DDT concentrations by as much as a factor of 100 or more. The fraction of the total mass of fine sediments and DDT that are deposited rather than flushed away could not be determined from the report. Please provide this fraction for the different cases analyzed to allow for a relatively quick check on the magnitude of the potential error.
      ii. The numerical model described in Jenkins and Waysl (2014) is an "example of a general flow modeling technique that is well-accepted and appropriate for this kind of flow environment, although no site-specific validation is presented for flow calculations."

iii. The final consolidation results are determined by the assumed values of bulk sediment density instead of calculated, which appears to be a reasonable approach.
iv. The boundary condition at the open boundary with San Diego Bay does not seem to allow for return of flow and sediment on the incoming tide that had been transported out of the domain on an earlier outgoing tide. Because flushing at the end of San Diego Bay is weak, it is important to represent the return process in some way.
v. The dispersion modeling in some cases uses highly empirical, and possibly dated, parameterizations of transport processes. In addition, error introduced by the assumptions about sediment resuspension and consolidation are not described or quantified. These assumptions are critical to the calculation the final concentration of DDT in the sediment deposited in the system. A sensitivity study to the parameter values would be appropriate to gain confidence in the overall calculation results.
vi. Calculations of "initial" and "final" concentrations of DDT included in Table 11 are not included and it is not clear why there are different values given that the mas of sediment and the mass of DDT remain constant. These calculations require clarification.

b. Ecotoxicology: Overall, Dr. Schwarzbach found that the methodologies employed to determine if contaminated sediments deposited in the proposed wetlands in a large flood event posed a risk to wildlife were appropriate, although several assumptions were not adequately justified and in some cases, result in sediment screening levels that are not adequately protective. His own analysis (discussed in further detail below) suggests that the range of appropriate sediment screening levels is likely narrower than the range of 2 to 240 ug/kg reported in the study and likely closer to 2 ug/kg than 240 ug/kg. Some key findings of Dr, Stolzenbach's review include (see Attachment 1 for full comments):
   i. The Otay watershed is scaled (linearly) to a much smaller watershed to determine the amount of eroded sediment. This leads to a conservative estimate of sediment dilution but likely leads to a significant overestimate of final DDT concentrations in the scenarios where upstream sediments are included.
   ii. In areas of the site where DDT was not detected in sediment samples, the DDT concentration is reported as 0 ug/kg. However, because zero can be approached but not measured, concentrations should be reported as the detection limit or some fraction of the detection limit.
   iii. Tidal redistribution of the DDT contaminated sediments within the intertidal marsh is not likely to result in a final uniform depth throughout the marsh and lower watershed, thus adding additional complexity and uncertainty to the likely DDT sediment concentrations in the marsh.
   iv. The two-component approach taken for assessing potential adverse impacts to fish and wildlife from possible DDT contamination (evaluating risk of direct toxicity to benthic organisms using established toxicity benchmarks, and the EPA approach that back calculates permissible sediment concentrations using dietary concentrations) is appropriate. However, potential toxic impacts of DDT on fish is underrepresented, especially given that one of the express purposes of the project is to mitigate for losses to fish.

v. The choice of the Light-footed Ridgeway Rail and the Snowy Egret as the wildlife receptors for this study are good choices, but may not tell the whole story of DDT risk in the ORERP. But the reasons for using these two species are not well explained. In addition, no equations are included, making it very difficult to check assumptions and calculations. For example, it is unclear what body weight is used for the Rail (attempts to recreate calculations resulted in a body weight of 315g which is much higher than the 270g weight communicated by the authors in a subsequent email and can have a significant effect on the final screening concentrations for sediment).

vi. The study's development of an NOAEL for eggshell thinning for brown pelican and double-crested comorant was appropriate and well justified. However, converting this NOAEL to a dietary concentration to determine a reference dose (TRV) for the rail and egret is poorly described and assumptions made to achieve the results are not well-justified. In addition, no equations or example calculations were provided. Please provide the relevant formulas and at least one example calculation. Also, please provide additional justification for assumptions made to determine the TRV (see page 13-14 of Dr. Schwarzbach's report for additional details and comments).

vii. The assumption that receiving sediments in the Otay basin will be at background levels is not supported by the results of the Zeeman 2008a that report elevated levels of DDT in mudsuckers in the Oay River as compared to the salt ponds. However, the likely underestimation of diluting upstream sediments (by scaling with Buena Vista watershed) may outweigh this factor.

viii. Dr Scharzbach was able to duplicate most of the risk assessment calculations. However, he believes that the weight of evidence suggests that exposure to rails to sediment with DDT concentrations in the upper end screening levels reported (i.e., 240 ug/kg) would have disastrous consequences and should be discarded as guidelines for safe levels of DDT in tidal marsh sediment.

ix. Dr Schwartzbach developed an excel tool first as an attempt to reproduce the results presented in the DDT study. In addition, he used the tool to expand the analysis to several different species and to assess the sensitivity of the results to several parameters (i.e., body weight, dietary DDT/sediment ratio, etc). His results indicate a safe level of exposure that ranges from 6 to 11ug/kg for rails and from 4 to 8 ug/kg for egrets. Results for additional species are included on pages 20-24 of Dr. Schwarzbach's report.

x. Based on the results described above, Dr. Schwarzbach concludes that for Rails, the species of greatest vulnerability and management interest, there may be some toxic risk from DDT bioaccumulation in the short term after a 50 or 100 year flood event. However, the risk appears to be very small in the long term. Thus, the benefits of the project to the Rail population as a whole would appear to outweigh impacts from potential DDT contamination.

As part of the CDP process, Commission staff will be analyzing the information presented in the DDT Analysis (Appendix I) as well as the findings of the reviews described above, to determine if the project is consistent with the Coastal Act and with the requirements of the approved MLMP. We have not completed our analysis but our initial review of information provided in the EIS and by the expert reviewers suggests that in the event of a major 50 to

100 year flood, there may be a risk that DDT laden sediment deposited in the proposed ORERP wetlands could result in adverse impacts to wildlife. To further reduce the risk of adverse impacts to wildlife, Commission staff would like to work with USFWS and other agency staff to explore additional mitigation measures. One possibility may be to incorporate a monitoring and remediation plan that requires Poseidon to measure DDT concentrations in wetland sediment and wildlife and remediate if concentrations reach harmful levels. It may also be desirable to include additional measures to ensure that the existing DDT contamination is not mobilized.

4. In addition to the comments above, I have a few specific comments on the Contaminants analysis presented in the NEPA document:
   a. Table 3.2.-11 (p. 3.2-49) – Please include the range of contaminant concentrations instead of just the average. The average isn't always a useful measure depending on the underlying data.
   b. It would be very useful to include a table in Section 3.2.10 with all potential contaminants listed, screening levels and the measured levels at Pond 15 and the Otay floodplain site.
   c. On page 3.2-53, it states that arsenic and lead levels in nearly all samples exceed the most conservative screening levels. For arsenic, this is a human health screening level, and thus probably not a concern. However, lead levels at Pond 15 exceed the wildlife-risk based screening level. This is dismissed as a concern in the document by stating that screening levels are conservative and the actual potential for adverse effects is probably much lower. Please provide evidence to back up this assertion. It is also stated that levels of arsenic and lead in Pond 15 are similar to levels of these contaminant in the south San Diego Bay. Are there data to support these conclusions? If so, please provide.
   d. On page 3.2-53, the document states that although individual soil samples exceeded mercury screening levels, because the Pond-15-wide mean concentration of mercury is below the most conservative screening level, it is also below levels of concern for aquatic organisms or aquatic-dependent wildlife. Does this mean that a population level effect is not expected? What about effects to individuals? Please provide additional explanation.
   e. It should be clarified in this section that Pond 15 is currently used as a salt pond, and thus does not support a wide variety of aquatic organisms. Furthermore, the proposed project will result in the burial of these contaminated sediments, thus significantly decreasing the availability of contaminated sediments, and resulting in a net benefit.

5. Biological Resources
   a. The wetland delineation presented in section 3.3.1.4 identifies certain vegetation communities, such as brackish water and former salt pond bottom and borrow area, as non-wetlands. However, on page 3.3.-25, the last sentence of the 4th paragraph states that all borrow areas meet the Commission's definition of a wetland. It is unclear from the analysis provided in this section why all areas within the borrow areas were not identified as wetlands under the Commission's jurisdiction. Under section 13577(b) of the Commission's Administrative Regulations, areas that do not exhibit hydrophytic vegetation or hydric soils as a result of high concentrations of salts are still generally considered wetlands if the site exhibits surface water or saturated substrate for some time during each year. For other projects, the Commission has found that to meet the

hydrology criteria, a site must be saturated for at least two weeks in an average water year. Unless there is solid evidence that these areas do not meet the wetland hydrology criteria, the assumption, based on the 2011 surveys, should be that these areas are CCC wetlands and should be indicated as such on the figures and throughout the text of the EIR. I will note that in section 4, the borrow areas, although described as non-wetlands, are essentially treated as wetlands in regards to determining impacts and mitigation. I agree that this is the right approach, but it makes the designation of wetlands and non-wetlands more confusing (at least from a CCC perspective).

b. Land cover type "SPBB" is not included in the legend on Figure 3.3-6
c. To ensure that impacts to eelgrass, fish and other marine species are minimized, we recommend that MM-HYD-1 require installation of a silt fence for the first 24 hours after the Pond 15 breach or until turbidity levels recede to ambient conditions instead of making it optional.
d. Section 4.2 (p. 4.2-53) describes the potential for deposition at the mouth of the Pond 15 opening because of the relatively low tidal velocities. However, because of the lack of a sediment source nearby, the impact is described as less than significant. Although this may be the case, in the unlikely event that sediment is deposited at the mouth of Pond 15, tidal flows to the wetland could be muted or cut-off, thus endangering the habitat the project is designed to create. I recommend including a mitigation measure that requires monitoring of the inlet and remediation in the event that tidal flows are significantly altered.
e. MM-BIO-8 requires that estuary seablite (*Suaeda esteroa*) be planted at a 2:1 mitigation ration in newly created mid-high marsh areas. Is it feasible to remove existing individual plants and transplant them into the new wetland? If so, we would recommend incorporating this into the mitigation measure. In addition, although California Box-thorn (*Lycium californicum*) and wooly seablite (*Suaeda taxifolia*) are lower on the rarity index and thus may not require the same level of mitigation, is it feasible and/or desirable to include these species in the planting palette for the new wetland areas as well? If so, we would recommend incorporating these species into MM-BIO-8 at a 1:1 ratio.

6. Mitigation measures
   a. MM-VIS-1 – Please insert "Executive Director of the" before "California Coastal Commission" in the third line. We also recommend explicitly including a requirement that the revegetation plan include a monitoring plan (with specific methodologies identified), interim and final success criteria, and adaptive management measures.
   b. MM-HYD-1 – we recommend requiring a silt fence for the first 24 hours after the Pond 15 breach or until turbidity levels recede to ambient conditions (see 5.c. above). At a minimum, please clarify who determines if a silt curtain is necessary after the Pond 15 breach.
   c. MM-HYD-4 – we recommend editing the second sentence as follows: "The soil transport monitoring plan shall include ~~monitoring~~ operational protocols to ensure that unanticipated spills of transported soil material ~~would~~ do not occur from conveyor belt or slurry pipeline operations and monitoring protocols to detect any spills that do occur."
   d. MM-BIO-2, 3, 5, 6 and 7 – we recommend revising these mitigation measures to delete the specific acreages associated with Alternative B and C in the event that an alternative other than Alternative B or C, or a slight variation of either alternative is implemented.

Stating that mitigation for conversion of wetlands will be provided at a 1:1 ratio and mitigation for permanent impacts to wetlands will be provided at a 4:1 ratio should achieve the same effect but allow for some flexibility. Another potential option would be to require that a mitigation plan be submitted to the Service for approval that mitigates for impacts to wetlands as described in the FRP and as described above.

e. MM-BIO-9 – although it is stated in the project description, we recommend including a requirement that construction will not occur during the nesting season and define how the start and end dates of the nesting season will be determined.

If you have any questions, please contact me at 415/396-9708.

Sincerely,

KATE HUCKELBRIDGE
Senior Environmental Scientist
Energy, Ocean Resources & Federal Consistency Division

Attachments

Cc via email:
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Lisa Honma, SDRWQCB
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December 1, 2016

Kate Huckelbridge
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Re: Review of DDT dispersion calculations

Dear Kate:

This letter is to convey my review of material in connection with the estimation of DDT dispersion and deposition in the OTAY River Floodplain as a result of a large flood event. I reviewed the following documents:

Everest. Otay River Estuary Restoration Project Fluvial Hydraulics Study. Prepared by Everest International Consultants, Inc. for Poseidon Water LLC, October 2014.

Jenkins, S. and J. Wasyl. Tidal Hydraulics of Wetlands Restoration Alternatives In the Otay River Flood Plain, Carlsbad Desalination Project Marine Life Mitigation Plan. Jenkins Consulting. September 2010.

Jenkins, S., Y. Poon, C. Zeeman, C. Roberts. Sensitivity Analysis of Potential DDT Deposition in the Otay River Estuary Restoration Plan (ORERP) Post-100 Year and 50-Year Floods. October 2015.

My review focused on the Jenkins et al. (2015) report. I did review the other two reports and have some comments I can include in a more detailed report.

The Jenkins et al. (2015) report calculates the dispersal of DDT contaminated sediments as a result of 100 and 50-year floods on the Otay River. DDT accumulation was specifically calculated in the Floodplain Tidal Basin of the Intertidal and Subtidal Alternatives and in the Pond 15 Tidal Basin of the Intertidal and Subtidal Alternatives. Attention was paid to the differences in DDT deposition between the Intertidal and Subtidal Alternatives.

The following is a summary and critique of the dispersion calculation assumptions methodology:

- Fine sediment loads carried by the floodwaters from the watershed are estimated on the basis of calculated loads from another watershed flooding study. This calculation uses only the relative areas of the two watersheds to make the extrapolation, and did not consider other factors such as land use differences. In addition, the percent of fine material in the discharge is assumed to be 50% for the 50-year flood and 60% for the 100-year flood, based on limited evidence. Calculations of DDT dispersal were made with and without the inclusion of fine sediment discharge from the watershed. When the fine sediments were included they were assumed to be free of DDT. These results indicated that inclusion of the watershed discharge results in greater overall sediment deposition and lower DDT concentrations because of the dilution of eroded contaminated sediments from the Otay River Floodplain (ORF) by the clean upstream material. However, the calculated difference between the two project alternatives is relatively small. This is not surprising given the relatively simplified assumptions about DDT erosion and initial dispersion within the ORF (see below) that cannot resolve the morphologic differences between the two alternatives. The potential error in this calculation of the fine sediment discharge would be of more concern if this differential impact between project alternatives were greater.

- The Everest (2014) report calculates that 100 and 50-year flood flow velocities would be sufficient to erode all size fractions of the sediments. For this reason the calculation of DDT dispersion assumes that erosion occurs uniformly over the entire Otay River Floodplain (ORF). Calculations were made for assumed erosion depths of 1, 2, and 3 ft, based on measured depths to less erodible material. These are highly simplifying assumptions, but are consistent with the overall resolution of the approach used in the DDT dispersion calculations. For each assumed depth of erosion measured values of percent fine material are used to compute the total mass of fine sediments eroded from the ORF.

- DDT contamination is assumed to be present only in a portion of the ORF where sediment sampling shows significant DDT calculations relative to background. DDT is assumed to exist only sorbed to fine sediments, a reasonable assumption, but it is not clear whether it is supported by the if the sediment DDT samples. Two separate data sets are used to quantify vertical DDT profiles in the contaminated area. Once the vertical profiles have been established the total mass of DDT and the average dry weight concentration of the sorbed DDT in the eroded material can be calculated for each assumed depth of erosion. These assumptions seem reasonable, although highly simplified.

- The calculation of the dispersal of the eroded DDT during the flood period assumes that all of the fine sediments are carried as suspended load with a uniform suspended solids concentration, and a uniform DDT dry weight concentration determined by assuming that the sorbed DDT is redistributed between the fine sediments eroded in the watershed and fine sediments eroded in the ORF. Furthermore, it is assumed that at the end of the flood period the floodwater volume occupies a region "throughout the floodplain and adjacent South San Diego Bay as far north as nodal points at the Chule Vista Wildlife Reserve." No justification is given for the assumed extent of the floodwater volume in San Diego Bay. It is certainly possible that some of the sediment eroded by a flood is transported beyond this region and does not deposit in the study area, but it is a conservative assumption to assume that it does. Deposition of eroded fine sediment is assumed negligible during the flooding period, which is likely to be a good assumption given the high velocities computed by in Everest (2014).

- As acknowledged in the report, the calculations of DDT dispersal do not include a representation of any contaminated sediment by bedload processes, i.e. by movement of sediment without sediment resuspension. Given the uncertainties in computing bedload sediment transport and the high tendency fine sediments to be resuspended, this is a reasonable and useful simplifying assumption.

- The fate of the DDT contaminated sediments after the flooding period is calculated using a numerical model that represents the tidal currents in the study area. The numerical model is described in Jenkins and Waysl (2014) and is an example of a general flow modeling technique that is well-accepted and appropriate for this kind of flow environment, although no site-specific validation is presented for the flow calculations.
- The numerical model contains terms representing the following sediment transport processes: i) horizontal advection by tidal currents, ii) deposition to the sediment surface, iii) resuspension from the sediment surface, iv) consolidation to a partially consolidated fluid mud layer from which resuspension does not occur (at least that is what I think they assume - I recommend obtaining more clarity about this), and v) long-term complete consolidation to a fluid mud layer. The deposition calculation is based on a single value of the settling velocity based on an assumed characteristic sediment size. The resuspension (Equation 3) and initial consolidation (Equation 8) of sediment are calculated using empirical approaches from the literature involving assumed values of parameters. The final consolidation does not involve a calculated rate so the layer thicknesses before and after consolidation are determined by the assumed values of bulk sediment density, which appear to be reasonable.
- I am concerned about several aspects of the sediment transport calculations. First, the boundary conditions at the open boundary with San Diego Bay do not seem to represent any possible return on an incoming tide of sediment transported out of the domain on earlier outgoing tides. This area is at the end of San Diego Bay where I suspect tidal flushing is weak, so it is important to represent this return process in some way. Second, I am afraid that I have no quick way to estimate the likely error introduced by the assumptions about sediment resuspension and consolidation. These assumptions are critical to the calculation of the fraction of the eroded DDT that is deposited in the study area and the depth of the deposited layer. It would seem to me that some sort of sensitivity study to the parameter values would be appropriate to gain confidence in the overall calculation results. Given the publication dates of the key literature sources of the equations for resuspension and consolidation, I am also wondering whether more recent studies might have developed more robust predictive equations.
- The ecosystem impact of the deposited DDT depends on the exposure to DDT to organisms feeding on an "active layer" of sediment. For this study, calculations were made for assumed active layer depths of 20mm, 40mm, and 80mm. Average DDT concentrations (dry weight) were computed by averaging the DDT present in the actual layer deposited during the flood over the active depth. This seems to be a reasonable procedure, however, the results presented in Table 11 include both "initial" and "final" values of average DDT concentration that are different, with the final values being smaller than the initial values. The report says clearly on page 27 that as the deposited layer consolidates no sediment mass is lost or gained, so the dry weight concentration of DDT stays the same. The consolidation just converts a thicker layer of less concentrated mud (200 g/L) into a thinner layer of more concentrated mud (1200 g/L), but the sediment mass and thus the DDT mass in the layer stays the same. I get that averaging this DDT over 20 mm (as a feeding organism does) will reduce the average dry weight concentration of DDT by dilution with clean sediments. This is why the concentrations in the last 6 columns of Table 11 are lower than 70.2 ug/kg (the average dry weight

concentration of the DDT in the deposited layer). What I don't get is the "initial" vs "final" values given that nothing is changing the amount of DDT to be averaged over the 20 mm. I have been unable to reproduce the calculated initial and final values, so I recommend obtaining clarification on this point.

- As discussed above, the calculation of DDT dispersion makes a number of assumptions that greatly simplify the actual sediment transport processes and/or utilize highly empirical, and possibly dated, parameterizations of transport processes without exploring the sensitivity of the results to parameter values. With additional time and effort I could try to determine if there are more recently developed representations that would be significantly more reliable.

- One way to get a perspective on the importance of potential errors in the DDT dispersion calculations is to consider the inherent range of possible results. The assumptions and methods underlying the dispersion calculations are equivalent to the following steps: 1) Specifying the quantity of fine sediment entering or eroded from the ORF during a flood, 2) Specifying the mass of DDT eroded from the ORF, 3) Assuming that all of the fine sediments and DDT initially remain in the project area after the flood, but then are either flushed out to San Diego Bay or deposited in the sediments, 4) The exposure of organisms to the DDT is assumed to be related to the DDT concentration calculated by assuming the remaining mass of DDT in the deposited sediments is mixed over a specified active layer depth. The detailed numerical model calculations in Step 3 could be completely replaced by an assumption about what fraction of the total mass of fine sediments and DDT are deposited rather than flushed away (the fraction would be the same for both). Once this fraction is specified, the mass of DDT in the sediments is determined and Step 4 can be completed. My point is that if the fraction of sediment and DDT that deposits is large, say 50%, uncertainties in the DDT concentrations could be low by no more than a factor of two. But if the fraction is small, say 1% or less, then the calculated value could underestimate the total value by as much as a factor of 100 or more. I was unable to calculate the fraction of fine sediment and DDT that deposits because I did not have a value for the total surface area on which deposition occurs. I recommend that the authors of the Jenkins et al. (2015) report be asked to provide this fraction for the different cases analyzed.

Please let me know if you need any additional information in this regard at this point. I would be happy to answer any questions you have and to discuss additional analyses that might be useful.

Submitted by,

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Technical Memorandum
To
California Coastal Commission Staff

A Review of the Sensitivity Analysis of Potential DDT Deposition in the Otay River Estuary Restoration Plan (ORERP) Post 100 year and 50 Year Floods.

Prepared By

Dr. Steven Schwarzbach

1 December 2016

Introduction

In this memorandum I provide a review of the report *Sensitivity Analysis of Potential DDT Deposition in the Otay River Estuary Restoration Plan (ORERP) Post-100 Year and 50-Year Flood*. This analysis evaluated the effects associated with erosion of soils from the Otay River flood plain contaminated with DDT. A restoration of tidal wetlands is proposed in the immediate downstream area as mitigation for an off site project. The goal of the original report is to inform decision makers on the risk of DDT contamination to the fish and wildlife benefits of the potential wetland restoration. The restoration plan is considering two different combinations of wetland alternatives for the Otay River Floodplain site. Alternative B, referred to as the intertidal alternative creates 29.61 acres of various intertidal wetland, mudflats, and low, mid and high elevation salt marsh, and 3.80 acres of adjacent upland habitat. Alternative C creates 33.51 acres of habitat with 4.1 being upland, 4.48 being sub-tidal and the remaining 24.93 acres as salt marsh.

The report includes hydraulic modeling of 100 year and 50 year flood events and the resulting scour of the contaminated upstream flood plain with a focus on downstream deposition depths, distribution and concentrations of DDT contaminants in the surficial sediments of intertidal and sub-tidal systems. Soil DDT concentrations in the floodplain to be scoured have been characterized to the projected scour depth of 1, 2, and 3 feet with potential for dilution from cleaner upstream sources a possibility and with dilution also excluded. The result is a suite of scenarios of DDT deposition in the downstream wetland sediments based on various assumptions and some measurements. These various DDT sediment concentration forecasts are then evaluated for their wildlife and benthic life hazards through a risk assessment process that considers both bioaccumulation to two species of birds likely to utilize the wetland and direct toxicity benchmarks for benthic life.

My intent is this review is to answer the following questions about the report:

- Is the approach taken appropriate?

- Are the methods clear and justified?
- Are the sediment DDT concentrations properly and completely evaluated with regard to risk to fish and wildlife?
- Are the conclusions drawn reasonable and justified?
- Are there important issues that were unaddressed?

Ecological risk assessment is a probabilistic approach to assessing the likelihood and magnitude of exposure and the likelihood of adverse toxic effects of such exposures to various key ecological receptors. With a chemical such as DDT that bioaccumulates the assessment must also include an assessment of bioaccumulation potential to higher trophic levels in the exposure component. The process necessarily relies on numerous assumptions and expertise and best professional judgments for assessing relevancy of studies, applicability of data, and extrapolating data from what is known to what may be less known. Ecological risk assessment is a stepwise process. In each step the risk assessor is essentially making a decision on how to deal with uncertainty.

In the ORERP report, the assessment is a multi-disciplinary effort with input from four experts from different fields. Both risk managers, and each of the risk assessors as well, need transparency on uncertainty as uncertainty can be magnified in each step. In the risk assessment process where uncertainty is great the application of the precautionary principle may lead to the application of an uncertainty factor or a modified decision that is called "conservative". Uncertainty factors, or "conservative decisions" are designed to err on the "safe side" of doing no harm. The application of these factors may however sometimes lead to a quantitative conclusion that is surely safe but may be unachievable so balance is sought, sometimes not explicitly. Best professional judgement is applied to achieve a balance in risk but needs to be done so with transparency. With ORERP risk decisions the situation is even more complex than say setting a site specific cleanup goal or a water quality criteria. Because of the potential wildlife habitat benefits, there is no safe side to err on. Falsely rejecting a project because of excess caution may cause harm from lost habitat that would have met dire needs for the management of an endangered species. Falsely accepting a project that presents true and significant toxic risk to endangered species could also cause harm. Risk balance depends upon understanding as precisely as possible the potential costs and benefits of all alternatives, including no project.

While there are well over a thousand published papers on the toxicity of DDT to wildlife there are still as yet many uncertainties. Species vary greatly in their toxic responses, many species are poorly studied, lab to field extrapolations are uncertain, field diets and thus bioaccumulation factors vary, there are multiple toxic endpoints and mechanisms of toxicity and eggshell thinning, while widely studied and sensitive endpoint in some species is not a perfect biomarker of all toxic effects to all bird species.

With all this in mind, my approach here is to provide a review of the analysis with comments where I provide my professional opinion and try to answer the review questions. I then present my own quantitative analysis of the wildlife risk component, which informs my review. I felt I needed to do an independent analysis to examine how changing risk decision assumptions propagate to the conclusions about risk. I provide that analysis to staff as a tool built in microsoft excel that allows one to manipulate the input and do the same evaluation. The excel tool provides multiple worksheets that evaluate both the wetland restoration alternative and the no project alternative with different target species and different changeable assumptions regarding diet, allometric scaling equations, species, body weight, TRV, and sediment/diet ratios to calculate estimates of harmful and non-harmful sediment concentrations to birds.

Part 1. Review and Comment

Section 1.0 Introduction

I think it important to mention climate change right up front in the introduction.. There are two mechanisms where climate change impacts the analysis, sea level rise, and the altered probabilities of 100 year storm events.

The international panel on climate change has identified coastal ecosystems as areas that will be disproportionally affected by climate change with sea level rise projections ranging widely between 0.57 and 1.9 meters in mean sea level by 2100. The USGS has assessed 13 tidal marshes in San Francisco Bay using the Wetland Accretion Rate Model for Ecosystem Resilience (WARMER) and found that 95.8% of marsh area in that study will loose marsh plant communities by 2100 and transition to a relative elevation range consistent with mudflat habitat. (Takekawa et al, 2013. http://pubs.usgs.gov/of/2013/1081 /) Figures are provided in the appendices of the ORERP report that show future marsh habitat in the restoration alternatives with sea level rise but these are not particularly useful except to show there is some habitat. What assumptions however do these figures represent of sea level rise? What model was used to project elevations?

How do habitat acreages change and how might the design of the wetland accommodate the projected sea level rise and maintain habitat benefits are just some questions.

Similarly climate change presents a challenge to the modeling of the probability of 100 year floods. Stationarity, the idea that natural systems fluctuate within an unchanging envelope of variability, the foundational assumption of water resource engineering and planning, is a concept now accepted as compromised by climate change. A poleward expansion of sub-tropical dry zone's is occurring due to climate warming and climate change is altering the means and extremes of precipitation, global evapotranspiration, and rates of river discharge. (Milly et al., 2008). Climate change scenarios have been done for California that indicate up to 10% change in precipitation (Cayan, 2008). I am not sure exactly how the risk assessment should take this into account but it should at least be acknowledged as a source of increased uncertainty.

The second paragraph of the introduction describes the report as having six sections. The final report has seven with the late addition of the 50-year flood analysis.

Section 2) Erosion Analysis 100-year flood

On page 8 it is noted that the Otay River watershed sediment discharge is scaled to a smaller watershed where sediment discharge has been measured. The Otay watershed is 2.42 times larger and as noted potentially more susceptible to soil erosion during a flood event as it has more open space. Buena Vista Creek watershed (19 mi^2) discharged 603,000 cy of material in a 100 year flood (Everest 2008). The report proceeds to scale up sediment discharge from Buena Vista creek to Otay River watershed (46 mi^2)in a linear fashion by multiplying by 2.42 to generate an estimate of 1,460,000 cy discharge from Otay River, below the Savage Dam. I'm no hydrologist but a linear scaling of a three dimensional watershed flow seems off. More likely the proper scaling between watersheds is exponential. (I believe USGS could assist with a more accurate sediment discharge estimate.) It seems to me upstream sediment dilution in the Otay basin is being underestimated, perhaps greatly underestimated, in an effort to be "conservative". This could lead to an overly high estimate of the final DDT concentrations in scenarios where upstream sediments contribute to final concentrations.

On page 12 under *section 2.2) DDT Concentrations of the Eroded Soils* there are estimates provided as Table 3 for DDT concentrations at erosion depths of 1, 2 and 3 feet. I am puzzled by the estimate that distinguishes the interval between 1 and 3 feet because the intervals between 1 and 2 and 2 and 3 were not measured, only the interval between 1 and 3 feet, yet average concentrations differ. How was this accomplished?

It may be worth noting that in the last flood of this approximate size, in 1916, the downstream sediment contribution from the upper Otay River watershed was greatly enhanced by the overtopping and eventual failure of the lower Otay Dam, which was of earthen construction. The drainage area above the dam includes another 98.6 mi^2. The flood was produced by a series of storms in rapid succession over a month long period with precipitation totaling about 19 inches. A peak flow into the reservoir of about 32,700 cfs was produced shortly before dam failure McGlashan and Ebert, 1918. By comparison the peak flow in the 100 year event stream channel below the dam is presumed to be 22,000 cfs. The replacement, Savage Dam, completed in 1919, was last inspected in 2007 and is considered to be safe.(http://inewsource.org/2016/05/16/san-diego-county-dams-old-but-still-passing-muster/) .

Section 3) Sensitivity Analysis of the post 100-year Flood DDT Deposition

On page 14, figure 17 shows the upper northeast cell at 0 ug/kg DDT. Because zero can be approached but not measured the statistical custom is to either use the detection limit or some fraction of the detection limit as the multiplier, not zero. The diagram reports to the nearest 0.01 ug/kg DDT concentration. If one uses 0.005 as the concentration in this cell that has 283,809 sf, then I calculate a DDT load estimate of 193 ug as the contribution from this cell to downstream loading. I recommend they remove the zero use some reasonable fraction of the detection limit if not the detection limit itself, but the actual number to use is a judgement call best made in consultation with the analytical chemist who knows the data or just go with half the dl.

Section 4) Suspended Sediment Transport and Deposition

I agree with the statement on page 16 that "deposition of the silts and clays that contain DDT will be a slow process which will extend for many tide cycles depending on the local water depth". My opinion is the tidal redistribution of DDT laden suspended sediments within the intertidal marsh is likely not going to result in a final uniform depth, throughout the marsh. This heterogeneity adds additional complexity and uncertainty to likely DDT sediment concentrations within the marsh. For an appreciation of tidal sedimentation dynamics please see Appendix 1. *Overview of Tidal Marsh Geomorphology by Josh Collins, 1999.*

Section 5) Post-Flood Tidal Deposition Simulations for the 100-Year Flood

Page 21, 3rd paragraph describes pond 15 tidal basin as having a longer residence time and a resulting sediment deposition thickness that is "nearly double". I was unclear about "double of what"? How is double thickness of pond 15 reflected in the final output in table 11? Or should it be?

Why are the intial post-flood concentrations in the top 20 mm of sediment identical in scenarios where the entire 3 feet erode and are diluted by the upper watershed to the 3 foot scenario where no dilution occurs? The footnote at the bottom of table about miscalculated columns is confusing.

Section 6) Post-Flood Tidal Deposition Simulations for the 50-Year Flood

Again I think the linear scaling of the watersheds between Buena Vista and Otay River underestimates sediment contributions from the upper watershed below Savage Dam in the 50-year flood.

Section 7) Biological Implications of the Post-Flood Deposition Simulations

7.0) The approach taken in the ORERP for assessing the potential for negative impacts to fish and wildlife in the report is twofold: The first is to evaluate risk of direct toxicity to benthic organisms using already established toxicity benchmarks for DDT adverse impacts in estuarine sediments and, the second, is to use the approach developed at the USEPA for setting wildlife criteria using a stepwise approach that back calculates permissible sediment concentrations using dietary concentrations associated with No Observed Adverse Effect Concentrations (NOAEC) or Low Observed Adverse Effect Concentrations in field or laboratory studies and allometric equations on rates of food consumption based on body weight. Extrapolations from one species to another is required and often from effect concentrations to no effect concentrations. I believe both approaches are appropriate for screening level assessments for DDT in this situation, but they are not without uncertainty. What remains under represented however, is any site-specific assessment of potential toxic impacts of DDT to fish in post 100-year and 50-year floods. This may be justifiable if screening level impacts to benthic biota and/or birds ends up sufficiently robust to drive a decision, but given that the restoration is mitigation for coastal fish entrainment it seems to at least some degree estuarine fish toxicity should be addressed.

7.1) How to consider the many forms of DDT and metabolites?

DDT occurs primarily as two isomers the ortho, para isomer, op DDT which is 10 to 20% of technical grade DDT, and para, para isomer or pp DDT. There are two persistent metabolites in the environment, DDE and DDD. DDE is most common and produced by bird liver, and pp DDE is the eggshell thinning agent. One mechanism of toxicity of DDT is to affect sodium channels in nerve tissue and I think most isomers and forms act upon this mechanism. Another mechanism the anti-androgenic effects of the pp isomer and another is the estrogenic effects of the op isomer which is five times more potent than the pp isomer however, it is about five times less abundant. The multiple mechanisms, the bioaccumulative nature and differences in species sensitivities and metabolic capacity for transformation of complex organics adds uncertainty to inter species comparisons. The authors of the ORERP wildlife assessment decide to lump all DDTs together for purposes of the risk assessment. In the end I think that is acceptable to do and to do otherwise would not change the outcome of the screening assessment.

7.2) Wildlife Receptors

The choice of wildlife receptors for the risk analysis can have large effects upon the conclusions. The authors chose to use two species based on the stated criteria that they were "representative of potentially most exposed aquatic-dependent wildlife to DDT in marsh sediments". These were the Light-footed Ridgway Rail (*Rallus longirostrus levipes*) [species name on page 36 is not correct] and the Snowy Egret (*Egretta thula*).

Criteria that I would use to evaluate the selection of target species include the following:

1. The species has a high likelihood of site-specific exposure.
2. The species has site-specific management significance.

3. The species, as part of a suite of species assessed, provides good trophic level representation for a bioaccumulative contaminant.
4. Exposure response data is available for the species or close taxanomic surrogates.
5. The species is relatively sensitive to the toxicant of interest so protecting it will protect many others.
6. The local population of the species is of regional conservation interest.
7. The species will spend most of its life cycle on site, including reproduction which a key toxic endpoint for DDT exposure in birds

The Ridgway Rail meets all of these criteria. It is a state and federally protected endangered species. This means that consideration for the protection of individuals, not just populations, is not only desirable, but also required by law. Field and laboratory data exist on the genus Rallus and good field data on exposure and response exist on this specific sub-species. The Snowy Egret is a common coastal species found in a wide variety of environments. It meets criteria 1, 3 and 4 and possibly 5. I suspect that the above criteria were also in the minds of the authors when they chose these species and I think they are good choices, but they may not tell the whole story of DDT risk in the ORERP.

The authors use the Ridgeway Rail as a representative of an exclusively invertebrate trophic pathway. In fact the Ridgeway rail is a an opportunistic omnivore that eats, lots of crustaceans clams worms and snails, but also will consume mice, fish, frogs, spiders, and even plant material as 4% of its diet. (Zembal and Fancher, 1988). They use the Snowy Egret as a representative of an exclusively piscivorous bird. In fact the snowy egret also consumes crabs and insects and other prey. Despite these dietary differences from the risk assessment assumptions I do think the egret and the rail are complementary trophic consumers. In my excel tool you will be able to adjust dietary assumptions regarding these species to see the impacts on the final sediment numbers.

The authors also note the choice of the snowy egret was in part because it is smaller bird than many other wading birds that could have been chosen as target species and that makes them a “conservative species”. They do not explain further. Because there is not a single equation shown in the section 7 risk assessment the reason for this may not be apparent to most readers. The reason is that the allometric equations which are used to scale food consumption in birds show that smaller birds consume proportionately more dry matter in their diet. The choice of body weight for the target animal of choice is therefore a key risk decision. I agree with the choice of egret but find they do a poor job of explaining it and are not transparent with equations and the numbers used. Supplemental information provided at my request showed the author used a 370 grams for the egret. The Birds of North America profile for the species gives the average weight for adult egrets as 369 g, which is what I use in my assessment. A 1 g difference is trivial so my results are similar to the authors. We both used the equation for Charadriiformes:

$$(\text{g Dry matter ingested/d}) = 0.522*(\text{bw g})^{0.769}$$

While egrets are not Charadriiformes I agree with the decision to use this equation because it seems the best fit of the 17 available. I include page 9R of Nagy, 2001 which shows all the equations as appendix to this review. It lists the formula for each bird grouping and importantly the error around the allometric regression. The species deviation for the formula for all birds is 30% for dry matter ingestion. For specific groups of birds the error is usually less. For Charadriiformes the deviation is 21%. That error propagates through the entire risk analysis but disappears when a single number is generated at the end.

For the LFFR I was able to obtain a range of weights from the Sea World web site. Sea world is engaged in a captive propagation effort for the rail so they have the best data on the species. Reported weights for rails ranged from 227 -398 g. (https://seaworld.org/Animal-Info/Animal-Bytes/Birds/Light-Footed-Clapper-Rail)Because in rails females are smaller and because protection of every individual is the goal for rail management I use the minimum weight, 227 g, in my assessment. In rails females are smaller than males. The female should be the choice for any assessment with reproduction as the toxic endpoint.
In a forwarded email from the authors by CCC staff I was provided supplemental information on rail weights and equations. The email states the assumed body weight for rails was 270g and the allometric equation used from Nagy, 2001 was again for Charadriiformes. When I duplicate the calculations I can only achieve the result used by the authors in section 2b, first paragraph on page 40 by using the body weight of 398g. When keeping everything else the same I find that at 227 g my sediment screening result is 14 ug/kg in sediment for the TRV of 0.014. The maximum weight used by authors results in a screening result of 16 ug/kg sediment for this TRV (15.6 but they rounded down and I rounded up). (See excel worksheet for LFFR). If you use their high TRV of 0.227 it also apparent that the highest body weight for rails was used. Table 10 shows this corresponds to 240 ug/kg in sediment, but that differs from my worksheet

which shows it should correspond to 253 ug/kg.   Using the lowest body weight for rails and the highest TRV results in a sediment screening level of 223 ug/kg. The higher the TRV the more difference a higher body weight assumption ends up making in the final result. A more conservative TRV means this assumption is less important.

7.3) General Approach   [for birds only]

This section of the report covers the rest of the steps needed to get to a sediment number after a wildlife target organism is selected. The authors discuss in this section the setting of tissue targets for Avian Eggs. They review available data specific to rails and egrets, some of it very useful for rails in particular from Goodbred, 1996. They review dietary targets for fish and invertebrate diets. They apply allometric scaling equations from Nagy, 2001 to determine ingestion rates (again without showing calculations or formula - numbers just appear). They select three possible TRVs (reference doses) based on a literature review and apply an uncertainty factor of 2 to the lowest which was already in the most sensitive species but not the highest TRV which is in a lab study with the least sensitive bird. They apply ingestion rates and the TRVs to determine three species-specific tissue screening levels for diet for both species using the following equation:

$$TRV_{mg/kg(bw)\text{-}d} / (\text{Ingestion Rate}_{g/d} / \text{Body Weight}_{g}) = \text{Screening level in diet}_{mg/kg}$$

They then divide their six screening levels in diet by either 3 or 27, the ratios they have determined to use for a diet of 100% invertebrates or fish respectively for the rail and the egret and generate 6 possible sediment screening levels that range from 2 to 240 ug/Kg.   My opinion, which I develop and support below, is a better answer is likely within a much narrower range somewhere between these numbers but is closer to 2 than 240.

DDT NOAELS, LOAELS and TRVs for Avian Reproduction

NOAELS and LOAELS are empirical benchmarks that have become key for ecological risk assessment purposes and are widely used in ecotoxicology. But not all NOAELS and LOAELS are of equal value and their use has increasingly been criticized (Landis and Chapman, 2011). By themselves they are single point estimates along a dose response curve that more accurately describes the relationship between dose and toxic response than any single point on that curve. In laboratory studies in particular there is weakness. Frequently NOAELS are just the lowest particular exposure not different form control or no-exposure. NOAELS are subject to the experimental choices of those doing the laboratory testing and are inconsistent between studies. Landis and Chapman, (2011) argue that NOAELS, because they represent only a small slice of the data ,by themselves don't have the power to show an effect that may really exist at that level of exposure and point to curve fitting as a better approach to generating a true NOAEL. A comparison between results obtained with curve fitting and the determination of LOAELs and NOAELs for the same data sets has been done. Moore and Caux (1997) demonstrated that NOAELs correspond to an EC 30 to EC10.

In my view, field derived NOAELS derived from wild animals exposed to a variety of concentrations through time have more power, particularly and especially if the effect can be duplicated with laboratory exposures. The work by Jeffrey Lincer in lab and field with Kestrels is a good example of this. Adverse impacts to avian reproduction in Falconiformes and Pelecaniformes provide the best information on NOAELS and LOAELs for DDT in the literature in my view.

The best example of a solid NOAEL is provided by the authors at the bottom of page 37 and the top of page 38 of the risk assessment, and comes from the work of D. Michael Fry (1994) on Pelicans and Cormorants in the Southern California Bight. The authors do exactly what you should do. They use a long term field study that can be used to develop a curve from which a true no effect concentration can be derived in two of the most sensitive bird species ever assessed for DDT, the Brown Pelican and the Double Crested Cormorant. The NOAEL is presented as the within egg concentration of DDT, 1.5 mg DDT/Kg ww for the Brown Pelican and 1.2 mg/kg ww for the Double Crested Cormorant. These concentrations need to be converted to dietary concentrations to serve as a TRV but they are the gold standard for a DDT NOAEL for eggshell thinning.

The authors proceed to convert the pelican egg concentration to a dietary concentration for the rail and the egret through a process that I can only characterize as magic. There is no formula provided just a cryptic footnote in table 8 on page 41 that says "based on field collections in southern California". It appears that the discussion of field data under 2a) leads to the ratio decisions but I honestly can't be sure even though I have read this many times. It is clear the Pelican egg concentration is divided by 50 to produce the target number for rails of 0.030 mg $_{DDT}$/kg diet based on eating 100% invertebrates and divided by 25 to produce a target diet of 0.060 for snowy egret based on eating 100% fish. This makes no sense to me that a strictly invertebrate diet would have a higher bioaccumulation factor than would a strictly fish diet.

The second TRV of 0.014 mg/kg-d again relies upon field studies of the Brown Pelican, this time by Dan Anderson in the 1970s as interpreted by the USEPA in 1995 for the Great Lakes Water Quality Initiative, but with a twist of applying an uncertainty factor of 2 rather than 3 to adjust from a LOAEL to a NOAEL as done in GLWQI or as recommended by the USEPA Region IX Biological Technical Advisory Group in California. There are actually two uncertainties here. One is the extrapolation from a LOAEL that was associated with a 30 % decrement in Pelican reproduction to a NOAEL for Pelicans. The second uncertainty is the application of the NOAEL between two different species. As the authors of this section note on page 38/39, "there is not data to determine if rails are less sensitive than pelicans."

I can accept applying an uncertainty factor of only 2 for the egret as reasonable for this site but I also think the risk assessor has an obligation to make the case to the risk manager audience about the circumstances that are driving that decision and be up front that it is counter to what is usually done. The USEPA Region IX biological Technical Assessment Group provides wildlife risk assessment guidance on the use of an uncertainty factor of 3 in adjustment of the DDT LOAEL to NOAEL. IF the guidance is followed the TRV for the Anderson study should be applied for NOAEL purposes as 0.009 mg/kg-d instead of 0.014 mg/kg-d. If EPA BTAG guidance is followed the SLV for the egret drops to 1 ug/kg to from 2 ug/kg. I already think 2 is too protective for reasons I discuss later (dietary assumptions) so I think this adjustment may be acceptable but not my preference in terms of approach because it is technically less rigorous and there are other ways to modify the risk assumptions that make more sense.

The TRV uncertainty factor adjustment for the rail risk assessment is more troubling. They list the rail SLV as 15 for the Pelican LOAEL to NOAEL derived TRV. Using BTAG guidance and the same body weight (398 g and sediment diet assumptions as in their calculation) the resulting SLV is calculated as 10 ug/Kg.

And the body weight issue here also troubles me. A 398 g Rail is a male bird. A reproductive risk assessment for DDT should be done using female body weights. (Females are smaller). Why does this matter? Body weight assumption is a risk decision, which becomes more important at the TRV multiplier is increased. The document did not provide body weights used for any risk calculation but I was able to deduce it through a back calculation of the Charadriiformes equation. SeaWorld lists body weights of rails as varying from 227 to 398. Changing BW to 227g lowers the SLV by another 2 ug/kg when applied to the 0.014 TRV and about 1 when applied to the 0.009 TRV.

The third TRV chosen is the most troubling. The TRV of 0.227 is 8.4 times the well document LOAEL in the Pelican at is taken from Ecological Soil Screening Levels developed for evaluating terrestrial sites as contaminated by DDT. The approach relied upon the geometric mean of NOAELs 105 laboratory studies done primarily in Galliform and Columbiform birds. The study which represents the chosen TRV (Cecil et al, 1978) is a study to compare the potential for DDT and PCBs to induce liver mixed Function Oxidase enzymes in white leghorn chickens. I wouldn't use this TRV even if the restoration was planning a free range chicken ranch on the wetland. This TRV has at least 4 uncertainty factors that should be applied. One would be for the using the most bullet proof, DDT resistant bird on the planet, the white leg horn chicken, as your test species. 2. Is the toxic endpoint of MFO induction which is unrelated to eggshell thinning. No effect of DDT was found at any Dose in the study so there is no does response toxicity curve at all if there were it would be unrelated to the enpoint we assess in this effort: embryotoxicity. 3. Is the uncertainty in converting data from an insensitive species to the requirements of protection of an endangered species at the individual not population level and 4. Is the application from laboratory to field. The result is of accepting such an unadjusted TRV is a sediment value so high as to be unprotective of all 8 species of wildlife for which I ran supplemental calculations. This TRV should be discarded from the analysis in my judgement.

Biota/Sediment Ratios for Calculating Wildlife Risk Sediment Concentrations

To connect dietary concentrations in fish and invertebrates to sediment concentrations the authors rely upon their experience and data collected in the

field to develop a ratio of 3 for invertebrates to sediment and 27 fo fish to sediment. Based upon my review of Goodbred et al., 1996 I'm willing to accept these ratios. Higher fish numbers could be justified, particularly for larger birds but for these birds in this setting I think 27 is reasonable. What I don't think is reasonable however is to continue to insist the rails eat only crabs and snails. The literature just does not support that (Zembal and Fancher, 1988). It also does not support the assumption that snowy egrets eat only fish. (https://birdsna.org/Species-Account/bna/species/snoegr/foodhabits#diet)
Keeping the ratio of 3 for invertebrates and 27 for fish one can recalculate a diet to sediment ratio by the proportion of each in the diet.
If one assumes that rails consume 80 % crustaceans and 20% fish the diet to sediment ratio changes from 3 to 7.8 using the equation
Proportion invert x (invert./sed.) + Proportion fish x (fish/sed.) = diet/sed

$$0.8(3) + 0.2(27) = 7.8$$

The egret ratio of diet to sediment should be like wise adjusted to reflect a 75% fish diet and 25% crustacean diet by the equation .75(27) + .25(3) = 21.

A Reality Check from Mugu Lagoon

What are some concentrations that make sense from what we know about the slope of the DDE curve in avian eggs from many species and field observations of how what might be harmful concentrations might relate to sediment concentrations. The place to look is Mugu Lagoon, a moderately contaminated site where Rails reproduction struggles and DDE in eggs is slightly elevated, and good data exist on DDE bioaccumulation in rail diet and sediments.

Lets assume that preventing harm to th clapper rail is the most important management endpoint for evaluating DDT Risk. Lets further use real data from Mugu Lagoon the SC coastal marsh with highest DDE concentrations in rails to derive sediment to diet and diet to egg concentrations and then examine various egg concentrations for potential harm and calculate associated sediment concentrations.

To see what happens to DDT predictions for sediment just using ratios, I will set four target concentrations for total DDE in Rail Eggs.

1.5 ppm The NOAEL in Pelican eggs will be set as a NOAEL for Rail Eggs. We can have the most confidence in this number

2.13 the mean in Rail eggs observed at Mugu Lagoon will be considered a threshold of harm to Rail reproduction or LOAEL. This is supported by the finding that eggshells at Mugu were 8% thinner than at other sites and that the thickness within Mugu eggs was correlated to DDE. This is an effect.

3.5 Assume the rail is maybe a little less sensitive than the Pelican and set 3.5 ppm as the mean associated with impacts to some individuals rather than population collapse as in the Brown Pelican at 3.0 ppm.

Assume the dose response curve is not as steep and set severe harm to a local population at a concentration of 12 ppm in eggs. These are assumptions that assess rails as sensitive but not quite as sensitive as cormorants and pelicans. This seems prudent given the precarious status and limited genetic diversity of the species. We can have more confidence in the lower numbers than the upper ones of course, but the problem is not how low you can go but how high.

Table 1. Mugu Lagoon projections derived from field ratios and concentrations.

Effect	Egg [DDE] Mg/kg	Egg/crab	Crab [DDE] mg/kg	Crab/Sed	Sed [DDE] mg/kg	[DDE] ug/kg ww
NOAEL	1.5	24	0.06	2	0.03	31.3
LOAEL	2.13	24	0.09	2	0.04	44.4
Harm to individuals	3.5	24	0.15	2	0.07	72.9
Severe Harm to local population	12	24	0.50	2	0.25	250.0

Using the maximum and minimum rail weights of 227 and 398, reasonable LOAEL and NOAELs derived from the Pelican data of Anderson that vary from 0.009 to 0.027 mg (DDT)/Kg (BW) –day, and either a diet composed only of invertebrates or invertebrates and fish one sees a range of potential sediment numbers for no to low effect that ranges between 3 to 16 ug/kg dw for rails. If you consider the allometric equation error is 21% you might adjust upwards to 19.36 ug/kd dw for the fattest, least sensitive, fish eating rail. Mugu ratio approach gets a NOAEC for sediment of about 55 ug/Kg dw. This approach generates what are also too high concentrations two to three orders of magnitude greater than that do not seem likely to be in the safe zone for rails to me. I am not proposing table 1 as an alternative because the methodology is flawed and it underestimates risk. However it does give us some clues about what ballpark protective and and ball park harmful concentrations might be in. The SLV of 240 ug/kg for rails should be discarded.

Benthic Community Sediment Targets

To evaluate toxicity to benthic organisms the authors of section 7 propose to use the marine and estuarine sediment guidelines from Long et al 1995. For DDT The ER-L is 1.58 ug/kg dw and the ER-M of 46.1 ug/kg. Above the ER-M bad things are sure to happen to benthic biota. However, Long et al caution that for total DDT and p,p' DDE there were relatively weak relationships between concentrations and the incidence of effects. It should also be noted that using p,p' DDE would have resulted in lower guidelines for the ER-M at 27 ug/kg.

Since p,p' DDE tends to make up most of the DDT in Total DDT this is worth keeping in mind as impacts are assessed. A statement in Zeeman et al 2008b, (F&G marsh study) on page 10 is of interest: "*Wildlife risk-based screening levels tend to be lower than the benthic community risk-based values for substances that have a high potential for bioaccumulation*."

Predicted Sediment Concentrations and Comparison with Screening Criteria

DDT Concentrations (0.0013 and 0.0016 mg/kg dw) measured in San Diego Bay mudflats in two unpublished studies by USFWS are offered as "background concentrations. These concentrations appear to be used for estimating concentrations that will be overlain by flood deposits. Table 11 provides estimated sediment concentrations that should be compared with wildlife Criteri a after mixing in receiving sediments. Calculated concentrations the upper 20mm of the sediment profile are highest and highest in the short term with concentrations as high as 29 ug/kg. Longer term concentrations do not exceed 7.9 ug/kg in any scenario in this layer. Many organisms feed at the surface and consume detritus. These consumers may receive higher doses of DDTs than other organisms. These concentrations are lower than the ER-M for effects to benthic organisms.

The assumption that receiving sediments will be a background levels is not supported by any on site data with the Otay River area. In fact, results from the Zeeman 2008a salt works report provides a strong indication that as one moves landward toward the Otay River, DDT increases in sediment. DDT in mudsuckers from the Otay river were 84.7 ppb ww while in Pond 11 the same species had a concentration 9.8 ppb ww. As Zeeman noted mud suckers have high site fidelity and burrow into the mud so this difference likely represents a local DDT contamination that already exists in the Otay River channel sediments. The assumption of clean receiving sediments has a powerful influence on the final sediment concentration estimates. However, it also seems highly unlikely that a 100 year flood would not mobilize diluting sediments from the lower watershed and the volume of that dilution may actually be underestimated by the linear extrapolation between Buena Vista Watershed and the Otay watershed.

Table 10 of the report summarizes the final wildlife risk based sediment numbers. I can duplicate all the risk assessment calculations using the body weights provided in the supplemental information except for the high dose rate for the rail which must have used a lower weight. I calculate 253 ug/kg dw in sediment and they calculate 240. Regardless, I believe the weight of evidence suggests concentrations this high would have disastrous consequences for rails and should be discarded as guidelines for safe levels of DDT in tidal marsh sediment.

My preferred estimates for Ridgeway Rail uses a higher diet to sediment ratio and the lowest possible body weight. The impact of these changes was to lower the estimated safe level for rails below the lowest value estimated in the report of 15 ug/kg to between **6 and 11 ug/kg dw**, though this overlaps with a LOAEL estimate of **10 ug/kg**. Another reason my rail calculation is about 5 ug/kg lower than the report's wildlife risk assessment is I used the lowest reported weight of a Ridgeway Rail and they used the highest – a 42% difference. The higher the TRV that is used the greater the influence of the body weight decision on the final safe calculation. I justify this in that females are the target sex of reproductive impairment form bioaccumulation in oviparous vertebrates and female rails weigh less than males. My assumption was intended to provide safety in the calculation to the most vulnerable of the rail population adults. The result is predicted final sediment concentration estimates in flood born deposits, which, if they stand further scrutiny, are in a safe zone for rails in all scenarios except during initial settling.

For egrets the effect of my dietary assumption change was to alter safe estimates from a range of 3 to 6 upwards to a range of **4 to 8 ug/kg dw**. Egrets appear at greater risk than rails even when their diet is adjusted to add invertebrates at a realistic ratio. The Ridgeway rail however is more vulnerable as a species and consequently of higher management concern. Protective egret concentrations would protect rails as well.

The proposed wetland restoration projects if successful in the goal of providing clean and suitable quality habitat will provide long term benefits to the Ridgeway Rail. Rail densities vary considerably and depend on suitable cover and nesting but densities up to 2 rails per hectare have been seen in the best habitat. (Zembal and Massey, 1987) Conceivably a 76 acres wetland complex as envisioned could support from dozens to over a hundred pair of rails per year. This would be an extraordinary accomplishment.

Part 2. Supplemental Wildlife Risk Assessment for Six Species

To evaluate the effect of target species selection on safe sediment estimates I conducted an assessment on 6 additional species using the same methods and many of the same assumptions. The salt works in southern San Diego Bay near the Otay river inputs are home to 94 species of birds, and 7 endangered species so many wildlife issues are in play. The additional species I chose included the Belding's Savannah Sparrow, Northern Hen Harrier, Black-necked Skimmer, Brown Pelican, Gull-billed Tern, and Black-necked Stilt.

Detailed results of all calculations are provided in an excel file submitted with this report where each species has its own worksheet. Potential body weights for the added birds range from 17 g to 2,824 g and diets items in the 6 additional species range from seeds and insects to fish, mammals and even Ridgeway Rails. By adding additional species I can also assess how risk varies with project and without project on different target species. I can also see how risk changes as I span more trophic levels more taxonomic diversity in diet and allometry and address risk to a greater range of wildlife conservation issues of concern. Allometric equations that I use from Nagy, 2001, include equations for Pelecaniformes, Charadriiformes, and Carnivorous birds and for the passerine I can use the direct data from the Savannah sparrow tests of Nagy.

Target Species and Results

The Belding's Sparrow (*Passerculus sandwichensis beldingi*) is a state listed endangered species and an obligate non-migratory endemic restricted to tidal marsh habitats. In San Diego County this amounts to 1,182 hectares (Unitt, 2004.). It meets my criteria 1,2,3,6 and 7 for selection of a target species. Nagy (2001) has used this species for his doubly labeled water study of field metabolic rate so an allometric equation is not required, so there is some greater precision to the feeding rate estimate which turn out to higher than the passerine equation predicts. The bird requires an enormous amount of food relative to its body mass but eats both seeds and insects. I have therefore assigned it a diet to sediment ratio of 1.5. Passerine birds are vulnerable to the anti-androgenic effects of in egg exposure to p,p' DDE with the result that song centers in the brain can be altered to impair the learning of song which has reproductive implications. The concentrations needed to produce these effects however are quite high compared with those that impair water birds (Iwaniuk, 2006). For this reason I believe the best TRV is the one for Kestrels of 0.11 mg (DDT)/Kg (BW) – day. Using this TRV I generate a predicted safe sediment concentration of **79 ug/kg dw.** This is a concentration well above those I believe safe for egrets and rails. Protecting Ridgeway Rail and Snowy Egret also protects Belding's Sparrow.

The Northern Hen Harrier (*Circus cyaneus hudsonius*) is a medium sized widely distributed low flying raptor. It consumes mostly small mammals and birds of grasslands and marshes, habitats where it also nests on the ground. In the nearby Tijuana estuary it is reported some harriers learned to specialize on the Ridgeway Rail as prey (Unitt, 2004). Harriers are sensitive to DDT and in the peak of the pesticide era, in the decades of the 1950s and 1960s, it suffered 19% eggshell thinning in the western U.S. and experienced a regional population decline due to DDE induced productivity impairments. The Harrier is not a listed species but is declining globally and is listed as endangered in 8 other states. Dependence on declining wetland and grassland habitat makes this species vulnerable. Restoration of large wetland tracts filled with safe to eat prey will benefit the species. This species meets my target species selection criteria 1,2,3,4,5,6 and for a few individuals perhaps 7 as well. I believe the kestrel TRV of 0.11 mg (DDT)/Kg (BW) –day to be the best and most representative TRV for this species. Its trophic position in the marsh is unique in that it eats birds that eat fish that eat invertebrates and it eats small terrestrial mammals that eat primarily plant material. I used the allometric equation for carnivores to estimate daily dry matter ingestion rates for an adult body weight of 515 g. I have examined the impact of two diet to sediment ratios of 30 and 10. These two values generate a NOAEC that varies between **22 and 66 ug/kg**. These safe concentrations are well above long-term post flood sediment concentrations but within the range of impacts of concentrations projected during initial settling periods. A short period of increased DDT exposure post flood could be reflected in a minor increase in risk to some individuals of this species. Long term DDT concentrations that protect the rail would protect the Harrier as well, in more ways than one.

The Black-necked Stilt (*Himantopus mexicanus*) is a breeding bird in the southern San Diego Bay Salt Works. Unitt describes the Salt Works as providing "the ideal foraging habitat, while the dikes offer ideal nest sites." I would add that providing inter-tidal marsh as part of the complex would create excellent habitat for newly hatched chicks to hide from predators like gulls and gull-billed terns. Stilts would thrive in such an environment if DDT is at safe concentrations. Stilt diet is nearly 97% invertebrates from wetland environments. The Black-necked stilt meets the first four of my 7 criteria for a good target species. Ideally I would like to have a toxic endpoint for growth and development of young birds to apply to a stilt chick that will forage in the marsh. This would assess another life stage but time does not permit this analysis.. Therefore, I apply a Charadriiformes allometric equation to the adult body weight of 169 g and used the diet sediment ratio of 3. Using the TRV of 0.014 developed in the report I calculate a Safe sediment concentration for protection of adult Stilts at **13 ug/kg**. Protections for the Snowy Egret and the Ridgeway Rail are also sufficiently protective to provide safety from adverse impacts of DDT to the Black-necked Stilt.

Target Wildlife Species that Forage in The Pelagic Food Chain

The following species evaluated for DDT risk will largely not be foraging in the restored intertidal marshes. If the intertidal marshes capture some of the post DDT flood and can safely sequester that DDT in other food chains it may reduce the post flood DDT risk to these species. Concentrations associated with the short term flux of DDT into pond 11 may be best comparison for risk for these species. Though deposition times were longer here and depths doubled. Generating sediment screening values will allow comparison of their relative sensitivity to the wetland species.

The Gull-billed Tern (*Gelochelidon nilotica*) nests at only two locations in the western United States, the San Diego Salt Works and the Salton Sea. The birds forage widely throughout the south San Diego Bay region. There were between 32 and 37 pairs nesting at the Salt Works in 2003 (Unitt, 2004). Only seven colonies are known in western Mexico thus this colony is of some conservation significance. This bird is an occasional predator of other birds eggs and chicks including those of Killdeer, Black-necked stilts, Least Terns and Snowy plovers. It eats small fish but will also forage in terrestrial systems taking lizards and crickets. Lizards and fish are the primary prey it feeds its young in San Diego Bay. There are only 600 pair of this subspecies at all known colonies and the bird is on the USFWS list of Birds of Conservation Concern (2002,2008) which identifies species that, without additional conservation actions are likely to become endangered. This species meets my target species criteria 1,2,3,4, 6 and maybe 5. Toxic effects of DDT to this species are not well documented. One egg collected from San Diego Bay had a concentration of 2.9 ppm DDE ww. (Birds of North America species profile). I use the Charadriiformes allometric equation of Nagy, (2001) and the Pelican based TRV of 0.014, and the diet to fish ratio of 27. This results in a sediment screening concentration of **1 ug/Kg**, the lowest of any species. Using the Pelican LOAEL generates an associated sediment value of **5 ug/kg.** If this species is as sensitive as the Brown Pelican, which it probably isn't, then it would be energetically and trophically very vulnerable to DDT. The uncertainty lies in how sensitive it may be.

The Brown Pelican (*Pellicanus occidentalis*) is the poster child for DDT impacts to populations of waterbirds. The double crested Cormorant may be slightly more sensitive but the decades of work by Dan Anderson on his favorite species has resulted in excellent documentation of LOAEL and NOAEL which are the basis of many of the risk calculations in this document. It was listed as a federally protected endangered species in 1970 in part due to DDT and delisted in 2009. In San Diego Bay the bird has winter roosts in the south bay. The nearest breeding colony to San Diego is an island offshore of northern Baja Mexico. Pelicans are in the Bay during the winter (flocks as large as 47 individuals). A 100 year or 50 year storm that flushes DDT into the bay would mostly likely occur in winter when Pelicans are present. Using the NOAEL generated by dividing the Pelican LOAEL by 2 (USEPA region IX recommends a factor of 3) a sediment screening level of **3 ug/kg** is calculated. Using the recommended UF of 3 a concentration of **2 ug/kg** is calculated. These are pretty low numbers, even for a Pelican, but they could go lower with different sediment/diet ratios. However the purpose here is to compare with the other species and this calculation has a pretty high confidence level, with the sed/fish ratio the greatest uncertainty. Another source of uncertainty is a wide variation in reported body weights used in previous risk calculations. USEPA 1995 used 3.5 Kg for the Great Lakes Water Quality calculations of Pelican risk factors. I used smaller body weights between 2.1 and 2.8 kg.

The Black Skimmer (Ry*nychops niger*) is one cool bird and the Salt Works have become a major colony with hundreds of birds. There are other colonies in Southern California and banding shows birds do move between colonies, but individual birds are unlikely to move after nesting has commenced unless there is a disturbance issue. The diet of this bird is almost exclusively small fish in the 3-12 cm length (BNA species profile). A 2008 study by Zeeman et al. was instigated because of high rates of post-hatch chick mortality and dented eggshells were observed. Hatched chicks have a yolk sac in the abdomen that they are still using for nutrition after hatch. DDTs are concentrated in the fatty yolk of the egg.When there is high enough DDT there can be toxic effects to the chick. The Zeeman study documented DDTs in failed eggs and three species of fish. Using mean fish concentrations I was able to calculate the associated daily DDT intake value for the Salt Works colony as 0.046 mg (DDT)/Kg (BW) –day. This intake is associated with 19.6% eggshell thinning, denting and post chick mortality. Using our standard fish sediment ratio of 27 for comparison this pretty severe effect translates to a sediment concentration of only 12 ug/Kg. Using a UF of 3 to set a NOAEL using the pelican data a protective concentration in sediment is estimated as **4 ug/kg**.

Table 2. Summary Table of Sediment Screening Levels

Species	Site Specific Sediment Concentration ug/kg dw			

ainty factor of 3 to convert the pelican LOAEL to a NOAEL

Interpretation of Avian Wildlife Risk from the Summary Table

The table is sorted top to bottom by lowest to highest sediment concentrations. Bird numbers are the SLV concentrations. Sediment numbers, shaded brown, are selected worst case initial or final maximum post-flood concentrations in sediment at the 20 mm layer for both 50 and 100 year events. Pelagic forager numbers are shaded blue. Marsh forager numbers are shaded green. Theoretically if a bird has a no effect TRV below this they could experience adverse effects long term. However we also need to consider associated uncertainty in the estmates and key biological characteristics and apply some transparent judgement and logic for final decisions about risk to each species on a case by case basis. All of my equations, assumptions and results are available in an excel worksheet submitted with this report. Those who know a different view or have additional relevant data can update and change this risk assessment by modify either the numbers or equations and calculate new SLVs. There are more results than reported in the summary table. The summary table highlights my choices of TRVs to use for my own risk assessment evaluation and those of the ORERP risk assessment.

The Ridgeway Rail (LFFR) has NOAELS that bracket the long term max concentrations in 50 and 100 year floods. Given the 21% error inherent around these numbers just from the feeding rate calculation I feel Ridgway Rails are not at long term risk even though it could be expected that some individuals will spend their entire lifetime foraging in the post-flood contaminated area. My calculated SLVs for this species are quite a bit lower than those calculated in the risk assessment of the ORERP. It does look to me That the Rail may be at some toxic risk from DDT bioaccumulation in the ORERP during the short term settling event for maximum concentrations in the 50 and 100 year flood events. This will not be a lifetime exposure for any individuals but it will likely come before the breeding season. The Ridgeway Rail is the species of greatest management interest in the marsh restoration and the marsh species at greatest risk, but the risk looks very small in the long term. When compared to the benefits of the project to the Rail in terms of rails lost vs rails produced by the marsh, even if worst case events happen the risk to rails on balance looks to me to favor marsh creation.

The Black-necked Stilt is similarly protected. The 21% error applied downward to the Stilt SLV shifts it's range down to 6.5 ug/kg, essentially in the concentration ballpark of the worst long term maximum with upstream dilution. Stilts however, unlike rails, are migratory. Stilt chicks might spend their first year foraging in the the post-flood contaminated marsh area while adults would forage more broadly and could include some less contaminated habitat within their foraging. The Stilt looks like it may be at some toxic risk from DDT bioaccumulation during the short term concentrations in the 50 and 100 year flood events but this would be a one season, not lifetime exposure rate for some vulnerable individuals based on their foraging preferences.

The Snowy Egret SLVs that I calculated for this species are a bit higher than two of three in ORERP risk assessment. I believe the lowest SLVs in the ORERP's risk assessment are over protective and the highest ones are under protective. The highest NOAEL generated SLV for egrets from my assessment is slightly below the worst long term concentration and even with a boost doesn't get above the worst case, but really this still looks like very low risk to this species to me in the long term. One factor in favor of a low risk determination is the Snowy Egret forages widely and is less likely to forage only in the one contaminated marsh, unlike the Rail. The risk calculations assume 100% of foraging is done in the contaminated area. The Egret looks like it may be at some toxic risk from DDT bioaccumulation during the short term concentrations in the 50 and 100 year flood events but this would be a one season, not lifetime exposure rate for some vulnerable individuals based on their foraging preferences.

The Northern Harrier has a range of possible SLVs presented between 22 and 66 ug/kg. This brackets only the worst of the worst case scenarios during the initial post-flood time period. The variation reflects my uncertainty about which sediment/prey ratio to use. A bird that specialized on eating Ridgeway Rails would be assigned the 22 SLV. One that preffered small mammals would be assigned the 66 value. The Harrier's forage widely but a rail consuming specialist individual focused on the ORERP marsh in the one one intial event could have exposures above a NOAEL. The DDT risk post flood to Northern Hen Harrier overall looks low.

The Belding's Savannah Sparrow also has a range presented for SLVs but all of them are well above the worst case scenario. Being an obligate marsh forager and nester it will receive a lifetime exposure, however, the trophic position, and sensitivity work in its favor. I believe risk to this species from post flood DDT is very low as there is no overlap of the SLV range with sediment concentrations in any scenario.

The Gull-billed Tern had the lowest wildlife risk SLV at 1. I calculated for any species. Sensitivity and trophic level put it at risk. The bird however forages very widely in the region and also takes terrestrial prey. Despite a literature search (limited) I am unaware of any current adverse effects to reproduction or elevated DDT concentrations in this species. The species could be at great risk if DDT concentrations rise in south San Diego Bay estuarine fish post flood but that is not an affect of the ORERP. Given the bird is a species of concern, a DDT releasing flood could add to the "concern". The species, however, is not expected to spend much time foraging in the ORERP marsh. It has not been quantified for analysis to my knowledge but if the marsh were to redistribute some DDT in floods to marsh plains rather than subtidal zones this could provide a moderating influence on fish bioaccumulation of DDT from subtidal sediment driven bioaccumulation in fish. If so then ORERP restoration would benefit the Gull-billed Tern.

The Brown Pelican as most sensitive species and a fish eating species has, as expected, a very low SLV. The Pelican however is probably much protected by its wide foraging range and thus limited exposure within the areas affected by the long term maximum DDT. Individuals that forage more exclusively in south San Diego Bay would be at particular risk during the short term concentrations in the 50 and 100 year flood events but this would occur with or without the restoration. An analysis of how wetlands may dilute DDT in the pelagic foragers diet by redistributing some flushed DDT to the marshes is unquantified to my knowledge but may be a signficant mitigating factor for the ORERP. Risk is uncertain and dependent on the degree that some individuals would focus their winter foraging more exclusively in this area.

The Black Skimmer has an SLV of 4 below the long term worst case scenario. An open water forager it has a more limited foraging range than the other two pelagic foragers looked at in this assessment. The ORERP sub-tidal component of the restoration would perhaps pose a post flood risk to black Skimmers in many of the scenarios by adding DDT to local foraging resources but even without the restoration the bird faces a significant post flood risk of increased DDT exposure. In this case the risk would not be to the assumed exposure of a clean population but an incremental increase to an already contaminated species. Local foraging habits are apparently putting this species at risk. Reproduction is adversely impacted with eggshell thinning, dented eggs and post hatch chick mortalities as documented by Zeeman et al 2008. Whole body fish concentrations of DDT in that study were 32.9 ug/kg. Using that dietary concentration and the estimated ingestion rate of 38.12 g/d we can calculate a the harmful daily dose the bird might be receiving with the following equation:

$$\text{Harmful dose} = \frac{\text{harmfuld diet [DDT] x IR}}{\text{Body weight}}$$

We can then use a fish to sediment ratio used throughout this assessment of 27 to estimate an associated benthic sediment for comparison with our SLVs for other species. The result is a calculated harmful (toBlack Skimmers) sediment concentration of 21ug/kg. I think this tells us two things. The worst case scenario for post-flood DDT concentrations in initial settling periods are high enough to produce a harmful pulse of contamination. DDT contamination in the Otay River drainage already exists.

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LFFR body weight (g)	dw ingestion rate for Charadriiformes (g/day) from Nagy 2001	TRV mg (DDT)/Kg (BW) -day	Resulting Tissue level in diet TRV/(IR/BW) on a wet weight basis	ww ingestion rate	ingestion bw ratio	Dietary DDT/Sediment ratio (ww)	Resulting Sediment Screening concentration mg DDT/Kg sediment (ww)	Resulting Sediment Screening concentration ug DDT/Kg sediment (dw)	EFFECT
398	52.1	**0.009**	0.017	208.47	0.524	**3**	**0.006**	**10.0**	NOAEL with UF=3 I consider this a population level NOAEL
398	52.1	**0.014**	0.027	208.47	0.524	**3**	**0.009**	**15.6**	NOAEL with UF = 2
398	52.1	**0.227**	0.433	208.47	0.524	**3**	**0.144**	**253.4**	inappropriately insensitive NOAEL with 4 unfactored uncertainties, inter spe
227	33.8	**0.009**	0.015	135.37	0.596	**3**	**0.005**	**8.8**	NOAEL with UF=3
227	33.8	**0.014**	0.023	135.37	0.596	**3**	**0.008**	**13.7**	NOAEL with UF = 2
227	33.8	**0.227**	0.381	135.37	0.596	**3**	**0.127**	**222.6**	inappropriately insensitive NOAEL with 4 unfactored uncertainties, inter spe
227	33.8	**0.009**	0.015	135.37	0.596	**7.8**	**0.002**	**3.4**	NOAEL with UF=3 I consider this a sensitive individual level NOAEL
227	33.8	**0.014**	0.023	135.37	0.596	**7.8**	**0.003**	**5.3**	NOAEL
227	33.8	**0.027**	0.045	135.37	0.596	**7.8**	**0.006**	**10.2**	LOAEL from Pelican (Anderson)
227	33.8	**0.03**	0.050	135.37	0.596	**7.8**	**0.006**	**11.3**	NOAEL from Pelican (Fry)
398	52.1	**0.009**	0.017	208.47	0.524	**7.8**	**0.002**	**3.9**	NOAEL
398	52.1	**0.014**	0.027	208.47	0.524	**7.8**	**0.003**	**6.0**	NOAEL
398	52.1	**0.03**	0.057	208.47	0.524	**7.8**	**0.007**	**12.9**	true LOAEL from Pelican (Anderson)
398	52.1	**0.227**	0.433	208.47	0.524	**7.8**	**0.056**	**97.5**	inappropriately insensitive NOAEL with 4 unfactored uncertainties, inter spe

northern Harrier

body weight (g)	dw ingestion rate from Carnivorous birds (g/day) from Nagy 2001	TRV mg (DDT)/Kg (BW) -day	Resulting Tissue level in diet (mg total DDT/Kg diet) on a wet weight basis TRV/(IR/BW)	ww ingestion rate g/day @75% moisture	ingestion bw ratio	Dietary DDT/Sediment ratio (ww)	Resulting Sediment Screening concentration mg DDT/Kg sediment (ww)	Resulting Sediment Screening concentration ug DDT/Kg sediment (dw)	EFFECT
513	33.56	0.014	0.053	134.24	0.262	30	0.0018	3.1	
513	37.66	0.03	0.102	150.63	0.294	30	0.0034	6.0	
513	37.66	0.11	0.375	150.63	0.294	30	0.0125	21.9	
513	37.66	0.11	0.375	150.63	0.294	10	0.0375	65.7	No Effect in Kestrels
513	37.66	0.227	0.773	150.63	0.294	30	0.0258	45.2	
513	37.66	1.1	3.746	150.63	0.294	30	0.1249	219.1	LOAEL in Kestrels

Sparrow

body weight (g)	dw ingestion from Savannah sparrow (Passerculus sandwichensis) data (g/day) from Nagy 2001	TRV mg (DDT)/Kg (BW) -day	Acceptable Tissue level in diet (mg total DDT/Kg diet) on a wet weight basis TRV/(IR/BW)	ww ingestion rate g/day @75% moisture	ingestion bw ratio	Dietary DDT/Sediment ratio (ww)	Resulting Sediment Screening concentration mg DDT/Kg sediment (ww)	Resulting Sediment Screening concentration ug DDT/Kg sediment (dw)
18.7	5.74	0.009	0.007	22.96	1.228	1.5	0.0049	9
18.7	5.74	0.014	0.011	22.96	1.228	2	0.0057	10
18.7	5.74	0.11	0.090	22.96	1.228	1.5	0.0597	105
18.7	5.74	0.11	0.090	22.96	1.228	2	0.0448	79
18.7	5.74	0.11	0.090	22.96	1.228	3	0.0299	52
18.7	5.74	0.227	0.185	22.96	1.228	2	0.0924	162

Brown Pelican body weight (g)	dw ingestion rate based on Pelecaniformes equation (g/day) from Nagy 2001	TRV mg (DDT)/Kg (BW) -day	Acceptable Tissue level in diet (mg total DDT/Kg diet) on a wet weight basis TRV/(IR/BW)	ww ingestion rate g/day @75% moisture	ingestion bw ratio	Dietary DDT/Sediment ratio (ww)	Resulting Sediment Screening concentration mg DDT/Kg sediment (ww)	Resulting Sediment Screening concentration ug DDT/Kg sediment (dw)	EFFECT
2824	229.92	0.009	0.028	919.70	0.326	27	0.001	2	NOAEL with UF=3 I consider this a population level NOAEL
2824	229.92	0.014	0.043	919.70	0.326	27	0.002	3	NOAEL with UF = 2
2824	229.92	0.027	0.083	919.70	0.326	27	0.003	5	true LOAEL from Pelican (Anderson)
2147	182.39	0.009	0.026	729.56	0.340	27	0.001	2	NOAEL with UF=3
2147	182.39	0.014	0.041	729.56	0.340	27	0.002	3	NOAEL with UF = 2
2147	182.39	0.027	0.079	729.56	0.340	27	0.003	5	true LOAEL from Pelican (Anderson)
3500	275.64	0.027	0.086	1102.56	0.315	27	0.003	6	true LOAEL from Pelican (Anderson)

Snowy Egret body weight (g)	dw ingestion rate for Charadriiformes (g/day) from Nagy 2001	TRV mg (DDT)/Kg (BW) -day	Acceptable Tissue level in diet (mg total DDT/Kg diet) on a wet weight basis TRV/(IR/BW)	ww ingestion rate g/day @75% moisture	IR bw ratio	Dietary DDT/Sediment ratio (ww)	Resulting Sediment Screening concentration mg DDT/Kg sediment (ww)	Resulting Sediment Screening concentration ug DDT/Kg sediment (dw)	Effect
370	49.3	0.009	0.017	197.10	0.533	27	0.001	1.10	NOAEL with UF=3 I consider this a population level NOAEL
370	49.3	0.014	0.026	197.10	0.533	27	0.001	1.71	NOAEL derived with UF=2
370	49.3	0.027	0.051	197.10	0.533	27	0.002	3.29	LOAEL from Pelican (Anderson)
370	49.3	0.03	0.056	197.10	0.533	27	0.002	3.66	NOAEL from Pelican (Fry)
370	49.3	0.227	0.426	197.10	0.533	27	0.016	27.69	inappropriately insensitive NOAEL with 2 unfactored uncertainties, for use with snowy egret
370	49.3	0.009	0.017	197.10	0.533	21	0.001	1.41	NOAEL
370	49.3	0.014	0.026	197.10	0.533	21	0.001	2.20	NOAEL derived with UF=2
370	49.3	0.027	0.051	197.10	0.533	21	0.002	4.23	LOAEL from Pelican (Anderson)
370	49.3	0.03	0.056	197.10	0.533	21	0.003	4.70	NOAEL from Pelican (Fry)
370	49.3	0.227	0.426	197.10	0.533	21	0.020	35.60	inappropriately insensitive NOAEL with 2 unfactored uncertainties, for use with snowy egret

Black Skimmer

body weight (g)	dw ingestion rate for Charadriiformes (g/day) from Nagy 2001	TRV mg (DDT)/Kg (BW) -day	Acceptable Tissue level in diet (mg total DDT/Kg diet) on a wet weight basis TRV/(IR/BW)	ww ingestion rate g/day @75% moisture	ingestion bw ratio	Fish to Sediment ratio	Resulting Sediment Screening concentration mg DDT/Kg sediment (ww)	Resulting Sediment Screening concentration ug DDT/Kg sediment (dw)	
265	38.12	0.009	0.063	152.5	0.144	27	0.002	4.07	NOAEL with UF=3 I consider this a population level NOAEL
265	38.12	0.014	0.097	152.5	0.144	27	0.004	6.32	NOAEL with UF = 2
265	38.12	0.027	0.188	152.5	0.144	27	0.007	12.20	predicted LOAEL from Pelicans (Anderson)
265	38.12	0.046	0.320	152.5	0.144	27	0.012	20.8	19.6% eggshell thinning, denting. Post hatch chick mortality. Zeeman et al., 2008

Gull Billed Tern

body weight (g)	dw ingestion rate for Charadriiformes (g/day) from Nagy 2001	TRV mg (DDT)/Kg (BW) -day	Acceptable Tissue level in diet (mg total DDT/Kg diet) on a wet weight basis TRV/(IR/BW)	ww ingestion rate g/day @75% moisture	ingestion bw ratio	Dietary DDT/Sediment ratio (ww)	Resulting Sediment Screening concentration mg DDT/Kg sediment (ww)	Resulting Sediment Screening concentration ug DDT/Kg sediment (dw)	EFFECT
187	29.16	0.009	0.014	116.62	0.624	21	0.001	1.2	NOAEL with UF=3 I consider this a population level NOAEL
187	29.16	0.014	0.022	116.62	0.624	21	0.001	1.9	NOAEL with UF = 2
187	29.16	0.027	0.043	116.62	0.624	21	0.002	3.6	predicted LOAEL from Pelicans (Anderson)
187	29.16	0.227	0.364	116.62	0.624	21	0.017	30.4	inappropriately insensitive NOAEL with 2 unfactored uncertainties, for use with snowy egret
187	29.16	0.03	0.048	116.62	0.624	21	0.002	4.0	NOAEL from Pelican (Fry)
187	29.16	0.046	0.074	116.62	0.624	21	0.004	6	adverse effect diet concentration for skimmers

0,044

Black-Necked Stilt

Black-Necked Stilt body weight (g)	dw ingestion rate for Charadriiformes (g/day) from Nagy 2001	TRV mg (DDT)/Kg (BW) -day	Acceptable Tissue level in diet TRV/(IR/BW) on a wet weight basis	ww ingestion rate	ingestion bw ratio	Dietary DDT/Sediment ratio (ww)	Resulting Sediment Screening concentration mg DDT/Kg sediment (ww)	Resulting Sediment Screening concentration ug DDT/Kg sediment (dw)	EFFECT
169	27.0	0.009	0.014	107.9	0.64	3	0.005	8.24	
169	27.0	0.014	0.022	107.9	0.64	3	0.007	12.82	
169	27.0	0.027	0.042	107.9	0.64	3	0.014	24.73	threshold for adverse effect in pelican
169	27.0	0.03	0.047	107.9	0.64	3	0.016	27.48	
169	27.0	0.227	0.356	107.9	0.64	3	0.119	207.94	

Mugu logic

Effect	Egg [DDE]mg/kg	Egg/crab	Crab [DDE]mg/kg	Crab/Sed	Sed [DDE]mg/kg	Sed [DDE] ug/kg
NOAEL	1.5	24	0.06	2	0.03	31.3
LOAEL (8% DDE related shell thinning Mugu Lagoon)	2.13	24	0.09	2	0.04	44.4
Harm predicted to individuals	3.5	24	0.15	2	0.07	72.9
Harm predicted to local population	12	24	0.50	2	0.25	250.0

table 11 calcs

Post flood mud [DDT] ug/kg	initial flood thickness mm	initial flood density kg/l	final flood thickness	final flood density kg/l	Thickness of Biologically relevant layer mm	predicted initial/final [DDT] relevant layer ug/kg by table 12	predicted [DDT] initial relevant layer ug/kg by volumetric ratio	predicted [DDT] initial relevant layer ug/kg by incorporating density	predicted [DDT] of final relevant layer ug/kg by incorporating density	Assumed background DDT concentration of receiving sediment	Assumed background density of marsh muds kg/l	thickness of initial non-flood layer	thickness of final non-flood layer	initial volumetric ratio of receiving layer	final volumetric ratio of receiving layer
70.2	3.3	0.2	0.5	1.2	20	13,3.5	12.9	3	0.6	1.6	0.149	16.7	19.5	0.835	0.975
70.2	8	0.2	1.4	1.2	20	29,6.4	29.0	6	1.1	1.6	0.149	12			
790	0.2	0.2	0.02	1.2	20	7.1, 2.4-4.0	13.8	2	1.0	6	0.149	19.8			
790	0.34	0.2	0.06	1.2	20	15	14.9	3	0.7	1.45	0.149	19.66			
		0.2		1.2	20		1.5			1.45	0.045	20			
		0.2		1.2	20		1.5			1.45	0.045	20			
		0.2		1.2	20		1.5			1.45	0.045	20			
		0.2		1.2	20		1.5			1.45	0.045	20			
		0.2		1.2	20		0.0			1.45	0.045				
		0.2		1.2	20		0.0			1.45					

Species	Site Specific Sediment Concentration ug/kg dw	Predicted Effect	Source	Reviewer opinion
Gull-billed Tern	1	NOAEL calculated with Pelican data and EPA R IX BTAG methodology guidance	Schwarzbach Review	protective
Snowy Egret	2	lowest NOAEL in ORERP risk assessment for any bird	ORERP wildlife risk assessment	May be overly protective
Brown Pelican	3	NOAEL calculated with Pelican data and EPA R IX BTAG methodology guidance	Schwarzbach Review	protective
Black Skimmer	4	NOAEL calculated with Pelican data and EPA R IX BTAG methodology guidance	Schwarzbach Review	protective
Snowy Egret	5	highest NOAEL (based on Pelican regression)	Schwarzbach Review	appropriately protective for the Egret population
LFFR	6	Highest Skinny Rail NOAEL that doesn't ovrerlap a LOAEL cacluation.		my recommended # for LFFR protection of individual rails. Upper boundary with 21% error in the diet calculatoin is 7.26
100 year flood	6.4	Highest 20 mm concentration calculated for long term	ORERP Sedimentation model with dilution	seems like the most likely scenario for 100 year flood because it includes upstream sources as dilution. This could get even smaller with watershed scaling adjustment that I think needs to be made.
50 year flood	7.1	Highest 20 mm concentration calculated for long term	ORERP Sedimentation model, (no dilution)	
100 year flood	7.9	Highest 20 mm concentration calculated for long term	ORERP Sedimentation model no dilution	
Black-Necked Stilt	8	NOAEL calculated with Pelican data and EPA R IX BTAG methodology guidance	Schwarzbach Review	very protective
LFFR	11	Highest Skinny Rail NOAEL	Schwarzbach Review	my recommended # for protection of local rail population
Black Skimmer	21	Toxic effects to chicks, dented eggshells, 19% eggshell thinning (Zeeman et al 2008)	Schwarzbach Review	Calculated from mean fish concentrations found by Zeeman et al. 2008
LFFR	15	lowest Wildlife SLV in ORERP risk assessment	ORERP wildlife risk assessment	too high due to flawed assumption on diet and use of maximum body weight of rail
50 year flood	26	Highest 20 mm concentration calculated for initial event	ORERP Sedimentation model	Short term hazard to color coded birds below this number

Snowy Egret	28	Highest NOAEL from ORERP for Egre	ORERP wildlife risk assessment	Under protective, and flawed TRV
100 year flood	29	Highest 20 mm concentration calculated for initial event	ORERP Sedimentation model with dilution	Short term hazard to color coded birds below this number
Northern Harrier	22-66	Highest NOAEL for this bird	Schwarzbach Review	protective
Beldling's Sparrow	52-105	Highest NOAEL for this bird	Schwarzbach Review	protective
LFFR	240	Highest NOAEL from ORERP for any bird.	ORERP wildlife risk assessment	inappropriately high. no uncertainty factors for a TRV derived from a study in a non-sensitive species for an irrelelevant toxic endpoint in a laboratory setting

Green shading indicates my recommended SLV for a marsh bird.

my recommended sediment SLV for a pelagic forager

Sediment. Dark brown is for long term and light brown is intial event.

commends using an uncertainty factor of 3 to convert the pelican LOAEL to a NOAEL