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Dredging Impacts on Food-Web Bioaccumulation of DDTs in San Francisco Bay, CA

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EXECUTIVE SUMMARY

This study represents a ‘first-order’, thought experiment with the goal of estimating the potential incremental impact of maintenance dredging activities on contaminant levels in resident biota. Existing field data on contaminant levels in Bay water and sediment were combined with information regarding routine maintenance dredging operations to estimate the incremental change in sediment and water contaminant concentrations as a function of distance from the dredge site.

Existing field data from a number of studies that have measured trace organic contaminant concentrations in Bay sediment and water (e.g., RMP, 2006; U.S. EPA, 2004; CalTrans, 2000; Battelle, 1992a,b,c; EVS, 1998; Battelle, 1994) were combined with field observations of dredging plume characteristics (e.g., MEC, Inc., 2004) to estimate the potential impact of dredging on food web bioaccumulation. Data were compiled into four different spatial regions (Figure 1A; dredge site, near-field, mid-field, and far-field) to address the question of ‘incremental’ impacts (i.e, How do the potential impacts change as a function of distance from the dredge site?).

A steady-state non-equilibrium food web model originally developed to assess uptake of non-polar organic contaminants in food webs (Gobas, 1993) was used to evaluate potential impacts on biaccumulation in resident biota. The model simulates organic contaminant transfer from sediment and water through a multi-species food web by combining contaminant kinematics in biota and food web dynamics (Gobas, 1993).

Six DDT compounds were modeled as part of this study (o,p-DDD, o,p-DDE, o,p-DDT, p,p-DDD, p,p-DDE, p,p-DDT). Eight different scenarios were modeled, representing the dredge site, near-, mid-, and far-fields both before and during dredging activities. The impacts of maintenance dredging on food web bioaccumulation were assumed to be driven by changes in water column exposure to DDTs.

Results indicate that the potential impact of dredging on contaminant levels in resident biota is 100 times greater at the dredge site than in the near-field. Impacts in the mid- and far-fields were negligable, with predicted percent increases in biota contaminant levels less than 1%.

I. INTRODUCTION

There is concern that incremental pollutant loads to the San Francisco Bay ecosystem from dredging and in-Bay disposal of dredged sediment could potentially impact aquatic biota. However, to date limited work has been conducted to put numeric bounds on just how much mass of a given pollutant could be potentially incorporated into the food web as a result of dredged sediment exposure. The few impairment assessments conducted for the San Francisco Bay (e.g., Hg, PCBs, and dioxins) have primarily focused on accumulation of pollutants in sport fish tissues, and several of the most impacted fish were identified as benthic foragers that are known to frequent harbor and marina environments. Thus, there remains a need to develop a method for measuring potential pollutant transfer to benthic-foraging fish species from dredging activities.

Dredgers move sediment around the Bay, from places where it obstructs business to places where it does not. Many of the priority contaminants in the Bay do not readily dissolve in water and tend to become associated with sediments. So for two reasons - navigation and pollution control - dredging and dredged material disposal are regulated by virtually every agency that has any connection with water and waterways. Sediments are regularly tested for contaminant levels, results of which govern when and how the material is dredged and where it is disposed. Often, federal permits require formal consultation under the Endangered Species Act, prompting dredgers to adjust their activities to allay fears that dredging will cause toxicity in endangered species or injure them through exacerbated bioaccumulation of contaminants in the Bay's food web. Still, no independent estimates exist on the potential for increased contaminant bioaccumulation in the Bay's food web resulting from dredging operations.

This study represents a 'first-order', thought experiment with the goal of estimating the potential incremental impact of maintenance dredging activities on contaminant levels in resident biota. Existing field data on contaminant levels in Bay water and sediment were combined with information regarding routine maintenance dredging operations to estimate the incremental change in sediment and water contaminant concentrations as a function of distance from the dredge site. The resulting estimates subsequently served as inputs to a sophisticated food web model to estimate the potential for increased contaminant bioaccumulation in resident biota resulting from dredging operations.

II. METHODS

A. COMPILATION OF EXISTING FIELD DATA

This study combined existing field data from a number of studies that have measured trace organic contaminant concentrations in Bay sediment and water (e.g., RMP, 2006; U.S. EPA, 2004;

CalTrans, 2000; Battelle, 1992a,b,c; EVS, 1998; Battelle, 1994) with field observations of dredging plume characteristics (e.g., MEC, Inc., 2004) to estimate the potential impact of dredging on food web bioaccumulation. Data were compiled into four different spatial regions (Figure 1A; dredge site, near-field, mid-field, and far-field) to address the question of 'incremental' impacts (i.e, How do the potential impacts change as a function of distance from the dredge site?). The methods of compositing/averaging the various data sources into the four fields are described in the following sections. After review of available data, DDTs were found to have the best overlap in all data sources. DDTs (including o,p-DDD, o,p-DDE, o,p-DDT, p,p-DDD, p,p-DDE, and p,p-DDT) were therefore selected as the contaminant evaluated in this case study. Additional contaminants can be evaluated in future studies.

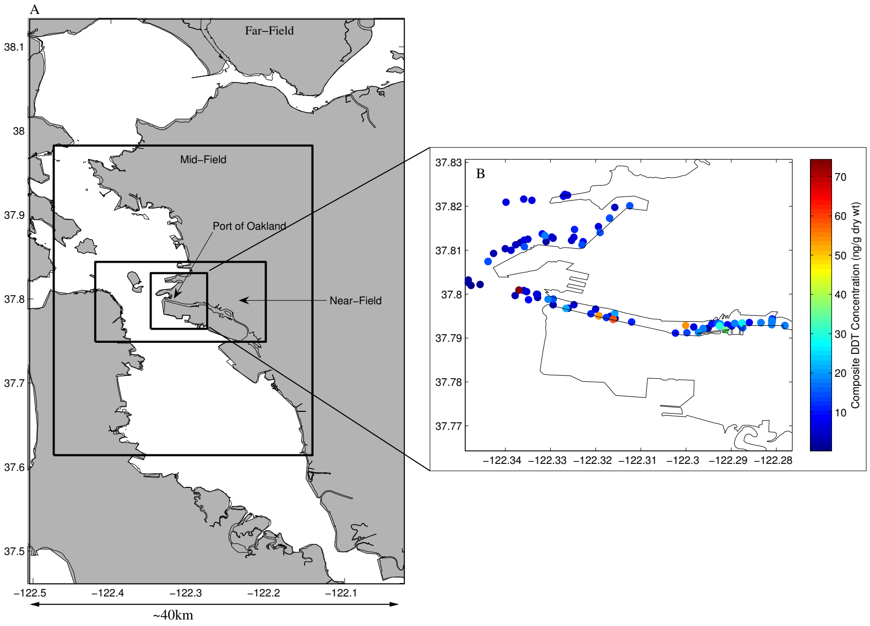

Figure 1: A) Map of San Francisco Bay indicating fields used in this study (Areas: dredge site $\approx 0.02km^2$, near-field $\approx 203km^2$, mid-field $\approx 1200km^2$, far-field $\approx 3200km^2$. B) Zoomed-in map of the Port of Oakland with locations of sediment cores sampled by various studies. Colors indicate the composite sediment DDT concentrations at each location.

1. Sediment

Port of Oakland

A number of studies have measured contaminant levels in sediments at the Port of Oakland (e.g., CalTrans, 2000; Battelle, 1992a,b,c; EVS, 1998; Battelle, 1994). These studies range from evaluation of maintenance dredging material to assessments of Port widening and deepening projects. Figure 1B indicates the locations of the sediment cores sampled by these studies. The concentrations reported in these studies represent the composite (average) contaminant concentration over the entire depth of the sediment core. The vertical distribution of DDT concentrations in sediment cores is illustrated in Figure 2. The data suggest that there is no significant trend of DDT concentration with depth. Thus, for the purpose of this study, the geometric mean DDT concentration (8.603 ng/g dry wt) is used to represent the concentration of DDTs in sediment at the Port of Oakland. This approach is reasonable as a first approximation, and fits well with sediment concentrations observed in Figure 3.

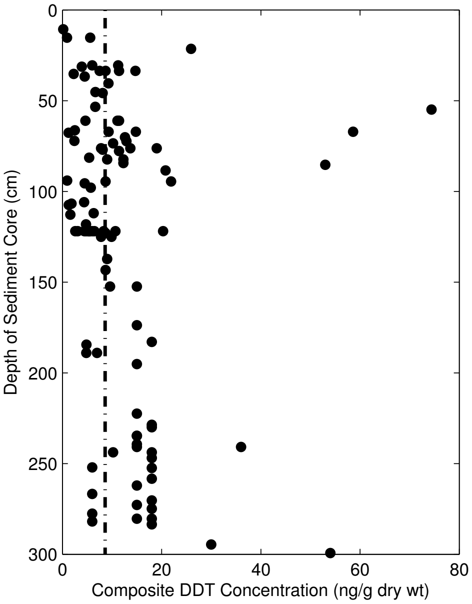

Figure 2: Vertical profile of DDTs in sediment cores from the Port of Oakland. The dashed line indicates the geometric mean of all samples (8.603 ng/g dry wt).

A number of assumptions were made when determining concentrations of DDTs to use at the Port of Oakland (also referred to as the dredge site). The first of which was mentioned above (i.e., using the geometric mean of all samples). The second assumption deals with sediment concentrations before and during dredging activities. This study is meant to evaluate the effects of maintenance dredging. Maintenance material is characterized by unconsolidated fine-grained sediments - typically young Bay muds - that are transported into the Port between routine dredging episodes (Port of Oakland, 2005). It is expected that maintenance material therefore has contaminant concentrations that are very similar to those found in nearby Central Bay. Thus, for the purpose of this study, it is assumed that there is no net change in sediment concentrations of DDTs during dredging operations. The incremental effect of dredging is therefore driven by changes in water column contaminant levels at the dredge site (see Section 3.).

The final assumption surrounding data at the Port addresses the fact that data for individual DDT compounds were not available, only data for the sum of DDTs. Since the food web model described in Section B. models individual DDT compounds, it was necessary to estimate the levels of these individual compounds at the Port. To fill this data gap, the ratios of individual DDT compounds to the sum of DDTs observed in EMAP data were used to estimate the levels of individual DDT compounds in maintenance material. The concentration of individual DDT compounds in sediment at the dredge site are shown in Figure 4.

The Bay

The two largest programs that monitor Bay surface sediments are the Regional Monitoring Program (RMP) and the Environmental Monitoring and Assessment Program (EMAP; U.S. EPA (2004)). The RMP monitors annually for trace contaminants with results reported in the Annual Monitoring Results publication. EMAP monitors the Bay intermittently, with the most recent large scale sampling in 2000 and 2001. This study used surface sediment DDT concentrations reported by EMAP as ambient levels for all fields. The choice to use EMAP data in this study was based on the high spatial coverage of the sampling protocol and the fact that the data were already processed and investigated as part of the San Francisco Bay synthesis paper of legacy pesticides submitted to *Environmental Research* in Connor *et al.* (2005).

The distribution of DDTs in Bay surface sediments from EMAP is illustrated in Figure 3. The contours seen in Figure 3 were generated by interpolating the field observations onto a regular grid by the method of krigging - a method of geospatial interpolation that relies on the spatial correlation structure of known data when estimating the value at unsampled locations (Journel and Huijbregts, 1981). The concentration of each individual DDT compound in each field was estimated by taking the geometric mean of all samples within the given field (Figure 4). It was assumed that dredging of maintenance material and the subsequent disposal at in-Bay disposal sites causes a zero net change on a field basis. That is, dredging simply moves material from one location within a field (near-, mid-, or far-field) to another location within the field. The geometric mean of all samples within that field is therefore unchanged. The resulting geometric mean concentrations then served as input to the food web model (Section B.).

2. Water

The distribution of total DDTs in the water column from RMP field observations is shown in Figure 5. The contours in Figure 5 were generated by interpolating the field observations onto a regular grid by the method of kriging. The concentration of each individual DDT compound at

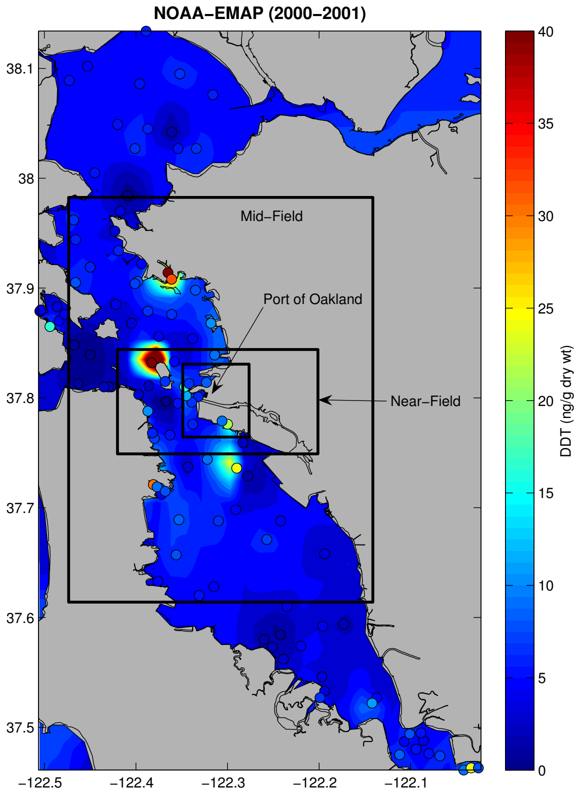

Figure 3: Distribution of DDTs in Bay surface sediments from NOAA-EMAP (U.S. EPA, 2004). Contours were generated by geospatial interpolation. Original sampling locations are included as scatter points.

each field, $\bar{C}_{tf}$, was estimated by taking the geometric mean of all points within the given field as follows

$$\log(\bar{C}_{t_f}) = \left(\sum_{i=1}^{N} \frac{\log(C_{t_i})}{N} + \frac{\log(C_{plume}) A_{plume}}{A_f} \right) \frac{A_f}{N(1 + A_{plume}/A_f)} \tag{1}$$

where N is the number of points (i.e., interpolated grid cells) within the given field, C_{ti} is the contaminant concentration at the i^{th} point within the field, C_{plume} is the contaminant concentration in the dredge plume, A_{plume} is the area of the plume, and A_f is the surface area of water in the given field. A_{plume} and C_{plume} are equal to zero for the **before dredging** scenarios. Estimation of A_{plume} and C_{plume} for the **during dredging** scenarios is described in Section 3. Water column concentrations of DDTs in the various fields **before dredging** are presented in Figure 6. The resulting concentration of individual DDT compounds in water for the various fields **before dredging** is

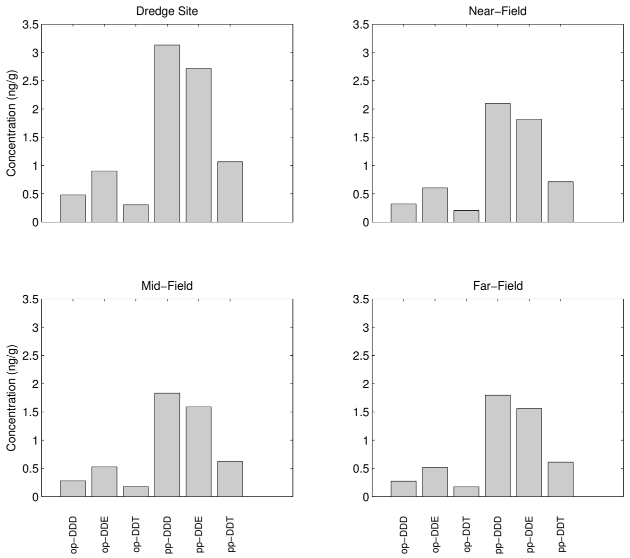

Figure 4: Geometric mean concentration of DDTs in surface sediment in the various fields used in this study. These concentrations served as input used as input to the food web model for scenarios with and without dredging activities.

indicated in Figure 6. Water concentrations of DDTs at the dredge site were not available, so the near-field concentrations were used to represent the ambient concentration of DDTs in water at the dredge site ($C_{ambient}$ in Equation 5).

3. Dredge Plume Characteristics

In order to estimate the incremental effect of dredging on contaminant levels in biota, it is necessary to know how much contaminant mass becomes suspended in the water column during dredging activities. Field data regarding contaminant concentrations in dredging plumes were not available at the time of writing. It was therefore necessary to estimate the concentration of contaminants in a dredging plume by combining field studies of the physical characteristics of a dredging plume with field data on the chemical characteristics of dredged material. MEC, Inc. (2004) conducted a detailed field study of a dredging plume in Oakland Harbor. Results showed a measurable plume area, A_{plume}, of approximately 5 acres (0.02 km^2) at the surface during dredging, with a somewhat wider plume at depth. Total suspended solids concentrations in the plume generally ranged from 75 to 100 mg/L, with occasional concentrations as high as 150 mg/L. Ambient suspended sediment concentrations were approximately 25 mg/L. Observations of plume duration (i.e., how long does the plume last?) were less successful, largely because the plume becomes indistinguishable from background within a tidal cycle (approx. 13 hours; (Port of Oakland, 2005)).

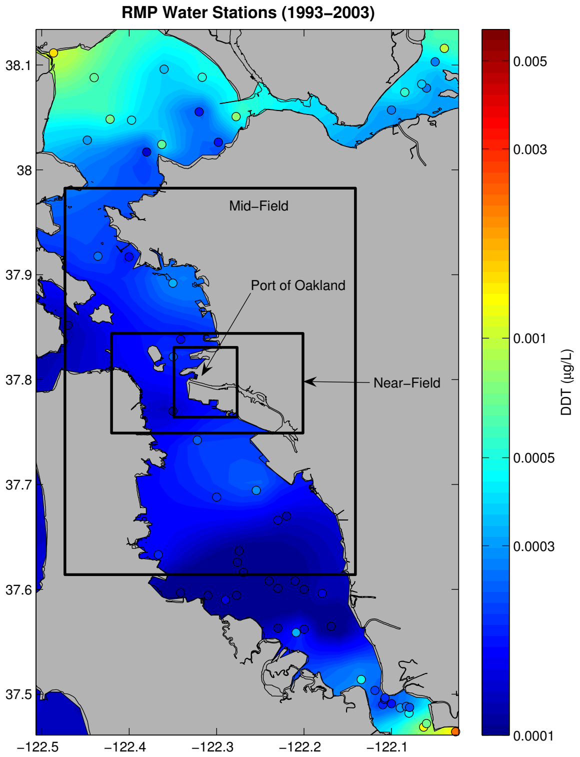

Figure 5: Distribution of total DDTs (particulate and dissolved) in RMP water samples. Contours were generated by geospatial interpolation. Original sampling locations are included as scatter points. Concentrations are colored on a logarithmic scale.

Given the results of MEC, Inc. (2004) and the mean concentration of DDTs in Port sediment, it is possible to estimate the change in total (particulate and dissolved) DDT concentration in the water column resulting from dredging activities. First, the change in particulate, ΔC_p, and dissolved, ΔC_d, concentrations relative to ambient levels can be estimated as follows

$$\Delta C_p = C_s \times \Delta SSC \tag{2}$$

$$\Delta C_d = C_s / (K_{ow} \times F_{oc}) \tag{3}$$

where C_s is the geometric mean concentration of DDTs in sediments at the Port determined in Section 1. (8.603 ng/g dry wt), ΔSSC is the change in suspended sediment concentration in the dredge plume relative to ambient levels (50 mg/L used for this study; O'Connor (1991) observed SSC of 50-80mg/L; MEC, Inc. (2004) reported SSC primarily in the range 75-100mg/L with an ambient SSC of approximately 25mg/L), K_{ow} is the octanol-water partitioning coefficient (see

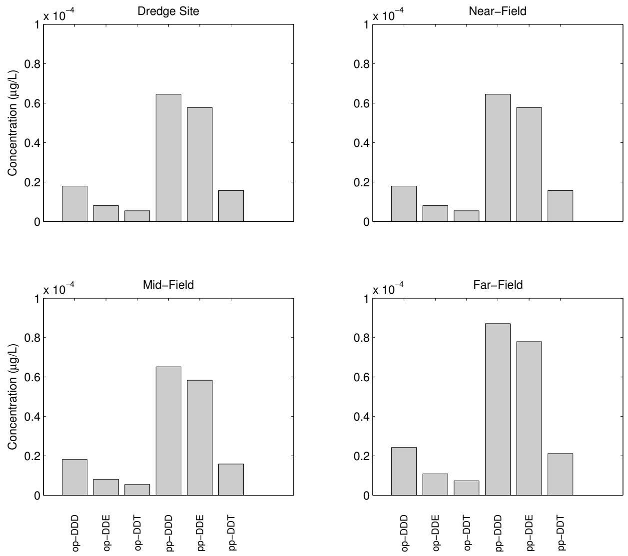

Figure 6: Geometric mean concentration of DDTs in water in the various fields used in this study. These concentrations served as input to the food web model for the **before dredging** scenarios.

Table 1), and F_{oc} is the fraction of organic carbon in sediment (0.01; Oros and Ross (2004)). The change in total concentration of DDTs in the plume relative to ambient, ΔC_t, is then

$$\Delta C_t = \Delta C_p + \Delta C_d. \tag{4}$$

The total concentration of DDT in the dredging plume is then estimated by adding ΔC_t to the ambient total concentration of DDT (i.e., before dredging or outside of the plume) as follows

$$C_{plume} = C_{ambient} + \Delta C_t. \tag{5}$$

Water concentrations of DDTs at the dredge site were not available, so the near-field concentrations before dredging were used to represent the ambient concentration of DDTs in water at the dredge site ($C_{ambient}$ in Equation 5). C_{plume} was used directly as input to the food web model for the dredge site scenario during dredging. The near-, mid-, and far-field water column concentrations during dredging were determined by Equation 1. The resulting concentrations used as input to the food web model for the **during dredging** scenarios are shown in Figure 7.

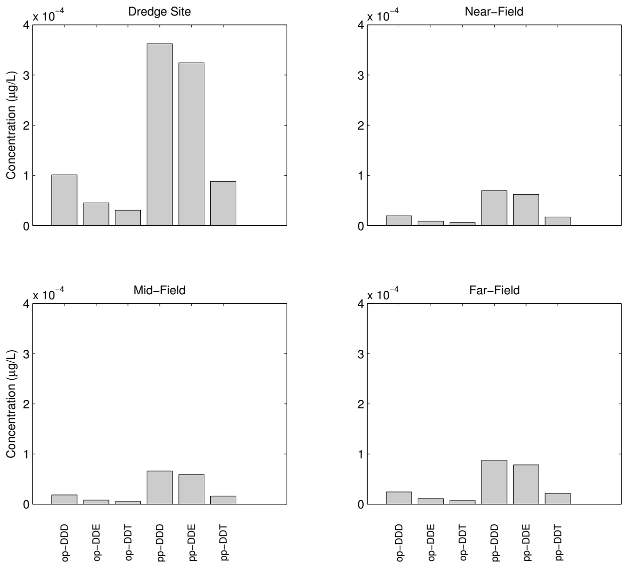

Figure 7: Geometric mean concentration of DDTs in water in the various fields used in this study. These concentrations served as input to the food web model for the **during dredging** scenarios.

B. BAY FOOD WEB MODEL

The mechanistic food web model used in this study was parameterized by Gobas and Arnot (2005) to calculate uptake of PCBs in selected fish and wildlife in San Francisco Bay as part of the PCB TMDL (Total Maximum Daily Load) study. Development included a detailed analysis of the Bay food web and existing field data to parameterize and validate the model for PCB uptake. The model was further developed for other bioaccumulative organic contaminants on the Section 303(d) list of TMDL priority contaminants (e.g., DDTs, chlordanes, dioxins, and PBDEs) by Greenfield *et al.* (2006). The food web model is a steady-state non-equilibrium model originally developed to assess uptake of non-polar organic contaminants in food webs (Gobas, 1993). This model simulates organic contaminant transfer from sediment and water through a multi-species food web by combining contaminant kinematics in biota and food web dynamics (Gobas, 1993). The original model has been used extensively in research and for regulatory applications (e.g., the TrophicTrace software program developed by the US Army Crops of Engineers for evaluation of dredge material disposal (Bridges and von Stakelberg, 2003)). The mechanistic model independently determines inputs of a particular chemical into an organism from water and sediment, making it a useful tool in evaluating the potential impacts of dredging, which affects water column and sediment concentration levels differently. Food web model parameters and equations are included as Appendix A.

1. Configuration for this Study

Six DDT compounds were modeled as part of this study (o,p-DDD, o,p-DDE, o,p-DDT, p,p-DDD, p,p-DDE, p,p-DDT). Eight different scenarios were modeled, representing the dredge site, near-, mid-, and far-fields both before and during dredging activities. As stated earlier, the impacts of maintenance dredging on food web bioaccumulation are assumed to be driven by changes in water column exposure to DDTs. To model these eight scenarios, the model required for each scenario inputs regarding concentrations of DDTs in water and sediment, octanol-water partitioning coefficients (K_{ow}; see Table 1), and suspended sediment concentrations. Estimation of concentrations of DDTs in water and sediment and of suspended sediment concentrations was described in Section A. The resulting estimates, which served as input to the food web model, are shown in Figures 4, 6, and 7.

Table 1: Octanol-water partitioning coefficients for individual DDT compounds.

DDT Compound	Kow
o,p-DDD	1.995×10^5
o,p-DDE	4.217×10^5
o,p-DDT	5.012×10^5
p,p-DDD	2.138×10^6
p,p-DDE	8.511×10^6
p,p-DDT	2.455×10^6

Source: (Mackay *et al.*, 1997; Shen and Wania, 2005; Leatherbarrow *et al.*, 2006)

2. Model Validation

Predicted concentrations of DDTs in biota were compared to field observations to assess model performance (Figure 8). Results indicate that the food web model is generally able to predict field observations within observed standard deviations. Furthermore, the model captures the general pattern of bioaccumulation through the Bay food web (i.e., contaminant levels in biota increase with an increase in trophic position). Such an agreement between predicted and observed concentrations lends credibility to the model framework and suggests the model is suitable for the case study of dredging presented here.

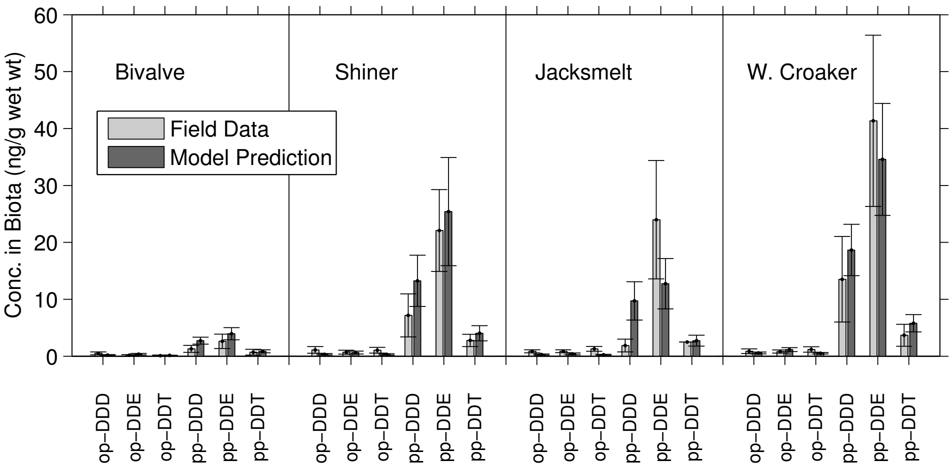

Figure 8: Comparison of food web model to field data for individual DDT compounds. Error bars indicate the standard deviation of field observations and the uncertainty of model results as estimated by a Monte Carlo analysis of model variables.

III. RESULTS AND DISCUSSION

A. CONTAMINANT LEVELS IN RESIDENT BIOTA

Four species (Table 2) at varying trophic levels in the San Francisco Bay food web (Figure A.1) were selected for evaluation of the potential impacts of dredging activity on food web bioaccumulation. Predicted concentrations of DDTs in biota for the various scenarios investigated are presented in the following sections.

Table 2: Species selected to evaluate potential impacts of dredging.

Scientific Name	Common Name	Feeding Mode	Movement
Ampelisca sp.	Amphipod	Epibenthic deposit feeder	Sedentary
Macoma nasuta	Bent-nosed Clam	Epibenthic sediment feeder	Sedentary
Cymatogaster aggregata	Shiner Surfperch	Epibenthic sediment feeder	Resident
Genyonemus lineatus	White Croaker	Epibenthic sediment feeder	Resident

1. Dredge Site

Prior to dredging, the predicted mean DDT concentration in biota at the dredge site was less than 2.5 ng/g for invertebrates (i.e., amphipod and bivalve), and 10-15 ng/g for the trophically higher fish species (Figure 9). Upon the initiation of dredging activity, the mean concentration increased to 6-10 ng/g for invertebrates and 50 ng/g for fish (Figure 10). DDTs in biota at the dredge site were therefore predicted to be approximately 200% greater than the concentration in the food web prior to dredging (Figure 11). Model results for the small deposit-feeding amphipod (*Ampelisca sp.*) indicated a similar average increase as the higher consumers (e.g., white croaker). Therefore, despite the large differences in DDT concentrations in these two biota themselves, the average percent increase was predicted to be similar. These comparable percent increases in DDT at the dredge site were likely due to a model assumption that the route of exposure for these species was similar (approx. 80% via water; Figure 12) and the fact that water column concentrations were so high at the dredge site.

2. Near-, Mid-, and Far-Fields

The average percent increase in DDTs differed between species in the near-field and beyond ($\geq 203km^2$). The bivalve (*Macoma nasuta*) was predicted to have less of an increase in DDTs, whereas shiner surfperch had the highest (Figure 11). The bivalve was least sensitive to changes in water-laden DDTs with increase in field area as the proportion of contaminant exposure was heavily weighted towards sediment (approx. 40% via sediment, Figure 12). Similarly, shiner surfperch was modeled with the highest proportion of water exposure (approx. 80%), and thus exhibited the highest percent change (recall that the model assumed the only route of increased exposure during dredging activity was through the water column).

The predicted impacts on food web bioaccumulation indicated an incremental reduction in the percent change of DDTs in biota from the dredge site to the far-field (Figure 11). In fact, the biota concentrations in the near-field were relatively similar to those predicted prior to dredging (Figures 9 and 10). The average percent increase due to dredging was only 2-4%, depending on species. This was two orders of magnitude less than the predicted increase around the dredge site itself. Furthermore, the average percent difference was minimal ($< 1\%$) in the mid-field and beyond ($\geq 1200km^2$), indicating that the contribution of water-laden DDTs to the food web from dredging activity was negligable at this scale.

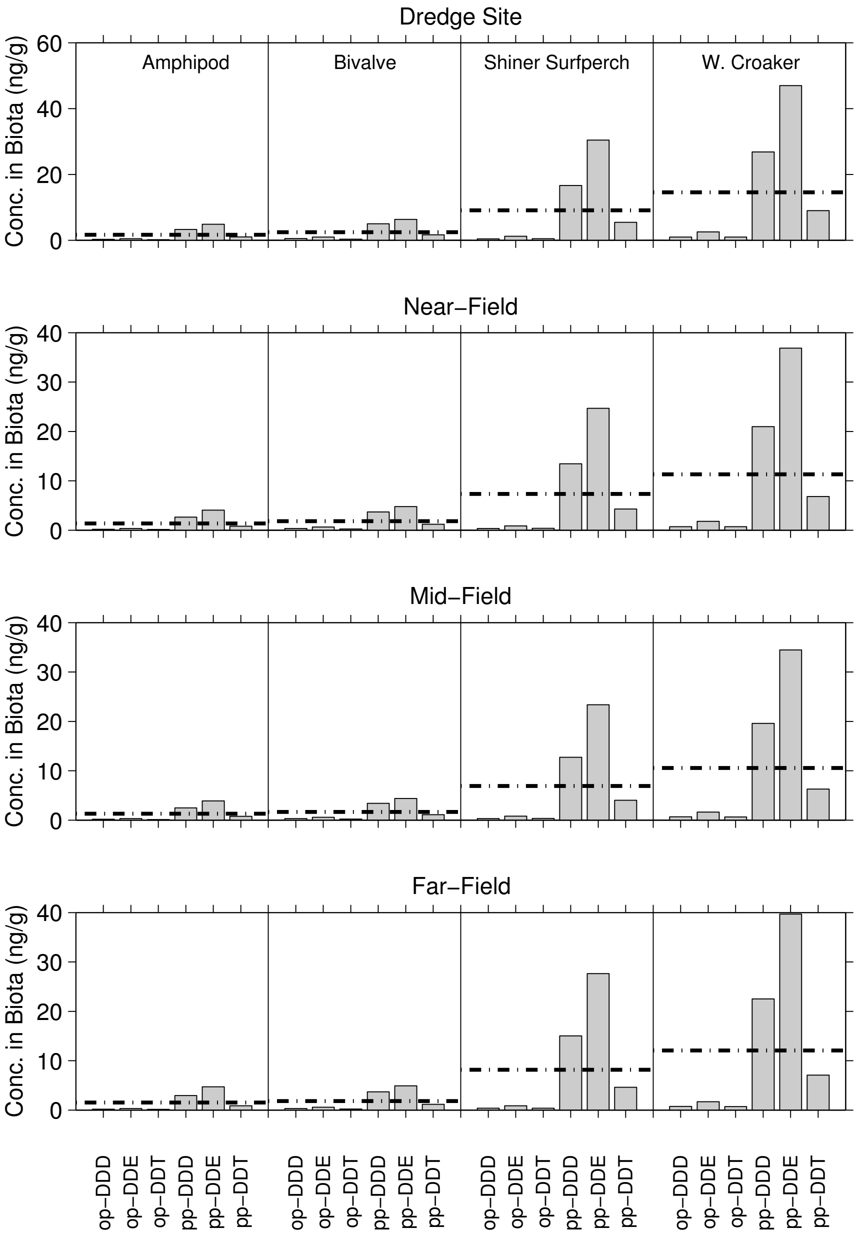

Figure 9: Predicted concentrations of DDTs in select biota prior to dredging activities. The dashed horizontal lines indicate the mean for each field and each biota.

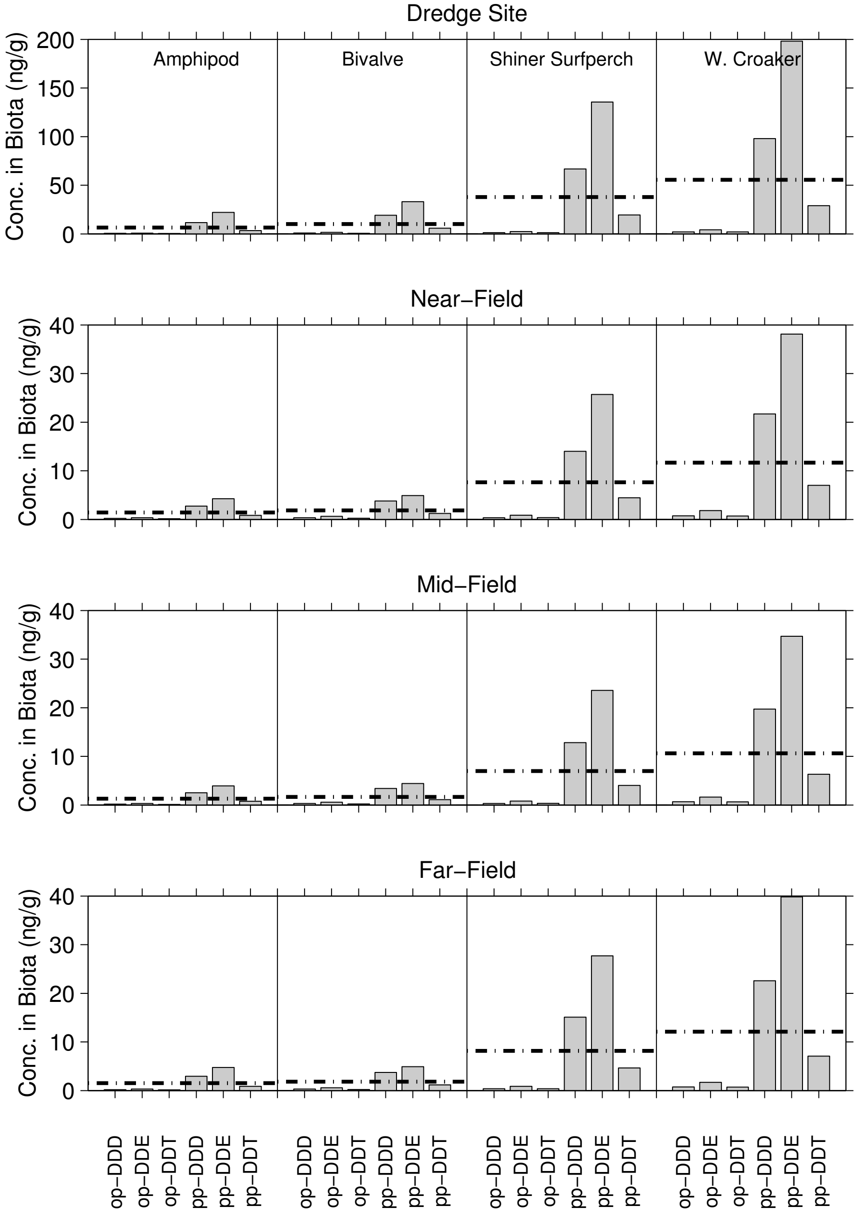

Figure 10: Predicted concentrations of DDTs in select biota during dredging activities. The dashed horizontal lines indicate the mean for each field and each biota.

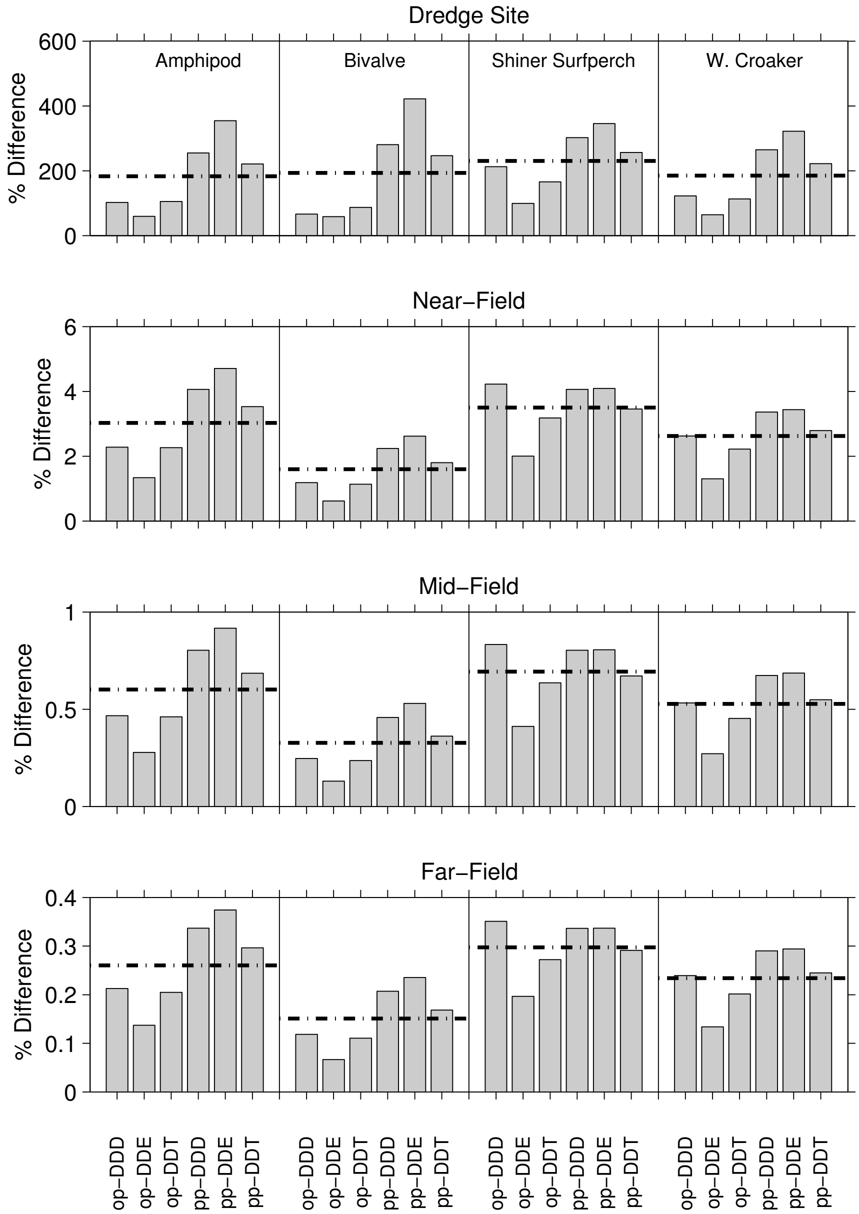

Figure 11: Predicted percent change in select biota resulting from dredging activities. The dashed horizontal lines indicate the mean percent change for each field and each biota.

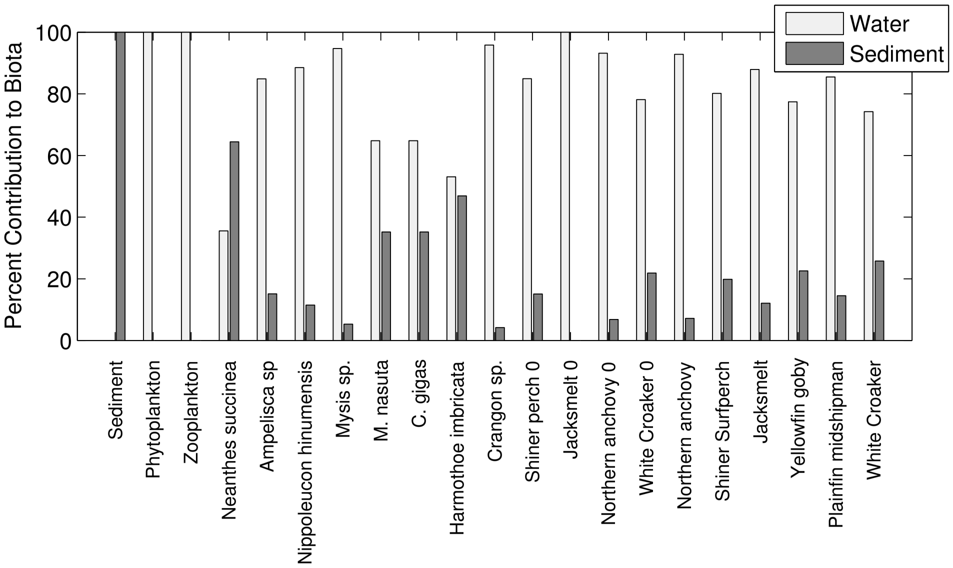

Figure 12: Relative contribution of water and sediment to food web bioaccumulation. Results are from a case study of p,p-DDE in San Francisco Bay presented in Greenfield *et al.* (2006).

B. SHORTCOMINGS AND POSSIBLE FUTURE STUDIES

The approach employed in this investigation has a number of known shortcomings. First, the steady-state food web model used is unable to estimate the impacts of episodic inputs of contaminants to the food web. Therefore, the **during dredging** scenarios represent 'worst case' scenarios in which dredging activities are assumed to be continuous. In reality, dredging operations must comply with strict regulations that limit their work windows to a few months out of the year. Furthermore, field observations of dredging plumes have noted that suspended sediment concentrations return to background levels within hours to days after the cessation of dredging. The food web model simply can not parameterize such a situation. Results of the food web model can not be directly scaled by time of exposure, making it difficult to estimate how much the during dredging predictions over estimate potential changes in bioaccumulation. However, even with this likely over-estimation, the model predicted a negligible increase (on the order of 1%) in bioaccumulation in the near-field.

Another shortcoming of the model as applied to this case study is the fact that it does not include the behavioral characteristic that fish will often avoid excessively turbid waters. O'Connor (1991) noted that 'fishes will avoid regions of excessively high turbidity, or will alter their distributions to conform to some preferred or tolerated range of suspended particulate matter.' Such behavior would minimize the potential increase in food web bioaccumulation caused by resuspension of contaminated dredge material in the immediate vicinity of the dredge site. Inclusion of this behavior would potentially reduce the predicted percent increase in concentrations of DDTs in biota at the dredge site. Effects on the near-field scale and beyond would likely be unchanged, as the area of the dredge site (or more appropriately the dredge plume) is small compared to the total area of the respective fields.

Due to budget constraints and data availability, this study focused solely on DDTs. Future studies could evaluate potential impacts from additional contaminants. The authors acknowledge that more data must be available to refine the estimates presented here and to investigate additional contaminants. Is it plausible that results would be significantly different for different contaminants. Further investigation is warranted.

Finally, some uncertainty surrounds the estimation of water and sediment DDT concentrations used as input to the food web model, particularly the estimation of changes in water column contaminant concentrations resulting from resuspension of dredged material. The methods used here are based on equilibrium partitioning of contaminants between water and sediment. This assumption could be corroborated (or refuted) by field observations of contaminant concentrations in dredging plumes.

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V. ACKNOWLEDGMENTS

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VI. APPENDICES

A. SUMMARY OF SAN FRANCISCO BAY FOOD WEB MODEL

A summary of parameters and equations for the mechanistic food web model used in this report is included below. A schematic of the Bay food web impacted by organic contaminants is included in Figure A.1. All model equations and assumptions are from Gobas and Arnot (2005).

Table A.1: Abiotic Input Parameters.

Variable	Description
C_{ox}	Dissolved oxygen concentration (mg O^2/L)
T	Mean water temperature
K_{ow}	Octanol-water partitioning coefficient
$K_{ow}TS$	K_{ow} corrected for temperature and salinity
C_{water}	Contaminant concentration in water
C_{sed}	Contaminant concentration in sediment
$salinity$	water salinity (PSU)
MCS	Molar concentration of seawater at 35ppt (0.5 mol/L)
SPC	Setschenow proportionality constant (0.0018 L/cm^3)
$ocsed$	Organic carbon fraction in sediment
vss	Concentration of suspended solids (kg/L)
$xpoc$	POC concentration in H2O (kg/L)
$xdoc$	DOC concentration in H2O (kg/L)
$dpoc$	Disequilibrium factor for POC partitioning (1.0)
$ddoc$	Disequilibrium factor for DOC partitioning (1.0)
$alphapoc$	Proportionality constant describing phase partitioning of POC and DOC (0.35)
$alphadoc$	Proportionality constant describing phase partitioning of POC and DOC (0.08)

Table A.2: Biotic Input Parameters.

Variable	Description
A, B	Constants for phytoplankton aqueous uptake rate($A = 6.0e - 5$, $B = 5.5$)
Wb	Body weight
$assimEff(i)$	Assimilation efficiency for lipid(i = 1) nlom (i = 2) and water (i = 3)
EdA	Constant A in dietary uptake efficiency equation ($8.5e - 8$)
EdB	Constant B in dietary uptake efficiency equation (2.0)
$lipid$	Tissue lipid content
$nlom$	Tissue non-lipid organic matter content
$beta$	Lipid-equivalency conversion factor for bioconcentration factor for non-lipid organic matter
$betap$	Plant lipid-equivalency conversion factor for bioconcentration factor (for nlom)
wc	Tissue water content
mo	Proportion of respiration or transpiration due to overlying water column
mp	Proportion of respiration or transpiration due to porewater
kM	Metabolic rate constant for contaminant in biota (set to zero in this study)
$scav$	Filter feeding particle scavenging efficiency
$preyprop$	Proportion of diet due to individual prey (calculated from prey proportion matrix)
vld	Proportion of diet that is lipid (calculated from prey proportion matrix)
vnd	Proportion of diet that is non-lipid organic matter (calculated from prey proportion matrix)
vwd	Proportion of diet that is water (calculated from prey proportion matrix)
$kGconst$	Growth rate equation constant
$scav$	Scavenging efficiency of particles absorbed from water by filter feeders (100%)

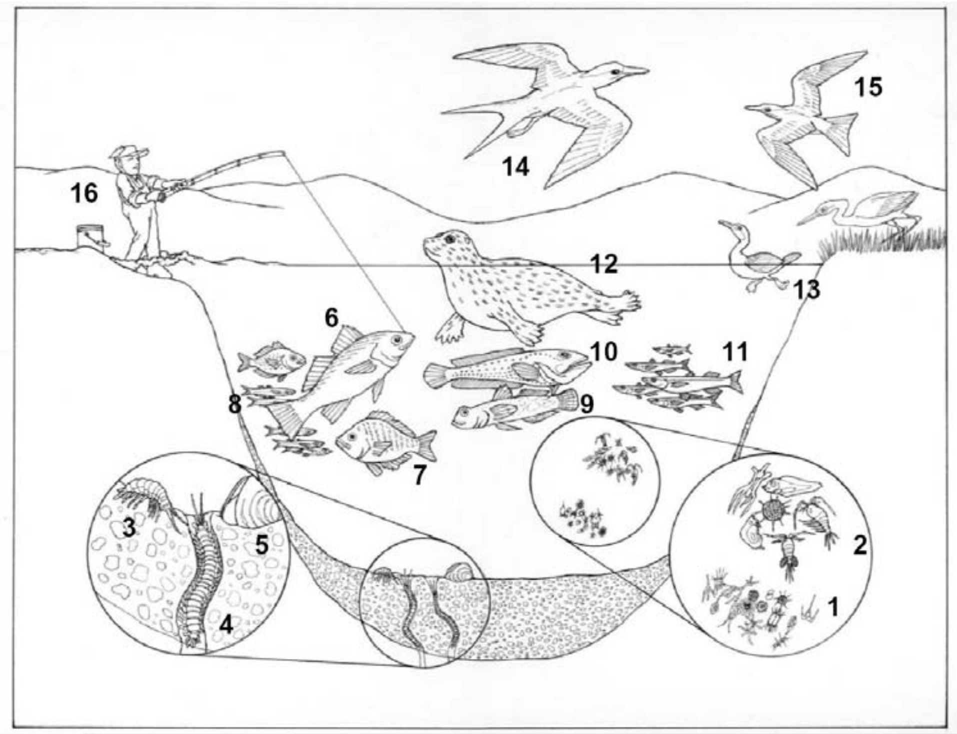

Figure A.1: Schematic of the Bay food web impacted by organic contaminants. Contaminants enter the food web primarily through accumulation by phytoplankton (1) at the base of the food web. Concentrations then increase with each step up the food web. Phytoplankton are consumed by small animals including zooplankton (2) and invertebrates such as amphipods (3), worms (4), or clams (5). Invertebrates in sediment also accumulate contaminants directly from sediment through ingestion of particles and from contact with sediment porewater. Fish consume zooplankton and invertebrates. People (16) and wildlife species consume fish such as yellow fin goby (9), plainfin midshipmen (10), anchovy (11), white croaker (6), shiner surfperch (7), and jacksmelt (8). Sensitive wildlife species include harbor seals (12), cormorants (13), Forster's terns (14), and least terns (15).

Table A.3: Model Variables.

Variable	Description
$k1$	Aqueous uptake rate constant
$k2$	Elimination rate constant
kG	Growth rate
Gv	Gill ventilation rate
Gd	Feeding rate
Gf	Fecal egestion rate
vlg	Lipid fraction of gut
vng	nlom fraction of gut
vwg	Water fraction of gut
kgb	Gut-biota partition coefficient
ke	Fecal egestion rate constant (1/d)
kd	Dietary uptake rate constant
Ew	Contaminant-specific gill chemical uptake efficiency
Ed	Contaminant-specific dietary chemical transfer efficiency (also called gut uptake efficiency)
phi	Freely dissolved contaminant fraction in overlying water column
cpw	Contaminant concentration in porewater
kbw	Biota-water partition coefficient (i.e., bioconcentration factor)
C_{biota}	Contaminant concentration in biota
C_{prey}	Contaminant concentration in prey diet

Model Calculations

$$E_w = 1/(1.85 + 1.55/K_{ow}TS)$$

$$E_d = 1/(EdA \times K_{ow} + EdB)$$

$$phi = 1/(1 + xpoc \times dpoc \times alphapoc \times K_{ow}TS + xdoc \times ddoc \times alphadoc \times K_{ow}TS)$$

$$cpw = C_{sed}/(ocsed \times 0.35 \times K_{ow}TS)$$

Calculations for Phytoplankton and Benthic Algae

$$k1 = 1/(A + B/K_{ow}TS)$$

$$kbw = (lipid \times K_{ow}TS + nlom \times betap \times K_{ow}TS + wc)$$

$$k2 = k1/kbw$$

$$C_{biota} = k1 \times (mo \times phi \times C_{water} + mp \times cpw)/(k2 + kG + kM)$$

Calculations for Invertebrates and Fishes

$$Gv = (1400 \times Wb^{0.65})/C_{ox}$$

$$k1 = Ew \times Gv/Wb$$

$$kbw = lipid \times K_{ow}TS + nlom \times beta \times K_{ow}TS + wc$$

$$k2 = k1/kbw$$

$Gd = 0.022 \times Wb^{0.85} \times \exp(0.06 \times T)$ (for non-filter feeders)

$Gd = Gv \times vss \times scav$ (for filter feeders)

$$kd = Ed \times Gd/Wb$$

$$kG = kGconst \times Wb^{-0.2}$$

$$Gf = Gd \times ((1 - assimEff(1)) \times vld + (1 - assimEff(2)) \times vnd + (1 - assimEff(3)) \times vwd)$$

$$vlg = (1 - assimEff(1)) \times vld/((1 - assimEff(1)) \times vld + ... \\ (1 - assimEff(2)) \times vnd + (1 - assimEff(3)) \times vwd)$$

$$vng = (1 - assimEff(2)) \times vnd/((1 - assimEff(1)) \times vld + ... \\ (1 - assimEff(2)) \times vnd + (1 - assimEff(3)) \times vwd)$$

$$vwg = (1 - assimEff(3)) \times vwd/((1 - assimEff(1)) \times vld + ... \\ (1 - assimEff(2)) \times vnd + (1 - assimEff(3)) \times vwd)$$

$$kgb = ((vlg \times K_{ow} + vng \times beta \times K_{ow} + vwg)/... \\ (lipid \times K_{ow} + nlom \times beta \times K_{ow} + wc))$$

$$ke = Gf \times Ed \times kgb/Wb$$

$$C_{prey} = preyprop \times C_{biota}$$

$$C_{biota} = (k1 \times (mo \times phi \times C_{water} + mp \times cpw) + kd \times C_{prey})/(k2 + ke + kG + kM)$$

B. Reviewer Comments

This section compiles reviewer comments and presents author responses where appropriate. Unfortunately, we do not have the resources to repeat/re-think the analysis to address all of reviewer comments. We would like to thank the reviewers for their time and constructive criticisms. Compilation of reviewer comments will aid future studies.

1. A. Jahn, April 2006

I like the approach with one exception, described below. The choice of DDT as the contaminant of interest seems well justified, although I think it would be useful to speculate how similar the results might be for a contaminant of greater current concern such as PCBs.

It seems to me that the problematic assumption of continuous dredging could be dealt with to some extent by applying a dilution factor, i.e., if dredging occurs for a single episode, or for a few months per year, then you could simply divide the number of dredging days by 365 and multiply this by the sediment concentration (though not so you'd reduce the latter below ambient).

There may be ways to scale the results of the food-web model to account for time of exposure. Unfortunately we do not have the resources to perform this analysis at this time.

I had a little trouble with units. In figure 7, it seems to me the units should be ng/L, and they should be in the tens. I guess this is what you mean by the "x 10 -4" at the top of each graph, but I always find such things ambiguous. I never know whether they mean the values have been so-multiplied, or if they should be so-multiplied. I think it is better to label them right to start with.

One more thing about Fig. 7 (&6): It is not clear to me why these have shapes that differ from that of Fig. 4, where op-DDE has a higher peak than op-DDD. I think you should explain.

Although the model and the report are very much what I had in mind when I first suggested the exercise, the apparent definition of "incremental" (p. iv, 2) is not what I meant by that term. I meant simply "over and above what would occur in the absence of dredging."

My quibble with the model is its treatment of "the dredge site." Unlike the other "fields", this tiny area (2 ha) moves. Not only does the dredge site move, but the fish and plankton (and dredge) will move through it, as well as (as you mention) around it, in the case of fish. These facts raise no great difficulties in the application at the larger scales, but they make it impossible for me to visualize the exposure scenario at the dredge site, particularly at higher trophic levels. Clearly, the disturbance near the dredge must have a calculable manifestation, but it seems more realistic to assign this to a large number of briefly exposed fish than to a small, captive population. Rather than express this increment as a percentage increase over a small area (except for the caption of Fig. 1, I do not see mention of the size of this area in the report), you might consider expressing it as a mass, and tabulating it along with the mass contributions at other scales. Can you run the Gobas model, with allowances for the proportion of time, as suggested above (my paragraph 2), and then back-calculate to a mass of contaminant? As difficult as "200% increase at the dredge site" is to conceptualize, it is also the one number in this that is most likely to be misused.

Re: Fig. 1. I don’t actually see the dredge site.

pp 2-3: I appreciate that good data are hard to find, but you may have mismatch here, in that the upper 3 m of Inner Harbor is just about exactly what was to be removed by the harbor deepening project. You might be talking about stuff that is no longer there; the Battelle refs, in particular, are quite old in this context. I don’t memorize these concentration data, but it seems worth checking with Beth Christian or someone at the Port to get an update on present concentrations in ”maintenance material” and confirm you are in the ball park.

nit picks:

p. 3, lines 12-13. Not a sentence.

p. 4, line 22: kriging misspelled

p. 4, lines 25-28: This is confusing. Unless the dredging occurs in the same field as the disposal, there must be a transfer between fields. No?

p12, line 13: might be a good place to remind the reader of the size of the modeled dredge site.

Figures 9-11 use the collective noun ’biota’ as a singular. Doesn’t work for me I suggest ’organism.’

2. US Army Engineer Research and Development Center

Comments by P.R. Schroeder, J.A. Steevens, T.S. Bridges.

ERDC has been requested to review the subject draft report by the San Francisco District under funding by the DOTS Assistance Program.

1. The report fails to present and justify all of its assumptions. It is unclear what are the differences in the physical, chemical and exposure processes between the near, mid. and far fields and how and why these areas were established. It’s not even clear what the physical dimensions are for these zones. The analysis and conceptual model for exposure are overly simplistic and inappropriate for the source. The method of estimating incrementally increased exposure is not technically sound.

The analysis is purposefully over-simplistic. This study represents a first-order look at the potential for bioaccumulation from dredging activities. We feel the estimates presented here represent an order-of-magnitude approximation.

2. The dredging and site assumptions need to be clearly specified. The dredging assumptions should include the sediment volumes removed, areas of dredging, lengths of dredging reaches, the durations of the dredging projects, the production rates, types of dredges, the use of overflow, grain size distributions of the sediments, the TOC of the sediments, DOC of the sediments, and contaminant levels in the sediments. The dredge plume data should include temporal and spatial

descriptions of TSS, TOC, and DOC data and, if possible, total and dissolved contaminant concentrations. The site conditions should include the average ambient TSS, TOC, DOC, total and dissolved contaminant concentrations as well as physical data such as velocity and bathymetry data. It is not clear from the presentation of the data to what degree these sources of information were used in the modeling. If disposal occurs in the Bay, similar data should be known for the disposal plumes and disposal sites.

Such details can be included in future reports.

3. The data given in Figures 1, 2 and 3 should also be expressed as normalized to organic carbon because bioavailability is related to normalized contaminant concentration. It is assumed that dredging does not alter the sediment contaminant concentration in the Bay sediments; therefore, contaminant release is only in the dredging zone due to the increased suspended sediment concentrations in the dredge plume. Then, these sediment solids partition their contaminants with the water. The dissolved and particulate contaminant releases are then dispersed and available for bioaccumulation. This conceptual model is simplistic, and inaccurate for predicting bioavailability. A more complete model would include partitioning with the whole water column including the background TSS, TOC and DOC. The resuspended sediment particles would tend to settle quickly due to their aggregated state in the sediment bed. Most of the solids would settle in the first hour and nearly all would settle in four hours. Particles remaining after the first few hours would aggregate with the entrained ambient solids and settle within a day. These settled solids will be very similar to the existing bed. The dissolved contaminants will partition with the entrained TSS, TOC and DOC and quickly become unavailable when they are incorporated into the sediment bed. Additionally, much of the contaminants associated with particulates, DOC and TOC are not readily bioavailable. However, as long as total contaminant concentration in the water column is elevated, the bioavailable dissolved contaminant concentration will also be elevated, but with background TSS concentrations as high as 25 mg/L the contaminated materials will settle quickly and will be incorporated in the sediment bed (Other data reported by SFEI show ambient TSS concentrations for San Francisco Bay ranging to concentrations between 25 and 100 mg/L for substantial periods of time). Elevated bioavailability may last as little as a day. Site data on particle settling and resuspension as well as near-field water column data are needed to verify the conceptual model. The combination of the expected short period of time that sediment particles will be suspended as a result of dredging activity along wit the presence of other binding phases in the water column (POC, DOC, background sediment) will result in substantially over-estimating the freely dissolved concentration of DDTs based on equilibrium partitioning as described as described in the report.

Time-varying food-web and water column models are better suited for such a detailed analysis. Unfortunately we do not have such tools, nor do we have the time or money to develop them for this project.

4. Equation 1 does not appear to be correct for computing the geometric mean of a field. More

information on its assumptions and development are needed. The contaminant concentration of the plume is not constant except perhaps in the near-field.

Equation 1 is correct for a spatially-weighted geometric mean.

5. The field data for dredge plumes are appropriate only for hopper dredges with overflow; the area is too large for other types of dredges and the concentration is too high for dredging without overflow. Plumes are not usually distinguishable beyond 1 to 4 hours except in waters with low ambient turbidity and low velocities (under 0.5 fps). The variability that would be expected among dredging projects limits extrapolating this analysis to include other dredging projects where plume characteristics could be substantially different.

This level of detail is outside the scope of this thought experiment. We would consider additional future studies if funding were available.

6. The method of calculating the change in dissolved contaminant concentration given in Equation 3 is not accurate. Calculation of the dissolved concentration must consider the overall contaminant concentration (both background and resuspended sediment) in the water column as well as the overall organic carbon content (dissolved and particulate associated, background and resuspended sediment). In addition, the partitioning relationship should use the Koc of the contaminants and not Kow, else the Kow should be adjusted to approximate the Koc. Ideally, the Koc should be determined for the actual sediment. As an example, Karickoff estimated the Koc to be equal to 0.617 Kow for PCBs.

Equation 3 assumes background concentration is negligable compared to concentration during dredging. Agreed that Koc is preferable to Kow.

7. The Foc of 0.01 is low for fine-grained maintenance sediments. Typical values are closer to 0.025.

Foc=0.01 is the Bay-wide average reported by Oros and Ross (2004) and was thought to be reasonable for first-order calculations.

8. ΔC_p as calculated in Equation 2 is equal to ΔC_t. Adding ΔC_d as computed in Equation 3 with ΔC_p to compute ΔC_t is double counting ΔC_d because ΔC_d comes from the computed ΔC_p.

Agreed. Merits re-analysis.

9. Due to the transient nature of the exposure, a dynamic, time-varying food web model would be a more appropriate choice for the type of analysis being attempted. Given the small size of typical dredging-induced sediment plumes, individual fish would be expected to remain within the plume for relatively short periods, much less than would be required to achieve steady-state concentrations of DDT. In addition, if the actual duration of dredging were considered a more accurate prediction could be made. If hopper dredges were used, incorporation of cycle time could be used to more accurately predict daily dose for the steady-state model.

A non-steady-state food web model would certainly be an improvement. Unfortunately we do not have such a tool at our disposal. The decision to use the steady-state model is consistent with this being a first-order calculation. This shortcoming is clearly stated in the report.

10. One significant problem with this study is, as the authors point out on page 17, is that exposure to DDT(s) in the water column are instantaneous and indefinite. While the authors indicate that the plume becomes indistinguishable from background within a tidal cycle, 13 hours the assumption is made that this plume does not decrease. As indicated above, information regarding the dredging efforts (i.e., dredge type, duration, etc) should be included to better characterize the temporal aspects of the exposure. In our lab, we have recently calculated the time to steady state for DDT compounds in Macoma nasuta (mean 102 days; range 86 to 122 days). Given the time that it would take for steady state to be achieved, the duration of the dredging project, and assumptions of indefinite exposures it is likely the model grossly overestimates the actual concentrations of DDT compounds in the SF Bay food web resulting from dredging activities.

We acknowledge the potential for gross over-estimation of bioaccumulation. Future work could address this issue.

11. Other issues that were identified by the authors include the avoidance of the area by higher trophic level organisms. If the authors make the assumption that suspension is continuous and indefinite, considerations should be made regarding the fraction of time that fish will reside/feed in this area. Verification of model outputs should be evaluated through a comparison to existing fish tissue data. This was mentioned briefly in the text; however, the information should be presented and discussed.

A more formal discussion is merited. A non-steady-state model would help the analysis as well.

12. Conclusions. There are a number of weaknesses in the draft report which severely limit its ability to be used in reaching conclusions about the role of dredging in DDT fate and transport within San Francisco Bay.

We agree that there are large uncertainties in these estimations. However, we tried, where possible, to error on the side that would result in increased bioaccumulation. Even so, bioaccumulation from dredging activities appears to be negligable (less than 1% in the mid- and far-fields).