FINAL
ECOLOGICAL RISK ASSESSMENT
FOR
THE PALOS VERDES SHELF

November 2003

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Acronyms and Abbreviations

μg/kg	micrograms per kilogram
μg/L	micrograms per liter
μm	micrometer
AF	accumulation factor
AI	Anacapa Island
ANP	Año Nuevo Point, California
ATC	advisory tissue concentration
BAZ	biologically active zone
BC	British Columbia
BERA	Baseline Ecological Risk Assessment
BEQ	benthic effect quotient
BPTCP	Bay Protection and Toxic Cleanup Program
BRI	benthic response index
BSAF	biota sediment accumulation factor
BTAG	Biological Technical Assistance Group
BW	body weight
C	Celsius
CA	California
CC	coastal California
CCC	criterion continuous concentration
CDFG	California Department of Fish and Game
CE	California-listed, endangered
CFP	California fully protected
CI	Channel Islands
cm/sec	centimeters per second
CMC	criterion maximum concentration

COC	chemical of concern
COEC	chemical of ecological concern
COPEC	chemical of potential ecological concern
CP	California-protected
CSC	California species of special concern
CSM	conceptual site model
CT	California-listed, threatened
DDD	dichloro-2,2-bis (p-chlorophenyl)ethane
DDE	dichlorodiphenyldichloroethylene
DDMU	1, 1′-(chloroethenylidene)bis(4-chlorobenzene)
DDNU	1, 1′-bis(p-chlorophenyl)ethylene
DDT	dichloro-diphenyl-trichloroethane
DTSC	[California] Department of Toxic Substances Control
DW	dry weight
EA	effluent-affected
EC	effect concentration
ECn	effect concentration (n = percent affected)
EC20	effect concentration at which 20 percent of the test population shows effect
EE/CA	engineering evaluation/cost analysis
EPC	exposure-point concentration
EqP	equilibrium partitioning
ERA	Ecological Risk Assessment
ER—L	effects range—low
ER—M	effects range—median
FE	federally listed, endangered
FT	federally listed, threatened
GERG	[Texas A&M University] Geochemical and Environmental Research Group
CIS	Geographic Information System
GOC	Gulf of California

HQ	hazard quotient
JWPCP	Joint Water Pollution Control Plant
km	kilometer
K_{oc}	organic carbon-water partition coefficient
K_{ow}	octanol-water partition coefficient
LA	Los Angeles
LACSD	Los Angeles County Sanitation District
LCI	Los Coronados Island
LC_{50}	lethal concentration, 50 percent
LOAEL	lowest observed adverse effect level
LOEC	lowest observed effect concentration
LOEL	lowest observed effect level
LW	lipid weight
m	meter
MATC	maximum allowable tissue concentration
mg/kg	milligrams per kilogram
mg/L	milligrams per liter
ml/g	milliliters per gram
MSRP	Montrose Settlements Restoration Program
MTSF	Monterey to San Francisco
MX	Mexico
NA	not available
ND	no data available
ng/g	nanograms per gram
ng/kg	nanograms per kilogram
ng/L	nanograms per liter
NOAA	National Oceanic and Atmospheric Administration
NOAEL	no observed adverse effect level
NOEC	no observed effect concentration

NOEL	no observed effect level
NPDES	National Pollutant Discharge Elimination System
NRDA	Natural Resource Damage Assessment
NS&T	National Status and Trends
OC	organic carbon
OEHHA	Office of Environmental Health Hazard Assessment
OR	Oregon
PAH	polycyclic aromatic hydrocarbon
^{210}Pb	lead-210
PCAPFRT	Pacific Coast American Peregrine Falcon Recovery Team
PCB	polychlorinated biphenyl
PD	proposed delisted
PE	federally proposed, endangered
PEL	probable effects level
POTW	public-owned treatment works
PT	federally proposed, threatened
PVS	Palos Verdes Shelf
RI/FS	Remedial Investigation / Feasibility Study
RTV	reference toxicity value
SA	saltwater acute
SBEI	San Benito Islands
SBI	Santa Barbara Island
SC	Southern California
SCAI	Santa Catalina Island
SCB	Southern California Bight
SCBPP	Southern California Bight Pilot Project
SCC	Santa Cruz County, California
SCCWRP	Southern California Coastal Water Research Project
SCI	Santa Cruz Island

SCT	State Candidate Threatened (California)
SEC	sediment effect concentration
SET	sediment effect threshold
SLERA	Screening-Level Ecological Risk Assessment
SMAI	San Martin Island
SMB	Santa Monica Bay
SMBRP	Santa Monica Bay Restoration Project
SMI	San Miguel Island
SRI	Santa Rosa Island
SWRCB	State Water Resources Control Board
tDDT	total DDT
tPCB	total PCBs
TEF	toxic equivalency factor
TEL	threshold effects level
TEQ	toxic equivalency quotient
TOC	total organic carbon
TRV	toxicity reference value
T20	toxic sediment response 20 percent of the time
UC	University of California
UCL	upper confidence limit
USACE	United States Army Corps of Engineers
USEPA	United States Environmental Protection Agency
USGS	United States Geological Survey
WA	Washington
WES	Waterways Experiment Station
WW	wet weight

Executive Summary

Introduction

The Palos Verdes Shelf (PVS) Superfund site comprises an area of effluent-affected (EA) marine sediment off the Palos Verdes peninsula (Figure ES-1) and within the greater Southern California Bight (SCB). The PVS area has been affected by industrial and municipal effluents that have been discharged mainly through the Los Angeles County Sanitation District's (LACSD's) Joint Water Pollution Control Plant (JWPCP) outfalls, which began operation in 1937. The PVS study area is defined for this assessment as extending from Point Fermin to Redondo Canyon, north of the Palos Verdes peninsula (Figure ES-1) and includes areas of the continental shelf and slope to a depth of approximately 800 meters (m). The chemicals of potential ecological concern (COPECs) for the PVS site are dichloro-diphenyl-trichloroethane (DDT) and its metabolites (referred to collectively as total DDT or tDDT) and polychlorinated biphenyls (tPCB). The larger SCB area encompasses the large foraging areas of marine mammal and avian receptors that may be exposed to COPECs originating at the PVS site.

This Ecological Risk Assessment (ERA) for the PVS Superfund site study area describes the risk of adverse effects of tDDT and tPCB on marine biota that inhabit or may use the PVS and SCB. These biota include benthic invertebrates, benthic and water-column fish, brown pelicans, double-crested cormorants, bald eagles, peregrine falcons, and sea lions and their pups. This assemblage of receptors includes the marine food web from contaminated sediments up through invertebrate and vertebrate prey to wide-ranging, higher order consumers.

The ERA is one component of the Remedial Investigation/Feasibility Study (RI/FS) and is designed to evaluate current levels of risk through identification and characterization of the current and potential risks to the environment from contaminants at the site. The GIS-based food web exposure model presented in this report can be used to estimate future risks if site conditions change as a result of remediation or under a no-action scenario. This ERA corresponds to the baseline ERA as described in USEPA guidance for ecological risk (USEPA 1997) and a Validation Assessment as described by California Department of Toxic Substances Control (DTSC 1996) guidance.

The baseline ERA incorporates a comprehensive literature review and database summary that provide a description of the physical, chemical, and ecological settings, and the methods for evaluation of ecological risk. The resulting selection of COPECs, key ecological receptors, habitats, pathways, and selection of methods for describing and evaluating ecological risk are summarized in the conceptual site model (Figure ES-2). The model describes the contaminant sources and release mechanisms, evaluates potential exposure pathways, and identifies the representative species that were used to assess potential ecological risk to those and other similar species. The primary mechanisms for exposure are from the sediment to resident invertebrates and bottom-dwelling fish. From there, bioaccumulated COPECs continue through the food web to benthic and water-column invertebrates and water-column fish and the fish-eating consumers (marine birds and sea

lions). In addition, ecological receptors were assessed for exposure and risk through consumption of sea lion carcasses (bald eagles) and seabirds (bald eagles and peregrine falcons).

The PVS study area, and to a lesser extent the entire SCB, have been the subject of intense investigations over a number of years. Those investigations have provided a comprehensive background for constructing the conceptual model and providing the data for this ERA. The goal for this ERA was to summarize data collected throughout the SCB but with an emphasis on PVS, from as many different sources as was practical, for the period 1990–2003 (birds summarized for 1985-2000). Although numerous studies have shown that sediment and organism concentrations of tDDT and tPCB in the SCB are among the highest ever reported for any coastal marine ecosystem (USEPA 2000), within the PVS study area, concentrations of these COPECs in surface sediments and tissues of marine organisms have decreased since the 1970s and have generally leveled off since the mid 1980s.

Assessment endpoints for this ERA were developed based on known information concerning the contaminants present, the study area, the ecological management goals, and risk hypotheses. In addition, they represent properties of the system that can be measured. The assessment endpoints chosen for the PVS ERA include:

- **Community-level**: Marine infaunal and benthic invertebrate survival, abundance and diversity (as assessed by sediment and porewater COPEC concentrations and effect levels, measures of community structure, and toxicity tests)
- **Population-level**: Marine fish survival, reproduction, abundance, and diversity (as assessed by water-column COPEC concentrations and effect levels, and fish tissue concentrations)
- **Individual-level**: Special-status bird species and sea lion survival and reproduction (as assessed by dietary exposure levels and estimates of tissue concentrations)

In addition to measured internal and external exposures, a food exposure model for birds and marine mammals was used to estimate the daily dosages of COPECs from diet. The model required a knowledge of dietary composition, ingestion rates, and foraging ranges as compared to the modeled geographic distribution of fish contamination. The bird and sea lion exposure model was based on the establishment of regression relationships between COPEC concentrations in sediment and fish tissues at locations throughout the SCB. The regressions were then used to estimate potential concentrations of COPECs in fish tissue for any SCB locations. Overlapping concentrations in a mixed dietary fish assemblage with foraging range yielded an estimated daily dosage of COPECs for the bird and sea lion receptors. Peregrine falcon exposure estimates required the additional step of estimating tissue concentrations in their seabird diet (as derived from estimated fish concentrations in the seabird diet). Bald eagle exposure required a combination of exposure through dietary fish as well as sea lion carcasses and seabirds (with tissue concentrations, in turn, as estimated from their fish diets). Sea lion pup exposures were estimated from maternal milk, as estimated from maternal dietary exposure and the use of literature equations for transfer to milk.

A combination of literature-derived and site-specific data were used to assess effects levels in benthic macroinvertebrates, fish, birds, and sea lions. Effects data for benthic macroinvertebrates included site-specific sediment and literature-derived water quality

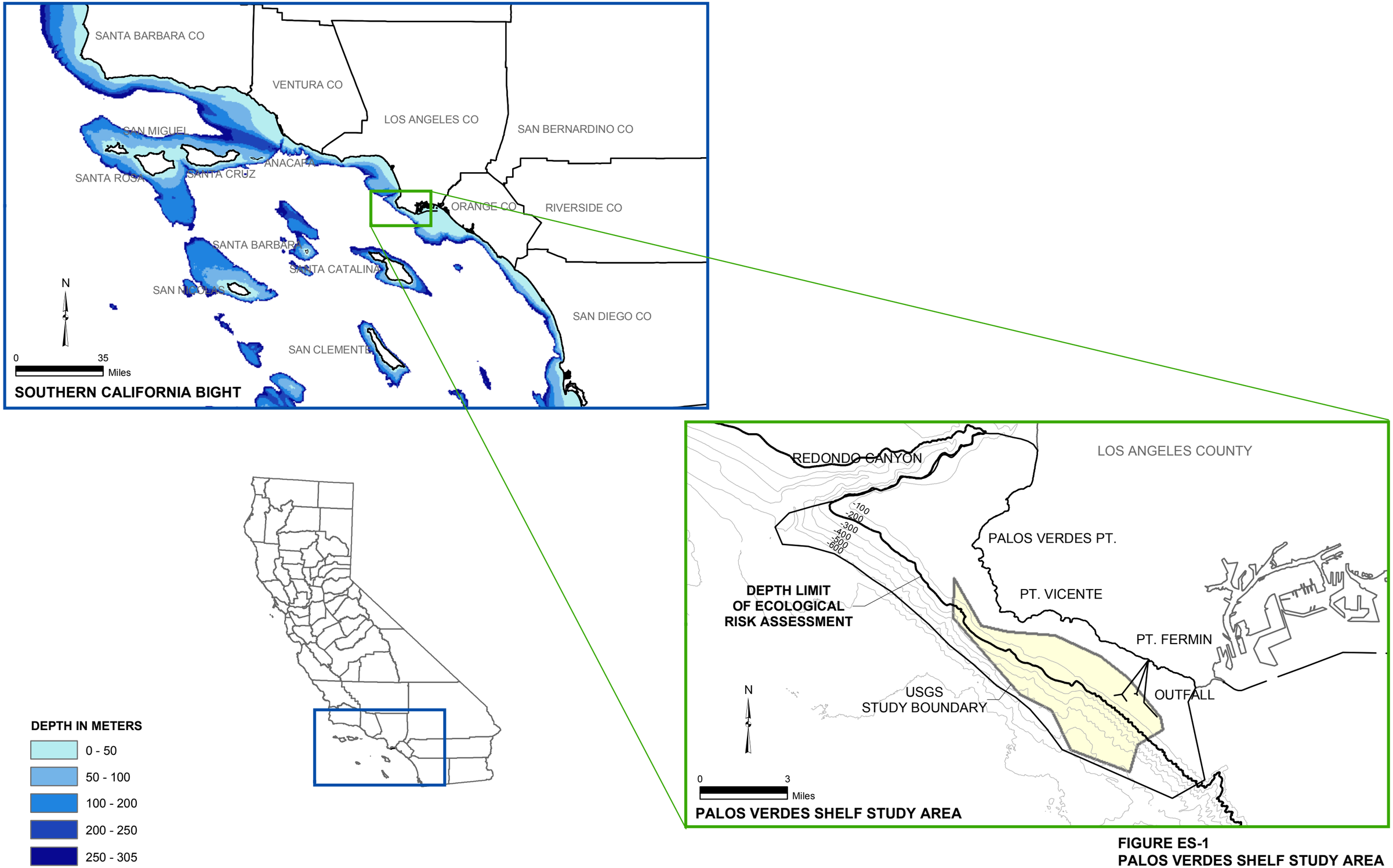

FIGURE ES-1
PALOS VERDES SHELF STUDY AREA
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

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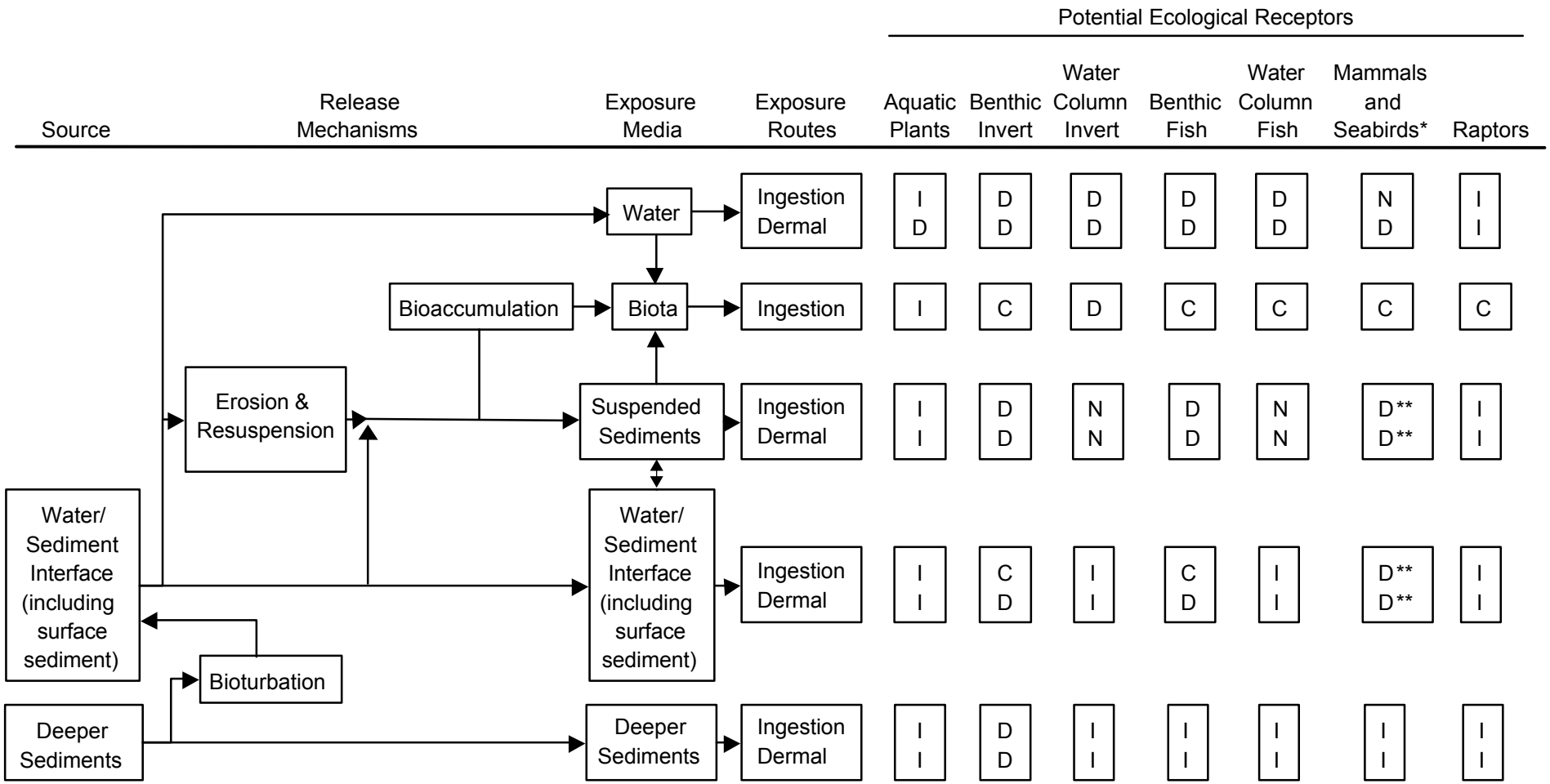

C = Complete exposure pathway
N = Possibly complete exposure pathway, but likely not significant
D = Possibly complete exposure pathway, but insufficient data for quantitative analysis
I = Incomplete exposure pathway or not applicable
* Pathways may not be quantitatively similar for mammals and seabirds; also, transfer of contaminants from mother to fetus and from mother to pup via milk ingestion is an important exposure pathway in marine mammals
** Not significant pathway for some seabirds

FIGURE ES-2
ECOLOGICAL CONCEPTUAL SITE MODEL
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

benchmarks, as well as site-specific toxicity tests and benthic community assessments. For fish, literature-derived water quality and tissue residue-based benchmarks were used to determine risk from external and internal exposure, respectively. Both oral exposure and target-organ-based internal tissue benchmarks (eggs for birds and blubber for sea lions) were developed from literature sources to assess external and internal risk to birds and mammals. In addition, site-specific studies outlining potential chronic effects from COPEC exposure (e.g., population declines, nest failures, juvenile mortality, impaired growth rates, etc.) were available for birds and mammals.

Risk Characterization

Estimates of risk were determined through comparison of measured and estimated chemical concentrations in abiotic media (measured sediment and surface water and estimated porewater), measured concentrations in biota (whole-body fish tissue, eggs for birds, and blubber for sea lions), and modeled bird and mammal exposures to site-specific and/or literature-based toxicity benchmarks. The Hazard Quotient (HQ) method employed is a simple ratio of exposure (i.e., concentrations in media/tissue or daily dosage) over effect levels (benchmarks). These quantitative risk estimates were evaluated along with the available site-specific field studies in a weight of evidence approach, and the results are described below by receptor.

Benthic Macroinvertebrates

Shallow sediments at PVS exceeded sediment quality benchmarks over 21.8 percent of the PVS study area for tDDT but only over 1.8 percent for tPCB. Similarly, PVS study area estimated porewater concentrations exceeded benchmarks over 79 percent of the area for tDDT but only 0.2 percent for tPCB. Sediment benchmark exceedances for both COPECs were relatively low at PVS, suggesting that risks are likely to be ecologically significant only for the most sensitive benthic invertebrate fauna. However, these risk estimates were supported by toxicity tests and benthic community assessments that demonstrated chronic toxicity and altered community structure (respectively) at the PVS locations nearest the outfall. All three lines of evidence support the conclusion that risk to infaunal and epibenthic invertebrates from exposure to sediment concentrations of tDDT and, to a lesser extent tPCB, is likely in the area of PVS. This risk is greatest in areas nearest to the outfalls.

Fish

Whole-body concentrations in individual fish collected in the vicinity of PVS exceeded literature-based no observed effects concentrations (NOECs) and lowest observed effects concentrations (LOECs), though the magnitude and extent of these exceedances varied by species. White croaker, a bottom-dwelling fish, had the highest percentage of tissue concentrations for individual fish exceeding benchmark values. Dover sole (another bottom-dwelling fish) and kelp bass (a water-column fish) had progressively fewer individuals that exceeded benchmark values for tissue concentrations. Most water-borne tDDT concentrations at PVS locations exceeded aquatic benchmark values, but fish did not appear to be at risk from water column concentrations of tPCB. These quantitative results indicate that risks to fish from tDDT in the immediate vicinity of PVS are likely, but are generally limited to bottom-dwelling species. In contrast, no risks are expected for PVS fish

exposed to tPCB. Site-specific field and laboratory studies provide equivocal information. Although some researchers observed reproductive effects in the laboratory, others have found that tDDT- and tPCB-related anomalies (e.g., skin lesions and tumors) have decreased to background levels. Therefore, risk to fish from exposure to tDDT cannot be eliminated; however, it is likely that this risk is greatest for bottom-dwelling fish and is localized in areas immediately adjacent to the outfalls.

Birds and Mammals

Total DDT concentrations in eggs exceeded toxicological benchmarks for brown pelicans at Anacapa and Santa Catalina Islands, double-crested cormorants at Anacapa Island, bald eagles at Santa Catalina Island, and peregrine falcons on Anacapa, San Miguel, Santa Rosa, and Santa Cruz islands. Egg tPCB benchmarks were exceeded by pelicans and cormorants on Anacapa Island, peregrines on Santa Rosa Island, and bald eagles on Santa Catalina Island.

Modeled exposure distributions and risk for brown pelicans and double-crested cormorants were similar, with high percentages of the modeled exposures (100 percent for tDDT and 24 to 100 percent for tPCB) exceeding no observed adverse effect levels (NOAELs) in both the breeding and non-breeding seasons. Exceedances of the lowest observed adverse effect levels (LOAELs) also occurred in the non-breeding season, but were absent or greatly reduced during the breeding season. Modeled oral exposures for bald eagles also exceeded tDDT and tPCB NOAELs, though LOAELs were not exceeded. Bald eagles obtain a large portion of their tDDT and tPCB exposure from their seabird and sea lion prey. Therefore, individuals are most at risk when feeding on seabird or sea lion prey that forage within areas near the PVS. For peregrines, all of the modeled oral exposures (depicted spatially as the foraging area of their seabird prey) for tDDT exceeded the NOAEL, and 16 percent exceeded the LOAEL. In general, percent exceedances of the tPCB benchmarks were lower, with less than 1 percent exceeding a LOAEL. As with bald eagles, risk to peregrines from seabird prey is greatest when those prey are foraging in areas near the PVS. Because the seabird prey generally forage between 20 to 80 km, peregrines nesting in the Northern Channel Islands would be exposed to lower dosages of tDDT and tPCB than those nesting near the PVS area (e.g., those in Los Angeles Harbor).

The quantitative risk estimates described above are supported by site-specific reproductive studies for the bird receptors. Increasing populations of pelicans and cormorants have been observed; however, current studies indicate that chronic low level exposure to tDDT is likely depressing reproduction and some individuals still have egg concentrations above thresholds for reproductive failure. Within the SCB, some breeding pairs of peregrine falcons and all breeding pairs of bald eagles on Santa Catalina Island are unable to reproduce without intervention.

The weight of evidence indicates that risk from tDDT and tPCB exposure to brown pelicans and double-crested cormorants is possible, but the risk from tPCB is expected to be lower than that from tDDT. For peregrines and bald eagles, risks are likely to be present from both COPECs, and adverse effects are probable for these species.

Limited, recent measures of adult female sea lion blubber were available for the comparison of tissue concentrations to toxicological benchmarks. None were available for pups.

However, 1991 results from adult tissues showed limited exceedances of benchmark values for DDT but none for tPCB. Modeled risks to sea lions varied by season. DDT exposures were highest near the PVS study area during the fall with exposures shifting more towards San Miguel, Anacapa, Santa Barbara, and Santa Cruz islands during the summer breeding season. PCB exposures were estimated to peak near the breeding areas on the islands. It is possible that tDDT and tPCB present risks to adult and pup sea lions as individuals, but that population effects are less probable. However, modeled exposure data indicate that the immune systems of some pups may be compromised by tDDT and tPCB exposure, though this effect on the population is unknown.

Uncertainties

Identification of uncertainties in the food web model and in the working assumptions can be found throughout the ERA and in the uncertainties section of the risk characterization. It is not always known whether a given uncertainty will result in an under- or overestimation of risk; however, every effort was made to be conservative when possible, such that confidence in a conclusion of no risk would be high. The following describes, in no particular order, several issues that were expected to contribute most to the uncertainty.

The PVS study area and SCB in general benefit from a high degree of data collection over recent years. However, the adequacy of sediment, fish, bird, and mammal data in characterizing exposure is of concern. In particular, the food web exposure model depends on modeled relationships between measured fish and sediment concentrations and the extrapolation of those relationships to unmeasured areas. There is uncertainty in the broad application of the model. All results have been summarized in the ERA as tDDT and tPCB, yet primary data sources provided either varying sums of isomers and congeners or measures of individual, dominant isomers (e.g., DDE). The choice of receptors and physiological constants (such as ingestion rates) are also a source of uncertainty for characterizing risks over such a large area using surrogates to model risk for many different species.

As always, the choice of toxicological benchmarks is a source of uncertainty in the ERA process unless detailed site-specific information is available. Even the best choice of benchmarks may not be adequate to characterize site-specific effects. It is recognized that HQs do not provide a precise quantification of risk, but rather indicate only that chosen benchmarks have been exceeded. Also, single-chemical risk estimates may be unrealistic in that natural, multi-chemical exposure may result in synergistic or additive effects.

Conclusions, Ecological Significance, and Recommendations

Multiple lines of evidence for several receptor groups show a gradient of ecological risk that is greatest in the vicinity of the PVS outfalls and along a band extending up the coast to the northwest. Intermediate risks were found in the immediate PVS vicinity and lowest risks were estimated for the more distant SCB locations. The receptor groups differ in spatial and temporal scales of exposure and, consequently, risk.

The fish and benthic invertebrate risk estimates show a predictable spatial pattern of risk for the PVS area with a significant relationship between sediment concentrations and risks

affecting the upper trophic-level consumers. Risks to fish and invertebrates were greater for tDDT than for tPCB and were greatest in the immediate area of the outfalls.

The exposure and risk from COPECs to birds and sea lions (as the representative marine mammal) in the SCB and PVS were important components of the ERA. Continuing risk is shown for birds throughout the SCB for both tDDT and tPCB. From the weight of evidence, it appears that tDDT (rather than tPCB) is associated with the greatest risk to birds.

Adult female sea lions show risk from the COPECs from both external and internal exposure pathways. Risks are measurable but small near PVS and on the Channel Islands. Sea lion pups receive high exposure to tDDT and tPCB from maternal milk, and these exposures are related to the mother's foraging range. The pups experience greater risk than the mothers; however, this increased risk from COPECs would be expected only in combination with stressful events (e.g., diseases, food shortages). Population effects may occur from this synergistic interaction.

It is recommended that chemical analysis of paired whole fish and fish fillets be conducted to develop local fillet/whole fish relationships by COPEC and species. These site- and species-specific relationships would decrease uncertainty associated with this portion of the food web exposure model and likely improve SCB-specific results. However, the current model is sufficient to predict risk based on changed levels of sediment contamination.

SECTION 1

Introduction

This report presents the baseline Ecological Risk Assessment (ERA) for current conditions in the Palos Verdes Shelf (PVS) study area. It provides qualitative and/or quantitative evaluations of the actual or potential effects of sediment concentrations of dichloro-diphenyl-trichloroethane (DDT), and its metabolites (the sum of the isomers 4,4′-DDT, 2,4′-DDT, 4,4′-DDD, 2,4′-DDD, 4,4′-DDE, and 2,4′-DDE, referred to collectively as total DDT or tDDT) and polychlorinated biphenyls (PCBs, also referred to as total PCB or tPCB) on marine biota that inhabit or may use the PVS. Some of these receptors range throughout the Southern California Bight (SCB); therefore, exposure and risk to these receptors are evaluated in this larger geographical area as well.

The PVS study area is an effluent-affected area of marine sediment off the Palos Verdes peninsula (Figure 1-1) and within the greater SCB. The PVS area has been affected by industrial and municipal effluents that have been discharged mainly through the Los Angeles County Sanitation District's (LACSD's) Joint Water Pollution Control Plant (JWPCP) outfall, which began operation in 1937. The PVS study area is defined for this assessment as extending from Point Fermin to Redondo Canyon, north of the Palos Verdes peninsula (Figure 1-1) and includes areas of the continental shelf and slope to a depth of approximately 800 meters (m). Within this study area a smaller region of contaminated sediment, about 44 square kilometers (km^2), was defined by the U.S. Geological Survey (USGS) as an area where tDDT concentrations in surface sediments (0-4 centimeters) exceeded one milligram per kilogram (mg/kg) (Lee 1994). In the report, USGS noted that, because virtually all of the area mapped has surface tDDT concentrations greater than 1 mg/kg, the actual area with surface sediment concentrations at or above this level is likely much greater than 44 km^2. Effluent-affected (EA) sediments are not physically confined to the PVS study area.

In July 1996, the U.S. Environmental Protection Agency (USEPA) authorized initiation of a response action under Superfund—specifically, the preparation of an engineering evaluation/cost analysis (EE/CA) to address the large deposit of contaminated sediments located on the PVS. A streamlined evaluation of ecological risks was completed as one element of the EE/CA (USEPA 2000). That Ecological Risk Evaluation Report provides the basis for this ERA, which has been expanded to include more spatial exposure modeling and evaluation of potential risks to ecological receptors from all levels of the food chain. The results of the ERA will provide the information necessary for risk managers to recommend remedial action within the study area.

The ERA is one component of the Remedial Investigation/Feasibility Study (RI/FS) process and plays an important role in decisionmaking for contaminated sites, as depicted conceptually in Figure 1-2 (not all elements of this figure are relevant to the PVS site). The ERA provides an estimate of the "observed or potential magnitude of adverse ecological effects at the site and the primary cause of the effects" (USEPA 1991a). The baseline ERA is used in the FS to aid in the selection of remedial goals and alternatives (including the "no

action alternative"), the development of preliminary remediation goals, the evaluation of alternate remediation scenarios, and the selection of ecological monitoring goals. It is not the role of the ERA to include those components of the FS, but rather to provide sufficient information for development of those components.

This ERA both references and partially incorporates the earlier ecological risk assessment document (i.e., USEPA 2000). The objectives, scope, approach, general assumptions, and organization of this ERA are presented below.

1.1 Objectives and Scope

The overall objective of this ERA is to qualitatively and/or quantitatively evaluate baseline or existing risks to ecological receptors caused by exposure to sediment concentrations of tDDT under existing environmental conditions on the PVS. Because prey species and receptors range throughout the SCB, both the PVS and the larger SCB area were evaluated.

The scope of this ERA includes presentation of the ecological management goals for the PVS study area; identification of potential habitats, ecological receptors, and chemicals of potential ecological concern (COPECs); selection of assessment endpoints and measures of exposure and effects; characterization of potential ecological exposures; selection of appropriate toxicity reference values (TRVs); and evaluation of potential risks, including correlation of sediment-based concentrations of COPECs with potential and/or actual adverse effects in members of the marine food chain. In addition, existing data were used to develop a Geographic Information System (GIS)-based model that can be used by risk managers to evaluate remedial alternatives as part of future studies or actions.

1.2 Approach

The approach for evaluating the PVS study area is modeled after that developed by USEPA (USEPA 1997) and the California Department of Toxic Substances Control (DTSC 1996) and is depicted in Figure 1-2. Ecological risk assessments are conducted in phases as recommended by the USEPA (1997 and 1998) and DTSC (1996). Each phase is more detailed and focused than the preceding one.

The first phase of risk assessment is the Screening-Level Ecological Risk Assessment (SLERA), which generally follows steps 1 and 2 of the Superfund program guidance (USEPA 1997, Figure 1-2). The SLERA provides a general indication of the potential for ecological risk and may be conducted for several purposes; these include estimating likelihood that a particular ecological risk exists, identifying the need for site-specific data collection efforts, or focusing site-specific ecological risk assessments where warranted (USEPA 2001).

The second phase of an ERA consists of a Baseline Ecological Risk Assessment (BERA) (USEPA 2001) that follows steps 3 through 7 of the Superfund program guidance (USEPA 1997, Figure 1-2). In this phase, the collection and interpretation of biological data are completed to further refine the conclusions of the SLERA.

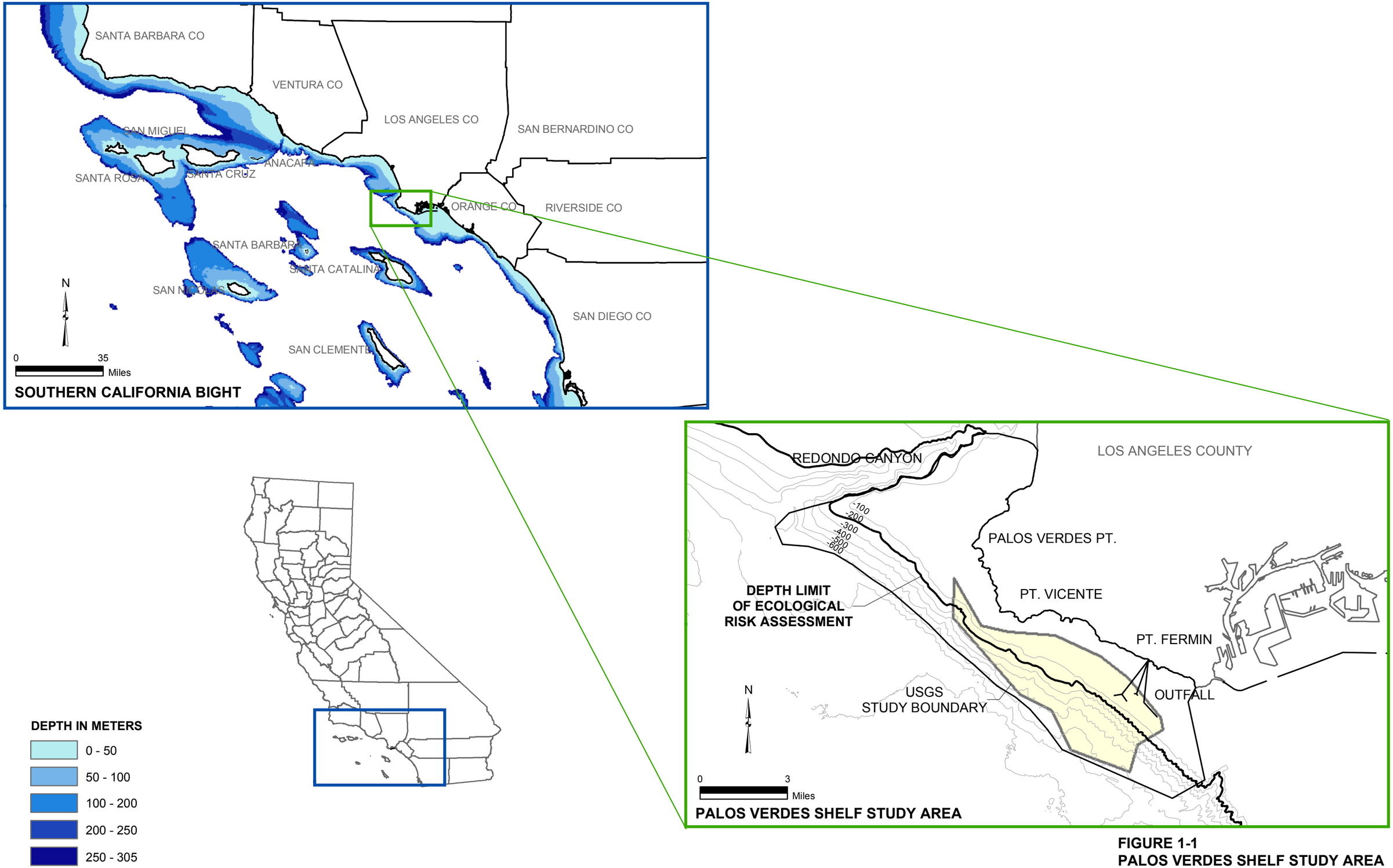

FIGURE 1-1
PALOS VERDES SHELF STUDY AREA
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

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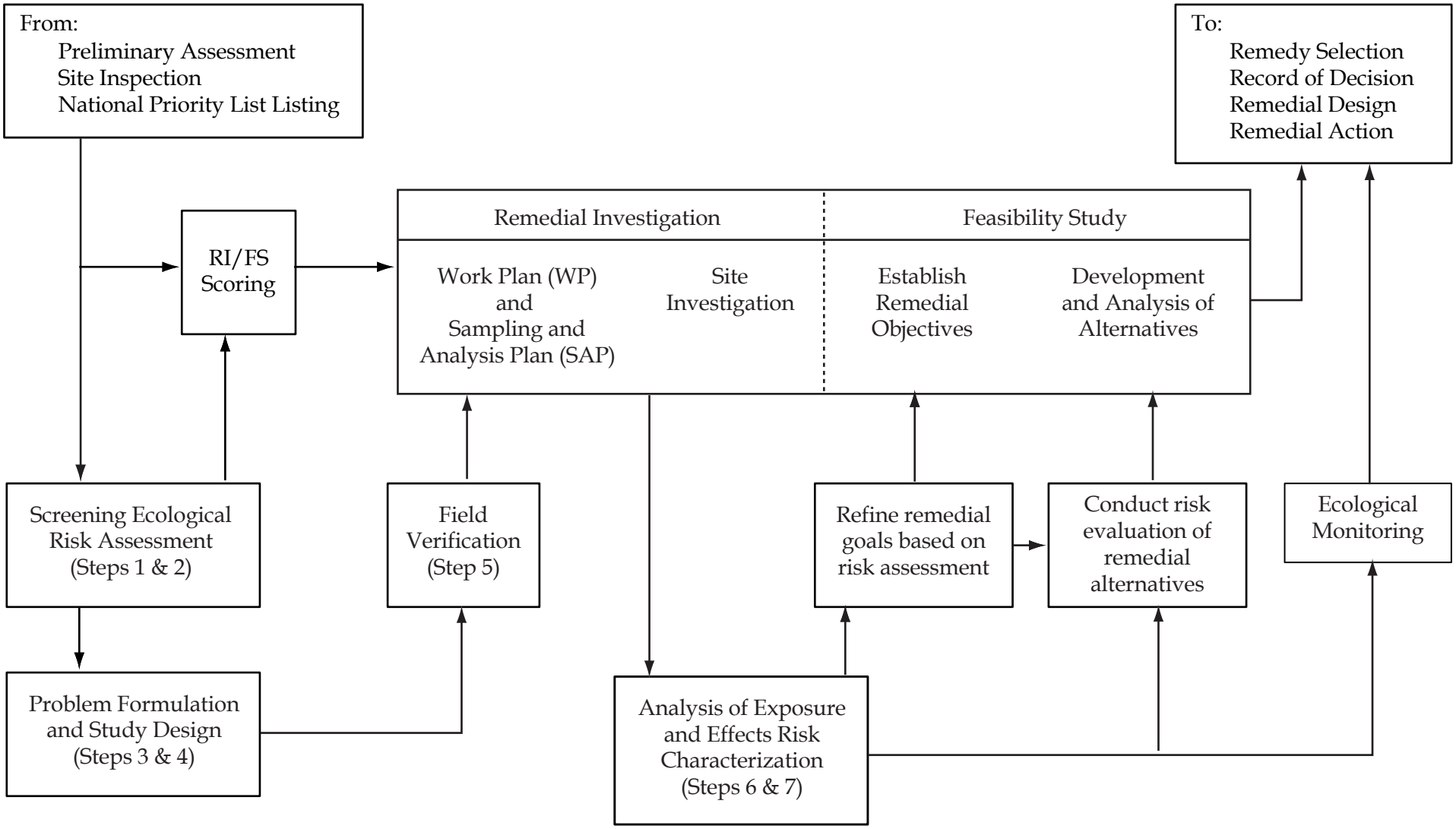

Figure 1-2
Ecological Risk Assessment in the Remedial Investigation/Feasibility Study (RI/FS) Process
Palos Verdes Shelf Ecological Risk Assessment

SOURCE: USEPA 1997

This ERA represents a complete risk assessment consistent with Steps 1 through 7 of the Superfund guidance and a Tier II Validation Assessment according to DTSC (1996). It is limited to current conditions and does not evaluate potential changes in risk over time. The earlier, streamlined ecological risk evaluation report (USEPA 2000) provided a screening-level assessment, but was limited in several areas that prevented it from fulfilling the essential requirements of an ERA as defined by USEPA (1997). These limitations included the following:

- Ecologically important receptors (seabirds, raptors, and marine mammals) with known sensitivities to the COPECs (tDDT and tPCB) were either evaluated in a limited manner or were not evaluated.
- Several of the available and relevant measurement endpoints (i.e., reports on reproductive effects in raptors in the PVS area) were not utilized in the risk analysis.

This ERA uses the information gathered for the Draft Ecological Risk Evaluation Report (USEPA 2000) and adds and/or updates several key components to produce a more complete assessment of exposures and risks in the PVS study area under current conditions as defined in the assumptions listed in Section 1.4. These updated components include the following:

- Evaluation of other available data, including residue data for invertebrate tissue, bird eggs, and marine mammal tissue
- A more comprehensive review and evaluation of bird and mammal exposure in the SCB, including distribution, migration, foraging, and diet
- Review and evaluation of fish species inhabiting the SCB and PVS, including home ranges, habitat preferences, and diet
- Literature review of DDT (and metabolites) and tPCB toxicity to fish, birds, and mammals
- Development of site-specific BSAF relationships of sediment to fish for tDDT and tPCB
- Evaluation of potential risks to ecological receptors at all levels of the marine food chain (primary, secondary, tertiary, and quaternary trophic levels)
- Development of a quantitative model linking bird and marine mammal exposure to tDDT and tPCB in sediments
- Development of a GIS-based model for mapping exposures and potential risks to facilitate remedial decisions (i.e., Food Web Exposure Model)

This ERA evaluates potential ecological risks using data from the PVS and SCB collected between 1990 and 2003 for sediment, fish, and marine mammals and collected between 1985 and 2000 for birds. A different time period was used for birds because of the availability of pertinent data for birds compared to data for the other receptors or sediment. The evaluation of the estimated risks results in one or more of the following outcomes:

- If the results of this ERA indicate that there is no risk to ecological receptors via a specific exposure pathway, that particular exposure pathway is recommended for no further ecological investigation. This situation may occur if estimated risks indicate that

contaminant levels are not toxic to the specified receptors(s) or if exposure pathways are not complete (e.g., contaminants are not present where ecological receptors would be exposed).

- If the results indicate that remedial action is needed, the results will be used in the GIS-based model to estimate reductions in risk to receptors following the implementation of various capping (or other sediment remediation) scenarios. This evaluation of alternative remediation scenarios (including the "no action alternative") is not included in the current ERA, but will be conducted after the baseline ERA is complete as a component of the FS process (Figure 1-2).
- If the results indicate potential for risk, but data gaps are present, then a Tier III Impact Assessment would be recommended. The Tier III ERA could include additional sampling and investigation to further delineate extent of contamination, to quantify the risks to specific ecological receptor groups, or to reduce uncertainty. The data collection that would be conducted as part of a Tier III ERA could include focused sampling of specific environmental media, field, and/or laboratory toxicity bioassays or tissue analyses.

1.3 Guidance

This ERA was performed in general accordance with the following guidance documents:

- *Guidance for Ecological Risk Assessment at Hazardous Waste Sites and Permitted Facilities* (DTSC 1996)
- *Framework for Ecological Risk Assessment* (USEPA 1992a)
- *ECO Updates*, Volume 1, Numbers 1 through 5 (USEPA 1991a, 1991b, 1992b, 1992c, and 1992d)
- *ECO Updates*, Volume 2, Numbers 1 through 4 (USEPA 1994a, 1994b, 1994c, and 1994d)
- *ECO Updates*, Volume 3, Numbers 1 and 2 (USEPA 1996a and 1996b)
- *Ecological Risk Assessment Guidance for Superfund Sites: Process for Designing and Conducting Ecological Risk Assessments, Interim Final* (USEPA 1997)
- *Final Guidelines for Ecological Risk Assessment* (USEPA 1998)
- *Ecological Risk Assessment and Risk Management Principles for Superfund Sites* (USEPA 1999)
- *The Role of Screening-Level Risk Assessments and Refining Contaminants of Concern in Baseline Ecological Risk Assessments* (USEPA 2001)

1.4 Assumptions

This assessment is based on the following general assumptions and constraints (additional assumptions made for specific evaluations and models are presented with the discussion regarding those models):

- The media of primary ecological concern are sediment and receptor tissue concentrations affected by bioaccumulation of COPECs.

- Current chemical concentrations are present at a steady state and will not change in the short term (defined in this assessment as 10 to 15 years). Steady-state conditions are represented by abiotic and biotic media limited to samples collected during the period 1990–2003, except birds, which have data collected during the period 1985-2000. Support for the use of these data under the current conditions definition is presented in the sediment and biota trend analyses (Sections 2.3.2.1 and 2.3.2.2, respectively).
- COPECs evaluated are limited to tDDT and tPCB. Although other chemicals have been detected (i.e., metals), they are not considered the primary risk drivers and are not included in this analysis per prior agreement with USEPA.
- Special-status species are present and are considered to be potential receptors of the most concern under current conditions.
- Exposure of most species found in the SCB study area is similar to the modeled and evaluated values for the representative receptors in this ERA.

1.5 Organization

This ERA is organized following the framework recommended by USEPA (1992a, 1997, and 1998), which consists of three main components: problem formulation, analysis, and risk characterization (shown conceptually in Figures 1-2 and 1-3). The usefulness of the ERA depends not only on the quality of available data and evaluations conducted, but also on ensuring that the ERA remains focused on the needs of the risk managers. The evaluations conducted as part of this ERA are described and discussed in the following sections. In addition, a conceptual flow diagram is presented within each section to outline the types of information used and how each type relates to the end-product(s) of each component. Additional sections consist of the following:

- **Section 2—Problem Formulation**—contains the information necessary to define the focus of the remainder of the ERA. It describes the site history, previous ecological investigations, physical and ecological setting, habitat, and observed/expected ecological species; identifies sources of contaminants and chemical data to be used in the ERA; discusses selection of assessment endpoints and measures of exposure and effects; and develops the ecological conceptual site model (CSM), including identification of representative species and exposure pathway analysis.
- **Section 3—Analysis**—presents the technical evaluation of potential exposures and adverse effects through the exposure characterization and the ecological effects characterization:
  - Exposure characterization presents a source evaluation, including fate and transport of COPECs, describes exposure assumptions and models for each representative species, and estimates the exposure for representative species.
  - Effects characterization presents an overview of the toxicity information available to derive TRVs for each representative species.

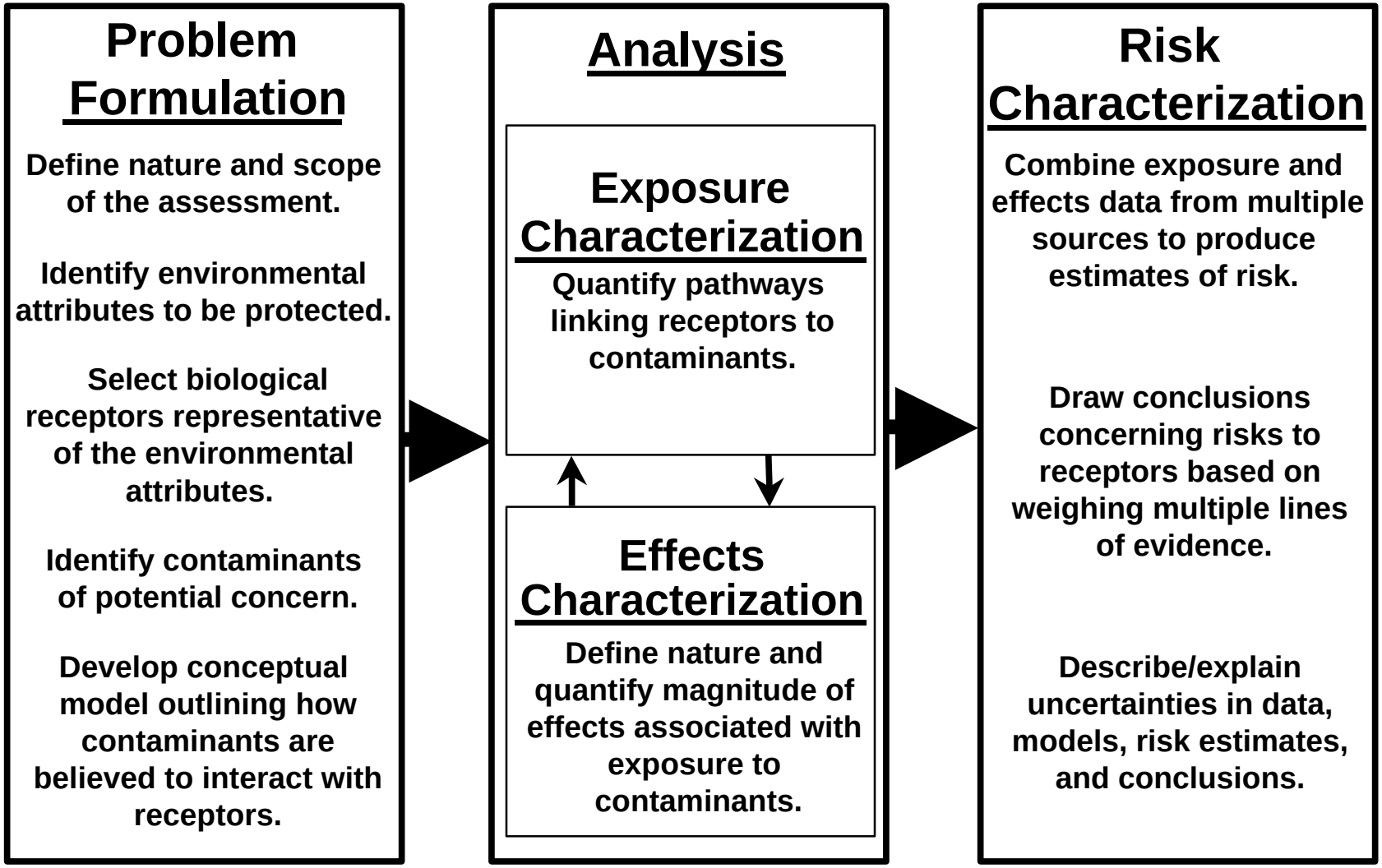

Figure 1-3
Ecological Risk Assessment Process
Palos Verdes Shelf Ecological Risk Assessment

- **Section 4—Risk Characterization**—integrates the problem formulation and the analysis to estimate the likelihood of impacts to ecological receptors from exposure to COPECs. It also discusses uncertainties and limitations associated with the risk assessment data and methodology.
- **Section 5—Conclusions and Recommendations**—summarizes the overall conclusions that can be drawn and recommendations that can be made about potential ecological risks associated with COPECs.
- **Section 6—References**—presents the literature citations for sources used to support this ERA.
- **Appendix A—Database**—includes the chemical database used for this ERA (available on CD-ROM).
- **Appendix B—Life-History Summaries for Representative Species**—summarizes receptor life-history information.
- **Appendix C—Food Web Exposure Model**—evaluates sediment contamination effects throughout the Southern California Bight.
- **Appendix D—Calculation of Oral Benchmarks for Birds and Mammals**—presents detailed toxicological information for marine birds and mammals.

SECTION 2

Problem Formulation

This section includes the historical background of the site, physical and ecological settings, sources of chemical and tissue data available for analyses, identification of COPECs, presentation of the ecological management goals for the PVS and SCB study areas and important aspects of the site to be protected (referred to as "assessment endpoints"), the means by which the assessment endpoints will be evaluated (measures of exposure and effects or "measures"), the habitats and ecological receptors present, and evaluation of potential exposure pathways. The need for both the PVS and SCB study areas is discussed on a receptor-specific basis, below. The end product of the problem formulation is a conceptual site model that describes the contaminant sources and transport mechanisms, evaluates potential exposure pathways, and identifies the representative species that will be used to assess potential ecological risk to those and other similar species. Integration of this information to develop the ecological conceptual site model is shown in Figure 1-3.

2.1 Site Background

This subsection briefly describes the physical location of the PVS and SCB study areas, relevant site history, and previous ecological investigations.

2.1.1 Location

The PVS study area is an effluent-affected area of marine sediment off the Palos Verdes peninsula and within the greater SCB. The PVS study area extends from Point Fermin to Redondo Canyon off the Palos Verdes peninsula and includes areas of the continental shelf and slope. The SCB, PVS, and previous USGS study areas are all depicted in Figure 1-1.

The continental shelf is very narrow in the PVS region, having a width of 1.5 to 4 km and a slope of 1 to 4 degrees. There is a steep increase in slope between 50 and 100 meters (m) in depth, producing an edge to the continental shelf in water depths of 70 to 100 m. Beyond the shelf break is the continental slope, which extends to a depth of approximately 800 m. The slope is also narrow and steep in this region, having a width of approximately 3 km and mean slope of 13 degrees (Lee 1994)[1].

In addition to the shelf and slope, the immediate area of the PVS also includes adjacent parts of the San Pedro Basin, which is south of the slope in water depths of at least 800 m (Lee 1994). Redondo Canyon occurs to the northwest of Point Vicente (Drake 1994) and to the north of the PVS study area. The Redondo and San Pedro shelves are to the west and east of the PVS, respectively (Lee 1994). The SCB contains all of these areas plus the offshore Channel Islands and intervening deep-ocean areas extending from Point Conception in the north to the U.S.-Mexico border in the south (Figure 1-1).

[1] Several studies cited in this ERA were produced as part of the *U.S. v. Montrose* CERCLA litigation regarding the PVS. In the listing of references, these studies are noted as being "Expert Report" or similar documents in *U.S. v. Montrose*.

2.1.2 Site History

Contamination to the PVS study area has historically occurred via municipal and industrial effluent entering from outfalls. The LACSD JWPCP outfall began discharging treated wastewater at a depth of 33 m from an outfall at White Point in 1937. Wastewater from the JWPCP was later discharged through submarine outfalls located approximately 2.5 km offshore to a water depth of 63 m (Drake 1994). The LACSD JWPCP outfalls have discharged approximately 4 million tons of suspended solids since 1937, half of which was discharged from 1964 to 1976. Wastewater contaminants included chlorinated hydrocarbons (e.g., tDDT and tPCB) as well as trace metals (cadmium, copper, lead, and zinc) and organic matter (Drake 1994; Stull 1995). The primary source of tDDT was wastewater from the Montrose Chemical Corporation of California, Inc. (Montrose) that was discharged to the JWPCP and then to the PVS study area via the submarine outfalls. Montrose operated a DDT manufacturing plant in Los Angeles County from 1947 to 1983. Total PCB from several sources in the greater Los Angeles area were also discharged from the JWPCP outfall and resulted in widespread contamination of sediments.

Bans on production and use of DDT and tPCB, along with improved source control and treatment technologies, led to a decline in mass emissions of these chemicals. Discharges of DDT declined from 100 tons per year in the late 1960s to 0.03 tons per year in 1985 (Drake 1994). By 1995, the solids mass emission was less than one-fifth of that discharged in 1971, and trace contaminant discharges were a small percentage of 1971 values (Stull 1995). Less-contaminated effluent and natural sediment have gradually buried the heavily contaminated sediment in the area of the outfalls. The EA sediments at the site have a lower density and finer grain size than the native sediment. The EA sediment ranges in thickness from 5 centimeters (cm) to greater than 60 cm, and is underlain by the firmer native shelf sediments. The total volume of the EA deposit is over 9 million cubic meters, with approximately 70 percent of this volume lying on the shelf and the remainder on the slope.

2.1.3 Previous Ecological Investigations

Several previous investigations have been conducted in or near the PVS study area to determine the extent of contamination. Numerous studies have shown that sediment and organism concentrations of tPCB and tDDT in the SCB are among the highest ever reported for any coastal marine ecosystem (USEPA 2000). Key aspects of previous ecological investigations specific to benthic invertebrates, fish, and marine birds and mammals are presented chronologically below, followed by an overview of ongoing regional studies.

2.1.3.1 Benthic Macroinvertebrates

Studies involving benthic macroinvertebrates have included both laboratory toxicity tests and field studies of abundance and diversity. Toxicity studies are presented first, followed by field studies.

Toxicity tests of Palos Verdes sediments have resulted in a 10-day acute 50 percent lethal concentration (LC_{50}) of 10.8 milligrams/kilograms (mg/kg) wet weight (WW) PCB using the benthic amphipod, *Rhepoxinius abronius* (Swartz et al. 1988), and acute and chronic threshold responses were found to range from 0.1 to 4 mg/kg dry weight (DW) PCB (Dexter and Field 1989).

Bay et al. (1994) performed tests on infaunal and epibenthic organisms to determine the toxicity and bioaccumulation characteristics of PVS sediment. Toxicity was found to coincide with distance from the outfall and depth. The study suggested that growth effects

were not caused by exposures to 4,4′-DDE, nor were metals likely to be significant causes of sediment toxicity.

Bioassays were performed using marine polychaete full-life-cycle tests as part of the Natural Resource Damage Assessment (NRDA). No observed effect concentrations (NOECs) in sediment, based on survival, growth, fecundity, and reproduction, were 8.51 mg/kg DW for DDT and 1.07 mg/kg DW for tPCB. Dispite this evidence for potential chronic toxicity values, amphipod tests showed no acute toxicity.

An extensive study by MacDonald (1997) established sediment effect concentration (SEC) thresholds for DDT, DDE, DDD, tDDT, Aroclor 1254, and tPCB using a tiered strategy and a weight-of-evidence approach. Exceedances of these established thresholds would indicate that survival and reproductive effects were likely. MacDonald concluded that some SCB sediments would likely cause injury to sediment-dwelling organisms.

Field studies assessing benthic community structure and function have been conducted since the 1970s. In addition to the effects of contaminants, other factors that have influenced the invertebrate communities include sediment inputs from the Portuguese Bend landslide; fluctuations in temperatures, currents, and storms associated with El Niño events; and biological disturbance such as the temporary invasion of and bioturbation by the echiuran worm (*Listriolobus pelodes*) from approximately 1973 to 1977 (LACSD 1995, 2001; Stull 1995; Drake 1994).

Field studies conducted in the mid 1990s indicated increased diversity and number of taxa across the entire shelf and slope; recolonization by certain pollution-sensitive species to areas from which they were previously absent (e.g., nearer the outfall); increased abundance of arthropods and echinoderms, many of which are pollution-sensitive; shared dominance among more species; and decreased abundance of opportunistic species (e.g., *Capitella capitata*) that often are indicators of stressed conditions (Stull 1995).

Multivariate (cluster) analysis of southern California trawl data from 1971 to 1984 classified the Palos Verdes samples as having a unique low-diversity assemblage. The analysis also indicated that the unique cluster group declined in the 1980s, and Palos Verdes samples were classified with the typical shelf assemblage for southern California (Stull 1995). The distribution and diversity of the epibenthic macroinvertebrates have increased since the 1970s. Some of the changes may be attributed to improved habitat quality, although other environmental variables (e.g., El Niño events) have had significant effects on these populations (Stull 1995). As examples, some species (e.g., *Eusicyonia ingentis* and *Pluroncodes planipes*) increased in abundance during El Niño years, while other populations (e.g., *Crangon alaskensis*, *Spirontocaris holmesi,* and *Astropectin verrilli*) decreased during these years (LACSD 1995).

The Southern California Coastal Water Research Project (SCCWRP) has attempted to define a reference benthic condition for benthic communities of the SCB (Bergen et al. 1999). Additional studies by SCCWRP have examined benthic condition as defined by a Benthic Response Index (BRI; Smith et al. 1999) at several sites on the PVS using LACSD data (Bay et al. 1998). These analyses indicated that, of the 24 stations examined, 14 experienced loss of biodiversity and 3 exhibited loss of community function. Only 5 stations were

characterized as having a benthic condition that fell within the range of regional reference sites.

A comprehensive study of marine benthos and their weighted average response to contaminant inputs in the SCB (through use of a BRI) confirms the relative stability of the benthic community in the PVS area over the last 10 years (Smith et al. 1999).

2.1.3.2 Fish

Fish studies have included field collection of fish to evaluate physical abnormalities, bioaccumulation, and relative abundance and diversity.

Trawling surveys from 1973 to 1995 indicated a decrease in the number of external anomalies (e.g., fin erosion, tumors, external parasites, skeletal deformities, black lesions, and color anomalies) in fish (LACSD 1999). It is not known whether these lesions are caused by exposure to tDDT or tPCB.

Field studies investigating the reproductive potential and ovarian tissue burdens of tDDT and tPCB in white croaker (*Genyonemus lineatus*) and kelp bass (*Paralabrax clathratus*) were conducted from 1985 to 1988 (Hose and Cross 1997; Hose et al. 1989). Reproductive potential was found to be impaired in 1985 and 1988, although tissue concentrations of tDDT and tPCB were significantly lower in 1988 and two of three measures of reproductive success were higher than in 1985. Effects of tDDT and tPCB on reproductive impairment in white croaker were compared in fish collected from San Pedro Bay versus Dana Point. In 1985, four of the five measures of reproductive potential were significantly lower in fish from San Pedro Bay compared to fish from Dana Point. Additionally, ovarian tDDT and tPCB concentrations were significantly higher in the San Pedro Bay fish than the reference population. A second study in 1988 also supported the conclusion that concentrations of tDDT and tPCB were causing reproductive impairment, but impacts were not as substantial. Two of three measures of reproductive success that were diminished in the 1985 study were significantly higher in 1988. Ovarian concentrations of tDDT and tPCB in San Pedro Bay croaker were also lower than those in 1985.

A fugacity model was developed to determine bioaccumulation in fish using a flatfish bioindicator exposed to PCB- and DDT-contaminated Bight sediments (Young et al. 1991). Tissue and sediment concentrations were measured to validate the model. Concentrations of tPCB and tDDT in sediment from the PVS were 38 and 300 times greater than those from reference sites, while fish tissue concentrations were 10 and 8 times higher. The median accumulation factors (AFs) for DDE from the PVS in liver and muscle were 1.7 and 2.0, respectively, while the AFs for liver and muscle from the reference site were 1.8 and 3.4, respectively.

One study by Hansen and Associates (2000) evaluated temporal trends in the concentrations of DDT and its metabolites in the tissues of white croaker. After reviewing various sources and performing several statistical analyses, the study team concluded that the concentrations of DDT (lipid-normalized) and its metabolites have been decreasing in the tissues of white croaker over the past 12 years. According to Hansen and Associates (2000), this indicates that the biological availability of DDT and its metabolites in the sediments has also been declining.

2.1.3.3 Birds and Mammals

Studies have also been conducted to determine effects of contaminants in the Bight on birds and mammals in the region. One study evaluated the injury of seabirds from DDT and PCB residues (Fry 1994). The study concluded that all species of seabirds investigated had elevated levels of DDT and PCB residues in eggs from SCB colonies when compared to colonies outside of the Bight. Gress (1994) evaluated the reproduction, eggshell thinning, and organochlorine data for brown pelicans (*Pelecanus occidentalis californicus*) and double-crested cormorants (*Phalacrocorax auritus*) in the Channel Islands. Both species were severely impacted by DDE, but have had major population increases since 1979 as DDE levels dropped in the food web. DDE levels in brown pelicans began to level off around 1985 and fluctuations in breeding success after this time were more influenced by factors other than DDE such as El Niño events and domoic acid toxicity (Gress et al. 2003). A study by Garcelon (1994) evaluated the effects of contaminants on bald eagle (*Haliaeetus leucocephalus*) reproduction on Santa Catalina Island. It was concluded that levels of DDT and DDE in bald eagles were adversely affecting bald eagle reproduction. A recent study (Sharpe 2003) reports the Santa Catalina Island breeding data for 1989 to 2002 and indicates that bald eagles are still not able to successfully hatch their own eggs. Hunt (1994) conducted a field study of peregrine falcons (*Falco peregrinus*) on the Channel Islands. Dietary studies indicated that local seabirds are the primary source of DDE in the peregrine falcon. Additionally, an extreme degree of shell thinning was noticed from 1992 to 1994 that corresponded with high levels of DDE.

In 1994, a damage assessment food-web/pathways study was performed, then revised in 1997 (HydroQual, Inc. 1997) as part of the Natural Resource Damage Assessment (NRDA) conducted by state and federal natural resource trustee agencies. The goal of the study was to determine whether sediment serves as the source of contamination for organisms that do not have direct contact with the contaminated sediment plumes. Results indicate that contaminants found in sediments, fish, and water of the SCB originated from the PVS. Similar results concluded that the DDE and PCB concentrations in the media are sufficient to account for all of the DDE and tPCB observed in fish living on the shelf. For marine mammals, while concentrations in prey observed at low-level contaminant sites (i.e., those areas farther from the outfalls such as Santa Catalina Island and the Northern Channel Islands) are insufficient to account for the DDE and PCB levels found in female sea lions (*Zalophus californianus*), concentrations in prey living closer to the outfalls are sufficient to account for the measured concentrations. Additionally, at least half of the contaminant exposure to the peregrine falcon and bald eagle originates from the Southern California Bight.

2.1.3.4 Regional Investigations

In addition to lab and field studies focusing on one group of receptors, several programs have focused on the ecology of the PVS region. Existing studies include those conducted by SCCWRP, LACSD, and National Oceanic and Atmospheric Administration (NOAA), as summarized in the following paragraphs.

SCCWRP has conducted several regional surveys. Two of these investigations include the Southern California Bight Pilot Project (SCBPP) in 1994 (Bight 94) and a larger-scale study in 1998 (Bight 98).

The SCBPP provided regional data to determine ecological conditions and alterations in soft-bottom habitats in the SCB at water depths of 10 to 200 m (LACSD 2000). Results from this survey indicated that water quality was good throughout the Bight although anthropogenic chemicals were present in 89 percent of the samples. Bight benthic infaunal communities varied with no severe community alterations, and fish communities were healthy even though DDT and tPCB were present in flatfish livers. Santa Monica Bay sediments had higher levels of contamination than the rest of the Bight, but all other measured conditions were similar. In discharge areas, levels of DDT, PCB, copper, mercury, and silver were found to be higher. Stormwater discharge areas were enriched in sediment contamination with minor alterations in benthic infaunal communities in 40 percent of the stormwater discharge areas. Overall, the report concluded that impacts from anthropogenic sources were relatively small.

The Bight 98 survey (SCCWRP 2002) served to build upon the initial SCBPP findings from Bight 94. The focus included coastal ecology, shoreline microbiology, and water quality. Results of the shoreline microbiology program indicated that bacteriological water quality was consistently good along the southern California shoreline. The Mexican beaches met California bacteriological water quality standards 75 percent of the time. Preliminary fish data from Bight 98 were used in the present analysis, along with partial Bight 98 sediment sample results.

LACSD conducts an extensive monitoring program at PVS. This program is a requirement of the JWPCP discharge permit. This Palos Verdes ocean water quality and bottom monitoring program investigates physical oceanography, sediments and infauna, invertebrate and fish populations, and bioaccumulation.

In developing a model monitoring program for the large public-owned treatment works (POTWs), such as LACSD, in southern California, SCCWRP and regulatory agencies decided that sampling fish contamination for human health risk concerns was a regional issue to be assessed in periodic (about 5 years) regional surveys.

The California Fish Contamination Project, funded by the State Water Resources Control Board (SWRCB) with Office of Environmental Health Hazard Assessment (OEHHA) oversight and samples collected and analyzed by California Department of Fish and Game (CDFG), has an ongoing survey of fish contamination along the California coast, sampling at piers and other sites. Data are available for two years of surveying.

NOAA published an inventory and assessment of contaminant trends in the SCB in 1991 (Mearns et al. 1991). This document serves to summarize existing information on contaminants measured by the National Status and Trends Program. It describes the data that exist for documenting the geographic distribution and long-term trends of groups or contaminants in sediments, mussels, fish, and other species of the SCB. It reviews ten trace elements, organotin compounds, polycyclic aromatic hydrocarbons (PAHs), tPCB, and several pesticides. Newport Bay, San Diego Harbor, Santa Monica Bay, Marina Del Rey, Los Angeles-Long Beach harbors, and PVS have the highest levels of DDT, tPCB, copper, lead, mercury, or zinc. The least contaminated areas for sediments are the coastal shelf areas near Santa Barbara, and between Orange County and Point Loma. Elevated sediment levels did not always coincide with elevated mussel or fish tissue concentrations. Long-term trend data indicate that contaminant concentrations are not increasing in sediments, mussels, or

fish in the Bight. While it seems evident that all contaminants have accumulated in sediment, not all have accumulated to toxic levels nor have all contaminants accumulated in mussels and other species.

2.2 Physical and Ecological Setting

This subsection provides an overview of the physical and ecological environments of the PVS study area and the wider SCB area, emphasizing current conditions.

2.2.1 Regional Setting

The PVS study area is in a region of the SCB characterized by warm, dry summers, tempered by ocean breezes, with mild winters. Data from Long Beach, California indicate that the annual average rainfall of about 13 inches occurs primarily from November to March (NOAA 2003). Fog and low clouds typically occur from February to April. In summer, morning fog and low clouds usually persist until mid-afternoon, keeping summer temperatures mild. In Long Beach, California, the average daily temperature in the summer is 23 degrees Celsius (°C), winter temperatures average 14°C, and record temperatures range from -3.8 to 44°C (NOAA 2003). The prevailing winds, which blow onshore from the southwest, help lower summer temperatures and dissipate the summer fog. In autumn, strong, gusty winds from the inland deserts, known locally as Santa Ana winds, cause unseasonably warm days.

2.2.2 Physical Oceanographic and Geological Conditions

Dominant circulation patterns in the SCB include the southward-flowing California Current, the northward-flowing California Countercurrent, and seasonal influences by the northward-trending Davidson Countercurrent (Drake 1994; Hickey 1992). Because the Palos Verdes peninsula extends prominently into the Bight, it is more directly affected by the California Countercurrent than are nearby bays. Surface and bottom waters are typically separated in spring through fall by a pycnocline (area of rapid change in temperature and salinity) occurring at depths of approximately 10 to 30 m. Currents below the pycnocline in the PVS study area, which would carry resuspended sediments, generally flow to the northwest, parallel to bathymetric contours. In contrast, surface currents flow predominantly southeastward, although they shift to a westerly flow in late autumn and winter when westerly winds weaken (Hickey 1992).

Sediment transport follows predominant current flow directions near the bottom, extending northwestward along the shelf (Drake 1994). This is also reflected in the shape of the effluent-affected sediment deposit and previously defined contaminant “footprint” away from the JWPCP outfall at White Point. Given the high wave energy, the effluent deposit does not at present extend shoreward of approximately 30 m depth. The mean circulation patterns in the SCB sweep up the coast from San Diego, past the PVS study area and Santa Monica Bay (SMB) and then out toward and circling the Channel Islands. On a smaller scale, the local PVS currents generally flow to the northwest from the PVS study area toward SMB, where they are immediately met with a southerly flowing current coming out of the bay. Thus, although currents may tend to move contaminants from PVS toward SMB, the flows from SMB tend to reduce further northward or northwestward movement (Hickey et al. 2002). The predominant

southerly flows out of SMB would tend to push particulate-associated PVS contaminants from the southern boundary of SMB to deeper water. The relatively narrow continental shelf at PVS and the southerly flows from SMB may act to push resuspended PVS sediments to deeper waters off the shelf into the PVS study area.

The predominant bottom currents over the PVS flow to the northwest and do not vary seasonally in direction (Noble et al. 2002). Current velocities in the vicinity of the JWPCP outfalls vary seasonally as a result of changes in wind patterns and periodic storms—typically, in winter or early spring. Shelf currents are generally of low velocity, with an average of 7 to 10 centimeters per second (cm/sec) throughout the year (LACSD 1995). Storm waves generated during the winter typically have maximum heights of 3 to 4 meters. However, wave heights up to 7 m were observed during major storms occurring in the 1980s (LACSD 1995).

The Portuguese Bend landslide (due to anthropogenic influences) and the JWPCP discharge have dominated the recent supply of sediment to the Palos Verdes Shelf. Since 1988, the rate of erosion from the Portuguese Bend landslide has decreased as a result of stabilization projects, which reduced movement to about 10 percent of former rates. Redondo Canyon and San Pedro Canyon bound the Palos Verdes Shelf to the northwest and southeast, respectively, and limit sediment transport from adjacent shelf areas. Los Angeles-Long Beach Harbor and its breakwater obstruct nearshore sediment transport (Drake 1994).

The thickness of shelf sediments varies, ranging from 32 m on the southeastern part of the shelf to less than 10 m near Point Vicente. Owing to strong currents, a patchy thin sediment layer with areas of bare rock occurs at the shelf break (Palermo 1994). Similar bedrock outcrops also occur over the sea floor to the east of the outfall and over the Redondo Shelf to the west (Lee 1994). Less than 1 m of sediment covers the Redondo Shelf (Drake 1994). In the immediate area of the PVS, sediments southeast of the outfall are eroding and depositing to the northwest (Zeng and Venkatesan 1999).

2.2.3 Habitat Evaluation and Observed/Expected Ecological Receptors

The PVS study area is characterized by (1) soft-bottom subtidal habitat, including invertebrate and fish communities, over most of the continental shelf and slope region to approximately 800 m; (2) hard-bottom habitat, including some kelp bed areas and associated invertebrate, fish, and algae communities, from shore to at least 20 m depth; and (3) pelagic (i.e., water column) zones, representing important habitat for fish, invertebrates, birds, and mammals from near the bottom to the surface. The exception to this pattern is the hard-substrate, artificial reef habitat represented by the wastewater outfall structure that extends primarily over soft bottom to a depth of approximately 60 m, some scattered hard-bottom areas on the shelf, and more extensive hard-bottom areas along some parts of the shelf break. Sediments near the outfall are fine-grained, organically enriched, and variably contaminated as a result of historic discharges (LACSD 2001). The present areas of highest chemical contaminant concentrations are located in the soft-bottom habitat. Diverse marine habitats and biological communities typify the PVS study area and the broader SCB region. The ecological receptors that have either been observed or are expected to occur in each habitat type in the PVS study area are presented below. Observed or expected special-status species are listed in Table 2-1.

TABLE 2-1
Summary of Special-Status Wildlife Species Potentially Occurring in the Southern California Bight
Palos Verdes Shelf Ecological Risk Assessment

Common Name	Scientific Name	Federal Status	State Status
Birds			
Ashy storm petrel	*Oceanodroma homochroa*		CSC
Baird's sandpiper	*Calidrus bairdii*		CSC
Bald eagle	*Haliaeetus leucocephalus*	FT	CE
Bank swallow	*Riparia riparia*		CT
Belding's savannah sparrow	*Passerculus sandwichensis beldingi*		CE
Black storm-petrel	*Oceanodroma melania*		CSC
Black tern	*Chlidonias niger*		CSC
Black swift	*Cypseloides niger*		CSC
California black rail	*Laterallus jamaicensis coturniculus*	FE	CT
California brown pelican	*Pelecanus occidentalis californicus*	FE	CE
California gull	*Larus californicus*		CSC
California least tern	*Sterna antillarum browni*	FE	CE
Common loon	*Gavia immer*		CSC
Double-crested cormorant	*Phalacrocorax auritus*		CSC
Elegant tern	*Sterna elegans*		CSC
Fork-tailed storm petrel	*Oceanodroma furcata*		CSC
Light-footed clapper rail	*Rallus longirostris levipes*	FE	CE
Long-billed curlew	*Numenius americanus*		CSC
Marbled murrelet	*Brachyrampus marmoratus*	FT	CE
Osprey	*Pandion haliaetus*		CSC
Peregrine falcon	*Falco peregrinus*	[a]	
Rhinoceros auklet	*Cerorhinca monocerata*		CSC
San Clemente loggerhead shrike	*Lanius ludovicianus mearnsi*	FE	
San Clemente sage sparrow	*Amphispiza nelli clementae*	FT	
Short-tailed albatross	*Phoebastrua albatrus*	FE	
Tufted puffin	*Fratercula cirrhata*		CSC
Western snowy plover	*Charadrius alexandrinus nivosus*	FT	CSC
Xantus's murrelet	*Synthliobramphus hypoleucus*		SCT
Mammals			
Blue whale	*Balaenoptera musculus*	FE	
California sea lion	*Zalophus californianus*		CP

TABLE 2-1
Summary of Special-Status Wildlife Species Potentially Occurring in the Southern California Bight
Palos Verdes Shelf Ecological Risk Assessment

Common Name	Scientific Name	Federal Status	State Status
Fin whale	*Balaenoptera physalus*	FE	
Guadalupe fur-seal	*Arctocephalus townsendii*	FT	CT
Harbor seal	*Phoca vitulina*		CP
Humpback whale	*Megaptera novaeangliae*	FE	
Island fox	*Urocyon littoralis*	PE	CT
Northern elephant seal	*Mirounga angustirostrus*		CFP
Northern fur-seal	*Callorhinus ursinus*		CP
Northern sea lion	*Eumetopias jubatus*	FT	CP
Sei whale	*Balaenoptera borealis*	FE	
Southern sea otter	*Enhydra lutris nereis*	FT	CFP
Sperm whale	*Physeter macrocephalus*	FE	

Notes:

[a] Peregrine falcon is a former Federally Endangered species that was delisted on August 25, 1999. Its current status is "Delisted Taxon, Recovered, Being Monitored First Five Years."

Sources: CDFG, Natural Diversity Data Base (2002); Ventura Fish and Wildlife Office web site.
Counties searched: Ventura, Santa Barbara and Los Angeles; and Channel Islands (*Anacapa, San Clemente, Santa Barbara, San Miguel, Santa Rosa, Santa Cruz, Santa Catalina, San Nicolas*)

Federal Status Codes:
FE federally listed, endangered
FT federally listed, threatened
PE federally proposed, endangered
PD proposed delisted
PT federally proposed, threatened
State Status Codes:
CE California-listed, endangered
CT California-listed, threatened
CFP California fully protected
CP California-protected
SCT State Candidate Threatened (California)

2.2.3.1 Soft-Bottom Subtidal Habitats

Soft-bottom habitats grading from sand to mud typify most of the sea bottom deeper than approximately 20 m off the Palos Verdes peninsula. Key inhabitants include infaunal and epifaunal invertebrates, both of which live in close association with the sediments and typically reside (especially infauna) in discrete areas as adults. Numerous bottom-feeding fish are also characteristic of these habitats but typically are much more motile than the invertebrates, and some fish species migrate over broad depth ranges.

Infaunal Community

The infaunal community (invertebrates living in soft sediments) on the shelf and slope is dominated by deposit feeders, primarily polychaete worms and small bivalves, but includes the full range of feeding types, including particle/suspension feeders and predators

representing numerous phyla (LACSD 1995, 2001). This community represents an important food source for many fish species and other invertebrates. The most abundant species are the bivalves *(Parvilucina tenuisculpta* and *Axinopsida serricata)*, and the polychaete (*Aphelochaeta marioni)* (LACSD 1995). Other common infauna include various polychaetes *(Aphelochaeta* sp. C, *Pista fasciata, Paraprionospio pinnata*, *Monticellina tesselata*, *Mediomastus* spp., *Glycera nana*, and *Pectinaria californiensis)* and the ostracod *(Euphilomedes carcharodonta)*.

Based on results from surveys by LACSD (1995), the greatest number of individuals occurs in the outfall area (>10,000 individuals/m^2), although the number of organisms is also high (>7,500 individuals/m^2) at stations off Point Vicente, Long Point, and Portuguese Bend (depths ranging from 30 to 152 m). Fewer individuals (<2,500 individuals/m^2) occur in the deeper (305 m) areas of the slope. Biomass is enhanced near and offshore of the outfall as a result of discharges of organic material. In general, the number of taxa and diversity are highest on the shelf and lowest on the slope. Diversity is also lower northwest of the outfall, in the general area of highest chemical contamination, although temporal trends have shown an increase in diversity in this area.

Epifaunal Invertebrate Community

In addition to the infaunal community surveys, LACSD has conducted trawl surveys of fish and invertebrates along the PVS shelf and slope since the early 1970s. Trawls were conducted at stations located at depths of 23, 61, and 137 m. The sea star (*Astropecten verrilli*), ridgeback prawn (*Eusicyonia ingentis*), white sea urchin (*Lytechinus pictus*), and California sea slug (*Pleurobranchaea californica*) were present in at least 50 percent of the samples collected by LACSD during 1995. The white urchin and sea star were the most abundant species along the 23-m isobath; white urchin and ridgeback prawn were the most abundant at 61 m; and ridgeback prawn, fragile sea urchin (*Allocentrotus fragilis*), pelagic red crab (*Pleuroncodes planipes*), gastropod (*Philine auriformis*), and shortkeel bay shrimp (*Neocrangon zacae*) were the most abundant species at 137 m. Other frequently collected epifauna included heart urchin (*Lovenia cordiformis*) and sea pen (*Virgularia bromleyi*) at 23 m; shrimp (*Crangon alaskensis*) and sea slug at 61 m; and a shrimp (*Spirontocaris holmesi*) and sea slug at 137 m (LACSD 1995, 2001).

Spatial patterns in the epifaunal community are primarily related to depth, sediment type, and (historically) effects from the wastewater discharge (Stull 1995). For example, many epifaunal species are more common in the organically enriched sediments of Redondo Canyon, the white urchin is most abundant on the shelf, and the shrimp *(Pandalus jordani)* is more common on the upper slope (Stull 1995). Populations of sea slug and *Platymera gaudichaudii* were enhanced near the outfall in the early 1970s, but decreased from 1978 to 1995. The abundance of several echinoderms, particularly white and fragile urchins and sea stars, has increased with time, likely as a result of improving quality of the wastewater discharges (LACSD 2001).

Fish Community

As described above, LACSD has been conducting trawl surveys of fish and invertebrates along the Palos Verdes shelf and slope since the early 1970s. The LACSD trawl catches have varied greatly over time. Conditions that may have influenced these changes include variations in water temperature, El Niño events, water mass movement (e.g., upwelling), kelp (*Macrocystis pyrifera*) coverage, food availability, habitat variability, and contaminants from the outfall (LACSD 2001).

During recent surveys (LACSD 2001), fish species that occurred in the greatest frequency throughout the study area included bigmouth sole (*Hippoglossina stomata*), plainfin midshipman (*Porichthys notatus*), and hornyhead turbot (*Pleuronichthys verticalis*). However, the most abundant species for all trawls combined were slender sole (*Eopsetta exilis*), Pacific sanddab (*Citharichthys sordidus*), yellowchin sculpin (*Icelinus quadriseriatus*), and Dover sole (*Microstomus pacificus*). Species that contributed to the largest percentage of biomass included slender sole, Dover sole, and California halibut (*Paralichthys californicus*). The highest abundance was observed in the fall, while diversity was greatest in the winter. The 23-m isobath had the lowest abundance and number of species observed during the 1995 survey (LACSD 1995). In general, flatfish were the most abundant fish in recent PVS study area trawls (LACSD 2001).

A few species—Dover sole, slender sole, Pacific sanddab, stripetail rockfish (*Sebastes saxicola*), plainfin midshipman, yellowchin sculpin, and speckled sanddab (*Citharichthys stigmaeus*)—contributed 64 percent of the total fish catch from 1973 to 1995. However, the dominance of these species and catches varied over time in apparent response to temperature changes. Some of the largest changes were attributed to El Niño events, which were accompanied by movement of new species northward and displacement of other species present in the area. Water temperature also had apparent effects on fish recruitment patterns. For example, there tended to be fewer rockfish and more flatfish when water temperatures were warmer (1980s and 1990s). Water depth also contributed to the distribution of fish species. For example, the blacktip poacher (*Xeneretmus latifrons*) was common in the 137-m trawls, and the greenblotched rockfish (*Sebastes rosenblatti*) was abundant in the Redondo Canyon. Speckled sanddabs were abundant at the 23-m isobath (LACSD 1995).

Fish species that were characteristic of the Palos Verdes outfall area during the early 1970s included shiner surfperch (*Cymatogaster aggregata*), curlfin sole (*Plueronichthys decurrens*), white seaperch (*Phanerodon furcatus*), English sole (*Pleuronectes vetulus*), and Dover sole. These species decreased from 1973 to 1995. In contrast, California tonguefish (*Symphurus atricauda*), hornyhead turbot, California halibut, and blackbelly eelpout (*Lycodopsis pacifica*) increased near the outfall (LACSD 2001). These species had been rare near the outfall in the early 1970s. This change in fish assemblages near the outfall is likely related to improved quality of the wastewater discharges, but warmer water temperatures may also be a factor (LACSD 1995, 2001).

The distribution of white croaker in relation to contaminants is of particular importance because of historically high levels of DDT bioaccumulation in the species, which has led to a commercial catch ban and recreational catch limits for this fish in the PVS study area. White croaker are captured sporadically, especially near the White Point outfall and Los Angeles Harbor (LACSD 1994). They are a dominant species in the Los Angeles Harbor and generally common in nearshore SCB waters (Cross and Allen 1994). White croaker was one of the top three fish species taken during LACSD (2001) trawls on the 23- and 61-m isobaths during the mid-1970s and early 1980s. However, croakers appear to be strongly influenced by ocean temperatures. However, during the ocean warming of the 1980s and 1990s, most species (including white croaker) decreased in abundance in the SCB (Herbinson et al. 2000). Local small-scale changes (such as at PVS trawl locations) may not match this overall trend precisely.

2.2.3.2 Hard-Bottom Habitats

Hard-bottom habitats include nearshore rocky intertidal to subtidal areas. Hard-bottom habitats in the PVS study area region are primarily documented from shore to approximately 20 m deep, although scattered outcrops and reefs also occur in some deeper shelf areas. Within these habitats, the most diverse communities, including numerous epifaunal invertebrate, fish, and plant (algae and surfgrass) species, are associated with kelp beds. These communities are generally shallower than the principal areas of chemical contamination on the PVS study area.

Kelp Community

The giant kelp (*Macrocystis pyrifera*) is a keystone species that provides refuge and a source of food for many fish and invertebrate species, although the extent of kelp beds has been extremely variable over time. In 1911 surveys, kelp canopy coverage near the PVS study area was estimated to be over 1,500 acres (LACSD 1995). By the late 1950s, giant kelp had disappeared from PVS rocky subtidal areas. This was attributed in part to wastewater discharges, which introduced toxicants, buried the substrate, and reduced light penetration (thus restricting photosynthesis) (Stull 1995). Transplantation efforts helped to reestablish kelp in the vicinity of the Palos Verdes peninsula, although the kelp beds suffered severe damage during winter storms in 1983 and 1988. Kelp beds near the PVS study area were estimated at 1,124 acres in 1989, but declined to 300 acres in 1993. This may have been caused by El Niño events and overgrazing by sea urchins (LACSD 1995). In addition, the Portuguese Bend landslide area also contributed to increased sedimentation and turbidity, which continued to impact some kelp bed populations (LACSD 2001; Stull 1995).

Common algae present within the kelp community include *Pterygophora californica*, *Eisenia arborea*, and *Agarum fimbriatum*, in addition to numerous species of red, green, and other brown algae.

Invertebrate Community

Common members of the invertebrate community associated with the kelp beds include three species of sea urchins that graze on kelp. These include the purple urchin (*Strongylocentratus purpuratus*), red urchin (*S. franciscanus*), and white urchin. Other species include various polychaetes, bivalves, sea stars, sea cucumbers, brittle stars, cnidarians (e.g., anemones and sea fans), and crustaceans.

Fish Community

Fish species common in the kelp beds include kelp bass, señorita (*Oxyjulis californica*), surfperch (e.g., *Embiotoca* species), half-moon (*Medialuna californiensis*), sculpins (Cottidae), blacksmith (*Chromis punctipinnis*), gobies (e.g., *Coryphopteru* species), and opaleye (*Girella nigricans*).

2.2.3.3 Pelagic Habitats

The pelagic environment includes the water column from near the bottom to the sea surface; it provides habitat for many species of plankton, invertebrates, fish, seabirds, and marine mammals.

Plankton Community

Phytoplankton and zooplankton communities in the Palos Verdes region exhibit species composition, abundance, and biomass changes that reflect seasonal variations in physical and

chemical parameters (e.g., light, temperature, salinity, available nutrients, upwelling, current regimes, and hydraulic conditions) (Cloern 1979). These community dynamics, in turn, have a strong influence on the feeding ecology of plankton-feeding organisms (e.g., many fish, invertebrates, and mysticete whales). Coastal phytoplankton typically observed in marine habitats include diatoms (Bacillariophyceae), coccolithophores (Haptophyta), dinoflagellates (Pyrrophyta), silicoflagellates (Chrysophyta), cryptomonads (Cryptophyceae), and green algae (Chlorophyceae). Because phytoplankton growth depends on light availability, they occur in the photic zone, generally the top 20 to 50 m in the water column in the SCB (Mullin 1986). Typically, dinoflagellates are the predominant phytoplankton in nearshore coastal waters, while diatoms are generally abundant in deeper, colder waters (Eppley et al. 1978).

Predominant zooplankton off southern California include various copepods, euphausiids, juvenile and larval fish and invertebrates, salps, chaetognaths (arrow worms), and pelagic molluscs. The abundance of zooplankton reflects changes in abundance of the phytoplankton. Salps and pelagic mollusks are generally observed offshore, while other zooplankton types are found both nearshore and offshore. Juvenile fish and crustaceans tend to be more prevalent in nearshore areas.

Fish Community

Pelagic fish include many relatively large, far-ranging species (e.g., tuna, mackerel, bonito, and barracuda) but also numerous small "bait" or forage fish (e.g., anchovies, sardines, and smelts). Mackerel and forage fish, in particular, can be abundant in the Palos Verdes region, although their transient nature and feeding ecology make them difficult to evaluate for the present risk evaluation. Analysis of the 1995-1996 CDFG recreational catch block data for Palos Verdes (Catch Blocks 719 and 720) indicates the most commonly caught pelagic fishes include ocean whitefish (*Caulolatilus princeps*), chub mackerel (*Scomber japonicus*), Pacific barracuda (*Sphryraena argentea*), Pacific bonito (*Sarda chiliensis*), and jack mackerel (*Trachurus symmetricus*). Other pelagic fish species collected in the Palos Verdes region include yellowtail (*Seriola lalandi*), skipjack tuna (*Euthynnus pelamis*), yellowfin tuna (*Thunnus albacares*), and a variety of sharks, such as thresher (*Alopias vulpinus*), blue (*Prionace glauca*), shortfin mako (*Isurus oxyrinchus*), and soupfin (*Galeorhinus zyopterus*).

Marine and Aquatic-Feeding Birds

Many migratory and resident marine and aquatic-feeding bird species occur in the vicinity of the PVS study area along the shelf and slope. These include a variety of pelagic, shelf, and nearshore species, many of which are far ranging throughout the Bight or are seasonally migratory. Some of the most common species are cormorants (*Phalacrocorax* spp.), California brown pelican, gulls (*Larus* spp.), terns (e.g., *Sterna* spp.), storm petrels (Hydrobatidae), murrelets and guillemots (Alcidae), and grebes (Podicipedidae). Juvenile bald eagles can disperse over large areas. For example, one fostered juvenile bald eagle was sighted in Klamath, California, more than 600 miles from its fledging site on Catalina Island (Sharpe 2003). However, the adult breeding bald eagles feed primarily on fish in the pelagic zone and have a relatively localized range. Populations that feed around Santa Catalina Island are generally found offshore from the PVS study area. Of these birds, those listed by the federal government and State of California as threatened or endangered species include bald eagle (threatened), peregrine falcon (recently delisted), and California brown pelican (endangered). (Note: Although the peregrine falcon was recently delisted by the federal government, it is still protected under a monitored status until 2005; see Table 2-1.) Brown

pelicans nest on the Northern Channel Islands (Anacapa and Santa Barbara Islands) offshore of southern California and feed in the vicinity of the PVS. Additional breeding colonies exist on Islas Los Coronados in Mexico off the Baja California peninsula. Historically, bald eagles nested on Santa Catalina, Santa Cruz, Anacapa, San Miguel, Santa Rosa, Santa Cruz, Santa Barbara, San Nicolas, San Clemente, and Los Coronados Islands (Kiff 1980). Peregrine falcons nested historically on Los Coronados, San Clemente, Santa Catalina, San Nicolas, Santa Barbara, Anacapa, Santa Cruz, Santa Rosa, and San Miguel Islands (Kiff 1980). Populations of peregrine falcons currently nest on the Channel Islands, including San Miguel, Santa Rosa, Santa Cruz, Anacapa, and Santa Barbara (Walton 1997).

Marine Mammals

Marine mammals (e.g., pinnipeds, dolphins, and toothed whales) are important predators of many fish species and some invertebrates, particularly in the pelagic zone, with many pinnipeds also feeding extensively in kelp bed habitats. In contrast, feeding by mysticete whales (e.g., blue whales, *Balaenoptera musculus*) is more exclusive on planktonic organisms in the pelagic zone.

Marine mammals occurring in the Palos Verdes region include both migratory and year-round residents (e.g., dolphins, porpoises, whales, seals, and sea lions). Seals and sea lions have well-established breeding colonies in several areas of the Bight, particularly in the northern Channel Islands, (e.g., San Miguel Island). Historically, sea otters (*Enhydra lutris nereis*) occurred in the region. Currently, there is a breeding population of transplanted sea otters near San Nicolas Island and a few transplanted individuals inhabiting areas around San Miguel Island (DeLong 2003). Some of the more common mammal species include harbor seals (*Phoca vitulina*), California sea lion, common dolphin (*Delphinus delphis*), Pacific white-sided dolphin (*Lagenorhynchus obliquedens*), bottlenose dolphin (*Tursiops truncatus*), and Minke whale (*Balaenoptera acutorostrata*) (Calambokidis and Francis 1994). Gray whales (*Eschrichtius robustus*) are sighted seasonally along the coast as they migrate between northern feeding and southern breeding and calving areas outside of the Bight; and blue whales have become more commonly sighted in the northern Bight.

Several threatened or endangered species of marine mammals are known to occur in the SCB. These include the humpback whale (*Megaptera novaeangliae*), blue whale (*Balaenoptera musculus*), fin whale (*Balaenoptera physalus*), sperm whale (*Physeter macrocephalus*), stellar sea lion (*Eumetopias jubatus*), sei whale (*Balaenoptera borealis*), and northern right whale (*Eubalaena glacialis).* Most of these species are uncommon in the Bight, and are found in deep waters offshore or near the Channel Islands, north of the Palos Verdes peninsula.

2.2.4 Current Ecological Setting

Numerous studies have shown that sediment and organism concentrations of tPCB and tDDT in the SCB are among the highest ever reported for any coastal marine ecosystem (USEPA 2000). However, within the PVS study area, concentrations of tDDT and tPCB in surface sediments and tissues of marine organisms have decreased since the 1970s. Surface sediment collected in 1977 from a 60-m site at the PVS area contained 175 mg/kg DDT DW. In 1985, surface sediment concentrations were 3 to 10 times lower than in 1977, but the DDT contamination still ranged from 1.4 to 91.6 mg/kg, despite the fact that DDT waste discharges had been curtailed for over 15 years. Generally, fish from the PVS study area are more contaminated, by a factor of 10 to $ 100, than macroinvertebrates from this area.

Coastal pelagic-zone fish from the PVS study area are less contaminated than croaker, flatfish (Dover sole and sanddabs), and shark (Mearns et al. 1991). Despite decreases in fish tissue concentrations since the 1970s, recent (1997) studies have confirmed that concentrations of tDDT in whole fish (sanddabs) collected from the PVS were up to three orders of magnitude higher than those in fish from other parts of the SCB (Allen et al. 1999).

2.3 Chemicals of Potential Ecological Concern

The COPECs for the PVS study area were identified from evaluation of existing data and discussions with USEPA. The available data were evaluated and reduced based on requirements for the ERA, and COPECs were identified for each exposure medium. Evaluation of all chemicals that have been detected in abiotic and biotic media in the PVS study area was not conducted as part of this ERA based on prior discussions with USEPA. The rationale for limiting the COPECs being evaluated is presented in subsection 2.3.3.

2.3.1 Data Sources

The goal for this ERA was to summarize data from as many different sources as practical, some throughout the SCB, for the period 1990–2003 (birds summarized for 1985-2000). Although there was a desire to carefully and fully characterize the PVS study area for exposure to resident invertebrates and fish, data were also needed to characterize SCB-wide exposure for wider-ranging, higher-order consumers (marine birds and mammals). The years of record were designed to capture recent historical conditions that would serve as a summary of current conditions. Data on sediment, mussels, fish, marine birds, and mammals were used in the assessment. All SCB contaminant data within the 1990–2003 period and avian data within the 1985-2000 period were used to estimate average tPCB or tDDT concentrations by location. Other data (abundance, distribution, benthic indices) were also collected within the same general time period. The PVS study area is particularly data-rich, but data were used from throughout the SCB to estimate the contaminant concentrations of sediment and the distribution and exposure of higher-order consumers from areas beyond the PVS. Data sources by media are listed in Table 2-2.

The sediment and fish contamination data are clustered in the PVS area with variable amounts of information available at more distant SCB locations. In particular, the LACSD data sets gathered as part of National Pollutant Discharge Elimination System (NPDES) monitoring and special studies have provided a wealth of information about conditions off the Palos Verdes peninsula, with some data available from every year in the 1990–2003 period. In contrast, the data of greatest spatial coverage came from the interagency-sponsored studies known as Bight 94 and Bight 98. Both yielded SCB-wide sediment concentrations, while Bight 98 provided a comprehensive dataset of flatfish contamination (as whole-body samples) throughout the SCB.

The USGS and LACSD sediment core results were used to create surface-sediment (0-15 cm) and deep-sediment (15–30 cm) composite contaminant concentration averages. The surface values were incorporated into our SCB-wide summary of surface-sediment quality, while the deeper-sediment results were presented by station location within the PVS to examine possible relationships between sediment contaminant concentrations and those in the tissues of resident fish.

TABLE 2-2
Use of Available Data for the Palos Verdes Shelf Study Area
Palos Verdes Shelf Ecological Risk Assessment

Medium/Tissue	Parameters	Date Collected	Source	Use in Draft Ecological Risk Evaluation Report (USEPA 2000)		Use in this ERA (CH2M HILL)
				Used	Justification	
Sediments						
Box core/gravity core	tDDT (sum of 2,4'-DDE, 4,4'-DDE, 2,4'-DDD, 4,4'-DDD, 2,4'-DDT, 4,4'-DDT, and DDMU); tPCB (sum of PCB congeners 8,18, 28, 44, 52, 66, 101, 105, 118, 128, 138, 153, 170, 180, 187, 195, 206, and 209 multiplied by 2); Total Organic Carbon (%)	1992/1993	Lee 1994	Yes	Considered the most thorough sampling of the EA sediment; used to calculate porewater concentrations using equilibrium partitioning.	Used to calculate porewater concentrations using equilibrium partitioning; used to determine whether trends exist 1990-2000; used as exposure-point concentrations (EPCs).
Gravity core	tDDT; tPCB	1990, 1991	Anderson et al. 1998	No	Not used.	Used as EPCs.
Gravity core	DDT isomers; PCB congeners (8, 18, 28, 44, 52, 66, 101, 105, 110/77, 118, 126, 128, 138, 153, 170, 180, 187, 195, 206)	1990, 1991	NOAA 1994	No	Not used.	Used as EPCs.
Gravity core	DDT isomers; PCB congeners (8, 18, 25, 44, 52, 66, 87, 101, 105, 118, 126, 138, 153, 170, 180, 187, 195, 206, 209)	1992	NOAA 1994	No	Not used.	Used as EPCs.
Surface sediments (Van Veen Grab—upper 2 cm) and gravity core	tDDT; tPCB (sum of Aroclors); Total Organic Carbon (%) (not consistently collected)	1973, 1980, 1985, 1988, 1990-2001	LACSD 1990-2001	Partial, 1992 and 1994 only	Used to fill gaps in USGS data; 1996 data compared to 1992 and 1994 data and determined to be similar; therefore, 1996 data were not used.	Used 1991-2001 core data only; Used to calculate porewater concentrations using equilibrium partitioning; used to determine whether trends exist 1990-2000; used as EPCs. Grab samples considered less accurate and unnecesary to characterize chemistry and therefore not used in this assessment.

TABLE 2-2
Use of Available Data for the Palos Verdes Shelf Study Area
Palos Verdes Shelf Ecological Risk Assessment

Medium/Tissue	Parameters	Date Collected	Source	Use in Draft Ecological Risk Evaluation Report (USEPA 2000)		Use in this ERA (CH2M HILL)
				Used	Justification	
Surface sediments (Van Veen Grab—upper 2 cm)	tDDT; Total Organic Carbon (%)	1994, 1998	Noblet et al. 2002, Schiff and Gossett 1997	No	Only 3 stations located in PVS area.	Used to fill in data gaps; used as EPCs. Pace laboratory data removed due to QA/QC concerns.
Box cores	4,4'-DDE	2002	SAIC 2002	No	Not Used.	Used to fill in data gaps; used as EPCs.
Vibracores	DDT isomers (4,4' isomers only)	2000	USACE 2000	No	Not Used.	Used to fill in data gaps; used as EPCs.
Vibracores	tDDT; tPCB	2002	Fredette et al. 2002	No	Not Used.	Used to fill in data gaps; used as EPCs.
Surface sediments (Van Veen Grab—upper 2 cm)	DDT isomers; PCB congeners (18, 28, 52, 49, 44, 37, 74, 70, 66, 101, 99, 119, 81/87, 110/77, 151, 123/149, 118, 114, 153, 168/105, 138, 158, 126, 187, 183, 128/167, 177, 156, 157/200, 180, 169, 170, 201, 189, 194, 206).	1997	SCCWRP 1999-2000	No	Not used.	Used to fill in data gaps; used as EPCs.
Invertebrates						
California mussel	DDE, tPCB	1985, 1986	Risebrough 1986	No		Not used. Not enough data to evaluate as a dietary item. Benthos community data and toxicity estimates used instead.
Yellow crab—muscle	tDDT; PCB congeners and Aroclors (1242, 1254, 1260); DDMU	1990	Connolly and Glaser 1994	No		Not used. Not enough data to evaluate as a dietary item. Benthos community data and toxicity estimates used instead.

TABLE 2-2
Use of Available Data for the Palos Verdes Shelf Study Area
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Medium/Tissue	Parameters	Date Collected	Source	Use in Draft Ecological Risk Evaluation Report (USEPA 2000)		Use in this ERA (CH2M HILL)
				Used	Justification	
Red urchin—gonads	tDDT; 2,4'-DDD, 2,4'-DDE, 2,4'-DDT, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT; tPCB (Aroclors 1242, 1254, 1260)	1985, 1988, 1990, 1994	LACSD 1985-2001	No		Not used. Not enough data to evaluate as a dietary item. Benthos community data and toxicity estimates used instead.
Rock crab—muscle	tDDT; 2,4'-DDD, 2,4'-DDE, 2,4'-DDT, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT; tPCB (Aroclors 1242, 1254, 1260)	1991	LACSD 1985-2001	No		Not used. Not enough data to evaluate as a dietary item. Benthos community data and toxicity estimates used instead.
Misc. species	tDDT; 2,4'-DDD, 2,4'-DDE, 2,4'-DDT, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT; tPCB (Aroclors 1242, 1254, 1260)	Pre-1991	LACSD 1985-2001	No		Not used. Not enough data to evaluate as a dietary item. Benthos community data and toxicity estimates used instead.
Spiny lobster	tDDT; 2,4'-DDD, 2,4'-DDE, 2,4'-DDT, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT; tPCB (Aroclors 1242, 1254, 1260)	1993	LACSD 1985-2001	No		Not used. Not enough data to evaluate as a dietary item. Benthos community data and toxicity estimates used instead.
Purple sea urchin	sperm cell fertilization test—interstitial water toxicity test	1992 and 1993	Bay et al. 1994	Yes		Not used. Removed as a line of evidence because of questionable test results.
Amphipod—*Rhepoxynius abronius*	10-day acute solid phase toxicity test	1992 and 1993	Bay et al. 1994	Yes		Used as a line of evidence in describing risk to sediment fauna.

TABLE 2-2
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Medium/Tissue	Parameters	Date Collected	Source	Use in Draft Ecological Risk Evaluation Report (USEPA 2000)		Use in this ERA (CH2M HILL)
				Used	Justification	
Amphipod—*Rhepoxynius abronius*	10-day acute solid phase toxicity test	1992 and 1993	EVS 1994	Yes		Used as a line of evidence in describing risk to sediment fauna.
Polychaete—*Neanthes arenaceodentata*	20-day acute and 120-day chronic solid phase toxicity test	1992 and 1993	EVS 1994, Chapman 1996	Yes		Used as a line of evidence in describing risk to sediment fauna.
Misc. mussels and other invertebrate species	DDE, tDDT, tPCB	1971-1993	Various (California Mussel Watch Program, NOAA's mussel watch, and NOAA's contaminant trends reports)	No		Not used.
Benthic macroinvertebrates	laboratory and field toxicity tests—established sediment effect thresholds for DDT, DDE, DDD, tDDT, Aroclor 1254, and tPCB		MacDonald 1997	Yes	Used as sediment benchmarks.	Used as sediment benchmarks.
Macroinvertebrates	species ID, enumeration	1992	LACSD 1985-2001	Yes		Used for benthic community metrics.
Macroinvertebrates	species ID, enumeration	1994	Schiff and Gossett 1997	Yes		Used to describe physical characteristics of sediments.
Macroinvertebrates	Benthic Response Index	1990-1996	LACSD 1985-2001, Schiff and Gossett 1997	Yes	Used as sediment benchmarks.	Cited. Used as measure of community response.
Fish (Other than LACSD and SCCWRP, all data are derived from SCB NRDA data files, thus parameters are limited to those listed.)						
White croaker, Dover sole, kelp bass—muscle	tDDT, tPCB	1990-1991	Connolly and Glaser, 1994	No	Not used.	1990-1991; Used as EPC and dietary item;

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Medium/Tissue	Parameters	Date Collected	Source	Use in Draft Ecological Risk Evaluation Report (USEPA 2000)		Use in this ERA (CH2M HILL)
				Used	Justification	
White croaker—muscle	tDDT; 2,4'-DDD, 2,4'-DDE, 2,4'-DDT, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT; tPCB (Aroclors 1242, 1254, 1260)	1985, 1988, 1990, 1992, 1994, 1995, 1996, 1997, 1999	LACSD 1985-2001	Partial, 1992 only	These data were used, but only to compare to concentrations predicted from a literature-based BSAF; Data also used to calculated site-specific % lipid.	1990-2001, Used as EPC and dietary item; Converted to whole body concentrations.
White croaker—muscle	tDDT/tPCB	1987	Pollock et al. 1991	No		Not used (collected before 1990)
White croaker—muscle	tDDT; PCB congeners and Aroclors (1242, 1254, 1260); DDMU	1990	SMBRP (Santa Monica Bay Restoration Project) 1992	No	Not used.	1990, Used as EPC and dietary item; Converted to whole body concentrations.
White croaker	evaluation of reproductive impairment in female white croaker	1985 and 1988	Hose and Cross 1997	No	Mentioned in text, but not discussed in the weight of evidence.	Not used (collected before 1990).
Dover sole—muscle	tDDT; 2,4'-DDD, 2,4'-DDE, 2,4'-DDT, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT; tPCB (Aroclors 1242, 1254, 1260)	1985,1990-1993, 1995, 1996, 1999	LACSD 1985-2001	Yes, 1992 only	These data were used, but only to compare to concentrations predicted from a literature-based BSAF; Data also used to calculate site-specific % lipid.	1990-2001, Used as EPC and dietary item; Converted to whole body concentrations.
Kelp bass—muscle, liver	tDDT; 2,4'-DDD, 2,4'-DDE, 2,4'-DDT, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT; tPCB (Aroclors 1242, 1254, 1260)	1985, 1988, 1990, 1992, 1995, 1996	LACSD 1985-2001	Yes	Used as a measure of exposure for aquatic-feeding birds. Historical average concentration used.	1990-2001, Used as EPC and dietary item; Converted to whole body concentrations.
Kelp bass—muscle only	DDT, tPCB	1987	Pollock et al. 1991	No		Not used.

TABLE 2-2
Use of Available Data for the Palos Verdes Shelf Study Area
Palos Verdes Shelf Ecological Risk Assessment

Medium/Tissue	Parameters	Date Collected	Source	Use in Draft Ecological Risk Evaluation Report (USEPA 2000)		Use in this ERA (CH2M HILL)
				Used	Justification	
Kelp bass—muscle, liver	DDE, tPCB	1985	Risebrough 1987	No	Also collected black perch and white croaker, but not on PVS.	Not used (collected before 1990).
Kelp bass—liver only	DDE, tPCB	1992, 1993	Spies et al. 1994	No	Collected to evaluate reproductive effects for NRDA, but not used in SERE weight of evidence.	Not used – need muscle or whole body data.
Kelp bass	evaluation of reproductive impairment	1985 and 1988	Hose et al. 1989	No	Mentioned in text, but not discussed in the weight of evidence.	Not used (collected before 1990).
Black surfperch—muscle	tDDT; 2,4'-DDD, 2,4'-DDE, 2,4'-DDT, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT; tPCB (Aroclors 1242, 1254, 1260)	1985, 1990, 1996	LACSD 1985-2001	No		1990-2001, Muscle only used as EPC and dietary item; Converted to whole body concentrations.
Surfperch—muscle	DDT, tPCB	1987	Pollock et al. 1991	No		Not used (collected before 1990).
Sanddabs: Pacific/longfin—liver (1 station only)	2,4'-DDT, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT; PCB congeners	1994	Schiff and Gossett 1997	No		Not used – need muscle or whole body data.
Sanddabs—muscle	DDT, tPCB	1987	Pollock et al. 1991	No		Not used (collected before 1990).
Other species: sculpin, Pacific bonito, mackerel, queenfish, corbina, rockfish—muscle	DDT, tPCB	1987	Pollock et al. 1991	No		Not used (collected before 1990).
California halibut	tDDT; 2,4'-DDD, 2,4'-DDE, 2,4'-DDT, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT; tPCB (Aroclors 1242, 1254, 1260)	1991	LACSD 1985-2001	No		Not used (other surrogates selected).

TABLE 2-2
Use of Available Data for the Palos Verdes Shelf Study Area
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Medium/Tissue	Parameters	Date Collected	Source	Use in Draft Ecological Risk Evaluation Report (USEPA 2000)		Use in this ERA (CH2M HILL)
				Used	Justification	
Turbot (and diet)	DDT, tPCB	1997	Zeng and Tran 2002	No	Example of fish uptake from invertebrates.	Not used –example only.
White croaker	tDDT; tPCB	1992; 1997	Anderson et al. 1998	No	Not used.	Muscle only used as EPC and dietary item; Converted to whole body concentrations.
White croaker, kelp bass, speckled sanddab	DDT isomers; tPCB as Aroclors (1248, 1254, and 1260)	1999-2000	OEHHA 2001	No	Not used.	Muscle only used as EPC and dietary item; Converted to whole body concentrations.
Kelp bass	tDDT; tPCB	1994	Costa et al. 1994	No	Not used.	Muscle only used as EPC and dietary item; Converted to whole body concentrations.
White croaker	tDDT; tPCB	1999	Hansen and Associates 2000	No	Not used.	Muscle only used as EPC and dietary item; Converted to whole body concentrations.
White croaker	DDT isomers; tPCB as Aroclors (1016, 1221, 1232, 1242, 1248, 1254, 1260)	1990	SMBRP 1992	No	Not used.	Muscle only used as EPC and dietary item; Converted to whole body concentrations.
Dover sole, English sole, longfin sanddab, Pacific sanddab, speckled sanddab	DDT isomers	1994	Schiff and Gossett 1997	No	Not used.	Liver data not used.

TABLE 2-2
Use of Available Data for the Palos Verdes Shelf Study Area
Palos Verdes Shelf Ecological Risk Assessment

Medium/Tissue	Parameters	Date Collected	Source	Use in Draft Ecological Risk Evaluation Report (USEPA 2000)		Use in this ERA (CH2M HILL)
				Used	Justification	
Speckled sanddab, longfin sanddab, Pacific sanddab, slender sole, California halibut	tDDT; tPCB	1988	Noblet et al. 2002	No	Not used.	Whole body used as EPC and dietary item.
Mackerel, Pacific bonito, barracuda	tDDT; tPCB	1992-1993	HydroQual 1997	No	Not used.	Used for calculating pelagic fish tissue concentrations.
Various species	Diets	1980	Allen 2002	No	Diets can be used to establish site-specific BAFs.	Not used. BSAFs estimated directly.
Birds						
Bald eagle—brain, liver, serum	DDE	1993	Garcelon and Thomas 1997	No	Not used.	Not used. Due to the paucity of data, brain, liver, and serum, data were not evaluated.
Bald eagle—blood, egg, carcass	DDE	1981-1987	Garcelon et al. 1989	No	Not used.	Eggs (1985-1987) used as EPC.
Bald eagle—blood, egg	tDDT and tPCB; described reproductive impairment	1989-1993	Garcelon 1997, 1994c	No	Mentioned in the text, but not discussed in the weight of evidence or used for HQ evaluation.	Eggs (1985-1993) used as EPC; also used in weight of evidence as site-specific field study.
Bald eagle—egg	DDE and total tPCB	1989-2000	Garcelon 2000	No	Not available.	Eggs (1994-2000 only) used as EPC (1989-1993 egg data included in the Garcelon 1997 and 1994c dataset).
Peregrine falcon	reproductive impairment described		Walton 1997; Hunt 1997	No	Mentioned in the text, but not discussed in the weight of evidence.	Used in weight of evidence as site-specific field study.

TABLE 2-2
Use of Available Data for the Palos Verdes Shelf Study Area
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Medium/Tissue	Parameters	Date Collected	Source	Use in Draft Ecological Risk Evaluation Report (USEPA 2000)		Use in this ERA (CH2M HILL)
				Used	Justification	
Peregrine falcon—eggs	DDE and tPCB	1978-1980	Peakall et al. 1983	No	Not used.	Only used to calculate historical mean.
Peregrine falcon—eggs	tPCB, PCDDs, and PCDFs	1983-1988	Jarman et al. 1993a	No	Not used.	Eggs (1985-1988) used as EPC.
Peregrine falcon—eggs	tDDT, tPCB, chlordane, and related chemicals	1986-1989	Jarman et al. 1993b	No	Not used.	Eggs (1986-1989) used as EPC.
Peregrine falcon—eggs		1987-1992	Jarman 1994	No	Not used.	Eggs (1987-1992) used as EPC.
Peregrine falcon—eggs	tDDT and tPCB	1992-1993	Hunt 1994 and 1997	No	Mentioned in text, but not discussed in the weight of evidence.	Eggs (1992-1993) used as EPC.
Brown pelican—egg	tDDT	1969	Lamont et al. 1970	No	Not used.	Only used to calculate historical mean.
Brown pelican—carcass, breast muscle, wing muscle	DDE	1980	Ohlendorf et al. 1985	No	Not used.	Only used to calculate historical mean.
Brown pelican—egg	tDDT, tPCB, Dieldrin, and Endrin	1969-1971	Jehl 1973	No	Not used.	Only used to calculate historical mean.
Brown pelican—egg	tDDT, tPCB, HCB, mercury, lead, cadmium	1973-1975	Anderson et al. 1977	No	Not used.	Only used to calculate historical mean.
Brown pelican—egg	tDDT and tPCB	1977-1984, 1986-1990, 1992	Gress 1994	No	Not used.	Eggs (1986-1990, 1992) used as EPC.
Brown pelican—egg	tDDT and tPCB	1991-1992	Fry 1994	No	Not used.	Eggs (1991-1992) used as EPC.

TABLE 2-2
Use of Available Data for the Palos Verdes Shelf Study Area
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Medium/Tissue	Parameters	Date Collected	Source	Use in Draft Ecological Risk Evaluation Report (USEPA 2000)		Use in this ERA (CH2M HILL)
				Used	Justification	
Double-crested cormorant—eggs	tDDT and tPCB	1992	Peakall 1994	No	Mentioned in the text, but not discussed in the weight of evidence.	Eggs (1992) used as EPC. Also used in weight of evidence as a site-specific field study.
Double-crested cormorant— yolk lipids	DDE	1969	Gress et al. 1973	No	Not used.	Not used. Due to the paucity of data, yolk lipid data was not evaluated.
Double-crested cormorant—eggs	tDDT and tPCB	1991-1992	Fry 1994	No	Not used.	Eggs (1991-1992) used as EPC.
Bald eagle, peregrine falcon, and double-crested cormorant	exposure pathways analysis		HydroQual 1997	No	Mentioned in text, but not discussed in the weight of evidence.	Supports link between egg concentrations and sediment/fish concentrations at PVS.
Bald eagle, peregrine falcon, and double-crested cormorant	DDE, egg shell thinning		Kiff 1994	No	Not used.	Used in weight of evidence as a site-specific field study.
Brown pelican and double-crested cormorant	reproductive performance		Gress, 1994	No	Not used.	Used in weight of evidence as a site-specific field study.
Ashy storm petrel—eggs	DDE, egg shell thinning	1995-1997	Welsh et al. 2001	No	Not available	Ashy storm petrels are not a receptor that was evaluated in this ERA. However, this information was included in the site-specific studies section.

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Use of Available Data for the Palos Verdes Shelf Study Area
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Medium/Tissue	Parameters	Date Collected	Source	Use in Draft Ecological Risk Evaluation Report (USEPA 2000)		Use in this ERA (CH2M HILL)
				Used	Justification	
Aquatic birds (approximately 22 species)—eggs and whole body	tDDT and tPCB; estrogenic effects	1991-1992	Fry 1994	No	Not used.	Used whole body values as prey for peregrine and bald eagle. Also used as site-specific field study in the weight-of-evidence.
Marine Mammals						
California sea lions	exposure pathways analysis		HydroQual 1997	No	Mentioned in text, but not discussed in the weight of evidence.	Supports link between blubber concentrations and sediment/fish concentrations at PVS.
California sea lions— blubber, liver, brain	tDDE and tPCB	1970	DeLong et al. 1973	No	Not used.	Used to estimate concentrations in 1991 samples.
California sea lions— blubber, brain	DDT	1970	Le Boeuf and Bonnell 1971	No	Not used.	Not used. Adult female sea lions are evaluated in the risk assessment. These data present means for mixed sex/age class animals (i.e., males and females and adults and juveniles).
California sea lions— blubber	tDDT and tPCB	1991	Costa et al. 1994	No	Not used.	Exact values not used because require re-analysis. Trend used.
California sea lions— blubber, liver	DDE and tPCB	1972	Gilmartin et al. 1976	No	Not used.	Used to estimate concentrations in 1991 samples.

TABLE 2-2
Use of Available Data for the Palos Verdes Shelf Study Area
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Medium/Tissue	Parameters	Date Collected	Source	Use in Draft Ecological Risk Evaluation Report (USEPA 2000)		Use in this ERA (CH2M HILL)
				Used	Justification	
California sea lions— liver	tDDT, tPCB, lead, mercury, cadmium, and selenium	1973-1981	Britt and Howard 1983	No	Not used.	Not used. Due to the paucity of post-1981 data, liver data were not evaluated.
California sea lions— blubber	tDDT and tPCB	2000	Le Boeuf et al. 2002	No	Not available.	Adult female concentrations from the SCB were incorporated for comparison.
California sea lions— blubber	DDT	2000	Le Boeuf 2002	No	Not available.	Data for males only; however, was used for comparison to model output for bald eagle sea lion prey.
California sea lions— blubber, liver	tDDT, tPCB, and other organochlorines	1991-1997	Kajiwara et al. 2001	No	Not available.	Not used. All data from central and northern California.
California sea lion	DDT and population trends	1970, 1992	Lieberg-Clark et al. 1995	No	Not used.	Used in weight of evidence as site-specific field study.
California sea lion	DDT and population trends	1970, 1992	O'Shea and Brownell 1998	No	Not used.	Used in weight of evidence as site-specific field study.
Various marine mammals—various tissues	tDDT and tPCB, distribution, occurrence status, and trends of populations in SCB.	various	Calambokidis and Francis 1994	No	Not used.	Used to identify data sources

Benthic invertebrate and fish communities and sediment toxicity data were summarized from LACSD monitoring results to evaluate effects of exposure to the benthos and resident fish communities.

Recent marine bird and mammal studies were evaluated to document current levels of bioaccumulated COPECs and to present seasonal and home range information for use in the exposure model.

2.3.2 Data Evaluation and Reduction

This subsection discusses the spatial and temporal trends of selected sediment and biota data. Most of the data evaluated were collected in the area south and east of Palos Verdes Point and in areas west of White Point. Tables 2-3 and 2-4 summarize the reduced site-specific data that were considered for use in this evaluation.

The available data on contamination of biota include information on several fish species, mussels, and marine birds and mammals. LACSD has collected biota data at the site since the 1970s. Since 1988, specific protocols have existed for sampling fish/invertebrates as part of their bioaccumulation and benthic community studies; these protocols were further refined in 1994. Significant quantities of data are available for white croaker, Dover sole, and kelp bass. Tissue chemistry data have generally included DDT plus metabolites, and either PCB congeners or specific Aroclors. LACSD has also conducted benthic infaunal surveys at multiple locations on the PVS since the early 1970s. LACSD sediment chemistry data have been collected annually, and were used in this evaluation to determine whether tDDT concentrations in sediment have changed since 1991. Other sediment and fish tissue datasets are listed in Table 2-3. Sediment results are reported on a dry-weight basis and fish results on a wet-weight basis throughout this ERA.

Several studies were available from SCCWRP, including the Bight 94 Pilot Project (Schiff and Gossett 1997), the SCCWRP 1999-2000 Annual Report (SCCWRP 1999-2000), and the Bight 98 Report (Noblet et al. 2002). In the Bight 94 study, tDDT concentrations were calculated by analyzing surface sediment and fish livers for DDT isomers. In the SCCWRP 1996 study, tDDT and tPCB concentrations were calculated by analyzing surface sediment for DDT isomers and PCB congeners. In contrast, the Bight 98 study (Noblet et al. 2002) provided data for contaminants throughout the SCB for tDDT and tPCB in surface sediment and whole fish. Data from the Bight 98 study are of most interest, because they provide an SCB-wide distribution of contaminants in whole fish (sanddab guild) that are likely dietary items for marine birds and mammals.

Several smaller studies, or subsets of data from these studies, were available from the United States Army Corps of Engineers (USACE) (USACE 2000; Fredette et al. 2002) and SAIC (SAIC 2002). USACE collected sediment samples with a Vibra-core from five nearshore locations in Portuguese Bend and one location in Smuggler's Cove. The 4,4'-DDD, 4,4'-DDE, and 4,4'-DDT concentrations, as well as tDDT calculated from these isomers, were reported for the upper, middle, and lower subsamples from each core (USACE 2000). The Fredette et al. (2002) dataset contains total DDT and tPCB sediment concentrations from the USACE pilot capping study from the A-III borrow site. Total DDT concentrations were reported as the sum of 4,4'-DDD, DDE, and DDT. Total PCB concentrations were reported as the sum of Aroclors (1016, 1221, 1232, 1242, 1248, 1254, and 1260). SAIC (SAIC 2002)

TABLE 2-3
Site-Specific Sediment and Fish Data Sources
Palos Verdes Shelf—Ecological Risk Assessment

	No. Sediment Samples				No. Fish Tissue Samples			
Source	**Year(s)**	**tDDT**	**tPCB**	**Total**	**Year(s)**	**tDDT**	**tPCB**	**Total**
Anderson et al. 1998	1992; 1996-97	24 [a]	24 [b]	48	1992; 1997	4 [a]	4 [d]	8
Connolly and Glaser 1994	--	--	--	--	1990-91	105 [a]	104 [c]	209
Costa et al. 1994	--	--	--	--	1994	131 [d]	131 [d]	262
Fredette et al. 2002	2000	2[a]	2[c]	4	--	--	--	--
Hansen and Associates 2000	--	--	--	--	1999	104 [d]	104 [d]	208
LACSD 1990-2001	1991-2001	70 [e]	--	70	1990-2001	464 [d]	464 [d]	928
Lee 1994	1992	40 [a]	40 [b]	80	--	--	--	--
NOAA 1994	1990-91	20 [a]	20 [b]	40	--	--	--	--
NOAA unpubl.	1992	45 [a]	44 [b]	89	--	--	--	--
Noblet et al. 2002	1998	132 [a]	75 [b]	207	1998	198 [a]	114 [d]	312
OEHHA 2001	--	--	--	--	1999-2000	36 [a]	36 [c]	72
SAIC (2002)	2002	2[e]	--	2	--	--	--	--
SCCWRP 1999-2000	1997	25 [a]	25 [b]	50	--	--	--	--
Schiff and Gossett 1997 (Bight 94)	1994-95	167 [a]	--	167	--	--	--	--
SMBRP 1992	--	--	--	--	1990	59 [a]	60 [c]	119
USACE 2000	2000	6[d]	--	6	--	--	--	--
Total	**1990-2001**	**533**	**757**	**763**	**1990-2001**	**1,069**	**1,091**	**2,260**

[a] Reported as the sum of DDE, DDD, and DDT isomers
[b] Reported as the sum of PCB congeners
[c] Reported as the sum of Aroclors
[d] Reported as total
[e] Reported as 4,4'-DDE

collected two box cores from uncapped locations between the cells and the Y-shaped diffuser on the JWPCP outfall. Only 4,4'-DDE sediment concentrations were reported (no other isomers).

Additional sediment data have been collected in the SCB by the USGS (Lee 1994). In support of the NRDA, USGS collected extensive sediment profile data in 1992-93. These samples were analyzed primarily for DDT and its metabolites, as well as PCB congeners. Box cores and gravity cores were taken at 70 sites on the shelf, slope, and basin to collect data on contaminants, organic carbon, and texture. Box cores are generally shorter than gravity cores, but they provide an undisturbed examination of both the geologic and biologic structure of the sediments. Tissue data were also collected and analyzed as part of the NRDA (Costa et al. 1994; Connolly and Glaser 1994). Results were either reported as tDDT and tPCB in kelp bass fillets, or as DDT isomers and Aroclors in kelp bass, Dover sole, and white croaker.

TABLE 2-4
Summary of Available Marine Bird and Mammal Tissue Data
Palos Verdes Shelf Ecological Risk Assessment

Agency/Author	Year	Tissue	Number of Samples	Location	SCB Information Yes/No	Comments
Brown Pelican						
Lamont et al. 1970	1969	egg	10	AI	Yes	
Ohlendorf et al. 1985	1980	carcass, breast muscle, wing muscle	1	SCB	Yes	mean only
Jehl 1973	1969-1971		4	LCI, SBI, SMAI	Yes	mean only
Anderson et al. 1977	1973-1975	egg	4	AI	Yes	mean only
Gress 1994	1977-1984, 1986-1990, 1992	egg	10	CI	Yes	expert report, mean only
Fry 1994	1991-1992	egg	--	AI, SBI	Yes	expert report
Double-Crested Cormorant						
Gress et al. 1973	1969	yolk lipids	3	AI, LCI, SMAI	Yes	mean only
Fry 1994	1991-1992	egg	--	NC, OR, BC, SCB	Yes	expert report
Elliott et al. 1989	1970, 1979, 1985	egg	4	BC	No	mean only
Ohlendorf et al. 1982	1973-1976	egg	2	AK	No	mean only
Bald Eagle						
Garcelon and Thomas 1997	1993	brain, liver, serum	1	SCAI	Yes	
Garcelon et al. 1989	1981-1987	blood, egg, carcass	11	SCAI	Yes	
Garcelon 1994c	1989-1993	blood, egg	--	SCAI	Yes	expert report
Garcelon 2000	1989-2000	egg	26	SCAI	Yes	expert testimony; data provided by Beckye Stanton at the USFWS
Wiemeyer et al. 1984	1969-1979	egg	6	14 states	No	
Wiemeyer et al. 1993	1980-1984	egg	1	15 states	No	
Anthony et al. 1993	1980-1987	egg, blood, carcass	4	Columbia River Estuary	No	
Elliott et al. 1996a	1989-1994	carcass, liver	24	BC	No	

TABLE 2-4
Summary of Available Marine Bird and Mammal Tissue Data
Palos Verdes Shelf Ecological Risk Assessment

Agency/Author	Year	Tissue	Number of Samples	Location	SCB Information Yes/No	Comments
Elliott et al. 1996b	1990-1994	egg	4	Canada—Pacific coast	No	mean only
Elliott and Norstrom 1998	1991-1995	blood	8	BC—islands	No	mean only
Peregrine Falcon						
Peakall et al. 1983	1978-1980	egg	10	CA	Yes	
Jarman et al. 1993a	1983-1988	egg	6	CA	Yes	
Jarman et al. 1993b	1986-1989	egg	5	CA	Yes	
Jarman 1994	1987-1992	egg	--	CI, CA, OR	Yes	expert testimony
Hunt 1994	1992-1993	egg	--	AI, SCI, SRI, SMI	Yes	expert report
Nelson 1990	1968-1989	egg	2	Canada	No	
Other Birds						
Garcelon et al. 1989	1981-1987	breast muscle	11	SCAI	Yes	
Garcelon 1994c	1985-1993	breast muscle, whole body, egg	--	SCAI	Yes	expert report
Welsh et al. 2001	1995-1997	egg	23	SCI	Yes	abstract, mean only
Fry 1994	1991-1992	egg	--	SCB, GOC, MX, NC, OR, WA, BC	Yes	from NRDA
California Sea Lion						
DeLong et al 1973	1970	blubber, liver, brain	6	SMI	Yes	mean only
Le Boeuf and Bonnell 1971	1970	blubber, brain	3	SMI, ANI, CC	Yes	mean only
Costa et al. 1994	1991	blubber	--	SCB	Yes	from NRDA
Gilmartin et al. 1976	1970-1975	blubber, liver	20	SMI	Yes	
Britt and Howard 1983	1973-1981	liver	69	LA	Yes	
Le Boeuf et al. 2002	2000	blubber	7	SC	Yes	
Le Boeuf et al. 2002	2000	blubber	29	CC, NC	No	

TABLE 2-4
Summary of Available Marine Bird and Mammal Tissue Data
Palos Verdes Shelf Ecological Risk Assessment

Agency/Author	Year	Tissue	Number of Samples	Location	SCB Information Yes/No	Comments
Le Boeuf 2002	2000	blubber	6	SCAI	Yes	some overlap with Le Boeuf et al. 2002
Kajiwara et al. 2001	1991-1997	blubber, liver	15	CC, NC	No	
Bacon et al. 1992	1988	milk	1	CA	No	
Buhler et al. 1975	1970-1973	liver, muscle, brain, fat	4	OR	No	mean only
Lieberg-Clark et al. 1995	1988-1992	blubber	1	ANP, SCC	No	mean only
Hayteas and Duffield 1997	1991-1995	blubber	5	OR	No	

Notes:
-- part of NRDA dataset
AI Anacapa Island
ANP Año Nuevo Point, CA
BC British Columbia
CA California
CC Coastal California
CI Channel Islands
GOC Gulf of California
LA Los Angeles
LCI Los Coronados Island, Mexico
MTSF Monterey to San Francisco
MX Mexico
NA not available
NC Northern California
OR Oregon
SBEI San Benito Islands
SBI Santa Barbara Island
SC Southern California
SCAI Santa Catalina Island
SCB Southern California Bight
SCC Santa Cruz County, CA
SCI Santa Cruz Island
SMAI San Martin Island
SMI San Miguel Island
SRI Santa Rosa Island
WA Washington

Data were collected by the SMBRP to evaluate temporal trends in seafood contamination for the SCB by comparing contaminant concentrations in white croaker edible tissue to historical data (SMBRP 1992). Fillets from white croaker were analyzed after the skin was removed, and only these data were incorporated into the SCB fish tissue database. Total DDT was calculated as the sum of the DDT isomers, and tPCB was calculated as the sum of 7 Aroclors (1016, 1221, 1232, 1242, 1248, 1254, and 1260).

NOAA conducted sediment and fish surveys of coastal regions, including harbors, for their benthic surveillance and bioeffects assessment programs. The bioeffects assessment program (NOAA unpublished) was conducted to assess the potential adverse biological effects in San Pedro Bay, Los Angeles Harbor, Long Beach Harbor, Anaheim Bay, Alamitos Bay, and Huntington Harbor. Adverse effects were assessed by characterizing magnitude and relative spatial extent of toxicant-associated bioeffects in nearshore waters, determining relationships between concentrations and mixtures of sediment-associated toxicants and the occurrence and severity of biological effects, and by distinguishing more severely impacted sediments from less severely impacted sediments. NOAA's benthic surveillance program (NOAA 1994) monitored levels of contaminants in sediment and bottomfish and prevalences of pathological conditions in bottomfish along the U.S. coastline. Total DDT and tPCB surface sediment concentrations were calculated from DDT isomers and PCB congeners from data collected for both programs.

The California Fish Contamination Project, a joint project funded by the SWRCB and performed by OEHHA, collected fish tissue samples throughout coastal California waters for contaminant analysis (OEHHA 2001). Total DDT and tPCB were calculated from the sum of DDT isomers and PCB congeners measured in kelp bass, speckled sanddab, and white croaker muscle tissue.

The State of California, Bay Protection and Toxic Cleanup Program (BPTCP) database of sediment and fish tissue samples collected during the 1990s was used for inner harbor and marina sampling locations along the SCB coastal area, particularly in Los Angeles and Long Beach harbors. Sediment tPCB and tDDT were available, but only a limited number of fish samples were collected (Anderson et al. 1998).

One additional fish tissue dataset was supplied by Milton Love (Hansen and Associates 2000), a marine biologist at the University of California, Santa Barbara. White croaker fillets were collected in shallow water along the coast of the PVS during 1999. These samples were collected to determine whether tDDT and tPCB found in sediments from deeper waters on the shelf were accumulating in white croaker living in shallow water. Total DDT and tPCB in fish muscle tissue were reported.

Original datasets from the sources listed in Table 2-3 were reduced prior to performing statistical analyses. Data were reduced in steps, to prevent data loss. Sediment cores were normalized by either density or percentage of solids to combine multiple core segments into one of two general layers—surface sediments and deep sediments. Sediment cores containing strata between 0 and 16 cm deep were combined into a surface layer of 0 to 15 cm. Sediment cores containing strata from 16 cm to 31 cm were combined into a deeper layer of 15 to 30 cm. Data from grab samples were categorized as surface-layer sediments and, therefore, identified as belonging to the 0 to 15 cm sediment layer.

Before reducing the sediment dataset further, sediment tDDT data were analyzed to determine whether concentrations measured by LACSD had been decreasing between 1991 and 2001. At almost all stations, the sediment concentrations were unchanged. Geometric means were calculated for co-located sediment data, which were then incorporated into the sediment dataset.

Marine birds and mammals within the SCB have been studied since the early 1970s, whereas studies of raptors inhabiting the Channel Islands began in the late 1970s. The years, tissue types collected, number of samples collected, and sampling locations of these data are outlined in Table 2-4. Though many efforts have been made over the years to collect data for birds and mammals utilizing the SCB, the total number of samples is small compared to the sediment and fish samples collected over the same time period. To maximize the available data, the bird and mammal tissue residue data collected from 1985 to 2000 were considered to represent current conditions within the SCB. This assumption is supported by data trend analysis of the sediment, which indicated that conditions have remained relatively unchanged from the mid 1980s to the present (HydroQual, Inc. 1997). Therefore, data from 1985 to 2000 were used to derive current tissue concentrations within the SCB and outside the SCB. Data prior to 1985 were considered to be historic and were used to derive historic tissue concentrations within the SCB. Historic data outside the SCB were not evaluated.

Table 2-5 summarizes the DDT isomers and PCB congeners used to calculate tPCB and tDDT in sediment samples. Schiff and Gossett (1997), SCCWRP (2002), Anderson et al. (1998), LACSD (1990-2001), and USGS (Lee 1994) reported DDT and PCB concentrations as tDDT and tPCB. The NOAA benthic surveillance program and bioeffects assessment program reported DDT and PCB concentrations as the sum of DDT isomers and the sum of PCB congeners.

Table 2-6 summarizes the DDT isomers and PCB congeners used to calculate total tPCB and total tDDT in fish tissue samples. Anderson et al. (1998), Costa et al. (1994), LACSD (1990-2001), Hansen and Associates (2000), and SCCWRP (2002) reported DDT and PCB concentrations as total tDDT and total tPCB. Connolly and Glaser (1994), OEHHA (2001), and HydroQual, Inc. (1997) reported total DDT concentrations as the sum of DDT isomer concentrations, and reported tPCB concentrations as the sum of Aroclor concentrations. Schiff and Gossett (1997) also reported total DDT concentrations as the sum of DDT isomer concentrations, but reported tPCB concentrations as the sum of PCB congener concentrations.

2.3.2.1 Abiotic Data

The studies listed in Table 2-2 were selected to represent current conditions at the PVS from 1990 to 2003. Summary statistics for sediment data prepared from these studies are listed in Table 2-3, and raw data are supplied in Appendix A. Where possible, equilibrium partitioning was used to calculate porewater concentrations. Non-detects were incorporated into the respective datasets at half the detection limit. If detection limits were not available, the lowest detected value was assumed to be the detection limit.

Sediment Data

The spatial distributions of tDDT and tPCB in shelf and slope sediments were extensively evaluated by the USGS as part of the Trustees' site investigation for the NRDA; relevant findings and conclusions concerning the spatial distribution of contaminants are presented by Lee (1994). Sediment data collected by the USGS and LACSD provide the most complete

TABLE 2-5
DDT Isomers and PCB Congeners Used in Calculating tDDT and tPCB Concentrations in Sediment
Palos Verdes Shelf Ecological Risk Assessment

Source	DDT Isomers	PCB Congeners
Anderson et al. 1998	Reported as tDDT	Reported as tPCB
Fredette et al. 2002	4,4'-DDD; 4,4'-DDE; 4,4'-DDT	Aroclors 1016, 1221, 1232, 1242, 1248, 1254, 1260)
LACSD 1990-2001	Reported as tDDT	Reported as tPCB
Lee 1994	Reported as tDDT	Reported as tPCB
NOAA (1994)	2,4'-DDE; 4,4'-DDE; 2,4'-DDD; 4,4'-DDD; 2,4'-DDT; 4,4'-DDT	8, 18, 28, 44, 52, 66, 101, 105, 110/77, 118, 126, 128, 138, 153, 170, 180, 187, 195, 206
NOAA (unpublished)	2,4'-DDE; 4,4'-DDE; 2,4'-DDD; 4,4'-DDD; 2,4'-DDT; 4,4'-DDT	8, 18, 25, 44, 52, 66, 87, 101, 105, 118, 128, 138, 153, 170, 180, 187, 195, 206, 209
SAIC (2002)	4,4'-DDE	Not measured
SCCWRP 1999-2000	2,4'-DDE; 4,4'-DDE; 2,4'-DDD; 4,4'-DDD; 2,4'-DDT; 4,4'-DDT	18, 28, 52, 49, 44, 37, 74, 70, 66, 101, 99, 119, 81/87, 110/77, 151, 123/149, 118, 114, 153, 168/105, 138, 158, 126, 187, 183, 128/167, 177, 156, 157/200, 180, 169, 170, 201, 189, 194, 206
Noblet et al. 2002	Reported as tDDT	Reported as tPCB
Schiff and Gossett 1997 (Bight 94)	2,4'-DDE; 4,4'-DDE; 2,4'-DDD; 4,4'-DDD; 2,4'-DDT; 4,4'-DDT	Not measured
USACE 2000	4,4'-DDD; 2,4'-DDT; 4,4'-DDT	Not measured

Notes:
DDD 1,1-dichloro-2,2-bis (p-chlorophenyl) ethane
DDE 2,2-bis (p-chlorophenyl) 1,1-dichloroethane
DDT 1,1,1-trichloro-2,2-bis (p-chlorophenyl) ethane
Numbers not listed as Aroclors represent PCB congeners

TABLE 2-6
DDT Isomer and PCB Congeners Used in Calculating tDDT and tPCB Concentrations in Fish Tissue
Palos Verdes Shelf Ecological Risk Assessment

Source	DDT Isomers	PCB Congeners
Anderson et al. 1998	Reported as tDDT	Reported as tPCB
Costa et al. 1994	Reported as tDDT	Reported as tPCB
LACSD 1990-2001	Reported as tDDT	Reported as tPCB
Hansen and Associates 2000	Reported as tDDT	Reported as tPCB
Connolly and Glaser 1994	2,4'-DDE; 4,4'-DDE; 2,4'-DDD; 4,4'-DDD; 2,4'-DDT; 4,4'-DDT	Aroclors 1016, 1221, 1242, 1254, 1260
OEHHA 2001	2,4'-DDE; 4,4'-DDE; 2,4'-DDD; 4,4'-DDD; 2,4'-DDT; 4,4'-DDT	Aroclors 1248, 1254, 1260
Noblet et al. 2002	Reported as tDDT	Reported as tPCB
SMBRP 1992	2,4'-DDE; 4,4'-DDE; 2,4'-DDD; 4,4'-DDD; 2,4'-DDT; 4,4'-DDT	Aroclors 1016, 1221, 1232, 1242, 1248, 1254, 1260

Notes:
DDD 1,1-dichloro-2,2-bis (p-chlorophenyl) ethane
DDE 2,2-bis (p-chlorophenyl) 1,1-dichloroethane
DDT 1,1,1-trichloro-2,2-bis (p-chlorophenyl) ethane
Numbers not listed as Aroclors represent PCB congeners

spatial coverage of the PVS through 2001. The PCB congeners and DDT isomers summarized in this ERA are given in Table 2-5 for sediment and Table 2-6 for fish.

The focus of evaluating sediment and TOC data for exposure pathway analysis is to obtain a spatial picture of site contamination in the mixed sediment layer. The mixed surface-sediment layer represents the biologically active zone (BAZ), that portion of the sediment where benthic infauna are most abundant and where the greatest likelihood exists for exposure of benthic organisms and contaminant transfer up the food web. Sediment sampling locations for tDDT and tPCB concentrations in the BAZ (surface layer, 0-15 cm) are presented in Figure 2-1. Figure 2-2 and Figure 2-3 present an SCB-wide and close-up view of the greater PVS area that show the spread of tDDT in relation to depth contours. Figures 2-2 and 2-4 expand the depiction to encompass samples in water up to 800 m in depth in the PVS study area. Figures 2-3 and 2-5 limit the characterization to ecological risk determination cut-off at 200 m in water depth. All stations sampled showed some level of contamination. While the area of greatest contamination is in the vicinity of the LACSD outfalls, contaminant concentrations greater than 1 mg/kg DW cover large areas of the PVS, and some sediments contain contamination at levels greater than 10 mg/kg DW. Concentration gradients decline steeply away from the outfall toward nearshore areas. Toward the northwest, the concentration gradient is more gradual, and large areas, including a region beyond Point Vicente, are contaminated at levels greater than 4 mg/kg DW, reflecting the direction of the predominant current flow. Figure 2-4 and Figure 2-5 present the PCB contamination in the SCB and over the greater PVS area. The areas near the outfalls have the highest concentrations (greater than 1 mg/kg DW), while concentrations less than 1 mg/kg DW cover most of the remaining shelf and slope. The distribution of TOC in sediment is shown in Figure 2-6. Note the higher concentrations in EA sediments near the PVS outfalls.

USGS (Lee 1994) presented data on the peak concentrations (from 4 cm increments) of tDDT and tPCB from sediment cores. The highest concentrations are typically centered on the 60-m isobath, to the southeast of the White Point outfalls, and extend several kilometers to the northwest over the shelf and west over a portion of the slope. The most contaminated sediments (tDDT concentrations greater than 200 mg/kg DW) occur in sediment layers greater than 30 cm below the surface. The 4,4'-DDE isomer is the predominant isomer on the PVS, representing 70 percent of the total detected DDT.

Total PCB concentrations are also highest in areas adjacent to the outfalls. Peak total tPCB greater than 20 mg/kg DW occur at sediment depths greater than 30 cm. There is a much smaller gradient in PCB concentrations as compared to that for total DDT.

Contaminant concentration data averaged over the uppermost 15 cm show a strong relationship between concentrations in both the surface and the deeper, contaminated sediments, reflecting the fact that contaminants at depth are being remobilized to the surface. The BAZ is assumed to be the same as the depth of the mixed layer, which is defined as 15 cm. The assumption of a 15-cm depth for the BAZ was made by the U.S. Army Corps of Engineers, Waterways Experiment Station (WES) (Palermo et al. 1999), in their modeling of diffusive flux at the site, and the value is used for consistency with that modeling effort. WES consulted with a local panel of experts familiar with bioturbation processes in the PVS region. It was concluded that most benthic organisms inhabiting the PVS are "shallow" bioturbators that dwell in the uppermost 15-20 cm of the sediment

column, within which sediment mixing largely occurs. Sediment mixing to 30 cm also occurs, at least occasionally, although the rates are expected to be relatively low. This risk evaluation recognizes the potential for transport of sediments from below the 15-cm horizon to the surface caused by bioturbation activities of deep burrowers. However, the impacts of such activities are assessed through characterization of sediment concentrations, effects to benthos, and bioaccumulation in fish.

Deeper sediment layers (15-30 cm) were evaluated from the USGS and LACSD sediment core databases for their ability to explain variability in tissue concentrations in benthic fish (Appendix C). Surface (0-15 cm) and deeper (15-30 cm) layers were found to be tightly correlated in DDT and PCB contamination. Regression relationships between fish and surface sediments explained more variation than did those based on deeper sediments. Thus, modeled concentrations of deeper sediments and modeled concentrations of surface sediment, not used separately as part of the ecological exposure and risk estimates, are presented in Table 2-7.

TABLE 2-7
Sediment Summary Statistics from Gridded GIS Output for the PVS Study Area (less than 200 m in water depth)
Palos Verdes Shelf Ecological Risk Assessment

Sediment Layer	Analyte	Geometric Mean (mg/kg DW)	Standard Deviation	Minimum	Median	Maximum	95% UCL of the Geometric Mean
0-15 cm	tDDT	0.70	11.44	<0.001	1.053	64.13	14.90
15-30 cm		0.50	34.52	0.004	1.205	209.31	--[a]
0-15 cm	tPCB	0.06	22.08	<0.001	0.082	2.53	--[a]
15-30 cm		0.12	20.39	0.013	0.095	8.32	--[a]

Notes:
[a] Due to the large standard deviation, calculation of the 95% UCL of the geometric mean was not appropriate.
Mean and standard deviation are calculated from ln-transformed data.
95% UCL = 95th Upper Confidence Limit of the geometric mean.

Sediment Data Trends

A separate analysis of LACSD surface-layer tDDT was conducted to determine whether COPEC concentrations increased, decreased, or remained unchanged in recent years (1991-2001). Previous investigations (HydroQual 1997; USEPA 2000) demonstrated that COPEC concentrations remained relatively static from 1986 through 1995. However, the period from 1995-2001 was not previously evaluated. Figure 2-7 summarizes tDDT concentrations in surface-layer sediment at LACSD sediment sampling stations from 1991 through 2001.

Data from 48 LACSD sediment stations were plotted to determine whether trends in sediment tDDT concentrations existed between 1991 and 2001. Twenty-nine of these stations had less than three samples, and were not evaluated for trends. Nine of the remaining stations (3BC, 5CD, 5DC, 6B, 6BC, 6DC, 7CD, 8B, and 9C) had 3 samples. Seven of the remaining 10 stations had 4 samples (1C, 3C, 6C, 6CD, 7C, 8CD, 9CD) whereas the rest (4C, 5C, 8C) had 5 samples.

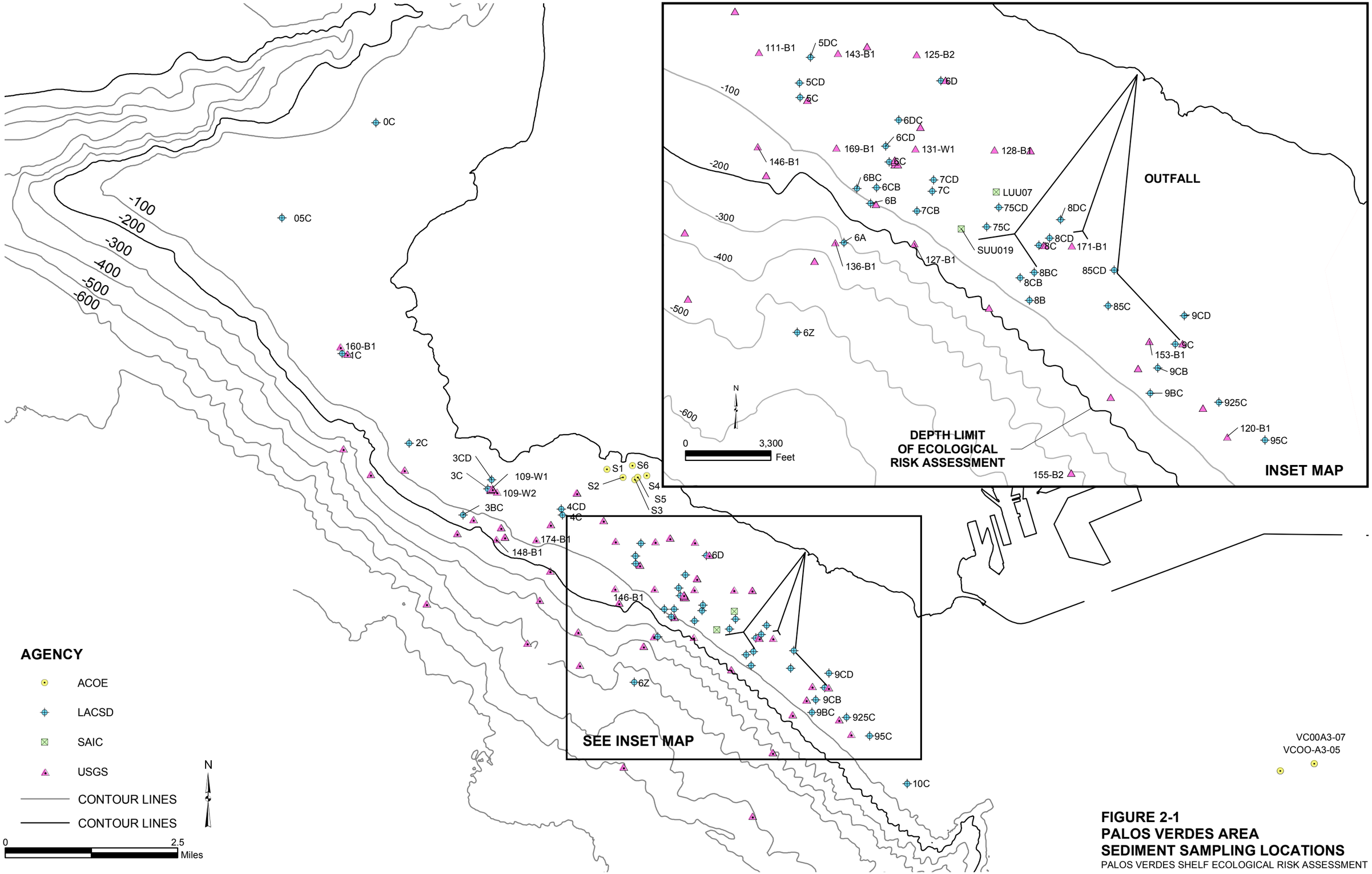
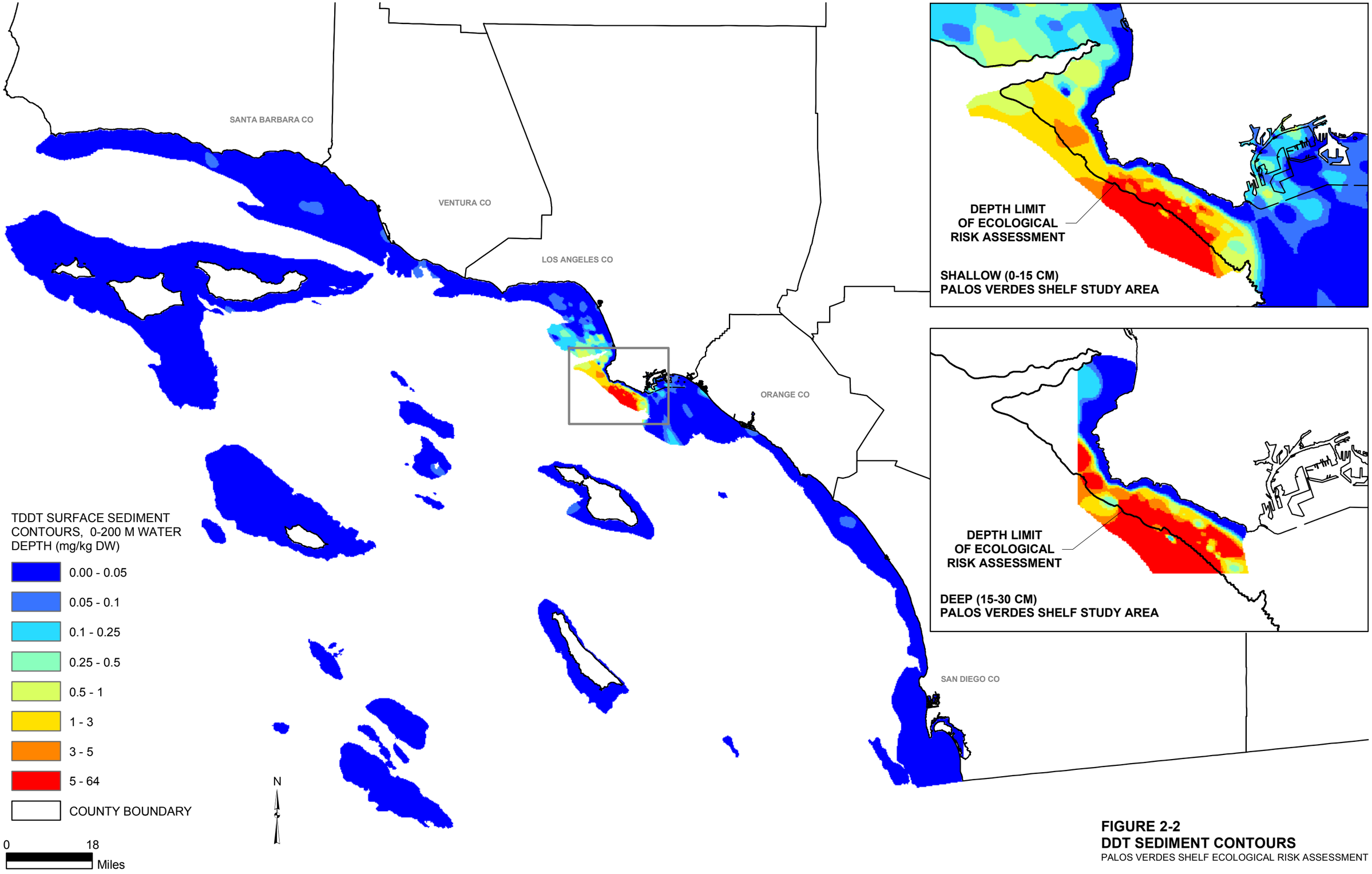


FIGURE 2-1
PALOS VERDES AREA
SEDIMENT SAMPLING LOCATIONS
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

RDD \\LOKI\PROJECTS\RDDGIS\PVS\PVS_SAMPLE_ZONE.MXD PVS_SAMPLE_ZONE.PDF 10/27/2003 22:06:56



RDD \\LOKI\PROJECTS\RDDGIS\PVS\PVS_DDT.MXD PVS_DDT.PDF 10/29/2003 13:30:46

FIGURE 2-2
DDT SEDIMENT CONTOURS
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

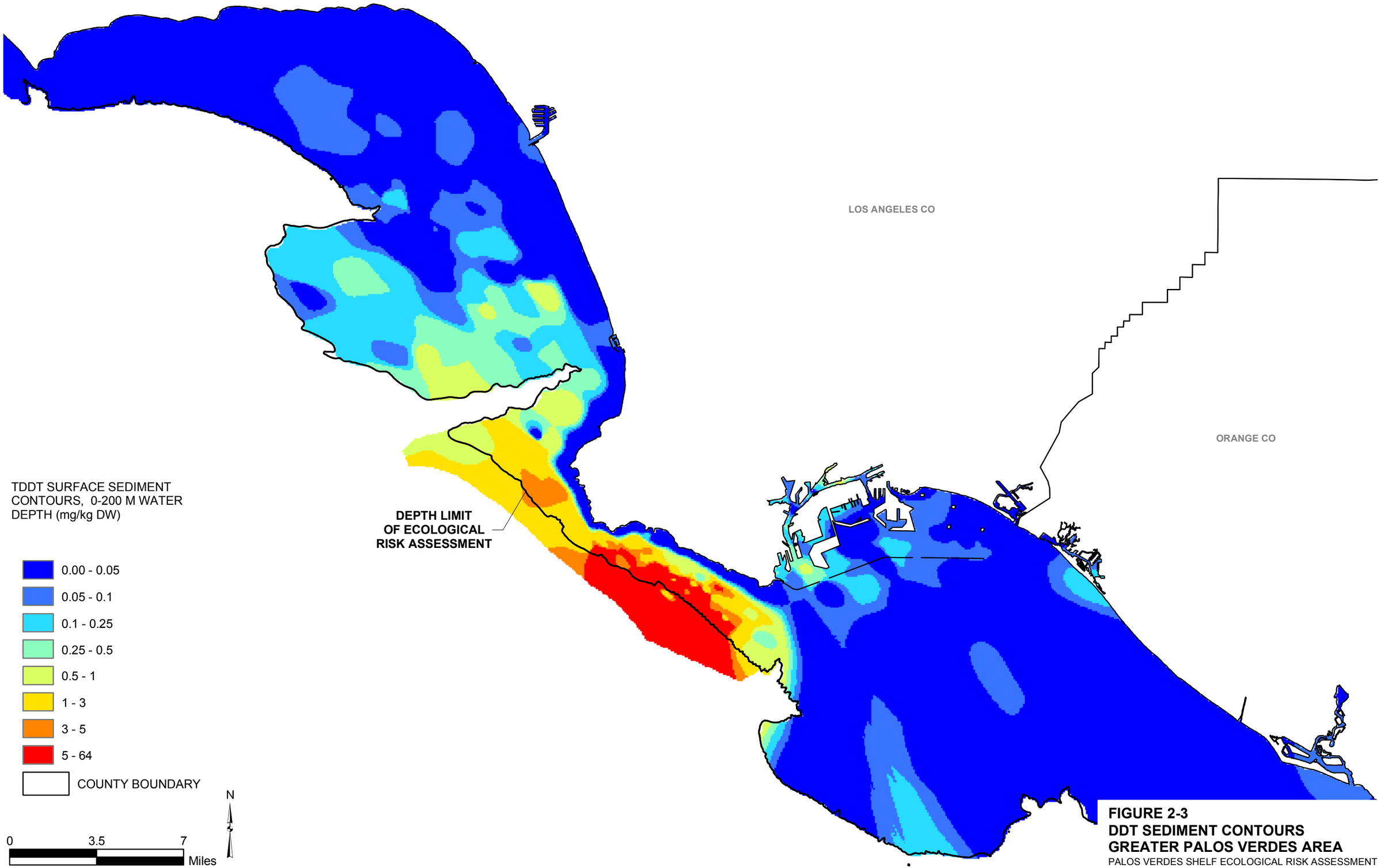


FIGURE 2-3
DDT SEDIMENT CONTOURS
GREATER PALOS VERDES AREA
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

RDD \\LOKI\PROJECTS\RDDGIS\PVS\PVS_DDT_INSET.MXD PVS_DDT_INSET.MXD 10/27/2003 22:26:04

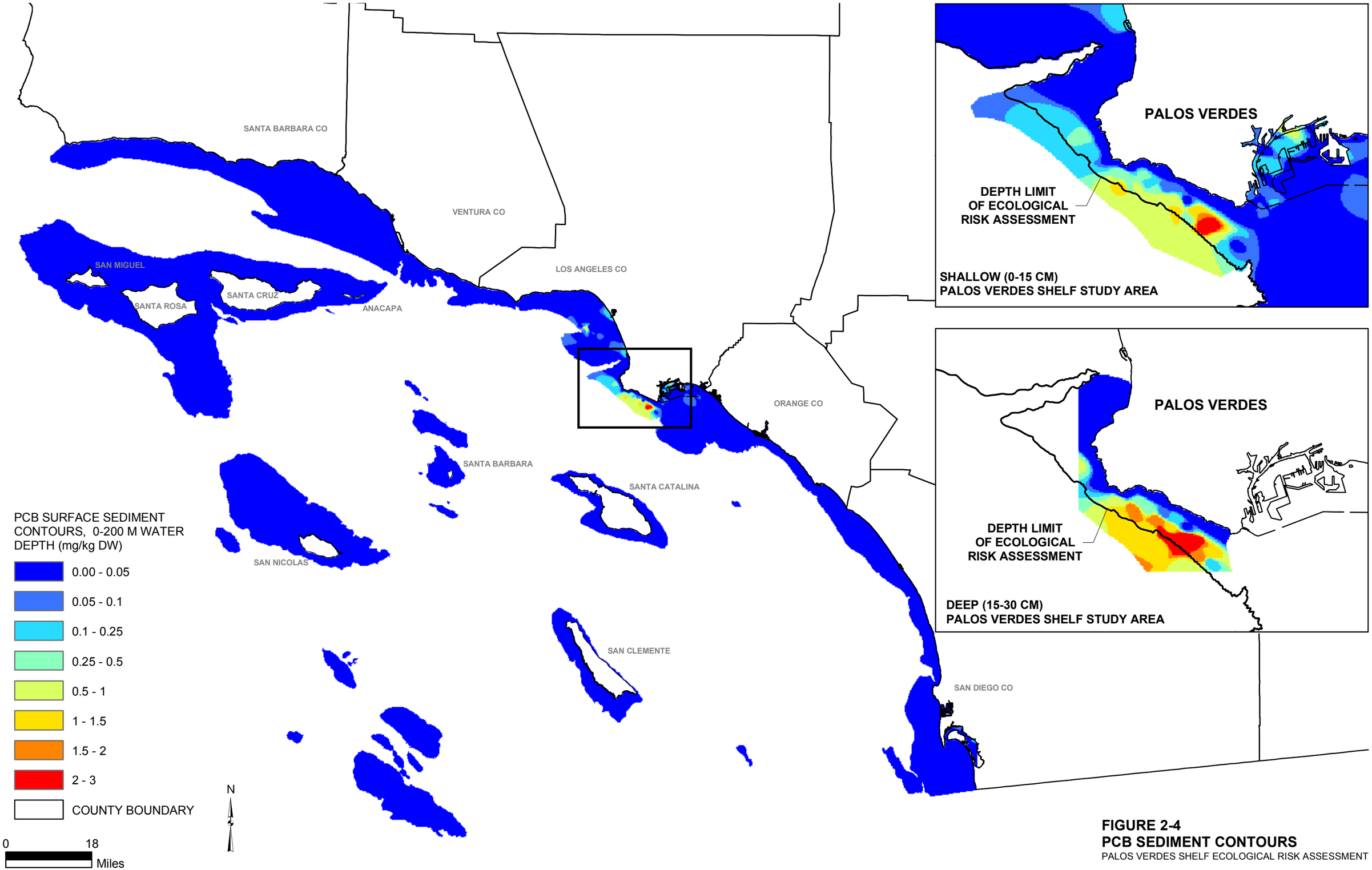


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FIGURE 2-4
PCB SEDIMENT CONTOURS
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

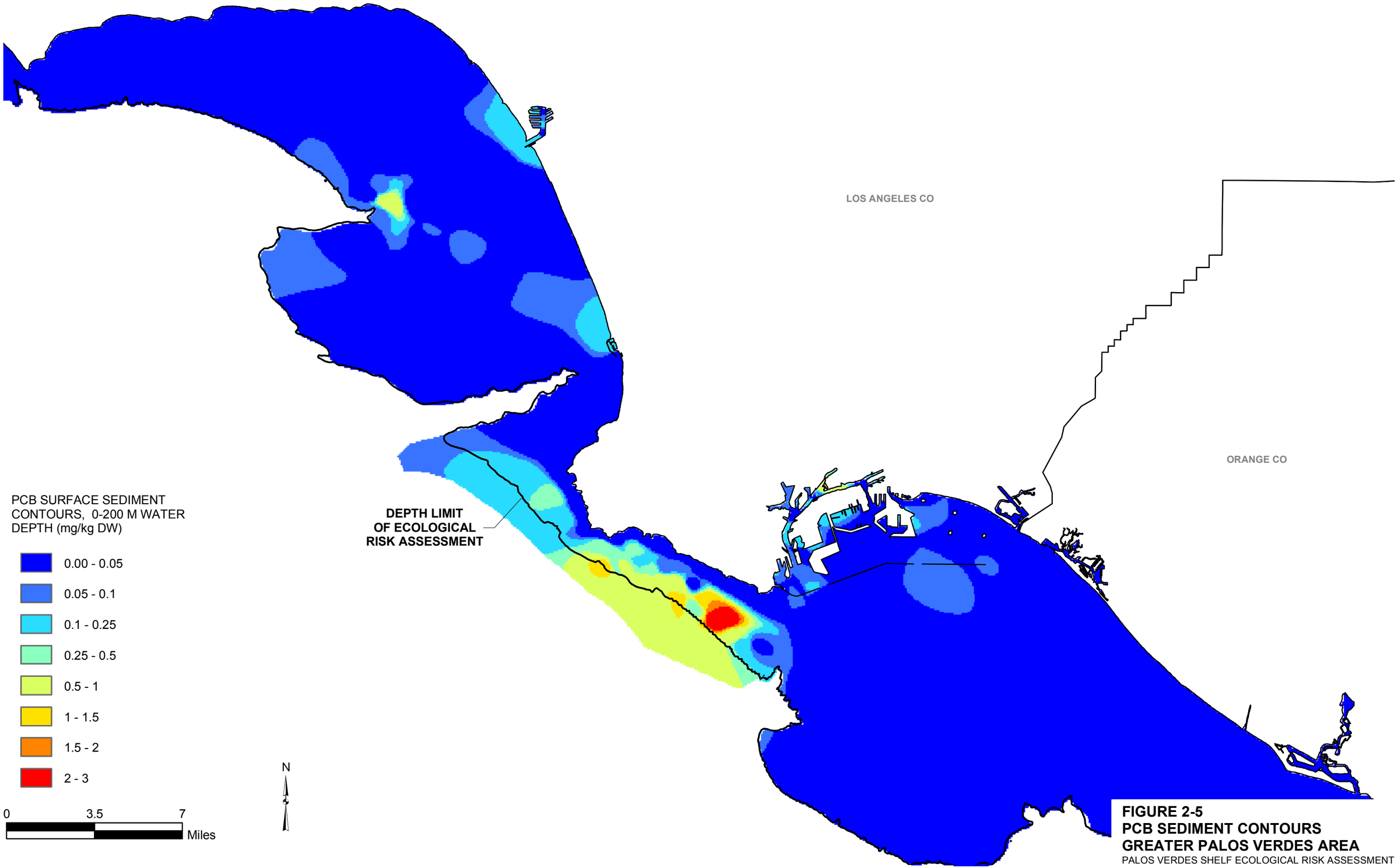

FIGURE 2-5
PCB SEDIMENT CONTOURS
GREATER PALOS VERDES AREA
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

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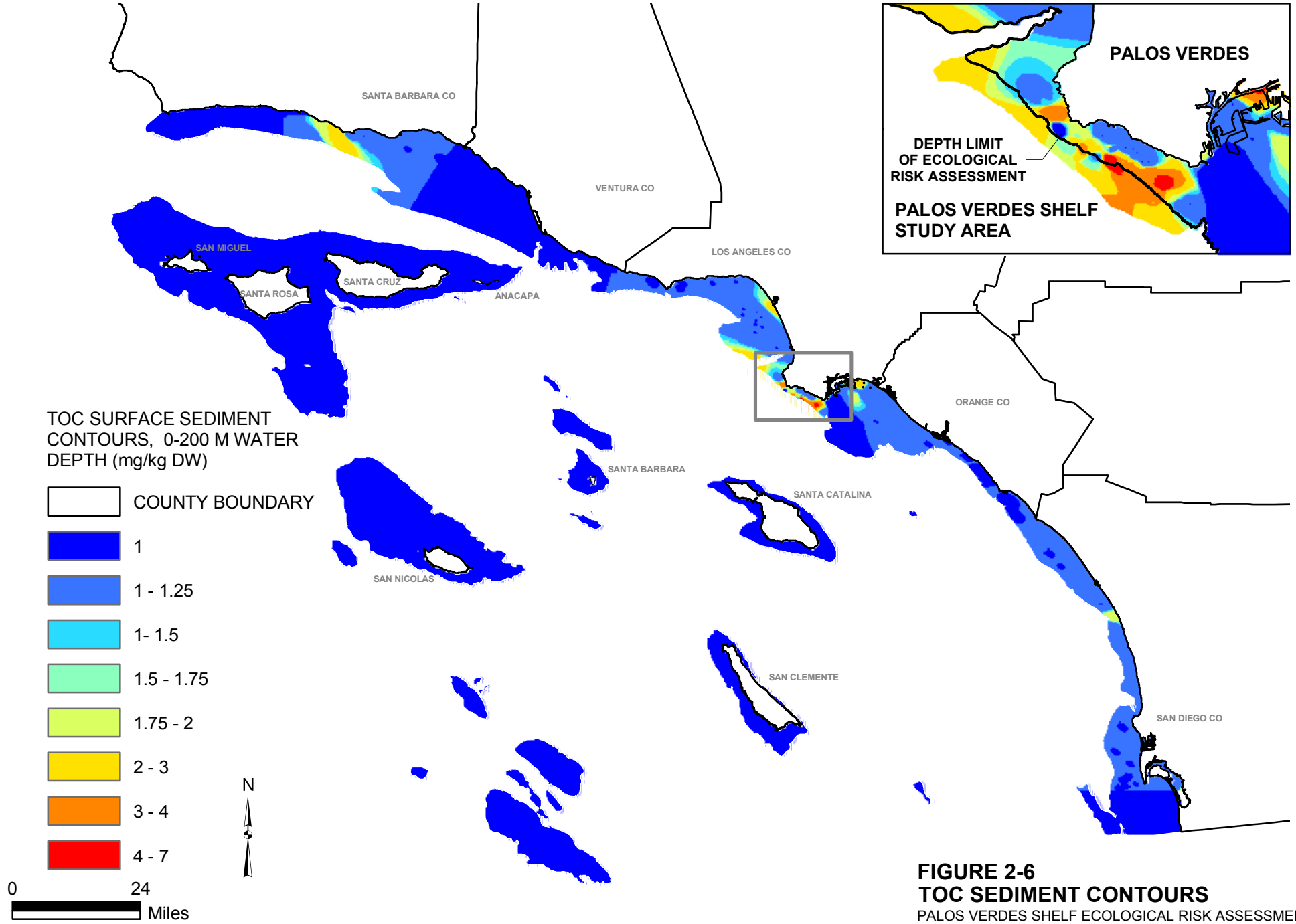

FIGURE 2-6
TOC SEDIMENT CONTOURS
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

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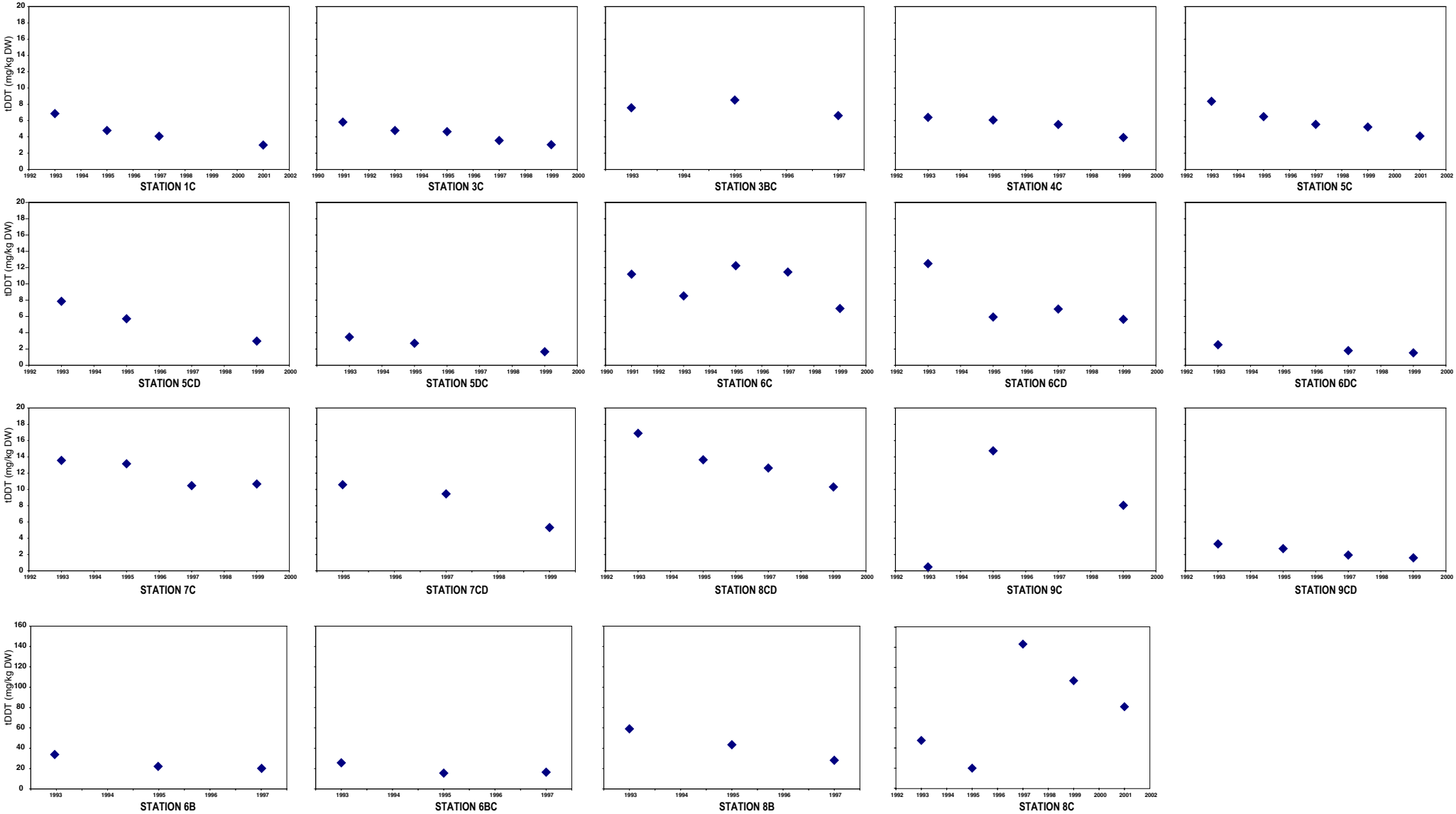

FIGURE 2-7
LACSD SEDIMENT CORES (1991-2001)
0-15 CM DEEP
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

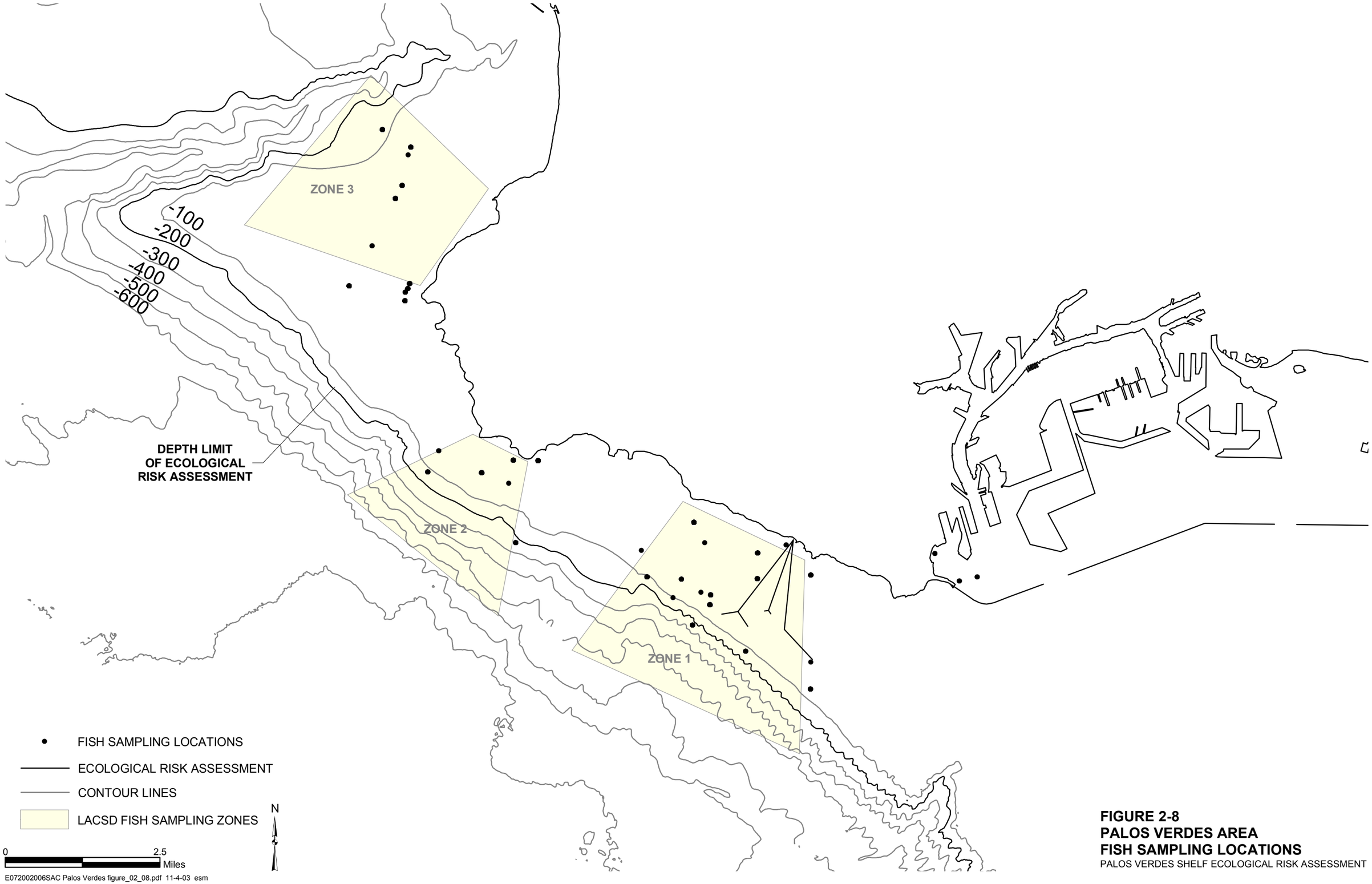

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FIGURE 2-8
PALOS VERDES AREA
FISH SAMPLING LOCATIONS
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

A decreasing trend is suggested at a few stations (1C, 4C, 5C, 5CD, 6CD, BCD, and 8B). All other stations, except 8C, remained relatively unchanged from 1991 through 2001 (or the latest sample date). Station 8C shows an initial decrease, then an increase followed by a decrease in recent years (1997-2001).

With the exception of these three stations, DDT concentrations in surface layer sediment appear to be relatively constant as represented by the LACSD cores collected between 1991 and 2001. The observed trends at 8B and some low concentration stations, and variability at 8C over time may be due to sediment burial, erosion, or transport, or to the inherent small-scale heterogeneity of sediment quality.

Water Column and Porewater

Few data on DDT and PCB concentrations in porewater or the water column above the PVS are available. Green et al. (1986) reported DDT concentrations of 3.0 ng/L at 20-m depth off an unused JWPCP outfall. Studies by Zeng and Tran (2002) report ranges of tDDT at Station 7C off the LACSD outfalls in 60 m of water, 2 m above the bottom. Samples taken in the fall of 1995 contained 3.9-5.4 ng tDDT/L, and those in the spring of 1996 contained 6.8-11.7 ng tDDT/L. Comparable samples taken in the spring off Dana Point, to the south of PVS, contained 0.2 ng tDDT/L.

In a more recent study conducted by Zeng et al. (1999), water column samples from approximately 1 m off the sea floor were collected during winter and summer of 1997 off the Palos Verdes peninsula at eight LACSD monitoring stations and analyzed for tDDT and tPCB. Six stations were located at the 60-m contour C stations and two each at the 30-m (D stations) and 150-m contours. At one mid-shelf station (6C), samples were taken from approximately 2, 5, 20, and 35 m above the bottom. These data for tDDT and tPCB are presented in Tables 2-8 and 2-9, respectively. For both tDDT and tPCB in winter and summer, there was a distinct gradient whereby dissolved, particulate, and total contaminant concentrations were highest near the bottom. In general, the water concentrations reflected sediment concentrations at the site and ranged from 0.6-15.8 ng/L and 0.06-1.14 ng/L for tDDT and tPCB, respectively. These data indicate elevated tDDT and tPCB concentrations in the water column at the site.

No reliable porewater data are available for use in this evaluation; instead, porewater concentrations of COPECs were estimated from bulk, surface-sediment concentrations of contaminants and organic carbon content using established equilibrium partitioning models (see Section 3).

2.3.2.2 Biological Data

Biological data for this assessment include fish, avian, and sea lion tissue tDDT and tPCB concentrations. Summary statistics for fish and birds are provided in Tables 2-10, 2-11, and 2-12, respectively. Data for marine mammals were not combined across studies, as much of the data came from literature studies, and descriptive data (such as age) were not always available for similar groupings of animals (e.g., age, sex, location). Fish sampling locations at PVS are shown in Figure 2-8.

Earlier studies have shown that muscle tissues of bottom-dwelling fish and invertebrates from the PVS area usually have the highest concentrations of COPECs in the SCB (SMBRP 1992). Concentrations in pelagic fish, which travel over a wide range of the Bight, were more uniform. Of the various fish species analyzed, white croaker had the highest tDDT and tPCB residues at every location in studies conducted between 1990 and 2001. The high

TABLE 2-8
Water-Column Concentrations of tDDT
Palos Verdes Shelf Ecological Risk Assessment

Station	Station Depth (m)	Sampling Depth (m)	Season	Concentration (ng/L)
0C-1M	60	59	Winter	2.3
3C-1M	60	59	Winter	4.5
5C-1M	60	59	Winter	9.2
6C-1M	60	59	Winter	14.5
6C-2M	60	58	Winter	15.8
6C-5M	60	55	Winter	7.6
6C-20M	60	40	Winter	2.8
6C-35M	60	25	Winter	0.8
7C-1M	60	59	Winter	9.9
9C-1M	60	59	Winter	5.3
6B-1M	150	149	Winter	5.4
6D-1M	30	29	Winter	7.2
0C-1M	60	59	Summer	4.3
3C-1M	60	59	Summer	7.6
5C-1M	60	59	Summer	10.4
6C-1M	60	59	Summer	8.7
6C-2M	60	58	Summer	10.3
6C-5M	60	55	Summer	8.6
6C-20M	60	40	Summer	2
6C-35M	60	25	Summer	0.6
7C-1M	60	59	Summer	5.5
9C-1M	60	59	Summer	5
6B-1M	150	149	Summer	5.6
6D-1M	30	29	Summer	3

Notes:
Winter samples taken Jan-March, 1997
Summer samples taken Jun-Jul, 1997
Data from Zeng et al. (1999)

TABLE 2-9
Water-Column Concentrations of tPCB
Palos Verdes Shelf Ecological Risk Assessment

Station	Station Depth (m)	Sampling Depth (m)	Season	Concentration (ng/L)
0C-1M	60	59	Winter	0.14
3C-1M	60	59	Winter	0.28
5C-1M	60	59	Winter	0.51
6C-1M	60	59	Winter	0.88
6C-2M	60	58	Winter	0.89
6C-5M	60	55	Winter	0.41
6C-20M	60	40	Winter	0.21
6C-35M	60	25	Winter	0.06
7C-1M	60	59	Winter	0.65
9C-1M	60	59	Winter	0.31
6B-1M	150	149	Winter	0.33
6D-1M	30	29	Winter	0.48
0C-1M	60	59	Summer	0.41
3C-1M	60	59	Summer	0.94
5C-1M	60	59	Summer	1.14
6C-1M	60	59	Summer	0.84
6C-2M	60	58	Summer	1.11
6C-5M	60	55	Summer	0.94
6C-20M	60	40	Summer	0.28
6C-35M	60	25	Summer	0.21
7C-1M	60	59	Summer	0.56
9C-1M	60	59	Summer	0.30
6B-1M	150	149	Summer	0.52
6D-1M	30	29	Summer	0.67

Notes:
Winter samples taken Jan-March, 1997
Summer samples taken Jun-Jul, 1997
Data from Zeng et al. (1999)

TABLE 2-10
Palos Verdes Study Area Fish Tissue Summary Statistics (1990 – 2003)
Palos Verdes Shelf Ecological Risk Assessment

Analyte	Species Common Name	Tissue Type	Non-Detects	n	Geometric Mean (mg/kg WW)	Standard Deviation	Minimum	Median	Maximum	95% UCL of the Geometric Mean
tDDT	Dover sole	fillet	0	110	1.784	2.945	0.229	1.941	34.295	4.008
	Kelp bass	fillet	0	227	0.347	3.945	0.030	0.299	17.222	1.130
	White croaker	fillet	0	379	6.709	3.452	0.190	6.671	134.963	16.744
	Sanddab "Guild"	whole body	0	7	3.510	3.151	0.384	4.938	10.462	--[a]
tPCB	Dover sole	fillet	0	110	0.221	2.292	0.029	0.232	2.550	0.366
	Kelp bass	fillet	26[b]	227	0.082	4.811	0.005	0.073	3.948	0.368
	White croaker	fillet	0	380	0.703	2.618	0.0200	0.690	10.196	1.240
	Sanddab "Guild"	whole body	0	7	0.229	2.813	0.0320	0.309	0.710	--[a]

Notes:
[a] Due to the small sample size and large standard deviation, calculation of the 95% UCL of the geometric mean was not appropriate.
[b] A value of one-half the method detection limit (0.005 mg/kg) was used for non-detects to calculate summary statistics for tPCB in kelp bass.
Mean and standard deviation calculated from ln-transformed data.
95% UCL = 95th Upper Confidence Limit of the geometric mean
Sanddab "Guild" includes English sole, longfin sanddab, Pacific sanddab, speckled sanddab, and slender sole.

concentrations have been attributed to the high lipid content in croaker muscle tissue, their feeding habits, or both.

Tables 2-3 and 2-4 summarize the available DDT and PCB data sources for various species historically sampled within the site. Analyses for concentrations of tDDT and tPCB in fish have focused on muscle tissue. The metabolite 4,4'-DDE was the dominant DDT isomer in the biota sampled, representing about 90 percent of the total DDT. LACSD (1995) reports that Aroclor 1254 represents more than 50 percent of the tPCB, and Aroclors 1242 and 1260 were present at lower levels. Few data exist for invertebrates that are found in the soft-bottom sediment habitats.

Spatial trends for tissue contaminant residues were also evaluated within the site. Three fish species (kelp bass, Dover sole, and white croaker) were evaluated for tDDT and tPCB based on fish trawl data collected by LACSD from 1990 to 2000. White croaker and Dover sole were collected from trawls and kelp bass were collected using hook and line or divers in three zones (zone 1—White Point to Bunker Point, zone 2—Long Point to Point Vicente, and zone 3—Palos Verdes Point to Bluff Cove).

Contaminant data are also available for brown pelicans, double-crested cormorants, peregrine falcons, bald eagles, and sea lions. Data for western gulls and ashy storm petrels are included for possible future use. These data are listed in Tables 2-11 and 2-12; calculated TEQs are presented in Table 2-13. (Note: mean values for ashy storm petrel eggs collected

TABLE 2-11
Avian Summary Statistics for the Southern California Bight, 1985 – 2000
Palos Verdes Shelf Ecological Risk Assessment

					(mg/kg WW)						
Analyte	**Specific Island**	**Full Species Name**	**Tissue**	**n**	**Mean**	**Standard Deviation**	**Minimum**	**Median**	**P90**	**Maximum**	**95% UCL**
tDDT	Anacapa Island	Brown pelican	Egg	24	3.64	4.67	0.742	2.690	5.09	24.1	5.51
tDDT	Anacapa Island	Double-crested cormorant	Egg	13	8.34	7.41	1.83	4.82	17.70	25.60	12.37
tDDT	Anacapa Island	Peregrine falcon	Blood	1	0.59	--	0.59	0.59	0.59	0.59	--
tDDT	Anacapa Island	Peregrine falcon	Egg	2	52.55	24.68	35.10	52.55	70.00	70.00	86.75
tDDT	Anacapa Island	Western gull	Egg	12	1.35	0.50	0.582	1.36	1.810	2.140	1.63
tDDT	Mainland Los Angeles Harbor	Peregrine falcon	Body	1	2.54	--	2.54	2.54	2.54	2.54	--
tDDT	Mainland Los Angeles Harbor	Peregrine falcon	Brain	1	3.24	--	3.24	3.24	3.24	3.24	--
tDDT	Mainland Los Angeles Harbor	Peregrine falcon	Liver	1	6.68	--	6.68	6.68	6.68	6.68	--
tDDT	San Clemente Island	Western gull	Egg	14	6.63	4.60	2.42	5.36	9.43	20.9	9.05
tDDT	San Miguel Island	Peregrine falcon	Blood	1	0.99	--	0.99	0.99	0.99	0.99	--
tDDT	San Miguel Island	Peregrine falcon	Egg	6	20.12	4.85	14.80	19.10	26.30	26.30	24.00
tDDT	San Miguel Island	Western gull	Egg	11	7.98	8.03	2.59	4.40	23.90	24.00	12.72
tDDT	San Miguel Island	Western gull	Other	1	9.70	--	9.70	9.70	9.70	9.70	--
tDDT	Santa Barbara Island	Western gull	Body	1	18.90	--	18.90	18.90	18.90	18.90	--
tDDT	Santa Barbara Island	Western gull	Egg	12	5.96	9.59	2.22	3.13	5.18	36.30	11.39
tDDT	Santa Catalina Island	Bald eagle	Blood	4	0.07	0.05	0.02	0.07	0.11	0.11	0.11
tDDT	Santa Catalina Island	Bald eagle	Body	1	0.66	--	0.66	0.66	0.66	0.66	--
tDDT	Santa Catalina Island	Bald eagle	Brain	2	214.80	3.11	212.60	214.80	217.00	217.00	219.11

TABLE 2-11
Avian Summary Statistics for the Southern California Bight, 1985 – 2000
Palos Verdes Shelf Ecological Risk Assessment

					(mg/kg WW)						
Analyte	**Specific Island**	**Full Species Name**	**Tissue**	**n**	**Mean**	**Standard Deviation**	**Minimum**	**Median**	**P90**	**Maximum**	**95% UCL**
tDDT	Santa Catalina Island	Bald eagle	Egg	36	27.19	14.57	6.83	22.62	47.80	67.20	31.95
tDDT	Santa Catalina Island	Bald eagle	Liver	2	844.70	8.91	838.40	844.70	851.00	851.00	857.05
tDDT	Santa Catalina Island	Bald eagle	Muscle	1	321.00	--	321.00	321.00	321.00	321.00	--
tDDT	Santa Catalina Island	Bald eagle	Other	1	18.30	--	18.30	18.30	18.30	18.30	--
tDDT	Santa Catalina Island	Bald eagle	Serum	3	35.66	30.86	0.02	53.25	53.70	53.70	70.58
tDDT	Santa Catalina Island	Bald eagle	Skeletal Muscle	1	317.52	--	317.52	317.52	317.52	317.52	--
tDDT	Santa Catalina Island	Bald eagle	Yolk	1	57.90	--	57.90	57.90	57.90	57.90	--
tDDT	Santa Catalina Island	Brown pelican	Egg	1	3.14	--	3.14	3.14	3.14	3.14	--
tDDT	Santa Catalina Island	Double-crested cormorant	Egg	4	1.40	0.45	0.88	1.38	1.97	1.97	1.85
tDDT	Santa Catalina Island	Western gull	Egg	12	6.69	2.62	3.00	6.59	9.48	10.70	8.17
tDDT	Santa Cruz Island	Ashy storm petrel	Egg	14	13.38	5.79	3.45	12.05	19.70	26.60	16.41
tDDT	Santa Cruz Island	Peregrine falcon	Blood	1	0.40	--	0.40	0.40	0.40	0.40	--
tDDT	Santa Cruz Island	Peregrine falcon	Egg	7	15.18	6.56	7.14	17.50	22.40	22.40	20.04
tDDT	Santa Cruz Island	Peregrine falcon	Serum	1	3.70	--	3.70	3.70	3.70	3.70	--
tDDT	Santa Cruz Island	Western gull	Egg	12	1.27	0.36	0.65	1.26	1.78	1.93	1.48
tDDT	Santa Rosa Island	Peregrine falcon	Egg	1	32.50	--	32.50	32.50	32.50	32.50	--
tDDT	Santa Rosa Island	Western gull	Body	1	6.94	--	6.94	6.94	6.94	6.94	10.33
tPCB	Anacapa Island	Brown pelican	Egg	24	2.22	2.36	0.48	1.30	6.37	9.08	3.17
tPCB	Anacapa Island	Double-crested cormorant	Egg	13	5.05	5.89	0.55	2.20	13.40	18.60	8.26

TABLE 2-11
Avian Summary Statistics for the Southern California Bight, 1985 – 2000
Palos Verdes Shelf Ecological Risk Assessment

					(mg/kg WW)						
Analyte	**Specific Island**	**Full Species Name**	**Tissue**	**n**	**Mean**	**Standard Deviation**	**Minimum**	**Median**	**P90**	**Maximum**	**95% UCL**
tPCB	Anacapa Island	Peregrine falcon	Blood	1	0.49	--	0.49	0.49	0.49	0.49	--
tPCB	Anacapa Island	Peregrine falcon	Egg	2	19.80	9.90	12.80	19.80	26.80	26.80	33.52
tPCB	Anacapa Island	Western gull	Egg	12	0.60	0.22	0.31	0.56	0.87	1.05	0.72
tPCB	Mainland Los Angeles Harbor	Peregrine falcon	Body	1	1.37	--	1.37	1.37	1.37	1.37	--
tPCB	Mainland Los Angeles Harbor	Peregrine falcon	Brain	1	3.19	--	3.19	3.19	3.19	3.19	--
tPCB	Mainland Los Angeles Harbor	Peregrine falcon	Liver	1	5.13	--	5.13	5.13	5.13	5.13	--
tPCB	San Clemente Island	Western gull	Egg	14	2.62	1.28	1.23	2.37	3.20	6.56	3.29
tPCB	San Miguel Island	Peregrine falcon	Blood	1	0.61	--	0.61	0.61	0.61	0.61	--
tPCB	San Miguel Island	Peregrine falcon	Egg	6	9.15	1.31	7.23	9.24	10.60	10.60	10.19
tPCB	San Miguel Island	Western gull	Egg	11	3.16	3.63	1.09	1.69	9.91	11.00	5.30
tPCB	San Miguel Island	Western gull	Other	1	4.93	--	4.93	4.93	4.93	4.93	--
tPCB	Santa Barbara Island	Western gull	Body	1	5.48	--	5.48	5.48	5.48	5.48	--
tPCB	Santa Barbara Island	Western gull	Egg	12	1.73	2.97	0.47	1.00	1.30	11.10	3.41
tPCB	Santa Catalina Island	Bald eagle	Blood	1	0.06	--	0.06	0.06	0.06	0.06	--
tPCB	Santa Catalina Island	Bald eagle	Body	1	0.42	--	0.42	0.42	0.42	0.42	--
tPCB	Santa Catalina Island	Bald eagle	Brain	2	77.65	26.94	58.60	77.65	96.70	96.70	114.99
tPCB	Santa Catalina Island	Bald eagle	Egg	36	8.66	5.97	1.82	6.47	17.24	29.70	10.61
tPCB	Santa Catalina Island	Bald eagle	Liver	2	294.00	0.00	294.00	294.00	294.00	294.00	294.00
tPCB	Santa Catalina Island	Bald eagle	Muscle	1	140.00	--	140.00	140.00	140.00	140.00	--

TABLE 2-11
Avian Summary Statistics for the Southern California Bight, 1985 – 2000
Palos Verdes Shelf Ecological Risk Assessment

					(mg/kg WW)						
Analyte	**Specific Island**	**Full Species Name**	**Tissue**	**n**	**Mean**	**Standard Deviation**	**Minimum**	**Median**	**P90**	**Maximum**	**95% UCL**
tPCB	Santa Catalina Island	Bald eagle	Other	1	5.87	--	5.87	5.87	5.87	5.87	--
tPCB	Santa Catalina Island	Bald eagle	Serum	2	26.00	0.00	26.00	26.00	26.00	26.00	26.00
tPCB	Santa Catalina Island	Bald eagle	Skeletal muscle	1	3.90	--	3.90	3.90	3.90	3.90	--
tPCB	Santa Catalina Island	Bald eagle	Yolk	1	22.10	--	22.10	22.10	22.10	22.10	--
tPCB	Santa Catalina Island	Brown pelican	Egg	1	1.43	--	1.43	1.43	1.43	1.43	--
tPCB	Santa Catalina Island	Western gull	Egg	12	2.52	0.79	1.36	2.62	3.22	3.91	2.97
tPCB	Santa Cruz Island	Ashy storm petrel	Egg	14	9.39	3.74	3.28	9.03	12.60	19.40	11.35
tPCB	Santa Cruz Island	Peregrine falcon	Blood	1	0.38	--	0.38	0.38	0.38	0.38	--
tPCB	Santa Cruz Island	Peregrine falcon	Egg	7	9.40	5.01	3.66	11.30	16.10	16.10	13.11
tPCB	Santa Cruz Island	Peregrine falcon	Serum	1	2.24	--	2.24	2.24	2.24	2.24	--
tPCB	Santa Cruz Island	Western gull	Egg	12	0.76	0.11	0.53	0.76	0.88	0.98	0.83
tPCB	Santa Rosa Island	Peregrine falcon	Egg	1	13.60	--	13.60	13.60	13.60	13.60	--
tPCB	Santa Rosa Island	Western gull	Body	1	1.93	--	1.93	1.93	1.93	1.93	--

Notes:
-- denotes no data

P90 90th percentile value
95% UCL 95th Upper Confidence Limit of Mean

TABLE 2-12
Avian Summary Statistics from Samples Collected in Areas Outside Southern California Bight, 1985 – 2000
Palos Verdes Shelf Ecological Risk Assessment

					(mg/kg WW)						
Analyte	**Location**	**Species**	**Tissue**	**n**	**Mean**	**Standard Deviation**	**Minimum**	**Median**	**P90**	**Maximum**	**95% UCL**
tDDT	British Columbia (BC)	Bald eagle	Liver	5	2.25	1.99	0.62	1.53	5.67	5.67	3.99
tDDT	British Columbia (BC)	Bald eagle	Blood Plasma	8	0.02	0.01	0.01	0.01	0.03	0.03	0.02
tDDT	British Columbia (BC)	Bald eagle	Yolk sacs	4	17.21	6.08	10.64	16.44	25.29	25.29	23.16
tDDT	British Columbia (BC)	Double-crested cormorant	Egg	14	0.57	0.37	0.19	0.47	1.14	1.15	0.76
tDDT	British Columbia (BC)	Peregrine falcon	Egg	1	6.74	--	6.74	6.74	6.74	6.74	--
tDDT	Coos, Curry, Other (OR)	Bald eagle	Blood	3	0.95	1.30	0.058	0.36	2.45	2.45	2.43
tDDT	Coos, Curry, Other (OR)	Bald eagle	Egg	1	11.10	--	11.10	11.10	11.10	11.10	--
tDDT	Coos, Curry, Other (OR)	Double-crested cormorant	Body	5	0.34	0.32	0.12	0.20	0.89	0.89	0.62
tDDT	Coos, Curry, Other (OR)	Double-crested cormorant	Egg	14	1.20	0.63	0.28	1.30	2.01	2.18	1.53
tDDT	Coos, Curry, Other (OR)	Double-crested cormorant	Other	15	3.67	2.84	0.92	1.89	6.24	10.70	5.11
tDDT	Coos, Curry, Other (OR)	Western gull	Egg	12	0.66	0.44	0.26	0.54	1.44	1.86	0.91
tDDT	Humboldt (CA)	Double-crested cormorant	Egg	12	3.65	4.93	0.57	1.59	7.75	17.90	6.44
tDDT	San Francisco, Sonoma, Yolo (CA)	Ashy storm petrel	Egg	1	9.37	--	9.37	9.37	9.37	9.37	--
tDDT	San Francisco, Sonoma, Yolo (CA)	Double-crested cormorant	Egg	12	2.50	1.38	1.15	2.13	3.48	6.27	3.28

TABLE 2-12
Avian Summary Statistics from Samples Collected in Areas Outside Southern California Bight, 1985 – 2000
Palos Verdes Shelf Ecological Risk Assessment

					(mg/kg WW)						
Analyte	**Location**	**Species**	**Tissue**	**n**	**Mean**	**Standard Deviation**	**Minimum**	**Median**	**P90**	**Maximum**	**95% UCL**
tDDT	San Francisco, Sonoma, Yolo (CA)	Peregrine falcon	Egg	10	11.50	7.62	4.10	8.98	23.59	29.00	16.23
tDDT	San Francisco, Sonoma, Yolo (CA)	Western gull	Egg	7	0.52	0.19	0.32	0.48	0.85	0.85	0.66
tDDT	San Luis Obispo (CA)	Double-crested cormorant	Egg	12	3.30	1.68	0.99	3.06	5.76	6.26	4.25
tDDT	San Luis Obispo (CA)	Peregrine falcon	Blood	1	1.57	--	1.57	1.57	1.57	1.57	--
tDDT	San Luis Obispo (CA)	Peregrine falcon	Egg	9	9.29	8.60	1.29	5.55	21.90	21.90	14.91
tDDT	San Luis Obispo (CA)	Peregrine falcon	Serum	1	0.13	--	0.13	0.13	0.13	0.13	--
tDDT	Washoe (NV)	Bald eagle	Brain	1	2.29	--	2.29	2.29	2.29	2.29	--
tDDT	Washoe (NV)	Bald eagle	Fat	1	81.90	--	81.90	81.90	81.90	81.90	--
tDDT	Washoe (NV)	Bald eagle	Liver	1	5.76	--	5.76	5.76	5.76	5.76	--
tDDT	Washoe (NV)	Bald eagle	Muscle	1	9.25	--	9.25	9.25	9.25	9.25	--
tPCB	British Columbia (BC)	Bald eagle	Liver	5	1.48	1.10	0.61	1.15	3.36	3.36	2.45
tPCB	British Columbia (BC)	Bald eagle	Blood Plasma	8	0.02	0.02	0.01	0.02	0.06	0.06	0.03
tPCB	British Columbia (BC)	Bald eagle	Yolk sacs	4	50.87	13.77	33.79	52.84	64.00	64.00	64.36
tPCB	British Columbia (BC)	Double-crested cormorant	Egg	14	7.04	16.25	1.14	2.75	4.76	63.40	15.55
tPCB	British Columbia (BC)	Peregrine falcon	Egg	1	8.58	--	8.58	8.58	8.58	8.58	--
tPCB	Coos, Curry, Other (OR)	Bald eagle	Blood	3	0.99	1.25	0.04	0.53	2.40	2.40	2.40
tPCB	Coos, Curry, Other (OR)	Bald eagle	Egg	1	12.70	--	12.70	12.70	12.70	12.70	--

TABLE 2-12
Avian Summary Statistics from Samples Collected in Areas Outside Southern California Bight, 1985 – 2000
Palos Verdes Shelf Ecological Risk Assessment

					(mg/kg WW)						
Analyte	**Location**	**Species**	**Tissue**	**n**	**Mean**	**Standard Deviation**	**Minimum**	**Median**	**P90**	**Maximum**	**95% UCL**
tPCB	Coos, Curry, Other (OR)	Double-crested cormorant	Body	5	0.47	0.40	0.16	0.29	1.13	1.13	0.82
tPCB	Coos, Curry, Other (OR)	Double-crested cormorant	Egg	14	1.60	1.30	0.37	1.16	3.67	4.59	2.28
tPCB	Coos, Curry, Other (OR)	Double-crested cormorant	Other	15	4.46	3.83	1.13	2.33	8.90	14.50	6.39
tPCB	Coos, Curry, Other (OR)	Western gull	Egg	12	0.72	0.45	0.36	0.54	1.19	1.93	0.97
tPCB	Humboldt (CA)	Double-crested cormorant	Egg	12	2.02	1.18	0.55	2.14	3.47	4.30	2.69
tPCB	San Francisco, Sonoma, Yolo (CA)	Ashy storm petrel	Egg	1	6.93	--	6.93	6.93	6.93	6.93	--
tPCB	San Francisco, Sonoma, Yolo (CA)	Double-crested cormorant	Egg	12	5.18	3.54	0.75	4.86	8.42	13.70	7.18
tPCB	San Francisco, Sonoma, Yolo (CA)	Peregrine falcon	Egg	10	6.72	3.00	2.50	6.40	11.30	13.00	14.91
tPCB	San Francisco, Sonoma, Yolo (CA)	Western gull	Egg	9	0.58	0.27	0.17	0.59	0.92	0.92	0.75
tPCB	San Luis Obispo (CA)	Double-crested cormorant	Egg	12	1.79	1.16	0.37	1.87	3.58	3.58	2.45
tPCB	San Luis Obispo (CA)	Peregrine falcon	Blood	1	1.03	--	1.03	1.03	1.03	1.03	--
tPCB	San Luis Obispo (CA)	Peregrine falcon	Egg	9	4.71	4.26	0.74	2.24	10.20	10.20	7.49
tPCB	Washoe (NV)	Bald eagle	Brain	1	0.83	--	0.83	0.83	0.83	0.83	--
tPCB	Washoe (NV)	Bald eagle	Fat	1	25.00	--	25.00	25.00	25.00	25.00	--

TABLE 2-12
Avian Summary Statistics from Samples Collected in Areas Outside Southern California Bight, 1985 – 2000
Palos Verdes Shelf Ecological Risk Assessment

					(mg/kg WW)						
Analyte	**Location**	**Species**	**Tissue**	**n**	**Mean**	**Standard Deviation**	**Minimum**	**Median**	**P90**	**Maximum**	**95% UCL**
tPCB	Washoe (NV)	Bald eagle	Liver	1	2.17	--	2.17	2.17	2.17	2.17	--
tPCB	Washoe (NV)	Bald eagle	Muscle	1	2.96	--	2.96	2.96	2.96	2.96	--

Notes:
-- denotes no data

P90 90th percentile value
95% UCL 95th Upper Confidence Limit

TABLE 2-13
Summary Statistics for Calculated TEQs in Avian Tissue, by Species and Specific Island, in the SCB, 1985 - 2000
Palos Verdes Shelf Ecological Risk Assessment

Island	Species	Tissue	n	Mean ng/kg WW	Standard Deviation ng/kg WW	Minimum ng/kg WW	Median ng/kg WW	P90 ng/kg WW	Maximum ng/kg WW	95% UCL
Anacapa Island	Brown pelican	egg	22	0.014	0.078	0.002	0.014	0.024	0.027	0.017
Anacapa Island	Double-crested cormorant	egg	13	0.026	0.022	0.005	0.020	0.068	0.070	0.038
Anacapa Island	Peregrine falcon	blood	1	0.003	--	0.003	0.003	0.003	0.003	--
Anacapa Island	Peregrine falcon	egg	2	0.069	0.037	0.043	0.069	0.094	0.094	0.119
Anacapa Island	Western gull	egg	12	0.007	0.003	0.003	0.006	0.012	0.014	0.009
Mainland/Los Angeles Harbor	Peregrine falcon	body	1	0.003	--	0.003	0.003	0.003	0.003	--
Mainland/Los Angeles Harbor	Peregrine falcon	brain	1	0.007	--	0.007	0.007	0.007	0.007	--
Mainland/Los Angeles Harbor	Peregrine falcon	liver	1	0.014	--	0.014	0.014	0.014	0.014	--
San Clemente Island	Western gull	egg	14	0.020	0.005	0.008	0.023	0.025	0.027	0.023
San Miguel Island	Peregrine falcon	blood	1	0.003	--	0.003	0.003	0.003	0.003	--
San Miguel Island	Peregrine falcon	egg	6	0.022	0.004	0.018	0.022	0.028	0.028	0.026
San Miguel Island	Western gull	egg	11	0.012	0.005	0.003	0.012	0.015	0.021	0.014
San Miguel Island	Western gull	other	1	0.014	--	0.014	0.014	0.014	0.014	--
Santa Barbara Island	Western gull	body	1	0.016	--	0.016	0.016	0.016	0.016	--
Santa Barbara Island	Western gull	egg	12	0.013	0.013	0.005	0.010	0.015	0.053	0.020
Santa Catalina Island	Bald eagle	blood	5	0.0007	0.0002	0.0004	0.0007	0.0008	0.0008	0.0008
Santa Catalina Island	Bald eagle	body	1	0.003	--	0.003	0.003	0.003	0.003	--
Santa Catalina Island	Bald eagle	brain	1	0.286	--	0.286	0.286	0.286	0.286	--
Santa Catalina Island	Bald eagle	egg	10	0.039	0.020	0.020	0.034	0.071	0.084	0.051
Santa Catalina Island	Bald eagle	liver	1	0.703	--	0.703	0.703	0.703	0.703	--

TABLE 2-13
Summary Statistics for Calculated TEQs in Avian Tissue, by Species and Specific Island, in the SCB, 1985 - 2000
Palos Verdes Shelf Ecological Risk Assessment

Island	Species	Tissue	n	Mean ng/kg WW	Standard Deviation ng/kg WW	Minimum ng/kg WW	Median ng/kg WW	P90 ng/kg WW	Maximum ng/kg WW	95% UCL
Santa Catalina Island	Bald eagle	muscle	1	0.245	--	0.245	0.245	0.245	0.245	--
Santa Catalina Island	Bald eagle	other	1	0.020	--	0.020	0.020	0.020	0.020	--
Santa Catalina Island	Bald eagle	serum	2	0.025	0.034	0.001	0.025	0.050	0.050	0.073
Santa Catalina Island	Bald eagle	yolk	1	0.051	--	0.051	0.051	0.051	0.051	--
Santa Catalina Island	Brown pelican	egg	1	0.006	--	0.006	0.006	0.006	0.006	--
Santa Catalina Island	Double-crested cormorant	egg	4	0.004	0.001	0.002	0.003	0.006	0.006	0.005
Santa Catalina Island	Western gull	egg	12	0.024	0.008	0.010	0.025	0.031	0.040	0.029
Santa Cruz Island	Ashy storm petrel	egg	14	0.042	0.025	0.016	0.034	0.086	0.105	0.056
Santa Cruz Island	Peregrine falcon	blood	1	0.002	--	0.002	0.002	0.002	0.002	--
Santa Cruz Island	Peregrine falcon	egg	7	0.021	0.009	0.010	0.019	0.034	0.034	0.027
Santa Cruz Island	Peregrine falcon	serum	1	0.027	--	0.027	0.027	0.027	0.027	--
Santa Cruz Island	Western gull	egg	12	0.009	0.003	0.003	0.010	0.012	0.013	0.011
Santa Rosa Island	Peregrine falcon	body	1	0.074	--	0.074	0.074	0.074	0.074	--
Santa Rosa Island	Peregrine falcon	egg	1	0.033	--	0.033	0.033	0.033	0.033	--
Santa Rosa Island	Western gull	body	1	0.004	--	0.004	0.004	0.004	0.004	--

Notes:
-- denotes no data
P90 90th Percentile Value
95% UCL 95 percent Upper Confidence Limit
Sum TEQ = [Avian TEF for PCB congener n* (congener n concentration in tissue)]. TEFs available for PCB congeners 81, 77, 126, 169, 105, 114, 118, 123, 156, 157, 167, 189 from Van den Berg et al. (1998).

from Santa Cruz Island during 1995 to 1997 were also available and are compared to the NRDA values in the trend analysis in Section 2.3.2.3.) The data compiled for NRDA activities were used as the primary source of data for bird species. In addition to the NRDA dataset, data from five studies were summarized for brown pelicans (29 samples), data from three studies were summarized for double-crested cormorants (9 samples), data from nine studies were summarized for bald eagles (85 samples), and data from four studies were summarized for peregrine falcons (23 samples). From these studies, egg residue levels were used as the primary data. Tissue and organ data were used to supplement egg residues when assessing risk to birds. The data were categorized so that comparisons could be made between samples collected prior to and after 1985, and samples collected within and outside the SCB.

Concentrations of 4,4′ DDE and PCB in blubber of sea lions from San Miguel Island have been measured in three separate studies (DeLong et al. 1973; Gilmartin et al. 1976; Costa et al. 1994). All of these studies focused on two groups of postpartum females: premature parturient and full-term parturient females. DeLong et al. (1973) included premature parturient and four full-term parturient females that were collected within 24 hours of parturition (April 20-24, 1970 for the premature females, and June 24-26, 1970 for the full-term females). This study found that the premature parturient females (mean age 8 years) had much higher 4,4′-DDE and PCB concentrations (2 to 8 times higher) than the full-term parturient females (mean age 12 years). A follow-up study was conducted by

Gilmartin and colleagues in 1972 (Gilmartin et al. 1976), in which 10 premature parturient females age 6 to 8 years (March 27 and 29, and April 25 and 27, 1972), and 10 full-term parturient females, age 10 to 14 years (June 13 and 15, 1972) were collected. The premature parturient group had 4,4′-DDE and PCB concentrations 4 to 8 times higher than the full-term parturient females. Both DeLong et al. (1973) and Gilmartin et al. (1976) suggested that the higher contaminant concentrations may be responsible for the premature parturition, although Gilmartin and coworkers found a viral infection that may have been a causative agent. In a third study, conducted in 1991 as part of the NRDA (Costa et al. 1994), 10 premature parturient and 10 full-term parturient females were collected. Blubber samples were analyzed two times for 4,4′-DDE, tPCB, and lipid content by the Texas A&M University Geochemical and Environmental Research Group (GERG) and once by the University of California (UC). Data analyzed by GERG have been previously termed "Old GERG" and "New GERG," depending on their analysis date. Although these data were presented and used in HydroQual, Inc. (1997), quality control measures have been questioned and these 20 samples are being reanalyzed (DeLong 2002). Concentrations used in this ERA represent those estimated from the 1970s values as per instructions from DeLong (2002) (i.e., the higher of either 1970 or 1972 value multiplied by a factor of 0.1).

In addition to the 1970s studies and the NRDA dataset, data for California sea lions (148 samples) from nine other studies were reviewed (Table 2-2 and 2-4). Because the 1991 data were collected from live sea lions of known age and breeding status from a known location, only these data were used in the risk estimation. However, partial data from several other studies (Kajiwara et al. 2001, Le Boeuf et al. 2002, and Le Boeuf 2002) were used for comparison to modeled sea lion concentrations derived for bald eagle exposure.

Biota Data Trends

Decreases in fish tissue concentrations of COPECs have been described for the period of 1970 to the early 1980s. In a review of historical data on seafood contamination for the SCB, including the PVS, SMBRP (1992) found that temporal trends of contamination in fish tissue for the Bight were difficult to assess owing to the variability in the data. For the PVS, SMBRP found the concentration of tPCB (Aroclors) in white croaker muscle tissue had decreased sharply from 2.78 mg/kg WW in 1975 to 0.383 mg/kg in 1980. Subsequent increases in tissue concentrations between 1987 and 1990 at White Point and northern PVS locations, possibly as a result of pre-versus post-spawning collection of the samples, led to the overall conclusion that there was little change in PCB concentration in white croaker muscle tissues. The lack of significant trends is evident in Figures 2-9 to 2-14. This provides support for the use of 1990 to 2001 fish concentration data as representing current conditions within the study area.

Egg tissue samples considered to represent current conditions ranged from 1985 to 2000. This use of time period is supported by the trends reported for brown pelicans and bald eagles nesting in the Channel Islands. Levels of DDE in pelicans have stabilized to low chronic levels since 1985 (Sharpe 2003) and levels in bald eagle eggs show no significant trend between 1989 and 2000 (Garcelon 2000). Additionally, mean values for 23 ashy storm petrel eggs collected on Santa Cruz Island from 1995 to 1997 were compared with eggs collected from 1991 to 1992. Mean DDE concentrations for 1995, 1996, and 1997 were 16.00, 11.01, and 8.60 mg/kg (WW), respectively (Welsh et al. 2001). Although this appears to be a decreasing trend, only eggs that had failed to hatch were collected in 1995 (i.e., these eggs are likely to be the most contaminated) and all means and the maximum concentration (20 mg/kg WW) are similar to the 1991-1992 mean (8.33 mg/kg WW) and maximum (16.88 mg/kg WW). Therefore, DDE levels in ashy storm petrel eggs have changed little into the late 1990s.

2.3.3 Identification of Chemicals of Potential Ecological Concern

This ERA focuses on DDT and PCB contamination in the Bight. The sediment concentrations of these contaminants in the PVS study area can be linked directly to the outfall discharges on the PVS that contained waste from the pesticide manufacturer, Montrose, and other sources. Other contaminant sources are not as clearly defined or extensive and do not affect the whole region as significantly.

Contaminants other than tDDT and tPCB are present in the SCB—specifically, trace metals and organic compounds. Among the trace metals are arsenic, cadmium, chromium, copper, lead, mercury, silver, tin, and zinc. The pesticides chlordane and dieldrin and PAHs have also been found in sediments on the PVS. Some of these contaminants (e.g., cadmium) have declined to below toxic concentrations since the late 1970s, while others (e.g., mercury, zinc) were not found to be elevated in biota as compared to reference sites (Mearns et al. 1991). SCCRWP 1997 data for PAHs indicated that concentrations were generally below detection limits except in a few cases. It should be noted, however, that the detection limits, for some

PAHs exceeded the Effects Range—Low (ER-L) and Effects Range—Median (ER-M) values used as ecological screening benchmarks (Long et al. 1995). The only PAHs detected in any location include napthalene, 1-methylnaphthalene, 2-methylnaphthalene, biphenyl, fluoranthene, 2,6-dimethylnaphthalene, phenanthrene, and pyrene. Of the contaminants with associated ER-L and ER-M values, 2-methylnapthalene was the only analyte to exceed both the ER-L and the ER-M. Although naphthalene exceeded the ER-L, it did not exceed the ER-M.

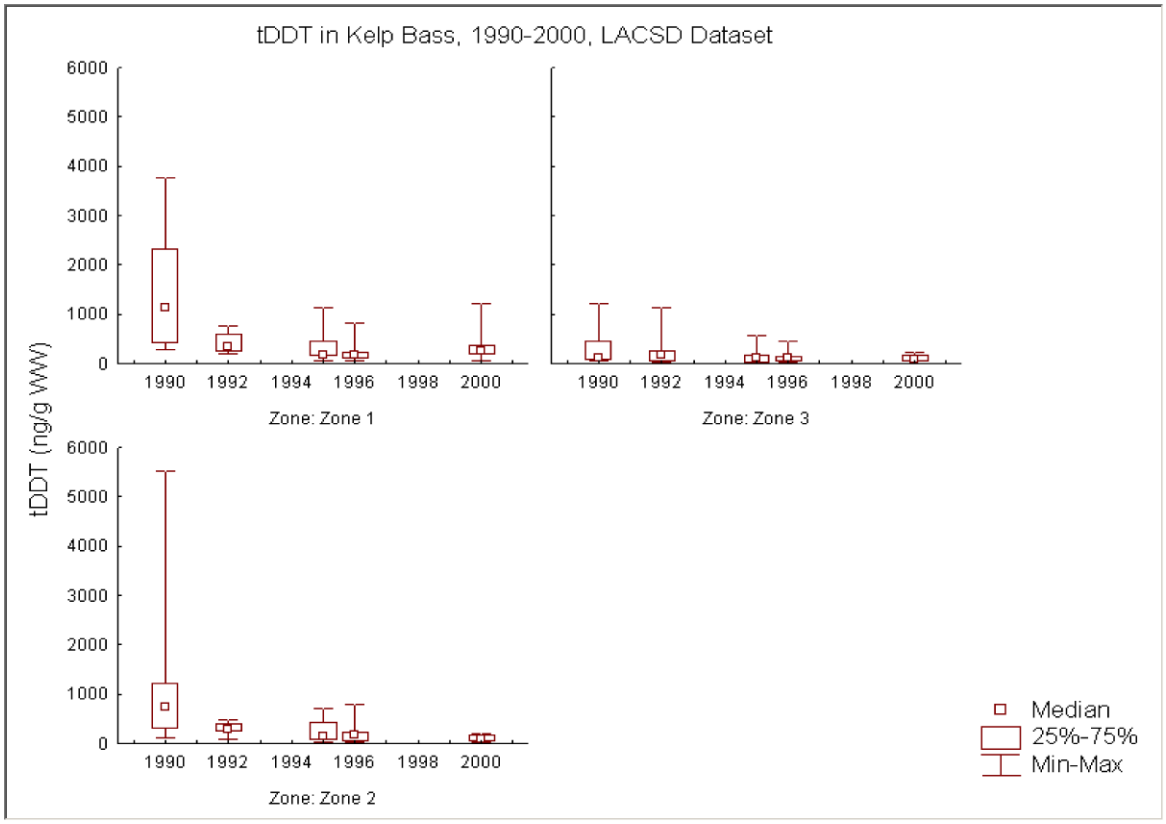


FIGURE 2-9
LACSD tDDT IN KELP BASS, 1990-2000
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

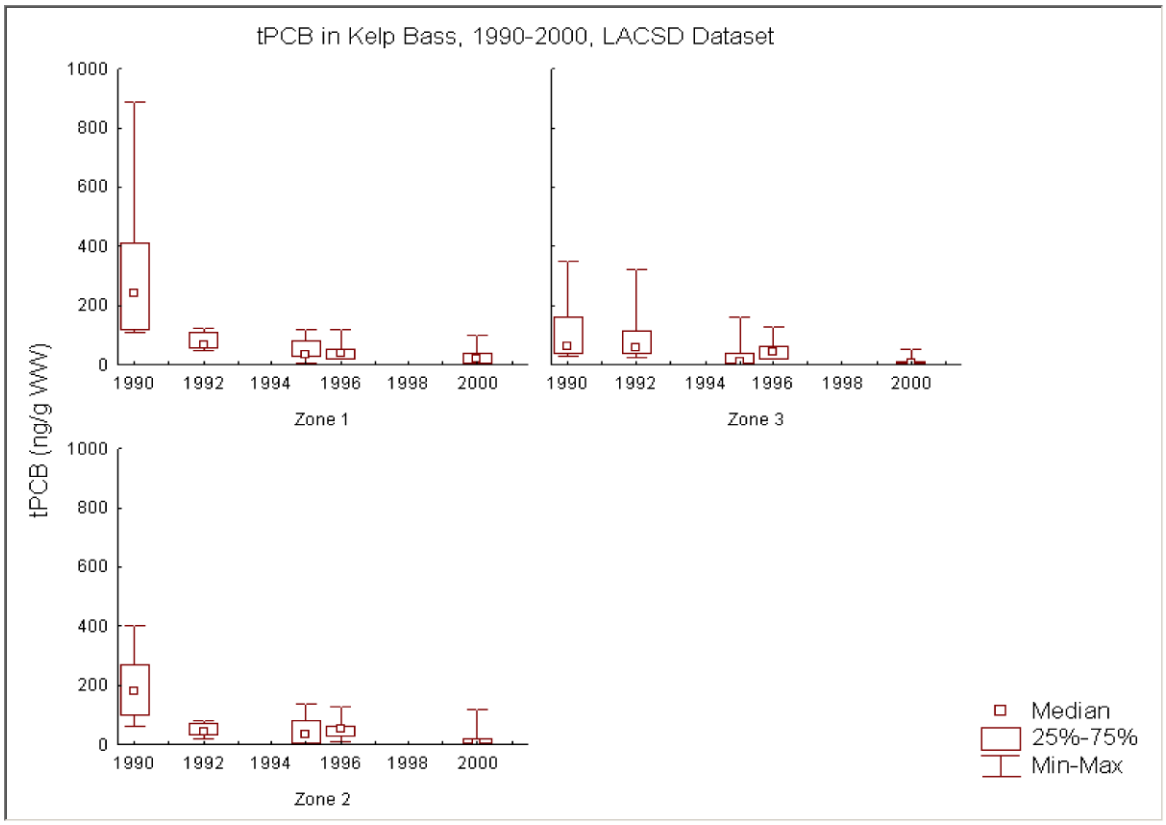


FIGURE 2-10
LACSD tPCB IN KELP BASS, 1990-2000
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

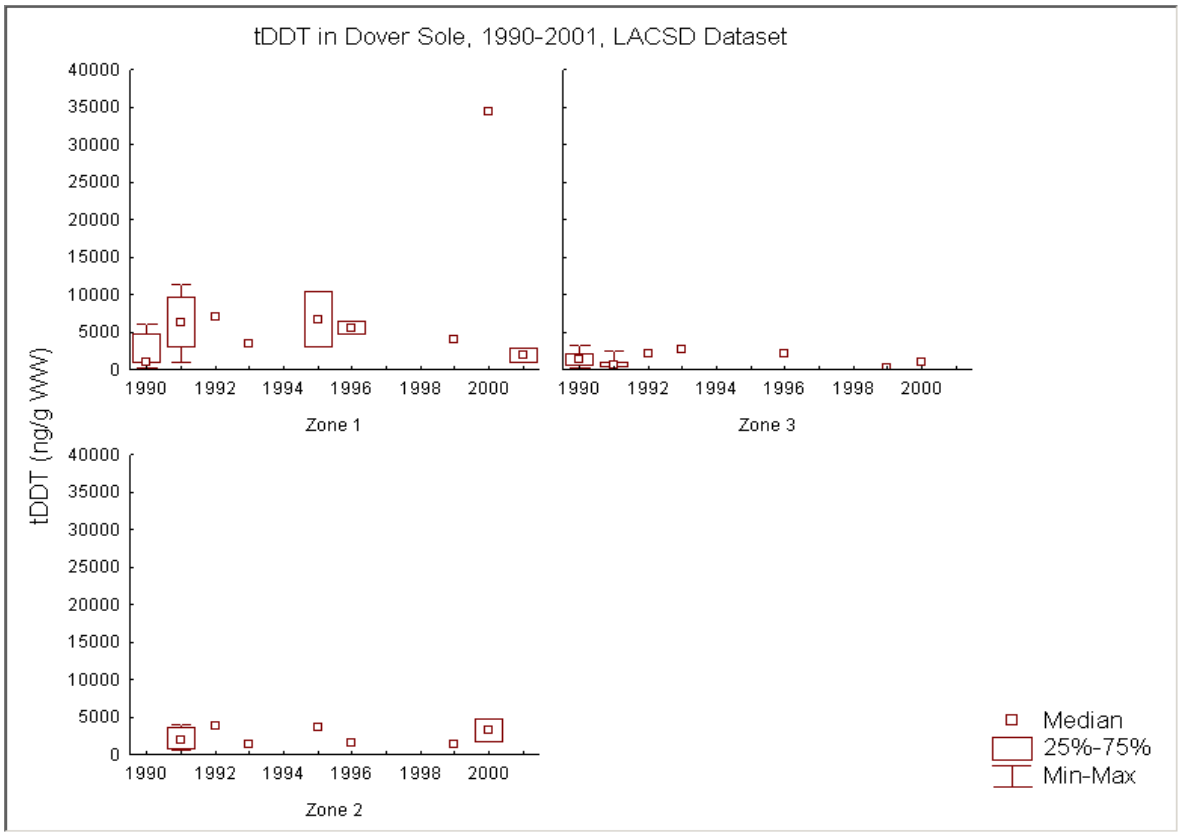


FIGURE 2-11
LACSD tDDT IN DOVER SOLE, 1990-2001
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

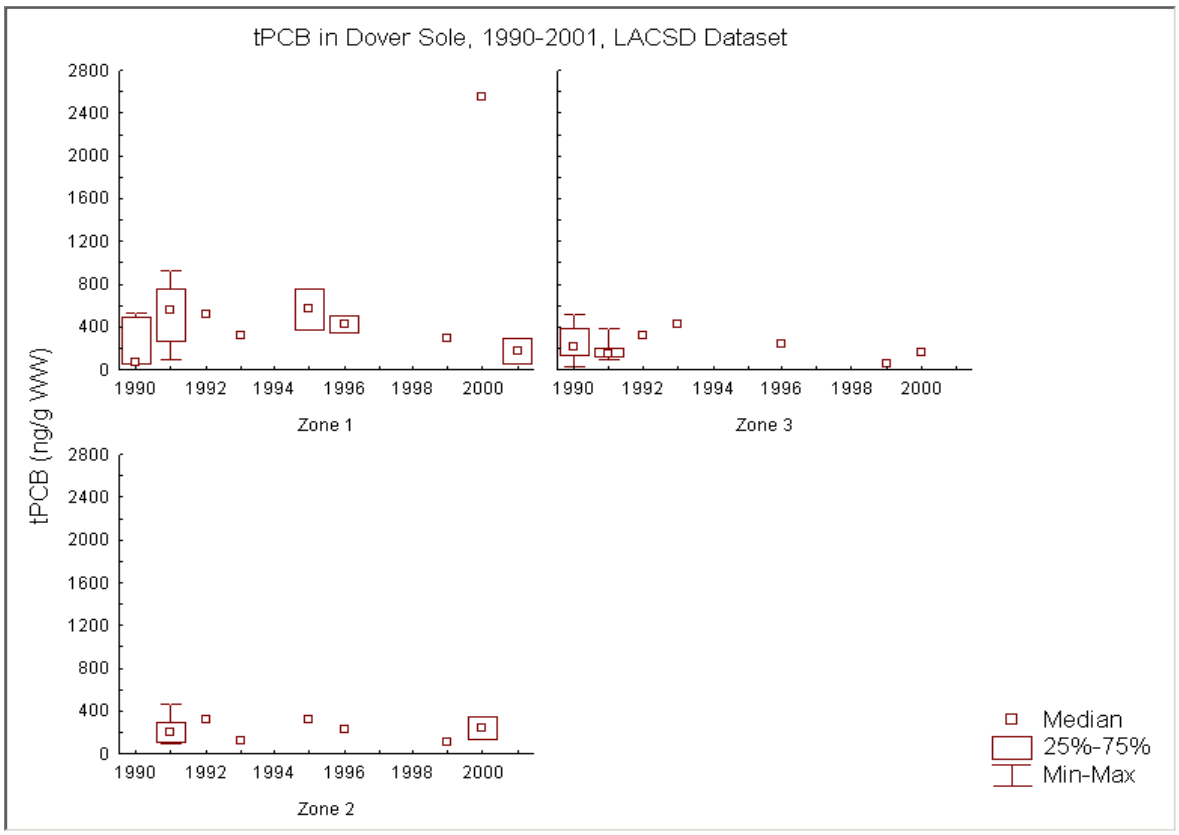


FIGURE 2-12
LACSD tPCB IN DOVER SOLE, 1990-2001
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

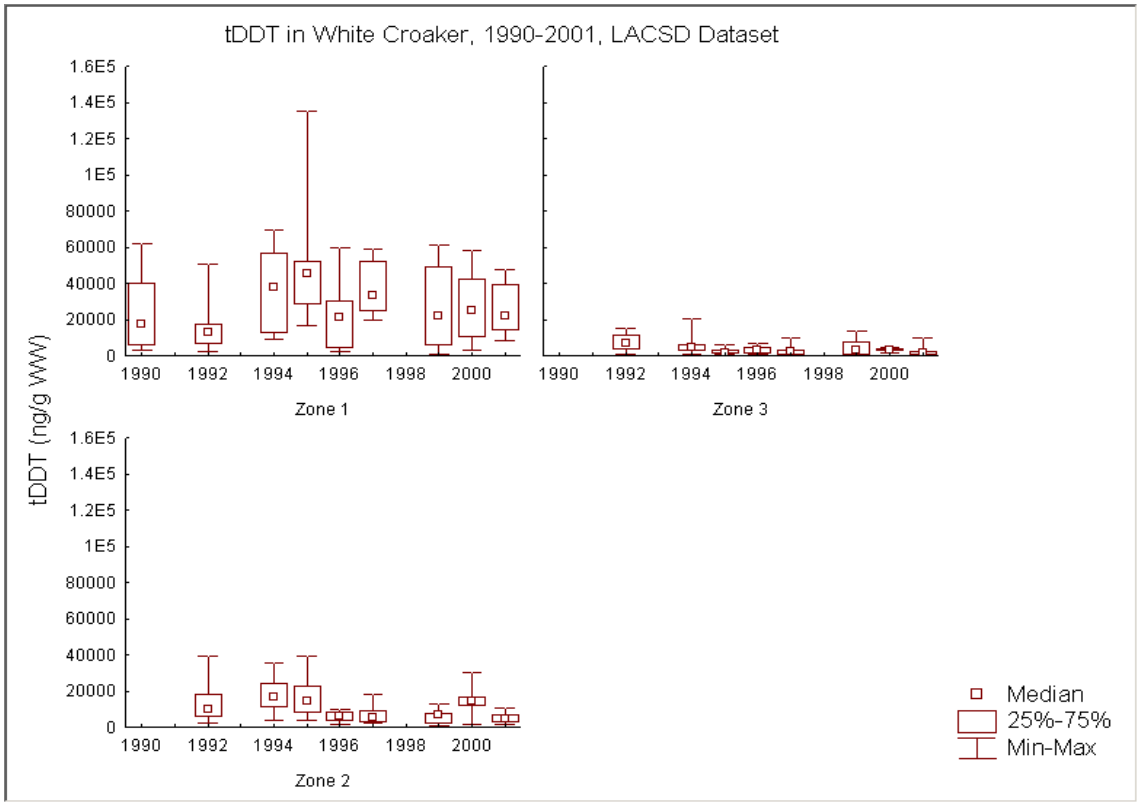


FIGURE 2-13
LACSD tDDT IN WHITE CROAKER, 1990-2001
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

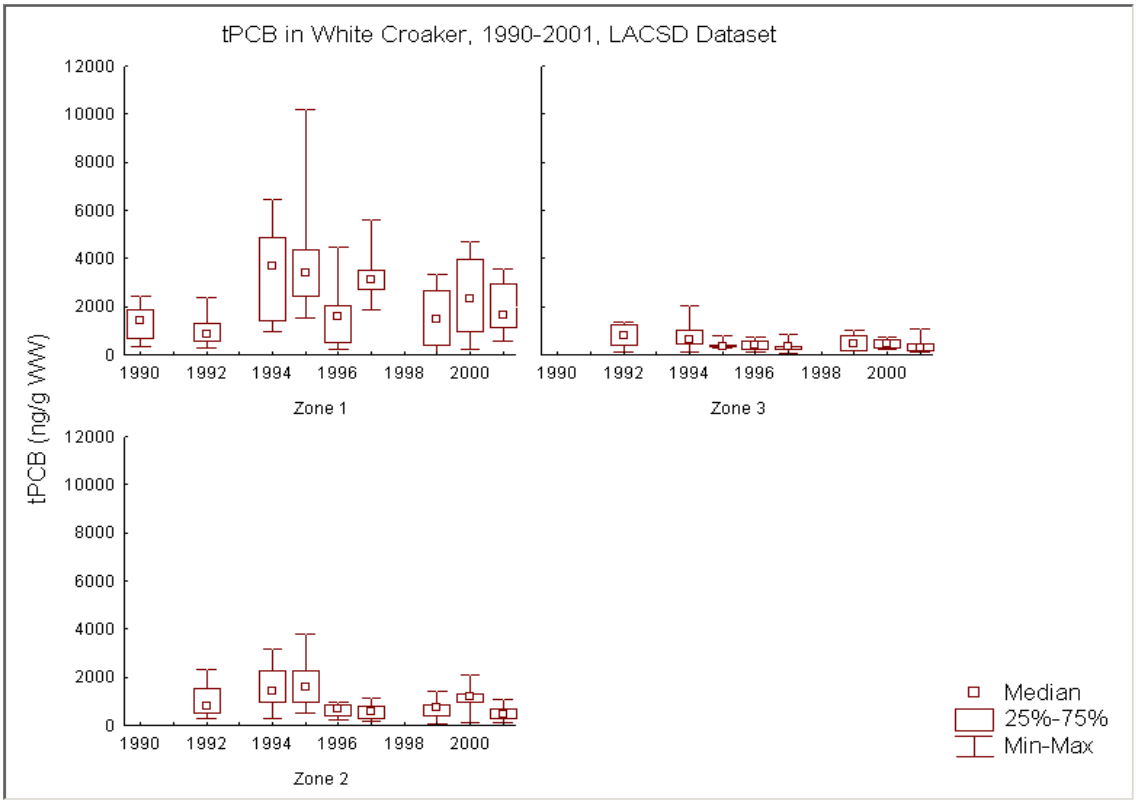


FIGURE 2-14
LACSD tPCB IN WHITE CROAKER, 1990-2001
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

According to the Bight 94 study (Schiff and Gossett 1997) sediment concentrations of arsenic, chromium, copper, lead, mercury, selenium, and zinc were comparable to concentrations observed nationally. This was based on the results of the National Status and Trends (NS&T) Program (NOAA 1994), which summarizes sediment contamination throughout the country. Concentrations of cadmium, silver, and total DDT, however, were three-fold higher in the SCB than other areas of the nation. Concentrations of antimony, total PCB, and total PAH were three-fold higher nationwide than in the SCB. Utilizing iron as a conservative tracer of natural contributions for eight metals of interest, it was estimated that 50 percent of the SCB was anthropogenically enriched in at least one trace metal. Cadmium, chromium, copper, lead, silver, zinc, and possibly mercury showed the greatest degree of anthropogenic enrichment; arsenic and nickel showed the least enrichment (Schiff and Gossett 1997).

The 1994 Bight report (Schiff and Gossett 1997) presents area-weighted means for each inorganic analyte, total DDT, total PCB, and total PAHs. Of these area-weighted averages, area in the SCB that had inorganic concentrations exceeding ER-Ls ranged from 0.5 to 7.3 percent. Area in the SCB exceeding contaminant thresholds for PAHs was 0 percent, whereas 63.7 and 15.3 percent of areas exceeded ER-Ls for total DDT and tPCB, respectively. Additionally, total DDT exceeded the ER-M in over 10 percent of the SCB area. This indicates that tDDT and tPCB in the sediments are of greatest concern in both spatial coverage and elevated concentrations throughout the SCB, particularly in the PVS study area.

2.4 Ecological Management Goals, Risk Hypotheses, Assessment Endpoints, and Measures

The identification of ecological management goals, risk hypotheses, assessment endpoints, and measures is critical to the ERA process. They provide the focus of the ERA and ensure that the methodologies and results of the ERA are technically sound. Most important, they ensure that the ecological values of the site are considered in the remedial decisionmaking process.

Ecological management goals, risk hypotheses, assessment endpoints, and measures for the PVS study area are described in the following subsections.

2.4.1 Ecological Management Goals

Ecological management goals are statements of the desired ecological conditions for the site. The ecological management goals for the PVS study area ERA include the following:

- Maintenance (or provision) of sediment, water quality, food source, and habitat conditions capable of sustaining a functional ecosystem for ecological receptors that comprise the marine food web, including receptors at primary, secondary, tertiary, and quaternary trophic levels
- Maintenance (or provision) of food source and habitat conditions supportive of individuals of special-status biota (including birds and mammals) and migratory birds (i.e., species protected under the Migratory Bird Treaty Act) that are likely to live or forage within the PVS study area and the SCB

Inherent in these ecological management goals is the need to reduce the toxicity and/or toxic effects of hazardous chemicals released into the PVS to special-status species, and also the need to provide habitat conducive to the recovery of special-status species. By protecting the integrity of the food chain, sediment, and other natural resources as well as habitat structure, the ecological management goals should be fulfilled. The assessment endpoints and measures to evaluate these goals are presented in the following subsections.

2.4.2 Risk Hypotheses

Risk hypotheses focus on the responses of the assessment endpoints when exposed to stressors and how the exposure could occur (USEPA 1998). Stressors in the PVS study area consist of chemicals that have been released from their primary sources to the environment via discharges of treated municipal and industrial effluent. Under current conditions, ecological receptors are exposed to contaminants in sediment, surface water, and/or biota. Based on the chemical stressors and potential exposure routes, the risk hypotheses for the PVS study area are:

- Total DDT and tPCB associated with historic effluent discharges are present in sediment and surface water at concentrations toxic to marine invertebrates, fish, birds, and mammals, as well as aquatic-feeding terrestrial birds.
- Total DDT and tPCB are bioaccumulating in forage and prey species for secondary, tertiary, and quaternary consumers, resulting in food-chain transfer of contaminants.

Under current conditions in the PVS, ecological receptors are exposed to COPECs by direct contact with sediments or surface water, or by incidentally ingesting them during activities such as feeding. If forage or prey species were contaminated from site-related chemicals, their consumers are secondary receptors via food-chain transfer. It is recognized that higher-order consumers (birds, marine mammals) can be exposed to PVS COPECs throughout the greater SCB area.

2.4.3 Assessment Endpoints and Measures of Exposure and Effects

The selection of assessment endpoints and measures is described below, followed by a summary of endpoints specific to community, population, and individual levels of assessment.

2.4.3.1 Assessment Endpoints

Assessment endpoints are an expression of the important ecological values that should be protected at a site (Suter 1990 and 1993; Suter et al. 2000; USEPA 1998). Assessment endpoints are developed based on known information concerning the contaminants present, the study area, the ecological management goals, and risk hypotheses. In addition, they must represent a property of the system that can be measured. There are three components to each assessment endpoint: an *entity* (e.g., migratory birds), an *attribute* of that entity (e.g., individual survival), and a *measure* (e.g., a measurable value, such as an effect level). Measures are described following the general description of assessment endpoints (USEPA 1998; Suter et al. 1995 and 2000).

The assessment endpoint entities for the PVS study area were selected based on the following principal criteria:

- Ecological relevance
- Political and societal relevance

- Susceptibility to known or potential stressors at the site
- Consistency with ecological management goals for the site

The attribute selected for each entity was based on the organizational level of the entity and the primary criteria that were used to select it. Entities and attributes were selected for community, population, and individual levels of assessment.

2.4.3.2 Measures of Exposure and Effects

The measures used in this ERA are predictive of the assessment endpoints (USEPA 1998). There are three categories of measures: exposure, effect, and ecosystem and receptor characteristics. Measures of exposure are used to evaluate how exposures could be occurring. Measures of effect are used to evaluate the response of the assessment endpoints when exposed to contaminants. Measures of ecosystem and receptor characteristics may be used to evaluate the ecosystem characteristics that could affect exposure or response to contaminants.

Measures identified for an ERA can be from one or more of these categories, depending on the complexity of the ERA. Criteria considered in the selection of measures are as follows:

- Corresponding to or predictive of an assessment endpoint
- Readily measurable or subject to evaluation
- Appropriate to the scale of the site
- Appropriate to the temporal dynamics
- Appropriate to the exposure pathway
- Associated with low natural variability
- Minimally disruptive to ecological community and species variability

The protection of assessment endpoints for the PVS and SCB region will be considered to result in a functional ecosystem if sediment, water quality, food source, and habitat conditions are capable of supporting natural, healthy populations of marine animals; there are no direct adverse effects on special-status species; and habitat conditions support recovery of special-status species. The assessment endpoints and measures are described in the following sections and are summarized in Table 2-14.

2.4.3.3 Community-level Endpoints

Assessment endpoints that pertain to potential effects within the PVS study area on marine infaunal and benthic macroinvertebrate communities characteristic of natural habitats in the region were identified.

- **Entity**—Marine infaunal and benthic macroinvertebrate communities
  - Attribute—Community composition, abundance, density, species diversity, proportion of pollutant-tolerant species, and community structure supportive of marine ecosystem processes (e.g., nutrient cycling, decomposition), as well as providing prey for higher trophic level receptors
  - Measures of exposure—Measurement of tDDT and tPCB in sediment, porewater, water column, and site-collected invertebrate tissues at PVS

TABLE 2-14
Assessment Endpoints and Measures
Palos Verdes Shelf Study Area Ecological Risk Assessment

Entity	Assessment Endpoint	Measures of Exposure	Measures of Effect
Community Level			
Sediment infauna and benthic macroinvertebrates	Community structure and function as shown by community composition, species abundance, density, diversity, proportion of pollution-tolerant species	Measured concentrations of COPECs in sediment, pore-water, and water column	Comparisons of sediment and porewater concentrations to benchmarks for growth rate, survival, or reproduction
		Biotic indices	Comparison of benthic invertebrate communities between impacted and reference sites to evaluate species richness, diversity, relative abundance, and presence/absence
	Support of higher trophic levels—as a source of prey	Observed toxicity to test species	Bioassays using site water and sediment to measure growth rate, survival, or reproduction
Population Level			
Benthic, water-column, and pelagic fish	Community structure and function as shown by species survival, reproduction, abundance, diversity	Measured concentrations of COPECs in water column	Comparisons of water-column concentrations to benchmarks for growth rate, survival, or reproduction
	Support of higher trophic levels—as a source of prey	Measured concentrations in tissues and target organs	Comparison to tissue benchmarks for fish
Individual Level			
Marine and aquatic-feeding birds (special-status)	Survival and reproduction	Measured concentrations of COPECs in sediment and water column; measured and estimated concentrations in prey	Comparisons of exposure-point concentrations to benchmarks for survival, reproduction, and/or other toxic effects that could affect survival or reproduction (NOAELs)
		Measured concentrations in tissues and target organs	Comparison to tissue benchmarks for birds (NOECs)

TABLE 2-14
Assessment Endpoints and Measures
Palos Verdes Shelf Study Area Ecological Risk Assessment

Entity	Assessment Endpoint	Measures of Exposure	Measures of Effect
Marine mammal (special-status)	Survival and reproduction	Measured concentrations of COPECs in sediment and water column; measured and estimated concentrations in prey; estimated concentrations in milk	Comparisons of exposure-point concentrations to benchmarks for survival, reproduction, and/or other toxic effects that could affect survival or reproduction (NOAELs)
		Measured concentrations in tissues and target organs	Comparison to tissue benchmarks for mammals (NOECs)

Notes:
EC20 effective concentration at which 20 percent of the test population shows effect.
LOAEL lowest observed adverse effect level
LOEC lowest observed effect concentration
NOAEL no observed adverse effect level
NOEC no observed effect concentration

- Measures of effects—Site-specific toxicity test results, assessment of benthic invertebrate abundance and community structure, comparisons to literature-derived sediment quality benchmarks

2.4.3.4 Population-level Endpoints

Assessment endpoints that pertain to potential effects on populations of fish species that are characteristic of natural habitats within the PVS region were identified.

- **Entity**—Fish
  - Attribute—Survival, growth, reproduction, species diversity, and abundance, supportive of community structure and function, as well as providing prey for higher trophic level receptors
  - Measures of exposure—Measurement of tDDT and tPCB in water column and site-collected fish tissues (whole body, fillet, and target organs) at PVS
  - Measures of effects—Comparisons to literature-derived toxicity benchmarks

2.4.3.5 Individual-level Endpoints

Assessment endpoints were identified based on potential effects on individuals of threatened or endangered species within the PVS/SCB region.

- **Entity**—Special-status marine and aquatic-feeding birds
  - Attribute—Survival, growth, and reproduction
  - Measures of exposure—Measurement of tDDT and tPCB in sediment, water column, and site-collected tissues (egg and target organs) throughout the SCB; measurement/estimation of tissue concentrations in prey items (invertebrates, fish, other birds, and marine mammals)
  - Measures of effects—Comparisons to literature-derived toxicity benchmarks for oral intake and target organ residues (including NOAELs and NOECs)
- **Entity**—Special-status marine mammals
  - Attribute—Survival, growth, and reproduction
  - Measures of exposure—Measurement of tDDT and tPCB in sediment, water column, and site-collected tissues (blubber and target organs) throughout the SCB; measurement/estimation of tissue concentrations in prey items (invertebrates and fish); estimation of concentrations in milk
  - Measures of effects—Comparisons to literature-derived toxicity benchmarks for oral intake and target organ residues (including NOAELs, NOECs)

2.5 Ecological Conceptual Site Model

This section presents the ecological conceptual site model for potential exposure by ecological receptors to site-related contaminants. The conceptual site model combines information on

potential ecological receptors; ecological management goals, assessment endpoints and measures; sources, fate, and transport of contaminants; potential exposure pathways; and selection of representative species to provide an overall picture of site-related exposures to focus the remaining evaluation of COPECs in this ERA (Figure 2-15).

2.5.1 Sources of Chemical Stressors

The PVS study area has been subject to discharges of municipal and industrial wastewater from several outfalls, beginning as early as 1937, from the LACSD JWPCP. The earliest outfall is located at White Point and discharged at a depth of 33 m, though this outfall is no longer in use. Additional outfalls constructed in later years range up to 60 m deep. Wastewater contaminants in the discharge have included chlorinated hydrocarbons (e.g., tDDT and tPCB) as well as trace metals (cadmium, copper, lead, and zinc) and organic matter. The primary source of tDDT was wastewater from Montrose, which manufactured DDT from 1947 to 1983. Sources of tPCB included various industries in the greater Los Angeles area.

The primary historical source of chemical contaminants to the PVS is effluent discharged through the White Point outfalls. Effluent-affected sediments in the area of the outfalls and those transported away from White Point by ocean currents and deposited on the ocean floor serve as the main repository for contaminants. Virtually all of the effluent deposits are contaminated with tDDT and tPCB.

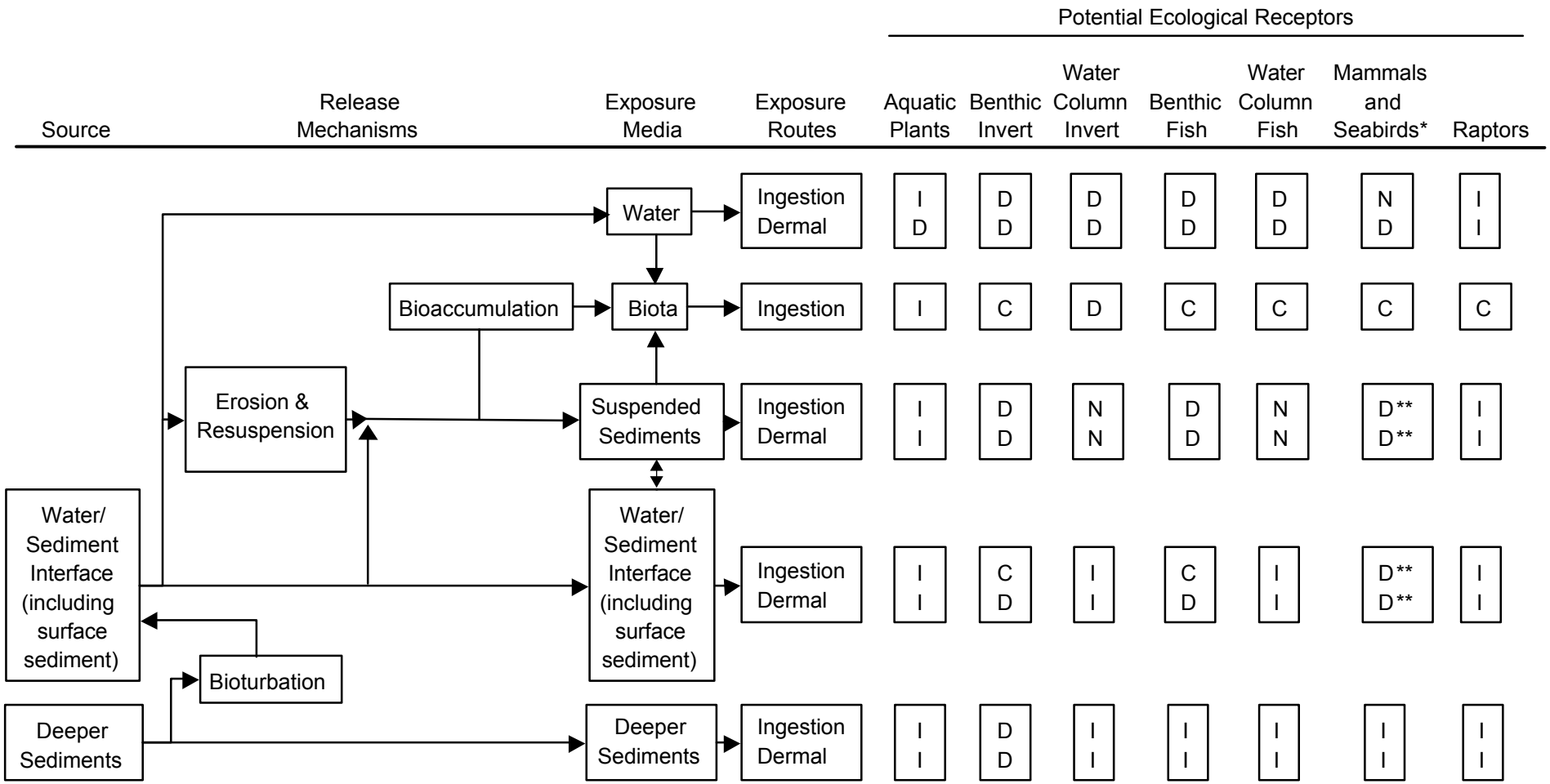
Although contaminant emissions decreased after the early 1970s and the contaminated sediments became buried by progressively cleaner deposits, surface-contaminant concentrations remain significantly elevated, reflecting mixing from deeper sediments. Contaminant concentrations in surface-sediment layers reflect both biological (bioturbation) and physicochemical (molecular) processes and the resuspension of particulates from waves, currents, and storms.

Contaminant flux from the sediment surface to the water column occurs, increasing contaminant concentrations in the water column above the sediments. Zeng et al. (1999) reported water-column concentrations ranging from 0.6-15.8 ng tDDT/L and 0.06-1.14 ng tPCB/L. These data were collected during the winter and summer of 1997 at 8 LACSD monitoring stations on the PVS. Based upon partitioning characteristics, Zeng et al. determined that the sediments were the source of these two chemicals in the water column and that flux of chemical into the water column is responsible for the decrease in tDDT mass in sediments of the shelf.

Other potential sources of contaminants (i.e., atmospheric deposition, stormwater runoff, and ocean currents) are considered minor in comparison to the existing reservoir of deposited contaminants (USEPA 2000).

2.5.2 Fate and Transport Processes

An understanding of the fate and transport mechanisms of the COPECs identified for the PVS is important for mapping potential exposure pathways. Fate and transport factors considered for the PVS sediment deposits include chemical persistence, stability in sediment and water, bioaccumulation potential, and chemical mobility via processes such as advection, molecular diffusion, and bioturbation



C = Complete exposure pathway
N = Possibly complete exposure pathway, but likely not significant
D = Possibly complete exposure pathway, but insufficient data for quantitative analysis
I = Incomplete exposure pathway or not applicable
* Pathways may not be quantitatively similar for mammals and seabirds; also, transfer of contaminants from mother to fetus and from mother to pup via milk ingestion is an important exposure pathway in marine mammals
** Not significant pathway for some seabirds

FIGURE 2-15
ECOLOGICAL CONCEPTUAL SITE MODEL
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

Long-term accumulation of effluent-affected sediments resulted in a surface layer of more recently deposited and moderately contaminated materials covering a buried layer of highly contaminated materials that were deposited prior to 1980. Decreases over time in concentrations of tDDT in surface sediments have been smaller than reductions in the combined mass emissions from the JWPCP and other potential sources. This implies that contaminants in historically deposited sediments are being remobilized and contribute to concentrations in the more recently deposited materials. The portions of the existing contaminant reservoir that may become biologically available are determined by the combined effects of several physical, chemical, and biological processes that control the fate and transport of contaminated sediments. Processes responsible for contaminant remobilization and burial are discussed in this subsection. A generic discussion of the physicochemical properties of tDDT and tPCB is provided, along with descriptions of the site-specific transport pathways (including estimated loss rates where available) that have implications for predicted future ecological risk. Much of this latter information is from the USGS process studies and natural recovery model (Lee 1994; Drake 1994).

2.5.2.1 Physicochemical Properties

Physicochemical properties (e.g., solubilities and partitioning tendencies) are important for evaluating environmental fate. The mobility and partitioning potentials of chemical analytes identified in PVS media were evaluated on the basis of selected physicochemical properties such as solubility in water and lipids; octanol-water partition coefficient (K_{ow}), organic carbon-water partition coefficient (K_{oc}), and vapor pressure (a measure of volatility). Physicochemical properties reflect the potential for adsorption to organic matter and binding to sediment. Partitioning between sediment and sediment porewater influences uptake, distribution, and storage in sediment-dwelling biota. Persistent, lipophilic, low water-soluble chemicals tend to have high bioaccumulation and biomagnification potentials, and exposure pathways that are generally food web-related.

Total DDT

Organochlorine pesticides (e.g., DDT and its degradation products, primarily DDE and DDD) are not easily metabolized by microorganisms and, therefore, persist in the environment. DDE is more stable and more persistent in the environment than DDT and DDD and is found more frequently at higher trophic levels (Stickel 1973). Other degradation products include DDMU, DDNU, and DDA. Total DDT have low water solubilities (1.2-140 µg/L in freshwater) and low vapor pressures (19×10^{-7} to 5.5×10^{-6} torr). Despite their low volatility, tDDT have been dispersed through the atmosphere to most parts of the world. Total DDT are highly soluble in lipids (indicating a high biomagnification potential) with log K_{ow} values from 6.1 to 6.76 (Karickhoff and Long 1995). High K_{oc} values for tDDT (1,600 to 15,200 ml/g) indicate tDDT will strongly sorb to organic matter in sediments. In aquatic systems, tDDT have strong affinities for particles and, therefore, are rapidly scavenged from water and deposited with carrier particles. This produces a strong tendency for these contaminants to accumulate in bottom sediments.

Total DDT readily bioaccumulate in aquatic organism tissues, especially lipid-rich tissues (i.e., vertebrate livers) and biomagnify in higher trophic-level organisms. Organisms exposed to contaminated sediments can be expected to accumulate tissue contaminant concentrations that are directly proportional to organic carbon-adjusted sediment contaminant levels. Depuration rates of tDDT are typically slow, although some species-specific differences

have been reported. Avian species and marine mammals that are highly piscivorous or omnivorous in their feeding habits bioaccumulate much higher concentrations in their body tissues than organisms at lower trophic levels.

Anaerobic degradation of DDE, the predominant DDT metabolite in PVS sediments, to products of presumed relatively lower toxicity (i.e., DDMU) has been demonstrated under laboratory conditions (Quensen et al. 1998, 2000). The metabolite DDMU is present in PVS sediments at concentrations that are proportional to sediment DDE concentrations, suggesting that *in situ* degradation of DDE may be occurring at the site. Assuming that degradation of DDE to DDMU occurs, the existing data are not sufficient to evaluate (1) when during the process of particle settling, deposition, and burial this conversion occurred; (2) the present rate of conversion; and (3) the rate limiting or enhancing factors. In general, the existence and rate of *in situ* degradation are very difficult to assess under realistic exposure conditions because the proof must demonstrate that the contaminant mass has decreased (as opposed to dispersing in the environment) and that microorganisms are the causative agents (Madsen 1991). Longer-term degradation rates may depend, in part, on the nature of the organic matrix of the sediment and availability of the DDE to microbial metabolism. Further studies are needed to determine whether and at what rates degradation of DDE is occurring in the buried sediments on the PVS and the potential implications for long-term changes in the present contaminant mass. In fact, detailed field studies have demonstrated that in-situ degradation of DDE in PVS sediments is 100 to 1,000 times slower than degradation rates from laboratory experiments (e.g. Quensen et al. 1998; Eganhouse et al. 2000). In addition, field assessments of DDE degradation at PVS indicate that degradation rates are highest in the areas of lowest DDE concentrations and quite variable over the PVA area (Sherwood et al. 2002). Sherwood et. al (2002) conclude that both losses and transformations are occurring at PVS but that burial and re-erosion are central processes in determining the pattern of contamination.

Total PCB

Since the mid-1970s, production and use of tPCB have been curtailed because of their chronic toxicity and environmental persistence. In the United States, PCB mixtures, called Aroclors, were identified according to the percentage of chlorine (e.g., Aroclor 1221 contains 21 percent chlorine). The group of tPCB includes 209 isomers/congeners of synthetically halogenated hydrocarbon compounds. Each of the 209 congeners differs in the number and position of chlorine atoms on the biphenyl ring, their toxicity, and their persistence in biota. Of the 209 individual congeners, 5 to 10 are commonly found that are considered the most toxic (Safe 1991). Structure-activity relationships have identified congeners, such as 2,5,2′,5′-tetrachlorobiphenyl (PCB 52) and 2,3,6,2′, 3′, 6′-hexachlorobiphenyl (PCB 136), as nontoxic, while others, such as 3,4,3′, 4′-tetrachlorobiphenyl (PCB 77) and 3,4,5,3′,4′,5′-hexachlorobiphenyl (PCB 169), are two of the most potent congeners (Hansen 1994).

The relative importance of environmental fate mechanisms for tPCB generally depends on the degree of chlorination, with persistence usually increasing with increasing chlorination. Mono-, di-, and trichlorinated biphenyls (primary composition of Aroclor 1221, 1232, and 1016) biodegrade relatively rapidly, tetrachlorinated biphenyls (Aroclor 1242) biodegrade slowly, and higher-chlorinated biphenyls (Aroclor 1248, 1254, 1260, and 1268) are more resistant to biodegradation. Consequently, residual concentrations of these congeners

are expected to be higher in weathered PCB Aroclors in sediments. The viscosity of PCB congeners increases and solubilities decrease as chlorine content increases (e.g., Aroclor 1221 is a mobile oil while Aroclor 1260 is a thick resin). Volatilization is generally not considered an important transport process for high-chlorine tPCB because of their high viscosity. Total PCB are generally hydrophobic (solubility of Aroclor 1260 in water is 0.0027 mg/L).

Adsorption onto particles is a primary process affecting the fate of tPCB with a relatively high organic carbon-water partition coefficient (K_{oc} values generally greater than 5,000; Aroclor mixtures about 45,400 ml/g) and low water solubility (Aroclor mixtures range from 0.027 to 0.59 mg/L). Total PCB, especially the higher-chlorinated congeners, do not readily partition from sediments into water. Factors promoting particle adsorption include higher organic content and smaller particle size. PCB concentration factors are inversely related to the surface area of adsorbents (Hiraizumi et al. 1979). Adsorption is enhanced in the presence of large amounts of suspended particles. Thus, PCB concentrations in sediment and suspended matter tend to be higher than in the associated water column.

Aquatic organisms have a strong tendency to accumulate tPCB from water and food sources. The bioconcentration factor for fish is approximately 50,000 (USEPA 1991). This factor represents the ratio of concentration in tissue to the ambient water concentration. Total PCB have high octanol-water partition coefficients (log K_{ow} ranges from 4.7 to 6.8 for Aroclor mixtures) and, therefore, are highly lipophilic and biomagnify through the food web. PCB concentrations in tissues of aquatic organisms will generally be greater than, or equal to, sediment concentrations (Neff 1984). Studies on Lake Ontario indicated a progressive increase in tissue PCB concentrations moving from organisms lower in the food web to top aquatic predators (Oliver and Niimi 1988). PCB concentrations in fish have been strongly correlated to their lipid content. Elimination of tPCB from organisms is related to the characteristics of the specific PCB congeners. Uptake rates increase while depuration rates decrease with increasing degree of chlorination. Elimination of tPCB from the body can occur during egg production and spawning in females of some species (Lech and Peterson 1983; Stout 1986). There is a limited capacity for fish and other aquatic organisms to transform or metabolize tPCB.

2.5.2.2 Molecular Diffusion

Under equilibrium conditions, some portion of the buried tDDT and tPCB will partition from sediments to porewaters, resulting in porewater concentrations that potentially are much higher than concentrations in the overlying water column. The concentration gradient from the porewater to waters above the sediment-water interface drives contaminants out of the sediments and into the water column. Rates of diffusion from porewater depend on the strength of the gradient, porosity of the sediments, and aqueous diffusivity of the dissolved compound.

2.5.2.3 Bioturbation

Bioturbation is a biologically mediated mechanism for redistribution/remobilization of buried contaminants in dissolved (colloidal) or particulate phases. Conceptually, removal of contaminants from sediments by organisms, via uptake from water, sediments, and/or suspended particles, represents an important process for food-chain transfers from the sediment reservoir to other, higher trophic-level organisms, including humans. There are

three general forms of bioturbation. Biodiffusion refers to a random displacement of particles by burrowing organisms within the mixed layer of surface sediments. This type of diffusion promotes contaminant transport in the direction of the concentration gradient. Biologically enhanced diffusion refers to porewater irrigation by tube-dwelling organisms. Non-local mixing refers to unidirectional advection of buried particles toward the sediment surface as a result of feeding behavior by infaunal organisms.

The degree of contaminant remobilization owing to bioturbation is related to the distribution of organisms within the sediment column and the extent of contamination within that area of the sediment. Boudreau (1998) states that the average depth of bioturbation in marine and lacustrine environments is 9.8 (± 4.5) cm. Stull et al. (1996) reported that most (>95 percent of individuals and 60-80 percent of the biomass) infaunal organisms on the PVS occur within the surface 10-cm layer, although they also observed burrowing organisms at depths of 35 cm below the surface. USGS observed spatial differences in the abundance of macrofaunal organisms within the surface 2-cm layer. In contrast, numbers of organisms at depths below 8 cm were spatially uniform, and organisms at depths greater than 20 cm were relatively rare throughout the area. Nevertheless, the presence of deep-burrowing organisms is important because these typically are larger species (e.g., thalassinid shrimp) that are capable of mixing relatively large amounts of sediments and, if present in reasonable abundance, may contribute substantially to the biological remobilization of buried contaminants.

This depth stratification for biological activity results in layers with varying mixing rates. The surface layer (0-15 cm) is referred to as the complete mixing layer. The deep layer (15-30 cm), which experiences periodic mixing by deep-burrowing organisms, is the biodiffusion zone. Sediments below the surface and subsurface mixing layers are within a zone of permanent burial. Deeper sediments (to 1 m) are considered for evaluation at PVS, but the surface and deeper layers (to 30 cm) capture the full spatial pattern of contamination required for this evaluation.

Rates of biological mixing are represented by a biodiffusivity coefficient and are estimated from measurements of vertical profiles of selected radionuclides with relatively short half-lives. Vertical mixing rates determined by USGS ranged from 6 to 71 cm^2/yr, with average rates for sites 3C and 6C of 49±12 cm^2/yr and 23±3 cm^2/yr, respectively. Relatively lower rates occurred near the JWPCP outfall, but rates increased with distance to the northwest. Mixing rates were assumed to be uniform within the surface 10 cm, decrease exponentially to 25 cm, and then decrease linearly to zero at 30 cm. This assumption is generally consistent with ^{210}Pb profiles presented by Swift et al. (1996).

2.5.2.4 Transport

Transport of resuspended sediments is considered an important process because contaminants such as tDDT and tPCB have strong affinities for particles. Thus, physical transport of sediment also results in dispersion and redistribution of adsorbed contaminants. Although organochlorines are considered particle-reactive, some losses caused by desorption from sediments may occur when sediments are resuspended by physical disturbance or biological activities.

Sediments can be transported by near-bottom currents as suspended particles or as bedload. Resuspension is the result of the combined force of oscillatory currents associated with surface waves and those of low frequency (e.g., tidal) currents. Wave-generated stresses typically are more important than mean currents in generating stresses to bottom sediment. Thus, transport events are generally associated with storms and the effects of storm-related waves. Current velocities must exceed threshold values to initiate movement; threshold values are relatively lower for smaller-sized particles and higher for larger particles. The frequency and depth within the sediment layer of physical mixing are greater in shallow water where oscillatory currents caused by surface gravity waves are stronger and exceed thresholds for erosion more frequently. Resuspension of sediments becomes less important at depths > 45 m, although the presence of ripples in surface sediments at 60-meter depths observed by Drake (1994) and extensive resuspension of bottom sediments up to 20 m above the bottom (in water depths of 60 meters) observed by Wu et al. (1994), are evidence for physical mixing in areas of the mid-shelf.

Contaminant desorption rates are related to: (1) rates of sediment resuspension, (2) duration of resuspension events, (3) distributions of contaminants with different particle sizes, and (4) desorption coefficients (rates) as a function of particle size. Smaller grain sizes contain proportionately higher contaminant concentrations because they have larger surface-to-volume ratios and contain relatively higher organic content than larger-sized particles. The nearshore, relatively low concentration of organochlorine contaminants in PVS sediments is probably closely related to the greater grain size of nearshore sediments (Lee, 1994). In addition, contaminant desorption is more rapid for fine-grained sediments than for coarser sediments. For the USGS natural recovery model, Wiberg (1994) assumed that all the DDE adsorbed to particles smaller than 30 µm was lost during resuspension, while none of the DDE adsorbed to particles larger than 30 µm was desorbed. Recent estimates of resuspended sediment dynamics at PVS indicate a desorption loss of 25 to 50 percent of the mass of DDE in resuspended particles (Wiberg and Harris 2002).

Wave-induced stresses produce reversing motions that initiate sediment movement but do not cause net transport. Horizontal transport is provided by the actions of long-term, low-frequency currents. Drake (1994) concluded that there was little evidence that large storms cause appreciable net erosion on the shelf. Instead, sediments eroded from the bottom appear to be redeposited locally. Nevertheless, desorption results in partitioning of contaminants from particulate to aqueous phases, which are then subject to current (advective) transport. Consequently, this process is responsible for the redistribution of contaminants from areas of high concentrations near the JWPCP outfall to upcoast areas of relatively lower contaminant concentrations with little displacement of the sediment deposit (Wiberg et al. 2002; Wiberg 1994).

In general, the most important processes governing the distribution and transport of sediment contaminants in the PVS area appear to be a complex pattern of burial of older deposits by cleaner surface sediments (Lee et al. 2002), coupled with resuspension and desorption of contaminants (Zeng and Venkatesan 1999), and redeposition of sediments and contaminants (Sherwood et al. 2002; Wiberg et al. 2002) following the predominant currents northwestward along the continental shelf (Noble et al. 2002).

2.5.2.5 Burial

Effluent-affected sediments are characterized as a two-layer deposit covering the "clean" native sediments that were present on the shelf and slope prior to the start of discharges of DDT—or PCB—containing effluent from JWPCP ocean outfalls. The effluent-affected deposit on the shelf comprises a buried layer with highly contaminated sediments (maximum tDDT and tPCB concentrations exceeding 200 mg/kg and 20 mg/kg, respectively) and a surface layer 10-30 cm thick containing relatively lower, but still significantly elevated, concentrations (e.g., > 1 mg/kg tDDT). This distribution generally reflects the deposition history, with some post-depositional alterations resulting from physical and biological mixing. The rates at which new sediments accumulate, or surficial layers are eroded, influence either the continued burial of contaminants or the remobilization of contaminants into areas where they become biologically available. Future changes in the supplies of particles to the shelf have important implications for potentials for net burial versus erosion at different locations on the shelf.

USGS (Lee et al. 2002; Drake 1994) concluded that rates of sediment deposition and accumulation on the shelf are controlled primarily by effluent solids emissions from the JWPCP and erosion of natural sediments from the Portuguese Bend landslide. Spatial differences across the shelf in net sediment accumulation rates reflect the relative contributions from these two sources. Accumulations of effluent solids are highest near the outfall and then decrease with increasing distance in upcoast and downcoast directions. For example, the solids deposition rate at a site 8 km from the outfall is approximately half of that at sites within 2 km of the outfall (Drake 1994). Because the net transport of effluent particles is predominantly to the northwest, the gradient for depositional rates is relatively steeper to the southeast. Within the inner-shelf region, sedimentation rates decrease to the southeast, in the direction of the nearshore littoral drift, at greater distances from Portuguese Bend (Wiberg et al. 2002).

Sedimentation rates on the mid- and outer-shelf regions, prior to initiation of effluent discharges from JWPCP, were estimated at 0.1 cm/yr (Drake 1994), while deposition rates on the inner shelf were probably less than 0.1 cm/yr. Deposition rates were higher in the 1940s, then increased dramatically in 1956 in response to the Portuguese Bend landslide and installation of the new LACSD ocean outfall off White Point. During the early 1970s, coinciding with periods of maximum solids mass emissions from JWPCP, deposition rates at site 6C may have reached 3-4 cm/yr. Subsequent declines of approximately 50-60 percent occurred during the 1980s at 60-m sites despite continuing erosion at the Portuguese Bend landslide. These decreases were attributable to reductions in solids emissions from the outfall (Drake 1994).

Sediment accumulation rates on the shelf during 1983 and 1991 were calculated from tDDT concentration profiles in cores collected biennially by LACSD (Lee 1994). Average deposition rates of 0.7 cm/yr. (standard error ±0.1 cm/yr.) and 0.4 cm/yr. (standard error ± 0.3 cm/yr.) were estimated for 60-m sites 6C and 3C, respectively. Evaluations of tDDT concentration profiles in cores from the slope suggested sedimentation rates of a few millimeters per year or less during the period 1983-1991 (Lee 1994), although longer-term (during 1970-1992) averages were estimated at 0.1 cm/yr.

Future (predicted) deposition rates are related to predicted changes in effluent solids mass emissions from LACSD and continued supply of sediments from the Portuguese Bend landslide (Drake 1994). Solids emission rates from the JWPCP are expected to decrease by approximately 30 percent from present levels because the JWPCP switched to full secondary

treatment in 2003. Sediment supply from the Portuguese Bend landslide is also expected to decline through 2003, due to erosion control measures. Sedimentation rates at site 6C are predicted to continue at the present rate (approximately 0.6 cm/yr.) until the year 2003, and then decrease to -0.6 cm/yr. (i.e., net erosion) following implementation of secondary treatment by the JWPCP. Net erosion is expected to occur during 2003 through 2035 (Drake 1994). Equilibrium conditions would be achieved after 2034, and the net sediment accumulation rate is expected to be zero, except possibly in areas within the immediate vicinity of the outfall, where a small positive accumulation rate will occur.

Other, more recent studies, provide somewhat contradictory predictions, as current comprehensive studies of ocean currents and sediment core contaminants suggest that most of the PVS area will continue to remain depositional, characterized by burial with new, relatively clean sediments (Sherwood et al. 2002). Future average deposition rates are expected to be approximately 0.30 cm/yr (60 percent of the current rate) with a range of 0.14 to 0.58 cm/yr. However, the area southeast of the outfalls is an active erosional zone that will act to expose deeper, more contaminated sediments, thus reintroducing historic DDT and PCB deposits to the upper sediment layers.

In summary, present deposition rates promote continued burial of subsurface pockets of highest contaminant concentrations in the mid- and outer-shelf regions, while sediment accumulation in slope areas is relatively reduced (Sherwood et al. 2002). Net accumulation of sediments in shallow areas is occurring at present, although selective erosion of effluent-affected sediments from inner-shelf areas likely is responsible for contaminant mass decreases in this portion of the shelf. Future changes in solids mass emissions from the JWPCP and rapid decreases in sediment supplies from the Portuguese Bend landslide are expected to reduce burial rates on the shelf. In fact, the USGS natural recovery model predicts net erosion at mid-shelf sites following implementation of full secondary treatment by the JWPCP. The effect of sediment erosion is to reduce the thickness of the surface layer that presently covers the pockets of highest contamination. In addition, some areas of buried contaminants may be exposed through erosion along the “leading edge” of predominant bottom currents that strike the deposited sediments from the southeast. The net result of these depositional and erosional processes has been very little change in the DDE inventory of PVS since 1990 (Zeng and Vankatesan 1999; Lee et al. 2002).

2.5.3 Exposure Pathway Analysis

Exposure pathways refer to the media and routes through which inorganic and organic contaminants may reach ecological receptors. Environmental media may be affected by contaminants originating from previously discussed primary or secondary sources. Exposure pathways are considered to be potentially complete or incomplete, depending on whether COPECs from primary or secondary sources have the potential to affect ecological receptors now or in the future.

Potential exposure pathways must meet specific criteria for an exposure to occur. Aside from necessary habitat for ecological receptors, a complete exposure pathway must satisfy the following elements:

- Contaminant source (e.g., chemicals in sediment)
- Mechanism for contaminant release and transport (e.g., erosion)
- Exposure point (e.g., sediment/water interface)

- Feasible route of exposure (e.g., ingestion)
- Receptor (e.g., bird)

Aquatic exposures and food-web relationships are the major exposure pathways in the PVS study area. Total PCB and tDDT are strongly sorbed to organic matter and sediments and can result in direct exposure to contaminants in the sediment and sediment porewater. However, owing to the highly lipophilic nature of these COPECs and their propensity to biomagnify through the food web, indirect contact through the consumption of food items is considered the major pathway of exposure to organisms in the upper trophic levels. Sediment contaminants can also be released to the water column during resuspension of sediments and through diffusional flux to the water column from the sediment surface, representing a secondary exposure pathway. The exposure pathway analysis is summarized for each receptor group in Table 2-15.

TABLE 2-15
Exposure Pathway Analyses
Palos Verdes Shelf Study Area Ecological Risk Assessment

Receptor	Exposure Medium	Potential Exposure Pathways	Exposure Pathway Potentially Complete	Pathway Retained for Analysis	Rationale
Aquatic plants	water column	absorbtion/ uptake	yes	no	Water column risks are assessed via food-web model (uptake via fish and higher trophic levels). Marine plants (e.g., kelp and other algae) are exposed to water-column contaminants rather than sediments and cannot be directly compared to sediment benchmarks. Marine plants are not particularly susceptible to toxicity from tDDT and tPCB.
Water column invertebrates	water column	injection (planktonic food web)	yes	no	Water-column invertebrates (zooplankton) are ingesting water-column plants (phytoplankton). Water-column data are limited and concentrations of COPECs are much lower than in sediments.
Sediment infauna and macroinvertebrates (benthic invertebrates)	sediment/ sediment porewater	ingestion	yes	yes	Ingestion of sediment/surface water is considered the primary route of exposure to COPECs.
		dermal contact	yes	yes	Invertebrates, especially benthic species, are often in direct contact with COPECs that can be readily transmitted through moist dermal layers.
		food-web transfer	yes	no	Food-web transfer could occur through intake of contaminated plants or aquatic invertebrates. Food-web transfer is assessed via evaluation of higher trophic levels that use invertebrates as prey items.
Benthic and water column fish	sediment/ water column	ingestion	yes	yes	Ingestion of sediment/surface water is considered the primary route of exposure to COPECs.
		dermal contact	yes	yes	Benthic species are often in direct contact with COPECs that can be readily transmitted through moist dermal layers.

TABLE 2-15
Exposure Pathway Analyses
Palos Verdes Shelf Study Area Ecological Risk Assessment

Receptor	Exposure Medium	Potential Exposure Pathways	Exposure Pathway Potentially Complete	Pathway Retained for Analysis	Rationale
		food-web transfer	yes	yes	Food-web transfer could occur through intake of contaminated marine plants and/or sediment infauna or benthic macroinvertebrates.
Marine and aquatic-feeding birds (seabirds and raptors)	sediment	ingestion	yes	no	Ingestion of sediment is considered a minor route of exposure to COPECs. Information for a quantitative analysis of this pathway is also lacking.
		dermal contact	yes	no	Dermal contact with sediment is considered a minor route of exposure, and little toxicity information exists on dermal contact with COPECs.
		food-web transfer	yes	yes	Food-web transfer could occur through intake of contaminated invertebrates and/or fish, as well as birds and/or marine mammals for raptors.
Marine mammals	sediment	ingestion	yes	no	Ingestion of sediment is considered a minor route of exposure to COPECs in these media. Information for a quantitative analysis of this pathway is also lacking.
		dermal contact	yes	no	Dermal contact with sediments is considered a minor route of exposure, and little toxicity information exists on dermal contact with COPECs.
		food-web transfer	yes	yes	Food-web transfer could occur through intake of contaminated invertebrates and/or fish. Additionally, transfer of contaminants from mother to pup could occur by direct transfer to the fetus and through intake of milk by the pup.

2.5.3.1 Food-Web Transfer

A generalized contaminant food-web transfer diagram is presented in Figure 2-16. Sufficient data are available to allow evaluation of the highlighted components of this food web. Primary trophic-level organisms form the basis of the food chain, and their exposures to contaminants occur in the sediments or in the overlying water column. These organisms include benthic deposit feeders, such as many polychaete worms and bivalves, and the sea cucumber *Parastichopus*; benthic particle/suspension feeders, such as many other polychaetes, crustaceans, bivalves, cnidarians, and other groups; zooplankton and phytoplankton, primarily in the water column/pelagic zone; and algae, including giant kelp (*Macrocystis*) and other brown, red, and green algae that may be exposed to dissolved contaminants within nearshore hard-substrate and wastewater outfall habitats.

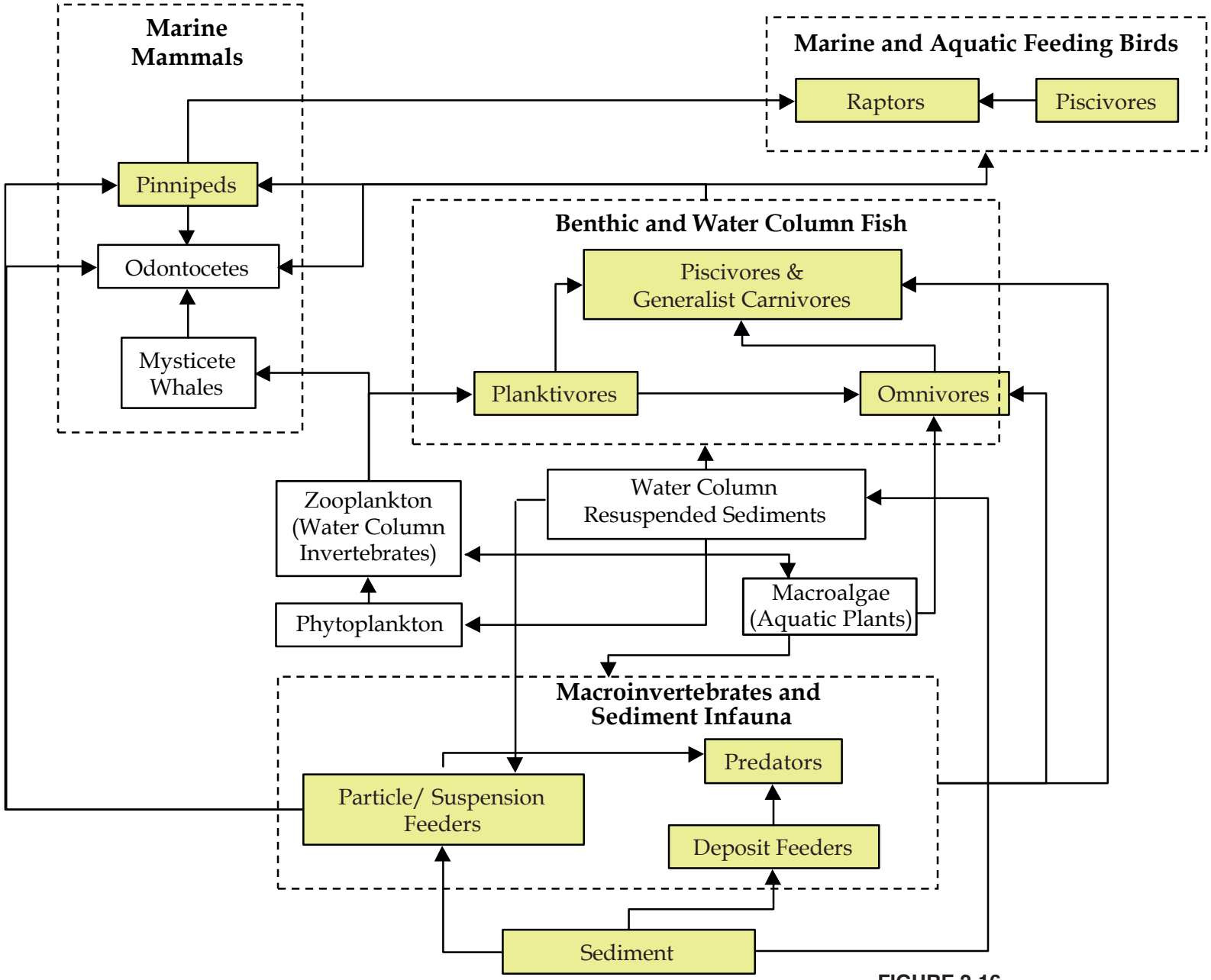


FIGURE 2-16
MARINE FOOD WEB
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

Note: Shaded components of the food web indicate those with sufficient data for evaluation in the risk assessment.

Secondary trophic-level organisms include several groups of animals, such as invertebrates, fish, and mammals (mysticete whales), that feed on the primary receptors including:

- Predatory invertebrates, such as some infaunal worms (e.g., *Nereis*), epifaunal organisms including most seastars and crabs, and pelagic species as exemplified by the market squid, *Loligo*, that feed in part on benthic invertebrates
- Pelagic invertebrates, such as mysid and euphausiid shrimp, that consume planktonic organisms
- Fish species (planktivores) that feed mainly on plankton in the water column
- Fish species (generalist carnivores), particularly bottom-feeding types such as flatfish, white croaker, and some rockfish that feed predominantly on soft-bottom invertebrates (i.e., primary receptors) and other fish, but also feed on species such as surfperch and rockfish
- Fish species (omnivores) such as kelp bass that frequent shallow (e.g., less than 20-m water depth) hard-substrate habitats having fish, invertebrate, and algae food sources
- Mysticete whales, including gray whales that feed by straining organisms from bottom sediments, and blue, fin, and sei whales that consume pelagic invertebrates and small fish that typically feed on planktonic organisms

Tertiary trophic-level organisms include some fish species, fish-eating birds, and toothed marine mammals (pinnipeds and odontocetes) that feed on the secondary-level animals. These animals include:

- Piscivorous fish, such as tuna, barracuda, and billfish, that are pelagic, very motile species
- Odontocetes (toothed whales and dolphins) that feed mostly on fish but, in the case of killer whales, also prey on pinnipeds and are pelagic and very motile
- Pinnipeds, such as sea lions and harbor seals, that prey predominantly on fish
- Piscivorous birds, such as brown pelicans and double-crested cormorants, that feed on fish and crustaceans

Quaternary trophic-level organisms include some avian species that feed on tertiary-level animals. Raptors (e.g., the bald eagle and the peregrine falcon) are considered quaternary-level animals.

2.5.3.2 Aquatic Exposure Pathways

At the PVS, the highest sediment-related exposures are likely to occur in the sediments in the area of the LACSD outfalls. Organisms associated with contaminated sediments (e.g., infaunal and epifaunal invertebrates) are directly exposed to contaminated sediments by multiple pathways. Benthic infaunal deposit feeders, such as many polychaete worms and bivalves, and the sea cucumber *Parastichopus*, are directly exposed via ingestion of and contact with contaminated sediments, exposure of contaminated porewater via gills and integument, and ingestion of contaminated prey items. Epibenthic invertebrates, either particle/suspension feeders or scavengers, such as many other polychaetes, crustaceans, bivalves, cnidarians, and other groups, live in close association with sediments and are

exposed through ingestion of contaminated resuspended particulates and respiration of water at the sediment-water interface. Consumption of these benthically coupled invertebrates by upper trophic-level animals forms the basis of most of the important exposure pathways.

Other primary trophic-level receptors, such as zooplankton, phytoplankton and algae, including the giant kelp (*Macrocystis*), are not associated with sediments, and their exposure pathway is via the water column and ingestion or absorption of contaminated particles suspended in the water column. Plankton are relatively transient and may be transported through the site by currents and tidal motions. Thus, their exposures are considered to be less than those of benthic and epibenthic organisms. The organisms that feed on them (e.g., pelagic invertebrates, such as, mysid and euphausiid shrimp, and planktivorous fish species that consume pelagic invertebrates) are not expected to bioaccumulate contaminants to the same degree as bottom-feeding fish. Water-column exposure pathways are of concern in that contaminants have been found to flux from the bottom sediments into the water column (Zeng et al. 1999), indicating there is a significant potential for contamination of pelagic food webs.

Exposure pathways for epibenthic fish species (e.g., demersal fish) include ingestion of contaminated prey, incidental ingestion of sediments, and respiration of water at the sediment-water interface. Some demersal fish bury into the sediment and are exposed via ingestion or contact with their gills to contaminants in porewater and resuspended particles. Pelagic organisms primarily accumulate pollutants via their gills or through consumption of contaminated prey items. Aquatic birds and marine mammals are tertiary receptors that are exposed predominantly via consumption of prey items. They can be exposed to contaminants at levels greater than those in the sediment as a result of biomagnification through the food web.

The feeding habits of selected bird species have been studied in conjunction with the NRDA. Bald eagles feed primarily on fish (e.g., California grunion [*Leuresthes tenuis*], kelp bass, surfperch) but also consume other birds (e.g., western gulls [*Larus occidentalis*], sooty shearwater [*Puffinus griseus*]) and mammal carrion (California sea lion, harbor seal) (Garcelon 1997; BioSystems 1994; Peakall 1994). The brown pelican feeds exclusively on fish, usually small schooling fish such as northern anchovy, Pacific sardine (*Sardinops sagax*), and chub (Pacific) mackerel (BioSystems 1994). Peregrine falcons in the Channel Islands feed primarily on seabirds, such as auklets and gulls, during the winter months when the female is forming eggs and there are few migrant prey birds (Hunt 1997). Auklets are fish-eating birds representative of a tertiary trophic level.

2.5.4 Identification of Representative Species

It is not feasible to evaluate every species that may be present and potentially exposed within the PVS study area and SCB. Consequently, receptors of high ecological or societal value or those believed to be representative of broader groups of organisms were selected for evaluation. Representative ecological receptors were selected based on current information on habitat presence, use, and distribution within the PVS study area and SCB. Each receptor was chosen to represent a trophic category and particular feeding behaviors that would represent different modes of exposure to COPECs. Thus, the species that were chosen for evaluation represent hundreds of similarly exposed species in the PVS/SCB region.

Representative receptors were selected using the following criteria:

- Special-status species (e.g., threatened or endangered)
- Small home range
- Representative of an ecological guild
- Susceptible to bioaccumulation or biomagnification of COPECs (e.g., higher trophic-level predators)
- Likely to be exposed to contaminants
- Habitat is available to support the selected receptor
- Known or suspected to be sensitive to contaminants
- Ecologically important

Representative species selected are described below and shown in Table 2-16. Representative species were selected from four trophic levels (primary, secondary, tertiary, and quaternary) and were assessed at one of three different levels: community, population, or individual. Species assessed at the individual level consist of special-status species of birds and mammals. Species assessed at the population level include fish for which protection of populations is important. Receptors assessed at the community level consist of sediment infauna and benthic invertebrates, which provide forage/prey for higher trophic levels or support habitat quality and require protection at a community level.

TABLE 2-16
Representative Species
Palos Verdes Shelf Ecological Risk Assessment

Trophic Level	Functional Group	Representative Species	Level of Assessment
Primary	sediment infauna	various invertebrates	community
	benthic macroinvertebrates	various invertebrates	community
Secondary	benthic fish	dover sole	population
	benthic fish	white croaker	population
	water column fish	kelp bass	population
Tertiary/Quaternary	piscivorous birds	double-crested cormorant[a]	individual
		brown pelican[a]	individual
	piscivorous mammals	California sea lion[a] (nursing pups and adult females)	individual
Quaternary	carnivorous birds	peregrine falcon[b]	individual
	piscivorous/carnivorous/scavenger	bald eagle[a]	individual

[a] Special-status species
[b] Although peregrine falcons were federally delisted in 1999, monitoring of the species is still required. Therefore, peregrine falcons are considered special-status for the purpose of this ERA.

Higher trophic levels generally are longer lived, and body burdens will remain in individuals throughout their lifetime. Species at lower trophic levels generally have higher rates of reproduction and shorter life spans, and are useful for monitoring improvement in benthic community structure and, in the case of fish species, the reduction of contaminants entering the food web. A brief profile for each selected receptor is presented below.

Note: Marine plants are primary receptors, but are not directly evaluated in this ERA. Marine plants (e.g., the giant kelp *Macrocystis pyrifera*, other attached algae, and phytoplankton) are exposed to water-column concentrations rather than sediments and, as such, can not be directly compared to sediment concentrations in the risk assessment as are benthic invertebrates. Water-column concentrations of the COPECs are generally unknown but are considerably reduced below sediment levels at PVS (Zeng et al. 1999). In addition, plants, in general, are not particularly susceptible to toxicity from these organochlorine compounds. Lowest chronic toxicity values for aquatic plants (0.3µg DDT/L, 0.144 µg PCB/L [Suter and Tsao 1996] are approximately 10-fold greater than COPECs in the PVS water column [Zeng et al. 1999]). Plankton (including algae, zooplankton, and larval fish) incur only transient exposure to PVS water-column contaminants. For the purposes of this ERA, the route of water-column exposure is tracked via the food web through exposure in the kelp bass and through pelagic fish and squid in the diet of higher-order consumers.

Sediment Infauna and Benthic Macroinvertebrates

Invertebrates that live in close association with the sediments are primary receptors and are assessed at the community level. Protection of sediment infauna and benthic macroinvertebrates that serve as an important food base to many higher trophic-level organisms is an assessment endpoint. These organisms are in direct contact with contaminants in the sediments and porewater. As adults, they are resident in the area and have small home ranges. Generally, most infaunal organisms occur within the surface 10- to 30-cm layer, and are not directly exposed to the highest concentrations of the buried effluent-affected sediments, although deeper contamination will contribute, via remobilization, to surface concentrations. Contaminants in the sediments may be affecting infaunal species abundance at the site. While maintenance of a healthy, sustained benthic community is an important endpoint, concerns for protecting the existing benthic community at the site may be outweighed by its role as a link in the exposure pathways of upper trophic-level organisms.

Benthic Fish

Benthic (or bottom-feeding) fish are secondary receptors and are assessed at the population level. Bottom-feeding fish reside in the area and feed on sediment-dwelling invertebrates. Substantial site-specific data on species abundance and tissue contaminant concentrations exist for the Dover sole and white croaker. Bottom-feeding fish from the PVS have some of the highest tissue levels of these COPECs within the SCB. These species are generally more contaminated than pelagic-zone fish species because of their feeding habits, residency in the area, and, in the case of white croaker, high lipid content.

Dover sole is a demersal species found from the Bering Sea to Baja California, Mexico (Eschmeyer et al. 1983; Love 1991). Spawning occurs annually, at depth, between September and April. Young of the year are pelagic until they settle out of the plankton at 2-3 inches (Love 1991). Fish that settle from the plankton inhabit the mud bottom (Eschmeyer et al. 1983), and most live at depth between 150—1,100 feet (Love 1991). Adults may be found in deeper

waters, up to 3,300 feet deep (Love 1991). Thus, it is likely that Dover sole inhabiting the PVS are present on the shelf for a small portion of their life, and move off the shelf into deeper waters as they mature. Dover soles ingest annelids, benthic crustaceans, echinoderms, and mollusks (Pearcy and Hancock 1978; Allen 2002).

White croakers are present along the west coast of North America from Baja California to Vancouver Island, British Columbia (Skogsberg 1939; Miller and Lea 1972; Hart 1973). They spawn in shallow and open waters of bays. Newly hatched larvae are pelagic, and descend to sand and gravel bottom substrates as they develop. Developing larvae may move into bays and estuaries until they become juveniles, at which point they move into the ocean (Smith 1971; Thomas 1971). However, it appears that white croakers associated with the PVS likely stay on the shelf, given the tissue residue concentrations found in white croaker in the SCB (Love 2002). White croakers can be found in coastal waters from the surf zone to as deep as 600 ft (Love 1991), and ingest polychaetes, small shrimps, crabs, and mollusks (Chao 1995). They are predominately found in shallow water (most are found in water of 0-20 m depth) (Allen 2002).

Water Column Fish

Water column fish (e.g., rockfish, surfperch, and kelp bass) that tend to feed on invertebrates and fishes are also important secondary or tertiary receptors. Many of these water column fish species are, in turn, important prey items for piscivorous fish, predatory mammals, and fish-eating birds (e.g., bald eagles). Like Dover sole and white croaker, substantial site-specific data are available for kelp bass. Kelp bass are present along the west coast of North America from Baja California to the Columbia River, Washington, in water at least 200 feet deep (Love 1991). Newly hatched larvae are pelagic and drift in the plankton (Fitch 1974; Love 1991). They are thought to settle out in kelp beds and other areas where structures exist (Love 1991). Juvenile and adult kelp bass associate with structures such as kelp, rocks, sewer pipes, and oil platforms, and tend to aggregate in large groups during the annual breeding season (Love 1991). Juvenile kelp bass are known to ingest benthic invertebrates (crustaceans), whereas adults ingest fish, plants, jellyfish, sponges, tunicates, polychaetes, cephalopods, shrimp, and zooplankton (Heemstra 1995; Love and Ebeling 1978; Allen 2002). Kelp bass are known to move; however, the extent of their movement is not well documented. The largest movement recorded for a tagged kelp bass is 282 miles (Love 1991). Their movement depends on food abundance and quality (Love 2002).

Marine and Aquatic-Feeding Birds and Marine Mammals

Tertiary and higher-level receptors are of greatest concern because of the biomagnification potential of the COPECs. These receptors include (1) seabirds and aquatic-oriented raptors that mainly feed on fish and/or other birds and (2) marine mammals (pinnipeds and odontocetes) that mostly feed on fish. Bald eagles and peregrine falcons are quaternary receptors, feeding on other birds and, in the case of the eagles, sea lion carcasses. Some marine mammals (sea lions) and aquatic-feeding birds (bald eagles) fulfill residency and complete exposure pathway requirements for representative tertiary receptors. Higher-order consumers with adequate foraging range and diet information or that have been identified as species of concern for organochlorine exposure in the SCB include brown pelicans, double-crested cormorants, bald eagles, peregrine falcons, and California sea lions (adult females and nursing pups). California least terns (*Sterna antillarum browni*) were federally listed as endangered in 1970. Although contaminants are thought to be among several factors contributing to population declines in the SCB, habitat loss and human-related disturbance and predation are thought to be the primary contributors (Hothem and Powell 2000). As a

piscivorous bird, the risk to least terns from exposure to tDDT and tPCB is represented in this ERA by brown pelicans and double-crested cormorants. Detailed summaries of life-history parameters for the selected receptors are included in Appendix B.

Brown Pelican—The brown pelican is a marine bird that can be either a tertiary or quaternary receptor. It is a special-status species, and is assessed at the individual level. They feed mostly on fish, but occasionally also on crustaceans, and their own young. Pelicans breed on the California Channel Islands and in Mexico from March to early August. After breeding, individuals leave colonies and disperse along the entire California coast (Zeiner 1990a).

Double-Crested Cormorant—The cormorant can be a tertiary or quaternary receptor. It is a special status-species, and is assessed at the individual level. Cormorants are year-long residents along the California coast, but are also found along the mountains and northeastern plateau in the summer. Cormorants feed primarily on fish, but also feed on crustaceans and amphibians (Zeiner 1990a).

Bald Eagle—Bald eagles are a protected quaternary receptor and are evaluated at the individual level. They feed on fish, but also eat various mammals, birds (especially waterfowl), and carrion. They sometimes pirate fish from ospreys. Bald eagles are found near seacoasts, rivers, lakes, and reservoirs. They typically use areas with a high amount of water-to-land edge where prey is concentrated or available. Bald eagles prefer areas with unimpeded views, such as margins of forest stands or trees projecting above the forest canopy (Palmer 1988). Within the SCB, bald eagles only nest on Santa Catalina Island, and, though juvenile eagles from these nests are known to disperse widely (Sharpe 2003), adult breeders are non-migratory (Garcelon et al. 1994b).

Peregrine Falcon—The peregrine falcon is a quaternary receptor and a special-status species. As such, it is assessed at the individual level. The peregrine falcon is formerly a federally listed endangered species that was delisted August 1999. Its current status is "Delisted Taxon, Recovered, Being Monitored First Five Years." Because this species is still being monitored, it is considered a special-status species in this assessment. Peregrine falcons feed on a variety of birds, mammals, insects, and fish; however, those inhabiting the Channel Islands feed only on birds (Kiff 1980) and have diets consisting of seabirds, waterfowl, shorebirds, and terrestrial birds (Hunt 1994). They migrate along the coast and in the western Sierra Nevada in spring and fall, and are occasionally found in the California Channel Islands during the winter (Zeiner 1990a). True migration, however, does not occur in the resident peregrine population in the Channel Islands (PCAPFRT 1982).

California Sea Lion—The California sea lion is a marine mammal that can be a tertiary or quaternary receptor. It is a special-status species and is assessed at the individual level. Sea lions feed on fish and cephalopods (squid and octopus), but diets shift to species that are locally and seasonally abundant (NOAA 2001; Zeiner 1990b). Foraging ranges for sea lions can be up to 54 km (Antonelis et al. 1984). Sea lions breed in the California Channel Islands and Mexico, rarely giving birth north of San Miguel Island. Males migrate southward for breeding season and north after breeding, while females and pups remain near breeding rookeries. Resident populations in southern California consist of females, pups, and juveniles (NOAA 2001; Zeiner 1990b). Therefore, adult female sea lions were evaluated in this ERA.

Contaminants in the female are transferred directly to the fetus and then via milk to the nursing pup postpartum. Because of the high lipid content of California sea lion milk (32 to

44 percent, Oftedal et al. 1987) relative to the average female lipid content (30 percent, Connolly and Glaser 2002), lipophilic compounds such as tDDT and tPCB are preferentially transferred to the milk. This results in exposures to nursing pups that are greater than those experienced by the adult female. Although pups may be nursed for up to a year, pups rely solely on the mother's milk for approximately the first seven months of their life and generally begin eating solid food after the first 200 days (Boness et al. 1991). Because this is a time of high exposure during a sensitive life-stage (rapid growth and development), pups up to age 7 months were evaluated in the current ERA.

SECTION 3

Analysis

3.1 Exposure Characterization

Exposure characterization is used to evaluate the relationship between receptors at the site and potential stressors (i.e., COPECs). Exposure is defined as the co-occurrence of a receptor and a stressor in both space and time. For risk to be present, there must be exposure. Exposure for ecological receptors (other than plants) may be characterized as either "external" to the animal (e.g., ingestion, inhalation, or dermal contact with contaminated media) whereby adverse effects are the result of intake of contaminated media or food; or they may be characterized as "internal" to the animal (e.g., COPEC concentrations within the receptor animal's tissues) whereby adverse effects may occur through various mechanisms including:

- Toxicity to a target organ (e.g., liver or kidneys where metabolism and excretion occur)
- Release from tissue reserves during episodic events (e.g., mobilization of contaminants from fat during migration or hibernation) to other tissues (e.g., the brain) where effects may occur

The information necessary to estimate exposure is described in this section, which includes an overview of the various sources, the spatial and temporal distribution of COPECs, and the models used to estimate exposure (external and/or internal, as appropriate). Both external and internal exposures are evaluated and quantified where data are available. Exposures are summarized as exposure-point concentrations (EPCs), either the concentration of the COPEC in a particular medium at the point of exposure to the ecological receptor (e.g. water, sediment, tissue) or the dosage experienced by ingestion. EPCs represent the concentrations of concern in determining ecological risk. The 95 percent UCLs were used as EPC values for individual data sets as summarized by medium and location.

3.1.1 Source Evaluation for Chemical Stressors

This subsection provides an overview of the sources of stressors (COPECs) in the PVS region, their spatial and temporal distribution, and models used to estimate porewater concentrations.

3.1.1.1 Sources

Sources of chemical stressors (tDDT and tPCB) were described in Section 2. The primary historical source of these COPECs to the PVS is effluent discharged through the White Point outfalls. EA sediments in the area of the outfalls and those transported away from White Point by ocean currents and deposited on the ocean floor serve as the main repository for contaminants. Virtually all of the effluent-affected area is contaminated with tDDT and tPCB. In addition, contaminant flux from the sediment surface to the water column occurs, increasing contaminant concentrations in the water column overlying sediments. Other

potential sources of contaminants (i.e., atmospheric deposition, stormwater runoff, and ocean currents) are considered minor in comparison to the existing reservoir of deposited contaminants, and those sources would provide diffuse contributions yielding ambient levels of contamination unrelated to the White Point discharge.

3.1.1.2 Spatial and Temporal Distribution

As discussed in Sections 1 and 2, the summary of exposure for the PVS and SCB study areas is based on a characterization of recent water, sediment, and fish tissue contamination from samples collected during the period 1990 through 2003. The spatial distribution varies by receptor group, with benthic and infaunal invertebrates and fish evaluated for the PVS study area and higher-order consumers (marine birds and mammals) evaluated for exposure over the entire SCB. PVS exposure is assessed in relation to water, sediment, and sediment porewater concentrations of COPECs. SCB-wide exposure is assessed in relation to estimated dietary concentrations of COPECs.

3.1.1.3 Estimation of Porewater Concentrations

Direct measurements of porewater tDDT and tPCB concentrations were not available. Therefore, porewater concentrations were calculated from the contoured surface-sediment layer (0 to 15 cm) where sediment tDDT and tPCB concentrations and organic carbon data were available. The porewater concentrations were calculated by the equilibrium partitioning (EqP) model (DiToro et al. 1991):

$$C_P = \frac{C_S}{(f_{OC} K_{OC})} \qquad \text{[Equation 3.1]}$$

Where: C_p = the calculated porewater concentration (µg/L)

C_s = the measured sediment concentration (µg/kg)

f_{oc} = the measured fraction of total organic carbon (TOC) in the site sediment (percent TOC/100)

K_{oc} = a calculated organic carbon-water partition coefficient for tDDT or tPCB.

Values for K_{oc} were determined from the relationship developed by USEPA (Karickhoff et al. 1995):

$$\log_{10} K_{oc} = 0.00028 + 0.9831 * \log_{10} K_{ow} \qquad \text{[Equation 3.2]}$$

Where: K_{ow} = the octanol/water partition coefficient for the chemical. The $\log_{10} K_{ow}$ values were obtained from the USEPA AQUIRE database, and were used to derive the values for K_{oc}:

Compound	$\text{Log}_{10} K_{ow}$	$\text{Log}_{10} K_{oc}$	K_{oc}
tDDT	6.530	6.42	2,625,850
tPCB	6.541	6.43	2,691,534

3.1.2 Exposure Estimation for Benthic Invertebrates

Exposure of benthic invertebrates to tDDT and tPCB occurs primarily through porewater and bulk sediment. Porewater concentrations of tDDT and tPCB are presented in subsection 3.1.1.3 and Figure 3-1. Sediment exposure is based on exposure to tDDT and tPCB in the surface layer (0-15 cm average concentration) and deep layer (15-30 cm average concentration) (Figure 2-3). Summary statistics for these data were presented in Table 2-7.

The risks to benthic invertebrates from sediment and porewater were estimated based on the chemistry of surface sediment deposits (0–15 cm) rather than the more limited data available from deeper cores. Figures 2-2 to 2-5 show the spatial distribution of tDDT and tPCB in surface and deep-layer sediments at the PVS. As with earlier reports (Lee 1994; USEPA 2000), the deeper layer of sediment (15–30 cm) has higher tDDT concentrations than the surface layer. This has been attributed to the burial of historically higher concentrations of tDDT from the sewer outfall. However, the ERA focuses on exposures from the shallow layer as this almost completely mimics the deeper layer in spatial pattern and is part of a larger sediment quality characterization that includes the entire SCB (in contrast to deeper layer characterizations). EPCs for benthic invertebrates are represented by the complete distribution of shallow sediment tDDT and tPCB within the PVS study area to the 200 m depth contour (Figures 2-3 and 2-5 respectively), as presented in Table 2-7.

3.1.3 Exposure Estimation for Benthic and Water Column Fish (Secondary Receptors)

3.1.3.1 External Exposure

External exposure to benthic and water column fish was estimated from measured tDDT and tPCB concentrations in the water column. Data from Zeng et al. (1999) were used as EPCs, and are summarized in Tables 2-8 (tDDT) and 2-9 (tPCB).

3.1.3.2 Internal Exposure

Internal exposures consist of measured concentrations of COPECs in tissues (e.g., whole body, fillet) of receptor fish in the PVS Study Area. Fillet data were available for white croaker, Dover sole, and kelp bass in and around the PVS, whereas whole-body data were available for sanddabs caught throughout the SCB. Whole-body data were calculated from fillet data for white croaker, Dover sole, and kelp bass, using simple regressions, as shown in Figure C-3 (Appendix C). These regression equations are presented in Table 3-1. To determine exposure-point concentrations for each analyte and species, distributions of measured and calculated whole-body concentrations were tested for normality. Summary statistics were calculated from the ln-transformed data, and the 95 percent UCL or maximum value (i.e., where the 95 percent UCL was greater than the maximum) (Table 2-10) was used to estimate internal exposure. The distributions of contaminant concentrations in fish at the PVS are shown in Figures 3-2 and 3-3.

TABLE 3-1
Equations Used to Convert Fillet Residues to Whole-Body Residues
Palos Verdes Shelf Ecological Risk Assessment

Species	tPCB	tDDT
Sanddabs	Concentration is already as whole body	Concentration is already as whole body
Dover sole	log(WB) = 0.155 + 0.775*log(F)	log(WB) = 0.138 + 0.845*log(F)
White croaker	log(WB) = 0.155 + 0.775*log(F)	log(WB) = 0.138 + 0.845*log(F)
Kelp bass	log(WB) = 0.155 + 0.775*log(F)	log(WB) = 0.138 + 0.845*log(F)

Notes:
Log_{10} used for transformations
F fillet concentration (mg/kg WW)
WB whole-body concentration (mg/kg WW)

3.1.4 Exposure Estimation for Birds and Marine Mammals

Birds and mammals experience both external and internal exposures. External exposure occurs through multiple pathways including ingestion of abiotic (e.g., sediment and surface water) and biotic (e.g., food) media as well as inhalation and dermal contact of abiotic media. Internal exposures consist of concentrations of COPECs in tissues of receptor species. Exposure and bioaccumulation models, associated assumptions, and exposure factors for external and internal exposures are presented below.

3.1.4.1 External Exposures

Modeling was used to address the multiple pathways of external exposure for birds and mammals. Distributions of exposure estimates for each representative species were generated based on model assumptions, modeled distributions of exposure concentrations in abiotic media and food sources, life-history parameters, and area use distributions described below.

The end product of the exposure estimate is a dosage (amount of chemical per kilogram receptor body weight per day [mg tDDT or tPCB/kg BW/d]) rather than a media concentration as was the case for benthic macroinvertebrates and fish. This is a function of both the multiple-pathway approach as well as the typical methods used in toxicity testing for birds and mammals (as described in Section 3.2). Summaries of pathway-specific and total (i.e., sum for all pathways) exposure estimates are presented and compared to toxicity values in Section 4.1.

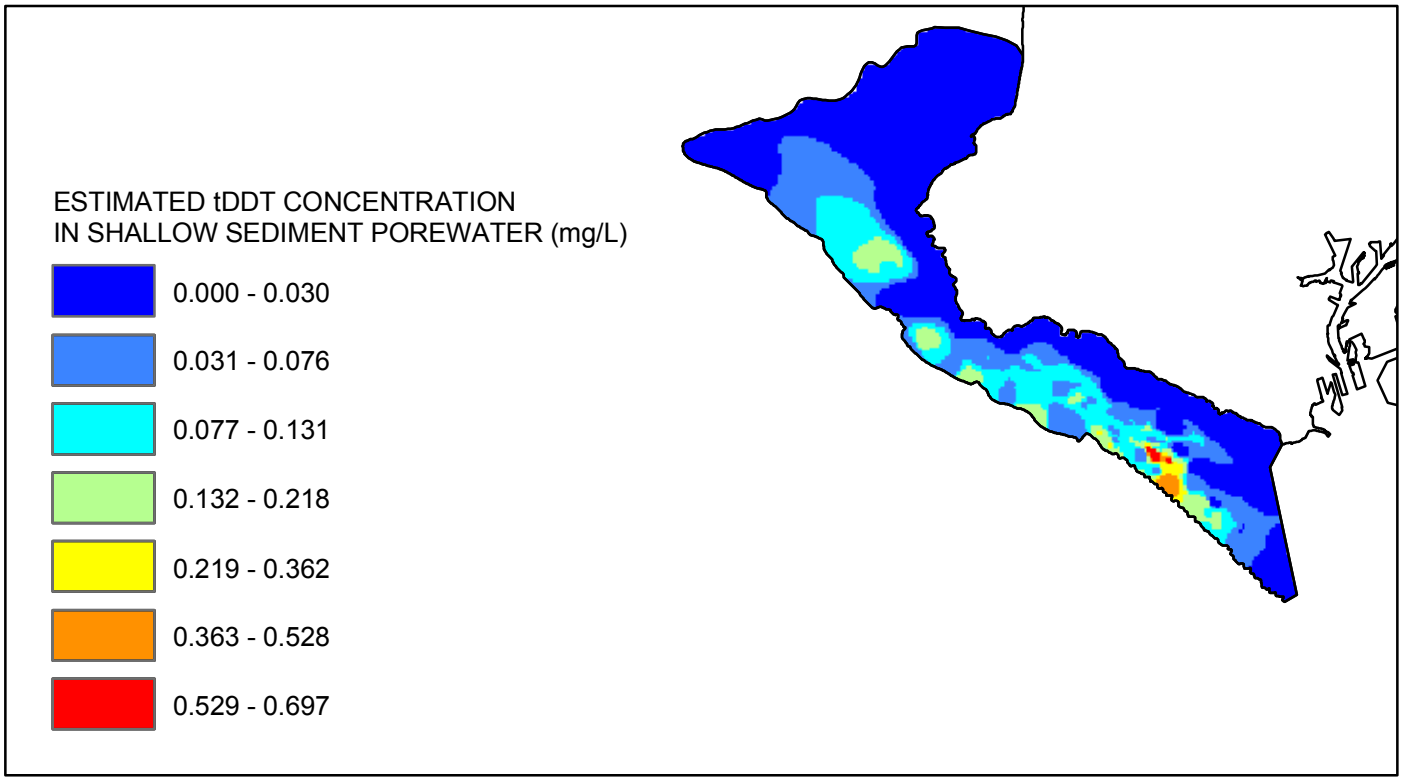
Exposure Model

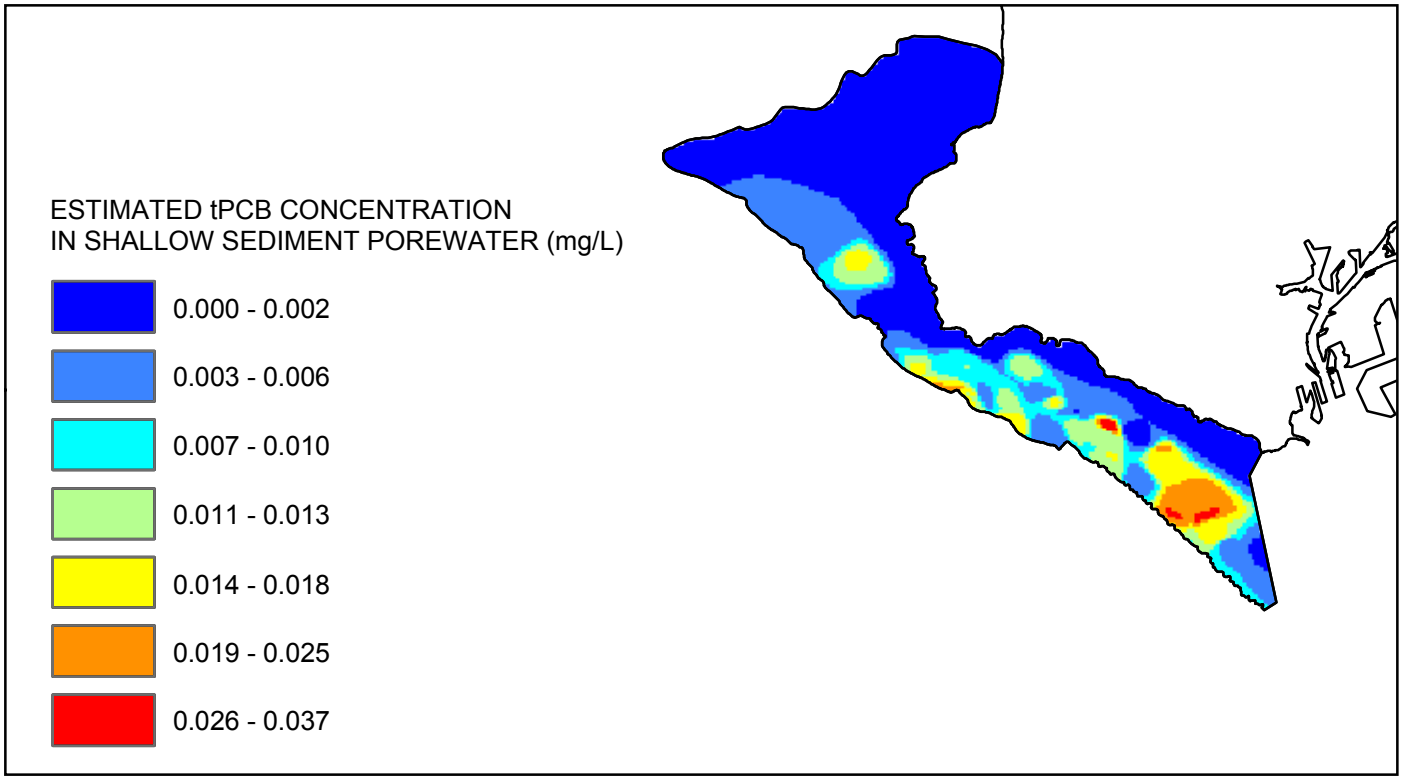
The general form of the model (Suter et al. 2000) used to estimate exposure of birds and mammals to COPECs in sediment, surface water, and food items is as follows:

$$E_t = E_o + E_d + E_I \qquad \text{[Equation 3.3]}$$

Where: E_t = the total chemical exposure experienced by wildlife

E_o, E_d, *and* E_i = oral, dermal, and inhalation exposure, respectively





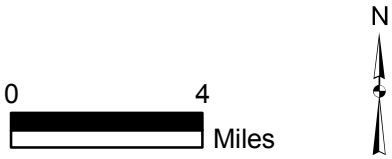


FIGURE 3-1
ESTIMATED POREWATER CONCENTRATION IN SHALLOW (0-15 CM) SEDIMENTS TO 200 METERS
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

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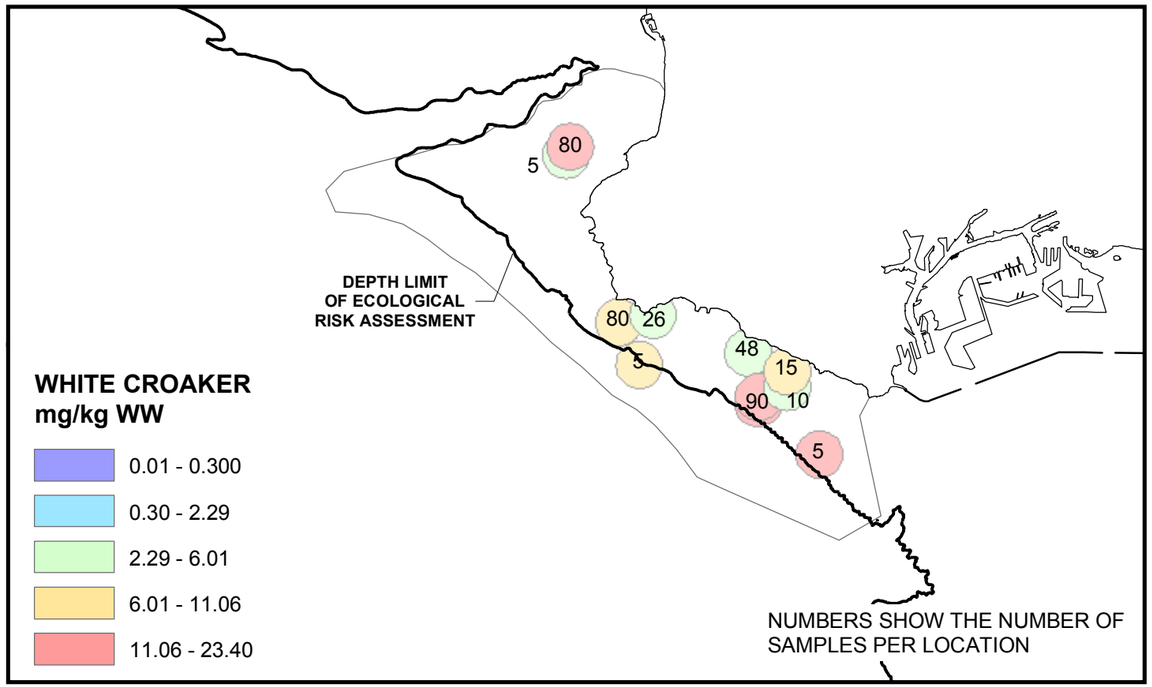

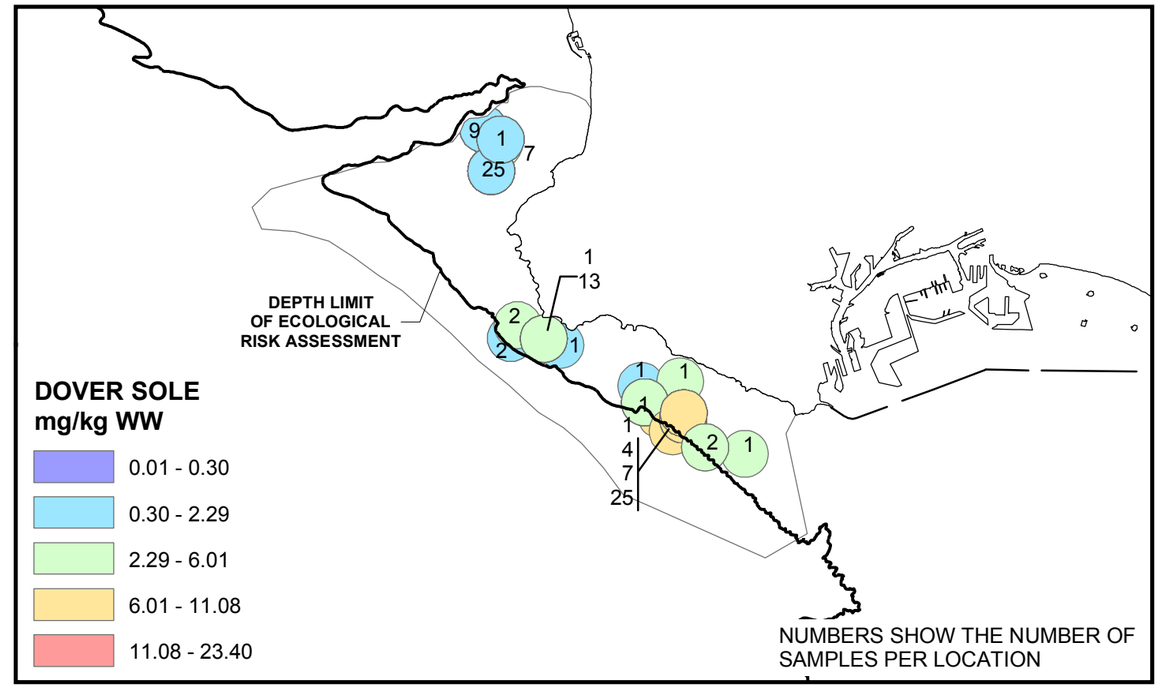

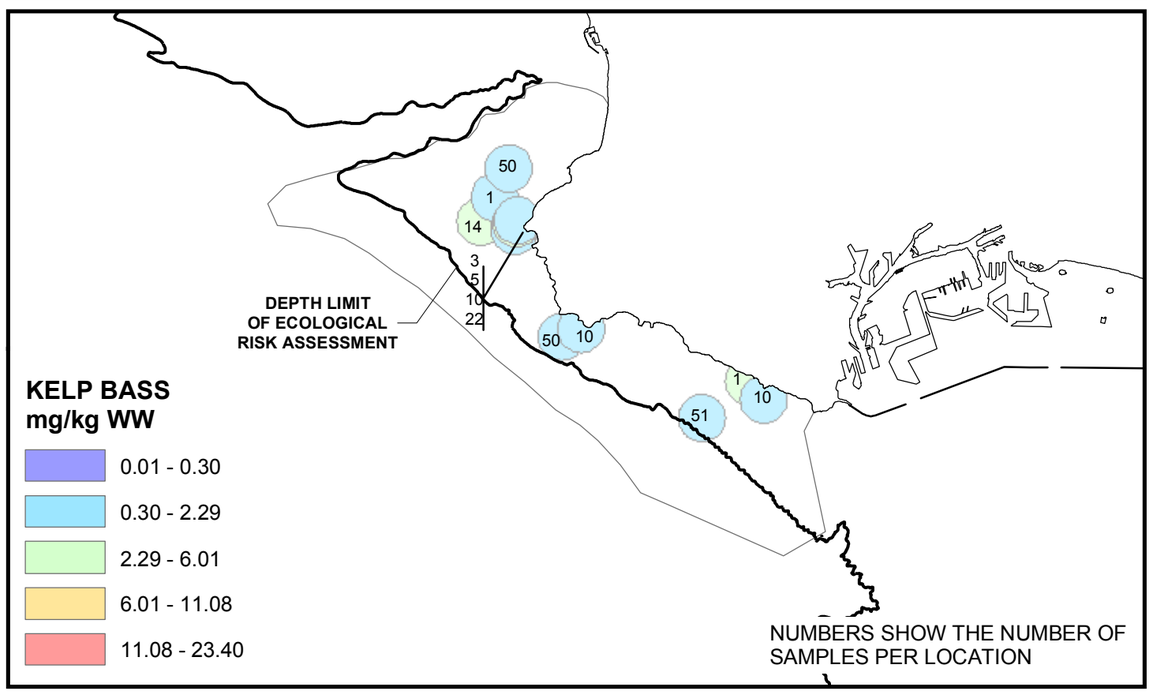

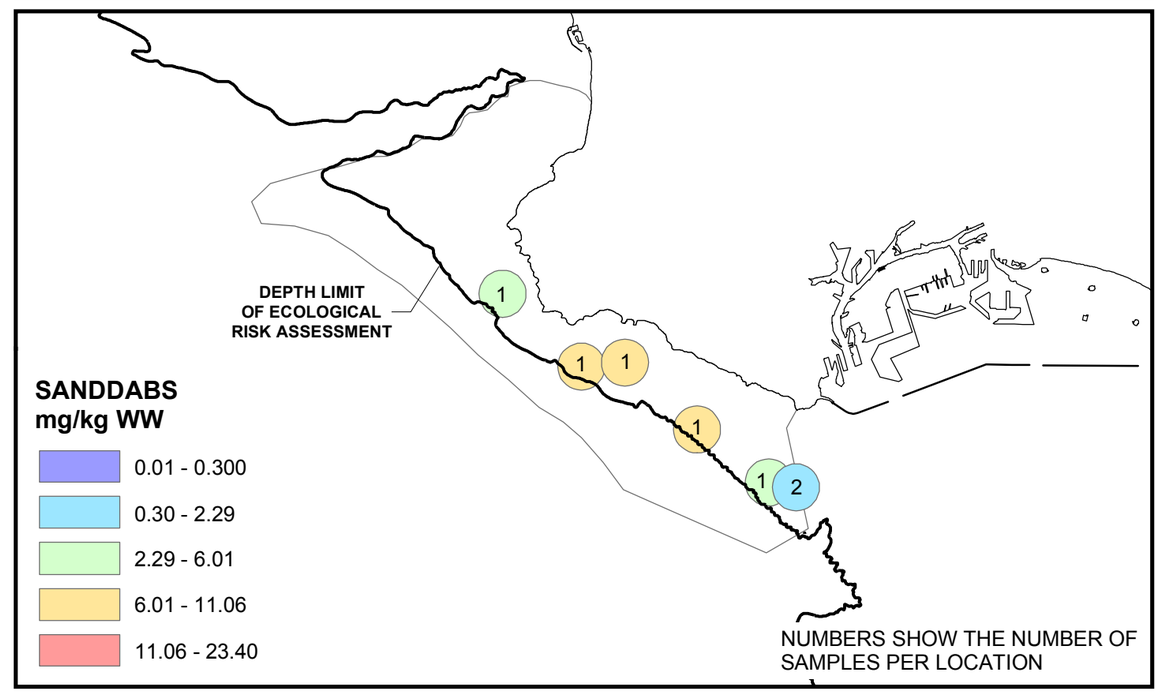

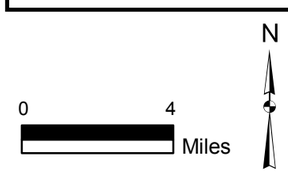

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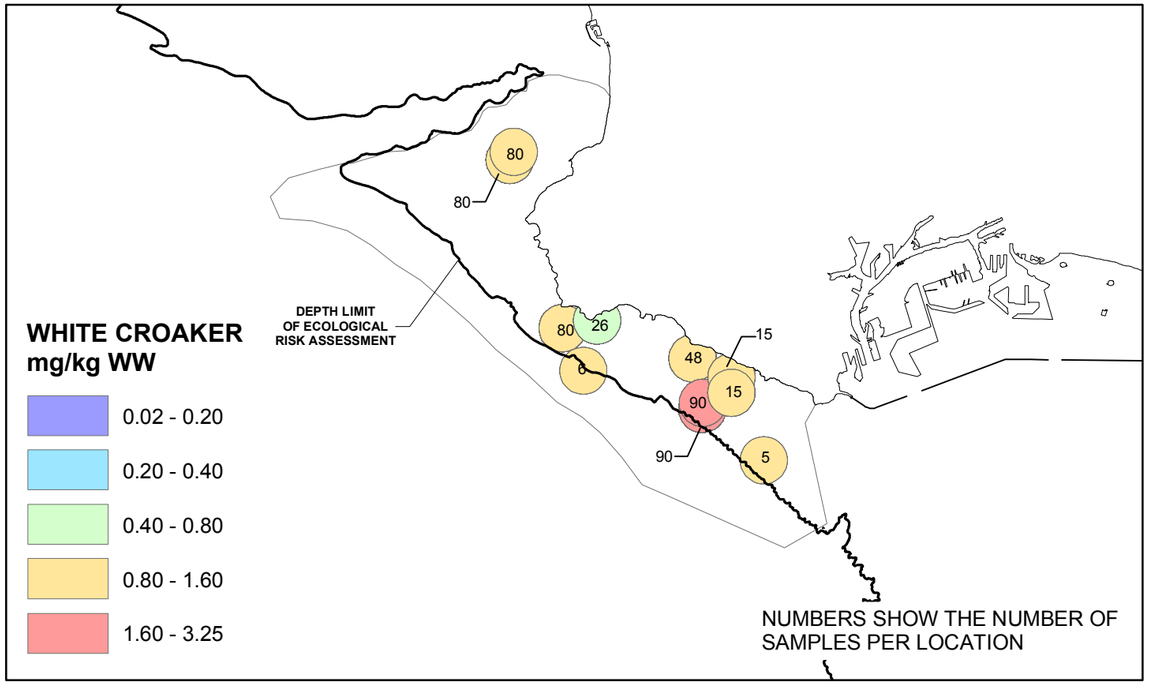
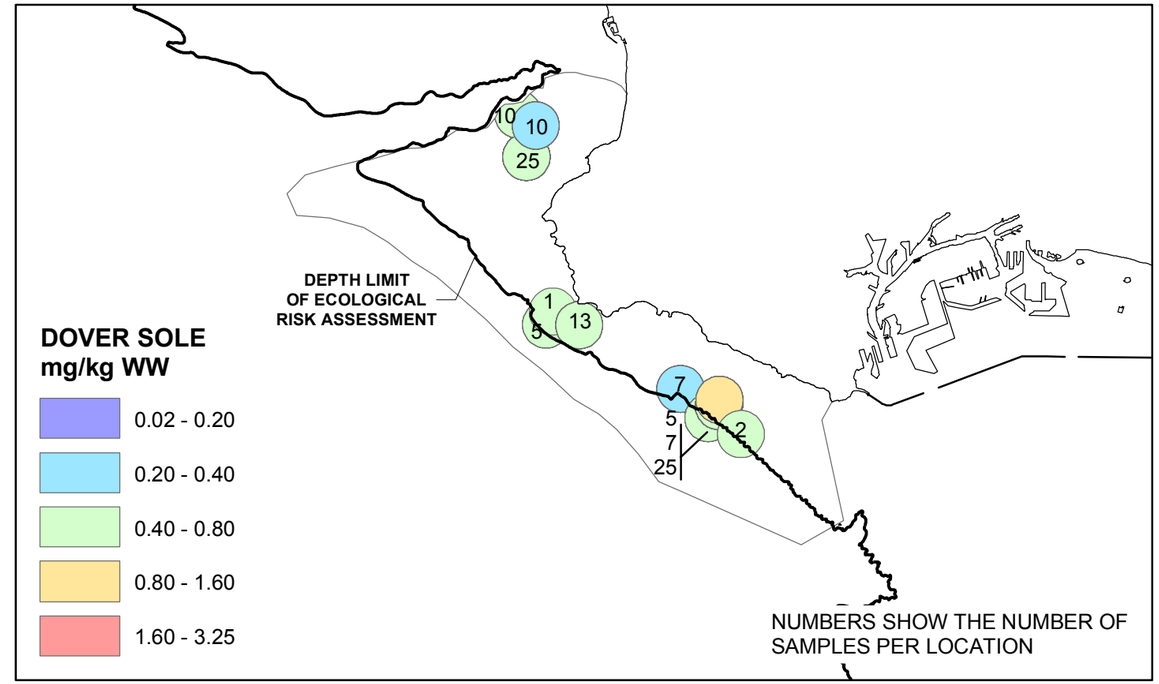
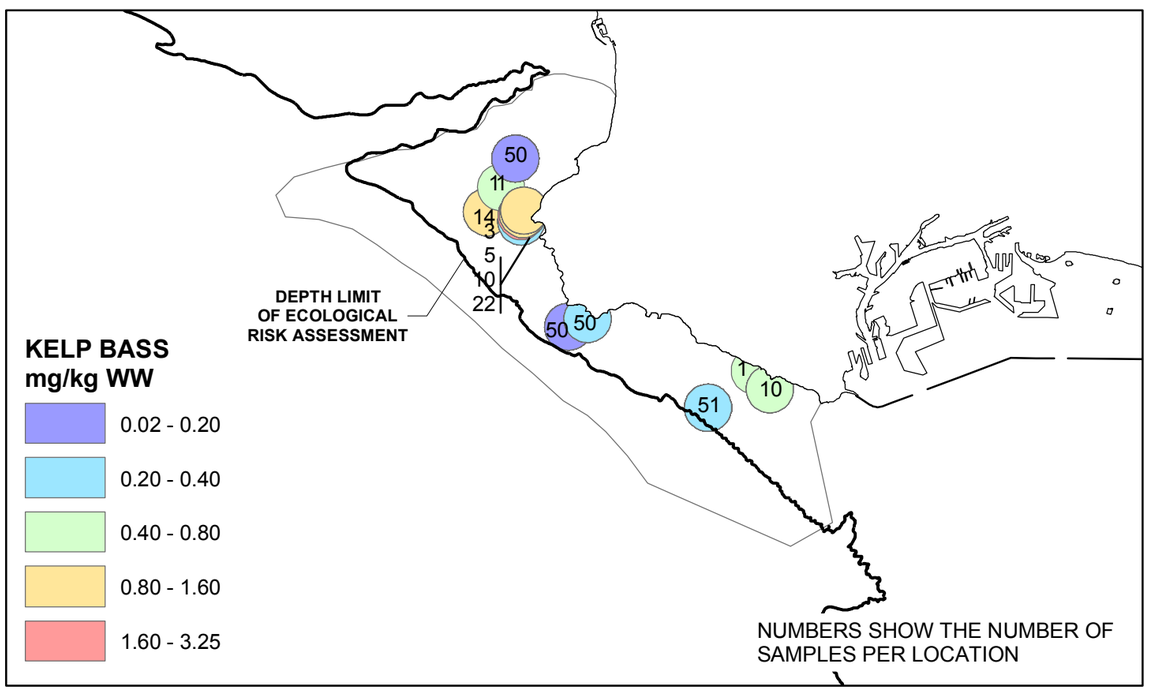
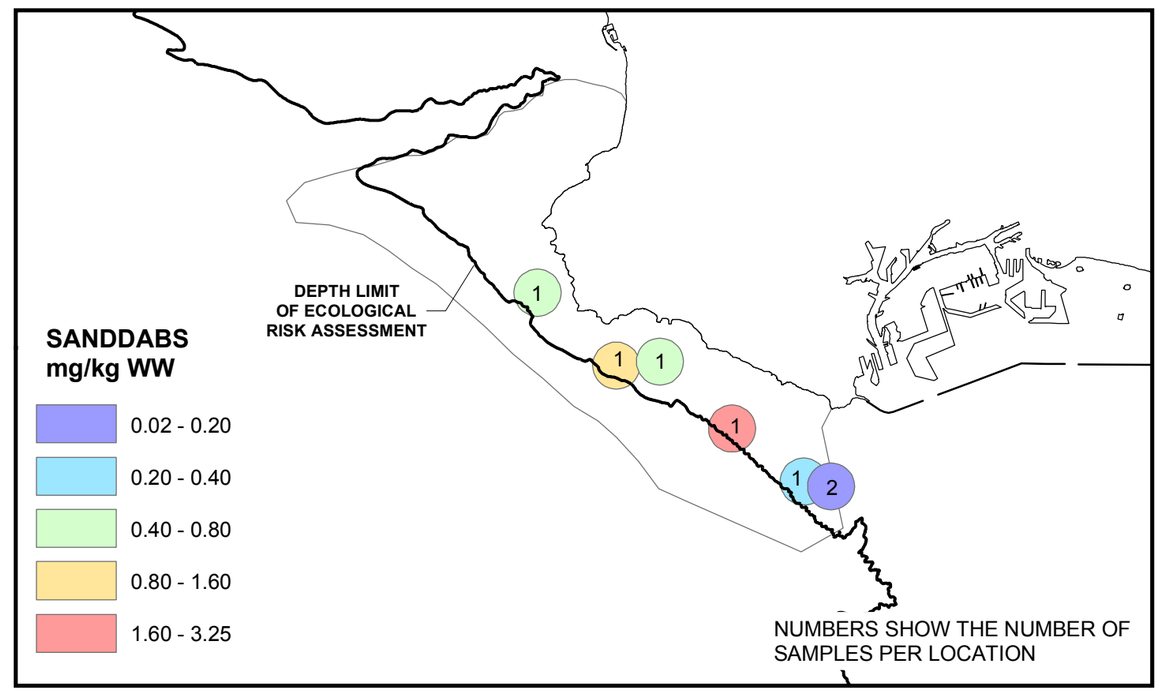
FIGURE 3-2
AVERAGE RESULTS OF DDT DETECTED IN INDIVIDUAL FISH FILLETS WITHIN THE PVS
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT









E072002006SAC Palos Verdes Figure_03_03.pdf 11-4-03 esm

FIGURE 3-3
AVERAGE RESULTS OF THE PCB DETECTED IN INDIVIDUAL FISH FILLETS WITHIN THE PVS
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

Oral exposure occurs through the consumption of contaminated food, water, or sediment. Dermal exposure occurs when contaminants are absorbed directly through the skin. Inhalation exposure occurs when volatile compounds or fine particulates are inhaled into the lungs.

Although methods are available for assessing dermal exposure to humans (USEPA 1992e), data necessary to estimate dermal exposure are generally not available for wildlife (USEPA 1993b). Similarly, methods and data necessary to estimate wildlife inhalation exposure are poorly developed or generally not available. Therefore, for the purposes of this assessment, both dermal and inhalation exposures are assumed to be negligible. As a consequence, most exposure must be attributed to the oral exposure pathway. Birds and mammals are exposed to contaminants through multiple oral pathways including sediments, food items, and surface water. The following generalized exposure model, modified from Suter et al. (2000), was used:

$$E_j = \{[Sed_j * P_s * FIR] + [\sum_{i=1}^{N} B_{ij} * P_i * FIR] + [Water_j * WIR]\} * AUF \qquad \text{[Equation 3.4]}$$

Where:

E_j	=	total exposure (mg/kg/day)
Sed_j	=	concentration of chemical in sediment (mg/kg DW)
$Water_j$	=	concentration of chemical in water (mg/L)
P_s	=	sediment ingestion rate as a proportion of diet
FIR	=	total food ingestion rate for the representative species (kg food/kg BW/day)
WIR	=	total water ingestion rate for the representative species (L/kg BW/day)
B_{ij}	=	concentration of chemical (j) in biota type (i) (mg/kg WW)
P_i	=	proportion of biota type (i) in diet
AUF	=	Area use factor (in this assessment, area use is defined as a distribution)

To establish parameters for the exposure model, various assumptions were necessary. These assumptions are outlined below.

Exposure Concentration Distributions and Bioaccumulation Models

A critical component of the estimation of exposure of birds and mammals to COPECs is the measurement of concentrations of COPECs in wildlife food and site media (e.g., sediment and water). Nursing California sea lion pups (up to seven months of age) rely solely on their mother's milk as a food source. Juvenile and adult California sea lions, brown pelicans, double-crested cormorants, and bald eagles are primarily piscivorous, whereas the peregrine falcon primarily forages on birds. Within the SCB, piscivorous seabirds (e.g., gulls and auklets) are the major food source for peregrines. Bald eagles also forage on seabirds in the SCB and occasionally consume sea lion carcasses that wash up on the beaches near their nesting areas. The foraging habits of these receptors (primarily taking prey in the water column or eating seabirds) indicate that sediment ingestion is unlikely or negligible.

Although sea lions and cormorants feed on benthic fish, these make up a relatively small portion of the diet and sediment ingestion is likely low for these receptors. Additionally, water is either obtained through the diet or from freshwater sources not associated with the site contamination.

These bird and mammal receptors are mobile, and they may experience exposure to contaminants in multiple locations where they range. Although the most highly contaminated sediment is largely restricted to the PVS study area (subsection 2.1.1 and Figure 1-1 outline the extent of this area), low levels of the COPECs can be found throughout the SCB. Through annual sampling programs, extensive sediment and fish data have been collected across the PVS and more limited data are available from areas throughout the SCB (mostly near the coast). Because the bird and mammal receptors forage throughout the SCB, measured sediment data were used to develop a GIS-based map depicting the distribution of COPEC concentrations in sediment throughout the SCB. Additionally, measured concentrations of COPECs in sediment and fish (white croaker, sanddabs, and kelp bass) were used to develop sediment-fish relationships resulting in distributions of COPECs in whole-body fish throughout the SCB (restricted to the 200-m depth contour across the SCB). Development of sediment and fish GIS-based models, including associated assumptions, is detailed in Appendix C.

Surrogate diets using white croaker, sanddabs, and kelp bass were developed to represent the fish component of each receptor's diet (see details below). Many wildlife species are highly mobile, covering large areas in search of food, water, and shelter. The exposure that individuals experience depends on the amount of time they spend at a contaminated site. Site use depends on the size of the site relative to an animal's foraging range. Exposure distributions were calculated for each receptor by overlaying foraging ranges of the receptors onto fish concentration distributions. Sediment and water were not included in the exposure calculation because ingestion of these media by receptors was expected to be negligible.

Linear regression relationships between fish tissue and sediment concentrations are given in Table C-6. The equations are of the form $y=mx + b$, where m = slope. The slopes are equivalent to biota-sediment accumulation factors (BSAFs), showing the relationships between log fish tissue and sediment concentrations. In addition, the regression technique for describing BSAFs provides an estimated strength of the relationship (e.g. r^2 terms in Table C-6).

Life-History Parameters

Species-specific life history factors are needed to estimate exposure to COPECs for each representative species. These include body weight; food, water, and media ingestion rates; and diet composition and respective proportion of each diet component. Information on site use is also necessary to the exposure estimation. These parameters were identified through an open literature search and review of expert reports compiled for the Southern California Bight Natural Resources Damage Assessment and are summarized in Tables 3-2 and 3-3. Detailed species accounts for each receptor are presented in Appendix B.

As described above, surrogate diets as well as foraging ranges are necessary to the development of the exposure distributions. The assumptions used to create surrogate diets and develop foraging ranges are outlined below by receptor.

TABLE 3-2
Distribution and Foraging Information for Avian and Mammalian Receptors
Palos Verdes Shelf Ecological Risk Assessment

Parameter	California Brown Pelican	Double-Crested Cormorant	Peregrine Falcon	Bald Eagle	California Sea Lion
Breeding Islands in SCB (Months)	Anacapa, Santa Barbara (March to August)	Anacapa, Santa Barbara, Islas Los Coronados, Mexico[a] (April to August)	San Miguel, Santa Rosa, Santa Cruz, Anacapa, and Santa Barbara Islands (March to August); on the mainland, found in Ports of Los Angles, Long Beach, San Diego Bay, and Point Loma	Santa Catalina (February to July)	San Miguel, San Nicolas Island (May to August)
Roosting/Loafing Islands in SCB (Months)	Santa Barbara Island (greatest number of roosting birds found), all Channel Islands with high numbers on Anacapa and Santa Cruz, and Southern California coast	Probably same as breeding islands	Probably same as breeding islands	Santa Catalina (Note: juveniles recently released on Santa Cruz and spotted near Anacapa and Santa Rosa)	Santa Catalina, San Clemente (September to April)
Other Locations (Months)	Southern California coast (June to October), Año Nuevo Island, Southeast Farallon Island, and Isla Coronado Norte (March to August)	Along the Pacific coast	Breed along the coast north of Santa Barbara Island	Along the Pacific coast	Males migrate north along Pacific coast (September to April)
Seasonal Migration	After breeding as early as mid-May, individuals leave colonies in the Channel Islands and Mexico, and disperse along the entire California coast. Most return to breeding colonies by March and April.	Some double-crested cormorants are year-long residents of the California coast. Summer residents of mountains and northeastern plateau are absent from about November to March; they presumably migrate west and south to lowlands, especially along the coast, where the population increases in winter. Double-crested cormorants that inhabit the Channel Islands are known as mostly sedentary with populations moving from offshore islands to inshore channels in the non-breeding season (September to March).	Occasionally found in Channel Islands in winter. Migrants occur along the coast and in the western Sierra Nevada in spring and fall. Resident as a breeder; other individuals breeding farther north migrate into California for winter.	Most birds that breed in the Pacific region probably winter in the vicinity of their nests. Some move short distances to lower elevations or inland food sources. Juveniles disperse in random directions from the nest sites. Fairly common as a local winter migrant at a few favored inland waters in southern California.	Male migrates southward to Channel Islands and Mexico for breeding season and north after breeding, while females and pups remain near breeding rookeries (little is known about female migration). Año Nuevo Island and the Farallon Islands provide major haul-out grounds for males throughout the year.

TABLE 3-2
Distribution and Foraging Information for Avian and Mammalian Receptors
Palos Verdes Shelf Ecological Risk Assessment

Parameter	California Brown Pelican	Double-Crested Cormorant	Peregrine Falcon	Bald Eagle	California Sea Lion
Percent time present in SCB	100 percent for resident populations, 50 percent for migratory populations	100 percent for resident populations, 42 percent for migratory populations	100 percent for resident breeders	100 percent	100 percent for adult females
Foraging Habitat	Offshore zone within 30 to 50 km; 30 km off coast though can be found up to 175 km; within 20 km of coastal and island shores and in waters less than 150 m deep, prefers fronts with sharp thermal gradients but not the warmest water.	Pursues prey underwater, often to great depths (maximum depth is about 20 m) feeding throughout the water column; prefer water less than 9 m deep with rocky or gravel bottom; neritic, nearshore, freshwater, and littoral-benthic zones	Shoreline and upland areas	Estuaries, pelagic zones, open water, and beaches	Kelp bed habitats; located over the continental shelf in the shallower waters of the neritic zone; typically found in waters with an average depth of 323 m and ranging from 18 to 1,556.
Foraging Range/Distance	Most numerous within 20 km of nesting islands; from June to October, they are common along the coast in southern California within 30 km of shore but are regularly seen out to 175 km.	Usually forages within 8-16 km of roost or nest colony. Cormorants forage near shore, and those nesting on the Channel Islands may even fly to the mainland coast to feed.	19 nests averaged 5.3 km from the nearest foraging marsh, and 12.2 km from the nearest marsh over 130 ha in area. Foraging flights may extend as many as 19.2 km from the nest during nesting season.	Foraging distances were recorded for the three colonies located on Santa Catalina Island and ranged from 2 to 4.5 km. Generally forage within 5 km of the nest.	The average foraging range for female adults between June and July at San Miguel Island was 54.2 km (Antonelis et al. 1990).

[a] Islas Los Coronados is located in Mexico near the southern tip of the SCB. The breeding population of brown pelicans on this island is not evaluated in this ERA.

TABLE 3-3
Body Weight, Ingestion Rates, and Dietary Items for Representative Species in the Southern California Bight
Palos Verdes Shelf Ecological Risk Assessment

Receptor	Body Weight (kg)[a]	Notes	Reference	Ingestion Rates Food (kg/kg/day) (WW)	Notes and Reference	Water (L/kg/day)	Reference	Dietary Items	Reference
California Brown Pelican	3.3	Minimum of range for one captive adult	Schreiber (1976) as cited in Cal/EPA (2001)	0.153	Allometric equation from Nagy (1987); water content of fish (75 percent) assumed for conversion to wet weight (USEPA 1993)	Negligible	Water requirements met through diet	Northern anchovies (80 percent), Pacific sardines, Pacific mackerel (varies with food availability)	Bonnell and Dailey (1993)
Double-Crested Cormorant	2.16	Adult females, N=41, January-April in LA	Glahn et al. (1995) as cited in Cal/EPA 2001	0.178	Allometric equation from Nagy (1987); water content of fish (75 percent) assumed for conversion to wet weight (USEPA 1993)	Negligible	Water requirements met through diet	Cormorant feeds mainly on fish, but also on crustaceans and amphibians	Zeiner et al. (1990a)
Western Gull[b]	0.760-female 0.981-male **0.875-both**	Average adult weights	Pierotti and Annett (1995)	0.257	Allometric equation from Nagy (1987); water content of fish (75 percent) and marine invertebrates (82 percent) assumed for conversion to wet weight (USEPA 1993)	Negligible	Water requirements met through diet	Fish (85 percent) such as northern anchovies, jack mackerel, saury, midshipman; marine invertebrates (15 percent) such as market squid	Pierotti and Annett (1995)
Peregrine Falcon	0.84			0.193	Allometric equation from Nagy (1987); water content of birds (68 percent) assumed for conversion to wet weight (USEPA 1993)	NA	Water intake assumed from freshwater sources unrelated to PVS	Birds (100 percent); bats in some areas are also taken	Kiff (1980)
Bald Eagle	5-female 4-male **4.5-both**	Average adult weights	Stalmaster (1987) as cited in Glaser and Connolly (2002)	0.132	Allometric equation from Nagy (1987); (water content of mammals and birds (68 percent) and fish (75 percent) assumed for conversion to wet weight (USEPA 1993)	NA	Water intake assumed from freshwater sources unrelated to PVS	Fish (77.2 percent), birds (8.9 percent), mammals (7.9 percent), invertebrates (5.8 percent) (averaged over one year); fish caught in water around islands, washed-up marine mammal carcasses, and seabirds around islands	Garcelon (1994a,b)
California Sea Lions	**100-female** 380-male 240-Average of male and female	Weight for adults	Antonelis et al. (1980)	0.116	11.6 percent of body weight according to Costa (1986), calculated using female body weight	Negligible	Water requirements met through diet	Fish (100 percent): market squid, Pacific whiting, shortbelly rockfish, jack mackerel, chub mackerel, and northern anchovy	Lowry et al. (1990)

Notes:
[a] When more than one value is presented for a parameter, the chosen value is bolded.
[b] Risks to western gulls were not evaluated. Life-history values used to estimate concentrations in western gulls foraging throughout the SCB for use in exposure estimates for peregrine falcons and bald eagles.
Bold values were selected for use in the risk evaluation.

California Brown Pelican

Brown pelicans primarily feed on northern anchovies (80 percent of diet), but can also take other pelagic fish (e.g., sardines and mackerel; Tables 3-3 and 3-4). Historically, brown pelicans have accumulated high concentrations of tDDT and have suffered reproductive failure that was directly related to this exposure to DDT (Gress 1994). Additionally, concentrations of tDDT in brown pelican eggs and tissues have decreased and reproductive success has increased with decreasing sediment concentrations (Gress 1994), indicating that some portion of the diet is tied to the more highly contaminated sediments at PVS. Therefore, 20 percent of the diet was assumed to be other water-column fish and is represented by the kelp bass regression in the model. Concentrations of COPECs in pelagic fish are generally low and do not vary significantly with varying sediment concentrations. Therefore, pelagic fish are represented as a constant (tDDT = 0.0147 mg/kg, tPCB = 0.0077 mg/kg) in the exposure model and are estimated as the geometric mean value of COPEC concentrations in mackerel, Pacific bonito, and barracuda from the SCB (HydroQual, Inc. 1997). Benthic fish are not a part of the brown pelican diet, and other infrequent components (e.g., carrion, crustaceans, and young brown pelicans; Table 3-4) were not included in the model. Details on these relationships are described in Appendix C.

There were no distribution maps for brown pelicans that showed a measure of foraging density across the SCB (as was available for sea lions); however, a map depicting the major foraging locations for brown pelicans in the SCB was available (Baird 1993; Figure 3-4). Life-history information on pelican foraging behavior in the breeding and non-breeding seasons was used to create two exposure distributions. During the breeding season (March to August), pelicans forage most heavily within 30 to 50 km of the nesting island (Anacapa or Santa Barbara Island; USFWS 1983) and during all seasons are usually found within 20 km of the shore and in waters less than 150 m deep (Briggs et al. 1983). (Note: Although the breeding population of brown pelicans on Islas Los Coronados, Mexico, has historically experienced effects from tDDT exposure attributed to foraging within the SCB [Gress 1994, Gress et al. 2003], this breeding population is not evaluated due to a lack of data from the immediate areas surrounding this island.) Therefore, exposure distributions were limited to 20-km radii of the nesting islands to the 150-m depth contour during the breeding season and 20 km of the shore (as shown in the foraging distribution map) to the 150-m depth contour during the non-breeding season. Total DDT and tPCB exposure distributions for brown pelicans in the SCB are depicted in Figure 3-5.

Double-Crested Cormorant

Double-crested cormorants primarily feed on schooling fish that occur from surface to near bottom (Ainley et al. 1981; Tables 3-3 and 3-4). As with brown pelicans, double-crested cormorants have experienced high tDDT accumulation and reproductive failure, and have shown improved reproductive success as the tDDT concentrations in sediment decreased. Therefore, a proportion of their diet was assumed to be tied to the sediment. Because schooling fish can encompass pelagic, water-column, and benthic species, the diet is presumed to consist of all three types of fish. Pelagic fish were considered to be the predominant type of prey (50 percent of diet) and are represented as a constant (tDDT = 0.0147 mg/kg, tPCB = 0.0077 mg/kg) (HydroQual, Inc. 1997) in the exposure model. Water-column fish (30 percent of diet) were represented by kelp bass regressions and benthic fish were divided between white croaker (5 percent of diet) and sanddabs (15 percent of diet) and were described by the respective regressions for these species. Infrequent components of the cormorant diet (e.g., crustaceans and amphibians) were not included in the model. Development of these relationships is outlined in Appendix C.

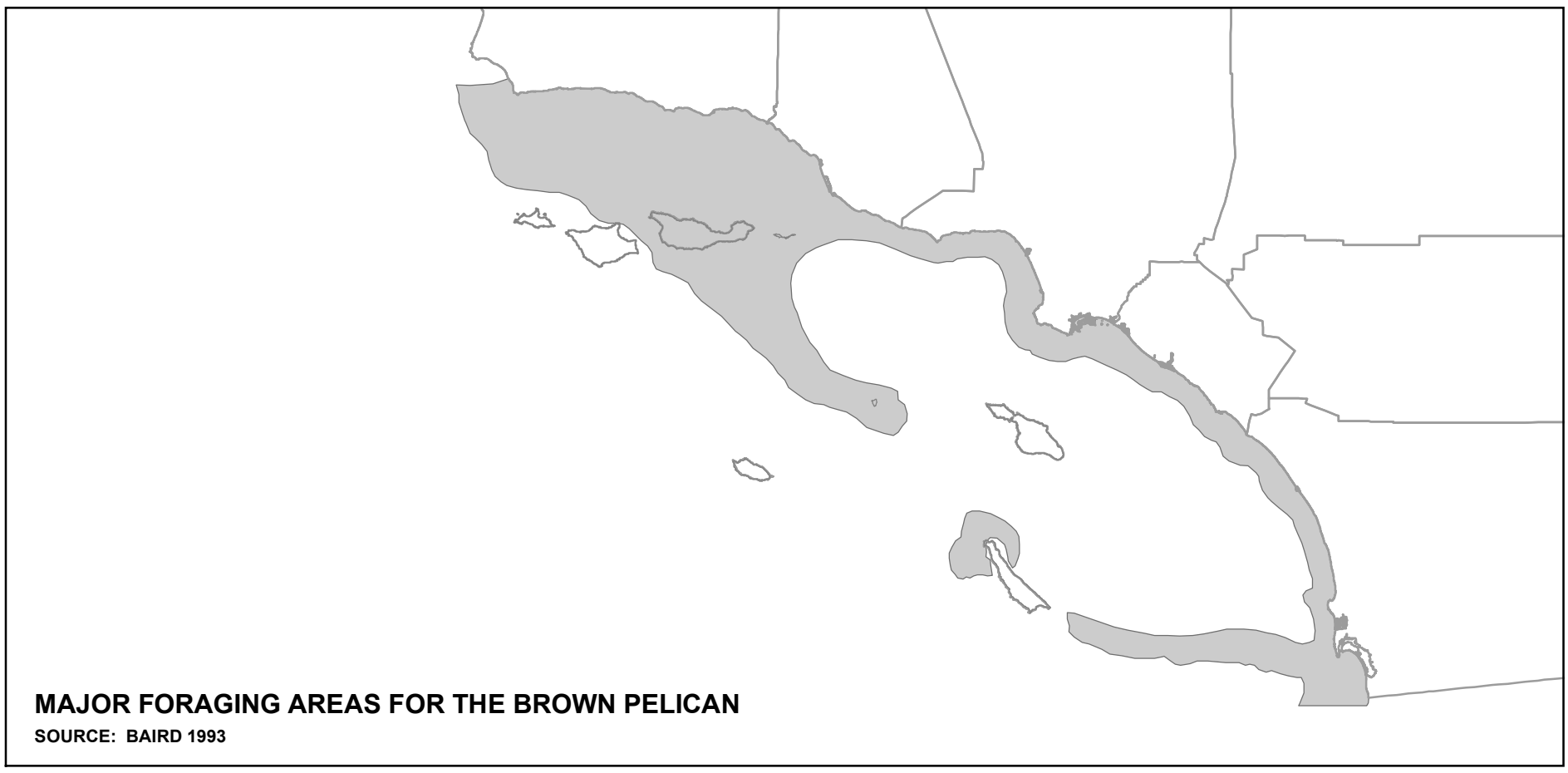
TABLE 3-4
Information Used for the Development of a Surrogate Diet for Each Bird and Mammal Receptor
Palos Verdes Shelf Ecological Risk Assessment

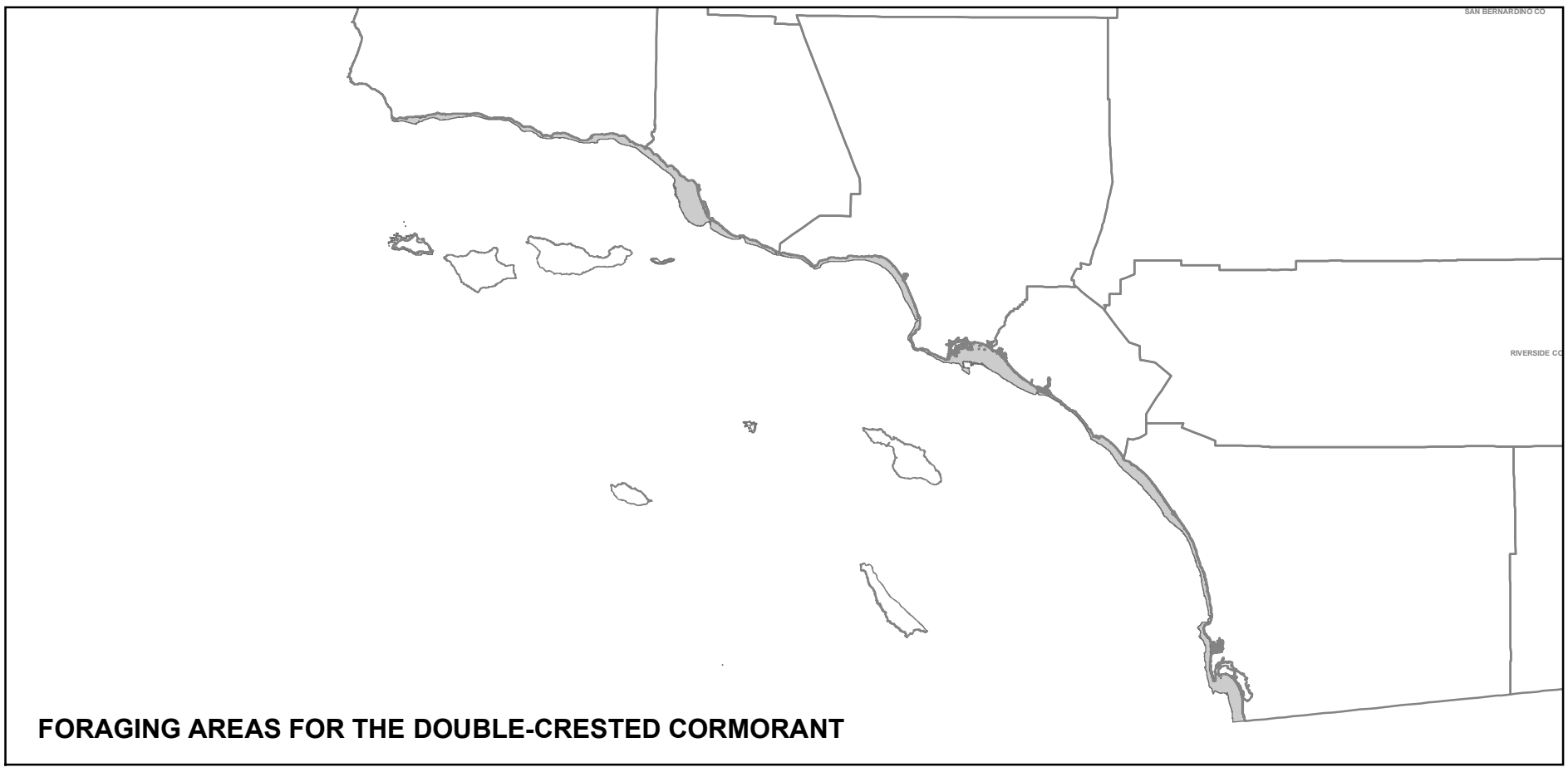
Receptor	General Diet	Diet	Proportion (wet weight basis)	Proportion (numerical basis)	Proportion (energy basis)	Surrogate Diet	Proportion
Brown Pelican[a]	Northern anchovy (80 percent); primarily fish, some crustaceans, carrion, and young pelicans	northern anchovy	0.8	—	—	pelagic fish	0.80
		other fish (Pacific sardines, Pacific mackerel)	0.2	—	—	kelp bass	0.20
Double-Crested Cormorant[b]	Cormorant feeds mainly on schooling fish that occur from surface to near surface with no relief, but also on crustaceans and amphibians	fish (water column)	0.7	—	—	pelagic fish	0.50
		fish (bottom)	0.3	—	—	kelp bass	0.30
						sanddabs	0.15
						white croaker	0.05
Peregrine Falcon[c]	Birds and bats (assume birds 100 percent)	western gulls	0.11	—	0.096	western gulls	0.68
		California gulls	0.092	—	0.094	land birds—resident	0.19
		Heermann's gulls	0.026	—	0.022	land birds—migratory	0.13
		Bonaparte's gulls	0.022	—	0.022		
		Cassin's auklets	0.17	—	0.18		
		other waterbirds	0.26	—	0.26		
		land birds—resident	0.19	—	0.19		
		land birds—migratory	0.13	—	0.13		
Bald Eagle[d]	Fish (77.2 percent), birds (8.9 percent), mammals (7.9 percent), invertebrates (5.8 percent) (averaged over one year)	fish and invertebrates		0.86	0.79	pelagic fish	0.21
		sea lions		0.027	0.058	kelp bass	0.50
		other mammals		0.006	0.0073	sanddabs	0.10
		birds—western gulls		0.026	0.033	sea lions	0.06
		other gulls		0.009	0.013	seabirds	0.13
		waterbirds		0.062	0.085		
		land birds—resident		0.005	0.0078		
		land birds—migratory		0.004	0.0053		

TABLE 3-4
Information Used for the Development of a Surrogate Diet for Each Bird and Mammal Receptor
Palos Verdes Shelf Ecological Risk Assessment

Receptor	General Diet	Diet	Proportion (wet weight basis)	Proportion (numerical basis)	Proportion (energy basis)	Surrogate Diet	Proportion
Western Gull[e]	Fish and crustaceans (will assume 100 percent fish)	northern anchovies, jack mackerel, Pacific saury, market squid		—	—	pelagic fish	0.70
		rockfish		—	—	kelp bass	0.10
		midshipmen, white croaker, spotted cusk eel		—	—	sanddabs	0.15
						white croaker	0.05
California Sea Lion[f]	Market squid, Pacific whiting, shortbelly rockfish, jack mackerel, chub mackerel, and northern anchovy (assume 100 percent fish; pelagic fish used as surrogate for squid[g])	northern anchovy		0.247	—	pelagic fish	0.50
		jack mackerel		0.118	—	kelp bass	0.20
		Pacific mackerel		0.046	—	sanddabs	0.25
		Pacific whiting		0.094	—	white croaker	0.05
		rockfish		0.092	—		
		market & other squid		0.09	—		
		pelagic red crab[h]		0.102	—		
		octopus		0.025	—		
		blacksmith		0.051	—		
		other fish		0.133	—		

[a] Sources for general diet are Bonnell and Dailey (9193) and Ziener et al. (1990a).
[b] Source for general diet is Ainley et al. (1981).
[c] Source for general diet is Hung (1994).
[d] Sources for general diet are Garcelon (1994a, b).
[e] Sources for general diet are Baird (1993) and Pierotti and Annett (1995). Risks to western gulls were not evaluated. Life-history values were used to estimate concentrations in western gulls foraging throughout the SCB for use in exposure estimates for peregrine falcons and bald eagles.
[f] Source for general diet is Lowry et al. (1990).
[g] See discussion in subsection 3.1.4.1 and Appendix B regarding the use of pelagic fish as a surrogate species for market squid.
[h] Not local, they migrate in from the south during El Niño years.





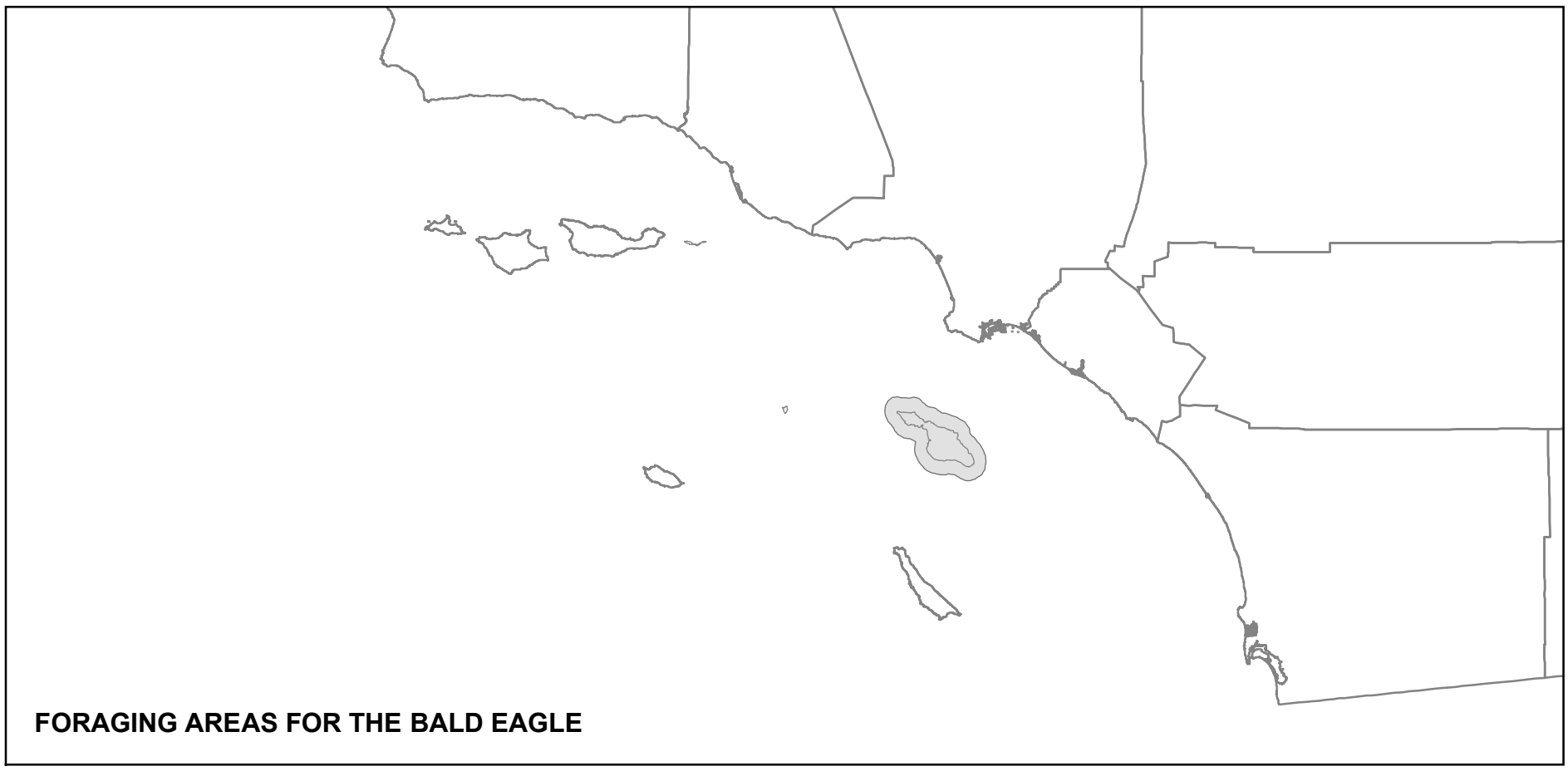


FIGURE 3-4
AVIAN FORAGING AREAS

PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

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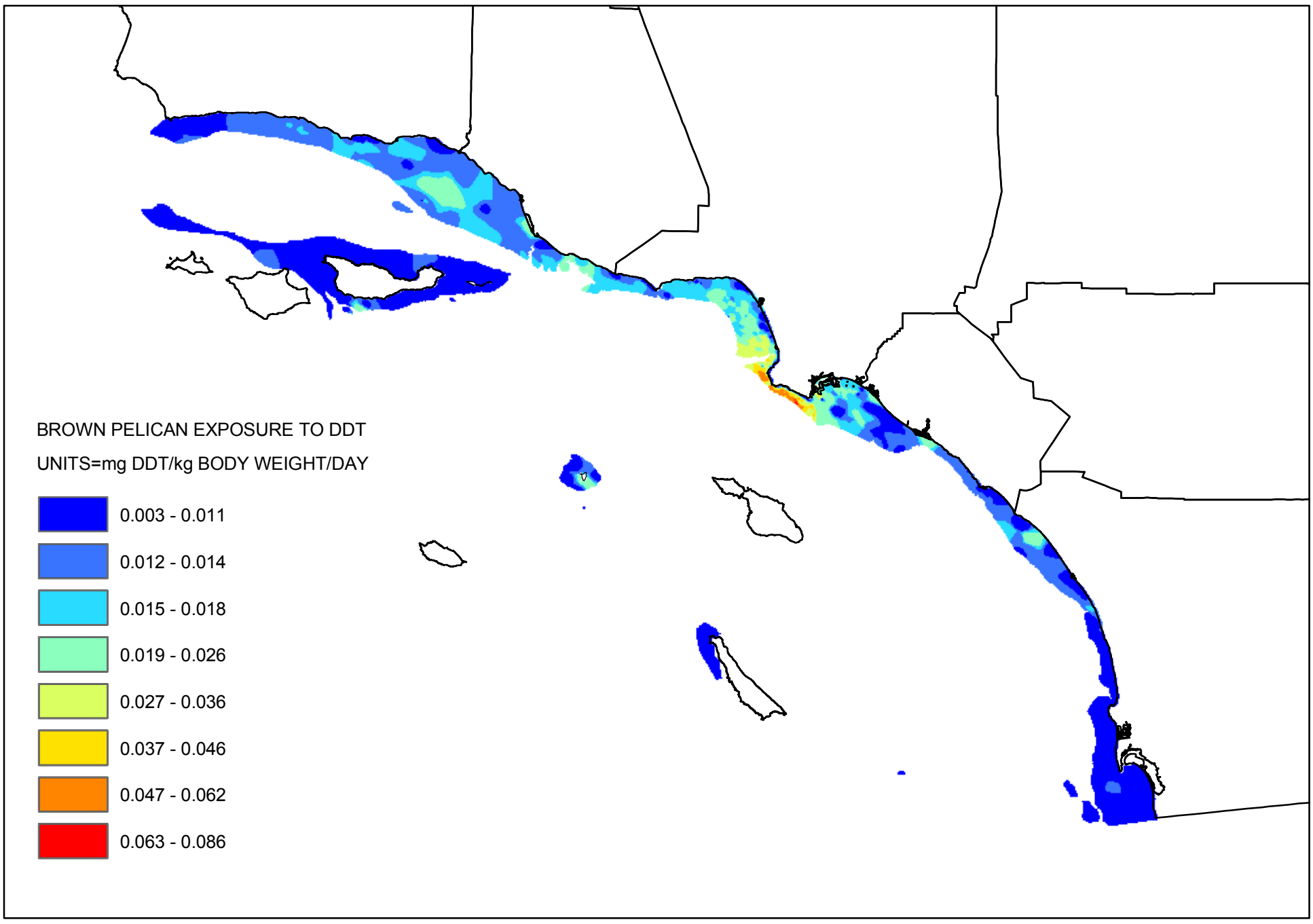

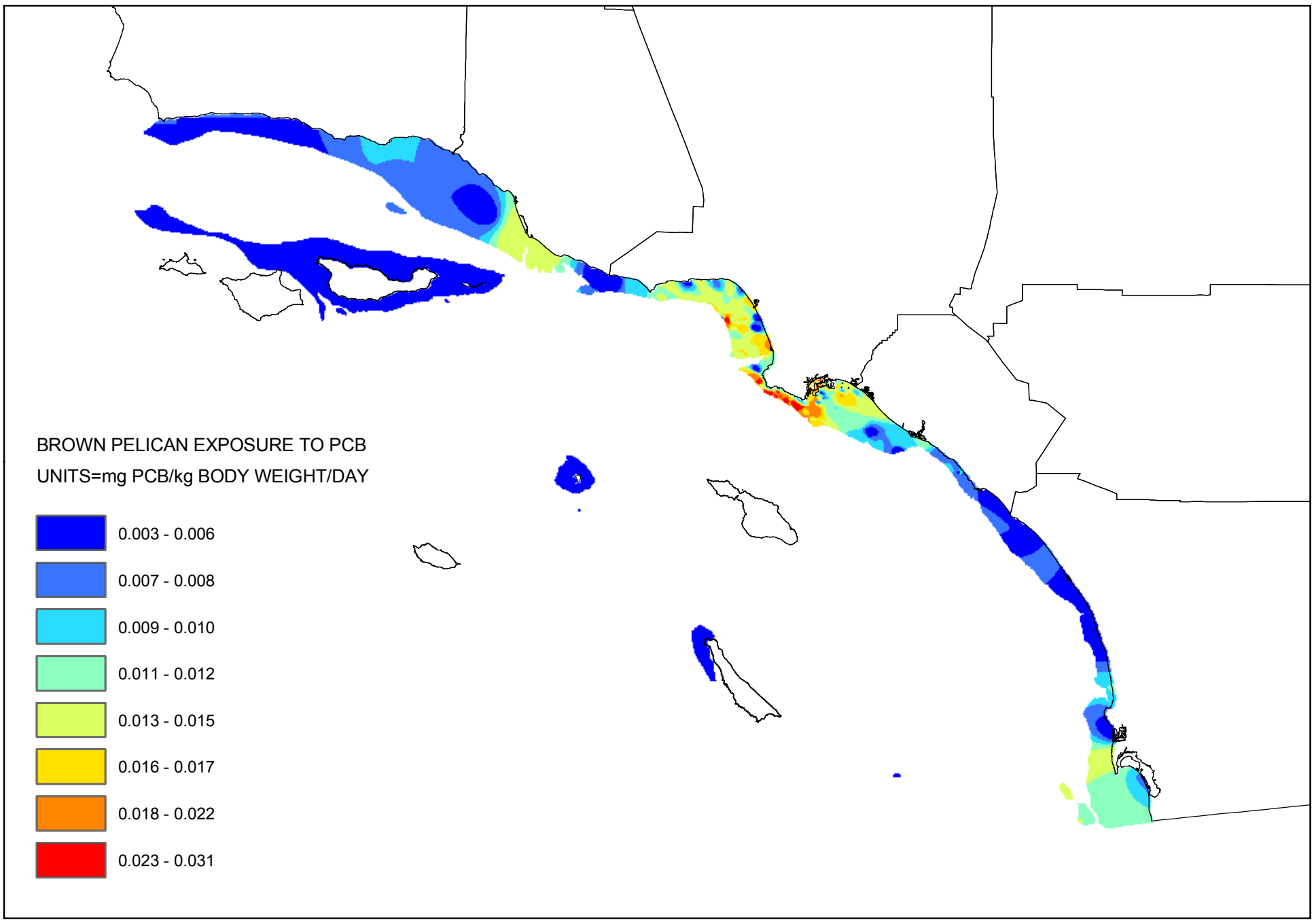

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SOURCE: BAIRD 1993

FIGURE 3-5
BROWN PELICAN
EXPOSURE VALUES TO 150 METERS

PALOS VERDES ECOLOGICAL RISK ASSESSMENT

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There were no distribution maps for double-crested cormorants that showed a measure of foraging density across the SCB (as was available for sea lions) or that depicted the major foraging locations (as for brown pelicans). Therefore, life-history information on cormorant foraging behavior in the breeding and non-breeding seasons was used to create a foraging map (Figure 3-4) and two exposure distributions. During the breeding season (April to August), cormorants forage within 16 km of the nesting island (Anacapa and Santa Barbara Islands) and may forage on the mainland coast nearest the breeding islands (Bonnell and Dailey 1993). Outside of the breeding season, cormorants forage throughout the SCB, and in both seasons prefer to forage in water with depths of 22 m or less (Zeiner et al. 1990a). Therefore, exposure distributions were limited to 16-km radii of the nesting islands up to the 22-m depth contour during the breeding season and along island and mainland coasts to the 22-m depth contour in the non-breeding season. Estimates of exposure of double-crested cormorants to the COPECs throughout the SCB are shown in Figure 3-6.

Peregrine Falcon

Peregrine falcons primarily feed on birds, and in the SCB these are made up of seabirds (e.g., gulls, auklets), other water birds (e.g., grebes, shearwaters, shorebirds), and resident and migratory land birds (Hunt 1994; Table 3-4). The proportions (energy basis) of these species or groups of species used in the exposure model correspond to those listed in the SCB avian food pathways analysis by HydroQual, Inc. (1997) and Glaser and Connolly (2002). Bats may also be consumed by peregrines occasionally, but these are not included in the model.

The resident land birds (19 percent of diet) and migratory land birds (13 percent of diet) are constant in the model. Total DDT and tPCB values (resident—tDDT and tPCB = 0 mg/kg; migratory—tDDT = 0.26 mg/kg and tPCB = 0.33 mg/kg) for these prey items presented in Glaser and Connolly (2002) were used in the exposure model. To estimate oral exposure in peregrines, it was necessary to estimate whole-body seabird concentrations from fish ingestion. The peregrine diet consists of a variety of seabirds and other waterbirds with varying concentrations of tDDT and tPCB (Table C-2, Appendix C). Because a foraging range (Figure 3-6) and specific life-history parameters (Table 3-3) were available for western gulls, a composite seabird that varies relative to western gull concentrations was developed. Whole-body COPEC concentrations in western gulls (mg/kg WW) were estimated using methods similar to those used for calculating oral exposure in pelicans and cormorants. That is, a surrogate diet was developed (Table 3-4) and the map of the major foraging areas of western gulls (Figure 3-7) was used to define a distribution of exposure. The western gull is primarily piscivorous, but also eats marine invertebrates, carrion, and garbage (Zeiner et al. 1990a). Pierotti and Annett (1995) report a diet of 85 percent fish and 15 percent squid. Fish include pelagic, water column, and benthic species, though percentages are not given for each type. Based on a diet comprised of northern anchovies, Pacific whiting, jack mackerel, Pacific sauries, rockfish, midshipmen, white croaker, and spotted cusk eel (Pierotti and Annett 1995), a surrogate diet assuming 70 percent pelagic fish, 10 percent water column fish, and 20 percent benthic fish was developed. Pelagic fish are a constant in the model (tDDT = 0.0147 mg/kg, tPCB = 0.0077 mg/kg; HydroQual, Inc. 1997), water column fish are represented by kelp bass, and benthic fish are divided between white croaker (5 percent) and sanddabs (15 percent, Table 3-4).

No distribution maps that showed a measure of foraging density across the SCB were available; however, a map depicting the major foraging locations for western gulls in the SCB (Baird 1993; Figure 3-7) was available. Western gulls forage in the top 1 to 2 meters of

the water column (Briggs et al. 1987), generally over much deeper waters. As with pelicans, it was assumed that western gulls do not forage in waters deeper than 150 m. After oral exposure distributions were developed for western gulls, it was necessary to estimate COPEC concentrations in gull tissue based on the oral exposure. Although HydroQual, Inc. (1997) and Glaser and Connolly (2002) do not present models specific for western gulls, those developed for cormorants can be used for gulls because excretion for gulls (an important model parameter) can be estimated from an allometric equation presented in HydroQual, Inc. (1997) and Glaser and Connolly (2002). Therefore, portions of the cormorant model were used to estimate whole-body tDDT and tPCB concentrations (mg/kg WW) in western gulls from the oral exposure distributions (see Appendix C for a more detailed description). These western gull concentrations were incorporated into the composite seabird model described below to produce a distribution of concentrations in the seabird portion of the peregrine's diet.

DDE and PCB concentrations and proportion of diet of known prey items (western gull, California gull, Heermann's gull, Bonaparte's gull, Cassin's auklet, and other water birds) collected throughout the SCB and outlined in HydroQual, Inc. (1997) and Glaser and Connolly (2002) were used and resulted in tDDT and tPCB equations for the composite seabird as follows:

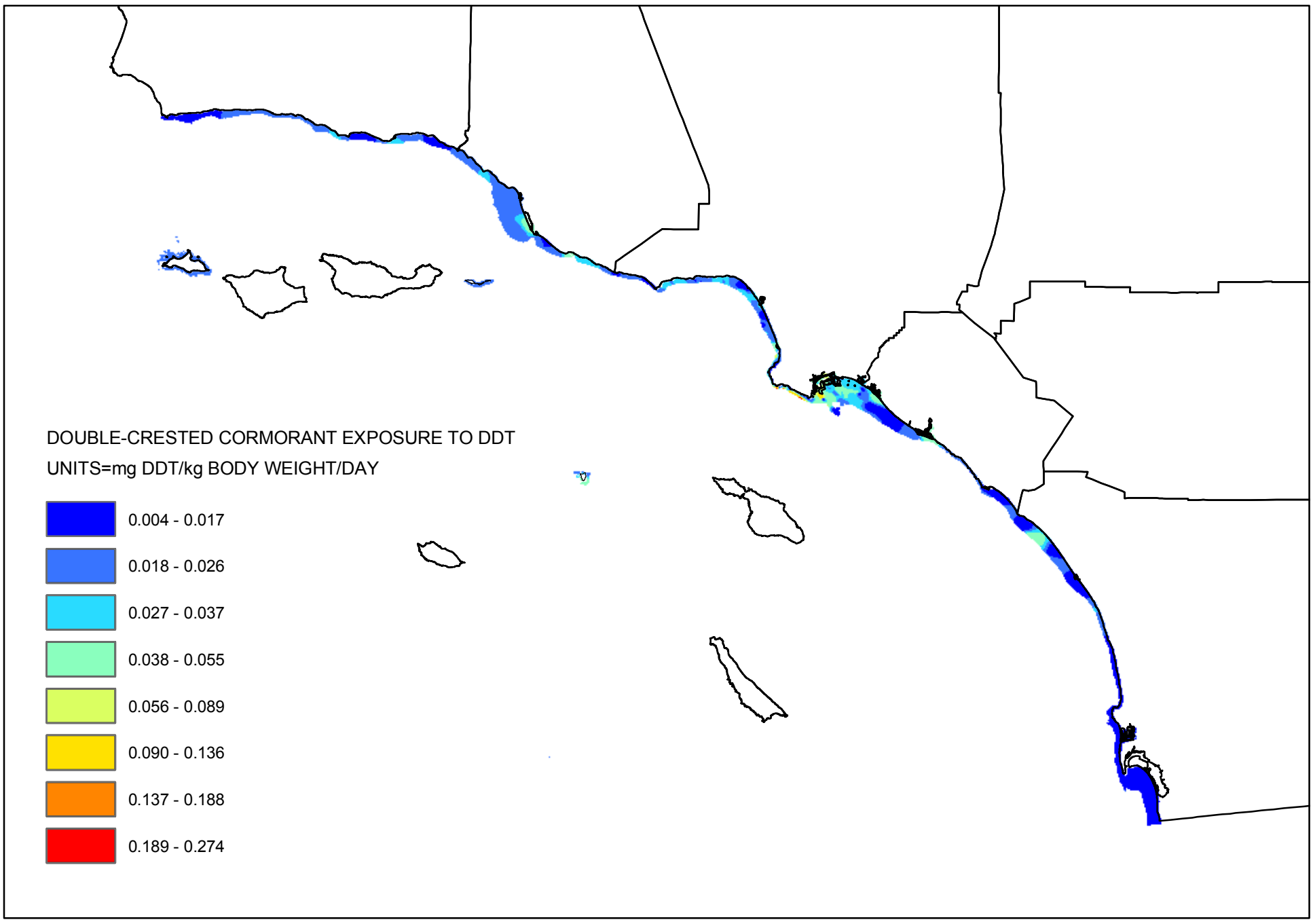
$$tDDT = 0.563(W) \text{ and } tPCB = 0.486(W) \qquad \text{[Equation 3.5]}$$

Where W = western gull tDDT or tPCB concentration (whole body, mg/kg WW)

Although some winter movements of Channel Island peregrines occur, there is no true migration of this species (PCAPFRT 1982). Peregrines breed on San Miguel, Santa Rosa, Santa Cruz, Anacapa, and Santa Barbara Islands in the SCB. They have also been found to breed under the Vincent Thomas Bridge in the Port of Los Angeles, as well as near the Port of Long Beach, San Diego Bay, and Point Loma. Peregrines forage approximately 12 km from the nesting site (Porter and White 1973, as cited in Zeiner et al. 1990a) and do not take prey over open water (Fry 2002); therefore, the foraging ranges of Channel Island peregrines are limited to the breeding island or other nearby islands within 12 km. Estimates of exposure of peregrine falcons to the COPECs throughout the SCB are shown in Figure 3-8. These distributions do not represent actual foraging locations of the peregrines (which do not forage over water), but rather depict foraging areas of the peregrine seabird prey (i.e., western gulls). Therefore, exposure to peregrines is discussed in the context of the foraging locations of their prey.

Bald Eagle

Bald eagles residing in the SCB primarily consume fish and, to a lesser extent, prey on gulls, other water birds (e.g., grebes, shearwaters, auklets), sea lion carcasses, resident and migratory land birds, and other mammals (Garcelon 1994a,b; Tables 3-3 and 3-4). The proportions (energy basis) of these species or groups of species used in the exposure model correspond to those listed in the SCB avian food pathways analysis by HydroQual, Inc. (1997) and Glaser and Connolly (2002). The land birds and other mammals make up less than 2 percent of the total diet and were excluded from the exposure model. It should also be noted that although sea lions make up a small numerical portion of the diet, they can substantially contribute to contaminant exposure in bald eagles (HydroQual, Inc. 1997).



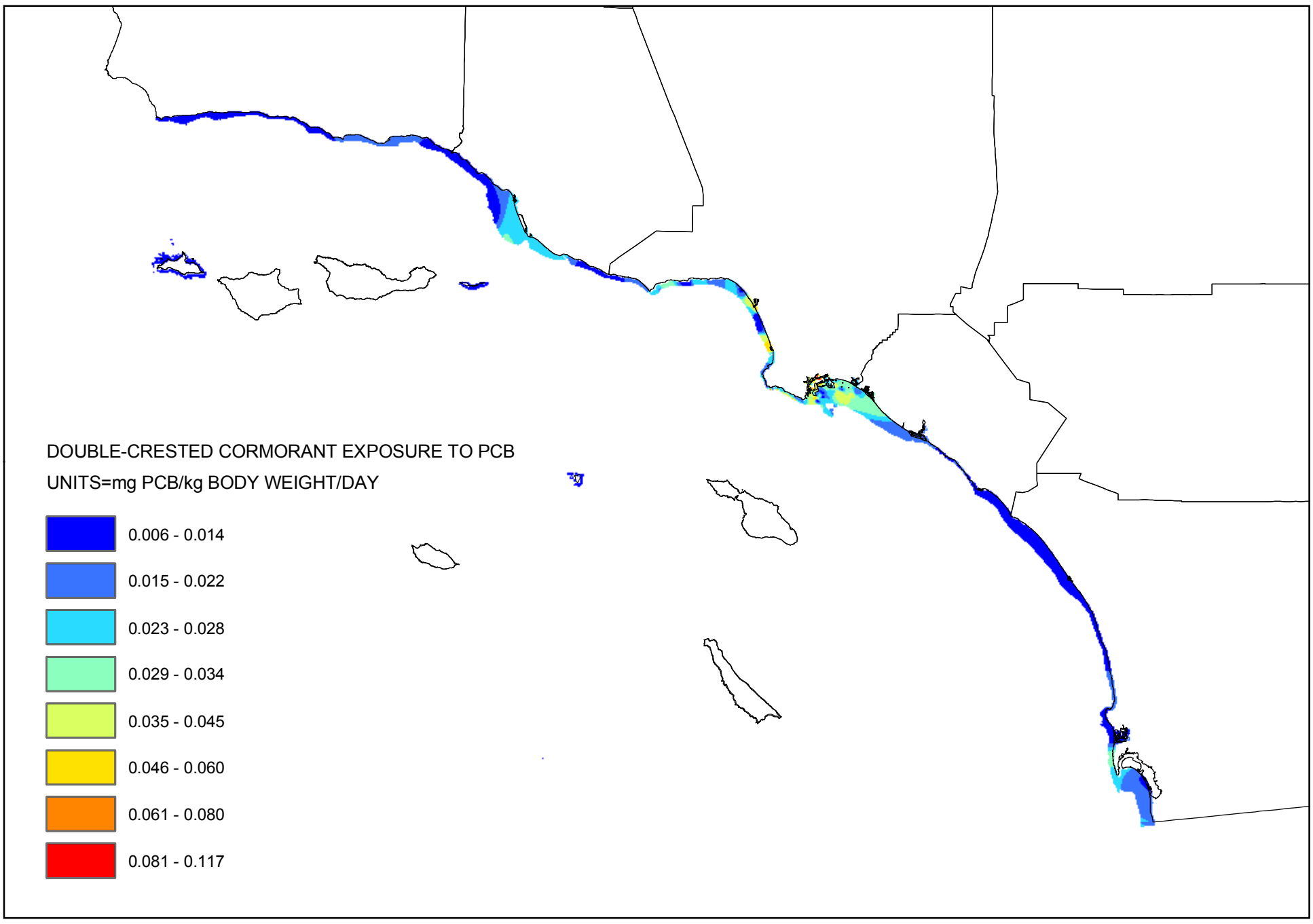


FIGURE 3-6
DOUBLE-CRESTED CORMORANT EXPOSURE VALUES TO 22 METERS

PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

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FIGURE 3-7
FORAGING AREAS FOR THE WESTERN GULL

PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

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Fish make up approximately 79 percent of the bald eagle diet at Santa Catalina Island. Of these, the majority are water column species (assumed 50 percent of diet) such as kelp bass, halfmoon, garibaldi, opaleye, and rockfish and were represented in the model by the kelp bass regression. Pelagic fish such as blacksmelt and mackerel are also important components of the bald eagle diet (assumed to be 21 percent of diet) and were represented by constants in the model (tDDT = 0.0147 mg/kg WW, tPCB = 0.0077 mg/kg WW) (HydroQual, Inc. 1997). Benthic fish are small components of the bald eagle diet (10 percent) and were represented in the model by the sanddab regressions.

To estimate bald eagle oral exposure from the seabird and sea lion components of their diet, it was necessary to estimate whole-body seabird and sea lion concentrations based on their consumption of fish. For sea lions, portions of the exposure model and associated parameters (e.g., excretion rate, assimilation efficiency) developed in HydroQual, Inc (1997) and Connolly and Glaser (2002) were used to determine body burdens in sea lions (mg/kg WW) using the oral exposure model developed for adult female sea lions as described below. Detailed methods for the estimation of whole-body sea lion concentrations are presented in Appendix C. A recent study (Le Boeuf 2002) indicated that most (64 percent) California sea lions observed on Santa Catalina Island from February to November 2000 were adult males; therefore, the model was modified to account for differences in excretion rates between adult (reproducing) females and adult males. It was also assumed that sea lions loafing on Santa Catalina Island were also those most likely to die and wash up on shore providing food for the bald eagles. Though it is acknowledged that breeding males do not forage in the SCB during the breeding season and generally migrate northward in the non-breeding season, the adult and juvenile males observed on Santa Catalina for nine months of the year are presumed to forage locally. To best represent these local individuals, the exposure distribution for adult male sea lions was limited to a 54 km radius (foraging distance of female sea lions during the breeding season; Antonelis et al. 1990) of the island and to a water depth of 200 m.

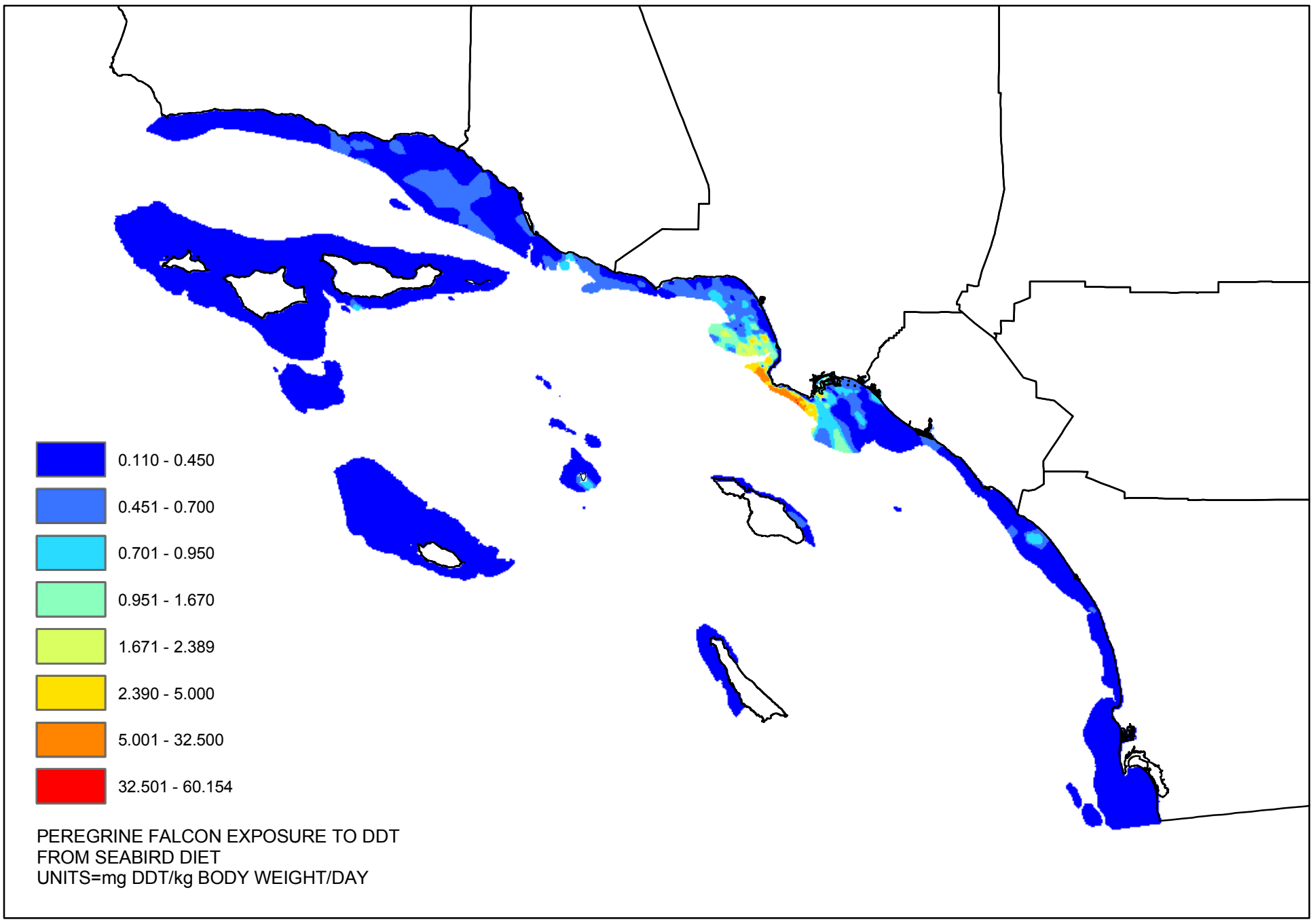
The western gull was used to estimate seabird concentrations in the bald eagle diet. To accomplish this, a "composite seabird" was developed that varies relative to the western gull concentrations and has a concentration of tDDT or tPCB at about one-half those in western gulls. COPEC concentrations in western gulls were estimated as described above for peregrine falcons.

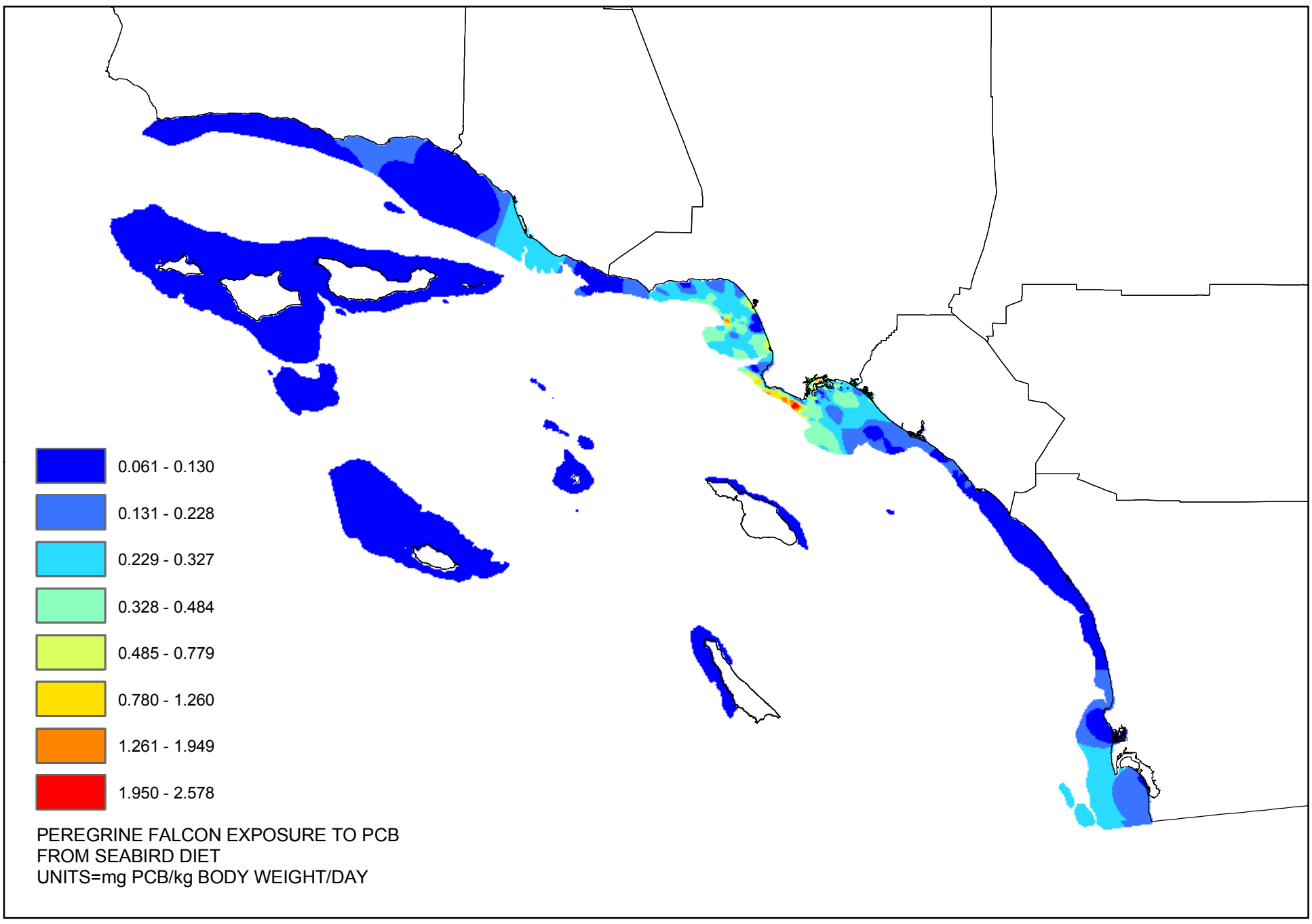
The DDE and tPCB concentrations and proportion of diet of known bald eagle prey items (western gull, other gulls, and other water birds) collected throughout the SCB and outlined in HydroQual, Inc. (1997) and Glaser and Connolly (2002) were used. Resulting tDDT and tPCB equations for the composite seabird are as follows:

$$tDDT = 0.458(W) \text{ and } tPCB = 0.477(W) \qquad \text{[Equation 3.6]}$$

Where: W = western gull tDDT or tPCB concentration (whole body, mg/kg WW)

It was assumed that western gulls nesting on Santa Catalina Island are most likely to become prey of bald eagles. Western gulls forage up to 80 km from the colony, though often within 20 km (Pierotti and Annett 1995). The size of the foraging range is dependent on food availability and dietary preferences of individuals. To account for these differences, a foraging range midway between 20 and 80 km (i.e., 50 km) was selected. Whole-body western gull





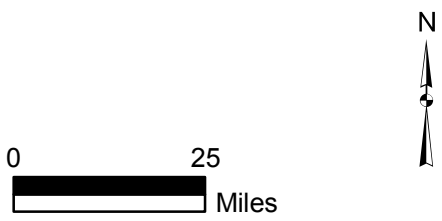


FIGURE 3-8
PEREGRINE FALCON
EXPOSURE VALUES TO 150 METERS

PALOS VERDES ECOLOGICAL RISK ASSESSMENT

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concentrations were modeled within a 50 km radius of Santa Catalina Island to the 150 m depth contour and limited to the major foraging areas depicted in Figure 3-7.

Channel Island bald eagles are non-migratory (Garcelon 1994b), and currently breed on Santa Catalina Island in the SCB. Juvenile bald eagles have recently been released on Santa Cruz Island and have been sighted near Anacapa and Santa Rosa Islands (Hoecker 2002). Bald eagles nesting at Santa Catalina Island forage up to 2.5 to 4.5 km from the nesting site (Garcelon 1994a). A radius of 5 km around each nesting site on Santa Catalina Island (Figure 3-4) was used to calculate exposure in the bald eagle. Because marine mammal and seabird prey of bald eagles forage over larger ranges than the bald eagle, exposure to the bald eagle is best depicted across the foraging ranges of the prey items as was done for the peregrine falcon. However, the three major components of the bald eagle diet (i.e., fish, marine mammals, and seabirds) have three different foraging ranges. Thus, it was not possible to incorporate all three distributions of whole-body concentrations of the prey items into the bald eagle exposure distribution. Instead, the 95 percent UCL of the geometric mean of the sea lion and the western gull whole-body concentration distributions were each incorporated into the model to estimate bald eagle exposure over the foraging range of the eagle. Due to limited variability in tDDT and tPCB concentrations in sediment surrounding Santa Catalina Island, the fish portion of the diet did not vary substantially. This resulted in essentially a single value for bald eagle exposure with tDDT ranging from 0.506 to 0.530 mg/kg/d and tPCB equal to 0.297 mg/kg/d.

To provide more spatial information on bald eagle exposure, the distributions of exposure from the sea lion and western gull portions of the bald eagle diet are depicted in Figures 3-9 and 3-10, respectively. These distributions do not represent actual foraging areas of the bald eagle (which are limited to 5 km of nesting site), but rather depict foraging areas of the sea lion and western gull prey. Therefore, exposure to bald eagles is also discussed in the context of the foraging locations of their prey.

Adult Female California Sea Lion

California sea lions are opportunistic feeders and ingest a large variety of fish species composed of pelagic (e.g., mackerel), water-column (e.g., rockfish), and benthic (e.g., flatfish, white croaker) fish. Invertebrates, such as market squid and pelagic red crabs, can also make up a large part of the diet at times. The diet presented by Lowry et al. (1990) was used to develop a surrogate diet for sea lions in the SCB. Of this diet, northern anchovy, jack mackerel, and Pacific mackerel were considered pelagic fish (40 percent of the diet), rockfish and blacksmith were considered water-column fish (20 percent of the diet), and other fish were considered to be benthic (13 percent of the diet).

Market squid are an important part of the sea lion diet in the SCB, making up 10 percent of the total diet. The measured COPEC data for market squid in the SCB are very limited (only 3 samples) and were not collected during 1990-2001 (collected in 1980 or 1981; Mearns et al. 1991). Therefore, pelagic fish were used as a surrogate species for market squid. Both species are water column feeders with often similar diets that forage over wide ranges (see Appendix C for a more detailed discussion).

Although pelagic red crab comprise 10 percent of the diet described by Lowry et al. (1990), these crabs are normally found off the Pacific coast of Baja and are only found in the SCB during El Niño events (Oceanlight 2003). Therefore, these crabs were assumed to be an

infrequent component of the sea lion diet and were not represented in the surrogate diet. Because sea lions are known to eat flatfishes and other benthic fish, the remaining 17 percent of the diet was considered to be benthic fish.

Concentrations of COPECs in pelagic fish are generally low and do not vary significantly with varying sediment concentrations. Therefore, pelagic fish are represented as a constant (tDDT = 0.0147 mg/kg, tPCB = 0.0077 mg/kg; HydroQual, Inc. 1997) in the exposure model. Water-column fish were represented by kelp bass regressions and benthic fish were divided between white croaker (5 percent) and sanddabs (25 percent) and were described by the respective regressions for these species. Development of these relationships is outlined in Appendix C.

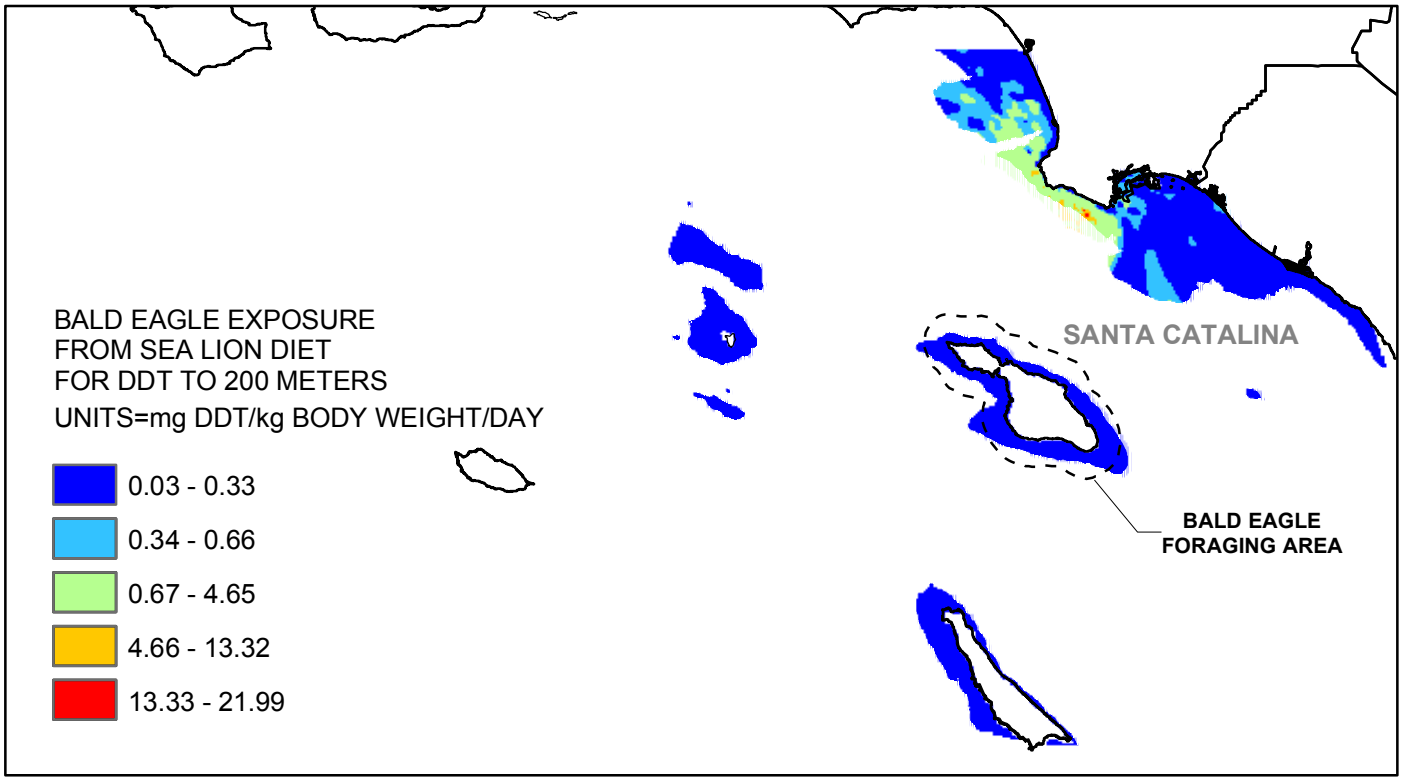
The second step in the exposure estimate was to overlay the California sea lion dietary exposure with the foraging areas. Seasonal distribution maps were available for sea lions in the SCB (Bonnell and Ford 1987), and are shown in Figure 3-11. These maps were created using data from 23 aerial surveys conducted over a 3-year period. Data from these surveys were pooled for three seasons (September through November, December through May, and June through August), and sightings were assigned to 30-foot by 30-foot grid cells centered on degree and half-degree lines of latitude and longitude. The distribution maps present density contours based on animals per square kilometer, and allow weighting of the sea lion exposure based on area use of the SCB. Foraging in the Channel Islands occurs to the 200 m depth contour (BLM 1981); therefore, exposure distributions are limited to this depth.

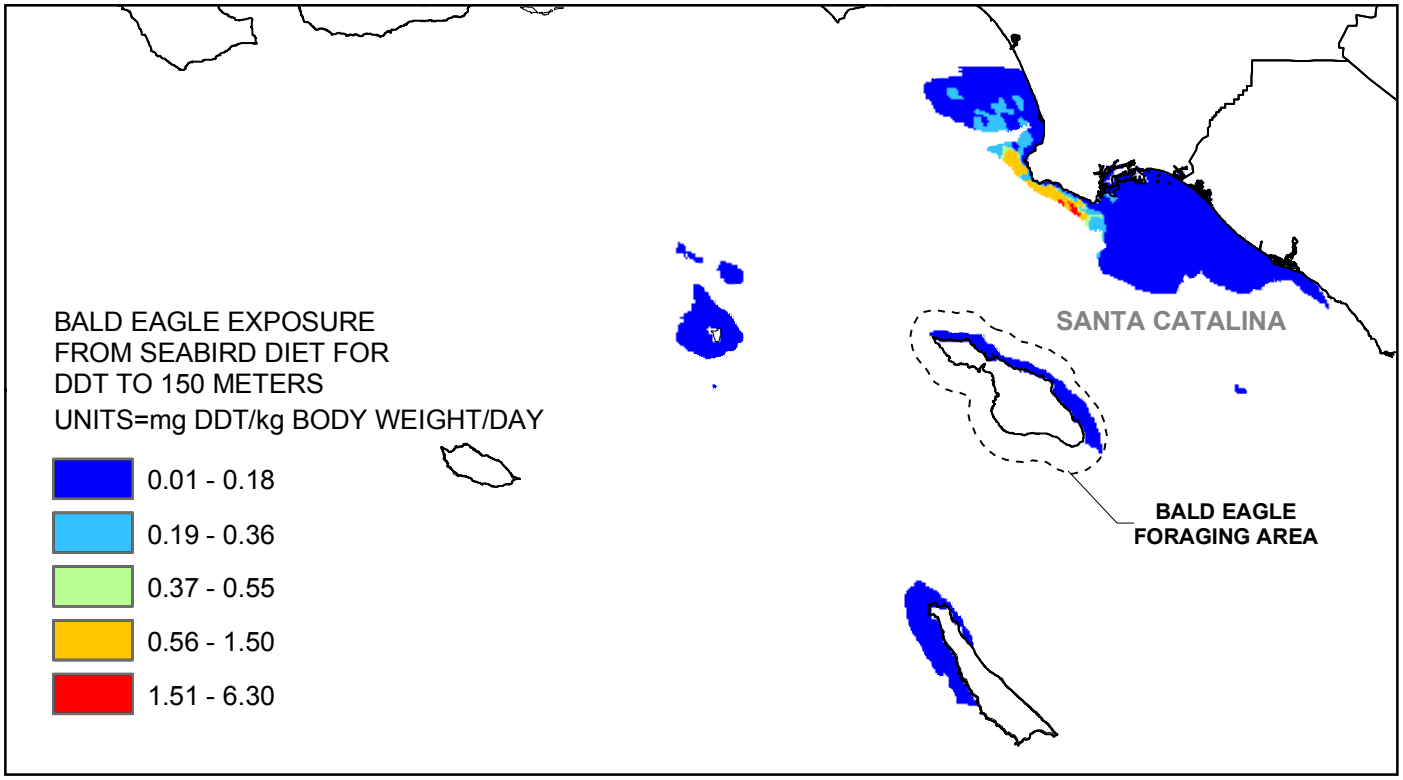
Finally, only adult female sea lions are evaluated. This is because adult male sea lions were excluded from the tDDT and tPCB studies on San Miguel Island and because the adult males migrate to the northwestern U.S. coast during the non-breeding season (Glaser and Connolly 2002). Seasonal estimates for female sea lion exposure to tDDT and tPCB throughout the SCB are shown in Figure 3-12 and 3-13, respectively.

Nursing California Sea Lion Pup

Contaminants in the female California sea lion are transferred directly to the fetus and then via milk to the nursing pup postpartum. Because of the high lipid content of California sea lion milk (32-44 percent; Oftedal et al. 1987) relative to the average female lipid content (30 percent, Connolly and Glaser 2002) lipophilic compounds such as tDDT and tPCB are preferentially transferred to the milk. This results in exposures to nursing pups that are greater than those experienced by the adult female. California sea lion pups rely soley on the mother's milk for approximately the first seven months of their life and generally begin eating solid food after the first 200 days (Boness et al. 1991). Because this is a time of high exposure during a sensitive life-stage (rapid growth and development), pups up to age 7 months were evaluated in the current ERA.

Estimation of oral exposure to nursing pups required determining the concentration of COPECs in adult female sea lion tissue and estimating the transfer of these concentrations to the milk. Portions of the exposure model and associated parameters developed in HydroQual, Inc (1997) and Connolly and Glaser (2002) were used to determine concentrations in sea lions (mg/kg WW) from the oral exposure distributions developed for adult sea lions as described above. Next, COPEC concentrations in milk were estimated from adult female sea lion tissue concentrations using the transfer model developed in





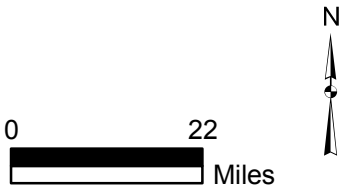


FIGURE 3-9
BALD EAGLE EXPOSURE VALUES
FROM SEABIRD AND SEA LION DIET FOR DDT
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

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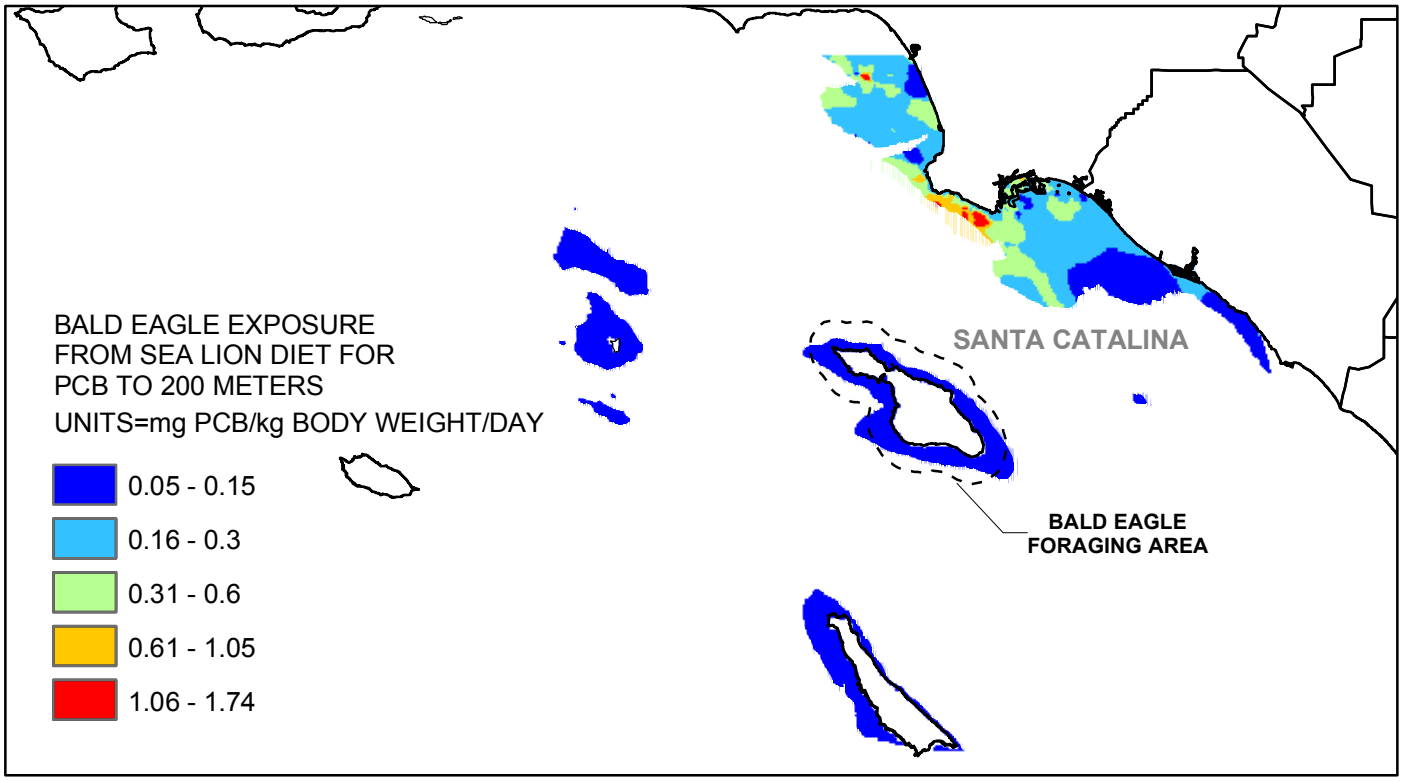

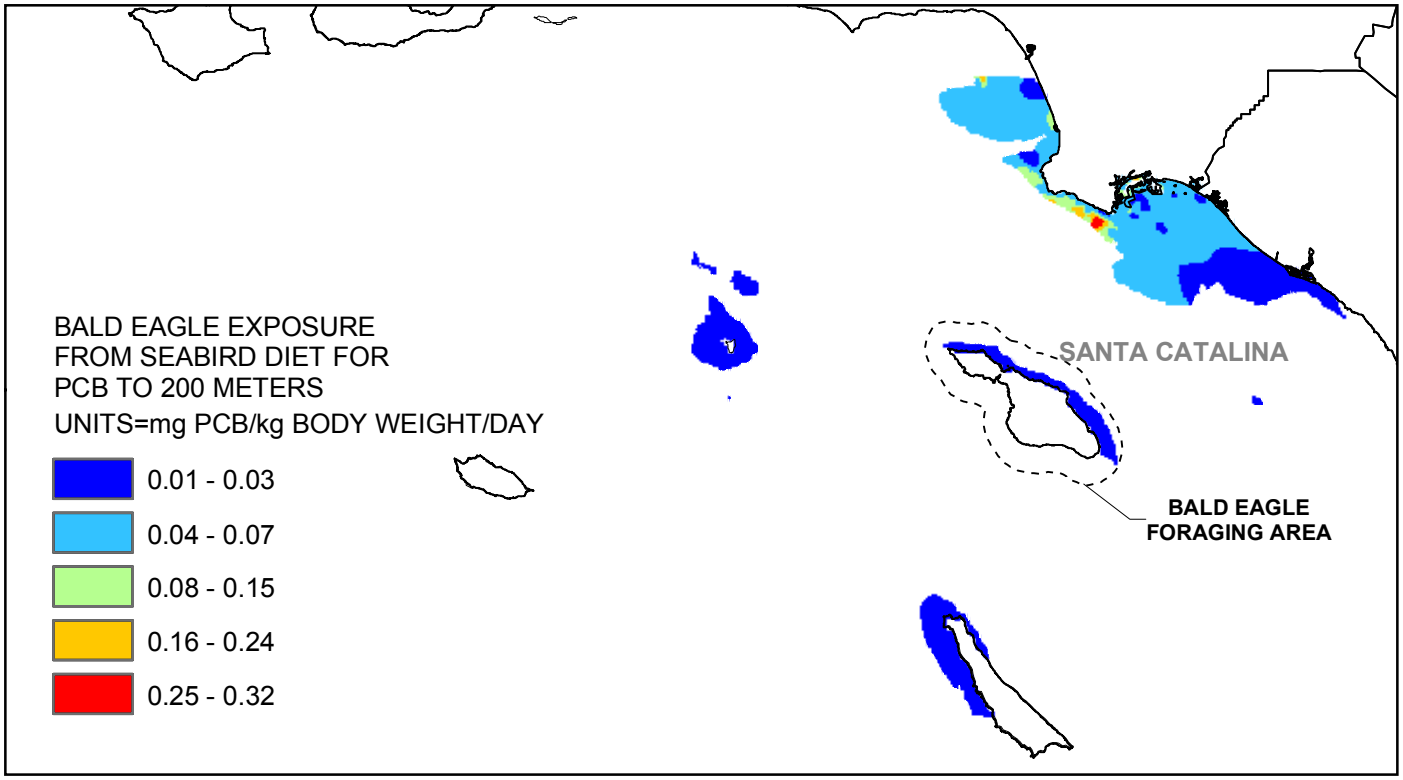

FIGURE 3-10
BALD EAGLE EXPOSURE VALUES
FROM SEABIRD AND SEA LION DIET FOR PCB
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

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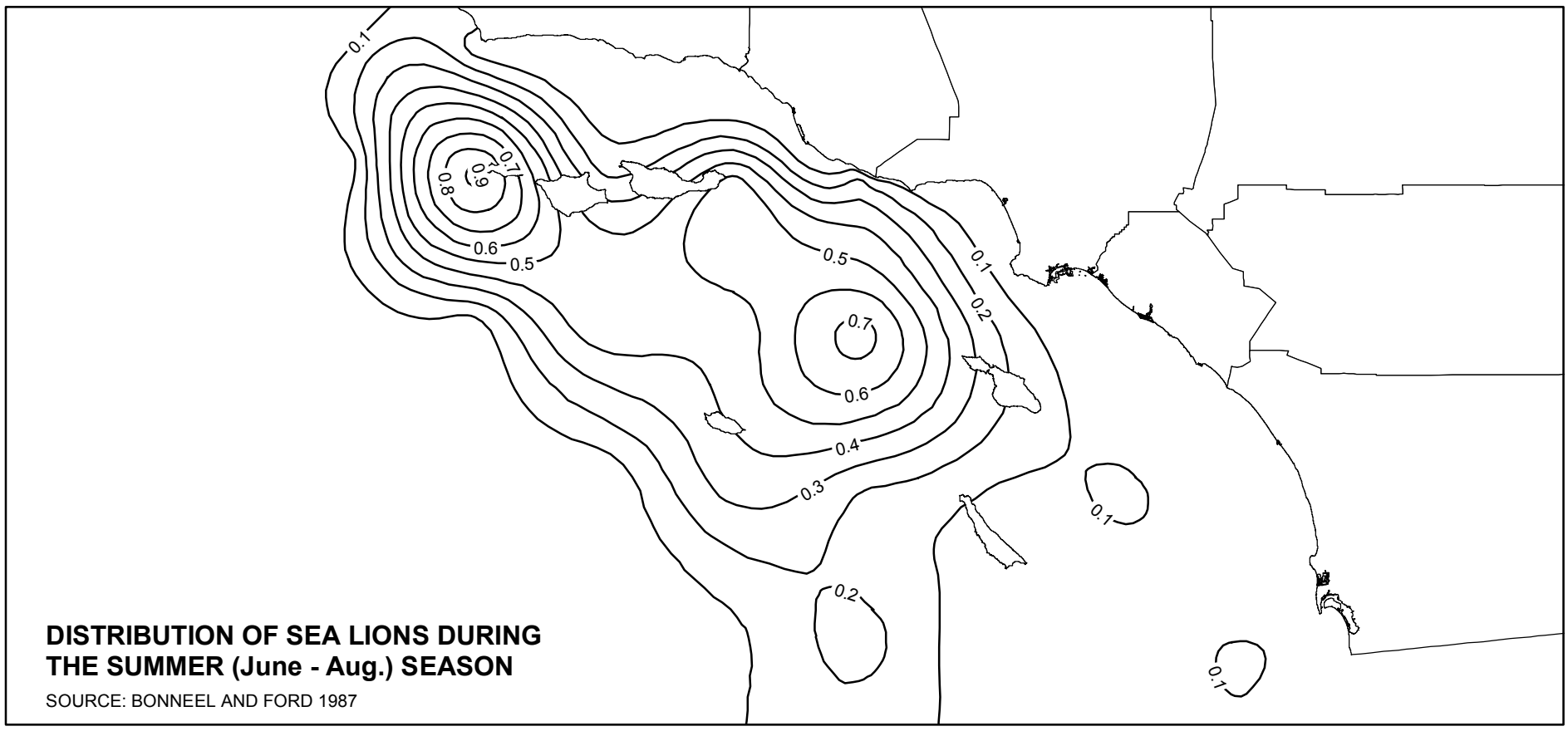
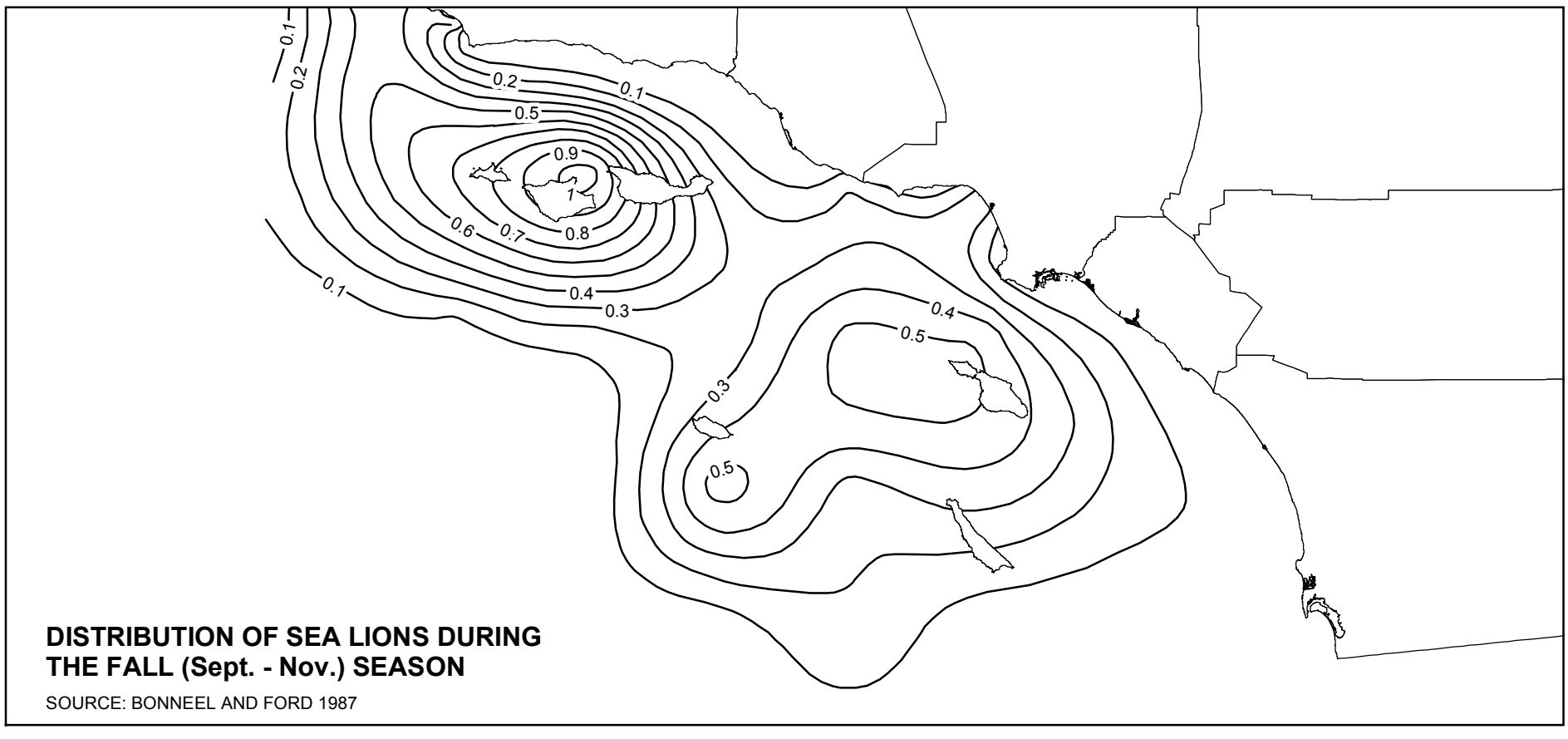


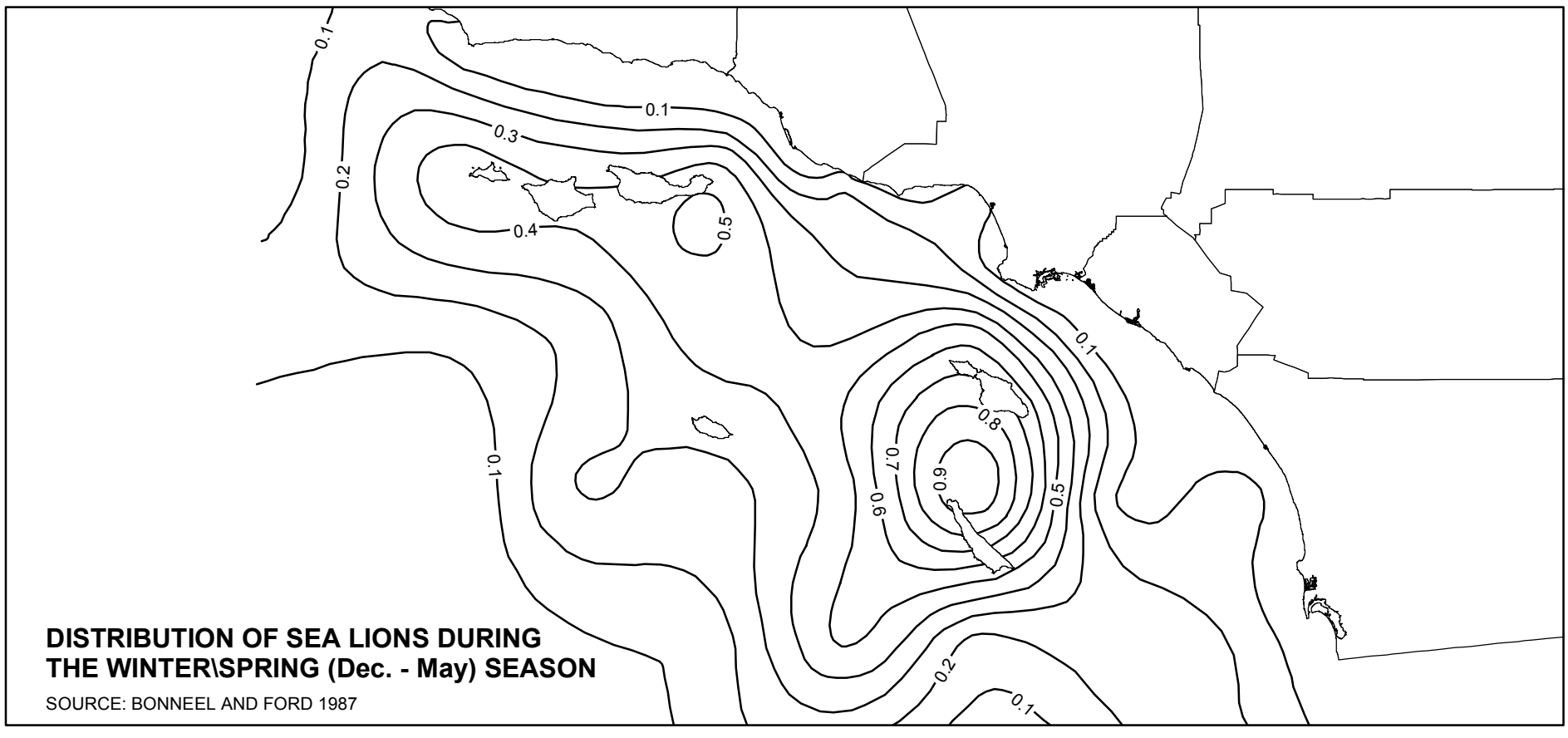
DISTRIBUTION OF SEA LIONS DURING THE SUMMER (June - Aug.) SEASON

SOURCE: BONNEEL AND FORD 1987



DISTRIBUTION OF SEA LIONS DURING THE FALL (Sept. - Nov.) SEASON

SOURCE: BONNEEL AND FORD 1987



DISTRIBUTION OF SEA LIONS DURING THE WINTER\SPRING (Dec. - May) SEASON

SOURCE: BONNEEL AND FORD 1987

N

0 35 Miles

FIGURE 3-11
SEASONAL DISTRIBUTION OF SEA LIONS

PALOS VERDES ECOLOGICAL RISK ASSESSMENT

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HydroQual, Inc (1997) and Connolly and Glaser (2002). Details of this development are described in Appendix C.

Although nursing sea lion pups do not leave the rookery, their oral exposure is related to the foraging areas of the mother. Therefore, estimates of exposure for nursing pups (Figures 3-14 and 3-15) do not represent actual foraging locations for pups, but rather depict the foraging areas of the adult female sea lion.

3.1.4.2 Internal Exposures

Internal exposures consist of concentrations of COPECs in tissues of receptor species. These concentrations may be measured directly from field-collected birds and/or mammals or they may be modeled using site-specific or literature-derived information. They can then be compared to available literature information about concentrations of chemicals in specific tissues that are associated with adverse effects. This provides another measure of the potential nature and magnitude of effects birds and mammals may experience in the PVS study area. Tissue concentrations that have been measured from field-collected species are discussed below, followed by a description of the modeling process for estimating tissue concentrations.

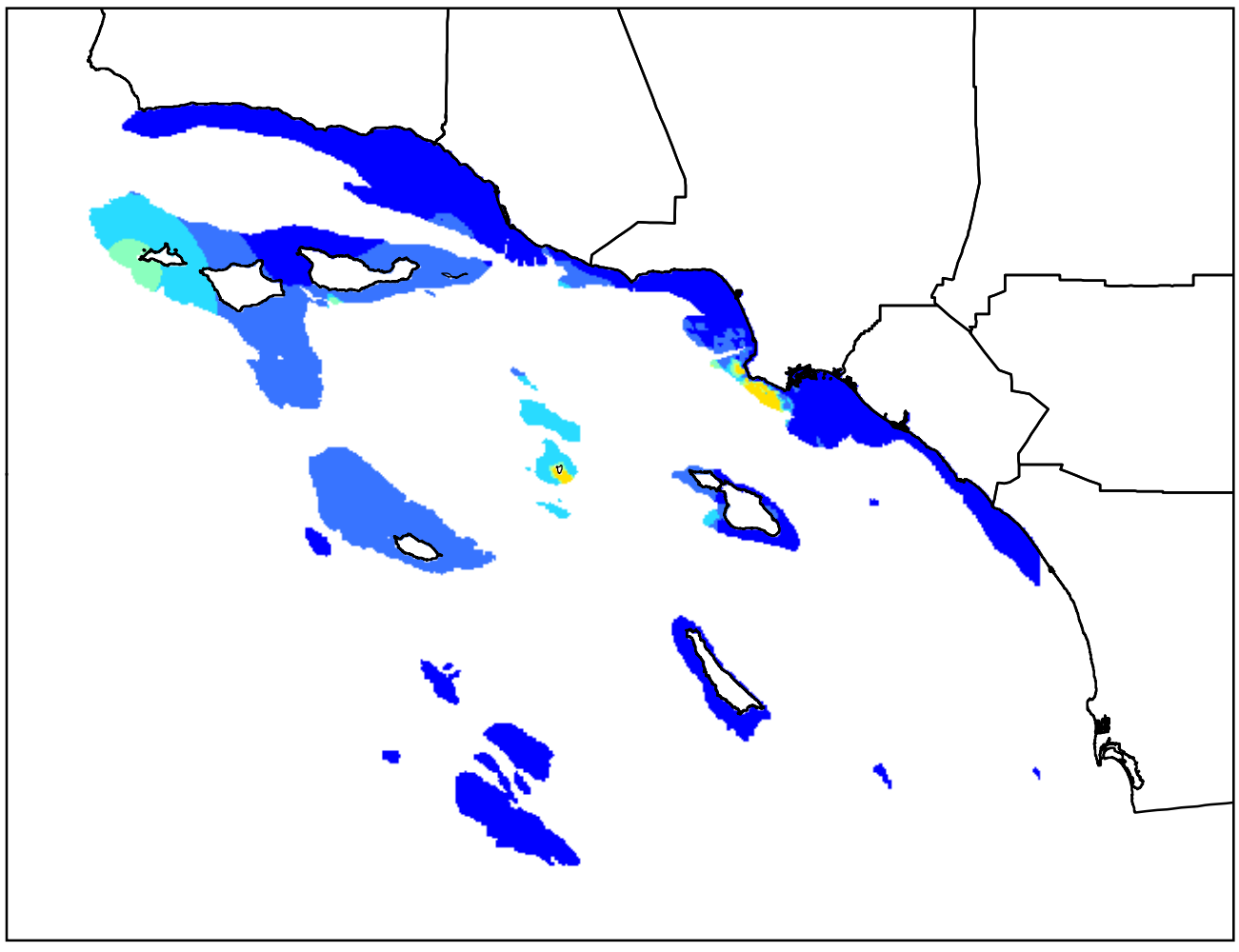
Measured Internal Exposures

Measured internal exposures for birds and mammals were compiled from both published and unpublished sources, with the studies completed during the NRDA process being the primary source. Data collected from 1985 to 2000 were included to represent current conditions within the SCB. Target organ concentration data were available for eggs of brown pelicans, double-crested cormorants, bald eagles, and peregrine falcons. Limited data were also available for blood, whole-body, brain, liver, muscle, serum, and yolk of bald eagles and blood, wholebody, brain, liver, and serum of peregrine falcons. Summary statistics for COPEC concentrations in organs from birds in the SCB are presented in Table 2-10.

Although blubber samples were collected and evaluated in 1991 as part of the NRDA, the values reported for tDDT and tPCB are not included in the Ecological Risk Assessment (ERA) at this time. These data are currently being re-evaluated by NOAA because certain quality control measures were not met (DeLong 2002). Despite the lack of exact values, blubber concentrations in sea lions are evaluated in the risk characterization using data collected in the early-1970s (DeLong et al. 1973; Gilmartin 1976). Current concentrations in blubber are one order of magnitude lower than those recorded in 1970 and 1972 (DeLong 2002). Three additional studies (Kajiwara et al. 2001, Le Boeuf et al. 2002, and Le Boeuf 2002) present contaminant concentrations in sea lions found dead along the California coast. These data represent various age and sex classes from southern, central, and northern California. One dataset has only animals collected in central and northern California (Kajiwara et al. 2001), one is all male or unknown sex animals (Le Boeuf 2002), and the last (Le Boeuf et al. 2003) has overlap of samples with Le Boeuf (2002) and only one adult female from southern California. Therefore, these data were not included in the internal risk estimation for female sea lions.

Modeled Internal Exposures

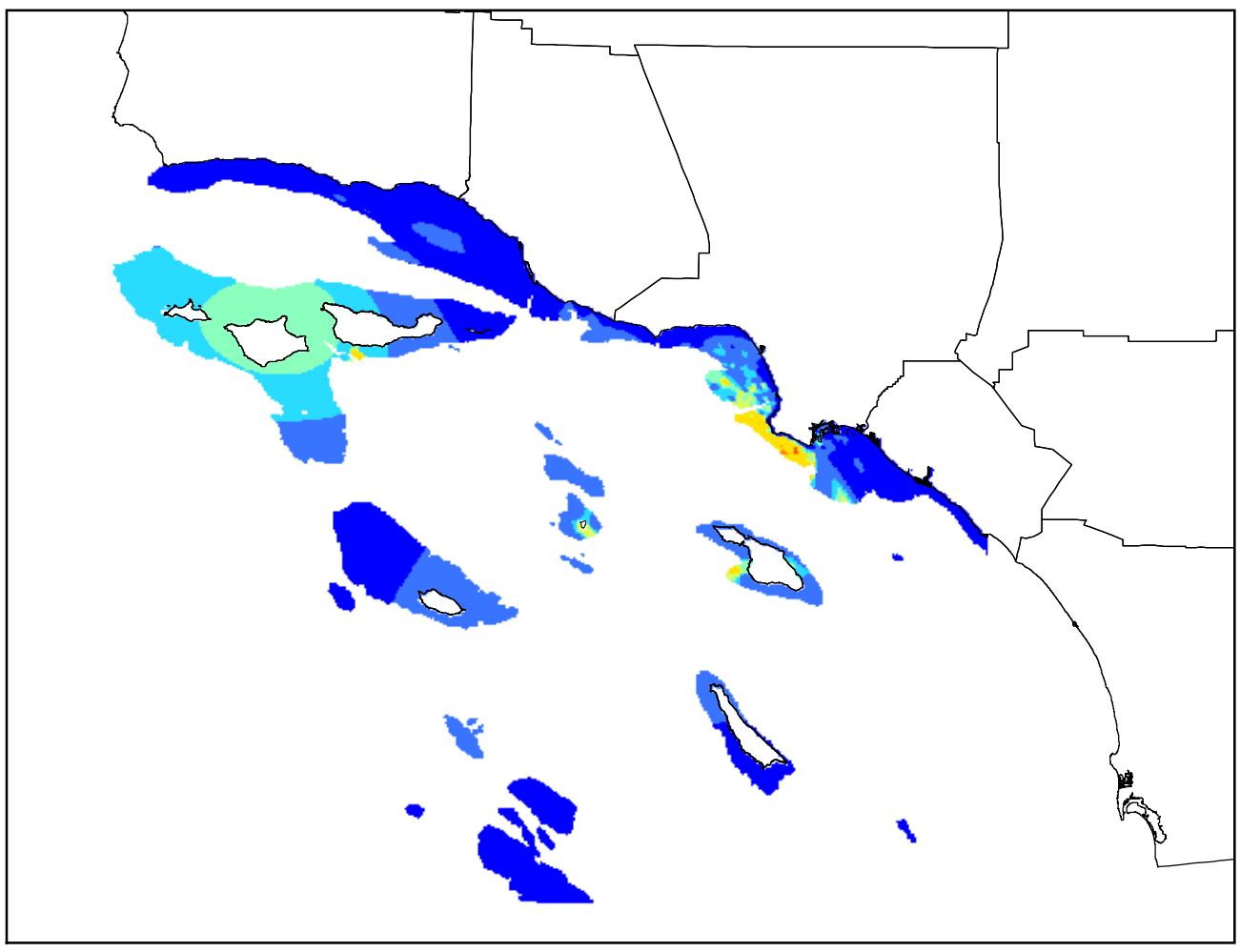
Models for estimation of COPEC concentrations in double-crested cormorant, bald eagle, and peregrine falcon eggs and California sea lion blubber from dietary exposure have been developed by HydroQual, Inc. (1997) and published in Glaser and Connolly (2002). Exposure was evaluated spatially, based on foraging ranges of the receptors. Only the



DDT-SUMMER (June-Aug.)

- 0.0001 - 0.0022
- 0.0023 - 0.0044
- 0.0045 - 0.0065
- 0.0066 - 0.0087
- 0.0088 - 0.0109
- 0.0110 - 0.0646
- 0.0647 - 0.1182
- 0.1183 - 0.1718

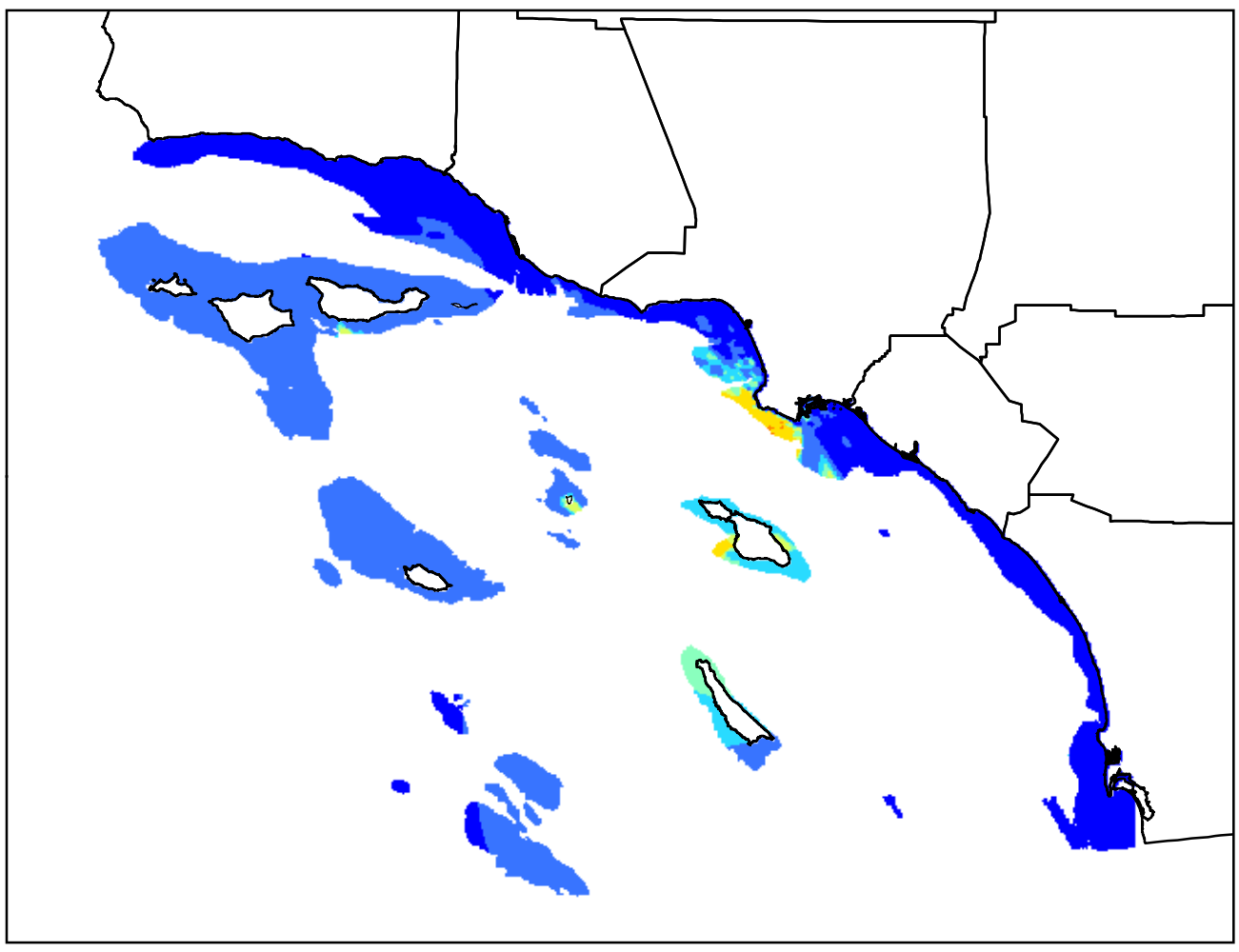
UNITS=mg DDT/kg BODY WEIGHT/DAY



DDT- FALL (Sept.-Nov.)

- 0.0003 - 0.0020
- 0.0021 - 0.0044
- 0.0045 - 0.0065
- 0.0066 - 0.0087
- 0.0088 - 0.0109
- 0.0110 - 0.0646
- 0.0647 - 0.1182
- 0.1183 - 0.1718

UNITS=mg DDT/kg BODY WEIGHT/DAY



DDT - WINTER\SPRING (Dec. - May)

- 0.0001 - 0.0022
- 0.0023 - 0.0044
- 0.0045 - 0.0065
- 0.0066 - 0.0087
- 0.0088 - 0.0109
- 0.011 - 0.0646
- 0.0647 - 0.1182
- 0.1183 - 0.1718

UNITS=mg DDT/kg BODY WEIGHT/DAY

N

0 35 Miles

FIGURE 3-12
ADULT FEMALE SEA LION EXPOSURE VALUES (DDT) TO 200 METERS

PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

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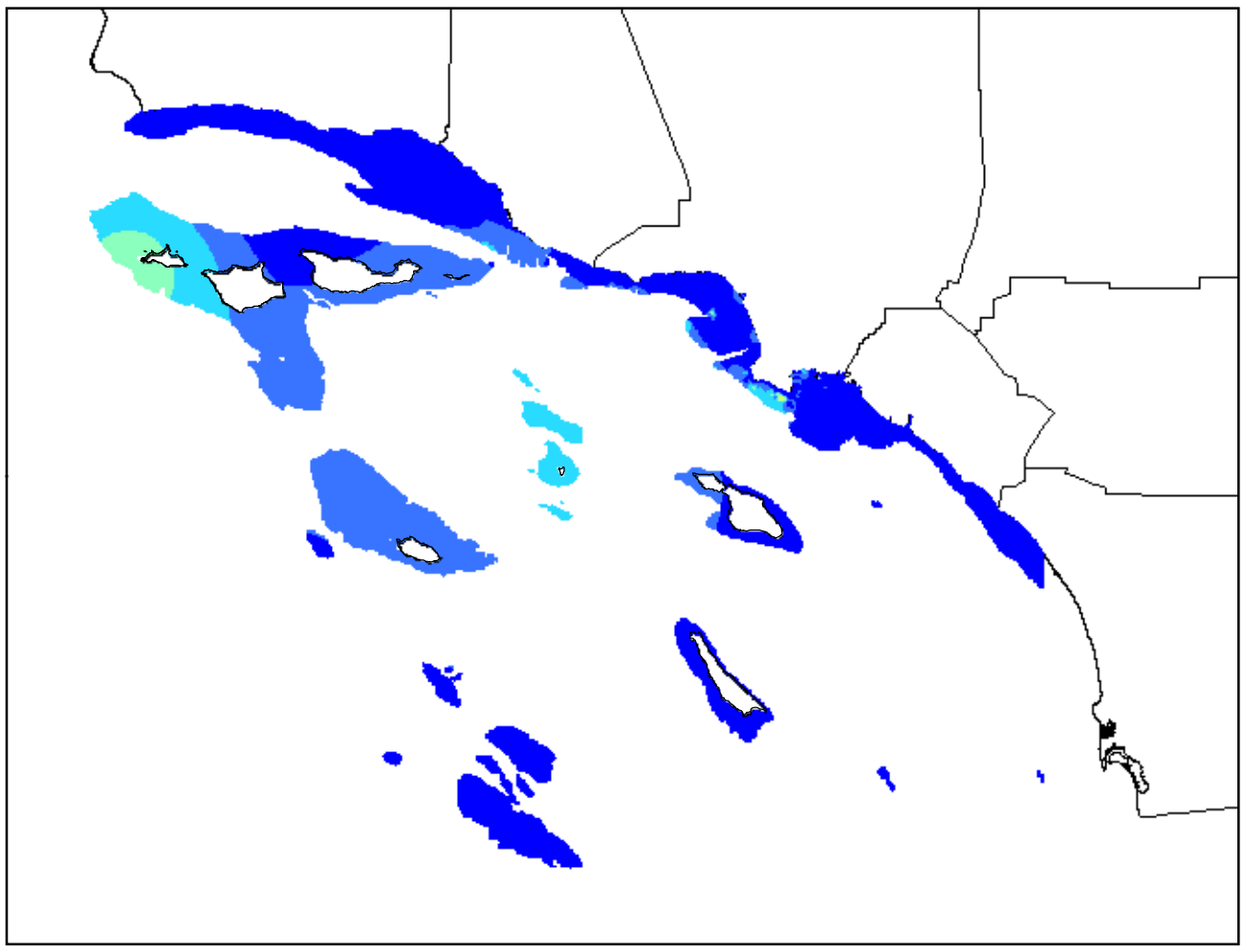
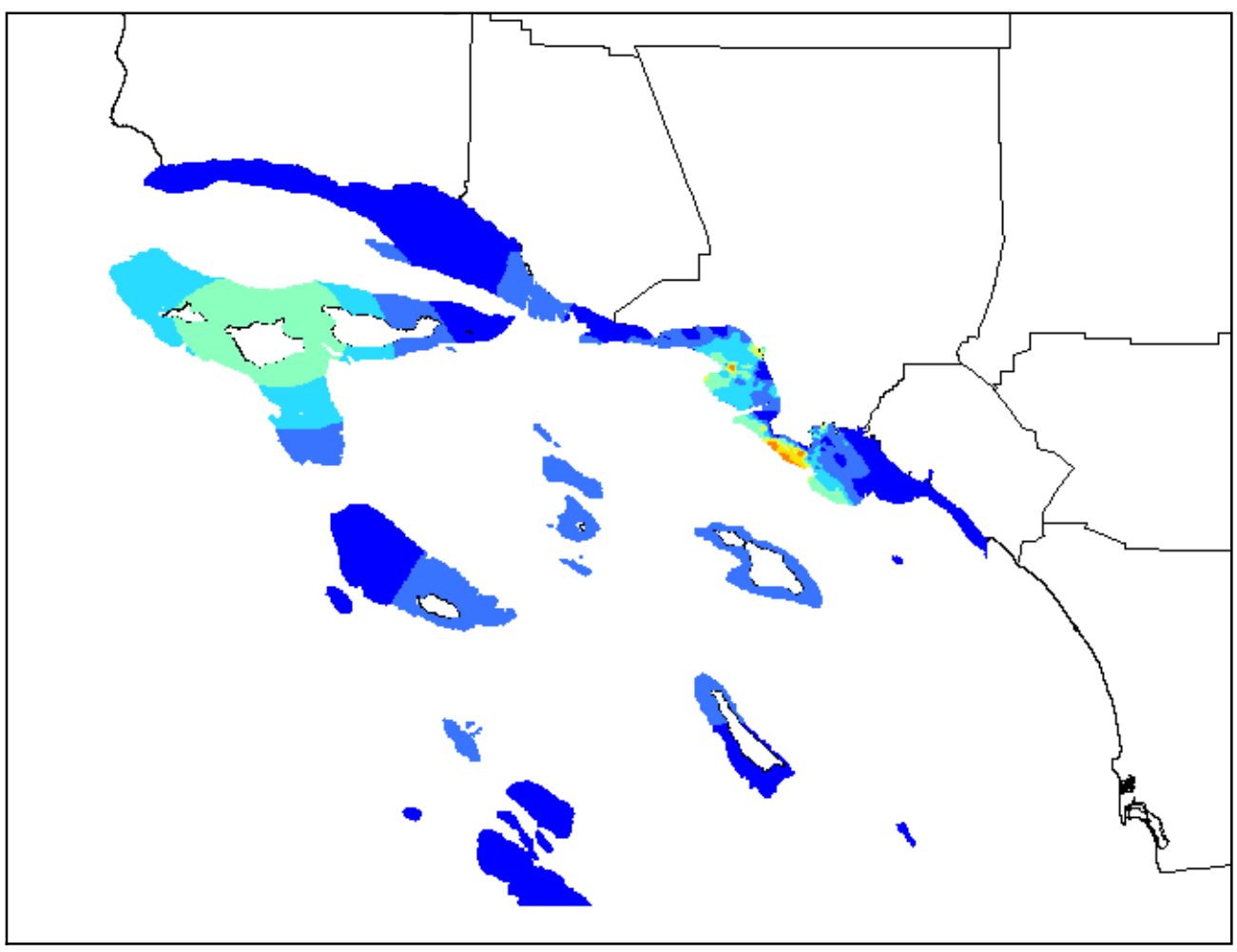


PCB - SUMMER (June - Aug.)

- 0.0001 - 0.0011
- 0.0012 - 0.0021
- 0.0022 - 0.0032
- 0.0033 - 0.0043
- 0.0044 - 0.006
- 0.0061 - 0.0078
- 0.0079 - 0.0148
- 0.0149 - 0.113

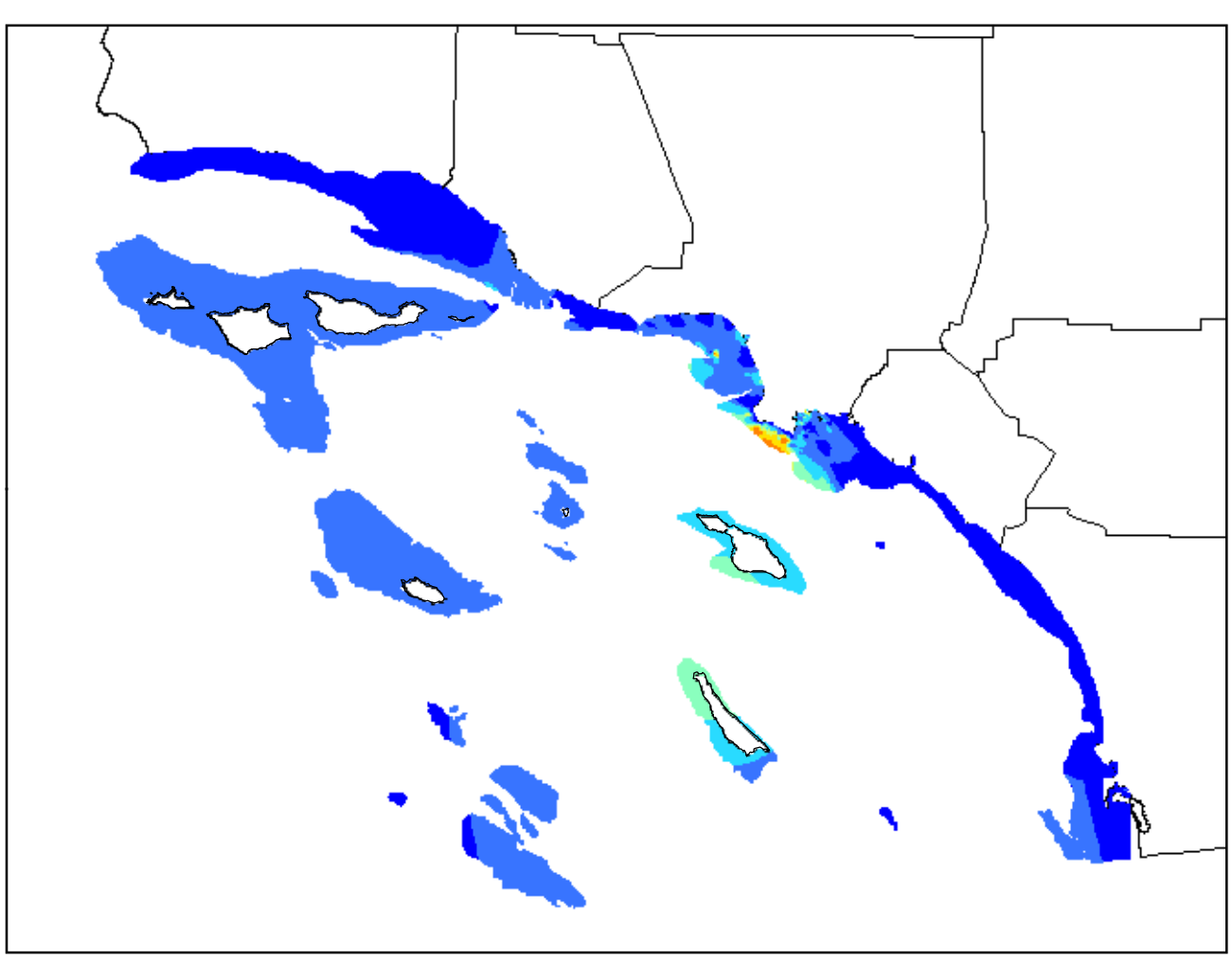
UNITS=mg PCB/kg BODY WEIGHT/DAY



PCB - FALL (Sept. - Nov.)

- 0.0001 - 0.0011
- 0.0012 - 0.0021
- 0.0022 - 0.0032
- 0.0033 - 0.0043
- 0.0044 - 0.0060
- 0.0061 - 0.0078
- 0.0079 - 0.0148
- 0.0149 - 0.113

UNITS=mg PCB/kg BODY WEIGHT/DAY



PCB - WINTER\SPRING (Dec. - May)

- 0.0001 - 0.0011
- 0.0012 - 0.0021
- 0.0022 - 0.0032
- 0.0033 - 0.0043
- 0.0044 - 0.006
- 0.0061 - 0.0078
- 0.0079 - 0.0148
- 0.0149 - 0.113

UNITS=mg PCB/kg BODY WEIGHT/DAY

0 35 Miles

N

FIGURE 3-13
ADULT FEMALE SEA LION EXPOSURE VALUES (PCB) TO 200 METERS

PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

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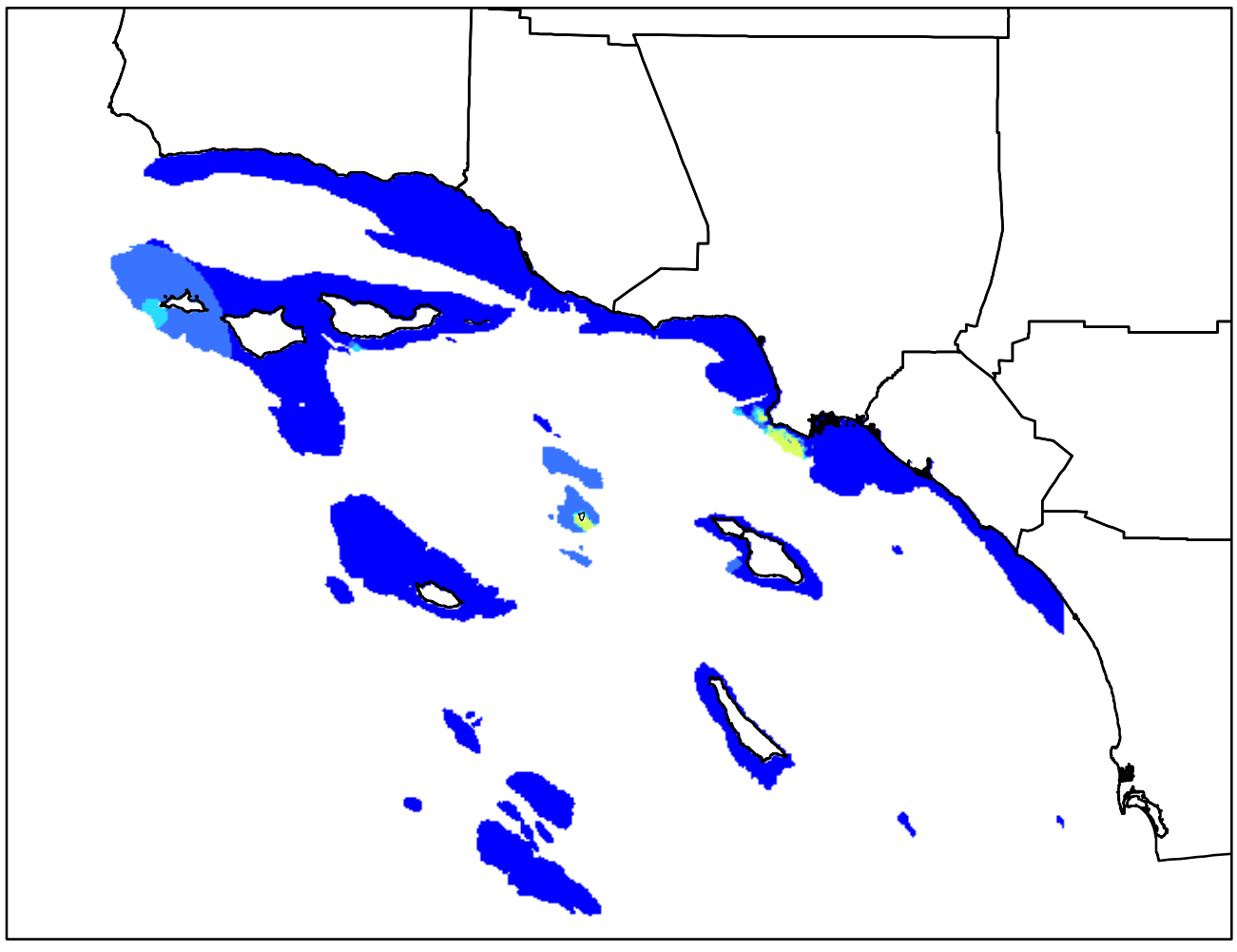
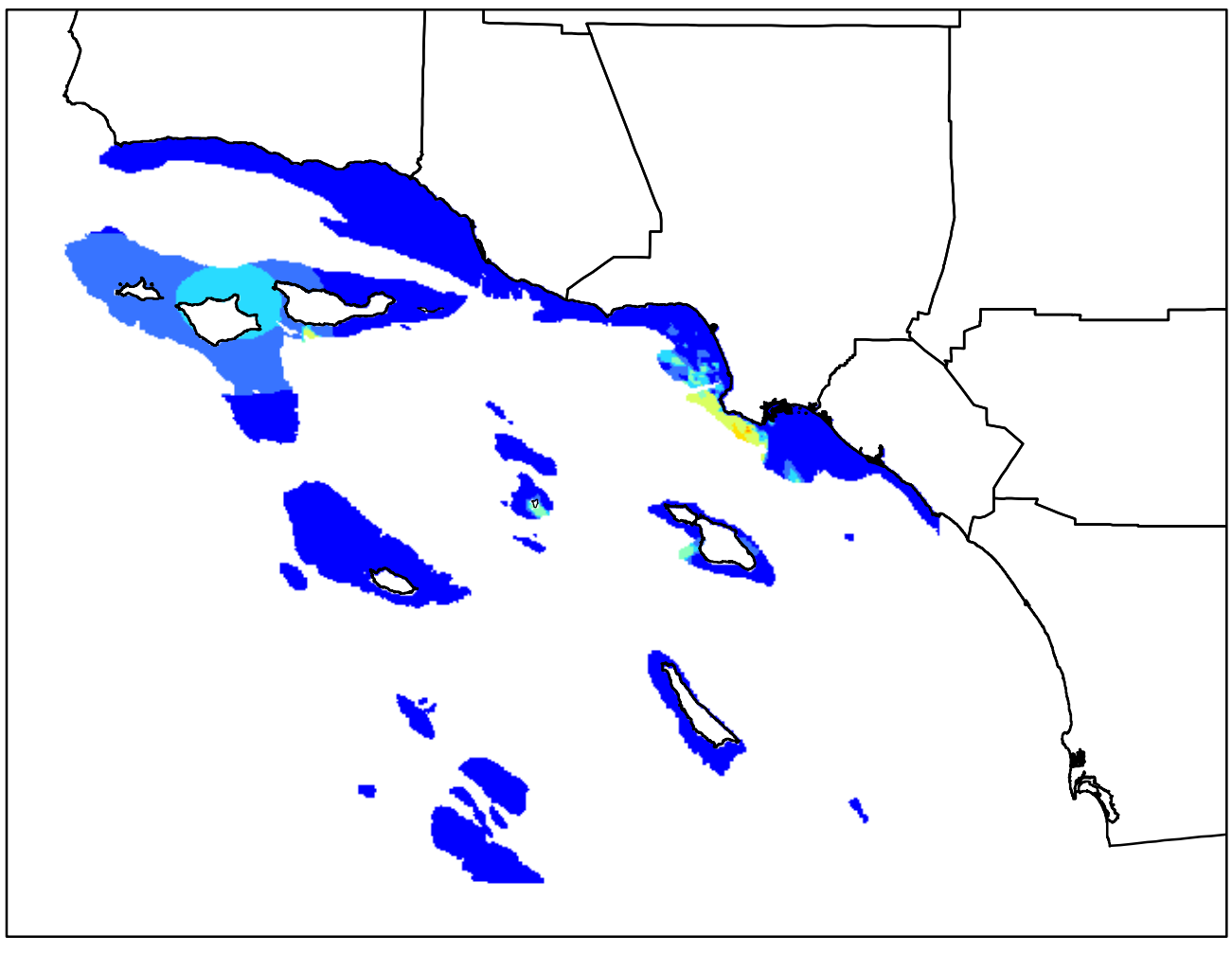


DDT - SUMMER (June - Aug.)

- 0.001 - 0.023
- 0.024 - 0.035
- 0.036 - 0.046
- 0.047 - 0.058
- 0.059 - 0.272
- 0.273 - 0.486
- 0.487 - 0.700
- 0.701 - 0.915

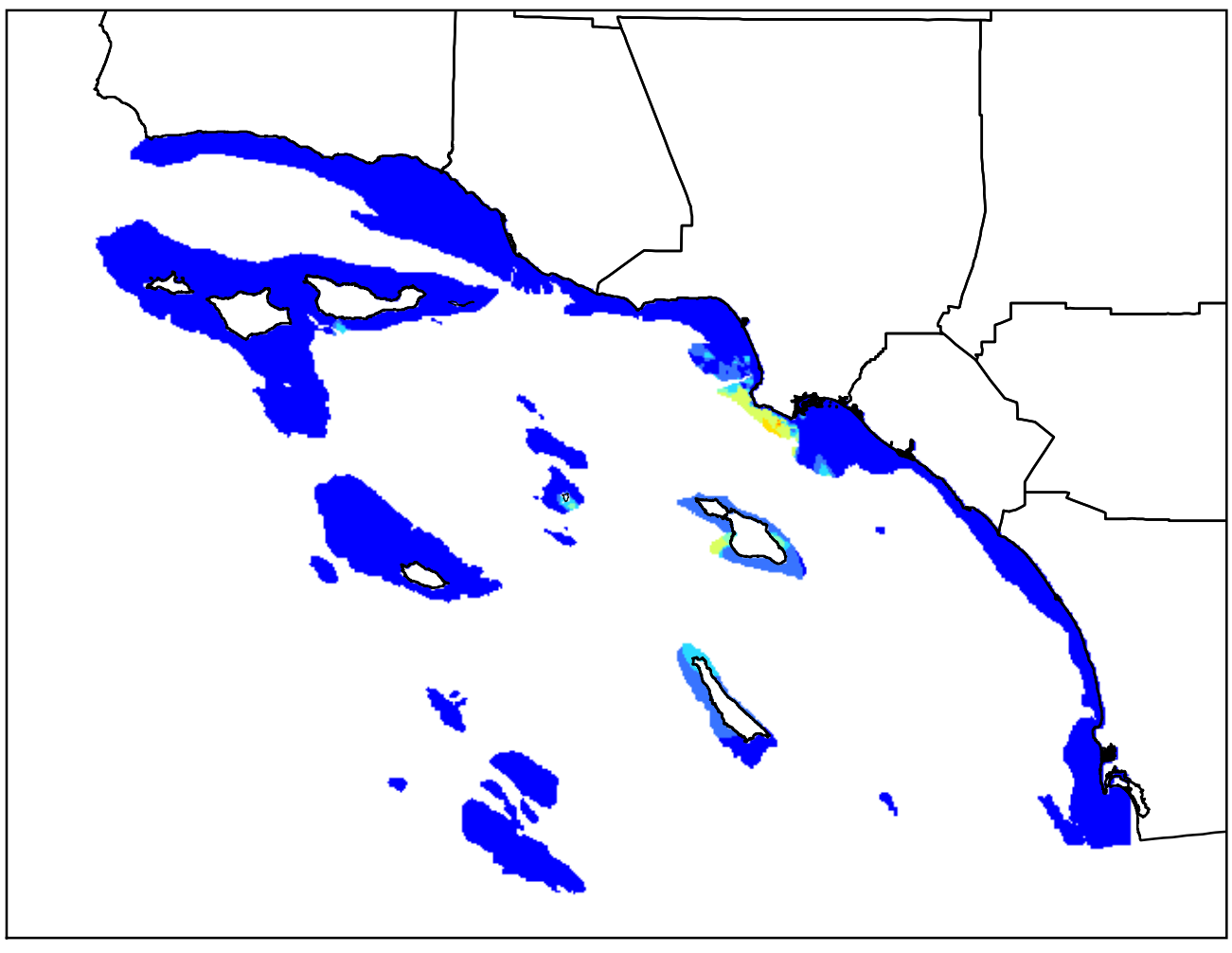
UNITS=mg DDT/kg BODY WEIGHT/DAY



DDT - FALL (Sept. - Nov.)

- 0.001 - 0.023
- 0.024 - 0.035
- 0.036 - 0.046
- 0.047 - 0.058
- 0.059 - 0.272
- 0.273 - 0.486
- 0.487 - 0.700
- 0.701 - 0.915

UNITS=mg DDT/kg BODY WEIGHT/DAY



DDT - WINTER\SPRING (Dec. - May)

- 0.001 - 0.023
- 0.024 - 0.035
- 0.036 - 0.046
- 0.047 - 0.058
- 0.059 - 0.272
- 0.273 - 0.486
- 0.487 - 0.700
- 0.701 - 0.915

UNITS=mg DDT/kg BODY WEIGHT/DAY

0 35 Miles

N

FIGURE 3-14
NURSING CALIFORNIA SEA LION PUP EXPOSURE VALUES (DDT) TO 200 METERS

PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

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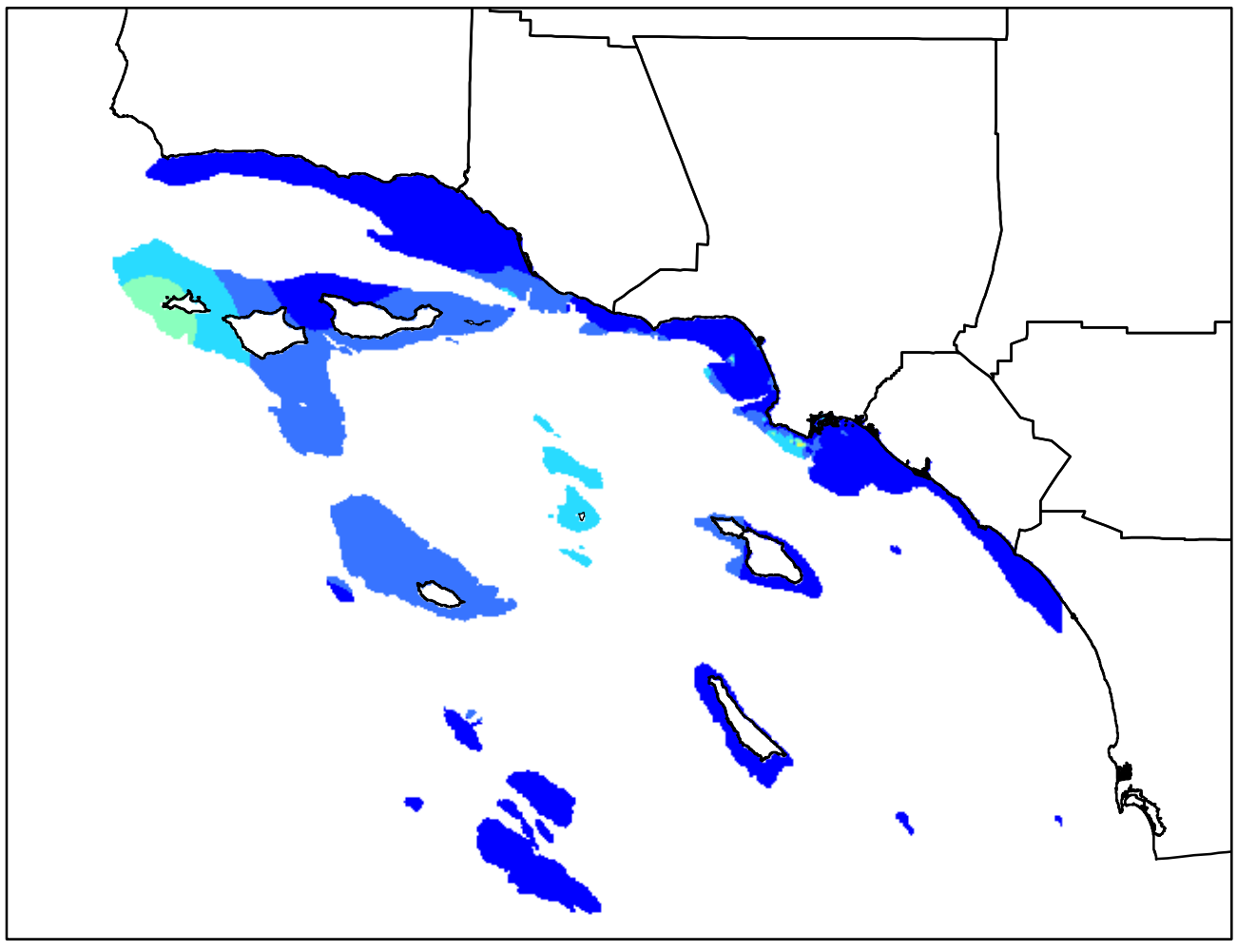
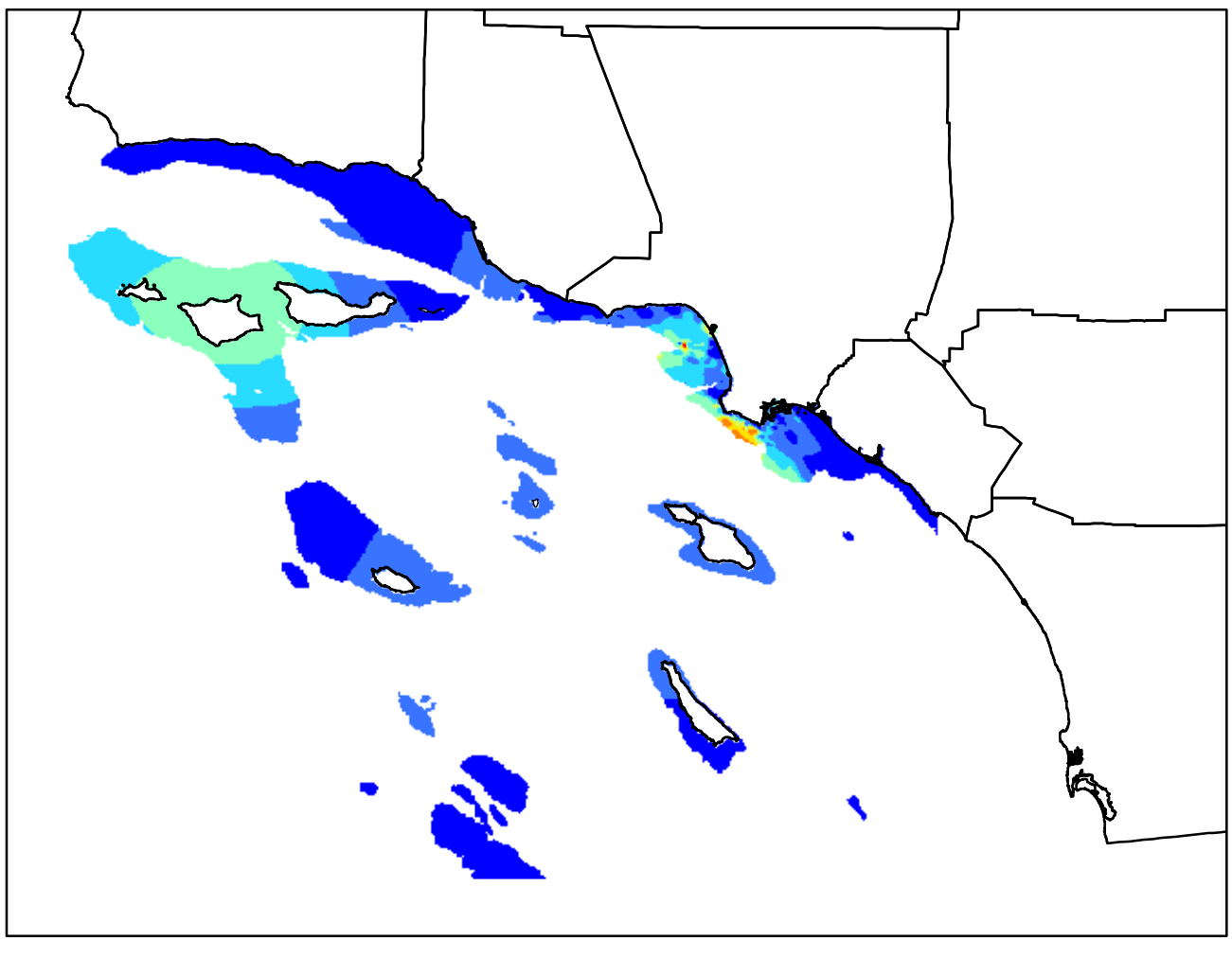


PCB - SUMMER (June - Aug.)

- 0.001 - 0.007
- 0.008 - 0.014
- 0.015 - 0.021
- 0.022 - 0.028
- 0.029 - 0.040
- 0.041 - 0.052
- 0.053 - 0.074
- 0.075 - 0.098

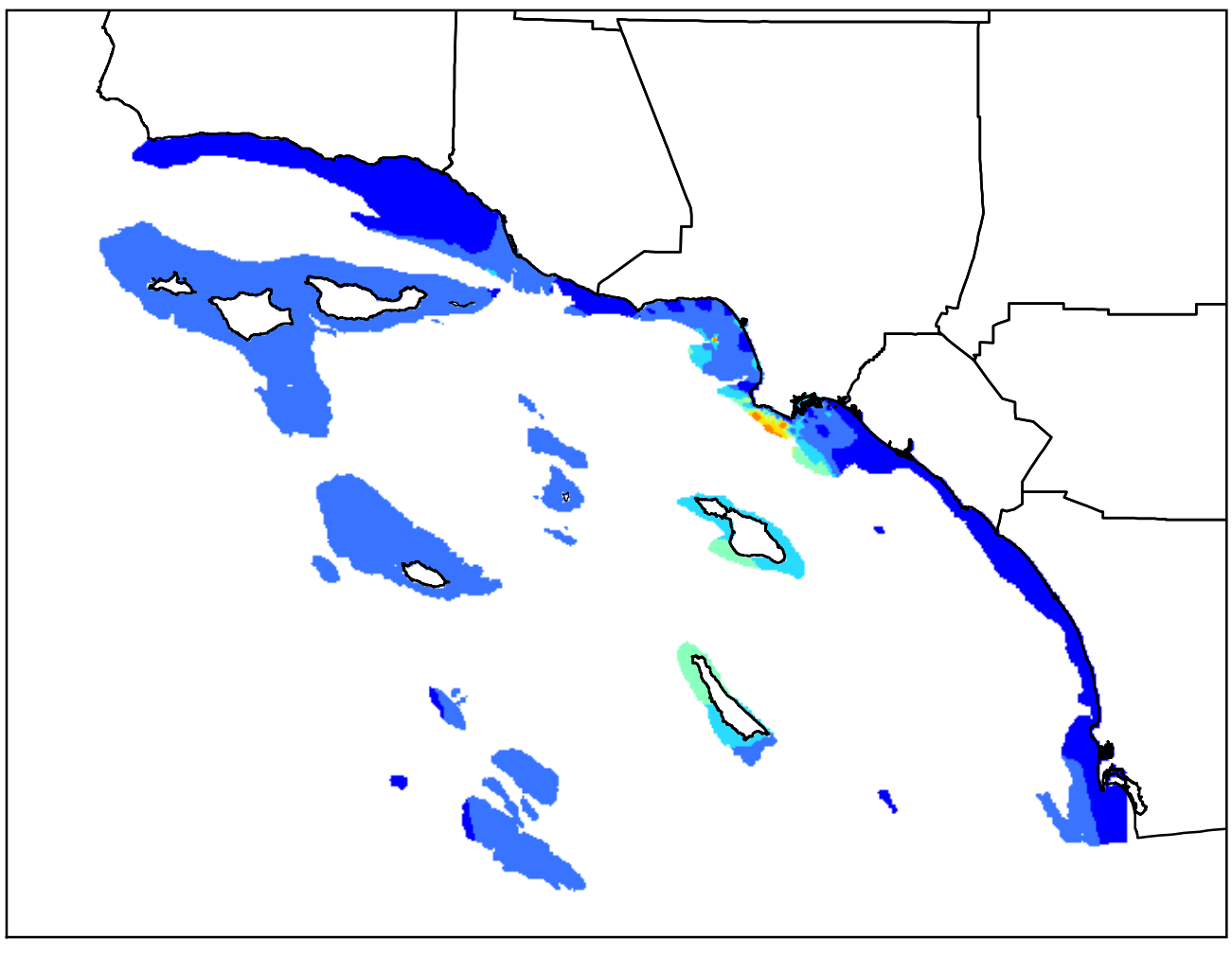
UNITS=mg PCB/kg BODY WEIGHT/DAY



PCB - FALL (Sept. - Nov.)

- 0.001 - 0.007
- 0.008 - 0.014
- 0.015 - 0.021
- 0.022 - 0.028
- 0.029 - 0.040
- 0.041 - 0.052
- 0.053 - 0.074
- 0.075 - 0.098

UNITS=mg PCB/kg BODY WEIGHT/DAY



PCB - WINTER\SPRING (Dec. - May)

- 0.001 - 0.007
- 0.008 - 0.014
- 0.015 - 0.021
- 0.022 - 0.028
- 0.029 - 0.040
- 0.041 - 0.052
- 0.053 - 0.074
- 0.075 - 0.098

UNITS=mg PCB/kg BODY WEIGHT/DAY

N

0 35 Miles

FIGURE 3-15
NURSING CALIFORNIA SEA LION PUP EXPOSURE VALUES (PCB) TO 200 METERS

PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

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measured values were used in the risk characterization for this ERA; however, these estimation models can be used in combination with the exposure models developed in the ERA to evaluate future capping scenarios at PVS.

3.2 Ecological Effects Characterization

3.2.1 Benthic Invertebrates

Epibenthic and infaunal invertebrates that are in close association with sediments may exhibit toxic effects associated with sediment and porewater contamination. Effects data for benthic invertebrates include literature-derived and site-specific toxicity data for external exposure to sediment, literature-derived toxicity data for external exposure to porewater, site-specific ambient media toxicity testing, site-specific field studies, and site-specific observations of ecological effects.

3.2.1.1 Literature-Derived and Site-Specific Toxicity Data

Sediment

Invertebrate species are generally more sensitive than fish species to effects associated with exposure to tDDT in the water column (USEPA 1980a). For example, in laboratory exposures to tDDT using 96-hour static tests, LC_{50}s ranged between 0.12 and 4.0 µg/L for crustaceans, and between 1.0 and 7.0 µg/L for insects (Verschueren 1983). USEPA (1980a) reports on studies with the eastern oyster (*Crassostrea virginica*) and Korean shrimp *(Palaemon macrodactylus*) that indicate water-borne tDDT to be up to 10 times more toxic than either of the two major metabolites DDE and DDD.

Sediments contaminated with pesticides, including tDDT, have been shown to affect benthic communities at low concentrations. Results of laboratory and field investigations suggest that chronic effects generally occur at tDDT concentrations in sediment exceeding 2 µg/kg (Long et al. 1995). Equilibrium partitioning methods predict that chronic effects occur at tDDT concentrations in sediment of 0.6 to 1.7 µg/kg (Pavlou et al. 1987).

Statistically significant responses to DDE were noted in tests of sediments from Puget Sound in an amphipod bioassay and in the evaluation of benthic community composition where DDE concentrations were 15 and 9 µg/kg, respectively (Beller et al. 1986). Statistically significant effects in the bivalve larval development and amphipod bioassays were observed in tests of sediments from San Francisco Bay with 2.2 µg/kg DDE (Long and Morgan 1990).

Swartz et al. (1994) examined effects thresholds for sediment toxicity and abundance of amphipods associated with tDDT-contaminated sediments at the United Heckathorn Superfund site. A threshold for acute toxicity of 300,000 µg/kg organic carbon (OC) was established, and decreased amphipod abundance was observed at 100,000 µg/kg OC.

Logistic regression models have been applied to a large North American database. Resulting T20 values—estimates yielding a toxic sediment response for invertebrates 20 percent of the time—were 35 µg/kg DW for tPCB and 3.1 µg/kg DW DDE (Field et al. 2002).

Responses to tPCB were observed in tests of marine sediments where concentrations ranged from 36.6 to 10,800 µg/kg (Long and Morgan 1990). In tests of Puget Sound sediment, statistically significant effects to amphipods, oyster larvae, and Microtox assays were

observed between 130 and 2,500 µg/kg tPCB. Statistically significant effects in bivalve larval development and amphipod bioassays were observed in tests of sediments from San Francisco Bay with 54 and 260 µg/kg tPCB, respectively (Long and Morgan 1990). Toxic responses (>80 percent amphipod mortality and >44 percent abnormal bivalve larval development) were noted in tests of Commencement Bay sediments with 38 and 368 µg/kg tPCB, respectively (Tetra Tech 1985). Negative growth was noted in nematode bioassays using Hudson-Raritan Estuary sediments with 638 µg/kg tPCB (Tietjen and Lee 1984).

Measurement endpoints for sediments typically include comparison of sediment contaminant values to sediment quality guidelines, such as the effects range values developed by NOAA (Long et al. 1995) or site- or region-specific effect threshold concentrations (e.g., State of Florida threshold effects concentrations described in MacDonald 1994). For the PVS, MacDonald (1997) has developed sediment effect concentrations (SECs) for tDDT and tPCB based upon effects data specific to the SCB (subsection 3.2.1.2, below). Other pertinent information has been provided by Swartz et al. (1994) regarding thresholds of tDDT that are expected to cause biological effects. These thresholds are based upon sediment toxicity to amphipods and benthic community studies conducted at both the United Heckathorn Superfund site and the PVS. In addition, the Field et al. (2002) results for amphipod toxicity were based on an extensive national database. Sediment effects thresholds for tDDT and tPCB determined in these studies are shown in Table 3-5.

TABLE 3-5
Bulk Sediment Toxicity Threshold Concentrations—Effects on Marine Invertebrates
Palos Verdes Shelf Ecological Risk Assessment

	Toxicity Threshold (µg/kg)	
Source	**tDDT**	**Total PCB**
NOAA ER-L (Long et al. 1995)	1.58	22.7
NOAA ER-M (Long et al. 1995)	46.1	180
Florida TEL (McDonald 1994)	3.89	21.6
Florida PEL (McDonald 1994)	51.7	189
T20 (Field et al. 2002)	3.1	35
Survival and reproduction SECs (MacDonald 1997)	2,000 @ 1% OC	577 @ 1 % OC
Acute[a] (Swartz et al. 1994)	3,000 @ 1 % OC	—
Chronic[b] (Swartz et al. 1994)	2,000 @ 1 % OC	—
PVS[c] (Swartz et al. 1994)	1,000 @ 1 % OC	—

Notes:
[a] Estimate of threshold of acute sediment toxicity to United Heckathorn site sediments
[b] Estimate of threshold for reduced amphipod abundance in United Heckathorn sediments
[c] Estimate of threshold for reduced amphipod abundance in PVS sediments (Swartz et al. 1994)
ER-L effects range—low
ER-M effects range—median
OC organic carbon
PEL probable effects level
TEL threshold effects level
SEC sediment effect concentration
T20 Threshold for 20% probability of toxicity
— denotes no data

It is clear from examination of these thresholds that the SEC and thresholds estimated by Swartz et al. (1994) are much higher than are the NOAA and Florida estimates. Long and MacDonald (1998) note that the variability of the tDDT data is very high and that reliability of the NOAA and Florida tDDT guidelines is lower than that for many other chemicals. Therefore, the SEC thresholds (MacDonald 1997) were selected for this analysis over those of Field et al. (2002), Long et al. (1995) or MacDonald (1994) because of their site-specific nature and because it has been noted that effects range values for tPCB and tDDT are not well correlated with observed biological effects. In contrast, the tDDT SEC benchmark was predictive of toxicity in 100 percent of the cases examined. Further, the California State Water Resources Control Board (Anderson et al. 1998) has recognized the unreliability of the Florida and NOAA benchmarks and established an effects screening concentration for tDDT of 1,000 µg/kg @ 1 percent OC, based upon Swartz et al.'s (1994) assessment of United Heckathorn and PVS sediments.

As noted previously, the 4,4'-DDE isomer accounts for approximately 70 percent of the tDDT mixture in PVS sediments. MacDonald's (1997) DDE benchmark was not utilized because his analysis indicated that insufficient data were available to make comparisons to an independent data set on toxicity. Swartz et al. (1994) noted the difference between United Heckathorn site sediments and those on the PVS for which they established an effects threshold of 1,000 µg/kg at 1 percent OC. Reductions in amphipod abundance were observed corresponding to the range of 1,000-2,000 µg/kg tDDT. United Heckathorn site sediments contain a significantly higher proportion of tDDT and a lower proportion of 2,4'-DDE and 4,4'-DDE isomers (tDDE) than do the PVS sediments. The higher SEC benchmark appears to account for reduced benthic toxicity of the relatively depleted tDDT and enriched tDDE components of PVS sediments. MacDonald's (1997) SEC values for the PVS are 2,000 µg/kg tDDT and 577 µg/kg tPCB, respectively, both at 1 percent TOC.

In this ERA, measured TOC values were used to develop TOC contours for tDDT and tPCB within the PVS study area (Figure 2-6). Therefore, the non-normalized SEC values (7,120 and 835 µg/kg for DDT and PCB, respectively, were used to create a TOC-normalized GIS layer for comparison to the sediment layer.

Porewater

Toxicity values for exposure of benthic invertebrates to porewater were derived from the National Ambient Water Quality Criteria (NAWQC) document (USEPA 2002), which provides acute (criterion maximum concentration [CMC]) and chronic (criterion continuous concentration, [CCC]) benchmarks for the protection of fresh and saltwater aquatic life. The CMC for tDDT is 0.13 µg/L and the CCC is 0.001 µg/L. The CMC is based on toxicity tests with brown shrimp (*Penaeus aztecus*) and is considered to be an appropriate acute benchmark for benthic invertebrates. The CCC is a residue-based value that was established to protect birds (developed from studies on brown pelicans by Anderson et al. 1975). Use of this benchmark would likely overestimate chronic risk to benthic invertebrates; therefore, a chronic benchmark was developed from the CMC by applying an acute-to-chronic factor of 18 as recommended in USEPA (1995). This results in a chronic benchmark for benthic invertebrates of 0.0072 µg/L. For tPCB, there was no CMC available and the CCC was 0.03 µg/L (USEPA 2002). This value was selected as the chronic benchmark for benthic invertebrates.

3.2.1.2 Site-Specific Ambient Media Toxicity Testing

Numerous studies conducted over the last two decades have shown that tDDT and tPCB concentrations in sediments and organisms of the SCB, particularly the PVS, were among the highest reported for any coastal marine ecosystem (Young et al. 1991). Historically, sediment PCB concentrations exceeded 100 µg/kg (DW) off the PVS. Tests conducted by Dexter and Field (1989) with Bight sediment identified acute and chronic threshold responses, which ranged from 100 to 4,000 µg/kg PCB (DW). Tests conducted by Swartz et al. (1988) using the benthic amphipod, *Rhepoxynius abronius*, identified a 10-day acute LC_{50} of 10,800 µg/kg (WW). Median concentrations for all tests ranged between 100 and 1,000 µg/kg (DW). Three toxic samples contained mean tDDT at 6,302 µg/kg DW and four non-toxic samples had 2,4'-DDT and 4,4'-DDT isomers (tDDT) concentrations at 2,842 µg/kg DW.

Bay et al. (1994) conducted studies to document the magnitude of PVS sediment toxicity and bioaccumulation in infaunal and epibenthic organisms. Interstitial water toxicity was measured using a sea urchin fertilization test. The toxicity of bulk sediment was measured using a 10-day amphipod survival test, a 28-day amphipod growth test, and a 35-day sea urchin growth test. The general spatial pattern of toxicity appeared to be influenced by distance from the outfall and depth (greater than 30 m). Amphipod tests conducted on a limited number of stations did not detect toxicity. Reductions in sea urchin growth and fertilization were produced at some stations. A spiked sediment test was conducted, which the authors felt demonstrated that growth effects were not caused by exposures to 4,4'-DDE. The authors also concluded that metals were not likely to be significant causes of sediment toxicity.

Three types of sediment toxicity tests were conducted in previous studies on sediment collected from a series of stations co-located with Palos Verdes stations included in this analysis. The purple sea urchin sperm cell fertilization test and the amphipod *Rhepoxynius abronius* 10-day solid phase test were conducted by Bay et al. (1994) on sediments collected in 1992 and 1993. Additional *Rhepoxynius* tests and the 120-day polychaete *Neanthes arenaceodentata* solid phase chronic test were conducted by EVS (1994) on sediments also collected in 1992 and 1993. These results are consistent with previous observations of no acute toxicity on the PVS presented in Swartz et al. (1985). Similarly, no acute toxicity was observed in 20-day exposures of *Neanthes arenaceodentata* to four sediments in the EVS study.

Indications of chronic toxicity were apparent at two stations tested with *Neanthes* where the number of emergent juveniles was depressed after 120-day exposures. Bay et al. (1994) observed effects on sea urchin fertilization success in five of nine sediments used in this comparison. These data suggest that sediments from LACSD stations 6B, 6C, 7C, 8B, 8C, and 9C pose a risk of chronic toxicity.

Bay et al. (1994) suggest that, based upon DDE-spiked water tests, it is unlikely that toxicity is associated with DDE contamination, although they caution that this interpretation cannot be extrapolated to other more toxic forms that were not tested, e.g., DDT. These authors further assert that observed toxicity may be attributable to hydrogen sulfide present in the samples; again, this conclusion is based upon sulfide-spiked water tests. This inference is made in spite of the fact that accurate hydrogen sulfide measurements were not made on the samples tested but on those collected from the same stations four months later. It also is

not clear whether the test exposures and sample handling were conducted in an oxygen-free environment to preserve *in situ* sulfide conditions.

Regardless, Bay et al. (1994) point out that chemical contaminants other than ammonia and sulfides did contribute to toxicity because one sample with no measured hydrogen sulfide exhibited toxicity. Another sample with the second-highest inferred hydrogen sulfide concentration was not toxic. The authors note that metals are not present in concentrations to be bioavailable, leading to the conclusion that polycyclic aromatic hydrocarbons (PAHs), tPCB, tDDT or unmeasured contaminant(s) are responsible. Examination of the PAH data in Bay et al. indicates that PAH concentrations in toxic samples are just above or below Long et al. (1995) ER-L values and thus are not likely to be a cause of toxicity.

Laboratory bioassays using the marine polychaete full life-cycle (120-day) test were conducted as part of the NRDA. Sediment NOECs for *Neanthes arenaceodentata* based on survival, growth, fecundity, and reproduction were 8.51 mg/kg DW (269 mg/kg OC) for DDT and 1.07 mg/kg DW (36.6 mg/kg OC) for tPCB (Chapman 1996; EVS 1994). Amphipod tests conducted showed no acute toxicity.

MacDonald (1997) conducted an exhaustive review of laboratory and field investigations related to the biological effects of DDT and tPCB to benthic macroinvertebrates exposed to sediments from the SCB. Using a tiered strategy and a weight-of-evidence approach, he established SEC thresholds for tDDT, DDE, DDD, tDDT, Aroclor 1254, and tPCB. Exceedance of the SEC would indicate that effects, e.g., reduced survival and reproduction, on sensitive species are likely to occur. Field data were used only if no information from controlled laboratory studies (i.e., spiked sediment bioassays using arthropods) with dose-response findings were available to determine SECs. Estimates of the reliability and predictability of these thresholds were derived by comparison of SECs to independent data sets. Estimated SEC values normalized to dry weight as µg/kg are as follows: DDT = 31; DDE = 6,580; DDD = 890; tDDT = 7,120; Aroclor 1254 = 400; and tPCB = 835. Based upon information available at the time, MacDonald concluded that the sediments in the SCB are sufficiently contaminated with these compounds to cause injury to sediment-dwelling organisms.

3.2.1.3 Site-Specific Field Surveys

Benthic invertebrate field surveys have been conducted at PVS for over 28 years (LACSD 2000). Benthic infauna are abundant and diverse on the PVS and water depth appears to be the largest determinant of species distributions; however, outfall effects (in general) at PVS have been documented. For example, infauna species richness is reduced in close proximity to the LACSD outfall. Larger, epibenthic invertebrates are also reduced in abundance and diversity in the area close to the outfall. However, the effect is not PVS-wide, as some species have been observed in greater abundance at PVS in comparison to other SCB sites.

An extensive, recent comparative survey of infauna and epibenthic invertebrates (and fish) throughout the SCB and PVS indicated that species distributions were most strongly influenced by water depth (Allen et al. 1997, Bergen et al. 1997). Depth and sediment grain size (a surrogate for hydrographic influences on sediment sorting and deposition) appeared to be the main determinant of species assemblages in the SCB, in general (Bergen et al. 1997). Nevertheless, comparison of the PVS benthic community to those at other SCB sites (251 sites total) revealed a noticeable effect of the outfalls on community structure.

3.2.1.4 Site-Specific Observations of Ecological Effects

Benthic community and sediment toxicity data were analyzed to establish the relationship between predicted and measured biological effects. Toxicity data were co-located with chemical measurements taken at USGS or LACSD stations. Benthic infaunal data were collected by LACSD (1992) at 16 stations within the USGS footprint and three stations to the east in July and August, 1992. Analyses of trends between the 1992-1994 chemical data indicated no real temporal trends between the 1992-1994 and the 1996 data sets. Thus, the 1992 benthic community data that are compatible with the USGS sediment chemical data were evaluated. Co-located sediment toxicity data were collected in 1993 and 1994 by Bay et al. (1994) and EVS (1994).

The relative abundance and species composition at these LACSD stations were compared by cluster analysis to those observed for reference sites identified for the SCBPP (Bergen et al. 1999). Twelve reference stations were selected within the geographic range of collection—four stations for each water depth stratum (30, 60, and 150 m) most similar to Palos Verdes sites with respect to particle size.

A benthic response index (BRI), developed as part of the Bight 94 project (Smith et al. 1999), was used to evaluate conditions at PVS. The index was based on four thresholds of biological response to pollution that utilize individual species pollution tolerance scores (Smith et al. 1999). The effect assessment was scaled relative to area reference conditions (USEPA 2000).

3.2.2 Fish

Effects data for fish include literature-derived toxicity data for external (water-mediated) and internal (body burden-based) exposure and site-specific observations of ecological effects.

3.2.2.1 Literature-Derived Toxicity Data

USEPA chronic water quality criteria, typically based on a large number of toxicity studies, were selected as effects-based toxicity benchmarks for both tDDT and tPCB measured in the water column. USEPA chronic criteria are 0.0001 µg/L and 0.03 µg/L for tDDT and tPCB, respectively (USEPA 2002). As discussed for benthic invertebrates (Section 3.2.1.1), the chronic benchmark (or CCC) for tDDT was derived for the protection of fish eating birds and is likely to overestimate risk to fish. Therefore, the chronic benchmark of 0.0072 µg/L developed by dividing the CMC by 18 was selected as a more appropriate benchmark for fish. Although the acute benchmark was based on data for an invertebrate species (brown shrimp), it is assumed to be protective of both benthic invertebrates and fish because benthic invertebrates are generally more sensitive to contaminant exposure than fish.

Whole-body concentrations of COPECs in fish that have been associated with effects in field or laboratory animals are used to evaluate body burden-based exposure data. Meador et al. (2002) list six criteria they used to select appropriate studies for their benchmarks. These methods were adapted for use at the PVS, and the following criteria were used to select benchmarks:

1. Species were selected to be the best available match for chosen receptors.
2. Results were from a controlled laboratory study.
3. Study duration was a minimum of 20 days to represent chronic effects.
4. Whole-body concentrations were reported, and exposure was by either water or diet.

5. Responses were statistically different from the controls.
6. Life stage was relevant.
7. Organisms were exposed to mixtures of tDDT or tPCB.

The highest whole-body concentration in fish with no observed effects was selected for the NOEC and the lowest whole-body concentration, greater than the selected NOEC, in which adverse effects were observed was selected for the LOEC. Effects considered included reductions in survival and reproduction. Although these concentrations represent levels at which adverse effects were observed in individual test fish, it is likely that the reduction in growth, reproduction, or survival of individuals will also reduce the reproduction and survival of fish communities. Selected NOECs and LOECs are presented in Table 3-6.

TABLE 3-6
Selected NOECs and LOECs for Whole-Body Fish Tissues
Palos Verdes Shelf Ecological Risk Assessment

Analyte	Endpoint	Value(mg/kg ww)	Chemical Tested	Reference
tDDT	NOEC	1.92	DDT	Macek and Korn (1970)
tDDT	LOEC	12.1	4,4'-DDT	Buhler et al. (1969)
tPCB	NOEC	21	Clophen A50	Bengtsson (1980)
tPCB	LOEC	170	Clophen A50	Bengtsson (1980)

3.2.2.2 Site-Specific Observations of Ecological Effects

Several species of fish have been the subject of studies conducted at the PVS. The occurrence of white croaker at the PVS has led to their frequent use in toxicity/impairment studies. Other fish species, such as kelp bass, also have been studied.

Spies and Thomas (1997) measured several reproductive endpoints in female kelp bass caught at the Palos Verdes peninsula and at Dana Point (considered a "reference" location), including endpoints related to endocrine disruption. They found that plasma estradiol was lower, whereas plasma testosterone levels were higher in female kelp bass caught at Palos Verdes, compared to fish from Dana Point. They also found, in relation to hepatic estradiol binding, that 2,4'-DDE and 2,4'-DDT were weak competitors, and could displace estradiol from its receptor about 100 to 1,000 times less effectively than unlabeled estrogen. Thus, the binding affinity of estrogen was slightly lower in fish from the PVS. Therefore, long-term, population-level effects could occur if reproductive success was significantly impaired by disruption of estradiol and testosterone-mediated processes, as suggested by the results of Cross and Hose (1988).

White croakers sampled at the PVS were found to have shorter life expectancy than white croakers caught at Dana Point (Moore 2000). Although growth rates did not appear to be affected, Moore postulated that life expectancy could have been lower for white croakers from the PVS. In addition to contaminant body burdens, recruitment, proximity to nursery grounds, and fishing pressure were all thought to contribute to the difference in age structures of white croaker populations in the two areas. Any changes in growth rates could, therefore, be attributed to more than just contaminant body burden.

Allen et al. (1997) studied the distribution, relative importance, and health of dominant fish species of the southern California mainland shelf. Pacific sanddab (*Citharichthys sordidus*), plainfin midshipman (*Porichthys notatus*), California halibut (*Paralichthys californicus*), slender sole (*Eopsetta exilis*), Dover sole (*Microstomus pacificus*), and white croaker were among the top six species in areal coverage, total abundance, or total biomass, suggesting these species of fish are relatively important ones in the shelf ecosystem. In addition to these population-level parameters, Allen et al. found that anomalies commonly found in fish caught at the PVS in the 1970s (such as tumors, fin erosion, lesions, and skeletal deformities) were rarely found along the mainland shelf in 1994. They suggested the measured 1 percent anomaly rate in the 1970s likely represented background conditions because demersal fish populations were relatively healthy.

3.2.3 Birds and Marine Mammals

Effects data for birds and marine mammals include literature-derived toxicity data for both external and internal exposures, site-specific field studies, and site-specific modeling studies. These data are described below.

3.2.3.1 Literature-Derived Toxicity Data

External Exposures

Single-chemical toxicity data for birds and mammals consist of NOAELs or LOAELs derived from toxicity studies reported in the literature. The selection of studies was based on an extensive search of primary literature, review papers, and electronic databases. Appropriate studies were selected based on the following criteria:

- Studies were of chronic exposures or exposures during a critical life-stage (i.e., reproduction).
- Exposure was oral through food, to ensure that data represented oral exposures expected for wildlife in the field.
- Emphasis was placed on studies of reproductive impacts because they could be significant and more sensitive than survival.
- Studies presented adequate information to evaluate and determine the magnitude of exposure and effects (or no effects concentrations).
- Data were extracted from original sources to verify levels of effects, quality of study design, magnitude of dose, and other study parameters. Secondary sources were not considered, except as a source for identification of primary literature. Avian toxicity data are summarized in Tables 3-7 and 3-8, and those for mammals are in Tables 3-9 and 3-10. Information concerning assumptions made as part of the extraction of data from each study is presented in Appendix D.

Multiple toxicity studies were available for both birds and mammals for each COPEC. Toxicity studies were selected to serve as the primary toxicity value if exposure was chronic or during reproduction, the dosing regime was sufficient to identify both a NOAEL and a LOAEL, and the study considered ecologically relevant effects (i.e., reproduction, survival, growth). If multiple studies for a given COPEC met these criteria, the study generating the lowest reliable toxicity value was selected to be the primary toxicity value. Primary toxicity

values were used for all initial evaluations of the exposure estimates and are highlighted in Tables 3-7 through 3-10. Additionally, toxicity studies using the selected bird and mammal receptors (e.g., brown pelicans, double-crested cormorants, bald eagles, and California sea lions) or closely related species were selected for additional evaluations of exposure estimates, even if not considered in the primary study. For example, an oral exposure of 0.027 mg DDT/kg/d has been associated with adverse reproductive effects (fledgling success 30 percent below that needed to maintain a stable population) in brown pelicans (Anderson et al. 1975; Anderson et al. 1977). This study was not chosen as the primary toxicity study because it is a field study where pelicans were exposed to multiple chemicals in the environment. Therefore, observed reproductive effects cannot be wholly attributed to DDT exposure. However, this is a site-specific study and provides a valuable measure to evaluate current pelican exposures. Although selected toxicity values represent levels at which adverse effects were observed in individual test species, it is likely that reduction in the growth, reproduction, or survival of individuals will also reduce the growth, reproduction or survival of receptor populations. Studies such as these that were also used in the risk evaluation are italicized in Tables 3-7 to 3-10.

NOAELs and LOAELs for mammalian and avian receptors were estimated from literature data using allometric scaling methods presented in Sample et al. (1996) and Sample and Arenal (1999). Using the following equation, no (or lowest) observed adverse effects levels for wildlife ($NOAEL_w$ or $LOAEL_w$) were determined for each species:

$$NOAEL_w = NOAEL_t \left(\frac{BW_t}{BW_w} \right)^{1-b} \text{ or } LOAEL_w = LOAEL_t \left(\frac{BW_t}{BW_w} \right)^{1-b} \qquad \text{[Equation 3.7]}$$

Where:

$NOAEL_t$	=	the NOAEL for a test species (obtained from the literature)
$LOAEL_t$	=	the LOAEL for a test species (obtained from the literature)
BW_t and BW_w	=	the body weights (in kg) for the test and wildlife species, respectively
b	=	the class-specific allometric scaling factor

Scaling factors of 0.94 and 1.2 were applied for mammals and birds, respectively (Sample and Arenal, 1999). These receptor-specific NOAELs and LOAELs are presented in Table 3-11.

Another measure used to evaluate the potential for adverse effects in piscivorous wildlife is tissue levels of contaminants in prey. For example, tPCB have been linked to reduced reproductive success in piscivorous birds in the Great Lakes region (Colborn 1991; Fox et al. 1991a and 1991b; Giesy et al. 1994a), and in bald eagles (Wiemeyer et al. 1984; Bowerman et al. 1993; Giesy et al. 1994b). The Great Lakes International Joint Commission (1988) recommends 0.1 mg/kg WW tPCB as a prey tissue level that will protect predatory birds and mammals. These tissue levels in prey of piscivorous wildlife are included with target-organ effects data in Tables 3-12 through 3-15.

Internal Exposures

Internal (or target-organ-based) exposures consist of measured concentrations of COPECs in target organs (e.g., egg or blubber) of receptor birds and marine mammals. Concentrations of COPECs in these target organs that have been associated with effects in field or laboratory animals are used to evaluate target-organ-based exposure data. Target-organ effects concentration data were derived from published studies.

Total DDT threatens eagles and other birds at high trophic levels by disrupting normal estrogen function, the hormone responsible for the deposition of calcium in egg shells. Eggshell thinning, embryo mortality, and decreased hatchling survival have been linked to chronic exposure to DDT and its metabolites in the diet of birds. Of the three compounds DDT, DDE, and DDD, evidence strongly indicates that DDE is responsible for most reproductive toxicity in avian species (Blus et al. 1972; Blus 1982; Blus 1996). Eggshell thinning has been described for more than 18 families of birds in North America (Hickey and Anderson 1968; Risebrough 1986; Anderson and Hickey 1972). Measurements of residues in eggs of birds reliably indicate adverse effects. There is a large amount of variability in sensitivity to tDDT and its metabolites among bird species, with waterfowl and raptor species showing the greatest sensitivities. Studies have shown the brown pelican to be the most susceptible toadverse effects, with eggshell thinning and depressed productivity occurring at 3.0 µg/g WW of DDE in the egg, and total reproductive failure when residues exceed 3.7 µg/g WW (Blus 1984; Blus 1996). Other species have experienced widespread, although less severe, declines attributable to DDE. They include the double-crested cormorant, osprey, peregrine and bald eagles (Anderson et al., 1969; Ames 1966; Sprint et al. 1973). Wiemeyer et al. (1993) found that reproductive impairment for eagles occurred when egg concentrations exceeded 3.6 µg/g WW DDE. Below this concentration, the production of young was normal. Between 3.6 and 6.3 µg/g WW, the production of young was halved and halved again when concentrations exceeded 6.3 µg/g WW. Critical egg residue DDE levels for reproductive impairment for osprey and peregrine falcons are 4 and 15 mg/kg WW, respectively (Wiemeyer et al. 1988; Peakall et al. 1975).

Similar information is available about effects levels for tPCB in birds and, to some extent, about organ-based concentrations in mammals. Tables 3-12 and 3-13 summarize target organ effect concentrations derived from published sources for birds, and Tables 3-14 and 3-15 summarize information for mammals. Effects considered include reductions in reproduction, eggshell thinning, and clinical toxicity (e.g., decreased immune function in seals). Effects such as reduced immune function are likely to reduce the survival of affected individuals in the field. Therefore, these are ecologically relevant endpoints.

The published target organ effect concentrations represent levels at which adverse effects were observed in individual test species, but it is likely that these adverse effects on individuals also will reduce the growth, reproduction, or survival of receptor populations. Selected target organ benchmarks are highlighted in Tables 3-12 to 3-15.

Toxic Equivalency Factors (TEFs) that are specific to avian species (e.g., Van den Berg et al. 1998) have been developed for tPCB. These were used to calculate Toxic Equivalency Quotient (TEQs) for egg residues of tPCB for avian receptors (see Table 2-13). Target organ effect concentration data for TEQs were also compiled for comparison to these calculated TEQs, and are presented in Table 3-13 along with the tPCB studies.

TABLE 3-7
tDDT Oral Exposure Benchmarks for Avian Receptors
Palos Verdes Shelf Ecological Risk Assessment

Test Species	Analyte	Effect	Study Type	Body Weight (kg)	Exposure	No. of Dosage Levels	Duration	NOEC	LOEC	Units	NOAEL (mg/kg/d)	LOAEL (mg/kg/d)	Primary Reference	Secondary Reference	Comments
Mallard	technical DDT	20 percent eggshell thinning	laboratory	0.94	oral	4	1 year	20	40	mg/kg diet	1.7	3.4	Davison and Sell 1974	—	—
Mallard	DDT	44 percent reduced duckling survival	laboratory	1	oral	3	1 year	10	25	mg/kg diet	1	2.5	Heath et al. 1969	—	Author's note that wet weight values are 1/3 of dry weight.
Mallard	DDE	13 percent eggshell thinning and embryo mortality; 75 percent decrease in duckling production per hen	laboratory	1	oral	2	1 year	—	10	mg/kg diet	0.1	1	Heath et al. 1969	—	Severely impaired reproductive success; eggshells 13 percent thinner: author's note that wet weight values are 1/3 of dry weight; LOAEL to NOAEL conversion 0.1.
Black duck	DDE	Duckling survival; 17.6 percent eggshell thinning	laboratory	1.25	oral	2	through reproduction	—	10	mg/kg diet	0.1	1	Longcore et al. 1971	Blus 1996	Mean residue in eggs was 46 µg/g WW; LOAEL to NOAEL conversion 0.1.
Brown pelican	*DDT*	*reproductive success; fledgling rate 30 percent below that needed for stable population*	*field*	*3.5*	*oral*	*1*	*5 years*	—	*0.15*	*mg/kg diet*	*0.0027*	*0.027*	*Anderson et al. 1975 and 1977*	*EFAW 1998*	*LOAEL to NOAEL conversion 0.1; BTAG recommends LOAEL to NOAEL of 0.33 and uses this NOAEL (0.009) as low TRV.*
Screech owl	DDE	12 percent-13 percent eggshell thinning	laboratory	0.181	oral	1	2 years	—	2.8	mg/kg diet	0.039	0.39	McLane and Hall 1972	—	LOAEL to NOAEL conversion 0.1.
Barn owl	*DDE*	*reproductive success; eggshell thinning; embryo mortality*	*laboratory*	*0.466*	*oral*	*1*	*2 years*	—	*3*	*mg/kg diet*	*0.04*	*0.4*	*Mendenhall et al. 1983*	—	*Mean residue in eggs was 12 µg/g WW after 1st year and 41 after 2nd year; LOAEL to NOAEL conversion 0.1.*

Notes:
— information that was not reported or not pertinent.
BTAG Biological Technical Assistance Group

The shaded study was chosen as the primary toxicity study.
Studies in italics, although not the primary study, were used in the risk evaluation because they used one of the ERA receptors or a closely related species.

TABLE 3-8
tPCB Oral Exposure Benchmarks for Avian Receptors
Palos Verdes Shelf Ecological Risk Assessment

Test Species	Analyte	Endpoint	Effect	Study Type	Body Weight (kg)	No. of Dosage Levels	Exposure	Duration	NOEC	LOEC	Units	TEQ NOEC	TEQ LOEC	Units	NOAEL (mg/kg/d)	LOAEL (mg/kg/d)	TEQ NOAEL (ng/kg/d)	TEQ LOAEL (ng/kg/d)	Primary Reference	Secondary Reference	Comments
American kestrel	Aroclor 1254	porphyrinogenic effects	hepatic APND and AE, CYP1A like proteins	laboratory	0.13	1	Oral	12 weeks								7		311.5	Elliott et al. 1997	—	Only one dosage level tested for chronic study
American kestrel	*Aroclor 1248*	*reproduction*	*eggshell thinning*	*laboratory*	*0.13*	*1*	*Oral*	*7 months*		*3*	*µg/g*	*753.9*		*ng/kg*	*0.115*	*1.15*		*289.9*	*Lowe and Stendell 1991*	—	*Significant eggshell thinning; NOAEL calculated by multiplying the LOAEL by 0.1*
American kestrel	Aroclor 1254	reproduction	male fertility	laboratory	0.13		—	—		33	mg/kg diet	1468.5		ng/kg diet		2.54		112.96	Bird et al. 1983	Barron et al. 1995	
American kestrel	*Aroclor 1254*	*reproduction*	*female fertility*	*laboratory*	*0.13*		—	—	*0.5*	*5*	*mg/kg diet*	*222.5*	*22.25*	*ng/kg diet*	*0.04*	*4*	*1.71*	*17.1*	*Lincer and Peakall 1970*	*Barron et al. 1995*	
Chicken	Aroclor 1254	reproduction	decreased egg production, female fertility	laboratory	1.5		Oral	28 weeks		5	mg/kg diet	222.5		ng/kg diet		0.35		1.59	Platonow and Rienhart 1973	—	
Japanese quail	Aroclor 1254	porphyrinogenic effects	HCPs, hepatic EROD, 4-CBH and ADND	laboratory	0.2	1	Oral	12 weeks		—	—	—		—		7		311.5	Elliott et al. 1997	—	Only one dosage level tested for chronic study
Japanese quail	Aroclor 1248	—	—	laboratory	0.15		—	—	50		mg/kg diet	12565		ng/kg	5.63		1415.6		Chang and Stockstad 1975	Barron et al. 1995	
Japanese quail	Aroclor 1254	—	—	laboratory	0.15		—	—	20		mg/kg diet	890		ng/kg diet	2.25		100.3		Scott 1977	Barron et al. 1995	
Mallard	Aroclor 1254	reproduction	eggshell thinning	laboratory	1.25		—	—		150	mg/kg diet	6675		ng/kg diet		15		667.5	Haseltine and Prouty 1980	Barron et al. 1995	
Mallard	Aroclor 1254	reproduction	reproductive success	laboratory	1.25	1	Oral	1 month		25	mg/kg diet	1112.5		ng/kg diet		2.5		111.25	Custer and Heinz 1980	—	Only one dosage level and a control
Mourning dove	Aroclor 1254	reproduction	behavior	laboratory	0.108		—	—		10	mg/kg diet	445		ng/kg diet		1.57		70.1	Tori and Peterle 1983	Barron et al. 1995	
Northern bobwhite	Aroclor 1254	—	—	laboratory	0.178		—	—	50		mg/kg diet	2225		ng/kg diet	30.9		1375		NAS 1979	Barron et al. 1995	
Ring dove	Aroclor 1254	neurological	decreased dopamine and norepinephrine levels	laboratory	0.155	3	Oral	8 weeks	1	10	mg/kg diet	—		—	0.072	0.74			Heinz et al. 1980	—	—
Ringed turtle dove	Aroclor 1254	reproduction	hatching success	laboratory	0.155		—	—		10	mg/kg diet	445		ng/kg diet		1.09		48.8	Peakall et al. 1972	Barron et al. 1995	
Ring-necked pheasant	Aroclor 1254	reproduction	female fertility	laboratory	1		—	—		50	mg/kg diet	2225		ng/kg diet		2.91		129.5	Roberts et al. 1978	Barron et al. 1995	
Screech owl	*Aroclor 1248*	*reproduction*	*parameters*	*laboratory*	*0.181*	*1*	*Oral*	*2 breeding seasons*	*3*		*µg/g*	*753.9*		*ng/kg*	*0.41*		*104.1*		*McLane and Hughes 1980*	—	*No detectable effect on screech owl production*
White leghorn hens	Aroclor 1242	reproduction	reduced hatchability	laboratory	1.5		Oral	—		—	—	—		—	0.0072	0.072	1.68912	16.8912	Lillie et al. 1974	CCMOE 2001	Assuming BW of 1 kg and food ingestion of 0.06 kg/day
White leghorn hens	Aroclor 1254	physiologic	growth	laboratory	1.5		Oral	—		—	—	—		—		0.12		5.3	Lillie et al. 1974	CCMOE 2001	10 percent reduction in growth
White leghorn hens	Aroclor 1254	physiologic	growth	laboratory	1.5		Oral	—		—	—	—		—		0.021		0.95	Lillie et al. 1974	CCMOE 2001	10 percent reduction in growth
White leghorn hens	Aroclor 1242	reproduction	decreased hatchability	laboratory	1.5	5	Oral	6 weeks	5	10	mg/kg diet	222.5	445	ng/kg diet	0.3	0.6	1.35	26.7	Britton and Huston 1973	.	Assuming BW of 1 kg and food ingestion of 0.06 kg/day

Notes:
— information that was not reported or not pertinent.
The shaded study was chosen as the primary toxicity study.
Studies in italics, although not the primary study, were used in the risk evaluation because they used one of the ERA receptors or a closely related species.

TABLE 3-9
tDDT Oral Exposure Benchmarks for Mammalian Receptors
Palos Verdes Shelf Ecological Risk Assessment

Test Species	Analyte	Endpoint	Effect	Study Type	Body Weight (kg)	Exposure	No. of Dosage Levels	Duration	Tissue	NOEC	LOEC	Units	NOAEL (mg/kg/d)	LOAEL (mg/kg/d)	Primary Reference	Secondary Reference	Comments
Rat	DDT	reproduction	reduced production of young	laboratory	0.32	oral	4	2 years	dietary	10	50	—	0.8	4	Fitzugh 1948	EFAW 1998	NOAEL recommended as low TRV by BTAG
Harbor seal	*DDT*	*physiologic*	*growth*	*semi-field*	*50*	*oral*	*2*	*93 weeks*	*dietary*	*—*	*—*	*—*	*0.009*	*—*	*de Swart et al. 1994*	*Kannan et al. 2000*	*Seals remained healthy and exhibited normal growth patterns during the study*
Beagle dog	technical DDT	reproduction	length of gestation, fertility, litter size etc.	laboratory	—	oral	3	Multiple generations	—	—	—	—	10	—	Ottoboni et al. 1977	—	—
Mice	DDT	reproduction	reproductive success	laboratory	0.03	oral	2	3-5 months	dietary	—	300	mg/kg diet	—	55	Bernard and Gaertner 1964	—	—
Mice	DDT	reproduction	lactation index	laboratory	0.03	oral	1	Through generations	dietary	—	100	mg/kg diet	—	18.3	Del Pup et al. 1978	—	—
Rat	DDT	liver	hepatic cell alteration	laboratory	0.35	oral	4	15-27 weeks	dietary	—	5	mg/kg diet	—	0.4	Laug et al. 1950	—	Seen to greater degree in male
Rat	DDT	reproduction	reproductive success	laboratory	0.35	oral	4	Two generations	dietary	40	—	mg/kg diet	3.2	—	Wrenn et al. 1971	—	—

Notes:
— information that was not reported or not pertinent.

The shaded study was chosen as the primary toxicity study.
Studies in italics, although not the primary study, were used in the risk evaluation because they used one of the ERA receptors or a closely related species.

TABLE 3-10
tPCB Oral Exposure Benchmarks for Mammalian Receptors
Palos Verdes Shelf Ecological Risk Assessment

Test Species	Analyte	Endpoint	Effect	Study Type	Body Weight (kg)	Exposure	No. of Dosage Levels	Duration	NOAEL (mg/kg/d)	LOAEL (mg/kg/d)	TEQ NOAEL (mg/kg/d)	TEQ LOAEL (mg/kg/d)	Primary Reference	Secondary Reference	Comments
Mice	Aroclor 1254	reproduction	PROD activity, liver weight, pentobarbitol-induced sleep time	laboratory	0.028	Oral	—	21 days	0.36	—	10.836	—	Simmons and McKee 1992	EFAW 1998	Recommended as low TRV
Mink	tPCB	reproduction	—	laboratory	1.12	—	—	—	0.004	0.13	0.25	3.6	Heaton et al. 1995	Kannan et al. 2000	—
Mink	tPCB	reproduction	kit growth	laboratory	1.12	—	—	—	0.026	0.15	0.8	4.5	Wren et al. 1987	CCMOE 2001	—
Mink	tPCB	reproduction	—	—	1.12	—	—	—	—	—	0.42	—	Tillet et al. 1996	Kannan et al. 2000	Threshold dietary [ ]
Mink	Aroclor 1242	reproduction	total reproductive failure	laboratory	1	Oral	4	7 months	0.069	0.69			Bleavins et al. 1980		
Mink	Aroclor 1254	reproduction	number of live offspring	laboratory	1	Oral	3	4.5 months	0.14	0.69			Aulerich and Ringer 1977		
Oldfield mouse	Aroclor 1254	reproduction	number of litters, offspring weight and survival	laboratory	0.014	Oral	1	12 months	0.068	0.68			McCoy et al. 1995		
Rat	Aroclor 1254	hormonal	thyroid hormone and vitamin A levels	laboratory	0.35	Oral	1	2 weeks	—	10	—	301	Hallgren et al. 2001	—	Only one dose tested for Aroclor
Rhesus monkey	Aroclor 1248	reproduction	pregnancy and live births	laboratory	5	Oral	2	14 months	—	0.087	—	1.1136	Barsotti et al. 1976	—	Intake =75.3 mg over 173 days therefore 0.43 mg/d
Harbor seal	*PCB*	*reproduction*	*reproductive success*	*semi-field*	*50*	*oral*	*2*	*2 years*	*0.005*	*0.03*	—	—	*Reijinders 1986; Boon et al. 1987; Brouwer et al. 1989*	*Kannan et al. 2000*	—
Harbor seal	*PCB*	*immune function*	*decreased vit A levels, NK cell activity, and lymphocyte proliferation*	*semi-field*	*50*	*oral*	*2*	*93 weeks*	*0.0052*	*0.0292*			*de Swart et al. 1994*	*Kannan et al. 2000*	—

Notes:

— information that was not reported or not pertinent.

The shaded study was chosen as the primary toxicity study.
Studies in italics, although not the primary study, were used in the risk evaluation because they used one of the ERA receptors or a closely related species.

TABLE 3-11
Receptor-specific NOAELs and LOAELs for Oral Exposure (mg tDDT or tPCB/kg BW/day) in Avian and Mammalian Receptors[a]
Palos Verdes Shelf Ecological Risk Assessment

	tDDT			tPCB		
Receptor	**NOAEL**	**LOAEL**	**Study**	**NOAEL**	**LOAEL**	**Study**
Primary Benchmark						
California brown pelican	1.348	3.370	Health et al. 1969	0.365	0.731	Britton and Huston 1973
Double-crested cormorant	1.212	3.031	Health et al. 1969	0.329	0.657	Britton and Huston 1973
Peregrine falcon	0.957	2.393	Health et al. 1969	0.260	0.519	Britton and Huston 1973
Bald eagle	1.456	3.641	Health et al. 1969	0.395	0.790	Britton and Huston 1973
Adult female California sea lion	0.567	2.834	Fitzhugh 1948	0.106	0.523	Aulerich and Ringer 1977
Nursing California sea lion pup	0.625	3.124	Fitzhugh 1948	0.117	0.577	Aulerich and Ringer 1977
Other Benchmarks						
California brown pelican	0.0027	0.027	Anderson et al. 1975 and 1977			
Peregrine falcon	0.046	0.463	Mendenhall et al. 1983			
Bald eagle	0.071	0.705	Mendenhall et al. 1983			
Peregrine falcon				0.183	1.834	Lowe and Stendell 1991
Bald eagle				0.279	2.789	Lowe and Stendell 1991
Peregrine falcon				0.064	6.377	Lincer and Peakall 1970
Bald eagle				0.097	9.702	Lincer and Peakall 1970
Peregrine falcon				0.602		McLane and Hughes 1980
Bald eagle				0.916		McLane and Hughes 1980
Adult female California sea lion	0.009		de Swart et al. 1994			
Adult female California sea lion				0.0052	0.028	Reijinders 1986; Boon et al. 1987; Brouwer et al. 1989
Adult female California sea lion				0.005	0.029	de Swart et al. 1994
Nursing California sea lion pup	0.010		de Swart et al. 1994			
Nursing California sea lion pup				0.0053	0.032	Reijinders 1986; Boon et al. 1987; Brouwer et al. 1989
Nursing California sea lion pup				0.0055	0.031	de Swart et al. 1994

[a] When test species differed from receptor, NOAELs and LOAELs were calculated using allometric scaling factors of 0.94 and 1.2 for mammals and birds, respectively (Sample and Arenal 1999).

TABLE 3-12
tDDT Internal (Target-Organ Based) Benchmarks for Avian Receptors
Palos Verdes Shelf Ecological Risk Assessment

Test Species	Analyte	Endpoint	Effect	Study Type	Tissue	NOEC	LOEC	Critical Threshold	Units	Basis	Primary Reference	Secondary Reference	Comments
Wildlife consumers	tDDT	—	—	—	aquatic biota tissue	—	—	14	µg/kg diet	WW	Environment Canada 1998	CCMOE 2001	Tissue residue guideline for protection of wildlife consumers of aquatic biota
Bald eagle	DDE	reproduction	productivity	modeled based on field info	blood		40	—	µg/kg	—	Wiemeyer et al. 1993, Elliot et al. 1998	Elliot and Norstrom 1998	Productivity at 0.52 young/nest
Bald eagle	DDE	reproduction	mean 5 year production at the rate of 1	modeled based on field info	egg	1.5	—	—	mg/kg	—	Wiemeyer et al. 1993		Used unweighted linear regression equation to predict, highest level associated with a healthy population
Bald eagle	DDE	reproduction	mean 5 year production at the rate of 1	modeled based on field info	egg	1.8	—	—	mg/kg	—	Wiemeyer et al. 1993		Used weighted linear regression equation to predict, highest level associated with a healthy population
Bald eagle	DDE	reproduction	mean 5 year production at the rate of 1	modeled based on field info	egg	2.7	—	—	mg/kg	—	Wiemeyer et al. 1993		Used weighted sigmoidal regression equation to predict, highest level associated with a healthy population
Bald eagle	DDE	reproduction	mean 5-year productivity of 0.7	modeled based on field info	egg	3.6	—	—	mg/kg	—	Wiemeyer et al. 1993		Used unweighted linear regression equation to predict, highest concentration associated with population stability
Bald eagle	DDE	reproduction	mean 5-year productivity of 0.7	modeled based on field info	egg	4.2	—	—	mg/kg	—	Wiemeyer et al. 1993		Used weighted linear regression equation to predict, highest concentration associated with population stability
Bald eagle	DDE	reproduction	mean 5-year productivity of 0.7	modeled based on field info	egg	4.3	—	—	mg/kg	—	Wiemeyer et al. 1993		Used weighted sigmoidal regression equation to predict, highest concentration associated with population stability
Bald eagle	DDE	reproduction	mean 5 year production at the rate of 0.5	modeled based on field info	egg	—	5.9	—	mg/kg	—	Wiemeyer et al. 1993		Used weighted sigmoidal regression equation to predict, productivity of 0.5
Bald eagle	DDE	reproduction	mean 5 year production at the rate of 0.5	modeled based on field info	egg	—	6.7	—	mg/kg	—	Wiemeyer et al. 1993		Used unweighted linear regression equation to predict, productivity of 0.5
Bald eagle	DDE	reproduction	mean 5 year production at the rate of 0.5	modeled based on field info	egg	—	7.4	—	mg/kg	—	Wiemeyer et al. 1993		Used weighted linear regression equation to predict, productivity of 0.5
Bald eagle	DDE	reproduction	eggshell thinning	regression analysis	egg	—	—	60	mg/kg	WW	Wiemeyer et al. 1993	Blus 1996	Estimated as causing 20 percent eggshell thinning
Barn owl	DDE	reproduction	eggshell thinning	—	egg	—	—	12	mg/kg	WW	Mendenhall et al. 1983	Blus 1996	Mean residue with mean eggshell thinning of 20 percent
Black duck	DDE	reproduction	eggshell thinning	—	egg	—	—	46	mg/kg	WW	Longcore et al. 1971	Blus 1996	Mean residue with mean eggshell thinning of 18 percent
Black-crowned night-heron	DDE	reproduction	reproductive failure	modeled based on field info	egg	—	—	12	mg/kg	—	Blus 1984		Determined using sample egg technique
Black-crowned night-heron	DDE	reproduction	eggshell thinning	regression analysis	egg	—	—	36	mg/kg	—	Blus 1984		Critical threshold associated with 18 percent thinning, regression analysis
Black-crowned night-heron	DDE	reproduction	eggshell thinning	regression analysis	egg	—	—	54	mg/kg	WW	Blus 1984, Henny et al. 1984	Blus 1996	Estimated as causing 20 percent eggshell thinning
Black-crowned night-heron	DDE	reproduction	eggshell thinning	regression analysis	egg	—	—	54	mg/kg	—	Blus 1984		Critical threshold associated with 20 percent thinning, regression analysis
Brown pelican	DDE	reproduction	reproduction	regression analysis	egg	2	3		mg/kg	WW	Blus 1984		40 percent decrease in nest success associated with LOEC; greatest reproductive success seen when eggs had 1-2 mg/kg DDE
Brown pelican	DDE	reproduction	reproductive failure	modeled based on field info	egg	—	—	3.7	mg/kg	—	Blus 1984		Determined using sample egg technique

TABLE 3-12
tDDT Internal (Target-Organ Based) Benchmarks for Avian Receptors
Palos Verdes Shelf Ecological Risk Assessment

Test Species	Analyte	Endpoint	Effect	Study Type	Tissue	NOEC	LOEC	Critical Threshold	Units	Basis	Primary Reference	Secondary Reference	Comments
Brown pelican	DDE	reproduction	eggshell thinning	regression analysis	egg	—	—	5	mg/kg	—	Blus 1984		Critical threshold associated with 18 percent thinning, regression analysis
Brown pelican	DDE	reproduction	eggshell thinning	regression analysis	egg	—	—	8	mg/kg	WW	Blus 1984	Blus 1996	Estimated as causing 20 percent eggshell thinning
California condor	DDE	reproduction	eggshell thinning	regression analysis	egg	—	—	5	mg/kg	WW	Kiff et al. 1979	Blus 1996	Estimated as causing 20 percent eggshell thinning
Common loon	DDE	reproduction	eggshell thinning	regression analysis	egg	—	—	14	mg/kg	WW	Price 1977	Blus 1996	Estimated as causing 20 percent eggshell thinning
common loon	DDE	reproduction	eggshell thinning	regression analysis	egg	—	—	47	mg/kg	WW	Fox et al. 1980	Blus 1996	Estimated as causing 20 percent eggshell thinning
Double crested cormorant	4,4'-DDE	reproduction	eggshell thinning	—	egg	—	—	10	mg/kg	WW	Pearce et al. 1979, Weseloh et al. 1983	Dirksen et al. 1995, Elliott et al. 1989	Concentration that causes 20 percent eggshell thinning (empirical pop. threshold)
Double crested cormorant	4,4'-DDE	reproduction	eggshell thinning	—	egg	4			mg/kg		Dirksen et al. 1995, Koeman et al. 1972, 1973		
Eurasion sparrowhawk	DDE	reproduction	eggshell thinning	regression analysis	egg	—	—	10	mg/kg	WW	Newton et al. 1986	Blus 1996	Estimated as causing 20 percent eggshell thinning
Great blue heron	DDE	reproduction	eggshell thinning	regression analysis	egg	—	—	19	mg/kg	WW	Blus 1984	Blus 1996	Estimated as causing 20 percent eggshell thinning
Herring gull	DDE	reproduction	eggshell thinning	field	egg	—	—	162	mg/kg	—	Keith and Gruchy 1972	Elliot et al. 1989	Calculated based on 20 percent eggshell thinning
Leach's storm petrel	DDE	reproduction	eggshell thinning	calculated value	egg	—	—	12	mg/kg	WW	Pearce et al. 1979	Elliot et al. 1989	Calculated based on 20 percent eggshell thinning
Merlin	DDE	reproduction	eggshell thinning	—	egg	—	—	5	mg/kg	—	Fyfe et al. 1988	Noble and Elliott 1990	Significant eggshell thinning
Merlin	DDE	reproduction	eggshell thinning	regression analysis	egg	—	—	16	mg/kg	WW	Newton et al. 1982	Blus 1996	Estimated as causing 20 percent eggshell thinning
Osprey	DDE	reproduction	eggshell thinning	—	egg	—	—	4	mg/kg	—	Wiemeyer et al. 1988	Noble and Elliott 1990	15 percent eggshell thinning, max conc. For sustainable population
Osprey	DDE	reproduction	eggshell thinning	regression analysis	egg	—	—	9	mg/kg	WW	Wiemeyer et al. 1988	Blus 1996	Estimated as causing 20 percent eggshell thinning
Osprey	DDE	reproduction	eggshell thinning	regression analysis	egg	—	—	29	mg/kg	WW	Wiemeyer et al. 1988	Blus 1996	Estimated as causing 20 percent eggshell thinning
Osprey	DDE	reproduction	eggshell thinning	regression analysis	egg	—	—	41	mg/kg	WW	Spitzer et al. 1978	Blus 1996	Estimated as causing 20 percent eggshell thinning
Peregrine falcon	DDE	reproduction	population decline	—	egg	—	—	15	mg/kg	WW	Peakall et al. 1975	Peakall and Kiff 1988	Range is 15 to 20 ppm WW
Peregrine falcon	DDE	reproduction	eggshell thinning	regression analysis	egg	—	—	15	mg/kg	WW	Peakall et al. 1975	Blus 1996	Estimated as causing 20 percent eggshell thinning
Peregrine falcon	DDE	reproduction	eggshell thinning	regression analysis	egg	—	—	18	mg/kg	WW	Pruett-Jones et al. 1980	Blus 1996	Estimated as causing 20 percent eggshell thinning

TABLE 3-12
tDDT Internal (Target-Organ Based) Benchmarks for Avian Receptors
Palos Verdes Shelf Ecological Risk Assessment

Test Species	Analyte	Endpoint	Effect	Study Type	Tissue	NOEC	LOEC	Critical Threshold	Units	Basis	Primary Reference	Secondary Reference	Comments
Peregrine falcon	DDE	reproduction	eggshell thinning	regression analysis	egg	—	—	20	mg/kg	WW	Enderson and Wrege 1973	Blus 1996	Estimated as causing 20 percent eggshell thinning
Peregrine falcon	DDE	reproduction	eggshell thinning	regression analysis	egg	—	—	22	mg/kg	WW	Cade et al. 1971	Blus 1996	Estimated as causing 20 percent eggshell thinning
Prairie falcon	DDE	reproduction	eggshell thinning	—	egg	—	—	1.2	mg/kg	—	Fyfe et al. 1988	Noble and Elliott 1990	Significant eggshell thinning
Prairie falcon	DDE	reproduction	eggshell thinning	regression analysis	egg	—	—	7	mg/kg	WW	Enderson and Wrege 1973	Blus 1996	Estimated as causing 20 percent eggshell thinning
Red-tailed hawk	DDE	reproduction	eggshell thinning	—	egg	—	15	—	mg/kg	—	Seidensticker and Reynolds 1971	Noble and Elliott 1990	Minor eggshell thinning
Swainson's hawk	DDE	—	—	—	egg	2.5	—	—	mg/kg	—	Henny et al. 1984	Noble and Elliott 1990	Evidence that Swainson's hawk can tolerate concentrations in excess of 2.5 mg/kg
White-faced ibis	DDE	reproduction	eggshell thinning	regression analysis	egg	—	—	7	mg/kg	WW	Henny and Herron 1989	Blus 1996	Estimated as causing 20 percent eggshell thinning
Raptors	DDE	—	—	—	liver	—	—	100	mg/kg	—	Cooke et al. 1982	Noble and Elliott 1990	—
Peregrine falcon	DDE	reproduction	reproductive success	—	prey	—	—	1	mg/kg	—	Enderson et al. 1982	Elliot et al. 1989	Represents a significant threat to reproductive performance

Notes:
— information was not reported or not pertinent

TABLE 3-13
Total PCB Internal Exposure (Target-Organ Based) Benchmarks for Avian Receptors
Palos Verdes Shelf Ecological Risk Assessment

Test Species	Analyte	Endpoint	Effect	Study Type	Tissue	NOEC	LOEC	EC	Units	NOEC TEQ	LOEC TEQ	Units	Basis	Primary Reference	Secondary Reference	Comments
Ring dove	tPCB	neurological	brain neuorotranmitter levels	laboratory	brain	1	2.82	—	mg/kg	—	—	—	—	Heinz et al. 1980		—
Ring dove	tPCB	reproduction	—	laboratory	brain	—	5.5	—	mg/kg	—	—	—	—	Peakall and Peakall 1973	Barron et al. 1995	—
Bald eagle	tPCB	reproduction	population size or reproductive success	modeled based on field info	egg	1.3	4	—	mg/kg	—	—	—	—	Wiemeyer et al. 1984	Barron et al. 1995	Derived by Ludwig et al. 1993
Bald eagle	tPCB	reproduction	productivity rate of 1	modeled based on field info	egg	4	—	—	mg/kg	—	—	—	WW	Wiemeyer et al. 1984	Bowerman et al. 1998	Determined using weighted sigmoidal regression analysis
Bald eagle	tPCB	reproduction	population size or reproductive success	modeled based on field info	egg	—	7.2	—	mg/kg	—	—	—	—	Wiemeyer et al. 1984	Barron et al. 1995	—
Bald eagle	tPCB	reproduction	reduced nesting success	—	egg	—	12	—	ug/g	—	—	—	—	Nisbet and Risebrough 1994	Clark et al. 1998	—
Bald eagle	tPCB	reproduction	population size or reproductive success	—	egg	—	13	—	mg/kg	—	—	—	—	Bosveld and Van den Berg, in press	Barron et al. 1995	—
Black-crowned night-heron	tPCB	reproduction	reproductive success	—	egg	10.9	—	—	mg/kg	—	—	—	—	Tremblay and Ellison 1980	Barron et al. 1995	Based on no apparent adverse effects in the field population
Chickens	Aroclor 1254	reproduction	hatching success	laboratory	egg	13.2	—	—	mg/kg	587.4	—	ng/kg	—	Cecil et al. 1977	Rice and O'Keefe 1995	—
Dipper	tPCB	reproduction	breeding performance and survival	field	egg	0.49	—	—	µg/g	—	—	—	WW	Omerod et al. 2000		Geometric mean
Double crested cormorant	PCB	reproduction	reproductive success	—	egg	—	3.5	—	mg/kg	—	—	—	—	Tillitt et al. 1993	Barron et al. 1995	—
Double crested cormorant	tPCB	deformities	—	field	egg	—	—	5.2 (7)	mg/kg	—	—	—	—	Ludwig et al. 1996		Eggs also measured for TCDD-Eqs so may be confounding results
Double crested cormorant	tPCB	deformities	—	field	egg	—	—	6 (3)	mg/kg	—	—	—	—	Ludwig et al. 1996		Eggs also measured for TCDD-Eqs so may be confounding results
Double crested cormorant	tPCB	deformities	—	field	egg	—	—	7.3 (5)	mg/kg	—	—	—	—	Ludwig et al. 1996		Eggs also measured for TCDD-Eqs so may be confounding results
Double crested cormorant	tPCB	deformities	—	field	egg	—	—	8.1(6)	mg/kg	—	—	—	—	Ludwig et al. 1996		Eggs also measured for TCDD-Eqs so may be confounding results
Double crested cormorant	tPCB	deformities	—	field	egg	—	—	8.75 (4)	mg/kg	—	—	—	—	Ludwig et al. 1996		Eggs also measured for TCDD-Eqs so may be confounding results
Double crested cormorant	tPCB	deformities	—	field	egg	—	—	11(8)	mg/kg	—	—	—	—	Ludwig et al. 1996		Eggs also measured for TCDD-Eqs so may be confounding results
Double crested cormorant	tPCB	deformities	—	field	egg	—	—	12.1 (7)	mg/kg	—	—	—	—	Ludwig et al. 1996		Eggs also measured for TCDD-Eqs so may be confounding results
Great blue heron	tPCB	reproduction	reproductive success	—	egg	7.8	—	—	mg/kg	—	—	—	—	Boily et al. 1994	Barron et al. 1995	Based on no apparent adverse effects in the field population
Herring gull	tPCB	reproduction	population size or reproductive success	—	egg	—	5	—	mg/kg	—	—	—	—	Ludwig et al. 1993	Barron et al. 1995	—
White leghorn hens	Aroclor 1254	reproduction	hatching success	laboratory	egg	—	4	—	mg/kg	—	178	ng/kg	—	Tumasonis et al. 1973	Rice and O'Keefe 1995	—
Ring dove	tPCB	reproduction	—	laboratory	fat	—	736	—	mg/kg	—	—	—	—	Peakall and Peakall 1973	Barron et al. 1995	—
American kestrel	tPCB	reproduction	male fertility	laboratory	liver	—	91.6	—	mg/kg	—	—	—	—	Bird et al. 1983	Barron et al. 1995	—
Ring dove	tPCB	reproduction	—	laboratory	liver	—	15	—	mg/kg	—	—	—	—	Peakall and Peakall 1973	Barron et al. 1995	—
Ring dove	tPCB	reproduction	—	laboratory	muscle	—	8	—	mg/kg	—	—	—	—	Peakall and Peakall 1973	Barron et al. 1995	—
Chickens	Aroclor 1254	reproduction	hatching success	laboratory	yolk	10	—	—	mg/kg	445	—	ng/kg	—	Cecil et al. 1977	Rice and O'Keefe 1995	—

Notes:

— information not reported or not pertinent.
EC effect concentration; percent affected in parentheses

The shaded study was chosen as the primary toxicity study.

TABLE 3-14
Total DDT Internal (Target-Organ Based) Benchmarks for Mammalian Receptors
Palos Verdes Shelf Ecological Risk Assessment

Test Species	Analyte	Endpoint	Effect	Study Type	Tissue	LOEC	Tissue Residue Guideline	Units	Basis	Primary Reference	Secondary Reference	Comments
Dolphin	tDDT	immunologic	proliferative response	laboratory	blood	13		Ng/g	WW	Lahvis et al. 1995	Kannan et al. 2000	Negative relationship between lymphocyte proliferative responses to mitogens and concentrations of tPCB and tDDT in the blood though the sample size was only 5 and the study lacked a reference population. Additionally, co-contaminants were also found in the blood.
Wildlife consumers	tDDT	—	aquatic biota tissue		dietary		14	µg/kg diet	WW	Environment Canada 1998	CCMOE 2001	Tissue residue guideline for protection of wildlife consumers of aquatic biota.

Notes:

— information that was not reported or not pertinent

The shaded study was chosen as the primary toxicity study.

TABLE 3-15
Total PCB Internal Exposure (Target-Organ Based) Benchmarks for Mammalian Receptors
Palos Verdes Shelf Ecological Risk Assessment

Test Species	Analyte	Endpoint	Effect	Study Type	Body Weight (kg)	Exposure	Duration	Tissue	NOEC	LOEC	Critical Threshold	Units	TEQ NOEC	TEQ LOEC	TEQ Critical Threshold	Units	Basis	Primary Reference	Secondary Reference	Comments
Harbor seal	tPCB	reproduction	reproductive success	semi-field	50	oral	2 years	Blood			4.5	ng/g		—		—	WW	Boon et al. 1987, Brouwer et al. 1989	Kannan et al. 2000	Maximum allowable tissue concentration.
Marine mammals	tPCB	—	—	—	—	—	—	Blood			8.7	mg/kg		—		—	lipid	Kannon et al. 2000	Kannan et al. 2000	Threshold [ ] in blood (derived from geometric mean of 3 values for seal, otter, and mink).
Harbor seal	tPCB	reproduction	reproductive success	semi-field	50	oral	2 years	Blood		16		ng/g		—		—	WW	Boon et al. 1987, Brouwer et al. 1989	Kannan et al. 2000	—
Dolphin	tPCB	immunologic	proliferative response	field	—	—	—	Blood		26		ng/g		—		—	WW	Lahvis et al. 1995	Kannan et al. 2000	Negative relationship between lymphocyte proliferative responses to mitogens and concentrations of PCB and tDDT in the blood though the sample size only 5, and study lacked reference population.
Marine mammals	tPCB	—	—	—	—	—	—	Blubber	—	—	17	mg/kg		—		—	lipid	Kannon et al. 2000	Kannan et al. 2000	Threshold [ ] in blubber (applied factor of 2 to blood threshold).
Harbor seal	tPCB	immunologic	proliferative response, thyroid hormone conc. etc.	semi-field	50	oral	93 weeks	Blubber		—		—	90			pg/g	lipid	de Swart et al. 1994	—	Seals remained healthy and exhibited normal growth patterns during the study.
Harbor seal	tPCB	immunologic	proliferative response, thyroid hormone conc. etc.	semi-field	50	oral	93 weeks	Blubber		—		—	286			pg/g	lipid	de Swart et al. 1994	.	Seals remained healthy and exhibited normal growth patterns during the study.
Rhesus monkey	Aroclor 1248	reproduction	conception rates	laboratory	—	oral	7 months	Dietary		5		mg/kg diet		64		ng/kg	—	Barsotti et al. 1976	Barsotti et al. 1976	Intake=284.56 mg over 310 days therefore 0.92 mg/day.
Otter	tPCB	liver	hepatic retinoids	semi-field	—	—	—	prey (fish)	0.012	0.033		mg/kg	0.001	0.002		ng/g	WW	Smit et al. 1996, Murk et al. 1998	Kannan et al. 2000	Dietary NOAEL; BMF for PCB was 14 on a WW basis.
Harbor seal	tPCB	reproduction	reproductive success	semi-field	50	oral	2 years	prey (fish)	0.1	0.2		mg/kg		—		—	WW	Boon et al. 1987, Brouwer et al. 1989	Kannan et al. 2000	—
Otter	tPCB	liver	hepatic retinoids	semi-field	—	—	—	Liver	0.17			mg/kg	0.042			ng/g	WW	Smit et al. 1996, Murk et al. 1998	Kannan et al. 2000	Based on an EC1 concentration.
Otter	tPCB	liver	hepatic retinoids	semi-field	—	—	—	Liver	0.46			mg/kg	0.084			ng/g	WW	Smit et al. 1996, Murk et al. 1998	Kannan et al. 2000	Based on an EC90 concentration.
Mink	tPCB	reproduction	—	laboratory	—	—	—	Liver	2.03			mg/kg	11			pg/g	lipid	Heaton et al. 1995, Tillitt et al. 1996	Kannan et al. 2000	Lipid content of liver assumed to be 5 percent.
Marine mammals	tPCB	—	—	—	—	—	—	Liver			8.7	mg/kg		—		—	lipid	Kannon et al. 2000	Kannan et al. 2000	Threshold [ ] in liver (derived from geometric mean of 3 values for seal, otter, and mink).

TABLE 3-15
Total PCB Internal Exposure (Target-Organ Based) Benchmarks for Mammalian Receptors
Palos Verdes Shelf Ecological Risk Assessment

Test Species	Analyte	Endpoint	Effect	Study Type	Body Weight (kg)	Exposure	Duration	Tissue	NOEC	LOEC	Critical Threshold	Units	TEQ NOEC	TEQ LOEC	TEQ Critical Threshold	Units	Basis	Primary Reference	Secondary Reference	Comments
Mink	tPCB	reproduction	—	laboratory	—	—	—	Liver		44.4		mg/kg		324		pg/g	lipid	Heaton et al. 1995, Tillitt et al. 1996	Kannan et al. 2000	Lipid content of liver assumed to be 5 percent.
Mink	tPCB	reproduction	—	—	—	—	—	Liver		—		—			60	pg/g	WW	Tillet et al. 1996	Kannan et al. 2000	Or 1.2 ng/g lipid weight assuming 5 percent lipid in liver.
Mink	tPCB	reproduction	—	laboratory	—	—	—	Prey	15			ng/g	1.03			pg/g	WW	Heaton et al. 1995	Kannan et al. 2000	—
Mink	tPCB	reproduction	—	laboratory	—	—	—	Prey		250		ng/g			1.9	pg/g	WW	Tillitt et al. 1996, Restum et al. 1998	Kannan et al. 2000	—
Mink	tPCB	reproduction	—	laboratory	—	—	—	Prey		720		ng/g		19		pg/g	WW	Heaton et al. 1995	Kannan et al. 2000	—
Otter	tPCB	—	—	—	—	—	—	Whole body			50	mg/kg		—		—	lipid	Olsson and Sandegren 1983, Jensen et al. 1977	Kannan et al. 2000	Proposed as a critical level in the early 1980's.

Notes:

— information that was not reported or not pertinent
MATC maximum allowable tissue concentration
ECn effect concentration (n=percent affected)

The shaded study was chosen as the primary toxicity study.

3.2.3.2 Site-Specific Field Surveys

Several previous studies have been conducted in the SCB to investigate species population trends and impacts related to tDDT and tPCB contamination. Most of these studies were completed to support NRDA efforts.

Birds

The most devastating example of reproductive failure in brown pelicans occurred on Anacapa Island in southern California in 1969. Eggs collapsed under incubating birds with only five young produced from 1,300 nesting attempts (Risebrough et al., 1970). DDE concentrations as high as 2,500 mg/kg lipid weight (about 145 mg/kg WW) were measured in eggs with the thinnest shells from brown pelicans on Anacapa Island.

Relatively high levels of tDDT persist in the environment, and levels within the brown pelicans remain at a low chronic level (BioSystems 1994). Widespread contamination by tDDT also contributed historically to the decline of the bald eagle population (Garcelon 1997). Currently, DDE levels continue to be of concern for successful natural reproduction of bald eagles. Both eggshell thinning and abnormal eggshell structure, leading to excessive water loss and subsequent dehydration of the eggs, are identified as the cause of reproductive failure among bald eagles. For the Santa Catalina Island population, eggs must be artificially reared in the laboratory for chicks to hatch successfully, and the hatching success is poor (Peakall 1994; Garcelon 1997). From 1989 to 1993, only three of 17 (18 percent) artificially incubated eggs were successfully hatched. All the unhatched eggs were analyzed and had DDE concentrations significantly above the critical threshold (3 to 5 mg/kg WW) for successful reproduction in bald eagles. Wiemeyer et al. (1993) found that concentrations of DDE at 20 to 30 mg/kg WW in eggs results in 0.1 eaglet per active nest, which is less than required to keep the population from declining.

As a result of eggshell thinning associated with organochlorine (e.g., DDE) contamination and its subsequent bioaccumulation into falcon tissues, the peregrine falcon population began to decline in the late 1940s. As with the bald eagle, peregrine falcons continue to lay thin-shelled eggs, and artificial manipulation is necessary to ensure successful hatching (BioSystems 1994; Peakall 1994). Peregrines on the Channel Islands remain highly contaminated with DDE. In 1992, the mean concentration of DDE in 16 eggs from 7 clutches was 17.4 mg/kg WW (Peakall 1994). Much of the peregrine falcon's prey acquires its body burden from the SCB. Other available site-specific field investigations are summarized below.

The estrogenic effects of DDT to seabirds in the SCB were described by Fry (1994). This study addressed the long-term injury birds sustained from DDT and its metabolites as well as the estrogenic effects of tDDT on the western gull. Fry developed experiments to test the theory that 2,4'-DDT causes abnormal development of the reproductive system of gull embryos (Fry and Toone 1981). Those experiments, along with field observations, led to the conclusion that tDDT in the SCB caused a multigenerational decline in western gulls.

Fry (1994) presents extensive field data suggesting that gulls and other seabirds in the SCB were highly contaminated with DDT and DDE residues through the 1960s and early 1970s. Additionally, extensive eggshell thinning in brown pelicans and double-crested cormorants led to complete reproductive failure in the 1960s and early 1970s. Gulls, however, are relatively insensitive to eggshell thinning, so they laid eggs with high contaminant levels;

these eggs did not break. Consequently, gull embryos were exposed to high DDT and PCB levels during development. This had a delayed effect on population impacts because gulls were still hatching and population levels did not decrease as significantly as those of the brown pelicans and double-crested cormorants.

Ultimately, the population-level effect in gulls came many years after the pelican and cormorant populations had already begun to recover. Additionally, an increase in supernormal clutches (twice the number of eggs), female-female pairing, and polygamous trios among breeding pairs were observed in gull populations. The relationship between estrogenic effects and unusual breeding behavior was highly correlated. Experimental studies on bird embryos exposed to high levels of estrogens have shown feminization of the male reproductive tract or demasculinization of the developing brain.

Both feminization of male chicks and alterations in the anatomy of female chicks occurred when gull eggs were dosed with DDT (Fry and Toone 1981; Fry et al. 1987). Both the field data and the experimental data support the theory that there is a causal relationship between DDT and second-generation breeding failure in gulls, and that reproductive failures and abnormal sex ratios of the breeding population of western gulls in the SCB were a direct result of embryonic development caused by organochlorine pollutants. Welsh et al. (2001) report DDE concentrations in 23 ashy storm petrel eggs collected from 1995 to 1997. Three eggs that failed to hatch were collected in 1995 and 10 randomly collected eggs each were collected in 1996 and in 1997. Average DDE concentrations for 1995, 1996, and 1997 were 16.0, 11.0, and 8.60 mg/kg (WW), respectively. Although this appears to be a decreasing trend, only eggs that had failed to hatch were collected in 1995 (i.e., these eggs are likely to be the most contaminated) and all means and the maximum concentration (20 mg/kg WW) are similar to the 1991-1992 mean (8.33 mg/kg WW) and maximum (16.9 mg/kg WW). Therefore, DDE levels in ashy storm petrel eggs have changed little during the 1990s. Moreover, eggshells from 1995 to 1997 were 3.5 to 5.6 percent thinner than the historic mean. The authors concluded that these values for DDE and eggshell thickness are not expected to result in large-scale reproductive failure, but the success of some individual birds may be reduced.

Gress (1994) reviewed the literature on reproductive performance, eggshell thinning, and organochlorines in brown pelicans and double-crested cormorants in the Channel Islands. Unpublished data on pollutants and eggshell thinning in brown pelicans in the SCB are also presented.

Initial speculation that brown pelican populations were experiencing reproductive problems came from shipboard surveys in 1968 (Gress 1994). Nesting activity was found to be limited based on historical observations. Visits to Anacapa in 1969 suggested that breeding efforts had almost completely failed. Eggshells collected were up to 50 percent thinner than pre-1947 specimens. Reproductive failures were also noticed farther south on Los Coronados, just south of the SCB, off northern Baja California, Mexico.

In 1970, studies were initiated to determine the cause and impact of the reproductive failure (Gress 1994). As part of those studies, long-term conservation measures and management plans were proposed and implemented. Low fledging rates were observed until the mid-1970s, when the discharge of DDT was reduced. In 1974 and 1975, breeding efforts and performance improved and was attributed to reduced input of DDT, recruitment of first-time breeders from more successful colonies outside the SCB that were not impacted by

pollutants, large increases in the abundance of northern anchovies (a prey species for pelicans) in SCB water, subsequent decrease of DDE in the marine food web, and lower levels of DDE in brown pelicans than in previous years.

Double-crested cormorants, like the brown pelican, were severely impacted by DDT, and reproductive levels were slow to improve even after the decrease in DDT discharge (Gress 1994). Since the mid-1980s, breeding numbers and productivity have increased and populations appear to have stabilized at somewhat higher levels. In 1992, double-crested cormorant eggs were 18.7 percent thinner than the pre-1947 samples.

DDE residues in both species showed a north-south gradient in concentrations with higher levels in the Anacapa colonies (Gress 1994). In 1978, more intensive breeding studies were performed on the brown pelican. Colonies were observed and eggs were collected and, subsequently, analyzed for organochlorines. The study suggested that as DDE sources were reduced, pelican egg concentrations also decreased, and shell thickness increased. The concentration, however, began to stabilize and the problem turned from acute to low-level chronic. Shell thickness did, however, continue to linger in the range that was known to affect populations. Population levels have varied since the banning of DDT because of other factors, including El Niño events and food shortages. While the main factor in pelican populations is now food availability, the presence of DDE continues to contribute to low fledging rates. This conclusion for brown pelicans reported by Gress (1994) was reconfirmed in a recent Agency Review Draft report prepared for the American Trader Trustee Council (California Department of Fish and Game, U.S. Fish and Wildlife Service, and National Oceanic and Atmospheric Administration). This report (Gress et al. 2003) presents the breeding data for California brown pelicans nesting at West Anacapa Island and Scorpion Rock from 1969 through 2002 (Table 3-16), as well as trends in these data from 1985 to 2002.

Kiff (1994) presented a synopsis of studies on avian eggshell thinning in birds nesting on the Channel Islands. Eggshell thickness was analyzed for a minimum of 20 clutches of pre-1947 museum eggshell specimens and compared with post-1947 eggshells from the Channel Islands. Reference eggs were also collected in 1992 from more northern Pacific coast localities. Brown pelicans and double-crested cormorants were included among the species sampled. Significant decreases in eggshell thickness were observed between pre-1947 eggshells and those collected in 1992 from the Channel Islands and other Pacific coast locations for some species. Less eggshell thinning was found in double-crested cormorant eggs from more northern localities than in southern locations. Eggshell thinning in double-crested cormorant, pelagic cormorant, Brandt's cormorant, brown pelican, western gull, and pigeon guillemot exceeded 15 percent in eggs collected from 1948 to 1969. In bald eagles, eggshell thickness of eggs collected between 1987 and 1993 was 12.5 percent less than pre-1947 eggs. Peregrine falcon eggs were collected in the years immediately following the introduction of DDT and averaged 19 percent thinner than pre-1947 eggs, indicating a greater than 50 percent chance of failure, usually from breakage. Recovery of the peregrine began soon after the ban of DDT, but their recovery in the Channel Islands lagged behind that of other areas.

Hunt (1994) studied peregrine falcons on the Channel Islands to monitor aerie reoccupancy and reproductive status, eggshell thinning, and levels of pollution, as well as to attempt to determine the pathways of contaminant exposure through the food chain. Surveys began in March 1992 to observe reproductive behavior, monitor eggs, collect egg fragments and prey remains, and collect intact eggs for artificial incubation. Collected eggs were replaced with

dummy eggs which were then replaced with artificially incubated hatchlings. In 1993, four breeding adult peregrines were captured and fitted with radio transmitters to determine winter diets as a way of assessing DDE exposure. Based on several observations, the following six conclusions were presented:

1. Substantial numbers of breeding and wintering peregrines existed in coastal southern California prior to 1945.
2. Peregrines had disappeared from the SCB by the early 1950s.
3. Peregrines did not begin to breed on the Channel Islands again until the late 1980s.
4. The reestablishment of nesting peregrines in the Channel Islands was largely, if not entirely, a result of the release program.
5. Large amounts of DDE must still exist in the diets of female peregrine falcons nesting in the Channel Islands (based on concentrations in eggs).
6. DDE levels measured in seabirds from the Channel Islands in 1992 and 1993 were sufficient to explain the observed rates of DDE contamination and thinning of peregrine eggs.

TABLE 3-16
Annual Mean Breeding Data for California Brown Pelicans Nesting in the Anacapa Island Area (West Anacapa Island and Scorpion Rock), California, 1969-2002
Palos Verdes Shelf Ecological Risk Assessment

Year	Nest Attempts[a]	Young Fledged	Productivity[b]	Percent Abandoned Nests	Young/ Successful Nest	Percent Chick Mortality
1969	750	4	0.005	nd	nd	nd
1970	552	1	0.002	nd	nd	nd
1971	540	7	0.013	nd	nd	nd
1972[c]	261	57	0.22	nd	nd	nd
1973	247	34	0.14	nd	nd	nd
1974[c]	416	305	0.73	nd	nd	nd
1975[c]	292	256	0.88	nd	nd	nd
1976	417	279	0.67	53.5	1.44	nd
1977	76	39	0.51	55.0	1.14	nd
1978[d]	210	37	0.18	88.1	1.48	nd
1979	1260	980	0.78	41.5	1.33	nd
1980	2150	1440	0.67	44.2	1.20	5.8
1981	2950	1810	0.61	52.4	1.29	20.5
1982	1860	1180	0.63	56.5	1.46	27.2
1983	1860	1150	0.62	42.2	1.07	39.0
1984	628	530	0.84	48.5	1.64	0.4

TABLE 3-16
Annual Mean Breeding Data for California Brown Pelicans Nesting in the Anacapa Island Area (West Anacapa Island and Scorpion Rock), California, 1969-2002
Palos Verdes Shelf Ecological Risk Assessment

Year	Nest Attempts[a]	Young Fledged	Productivity[b]	Percent Abandoned Nests	Young/ Successful Nest	Percent Chick Mortality
1985	5150	6390	1.24	22.0	1.59	12.7
1986	5910	3990	0.68	53.1	1.44	40.7
1987	6330	4060	0.64	52.8	1.36	33.6
1988	2720	2470	0.91	25.6	1.22	10.6
1989	4990	2880	0.58	59.6	1.43	8.6
1990	2200	650	0.30	78.7	1.39	27.1
1991	5770	1600	0.28	79.0	1.32	9.5
1992	1490	372	0.25	77.7	1.12	33.7
1993	3640	2390	0.66	48.7	1.28	16.4
1994	4910	1930	0.39	68.6	1.25	33.9
1995[e]	4800	nd	nd	nd	nd	nd
1996	5440	5530	1.02	32.2	1.50	17.0
1997	5500	33200	0.60	47.0	1.14	49.3
1998	2540	2210	0.87	40.0	1.45	3.4
1999	5290	3020	0.57	47.1	1.08	35.0
2000	3670	3310	0.90	33.2	1.35	6.2
2001	3180	2550	0.80	40.2	1.34	17.4
2002	6440	3220	0.50	35.7	1.08	53.6

Source: Gress et al. 2003.
[a] Estimates of numbers of pairs nesting represent a compromise between maximum number present, numbers of nests constructed, reproductive behavior, and appearance of secondary sexual characteristics.
[b] Productivity defined as number of young fledged per nest attempt.
[c] Nesting occurred on Scorpion Rock in 1972 (112 nests; 31 fledged), 1974 (105 nests; 75 fledged), and 1975 (80 nests; 74 fledged); Scorpion Rock is located off northwestern Santa Cruz Island approximately 6 nautical miles west of West Anacapa Island.
[d] Probable renesting in 1978; 210 pairs built 340 nests.
[e] Based on nest counts at end of breeding season; young not censused in 1995.
nd no data available

Based on these conclusions, Hunt (1994) stated that it was too early to determine whether full recovery of the peregrines on the Channel Islands will occur without continued restoration efforts. He suggests that yearly surveys were warranted, as well as continued monitoring of contaminant concentrations.

Effects of organochlorine compounds on the bald eagle population residing at Santa Catalina Island were studied by Garcelon (1994c). The report also presents contaminant concentrations found in bald eagles, as well as in their eggs and their food. All unhatched eggs collected from 1989 through 1993 had high levels of DDE, and exceeded the critical

threshold of 3 to 5 mg/kg for successful reproduction in bald eagles. Catalina eggs had the highest reported DDE concentrations in the United States. Eggs showed average thinning of 12.5 percent. Because bald eagle populations north and south of the SCB are producing normally, this appears to be a local phenomenon. Eagle carcasses found on the island had high DDE residue levels, among the highest concentrations in the United States.

Concentrations in blood of eaglets were also elevated, with a mean concentration of 0.029 mg DDE/kg (Garcelon 1994c). Concentrations of DDE in the bald eagle's bird-prey species were estimated based on DDE concentrations in eggs of the prey species. Results indicated that average levels were 7.4 mg/kg DDE based on egg-to-body conversions. Actual tissue measurements resulted in an overall mean of 5.94 mg DDE/kg.

The mean concentration of DDE in the gulls from around Catalina Island exceeds the dietary level (2.8 mg/kg) found to cause significant eggshell thinning in American kestrels (Garcelon 1994c). In fish, mean tDDT levels were 0.06 mg/kg. Carcasses of California sea lions were found to have a mean DDE level of 3 mg/kg with a maximum of 23.6 mg/kg. Overall, concentrations of tDDT in bird and marine mammal components of the bald eagle's diet on Catalina Island are high and, in addition to the fish concentrations, are high enough to cause the observed reproductive failure (Garcelon 1994c). Both intervention and management strategies are required to sustain existing populations, because natural production was not occurring as of 2002 (Sharpe 2003). Breeding data for 1989 to 2002 (Table 3-17) indicate that bald eagle pairs are able to lay eggs and fledge young; however, the eggs generally fail to hatch, even though all eggs are collected and incubated by researchers. For example, in 2002, only 2 chicks were successfully hatched from the 7 eggs collected, but both of these chicks were successfully fledged by the foster adults (Table 3-17).

Marine Mammals

Female California sea lions at San Miguel Island with elevated tDDT levels had pathological changes in the uteri, implantation failure, and early abortion. During the 1970s, these reproductive effects in sea lions were linked to tDDT exposures. Surveys of California Channel Island sea lion populations found that nearly half of the pups in each breeding season died after premature births. Populations of sea lions have recovered since the 1970s, and tDDT concentrations in sea lion blubber have declined. Long-term effects from lower-level chronic exposures are still being studied.

In support of NRDA efforts, Calambokidis and Francis (1994) researched marine mammal exposure to tPCB and tDDT contamination in the SCB. Three groups of marine mammals were analyzed—odontocetes, mysticetes, and pinnipeds and sea otters. The report presents distribution and occurrence in the SCB, population size and trends, strandings, contaminant concentrations, and feeding habits of each identified species. Conclusions about each receptor were made regarding their exposure potential in the SCB.

Bottlenosed dolphins were assumed to have high exposure to tDDT based on their distribution and feeding habits (Calambokidis and Francis 1994). The common dolphin and killer whales were also expected to be exposed. Exposure by other odontocetes was either low or unknown. In general, receptors in the mysticetes group were not expected to be exposed, or exposure was low. Pinnipeds were the most highly exposed family with highest concentrations in the California sea lions, though harbor seals also have high levels. Those marine mammals that are common in the SCB (e.g., the California sea lion, common dolphin, bottlenose dolphin, and harbor seal) showed high tissue levels of both tDDT and tPCB.

TABLE 3-17
Summary of Bald Eagle Egg and Chick Manipulations on Santa Catalina Island, 1989-2002
Palos Verdes Shelf Ecological Risk Assessment

	Year													
	1989	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**
No. of Active Nests	1	2	2	3	3	2	3	3	2	3	4	3	4	4
No. of Eggs Laid	2	2-3	3	5	5-6	3	5	5-6	6	7	8	7	8	8
No. of Eggs Collected	1	1	3	5	4	3	5	4	5	6	6	4	7	7
No. of Catalina Island Eggs Hatched[a]	0	0	1	2	0	0	0	0	1	1	1[i]	2[j]	0	2
No. of Eggs Fostered into Nests on Catalina Island	0	0	2[b]	0	0	0	1	0	0	0	0	0	0	0
No. of Chicks Fostered Into Nests on Catalina Island	1	0	0	3	2[c]	2[d]	1	5[e]	1	4[g]	3[h]	4	5	7[j]
No. of Chicks Fledged From Nests on Catalina Island	1	0	2	3	1	1	1	2	1	3	2	4	5	6
No. of Eagles Hacked Onto Catalina Island	0	0	2	0	2	0	2	5[f]	0	4	2	0	4	0
No. of Island-Produced Eagles Breeding on Island	0	0	0	0	0	0	0	0	1	1	2	1	2	2
No. of Second Generation Eagles Fledged	0	0	0	0	0	0	0	0	1	0	0	0	0	0

Source: Sharpe 2003

[a] Hatched by the Santa Cruz Predatory Research Group (1991) or San Francisco Zoo (1992-Present).
[b] Both hatched.
[c] One chick died of asphyxiation on plastic bag.
[d] One chick died during severe storm.
[e] One injured chick was euthanized, one injured chick was treated and placed on hack tower, and one chick was killed by a red-tailed hawk.
[f] Includes a one-year-old rehabilitated eagle and an eaglet that was removed from a nest because of a leg injury.
[g] One chick killed by nesting female upon return to the nest following fostering.
[h] One chick disappeared from nest under unknown circumstances.
[I] Hacked by Ventana Wilderness Sanctuary in central California.
[j] One chick died in nest from unknown causes.

Lieberg-Clark et al. (1995) compared DDT concentrations in California sea lions collected along the coast of California in 1970 and 1992. Analysis of blubber concentrations indicated that tDDT in blubber from 1992 samples were greatly reduced from concentrations in 1970. High tDDT concentrations reported in 1970 were associated with reproductive problems, according to DeLong et al. (1973) and Gilmartin et al. (1976). Over the past 18 years, California sea lion populations were estimated to have increased by 133 percent. The California seal lion increase is attributed to the reduction in tDDT contamination at the sea lion breeding areas in the SCB, although a cause-effect link has not been unequivocally established between the decreased tDDT residues in sea lions and the observed population increase of the SCB.

O'Shea and Brownell (1998) studied the California sea lion populations and DDT contamination in southern California. The objective of that study was to evaluate the conclusion that the decrease in tDDT residues in California sea lions associated with the observed population increase, as presented in Lieberg-Clark et al. (1995). This paper aimed to address the following questions:

- Are the data reported in Lieberg-Clark et al. (1995) presented with a sufficiently thorough analysis to fully support a conclusion of declining tDDT residues in California sea lions?
- Is there supporting evidence unequivocally linking DDT exposure with reproductive impairment or mortality in sea lions or pinnipeds of any species?
- What is the strength of the previously reported reproductive problems in this species?
- What changes in sea lion population sizes have occurred over a broader time frame than that reported in the recent study?
- What other factors may have occurred over the period of DDT decline that may also affect population growth?

Data collected in 1970 for 12 males were compared to data collected in 1988 through 1992. Differences in tDDT concentrations were large, suggesting that changes had, in fact, occurred. This argument, however, could be strengthened if sample sets were larger and included sea lions of the same age and sex, as well as presenting DDT concentrations in prey species and stomach contents. There is no unequivocal experimental evidence that links DDT exposure and reproductive effects in pinnipeds. Additionally, some field studies suggest that reproductive effects were evident before the release of DDT. Other complications come into play when assessing field studies, including the age of the female and reproductive history, because female sea lions transfer most of their body burden to their first-born pup. So many confounding factors exist when interpreting studies on disease, organochlorine concentrations, and reproductive problems in California sea lions that it is difficult to attribute reproductive effects to DDT concentrations. O'Shea and Brownell (1998) suggest that populations of California sea lions were "generally increasing or stable before, during and after commercial development, manufacturing and use, and cessation of discharges of industrial DDT wastes off southern California." Some studies suggest that populations actually increased in southern California throughout the century, including times with high DDT concentrations. Other factors that are potentially attributable to changes in sea lion populations include federal protection as well as immigration of species from Mexico. Based on the confounding factors associated with determining the

impacts of DDT on the California sea lion population, the authors suggest that further research be conducted to supplement the previous inconclusive studies.

Three recent studies (Kajiwara et al. 2001, Le Boeuf et al. 2002, and Le Boeuf 2002) present contaminant concentrations in sea lions found dead along the California coast. These data represent various age and sex classes. One study only has animals collected in central and northern California (Kajiwara et al. 2001), one is all male or unknown sex animals (Le Boeuf 2002), and the last (Le Boeuf et al. 2002) has overlap of samples with Le Boeuf (2002) and only one adult female from southern California. Kajiwara et al. (2001) include 15 samples collected between 1991 and 1997. Six these are from adult females and blubber concentrations are only available for three (liver values are presented for the other three). Total DDT concentrations in these three females were 60, 130, and 140 mg/kg (lipid weight) and PCB concentrations were 18, 39, and 840 mg/kg (lipid weight), respectively in these samples. Adult males tended to have higher concentrations with some having tDDT levels as high as 2,300 µg/g (lipid weight [LW]) and tPCB up to 1,300 mg/kg (lipid weight).

Le Boeuf et al. (2002) present tPCB and tDDT data for 36 sea lions collected in 2000. Of these, only six are adult females and only one of these was found in southern California. This one female had a tDDT blubber concentration of 4.1 mg/kg lipid weight, which is quite low compared to those females collected in central California (range tDDT 39 to 1400 mg/kg lipid weight). As indicated by the authors, these females from central California were likely non-lactating females as breeding females remain near the rookeries in southern California throughout the year. The authors also note that although mean levels of tDDT are much lower in 2000 compared to 1970, the inter-annual variation remains high and several individuals from 2000 had concentrations within the 1970s range.

Le Boeuf (2002) studied sea lions utilizing Santa Catalina Island during February to November 2000 and report a maximum of 70 animals and a minimum around 35 animals. In general, these were mostly adult males (64 percent) and no pups were observed. Eight dead sea lions were observed during this period. Of these, blubber concentrations are reported for six (three of these also were reported in Le Boeuf et al. 2002), all of which were males or unknown sex. Total DDT concentrations in blubber samples from these carcasses ranged from 17.5 to 150 mg/kg (lipid weight) or 4.0 to 32.1 mg/kg (WW); however, the highest concentration on a lipid weight is difficult to interpret because the percent lipid in this animal was dramatically lower than the other samples (1.8 percent compared to 21.9 to 64 percent). Le Boeuf (2002) reports that bald eagles did not forage on these carcasses, but other researchers (Garcelon 1994a) have observed bald eagles foraging on sea lion carcasses in the past.

3.2.3.3 Site-Specific Modeling Efforts

Swartzman (1994) modeled effects of DDE on peregrines and bald eagles residing in the SCB, and HydroQual (1997) conducted a food-pathways analysis to determine whether double-crested cormorants, peregrines, and bald eagles were exposed to the higher contaminant concentrations in fish from the PVS.

Swartzman (1994) used an individual-based population dynamic model to simulate effects of DDE on Channel Island peregrine falcon and bald eagle populations. The simulation presents population trends from 1945 to 1994, with and without the effects of DDE. The effect of DDE on breeding success is portrayed in this model for bald eagles using DDE egg

concentrations and for peregrine falcons using eggshell thinning and average fecundity. The model was run for 50 years, assuming 4 breeding pairs and 4 subadults for both the eagle and the falcon. Results imply that no recovery of either species would have occurred on the Channel Islands without the eagle- and falcon-fostering programs. When compared with actual numbers of breeding falcon pairs observed in 1992, the model underestimated the number of mature falcons for 1992. Estimates from the model suggest that without DDE effects, populations would be higher by about 100 birds.

HydroQual, Inc. (1997) presented a food-web/pathways study. The purpose of the study was to "provide further quantitative evidence as to whether the contamination currently observed in the animals not directly exposed to the sediment plume may have originated from sources other than the Palos Verdes Shelf sediments" (HydroQual, Inc. 1997, page 1-4). Conclusions of this analysis were:

1. Measured concentrations in cormorant eggs from birds breeding on Anacapa Island were less than the computed egg levels, indicating that these birds spend less time than hypothesized feeding in the contaminated regions of the SCB. However, measured levels were greater than levels predicted from foraging in exclusively low contaminated areas; therefore, some exposure to PVS-contaminated prey is required to account for the measured egg residues in cormorants.

2. It was found that 78 percent of the contaminant dose received by peregrine falcons and 90 to 95 percent of the dose to bald eagles originated within the SCB.

3. Concentrations measured in fish collected for a low contamination area within the SCB (Santa Catalina Island) are "insufficient to account for the 4,4'-DDE and tPCB concentrations measured in the female sea lions" (HydroQual, Inc. 1997, page 1-6). However, prey contaminant concentrations in regions closer to the outfall are sufficient to account for measured blubber concentrations in female sea lions from San Miguel Island.

These results support a link between tDDT and tPCB concentrations in PVS sediment, and tissue concentrations in female California sea lion blubber, as well as in double-crested cormorant, peregrine falcon, and bald eagle eggs.

SECTION 4

Risk Characterization

Risk characterization is an evaluation of the evidence linking exposures to COPECs with their potential ecological effects for the representative species identified for the PVS and SCB study areas. This evaluation is completed through the integration of information gathered in the problem formulation, the results of the analysis, and other available lines of evidence. For this ERA, the evidence evaluated consisted of measured and predicted chemical concentrations in abiotic media (sediment, porewater and/or surface water), measured and modeled concentrations in biota (tissues and target organs), site-specific and literature toxicity information, field surveys, exposure and risk estimates for representative species, and results of the Food Web Exposure Model (Appendix C). There are three main components that comprise the risk characterization: risk estimation, risk description, and uncertainty analysis.

- The risk estimation quantifies potential risks associated with each combination of exposure and effects data presented in the analysis phase. Depending on the receptor group and the measures of effects that were evaluated, one or more risk estimates were derived**.** Risk estimates were evaluated for all receptor groups based on external exposures; in addition, internal pathway risk estimates were evaluated for fish, birds, and mammals.
- The risk description evaluates each of the different risk estimates (or lines of evidence) available for a given representative species to establish a weight-of-evidence for determination of chemicals of ecological concern (COECs). This includes identification of the COECs, which are the risk drivers (result in the greatest potential risk) and the representative species most at risk based on the evidence available. This section also discusses how the lines of evidence relate back to the assessment endpoints.
- The uncertainty analysis summarizes the uncertainties and limitations encountered in the ERA process, including those related to the problem formulation, analysis, and risk characterization. The overall impact of identified uncertainties and limitations is qualitatively evaluated and data gaps are identified.

These three components are used together to identify the final COECs for each receptor group of this assessment.

4.1 Risk Estimation

Risk estimation focuses primarily on quantitative methods to evaluate the potential for risk. For this ERA, these consisted of calculation of hazard quotients (HQs). Hazard quotients were developed for two types of comparisons using the indicated equations:

- Direct comparisons of measured concentrations in sediment, surface water, or tissue to the respective reference toxicity values (RTVs) for each COPEC. These comparisons were conducted for benthic macroinvertebrates, and fish for external exposures to sediment or surface water. In addition, this type of comparison was made for internal exposures (tissues and target organs) for fish, birds, and mammals.

$$HQ = \left(\frac{Exposure\ concentration\ (mg/kg\ or\ \mu g/L)}{RTV\ (mg/kg\ or\ \mu g/L)} \right) \qquad \text{[Equation 4.1]}$$

- Comparisons of estimated total exposure dosages via the food chain uptake model to effects dosage RTVs. These comparisons were conducted for birds and mammals exposed to sediment, surface water, and food.

$$HQ = \left(\frac{Exposure\ dosage\ (mg/kg\ BW/d)}{receptor-specific\ RTV\ (mg/kg\ BW/d)} \right) \qquad \text{[Equation 4.2]}$$

The risk estimates were based on the combinations of assessment endpoint level, representative species, exposure medium, exposure point concentration, and RTVs that were developed in the problem formulation and analysis. An HQ greater than 1 (unity) indicates potential adverse ecological effects. COPECs with HQs exceeding 1 were retained for further evaluation in the risk description.

4.1.1 Benthic Macroinvertebrates

4.1.1.1 Estimation of Effects Based on Exposure to Sediments and Porewaters

Two lines of evidence were used for this assessment endpoint: (1) comparison of COPEC concentrations in sediment with effects-based benchmark values for protection of benthic organisms; and (2) comparison of predicted porewater concentrations with water quality benchmark values derived from invertebrate toxicity results. The SEC-derived benchmark concentrations were developed for the PVS (MacDonald 1997), and PVS-wide percent total organic carbon (TOC), tDDT, and tPCB concentrations in surface sediments were available to compare TOC-normalized contaminant concentrations to the benchmarks.

Bulk sediment concentrations of COPECs were compared to SECs to compute HQs by normalizing the raw sediment values to 1 percent TOC in the following manner:

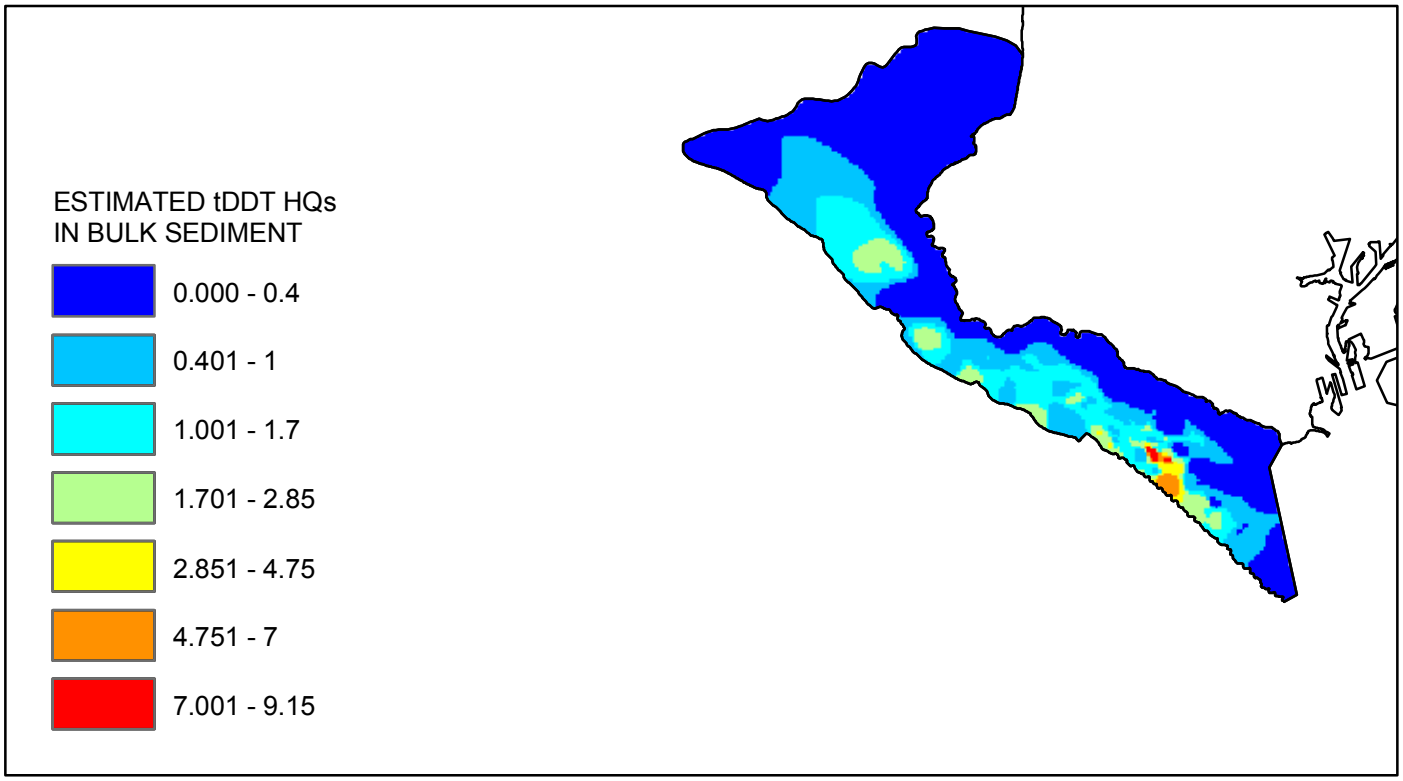
HQ (tDDT shallow sediment) = (tDDT [mg/kg]/TOC[%])/DDT-SEC [mg/kg]

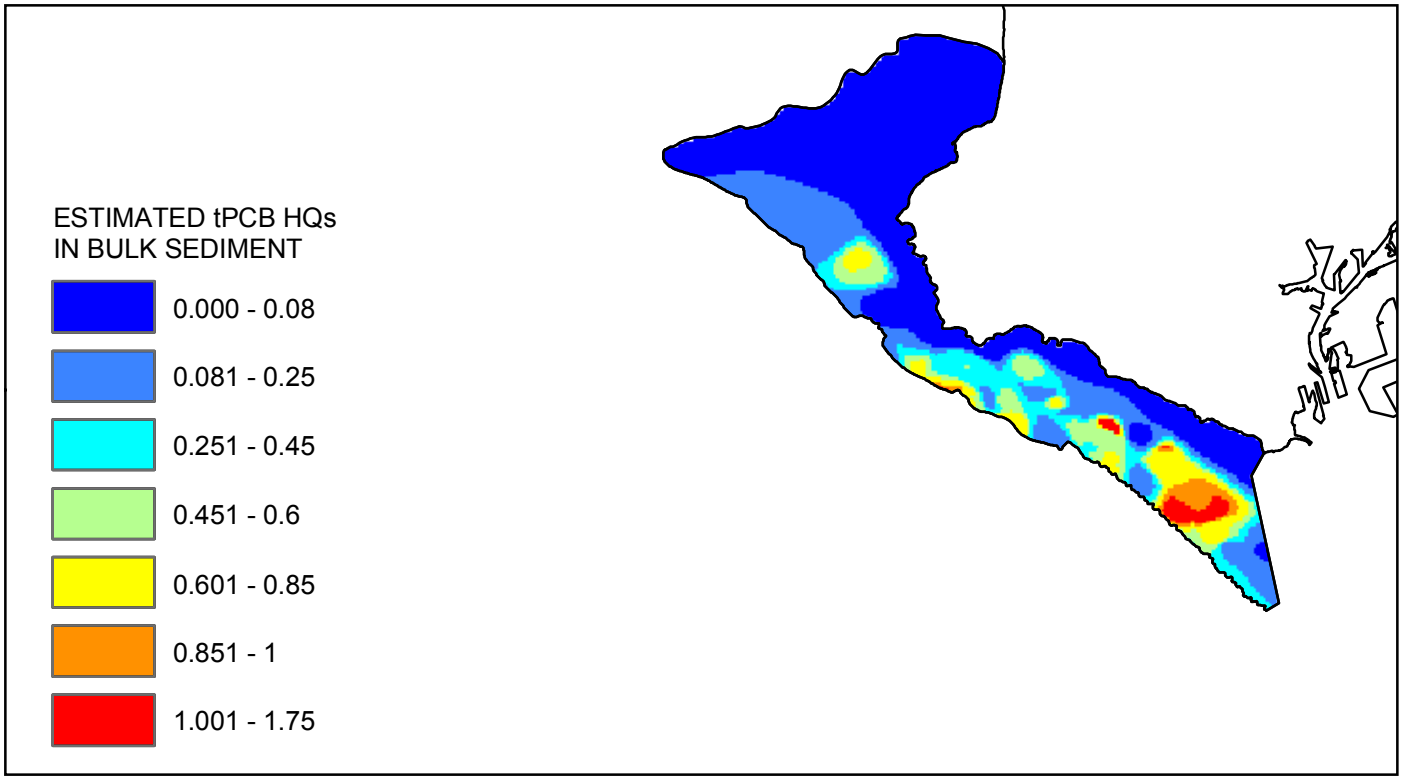
HQ (tPCB shallow sediment) = (tPCB [mg/kg]/TOC[%])/PCB-SEC [mg/kg]

Where the tDDT and tPCB SEC values for 1% TOC are 2.0 and 0.577 mg/kg, respectively.

The distributions of surface (0-15 cm) and deeper (15 – 30 cm) sediment concentrations are shown in Figures 2-3 through 2-5. As described in Section 3.1.2, shallow sediments were chosen as being most appropriate for this ecological risk evaluation. The COPEC-specific HQs that depict the relative exceedance of SEC benchmarks by area of the PVS are shown in Figure 4-1. The greatest risk due to exposure of benthic communities to the COPECs, based upon exceedance of benchmark values, exists in the area of the outfalls and to the northwest on the shelf. The elevated TOC in the sediments closest to the outfalls mitigates, in part, the elevated concentrations of tDDT and tPCB in that same general area and serves to reduce the relative toxicity of the COPECs.

Shallow sediments at PVS exceed sediment quality benchmarks (HQ > 1.0) over 21 percent of the PVS area for tDDT but less than 2 percent for tPCB (Table 4-1). The pattern also is reflected in the comparison to the 95 percent UCL value, which exceeds the benchmark for DDT but not for PCB. In contrast, deeper sediments are relatively more contaminated with COPECs and 95 percent UCLs for both tDDT and tPCB exceed the benchmarks.





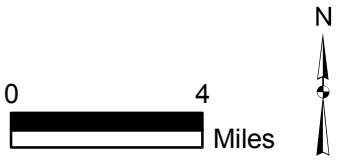


FIGURE 4-1
BULK SEDIMENT
HAZARD QUOTIENTS FOR
THE PVS STUDY AREAS TO 200 METERS
PALOS VERDES SHELF ECOLOGICAL RISK ASSESSMENT

TABLE 4-1
Minimum and Maximum HQs and Percentage of PVS Study Area with Shallow Sediments Exceeding Bulk Sediment Benchmarks
Palos Verdes Ecological Risk Assessment

Characteristic	Parameter	
	tDDT	tPCB
Minimum HQ	< 0.0001	0.0002
Maximum HQ	9.15	1.74
Percent of area with HQ > 1.0	21.77	1.80
Percent of area with HQ > 5.0	0.76	0

TABLE 4-2
Characteristics of PVS Study Area for Estimated Shallow Sediment Porewater Concentrations and Exceedances of Water Quality Benchmarks
Palos Verdes Ecological Risk Assessment

Characteristic	Parameter	
	tDDT	tPCB
Chronic water quality benchmark (μg/L)	0.0072	0.03
Minimum estimated porewater concentration (μg/L)	< 0.0001	< 0.0001
Maximum estimated porewater concentration (μg/L)	0.697	0.0373
Percent of PVS study area exceeding benchmark	79.13	0.20

HQs were calculated for the contoured sediment porewater concentration by comparison of the estimated porewater concentrations to water quality benchmarks for chronic exposure to marine invertebrates. Results are presented as the relative areal frequency of exceedance of benchmarks from the spatially contoured data.

Spatially averaged values from the plot show that the PVS study area shallow sediment porewater concentrations exceed benchmarks for tDDT over 79 percent of the area and tPCB for less than 1 percent of the area (Table 4-2) The highest exceedance values for both COPECs are in the immediate area of the outfalls. Relatively less contaminated areas are along the shoreline and to the north of Palos Verdes Point.

The patterns of porewater benchmark exceedances and degree of benchmark exceedances are similar to those for bulk sediments, indicating the comparability of the bulk sediment and water quality benchmarks in describing potential toxicity to benthic invertebrates. Porewater benchmark exceedances, as a line of evidence, suggest a high likelihood that concentrations of tDDT in porewater are sufficient to cause adverse effects to sensitive marine infaunal organisms. As evidenced from their low degree of benchmark exceedances for bulk sediment and porewater, tPCB appear less likely to pose a significant risk in surface sediments. Note that estimated porewater concentrations of tDDT also exceed the acute water quality criteria of 0.13 μg/L over 7.08 percent of the PVS study area (near the outfalls).

4.1.1.2 Effects Analysis—Benthic Community Analysis and Structure

Evaluation of observed effects at the PVS is an important element of understanding the link between predicted effects based on exposure to sediment and porewater (presented above) and estimated risks to benthic communities. The following evidence is used to characterize benthic community effects: comparison of site benthic community indices (species richness, diversity, evenness, similarity, and infaunal trophic index) with the same benthic community indices from a reference area, and direct evidence of sediment toxicity.

Infaunal species abundance data obtained by LACSD at various stations in the PVS were analyzed to evaluate changes in benthic community structure across the site and potentially detect environmental stress related to anthropogenic disturbance. For the present analysis, comparisons of community structure changes within the study area relative to reference locations outside the study site (i.e., as shown in USEPA 2000) were used as the basis for the assessment. In recognition of the fact that organic carbon content and depth may have significant effects on community composition, additional analyses were conducted in an attempt to evaluate these effects in relation to those due to potential impacts from COPECs.

Benthic Community Assessment Methods

Two complementary multivariate techniques were used to provide an "exploratory" analysis of variation in benthic community data with respect to abiotic environmental conditions to better isolate potential COPEC-related distributions (USEPA 2000). A four-level scoring scheme for evaluation of habitat and metric-specific data was developed following USEPA Rapid Bioassessment Protocols for use at PVS (USEPA 2000). The distribution of data was divided into "quarters", based on the range of data for a specific metric. After expressing the station-specific metric value as a percent of the corresponding reference location, each station/metric-specific value was assigned a number of "points" based on its relation to the corresponding reference location (USEPA 1989) as follows:

- equal to or greater than the reference value = 6 pts
- between 90 percent of and equal to reference value = 4 pts
- between 65 and 90 percent of reference value = 2 pts
- < 65 percent of reference value = 0 pts

A benthic effect quotient (BEQ) was calculated as the ratio of the sum of points for each station to total possible points (set = 30 [five metrics x 6 pts]). Normalization to the reference value and overall scoring is performed to facilitate expression of the combined metric characterization of the benthic community. Calculated metric values and BEQs are discussed in the following section. Station-specific sediment quality characteristics are shown in Table 4-3.

TABLE 4-3
Physical Characteristics of PVS Sediment
Palos Verdes Ecological Risk Assessment

Source[a]	Station	Depth	Mean % Fines (1994, 1997)	Mean % Organic Carbon (1994, 1997)
LACSD	5D-6D-7D	30 m	30, 37	0.9, 1.0
SCCWRP	899-1312-1328-1317	30 m	34	0.46
LACSD	10D	30 m	21, 28	0.7, 0.9
LACSD	1C-3C-5C-6C-7C-8C-9C	60 m	62, 62	3.0, 2.6
SCCWRP	136-232-1667-1892	60 m	62	1.12

TABLE 4-3
Physical Characteristics of PVS Sediment
Palos Verdes Ecological Risk Assessment

Source[a]	Station	Depth	Mean % Fines (1994, 1997)	Mean % Organic Carbon (1994, 1997)
LACSD	10C	60 m	13, 26	0.6, 0.6
LACSD	3B-5B-6B-7B-8B-9B	150 m	71, 82	3.8, 3.8
SCCWRP	1469-1551-1662-1908	172 m	69	1.1
LACSD	10B	150 m	47, 50	1.3, 1.3

[a] LACSD data are for summers of 1994 and 1997, to represent range of conditions. SCCWRP data are from Schiff and Gossett 1997.

Benthic Community Assessment Results

Three primary "groups" of stations were distinguished in the USEPA (2000) study: deep (B), shelf (C), and shallow water (D) communities. There was a strong influence of water depth (and associated habitat difference) in explaining the observed similarities and differences in benthic communities among stations (Table 4-4). In contrast, sediment TOC is more variable and does not account for large differences among the groups. This is because the ranges of TOC values within groups were similar to differences between groups. Thus, the analyses confirm that there was a strong depth-related dependency, which requires the separation of communities into depth-specific groups for further analysis of potential COPEC-related benthic community impacts (USEPA 2000).

Trends in the BEQ data suggest a more balanced indigenous community at the reference location than at various stations within the study area (USEPA 2000). Thus, while it cannot be determined that communities at the reference locations are free of site- (or non-site) related impacts (sediment chemistry data were not available), the data suggest that community attributes (diversity, richness, evenness, trophic structure) at reference locations serve as plausible benchmarks for discerning possible site-related impacts (USEPA 2000).

The pattern of BEQ results shows affected communities in the general vicinity of outfalls and less-affected stations in the northern and western portions of the study area. The results show BEQ scores less than 0.4 for the 60-meter station 9C and deep-water stations 7B, 8B, and 9B (Table 4-5). Stations with a BEQ equal to 0.4 are 6C (60-meter), 5B, and 6B (deep-water). Remaining stations with a BEQ equal to 0.5 to less than 0.6 were stations 5D, 3C, 8C, and 3B, while a BEQ greater than or equal to 0.6 was observed for stations 6D, 7D, 1C, 5C, and 7C.

Smith et al. (1999) described the development of a quantitative index applicable throughout the Southern California coastal shelf environment for the measurement of the condition of benthic assemblages. The BRI is based upon the relative abundance of pollution-tolerant species in the assemblage, and community response is defined in the following five categories: similar to reference; marginal deviation; loss of biodiversity; loss of community function; and defaunation.

BRI values for LACSD community data collected from 1990 to 1996 are shown along with the scale of impact in Table 4-5. Community attributes similar to reference or slightly different were found at the western-most stations, while the highest level of impact, loss of community function, was observed at the 60- and 150-m stations on transect 8 (adjacent to the outfall). Moderate impacts, i.e., loss of biodiversity, were found at all other stations for which these analyses were completed.

TABLE 4-4

Benthic Community Metrics and Derived Effects Rankings for Benthic Invertebrates

Palos Verdes Ecological Risk Assessment

	Shallow Stations						Reference Station	Shelf Stations (60m)														Reference Station	Deep Stations (150m)												Reference Station
	5D		554/6D		7D		10D	514/1C		522/3C		550/5C		556/6C		7C		564/8C		574/9C		10C	523/3B		5B		557/6B		7B		8B		9B		10B
Benthic Community Metrics	**Metric**	**Rank**	**Metric**	**Rank**	**Metric**	**Rank**	**Metric**	**Metric**	**Rank**	**Metric**	**Rank**	**Metric**	**Rank**	**Metric**	**Rank**	**Metric**	**Rank**	**Metric**	**Rank**	**Metric**	**Rank**	**Metric**	**Metric**	**Rank**	**Metric**	**Rank**	**Metric**	**Rank**	**Metric**	**Rank**	**Metric**	**Rank**	**Metric**	**Rank**	**Metric**
Total Number of Species	1.07	6	1.22	6	1.29	6	45.00	1.20	6	0.95	4	1.06	6	0.90	4	0.96	4	1.11	6	0.80	2	114.00	0.71	2	0.50	0	0.50	0	0.42	0	0.77	2	0.56	0	137.00
Margalef Species Richness/Ranking[a]	0.88	2	1.00	4	0.20	6	7.86	1.14	6	0.93	4	1.04	6	0.88	2	0.96	4	1.04	6	0.79	2	18.00	0.75	2	0.61	0	0.61	0	0.47	0	0.73	2	0.59	0	20.00
Shannon-Wiener Diversity/Ranking[a]	0.70	2	0.76	2	0.90	2	2.87	0.96	4	0.89	2	0.96	4	0.85	2	0.92	4	0.87	2	0.83	2	4.13	0.93	4	0.94	4	0.94	4	0.70	2	0.79	2	0.82	2	3.90
Pielou's Evenness/Ranking[a]	0.69	2	0.73	2	0.84	2	0.75	0.92	4	0.90	4	0.95	4	0.86	2	0.93	4	0.85	2	0.87	2	0.87	1.00	6	1.09	6	1.09	6	0.85	2	0.84	2	0.94	4	0.79
Infaunal Trophic Index	0.97	4	0.99	4	1.00	6	76.00	0.94	4	0.89	2	0.82	2	0.83	2	0.82	2	0.64	0	0.85	2	84.00	0.70	2	0.71	2	0.67	2	0.74	2	0.72	2	0.72	2	79.44
Benthic Effect Quotient/Ranking[b]	0.53	+	0.60	-	0.73	-		0.80	-	0.53	+	0.73	-	0.40	++	0.60	-	0.53	+	0.33	+++		0.53	+	0.40	++	0.40	++	0.20	+++	0.33	+++	0.27	+++	

a Metric-specific scores expressed as decimal fraction of reference within each depth (B, C, D) using assigned points: Station value>reference value (Station 10B, 10C, or 10D) = 6 pts; station value/reference value >0.90 but <1.00 = 4 pts.; station value/reference value >0.65 but <0.90 = 2 pts.; station value <0.65 = 0 pts.

b Benthic Effect Quotient expressed as decimal fraction of total possible points (30).
Station value >0.60 = "-"; station value >0.50 but <0.60 = "+"; station value >0.40 but <0.50 = "++"; station value <0.40 = "+++".

Source: Data from LACSD (1992).

TABLE 4-5
Benthic Response Index Values Derived from an Analysis of Benthic Infauna Data Collected by LACSD, 1990-1996
Palos Verdes Ecological Risk Assessment

Station[a]	Year	Benthic Response Index	Scale of Impact
0A	90	31.5	+
0A	92	19.1	-
0C	90	22.1	-
0C	92	17.9	-
0C	96	15.2	-
1C	92	29.8	+
1C	96	23.3	-
5B	92	41.1	++
5C	92	41.0	++
5C	96	37.5	++
6B	90	41.2	++
6C	92	46.3	++
6C	92	40.4	++
6C	96	42.6	++
6D	92	35.9	++
7C	90	42.7	++
7C	92	41.1	++
7C	96	43.4	++
8B	92	45.5	+++
8C	92	49.0	+++
8C	96	43.7	++
9C	92	35.6	++
9C	96	41.3	++
9D	92	37.9	++

Source: Bay et al. (1998); Smith et al. (1999)
[a] OA and OC are north of Palos Verdes. OA, OC, and 9D are outside of the USGS footprint. Water depths are as follows: B stations at 152 m; C Stations at 61 m; and D Stations at 30 m.
- 90th percentile of Reference condition
+ Marginal deviation from Reference condition
++ Loss of biodiversity
+++ Loss of community function

4.1.1.3 Sediment Toxicity

The toxicity of PVS sediments was assessed at 11 sites throughout the PVS study area using tests for acute toxicity to amphipods and chronic toxicity to sea urchin fertilization and growth (Schiff and Gossett 1997). No acute toxicity was observed at any location (Table 4-6). However, chronic toxicity was documented at stations nearest the outfall. Statistical analyses

indicated possible causal relationships to hydrogen sulfide, tDDT and tPCB, polycyclic aromatic hydrocarbons (PAHs), and metals concentrations in the surface sediment. Further studies discounted the relative importance of tDDT as determining toxicity. Sulfides were likely to have contributed the greatest toxicity (Schiff and Gossett 1997).

TABLE 4-6
Results of Sediment Toxicity Tests
Palos Verdes Ecological Risk Assessment

Station ID		Test-Specific Endpoint			Inferred Chronic Toxicity[d]
LACSD	USGS	REPOX[a] Acute	NEJ[b] Chronic	REPOX[c] Acute	
1C	514	NT	-	-	–
3C	522	-	NT	NT	–
5B		-	-	-	–
5C	550	NT	-	-	–
6D	554	-	-	-	–
6C	556	NT	NT	NT	+
6B	557	-	-	-	+
7C		-	T	NT	+
8C	564	NT	T	NT	+
8B		-	-	-	+
9C	574	-	-	-	+

[a] REPOX—amphipod (*Rhepoximius abronius*) 10-day bulk sediment test results reported by Bay et al. (1994).
Test endpoint: T = (< 80% control response), NT = non-toxic (≥ 80% control response).
[b] NEJ—Polychaete worm (*Neanthes arenaceodentata*) 120d number of emergent juveniles in sediment exposures.
Test results reported by EVS (1994).
Test endpoint: T = toxic (< 80% control response), NT = non-toxic (≥ 80% control response).
Study also reported no effects at 120 d on adult survival; adult growth, nest success, time-to-maturity, number of eggs/nest or EJ production.
[c] REPOX—amphipod (*Rhepoxinius abronius*) 10-day bulk sediment test results reported by EVS (1994).
Test endpoint: T = toxic (< 80% control response), NT = non-toxic (≥ 80% control response).
[d] – = no toxicity, + = toxic.

4.1.2 Fish

Risk to fish was estimated in two ways, including internal tissue residues and external exposures. The percentage of tDDT and tPCB concentrations in tissue samples in the 1990-2003 study period exceeding the NOEC and LOEC are presented in Table 4-7. HQs for external exposures to tDDT and tPCB in the water column are presented in Tables 4-8 and 4-9, respectively. White croaker at the PVS had the highest percentage of tissue concentrations that exceeded the NOEC and LOEC. Eighty-nine percent of white croaker tissue samples collected at the PVS study area exceeded the NOEC for tDDT and 30 percent exceeded the LOEC. Fifty-five percent of Dover sole tissue samples collected at the PVS study area exceeded the tDDT NOEC but less than one percent exceeded the LOEC. Fifteen percent of kelp bass tissue samples collected at the PVS study area exceeded the NOEC for tDDT and less than one percent exceeded the LOEC. Total DDT in 71 percent of tissue samples from sanddabs collected from the SCB exceeded the NOEC, whereas none exceeded the LOEC. None of the tissue samples for the four fish receptors collected at the PVS study area exceeded the tPCB NOEC or LOEC.

TABLE 4-7
Percentages of Benchmark Exceedances and Hazard Quotients for the 95 Percent UCL for Whole-Body[a] Fish Tissue Residues Collected from 1990 to 2003 in the PVS Study Area
Palos Verdes Ecological Risk Assessment

Species	Analyte	95% UCL[a] Whole- Body Concentration	NOEC	LOEC	Percentage of Total Samples Exceeding NOEC	Percentage of Total Samples Exceeding LOEC	NOEC HQ	LOEC HQ
Dover sole[b]	tDDT	4.4409	1.92	12.1	54.55	0.01	2.31	0.37
Kelp bass[c]	tDDT	1.5235	1.92	12.1	14.54	<0.01	0.79	0.13
White croaker[d]	tDDT	14.8649	1.92	12.1	88.92	29.82	7.74	1.23
Sanddab guild[e]	tDDT	10.462[b]	1.92	12.1	71.43	0.00	5.45	0.86
Dover sole[f]	tPCB	0.6557	21.0	170.0	0.00	0.00	0.03	<0.01
Kelp bass[g]	tPCB	0.6585	21.0	170.0	0.00	0.00	0.03	<0.01
White croaker	tPCB	1.6881	21.0	170.0	0.00	0.00	0.08	0.01
Sanddab guild	tPCB	0.710[b]	21.0	170.0	0.00	0.00	0.03	<0.01

Notes:
[a] Concentrations of fish in the sanddab guild were reported as whole-body concentrations, whereas those for Dover sole, Kelp bass, and white croaker were reported as fillet The 95% UCL fillet values (see Table 2-10) for these three species were converted to whole-body using equations presented in Table 3-1.
[b] The maximum concentration was selected for comparison to the benchmarks because the 95 percent UCL was greater than the maximum concentration.
NOEC and LOEC units are mg/kg WW.
Shaded cells indicate exceedance of benchmarks, (i.e., HQs>1).

TABLE 4-8
Risk from Exposure to Water Column Concentrations of tDDT
Palos Verdes Ecological Risk Assessment

Station	Station Depth (m)	Sampling Depth (m)	Season	Concentration (ng/L)	Chronic Benchmark[a] (ng/L)	HQ
0C-1M	60	59	Winter	2.3	7.2	0.32
3C-1M	60	59	Winter	4.5	7.2	0.63
5C-1M	60	59	Winter	9.2	7.2	1.28
6C-1M	60	59	Winter	14.5	7.2	2.01
6C-2M	60	58	Winter	15.8	7.2	2.19
6C-5M	60	55	Winter	7.6	7.2	1.06
6C-20M	60	40	Winter	2.8	7.2	0.39
6C-35M	60	25	Winter	0.8	7.2	0.11
7C-1M	60	59	Winter	9.9	7.2	1.38
9C-1M	60	59	Winter	5.3	7.2	0.74

TABLE 4-8
Risk from Exposure to Water Column Concentrations of tDDT
Palos Verdes Ecological Risk Assessment

Station	Station Depth (m)	Sampling Depth (m)	Season	Concentration (ng/L)	Chronic Benchmark[a] (ng/L)	HQ
6B-1M	150	149	Winter	5.4	7.2	0.75
6D-1M	30	29	Winter	7.2	7.2	1.00
0C-1M	60	59	Summer	4.3	7.2	0.60
3C-1M	60	59	Summer	7.6	7.2	1.06
5C-1M	60	59	Summer	10.4	7.2	1.44
6C-1M	60	59	Summer	8.7	7.2	1.21
6C-2M	60	58	Summer	10.3	7.2	1.43
6C-5M	60	55	Summer	8.6	7.2	1.19
6C-20M	60	40	Summer	2.0	7.2	0.28
6C-35M	60	25	Summer	0.6	7.2	0.08
7C-1M	60	59	Summer	5.5	7.2	0.76
9C-1M	60	59	Summer	5.0	7.2	0.69
6B-1M	150	149	Summer	5.6	7.2	0.78
6D-1M	30	29	Summer	3.0	7.2	0.42

Notes:
[a] The chronic benchmark was developed by dividing the acute water quality benchmark by an acute-to-chance factor of 18.
Winter samples taken Jan-March, 1997
Summer samples taken Jun-Jul, 1997
CCC criterion continuous concentration (USEPA 2002)
Shaded stations show HQ greater than 1.0.

TABLE 4-9
Risk from Exposure to Water Column Concentrations of tPCB
Palos Verdes Ecological Risk Assessment

Station	Station Depth (m)	Sampling Depth (m)	Season	Concentration (ng/L)	CCC (ng/L)	HQ
0C-1M	60	59	Winter	0.14	30	0.00
3C-1M	60	59	Winter	0.28	30	0.01
5C-1M	60	59	Winter	0.51	30	0.02
6C-1M	60	59	Winter	0.88	30	0.03
6C-2M	60	58	Winter	0.89	30	0.03
6C-5M	60	55	Winter	0.41	30	0.01
6C-20M	60	40	Winter	0.21	30	0.01

TABLE 4-9
Risk from Exposure to Water Column Concentrations of tPCB
Palos Verdes Ecological Risk Assessment

Station	Station Depth (m)	Sampling Depth (m)	Season	Concentration (ng/L)	CCC (ng/L)	HQ
6C-35M	60	25	Winter	0.06	30	0.00
7C-1M	60	59	Winter	0.65	30	0.02
9C-1M	60	59	Winter	0.31	30	0.01
6B-1M	150	149	Winter	0.33	30	0.01
6D-1M	30	29	Winter	0.48	30	0.02
0C-1M	60	59	Summer	0.41	30	0.01
3C-1M	60	59	Summer	0.94	30	0.03
5C-1M	60	59	Summer	1.14	30	0.04
6C-1M	60	59	Summer	0.84	30	0.03
6C-2M	60	58	Summer	1.11	30	0.04
6C-5M	60	55	Summer	0.94	30	0.03
6C-20M	60	40	Summer	0.28	30	0.01
6C-35M	60	25	Summer	0.21	30	0.01
7C-1M	60	59	Summer	0.56	30	0.02
9C-1M	60	59	Summer	0.3	30	0.01
6B-1M	150	149	Summer	0.52	30	0.02
6D-1M	30	29	Summer	0.67	30	0.02

Notes:
Winter samples taken Jan-March, 1997
Summer samples taken Jun-Jul, 1997
CCC Criterion Continuous Concentration (USEPA 2002)

HQs for tissue-based exposure are also presented in Table 4-7. Ninety-five percent UCLs for whole-body concentrations in Dover sole, kelp bass, and white croaker were calculated from measured fillet concentrations, whereas maximum concentrations for sanddabs were based on whole-body concentrations that were measured directly. The maximum concentrations were selected as EPCs for sanddabs because the 95 UCL for both tDDT and tPCB exceeded the maximum. For tDDT, 95 percent UCLs for Dover sole and white croaker and the maximum for sanddabs exceeded literature-derived NOEC (i.e., HQ>1), whereas only tDDT concentrations in white croakers exceeded the LOEC (Table 4-7). No tPCB benchmark was exceeded by EPCs for any fish receptor.

HQs for external exposure (i.e., water-column based exposure from Zeng et al. 1999) are presented in Tables 4-8 and 4-9. None of the tPCB concentrations measured in the water column exceeded the tPCB criterion continuous concentration for water quality criteria (CCC, Table 4-9). However, water-column tDDT concentrations exceeded the tDDT CCC at 42 percent (5/12) of the stations (Table 4-8).

4.1.3 Birds

4.1.3.1 External Exposure

The spatial distributions of oral exposures for brown pelicans and double-crested cormorants are shown in Figures 3-5 and 3-6. During the breeding season, the foraging range, and thus exposure, is limited to the areas surrounding the breeding colonies on Anacapa and Santa Barbara Islands. As shown in Figures 3-5 and 3-6, tDDT and tPCB exposures are low in these areas. Outside of the breeding season, pelicans and cormorants forage throughout the SCB along the mainland coast and in waters around several of the Channel Islands. Pelicans are exposed to higher concentrations of tDDT while foraging in the PVS study area with areas of moderate tDDT exposure north of PVS, near Ventura. Total PCB follows a similar trend with higher concentrations in the PVS study area and moderate concentrations near Ventura. There are also some areas of moderate tPCB exposure to the south of the PVS study area, near San Diego (Figure 3-5). Although foraging in shallower waters and on slightly different prey than pelicans, cormorants exhibit similar patterns of exposure during the non-breeding season (Figure 3-6).

Spatial exposure distributions for peregrine falcons are represented by the coastal foraging range (to the 150 m depth contour) of western gulls and are depicted in Figure 3-8. Greatest tDDT and tPCB exposure to peregrines is a result of foraging within the PVS study area by their prey species, though tPCB exposure is more widespread along the coast. Lower exposures are observed when prey are foraging around the peregrine breeding areas in the Northern Channel Islands, though those breeding in the Los Angeles Harbor area are likely to get the highest exposures. Bald eagles currently nest only on Santa Catalina Island and forage in the waters surrounding the island. Sediment concentrations of tDDT and tPCB in the areas immediately surrounding Santa Catalina Island varied little (tDDT) or did not vary at all (tPCB), resulting in little or no variation in the modeled fish concentrations within the bald eagle foraging range (5 km radius of the island). As a consequence, variation in bald eagle exposure is best represented by the variation in the seabird and sea lion portions of their diet as depicted in Figures 3-9 and 3-10 for tDDT and tPCB, respectively. These spatial depictions are represented by the foraging ranges of western gulls and sea lions breeding (gulls) or loafing (gulls and sea lions) on Santa Catalina Island. For gulls, this area is a 50 km radius around the island within the major foraging areas depicted in Figure 3-7 and to the 150 m depth contour. For sea lions, the area is a 54 km radius around the island and to the 200 m depth contour. Greatest tDDT and tPCB exposure to bald eagles is a result of foraging within the PVS study area by their prey species (Figures 3-9 and 3-10).

Two sets of benchmarks were used to evaluate risk to birds from tDDT and tPCB in the SCB. These primary and secondary oral exposure benchmarks (NOAELs and LOAELs; Table 3-11) were compared to modeled oral exposure data to determine the percentage of locations (by grid cell in the model) within the SCB that exceeded these benchmarks. These results are summarized in Tables 4-10 and 4-11. None of the modeled oral exposures for brown pelicans (breeding and non-breeding), double-crested cormorants (breeding and non-breeding), and bald eagles exceeded their respective primary NOAEL or LOAEL benchmarks. For peregrine falcons, 3.00 and 0.93 percent of modeled tDDT exposures exceeded the primary NOAEL and LOAEL, respectively. For tPCB, 11.80 percent of estimated exposures exceeded the primary NOAEL and 0.74 percent exceeded the primary LOAEL.

TABLE 4-10
Percentage of Modeled Data Exceeding Primary Oral Benchmarks for Birds and Marine Mammals
Palos Verdes Ecological Risk Assessment

Receptor	Range of Modeled Oral Exposure	Analyte	NOAEL (mg/kg/d)	LOAEL (mg/kg/d)	Percentage Exceeding NOAEL	Percentage Exceeding LOAEL
Brown pelican, non-breeding						
	0.0034-0.0862	tDDT	1.3480	3.3700	0.00	0.00
	0.0034-0.0305	tPCB	0.3650	0.7310	0.00	0.00
Brown pelican, breeding						
	0.0092-0.0197	tDDT	1.3480	3.3700	0.00	0.00
	0.005-0.0139	tPCB	0.3650	0.7310	0.00	0.00
Double-crested cormorant, non-breeding						
	0.0043-0.2744	tDDT	1.2120	3.0310	0.00	0.00
	0.0057-0.1171	tPCB	0.3290	0.6570	0.00	0.00
Double-crested cormorant, breeding						
	0.0153-0.0495	tDDT	1.2120	3.0310	0.00	0.00
	0.0091-0.0289	tPCB	0.3290	0.6570	0.00	0.00
Bald eagle						
	0.5061-0.5299	tDDT	1.47	3.64	0.00	0.00
	0.297[a]	tPCB	0.40	0.79	0.00	0.00
Peregrine falcon						
	0.1097-60.1539	tDDT	0.96	2.39	3.00	0.93
	0.0613-2.5776	tPCB	0.26	0.52	11.80	0.74
California sea lion (adult female), fall						
	0.0001-0.1848	tDDT	0.567	2.83	0.00	0.00
	0.0001-0.0148	tPCB	0.106	0.523	0.00	0.00
California sea lion (adult female), winter/spring						
	0.0001-0.1718	tDDT	0.567	2.83	0.00	0.00
	0.0001-0.0109	tPCB	0.106	0.523	0.00	0.00
California sea lion (adult female), summer						
	0.0001-0.0845	tDDT	0.567	2.83	0.00	0.00
	0.0001-0.005	tPCB	0.106	0.523	0.00	0.00
California sea lion (nursing pup), fall						
	0.0007-0.9148	tDDT	0.625	3.124	0.01	0.00
	0.0009-0.0979	tPCB	0.117	0.577	0.00	0.00
California sea lion (nursing pup), winter/spring						
	0.0007-0.8504	tDDT	0.625	3.26	0.01	0.00
	0.0009-0.072	tPCB	0.117	0.577	0.00	0.00
California sea lion (nursing pup), summer						
	0.0005-0.4186	tDDT	0.625	3.26	0.00	0.00
	0.0008-0.033	tPCB	0.117	0.577	0.00	0.00

Notes:
[a] Only one exposure value modeled for tPCB in bald eagles due to lack of tPCB variation in the sediment within a 5 km radius of Santa Catalina Island.

TABLE 4-11
Percentage of Modeled Data Exceeding Secondary Oral Benchmarks for Birds and Marine Mammals
Palos Verdes Ecological Risk Assessment

Receptor	Range of Modeled Oral Exposure	Analyte	NOAEL (mg/kg/d)	LOAEL (mg/kg/d)	Percentage Exceeding NOAEL	Percentage Exceeding LOAEL
Brown pelican, non-breeding						
	0.0034-0.0862	tDDT	0.0027	0.0270	100.00	2.57
	0.0034-0.0305	tPCB	0.0088	0.0877	32.54	0.00
Brown pelican, breeding						
	0.0092-0.0197	tDDT	0.0027	0.0270	100.00	0.00
	0.005-0.0139	tPCB	0.0088	0.0877	23.69	0.00
Double-crested cormorant, non-breeding						
	0.0043-0.2744	tDDT	0.0027	0.0270	100.00	20.37
	0.0057-0.1171	tPCB	0.0079	0.0789	96.24	0.04
Double-crested cormorant, breeding						
	0.0153-0.0495	tDDT	0.0027	0.0270	100.00	4.59
	0.0091-0.0289	tPCB	0.0079	0.0789	100.00	0.00
Bald eagle						
	0.5061-0.5299	tDDT	0.071	0.71	100.00	0.00
	0.297[a]	tPCB	0.280	2.79	100.00	0.00
		tPCB	0.097	0.97	100.00	0.00
		tPCB	0.916	--	0.00	--
Peregrine falcon						
	0.1097-60.1539	tDDT	0.046	0.46	100.00	16.45
	0.0613-2.5776	tPCB	0.180	1.83	20.96	0.05
		tPCB	0.060	6.38	100.00	0.00
		tPCB	0.600	--	0.53	--
California sea lion (adult female), fall						
	0.0001-0.1848	tDDT	0.0090	--	2.06	--
	0.0001-0.0148	tPCB	0.0052	0.028	0.63	0.00
		tPCB	0.0050	0.029	0.66	0.00
California sea lion (adult female), winter/spring						
	0.0001-0.1718	tDDT	0.0090	--	1.58	--
	0.0001-0.0109	tPCB	0.0052	0.028	0.50	0.00
		tPCB	0.0050	0.029	0.52	0.00
California sea lion (adult female), summer						
	0.0001-0.0845	tDDT	0.0090	--	0.84	--
	0.0001-0.005	tPCB	0.0052	0.028	0.00	0.00
		tPCB	0.0050	0.029	0.00	0.00

TABLE 4-11
Percentage of Modeled Data Exceeding Secondary Oral Benchmarks for Birds and Marine Mammals
Palos Verdes Ecological Risk Assessment

Receptor	Range of Modeled Oral Exposure	Analyte	NOAEL (mg/kg/d)	LOAEL (mg/kg/d)	Percentage Exceeding NOAEL	Percentage Exceeding LOAEL
California sea lion (nursing pup), fall						
	0.0007-0.9148	tDDT	0.0100	--	51.74	--
	0.0009-0.0979	tPCB	0.0053	0.032	61.66	0.74
		tPCB	0.0055	0.031	60.58	0.77
California sea lion (nursing pup), winter/spring						
	0.0007-0.8504	tDDT	0.0100	--	61.56	--
	0.0009-0.072	tPCB	0.0053	0.032	71.95	0.55
		tPCB	0.0055	0.031	71.51	0.56
California sea lion (nursing pup), summer						
	0.0005-0.4186	tDDT	0.0100	--	45.12	--
	0.0008-0.033	tPCB	0.0053	0.032	51.69	<0.01
		tPCB	0.0055	0.031	50.84	<0.01

Notes:
a Only one exposure value modeled for tPCB in bald eagles due to lack of tPCB variation in the sediment within a 5 km radius of Santa Catalina Island.
-- denotes no data.

The primary benchmarks were developed from single-chemical, chronic laboratory studies with multiple dosages and a clear dose-response relationship (Heath et al. 1969; Britton and Huston 1973). However, the test species for these studies were the mallard duck and the white-leghorn hen. Using benchmarks scaled from mallard ducks and chickens to estimate risk to pelicans, cormorants, eagles, and falcons may over- or underestimate risk; therefore, a set of NOAEL and LOAEL benchmarks available for species more closely related to the selected avian receptors (Table 3-11) were also evaluated. Studies from which secondary benchmarks were selected most often used only one dosage level. This study design does not allow for evaluation of a dose-response relationship. However, the selected studies reported a significant adverse effect to the test species, indicating that adverse effects are probable if the LOAEL is exceeded. In accordance with USEPA Superfund guidance (USEPA 1997), a chronic LOAEL was normalized to a chronic NOAEL by multiplying the LOAEL by 0.1. As discussed in subsection 3.2.3.1, field studies also introduce confounding factors associated with exposure to multiple chemicals in an uncontrolled environment; however, these may be appropriate for use in developing benchmarks when available for a selected receptor (e.g., brown pelican). The field-based benchmark for brown pelicans was developed for pelicans nesting in the SCB, which adds further support for the use of this value.

The secondary benchmarks for birds gave an additional line of evidence for risk evaluation, and the percent of locations within the SCB that exceeded these NOAELs and LOAELs are presented in Table 4-11. The NOAEL was exceeded by all of the modeled oral exposures for tDDT for breeding and non-breeding pelicans and cormorants. For tPCB, 32.54 and 23.69 percent of modeled exposures for non-breeding and breeding pelicans, respectively,

exceeded the NOAEL, whereas for cormants, the percent tPCB exceedances were 96.24 and 100 for non-breeding and breeding birds. The tDDT LOAEL was exceeded by 2.57 percent of the modeled oral exposures for non-breeding pelicans, but the tPCB LOAEL was not exceeded by any modeled data. Neither tDDT nor tPCB LOAEL was exceeded by modeled exposures for breeding pelicans. The LOAEL exceedances for cormorants followed a similar pattern: the magnitude of the non-breeding cormorant modeled oral exposure data exceeding the tDDT LOAEL (20.37 percent) is greater than the percentage exceeding the tDDT LOAEL in breeding cormorants (4.59 percent). All modeled tDDT oral exposures for the bald eagle and peregrine falcon exceeded the tDDT NOAEL, whereas none of the bald eagle and 16.47 percent of the peregrine falcon oral exposures exceeded the tDDT LOAEL. Of the three tPCB NOAELS for bald eagles, two were exceeded by all the modeled tPCB exposures. LOAELs for tPCB were not exceeded by the modeled tPCB oral exposures for bald eagles. The three secondary NOAELs for tPCB were exceeded by 20.96, 100, and 0.53 percent of the modeled oral exposures for peregrine falcons, and 0.05 percent exceeded the LOAEL associated with the first NOAEL. There were no exceedances of the LOAEL associated with the second tPCB NOAEL, and no LOAEL was available from the third study.

4.1.3.2 Internal Exposure

Measured tDDT and tPCB concentrations in eggs collected from various locations within the SCB (Table 2-9) were compared to literature-derived NOEC and LOEC benchmarks (highlighted values in Tables 3-12 and 3-13). Additionally, calculated sumTEQ values in these eggs (Table 2-11) were compared to the literature-derived NOEC and LOEC (Table 3-13). HQs calculated from these comparisons are presented in Table 4-12. The mean, maximum, and 95-percent UCL (as available) were chosen as EPCs.

The tDDT concentrations in brown pelican eggs collected from Anacapa and Santa Catalina Islands exceeded both the NOEC (HQ range 1.57 to 12.05) and LOEC (HQ range 1.05 to 8.03) for all EPCs. In contrast, only the maximum concentration of tPCB from eggs collected at Anacapa Island exceeded the LOEC (HQ = 2.59). Cormorant eggs were found to have tDDT concentrations that exceeded the NOEC at Anacapa Island, but not at Santa Catalina Island. Moreover, the maximum and 95-percent UCL concentration of tPCB in eggs collected at Anacapa exceeded the LOEC (HQs = 2.56 and 1.24, respectively); however, the mean tDDT concentration in these eggs was below the LOEC. Mean, maximum, and 95-percent UCL concentrations of tPCB exceeded the LOEC for cormorants nesting on Anacapa Island. It should be noted that a tPCB NOEC was not available for pelicans and cormorants. Additionally, the tPCB benchmark for double-crested cormorants was used as a surrogate for brown pelicans.

Peregrine falcon eggs collected from Anacapa, San Miguel, and Santa Cruz Islands had mean, maximum, and 95-percent UCL concentrations that exceeded the tDDT LOEC (HQ range 1.01 to 5.78). Only one egg was collected from Santa Rosa Island and it also exceeded the tDDT LOEC (HQ = 2.17). Both the LOEC and NOEC for tPCB were also exceeded by all EPCs (HQ range 2.29 to 25.78). Mean, maximum, and 95-percent UCL concentrations in bald eagle eggs at Santa Catalina Island exceeded the NOEC and LOEC for both tDDT and tPCB (HQ ranges of 4.61 to 18.67 and 2.16 to 22.85 for tDDT and tPCB, respectively).

TABLE 4-12
Hazard Quotient Calculations for Concentrations in Avian Eggs Collected in the Southern California Bight, 1985-Present
Palos Verdes Ecological Risk Assessment

Analyte	Specific Island	Full Species Name	n	Mean[a]	Maximum[a]	95% UCL[a]	NOEC[a]	LOEC[a]	Mean NOEC HQ	Mean LOEC HQ	Max NOEC HQ	Max LOEC HQ	95% UCL NOEC HQ	95% UCL LOEC HQ
tDDT	Anacapa Island	Brown pelican	24	3.640	24.1	5.506	2	3	1.82	1.21	12.05	8.03	2.75	1.84
tDDT	Anacapa Island	Double-crested cormorant	13	8.340	25.6	12.371	4	10	2.09	0.83	6.40	2.56	3.09	1.24
tDDT	Anacapa Island	Peregrine falcon	2	52.55	70.0	86.752	--	15	--	3.50	--	4.67	--	5.78
tDDT	San Miguel Island	Peregrine falcon	6	20.117	26.3	23.997	--	15	--	1.34	--	1.75	--	1.60
tDDT	Santa Catalina Island	Bald eagle	36	27.187	67.2	31.947	3	5.9	9.06	4.61	18.67	11.39	10.65	5.41
tDDT	Santa Catalina Island	Brown pelican	1	3.14	3.14	--	2	3	1.57	1.05	1.57	1.05	--	--
tDDT	Santa Catalina Island	Double-crested cormorant	4	1.402	1.97	1.847	4	10	0.35	0.14	0.49	0.20	0.46	0.18
tDDT	Santa Cruz Island	Peregrine falcon	7	15.18	22.4	20.037	--	15	--	1.01	--	1.49	--	1.34
tDDT	Santa Rosa Island	Peregrine falcon	1	32.5	32.5	--	--	15	--	2.17	--	2.17	--	--
tPCB	Anacapa Island	Brown pelican	24	2.224	9.08	3.170	--	3.5	--	0.64	--	2.59	--	0.91
tPCB	Anacapa Island	Double-crested cormorant	13	5.053	18.6	8.256	--	3.5	--	1.44	--	5.31	--	2.36
tPCB	Anacapa Island	Peregrine falcon	2	19.8	26.8	33.520	1.3	4	15.23	4.95	20.62	6.70	25.78	8.38
tPCB	San Miguel Island	Peregrine falcon	6	9.148	10.6	10.194	1.3	4	7.04	2.29	8.15	2.65	7.84	2.55
tPCB	Santa Catalina Island	Bald eagle	36	8.661	29.70	10.610	1.3	4	6.66	2.17	22.85	7.43	8.16	2.65
tPCB	Santa Catalina Island	Brown pelican	1	1.43	1.43	--	--	3.5	--	0.41	--	0.41	--	--
tPCB	Santa Cruz Island	Peregrine falcon	7	9.4	16.1	13.113	1.3	4	7.23	2.35	12.38	4.03	10.09	3.28
tPCB	Santa Rosa Island	Peregrine falcon	1	13.6	13.6	--	1.3	4	10.46	3.40	10.46	3.40	--	--
sumTEQ	Anacapa Island	Brown pelican	22	0.014	0.027	0.02	--	178	--	0.00	--	0.00	--	0.00
sumTEQ	Anacapa Island	Double-crested cormorant	13	0.026	0.070	0.04	--	178	--	0.00	--	0.00	--	0.00

TABLE 4-12
Hazard Quotient Calculations for Concentrations in Avian Eggs Collected in the Southern California Bight, 1985-Present
Palos Verdes Ecological Risk Assessment

Analyte	Specific Island	Full Species Name	n	Mean[a]	Maximum[a]	95% UCL[a]	NOEC[a]	LOEC[a]	Mean NOEC HQ	Mean LOEC HQ	Max NOEC HQ	Max LOEC HQ	95% UCL NOEC HQ	95% UCL LOEC HQ
sumTEQ	Anacapa Island	Peregrine falcon	2	0.069	0.095	0.12	--	178	--	0.00	--	0.00	--	0.00
sumTEQ	San Miguel Island	Peregrine falcon	6	0.022	0.028	0.03	--	178	--	0.00	--	0.00	--	0.00
sumTEQ	Santa Catalina Island	Bald eagle	10	0.039	0.084	0.05	--	178	--	0.00	--	0.00	--	0.00
sumTEQ	Santa Catalina Island	Brown pelican	1	0.006	0.006	--	--	178	--	0.00	--	0.00	--	--
sumTEQ	Santa Catalina Island	Double-crested cormorant	4	0.004	0.006	0.01	--	178	--	0.00	--	0.00	--	0.00
sumTEQ	Santa Cruz Island	Peregrine falcon	7	0.021	0.034	0.03	--	178	--	0.00	--	0.00	--	0.00
sumTEQ	Santa Rosa Island	Peregrine falcon	1	0.074	0.074	--	--	178	--	0.00	--	0.00	--	--

Notes:
-- denotes no data.
a Concentrations of tDDT and tPCB are in mg/kg WW and sumTEQ concentrations are in ng/kg WW.

tPCB benchmarks for double-crested cormorant used as a surrogate for brown pelicans and those for bald eagles used as a surrogate for peregrine falcons.
sumTEQ benchmark for white leghorn hens used as surrogate for all receptors.
Shaded cells indicate HQs greater than 1.

Overall mean egg concentrations of tDDT and tPCB measured in coastal areas outside the SCB for double-crested cormorants and peregrine falcons (location means listed in Table 2-12) were less than those recorded on islands within the SCB (Table 2-11). For cormorants, tDDT concentrations in eggs from Anacapa Island were 8.3 mg/kg WW compared to 2.2 mg/kg WW outside the SCB, and tPCB concentrations were 5.1 mg/kg WW compared to 3.6 mg/kg WW. Egg concentrations of tDDT for peregrine falcons nesting on islands within the SCB ranged from 1.5 to 3 times greater than outside the SCB, with mean values of 20.1, 15.8, and 32.5 mg/kg WW at San Miguel, Santa Cruz, and Santa Rosa islands compared to 10.3 mg/kg outside the SCB. Although several studies were found that reported tissue levels of tDDT and tPCB for bald eagles outside the SCB, these data were generally for blood and only one egg collected in coastal Oregon was represented (Table 2-12). Total DDT in this egg was 11.1 mg/kg WW compared to a mean of 27.2 mg/kg WW for eggs collected from Santa Catalina Island. However, tPCB concentration in this egg were greater than those from Catalina (12.7 mg/kg WW compared to 11.1 mg/kg WW).

The available measured data indicate that tDDT, and to a lesser extent tPCB, concentrations in eggs from certain avian species within the SCB are greater (often 2 or more times for tDDT) than those in eggs collected in other areas along the North American coast. This is further evidence that elevated concentrations of tDDT in eggs from avian species inhabiting the SCB are likely related to exposure to elevated sediment concentrations at PVS via the food chain. Elevated levels of tPCB in the SCB may also be related to elevated sediment concentrations at the PVS, but also are indicative of more widespread tPCB contamination observed in other areas along the SCB mainland coast.

Calculated sumTEQ values for all receptors were well below the TEQ LOEC and a NOEC was not available. However, it should be noted that this LOEC was derived for white-leghorn hens and concentrations for some PCB congeners required for calculating a PCB sumTEQ were not available.

4.1.3.3 Site-Specific Field Studies

These studies have been detailed in Section 3.2.3.2 and are briefly discussed here. Populations of brown pelicans and double-crested cormorants nesting in the Channel Islands suffered severe reproductive failure in the late 1960s; however, these populations began to recover soon after the ban of DDT (Gress 1994; Gress et al. 2003). Pelicans have shown marked recovery since about 1978, but chronic low-level DDE exposure continues to contribute to low fledging rates in the Anacapa colony. Since the mid-1980s, cormorant populations also have stabilized; however, eggshell thinning is still apparent and remains close to the 20 percent thinning threshold for reproductive impairment.

Peregrine falcons were extirpated from the Channel Islands by the early 1950s and did not begin breeding in the islands until the late 1980s. Continued high DDE levels in peregrine prey species (i.e., seabirds) suggest that peregrines are at risk (Hunt 1994), and, though not currently in practice, artificial incubation of eggs would benefit (i.e., increase nesting success) nesting pairs at some locations (Linthicum 2003). Bald eagles also required reintroduction to the Channel Islands and currently are breeding only on Santa Catalina Island. As with peregrines, intervention is still required to sustain the population (Garcelon 1994; Sharpe 2003). In fact, few eggs laid by these pairs from 1989 to 2002 were successfully hatched in the laboratory and chicks hatched from captive-bred eagles were fostered into

Catalina eagle nests (Sharpe 2003). Bald eagle egg DDT concentrations appeared to decrease in some breeding pairs in the late 1990s; however, monitoring of pairs that laid eggs in each year from 1995 to 2000 indicated that DDT levels have not decreased in these long-term residents (Garcelon 2000).

Effects were also observed in other species inhabiting the SCB. For example, eggs of ashy storm petrels collected from 1995 to 1997 were 3.5 to 5.6 percent thinner than the historic mean (Welsh et al. 2001). Though this is not expected to cause population-wide reproductive failure, the reproductive success of some individuals may be reduced. Although gull species are relatively insensitive to eggshell thinning, studies by Fry (1994) indicate that high embryonic exposure to DDT (i.e., high DDT concentrations in gull eggs) resulted in a delayed reproductive effect in the gull population. Fry (1994) observed an increase in estrogenic effects such as supernormal clutches (twice the number of eggs), female-female pairing, and polygamous trios among breeding pairs. It was later determined that the apparent female-female pairing was actually pairing between females and feminized males. In the laboratory, Fry and Toone (1981) and Fry et al. 1987) demonstrated that these types of estrogenic effects could be induced by dosing gull eggs with DDT.

Although these site-specific studies emphasize the effects of tDDT on the avian receptors, these birds are also exposed to tPCB, which can also affect reproduction. Therefore, PCB exposure may contribute to the reproductive and estrogenic effects described.

4.1.4 Marine Mammals

4.1.4.1 External Exposure

Risk from tDDT and tPCB to the adult female California sea lion and nursing California sea lion pup was estimated seasonally (fall, winter/spring, and summer), to account for changes in foraging behavior and exposure. (Note: for the pups, exposure distributions represent the foraging areas of the adult female.) As with birds, two sets of benchmarks were evaluated (primary and secondary). The primary benchmark for tDDT was based on a laboratory study with rats, and the tPCB benchmarks are derived from a laboratory study using mink. These primary benchmarks were developed from single-chemical, chronic laboratory studies with multiple dosages and a clear dose-response relationship (Fitzhugh 1948, Aulerich and Ringer 1977). However, the test species for these studies were the rat and the mink, and these species may not be appropriate surrogates for a marine mammal. Marine mammals have a much higher fat content than rats and mink and may store much higher concentrations of the contaminants than rats or mink. While stored, these chemicals are generally unavailable to the animal; however, fat stores are highly dynamic (particularly in marine mammals) and seasonal use of fat stores (e.g., starvation conditions experienced during mating) could release very high concentrations of chemicals into the blood stream. Therefore, it is difficult to determine if levels that cause effects in rats or mink would be protective of marine mammals and use of these benchmarks may over- or underestimate risk. Thus, a set of secondary NOAEL and LOAEL benchmarks (Table 3-11) available for species more closely related to the sea lion (i.e., harbor seals) were also evaluated. As previously discussed, field or semi-field studies introduce confounding factors associated with exposure to multiple chemicals. Additionally, the harbor seal studies were not conducted in the SCB; therefore, the mixture of chemicals likely varies. This introduces differences associated with the antagonistic and/or synergistic nature of the chemicals in the environment, and may result in under- or overestimation of risk to sea lions in the SCB.

None of modeled tDDT or tPCB oral exposures for adult female sea lions exceeded the primary NOAEL or LOAEL in any season (Table 4-10).

The percentage of modeled oral exposures exceeding the secondary tDDT NOAEL benchmark was 2.06 percent, 1.58 percent, and 0.84 percent for oral exposure modeled in the fall, winter/spring, and summer, respectively (Table 4-11). The percentage of modeled oral exposure in the sea lion exceeding the LOAEL was not calculated because a suitable LOAEL was not available for tDDT. For tPCB exposure, 0.63 percent and 0.50 percent of the fall and winter/spring modeled data exceeded one secondary NOAEL, and 0.66 percent and 0.52 percent, respectively, exceeded another secondary NOAEL (Table 4-12). None of the modeled oral exposures in summer exceeded the tPCB NOAELs, and none of the modeled exposures over all three seasons exceeded tPCB LOAELs.

Spatial distributions of oral exposures in adult female California sea lions are presented in Figures 3-12 and 3-13. These exposure distributions are weighted by the area use distributions presented in Figure 3-11. As depicted in Figure 3-12, DDT exposure varies by season with exposures being highest near the PVS study area during the fall. During the summer breeding season, higher exposure (relative to other areas in the same season) occurs around San Miguel, Anacapa, Santa Barbara, and eastern Santa Cruz Islands and within the PVS study area. The winter/spring season (December to May) exposure values are nearly as high as those in fall and are highest in the PVS study area and around Santa Catalina, Anacapa, San Clemente, eastern Santa Cruz, and eastern Santa Barbara islands.

PCB exposures (shown in Figure 3-13) also vary by season with the greatest exposures occurring around the breeding areas (San Miguel, Santa Rosa, Santa Cruz, and Santa Barbara Islands) in summer and fall. In the winter/spring, the greatest exposures occur near Santa Catalina and San Clemente islands. Adult female sea lions obtain greater exposures in the PVS study area during the fall and winter/spring when use of this area is greater.

For nursing California sea lion pups, 0.01 percent of modeled tDDT exposures exceeded the primary NOAEL for fall and winter/spring, though no exceedances were observed in summer. Total PCB oral exposures did not exceed the primary NOAEL in any season, and no modeled tDDT or tPCB oral exposures exceeded the primary LOAEL in any season (Table 4-10).

The percentage of modeled oral exposures exceeding the secondary tDDT NOAEL benchmark was 51.74 percent, 61.56 percent, and 45.12 percent for oral exposure modeled in the fall, winter/spring, and summer, respectively (Table 4-11). The percentage of modeled oral exposure in the sea lion pups exceeding the LOAEL was not calculated because a suitable LOAEL was not available for tDDT. For tPCB exposure, 61.66 percent, 71.95 percent, and 51.69 percent of the fall, winter/spring, and summer modeled data exceeded one secondary NOAEL, and 60.58 percent, 71.51 percent, and 50.84 percent, respectively, exceeded another secondary NOAEL (Table 4-11). The percentage of the modeled oral exposures exceeding the secondary tPCB LOAELs was significantly less than that exceeding the NOAELs (range: <0.01 to 0.77 percent of modeled values exceeded LOAELs).

Although nursing sea lion pups less than seven months of age do not forage, but rely solely on milk from the mother, spatial distributions of oral exposures in pups are depicted throughout the SCB. These distributions represent the pup exposure as a function of the seasonal adult female foraging areas and are presented in Figures 3-14 and 3-15. These exposure distributions

are weighted by the area use distributions presented in Figure 3-10. As depicted in Figure 3-15, DDT exposure varies by season with exposures being highest near the PVS study area during the fall. During the summer breeding season, higher exposure (relative to other areas in the same season) occurs from adult female foraging around San Miguel, Anacapa, Santa Barbara, and eastern Santa Cruz islands and within the PVS study area. The winter/spring season (December to May) exposure values are nearly as high as those in fall and are highest from female foraging in the PVS study area and around Santa Catalina, Anacapa, San Clemente, eastern Santa Cruz, and eastern Santa Barbara islands.

PCB exposures (shown in Figure 3-15) also vary by season with the greatest exposures occurring when females are foraging around the breeding areas (San Miguel, Santa Rosa, Santa Cruz, and Santa Barbara islands) in summer and fall. In the winter/spring, the greatest exposures result from adult foraging near Santa Catalina and San Clemente Islands. It should be noted that pups will likely begin foraging on their own in late December or January. Sea lion pups obtain greater exposures from the PVS study area during the fall and winter/spring when adult female use of this area is greater.

4.1.4.2 Internal Exposure

Concentrations of 4,4′-DDE and tPCB have been measured in blubber tissue from California sea lions collected at San Miguel Island during three periods (1970, 1972, and 1991). Although the values for the 1991 effort have been reported elsewhere (HydroQual, Inc. 1997), the quality of the data has been questioned and the samples are currently being re-analyzed (DeLong 2002). Despite the concerns over the quality of the data, the trends in the data (i.e., premature parturient females have lower mean ages and higher mean DDE and PCB concentrations than full-term parturient females) are valid and consistent with the data collected in 1970 and 1972. Additionally, the DDE and PCB values for 1991 are, on average, one order of magnitude lower than those from the early 1970s (DeLong 2002). Therefore, 1991 values were estimated by multiplying the highest mean value from either 1970 or 1972 by a factor of 0.1 (Table 4-13).

These historic (1970 and 1972) and estimated recent (1991) blubber PCB concentrations were compared to the critical threshold for marine mammals developed by Kannan et al. (2000) and resulting HQs are presented in Table 4-13. No blubber benchmark was available for DDE; however, a LOEC for blood is available from a dolphin study. Kannan et al. (2000) estimated that the blubber PCB threshold of 17 mg/kg lipid weight (LW) was approximately two times that of the blood threshold (8.7 mg/kg LW) based on comparisons of measured blood-to-blubber ratios. Although similar ratios are unknown for DDE, a benchmark can be calculated from the dolphin study by assuming that the blood-to-blubber ratio for DDE is similar to that measured for tPCB. Therefore, a DDE benchmark of 26 mg/kg LW was used to calculate blubber HQs for sea lions (Table 4-13).

DDE concentrations in blubber collected in 1970 and 1972 exceeded the threshold (HQ range 3.85 to 36.31); however, only DDE concentrations in blubber collected from premature parturient females in 1991 exceeded the threshold (HQ = 3.63). PCB concentrations in premature parturient females in 1970 and 1972 and in full-term parturient females in 1970 also exceeded benchmarks. Neither group of females sampled in 1991 is likely to have had PCB concentrations in blubber that exceeded the benchmark.

Internal measurements were not available for seven-month-old sea lion pups.

TABLE 4-13
Hazard Quotient Calculations for 4,4'-DDE and PCB Residues in Blubber Collected from California Sea Lions at San Miguel Island
Palos Verdes Ecological Risk Assessment

	Premature Parturient Females							Full-term Parturient Females						
Year	**Mean Age**	**4,4'-DDE[a]**	**PCB[a]**	**DDE Benchmark[a]**	**PCB Benchmark[a]**	**DDE HQ**	**PCB HQ**	**Mean Age**	**4,4'-DDE[a]**	**PCB[a]**	**DDE Benchmark[a]**	**PCB Benchmark[a]**	**DDE HQ**	**PCB HQ**
1970	8	944	138	26	17	36.31	8.12	12	109	20	26	17	4.19	1.18
1972	7.7	779	71	26	17	29.96	4.18	11.5	100	16	26	17	3.85	0.94
1991[b]	6.7	94.4	13.8	26	17	3.63	0.81	11.1	10.9	2	26	17	0.42	0.12

Notes:
[a] Measured values and benchmarks are in mg/kg lipid weight.
[b] Others (e.g., HydroQual 1997) have presented measured values for samples collected in 1991. However, quality control measures have been questioned and the samples are being reanalyzed (DeLong 2002). Therefore, 1991 values were calculated as (0.1)x (highest value from 1970 or 1972) as per instructions from Robert DeLong at NOAA.

Shaded values show values with HQ greater than 1.0.

4.1.4.3 Site-Specific Field Studies

In the early 1970s, tDDT and tPCB were implicated as factors contributing to an increase in premature births among young female sea lions (DeLong et al. 1973; Gilmartin et al. 1976). Liberg-Clark et al. (1995) report a decrease in blubber concentrations from 1970 to 1992 and a 133 percent increase in the sea lion population during this same time period. The authors suggest that decreased DDT concentrations contributed to this increase. O'Shea and Brownell (1998) confirmed the large decrease in contaminant loads among sea lions; however, they analyzed sea lion census data over the past 100 years and reported a general increase in sea lion population size before, during and after the time of highest DDT exposure. Despite these increasing populations, immigration of individuals from Mexico and the impact of federal protection of sea lions could not be excluded as confounding factors.

4.2 Risk Description

4.2.1 Benthic Invertebrates and Infaunal Organisms

The large spatial extent of sediment contamination at PVS represents some risk to epibenthic and infaunal invertebrates. The documentation of risk to invertebrates comes from several sources and can be compiled using a weight-of-evidence approach. All lines of evidence point towards the same general demonstration of risk.

Bulk sediment and porewater exposure assessments revealed the strong potential for toxicity to invertebrates from the COPECs at PVS with a gradient of effect centered on the outfalls (Figures 2-3, 2-5, and 2-6).

Benthic community studies confirm that invertebrates in the areas of the outfall appear to be more affected than at more distant sites. Significant reductions in biodiversity have been observed in the immediate vicinity of the outfall (Table 4-4).

Sediment toxicity studies also confirmed this basic pattern. Although acute toxicity was not found in PVS sediments, chronic toxicity was apparent near the outfalls (although interpretations are equivocal) and do not directly demonstrate toxicity from the COPECS.

As a general weight-of-evidence, there is demonstrable risk to epibenthic and infaunal invertebrates at PVS from both COPECs, although tDDT poses a higher risk than tPCB in terms of the overall level of exceedance of benchmarks and the areal extent of risk. In addition, there appear to be multiple, possibly synergistic "outfall effects" that contribute to impairment of the benthic community in the immediate vicinity of the outfalls.

Deep-dwelling invertebrates may be exposed to sediments and porewater at 15-30 cm depths. COPEC concentrations in bulk sediment at those depths are greater than in surface sediments (Figures 2-2 and 2-4). However, the deep sediments generally show the same spatial pattern of concentration as do the shallow sediments. Therefore, it is likely that they do not provide independent, additional risk in areas greater than those where the surface sediment exceeds toxic levels.

4.2.2 Fish

Studies (1980s to 2003) have demonstrated that an extensive deposit of tDDT and tPCB exists at the PVS (Lee 1994) to which fish may be exposed (USEPA 2000; Zeng et al. 1999). Several researchers have concluded that reproductive effects can occur at environmentally relevant concentrations (Hose and Cross 1988; Spies and Thomas 1997), and that life expectancy, but not growth rate, has been impacted (Moore 2000). However, others (Allen et al. 1997) have observed that the incidence and prevalence of DDT- and PCB-related anomalies such as skin lesions, tumors, and skeletal deformities have decreased to background levels (1 percent).

Fish tissue samples collected at the PVS show that exposure to tDDT occurs at levels sufficient for risk, whereas exposure to tPCB does not (Table 4-7). Hazard quotients based on tissue-residues (internal exposure) show that tDDT poses probable risk (i.e., both the NOEC and LOEC were exceeded) to white croaker at the PVS, possible risk (i.e., NOEC, but not LOEC, exceeded) to Dover sole and sanddabs at the PVS, and no risk (i.e., no exceedances) to kelp bass. No risk is posed by tPCB on the basis of tissue concentrations (Table 4-7). Hazard quotients for water-column-based (i.e., external) exposure show that no risk is posed by tPCB (Table 4-9), but that risk from exposure to tDDT is likely (Table 4-8).

Based on these lines of evidence (literature studies, external exposure, and internal exposure), no risk from tPCB to fish is expected to occur at the PVS or in the SCB. However, risk from tDDT, while small in the SCB, may be possible for Dover sole and sanddabs and probable for white croakers at the PVS.

4.2.3 Birds

In this section, each line of evidence (external and internal exposure and site-specific field studies) is evaluated and a weight-of-evidence level of risk is determined for each avian receptor.

Weight-of-evidence conclusions concerning presence or absence of risks were determined following processes outlined in Suter et al. (2000). Data were considered to be of unequal value in the weight-of-evidence. Greater weight was given to data displaying either responses of test biota exposed in the laboratory to site-specific field-collected media or site-specific field surveys of responses of resident biota to site-contamination. Comparison of literature-based toxicity data to site-specific exposure estimates received the lowest weight. Strength of risk conclusions was considered high if multiple lines of evidence, including site-specific field surveys and toxicity tests, were available for a given receptor and all lines of evidence were in agreement. Risk conclusions were considered to be of moderate strength if data consisted of literature-based toxicity and one site-specific line of evidence. If only literature-based toxicity data were available to evaluate risks, the strength of risk conclusions was considered to be low.

All of the avian species evaluated are federal and/or California special-status species (Table 2-1) and are protected at the individual level; therefore, exceedance of the NOAEL/NOEC indicates risk to the survival and reproduction of individuals. Exceedance of the LOAEL/LOEC further supports this risk and indicates that risk to populations is probable.

4.2.3.1 Brown Pelican

Oral exposure distributions were compared to literature-derived NOAELs and LOAELs. Although primary benchmarks (i.e., those based on single-chemical, chronic studies with

multiple dosages) were not exceeded, all modeled oral exposures exceeded secondary, but more receptor-appropriate, NOAELs for tDDT for breeding and non-breeding pelicans. For tPCB, 33 and 24 percent of oral exposures for non-breeding pelicans, respectively, exceeded the NOAEL. Additionally, 3 percent of modeled oral exposures exceeded the tDDT secondary LOAEL for non-breeding pelicans. Spatial analysis of these exposures indicates that exposures in areas around the breeding islands (Anacapa and Santa Barbara) are low and that exposure is reduced during the breeding season when foraging is restricted to these locations. During the non-breeding season, pelicans forage throughout the SCB within 20 km of the shore over water depths of 150 m or less. Individuals foraging within the PVS study area ingest prey with higher tDDT concentrations resulting in exceedance of the tDDT LOAEL. Additionally, pelican eggs collected from Anacapa exceeded NOECs and LOECs for tDDT and the maximum value exceeded the LOEC for tPCB. Finally, while the populations of pelicans nesting on Santa Barbara and Anacapa islands are increasing, chronic low-level exposure to DDE is likely depressing reproduction and some individuals still have egg concentrations above thresholds for reproductive failure.

Therefore, the weight-of-evidence is that both tDDT and tPCB present risk to individual brown pelicans; however, the risk from tPCB is expected to be lower than that from tDDT. Individuals primarily foraging within the PVS study area during the non-breeding season are at the greatest risk.

4.2.3.2 Double-Crested Cormorant

Results for the double-crested cormorant are similar to those described for brown pelicans. Oral exposure distributions were compared to literature-derived NOAELs and LOAELs, and primary benchmarks (i.e., those based on single-chemical, chronic studies with multiple doses) were not exceeded. However, all modeled oral exposures exceeded secondary, but more receptor-appropriate, NOAELs for tDDT for breeding and non-breeding cormorants. For tPCB, all oral exposures for the breeding season exceeded the secondary NOAEL and 96 percent of exposures for the non-breeding season exceeded the secondary NOAEL. Additionally, 20 percent and 5 percent of modeled oral exposures exceeded the tDDT secondary LOAEL for non-breeding and breeding cormorants, respectively, and less than 1 percent exceeded the tPCB secondary LOAEL for non-breeding cormorants. A greater percentage of exceedances for cormorants compared to pelicans is expected because the more contaminated flatfishes make up a portion of the cormorant diet. Spatial analysis of these exposures indicates that areas around the breeding islands (Anacapa and Santa Barbara) have low exposure and that exposure is reduced during the breeding season when foraging is restricted to these locations. During the non-breeding season, cormorants forage throughout the SCB to depths of 22 m. Individuals foraging within the PVS study area ingest prey with higher tDDT concentrations resulting in exceedance of the tDDT LOAEL. Exceedances of the tDDT LOAEL during the breeding season occur in the Anacapa colony and are related to foraging along the mainland coast by these birds. Individuals from the Santa Barbara colony do not forage near the mainland during the breeding season.

Eggs collected from Anacapa exceeded NOECs for tDDT and the LOEC for tPCB. Although the maximum and 95 percent UCL concentrations exceeded the tDDT LOEC, the mean value did not. Finally, while the populations of cormorants nesting on Santa Barbara and Anacapa islands are considered stable, chronic low-level exposure to DDE is likely depressing reproduction and some individuals still have egg concentrations above thresholds for reproductive failure.

Therefore, the weight-of-evidence is that both tDDT and tPCB present risk to individual double-crested cormorants, but the risk from tPCB is expected to be lower than that from tDDT. Individuals primarily foraging within the PVS study area during the non-breeding season are at the greatest risk, as are individuals foraging on the mainland across from Anacapa Island during the breeding season.

4.2.3.3 Peregrine Falcon

Oral exposure distributions were compared to literature-derived NOAELs and LOAELs. For primary benchmarks, 3 and 12 percent of oral exposures exceeded the tDDT and tPCB NOAELs, respectively, whereas less than 1 percent exceeded the tDDT and tPCB LOAELs. All modeled oral exposures exceeded the tDDT secondary NOAEL and one of three secondary PCB NOAELs. Less than 0.1 percent of the modeled exposures exceeded one of the secondary PCB LOAELs, and the tDDT LOAEL was exceeded by 16 percent of the modeled exposures. Spatial analysis of these exposures indicates that peregrines with the most risk from tDDT are those ingesting prey that forage within the PVS study area, while a larger exposure area for tPCB exists including the area around San Diego Bay. Breeding colonies of western gulls exist on Santa Catalina, Prince (off San Miguel), Santa Barbara, San Nicolas, Anacapa, Santa Cruz, and Gull (off Santa Cruz) islands. Therefore, breeding peregrines on Prince, Santa Barbara, Anacapa, and Santa Cruz islands likely capture much of their prey from the breeding gulls on their islands, but some peregrine aeries are close enough that foraging occurs on multiple islands. Given the foraging range of the western gull (generally 20 km, but not greater than 80 km), most peregrines would be exposed to the lower range of dosages in the seabird prey. However, even these lower concentrations in prey result in 100 percent exceedances of NOAELs and some LOAEL exceedances across the SCB. This suggests that risk to peregrines is possible. Additionally, concentrations in peregrine falcon eggs collected from Anacapa, San Miguel, and Santa Cruz islands exceeded the tDDT LOEC (no NOEC was available) and both the tPCB NOEC and LOEC. Field studies in the Channel Islands indicate that peregrines are suffering reproductive impairment and are not able to maintain a stable population at some breeding locations without intervention.

Therefore, the weight-of-evidence is that both tDDT and tPCB present risk to individual peregrine falcons nesting within the SCB, and that adverse effects are probable. Risk is not uniform throughout the SCB, with peregrines nesting in the Northern Channel Islands at lower risk than those nesting around the Los Angeles Harbor.

4.2.3.4 Bald Eagle

Primary benchmarks for oral exposure were not exceeded, but secondary (and likely more appropriate) benchmarks were exceeded. All modeled tDDT oral exposures exceeded the secondary tDDT NOAEL and two of the three secondary tPCB NOAELs. The tDDT LOAEL, one tPCB NOAEL, and tPCB LOAEL benchmarks were not exceeded. This indicates that risk from exposure to tDDT and tPCB is possible. Because seabird and sea lion prey items are likely to forage within the PVS study area (due to the close proximity of Santa Catalina Island to the PVS study area), exposure from these portions of the bald eagle diet present the greatest risk. Moreover, NOECs and LOECs for tDDT and tPCB were exceeded in egg tissue collected at Santa Catalina bald eagle nests, and bald eagles are not breeding without intervention.

Therefore, the weight-of-evidence is that both tDDT and tPCB present risk to individual bald eagles nesting within the SCB, and that adverse effects are probable.

4.2.4 Marine Mammals

In this subsection, each line of evidence (external and internal exposure and site-specific field studies) is evaluated and a weight-of-evidence level of risk is determined for California sea lions. As with the avian receptors, the California sea lion is a California state-protected species and is therefore protected at the individual level. Therefore, exceedance of the NOAEL/NOEC indicates risk to these individuals. Exceedance of the LOAEL/LOEC further supports this risk and indicates that risk to populations is probable.

4.2.4.1 Adult Female California Sea Lion

None of the modeled tDDT oral exposures for sea lions exceeded the primary NOAEL (no LOAEL was available) and none of the modeled tPCB oral exposures exceeded either of the primary NOAEL or LOAEL in any season. However, some modeled oral exposures exceeded secondary NOAELs for tDDT and tPCB. For tDDT the percentage of modeled oral exposures exceeding the NOAEL was highest in the fall, lowest in the winter/spring, and intermediate in the summer. The percentage for tPCB differed little among the three seasons. As noted for breeding and non-breeding birds, the difference in the percentage of modeled oral exposures exceeding benchmark toxicity values is largely due to changes in seasonal foraging areas. California sea lions are likely to eat fish with higher tDDT tissue concentrations in the fall than the winter/spring and summer because they spend more time foraging closer to the PVS study area in the fall. During the breeding season, higher exposure (relative to other areas in the same season) occurs around the breeding islands (San Miguel and San Nicolas) and within the PVS study area, which appear to have similar levels of weighted exposure. This reflects the heavy use of areas with low fish tDDT concentrations around the Channel Islands and the lower use of more heavily contaminated areas in the PVS study area. This suggests that tDDT in the PVS sediment, on a seasonal cycle, pose possible risk in the California sea lion through ingestion of tDDT in fish tissues, because the sediment at PVS can be a source of tDDT in fish tissues. This trend is not as apparent for tPCB.

tDDT and tPCB stored in blubber can serve as a source for internal exposure. Therefore, risk from internal exposures of tDDT and tPCB was estimated as an additional line of evidence. Only estimated DDE concentrations in blubber collected from premature parturient females in 1991 exceeded the threshold. Data from the 1970s are not evaluated in the risk characterization because they do not represent current or recent conditions in the SCB. Neither group of females sampled in 1991 had estimated PCB concentrations in blubber that exceeded the benchmark.

There are conflicting reports on the status of California sea lion populations over time, but all agree that sea lion populations are increasing and contaminant levels are decreasing. However, immigration of individuals from Mexico and the impact of federal protection of sea lions could not be excluded as confounding factors in explaining this increase.

Therefore, the weight of evidence is that tDDT and tPCB present risk to individual adult female California sea lions; however, population-level effects may not be occurring (i.e., sea lion populations are increasing). Sea lions foraging within the PVS study area and around the breeding islands during the summer and winter/spring for tDDT are at the greatest risk.

4.2.4.2 Nursing California Sea Lion Pups

Less than 1 percent of the modeled tDDT oral exposures for sea lion pups in fall and winter/spring and none of the modeled tDDT in summer and tPCB in any season exceeded the primary NOAEL or LOAEL. However, most modeled oral exposures exceeded secondary NOAELs for tDDT and tPCB across all seasons. For sea lion pups, the percentage of oral exposures exceeding the tDDT NOAEL was approximately equal in the fall and winter/spring and less in summer, though these differences were not great (range 1 to 2 percent). For tPCB, fall had the greatest percentage of exceedances, but this was very similar to winter/spring (less than 1 percent), and summer had no exceedances. These results suggest that adult females pick up concentrations sufficient to pose risk to sea lion pups in some of their foraging areas within the SCB. However, it should be noted that the toxicity benchmark for sea lions is based on adverse effect to immune function. Therefore, direct mortality is not likely from these exposures. However, sea lion pups with impaired immune function are susceptible to sudden stress events such as disease or food shortages (as occur in El Niño events).

Spatial analysis of the exposure distribution indicates that DDT exposure is highest in the PVS study area and around the breeding islands, where DDT concentrations in prey are high and where sea lions spend most of their time foraging, respectively. In contrast, tPCB are primarily being accumulated in the adult female sea lion from contaminated prey in the high use foraging areas around the breeding and loafing islands and not as much from the PVS area that receives less use. As indicated above, there are conflicting reports on the status of California sea lion populations over time; but all agree that sea lion populations are increasing and contaminant levels are decreasing.

Therefore, the weight of evidence is that tDDT and tPCB present risk to individual nursing California sea lion pups; however, population-level effects may not be occurring (i.e., sea lion populations are increasing). However, pups are likely experiencing adverse effects to their immune system that could result in high mortality when exposed to increased environmental stress (e.g., diseases or adverse weather).

4.3 Uncertainty Analysis

Uncertainties are inherent in all aspects of an ERA. The nature and magnitude of uncertainties depend on the amount and quality of the data available, the degree of knowledge concerning site conditions, and the assumptions made to perform the risk assessment. The weight-of-evidence approach to characterization of risk is effective in reducing uncertainty when the number of risk indicators is not contradictory, particularly with respect to location and magnitude of risk. A qualitative evaluation of the major uncertainties associated with this assessment, in no particular order of importance, is summarized below.

4.3.1 Problem Formulation

Adequacy of COPEC selection and chemical characterization. Risks at the Palos Verdes site have been evaluated only for tDDT and tPCB. Uncertainties associated with the calculated hazard indices may exist because they may not reflect all chemicals or activities of chemical

mixtures. The exclusion of other COPECs may have underestimated risks, and may have obscured the spatial extent and area of greatest risk.

Selection of representative species. Representative species were selected to reduce uncertainty and to focus on species that are both maximally exposed and representative of the wildlife using the site. However, differences between species, including physiology, reproductive biology, or foraging habits, can result in different exposures and sensitivities to different chemicals.

4.3.2 Analysis

Uncertainties associated with the analysis include those for the exposure characterization and the ecological effects characterization.

4.3.3 Exposure Characterization

Adequacy of fate and transport evaluations, including sample station selection, spatial sediment patterns (horizontal and vertical), and representative sample selection. The available data include vertical and horizontal samples covering a wide area. However, despite the consideration of multiple studies, spatial data gaps still remain for large (i.e., > 1 km^2) areas. Depending on chemical conditions at unmeasured locations, risks could either be underestimated or overestimated.

Adequacy of characterization of sediment disturbance events. This risk assessment addresses potential ecological effects associated with sediment contamination under equilibrium conditions that are not influenced by episodic small-scale disturbance such as infaunal and epifaunal bioturbation or localized erosion/deposition events. Although these phenomena are known to occur, the risk evaluation does not specifically consider their impacts beyond the degree to which they are represented by the mean condition observed at the site. This evaluation examined only large-scale contaminant distribution and potential biological effects.

Reliability of exposure point estimation methods, including porewater calculation methods. Concentrations of tDDT and tPCB in porewater were based on equilibrium partitioning predictions, measured sediment TOC concentrations, and sediment-to-porewater partitioning constants (K_{oc}). Uncertainty exists in the accuracy of these values for porewater prediction, possibly resulting in under- or overestimation of porewater concentrations and associated exposure to target receptors. Concentrations of tPCB in sediment and fish tissue samples were calculated from PCB congeners and Aroclor mixtures. However, the same congeners and Aroclors were not always available in the source data. A few data sources reported tPCB as tPCB, and not as a mixture of congeners. Therefore, uncertainty exists in the accuracy and comparability of calculated tPCB concentrations.

Estimation of COPEC concentrations in whole-body fish. Most whole-body fish concentrations were estimated based on a multi-species model for predicting whole-body COPEC concentrations from those in fillets. The relationship was not developed using PVS or SCB species; therefore, risk may be underestimated or overestimated. Future analytical results on SCB fillet and whole-body comparisons can be used to improve the accuracy of this estimate.

Adequacy of exposure model for birds. There is uncertainty associated with the estimation of risks to avian predators because multi-year averages of fish residue data were used instead of lowest or highest values. Although there is variability from year to year in this data set,

the absence of clear trends in decreasing concentrations with time did not allow for the selection of the most recent data for the risk analysis. If tissue concentrations do decrease in time, risk estimates would be concomitantly lower than those presented.

Adequacy of bioaccumulation and tissue-based effects models. Tissue residues can be used as an indication of exposure; however, their importance in ecological risk assessments is currently limited since evidence linking ecological effects directly with contaminant concentrations in tissue is limited. In addition, a more complete understanding of bioaccumulation and trophic transfer is required to evaluate the role of tissue residues in the status of natural resources, and to provide data for evaluating risks to wildlife associated with fish consumption.

Adequacy of literature-derived life-history parameters. Some food ingestion rates and body weights of avian and mammalian receptors were not specific to the SCB; therefore, exposure parameters were either modeled based on allometric relationships (e.g., food ingestion rates) or were based on data from the same species in other portions of its range. Because these values can differ among individuals and locations, published parameter values may not accurately reflect individuals present at the site. As a consequence, risk may be either overestimated or underestimated.

Adequacy of surrogate diet for bird and mammal receptors. Assumptions were made on percent of diet represented by fish species with modeled relationships to sediment (i.e., white croaker, kelp bass, Dover sole, and sanddabs). The extent to which these surrogate species represent concentrations of COPECs in other prey species is unknown; therefore risk may be over-estimated or underestimated.

4.3.4 Ecological Effects Characterization

Adequacy of tDDT water quality effects benchmarks. USEPA (1980a) established a saltwater criterion value for DDT that related to predicted toxicity from exposure to the 2,4′ and 4,4′ isomers of DDT. Toxicity data for total DDE (2,4′ and 4,4′ isomers) were summarized, but due to lack of available data a formal criterion value was not published. The existing data, however, do indicate that DDE may be less toxic than is DDT, although the degree of that difference is not well defined. The criterion value applied to the water quality benchmark for this evaluation employed the DDT value. Given that the DDE isomers represent approximately 70 percent of the tDDT mixture, the benchmark value should be viewed as a low threshold for effects and one that is conservative (protective).

Adequacy of sediment quality benchmarks to quantify risk. SEC-based porewater benchmark concentrations for tPCB and tDDT are based on the SECs derived in MacDonald (1997). SEC values (2,000 µg/kg for tDDT; 577 µg/kg for tPCB) were normalized to 1 percent TOC. However, TOC in sediment at PVS was measured between 1 and 7 percent. To compensate, the SEC was adjusted to account for the station-specific TOC concentrations. If none were available, a conservative value of 1 percent was used. Risk may be under- or overestimated when applying the SEC-based porewater benchmark where measured sediment TOC was unavailable. Because porewater concentrations were calculated, they are a less robust measurement of risk than measured concentrations. Therefore, risk may be over- or underestimated. SECs derived in McDonald (1997) were calculated for the PVS, and are therefore better estimators of effects thresholds than values developed for the entire SCB, or the United States overall (i.e., CCC values).

Adequacy of benthic community evaluations. The utility of community structure measurements, such as diversity, evenness, richness, similarity, and trophic structure, has considerable uncertainty with regard to applicability for detection of COPEC-related impacts. For instance, the seasonal and temporal sensitivity to pollutants has not been assessed, and leads to uncertainty given various life stages present at different times that may have different chemical sensitivity. The effluent-affected nature of the contaminated sediment deposit may also be a factor influencing these measurements. In addition, knowledge of community ecology, including natural history (e.g., size of feeding range and site use) of many species, species sensitivities to contaminants and trophic transfers, as well as natural changes and variability in biological/ecological systems is incomplete.

Appropriateness of wildlife toxicity benchmarks. Literature-derived toxicity data based on laboratory studies were used to evaluate risk to birds and marine mammals. It was assumed that effects observed in laboratory species were indicative of effects that would occur in wild species. The suitability of this assumption is unknown. Consequently, risk may be either overestimated or underestimated.

Primary toxicity studies were based on single-chemical, chronic laboratory studies with multiple doses and a clear dose-response relationship. Test species were standard laboratory species (i.e., mallard, chicken, rat, mink) that may not be representative of the receptors evaluated. Mink, and rats in particular, may not be appropriate surrogates for marine mammals because of the very different levels of fat content in these species. This difference may result in under- or overestimation of risk.

Secondary studies were based on laboratory (e.g., screech owl and American kestrel), field (e.g., brown pelican), or semi-field (e.g., harbor seal) studies with test species more closely related to the receptors evaluated. However, field studies introduce confounding factors associated with exposure to multiple chemicals in an uncontrolled environment, making it difficult to attribute an effect to one chemical. Additionally, all of these studies have only one dosage level; therefore, effects associated with concentrations above or below these dose levels are not known. Finally, studies from other areas (i.e., those on the harbor seal) have a different mixture of chemicals than those in the SCB, resulting in different synergistic or antagonistic forces on the COPECs evaluated. All of these factors may result in either under- or overestimation of risk.

Adequacy of extrapolation factors. Because toxicity data are not available for all avian and mammalian receptor species considered in this assessment, it was necessary to extrapolate toxicity values from test species to site receptor species. Although improved class-specific scaling factors were employed (Sample and Arenal 1999), these factors are not chemical-specific and are based on acute toxicity data. As a consequence, risk may be either overestimated or underestimated.

Adequacy and availability of national/regional benchmarks as reference toxicity values. Estimation of the magnitude of COPEC-related effects in the present risk assessment is based primarily on numerical comparisons of measured or predicted concentrations in sediment and water against national/regional sediment- and water-based benchmarks. For sediments and porewater, comparisons were made against regional benchmarks in order to cover a range of potential effects predictions. Porewater and water quality benchmarks are derived from national chronic water quality criteria. However, the true, site-specific value for protection

of target receptors is not known, leading to possible under- or over-estimation of risk for sediment-based exposure.

4.3.5 Risk Characterization

Use of HQ Approach. Potential ecological risks were quantified using the HQ approach. The magnitude of the HQ indicates potential for ecological risk, but is not an exact estimation of risk. For example, the actual risk from a chemical with an HQ of 70 could be less than that for a chemical with an HQ of 20 because of uncertainties involved in estimating exposure, selection of effects criteria, or other field conditions.

COPEC Evaluation Approach. In this assessment, risks from COPECs were considered independently for birds and mammals. Because chemicals may interact in an additive, antagonistic, or synergistic manner, evaluation of single-chemical risk may either underestimate or overestimate risks associated with chemical mixtures.

SECTION 5

Conclusions and Recommendations

Several lines of evidence, including sediment and porewater hazard quotients (HQs), benthic community effects, toxicity tests, effects on fish, and modeling of food-chain transfer to birds and mammals, were evaluated as part of this risk assessment. The results show highest risks in the vicinity of the PVS outfalls. Intermediate-risk areas are found generally to the south and southwest of the outfalls, as well as areas to the northwest off Point Vicente extending to Santa Monica Bay. Finally, low-risk areas occur at the far northern areas of the PVS study area and throughout the SCB. Results for birds and mammals indicate some far-reaching risks, including out to the Channel Islands, driven by their seasonal patterns of areal use.

The effects on marine birds and mammals are ultimately based on measured sediment concentrations throughout the SCB. However, fish tissue data at PVS confirm the pattern of highest bioaccumulation in the immediate area of the outfalls and the link between fish and sediments. In addition, benthic community and toxicity data confirm the basic patterns of risk. The benthic invertebrate data were evaluated for potential non-contaminant-related impacts (e.g., depth, total organic carbon [TOC]). Although non-contaminant-related effects exist, there are increased benthic impacts with increasing COPEC exposure; therefore, exposure to COPECs must be included as one of the important mechanisms explaining impairment of the benthic community.

The co-occurrence of benthic effects, as evidenced by community responses and sediment toxicity as well as fish tissue patterns, lends strong support to the establishment of the zone of highest relative risk at PVS. Sediment and predicted porewater data used to estimate potential risk are unequivocal. This agreement between exposure and effects lines of evidence, as well as the spatial representation of risk with respect to the presumed source of COPECs (i.e., wastewater released through the outfalls), provides confidence in the interpretation of the degree and spatial extent of risk. Areas of higher relative risk also represent sources of greater contamination to the food web, such as prey species consumed by birds, and are therefore of greater concern than areas of lower contamination.

The modeled results of bird and sea lion exposure levels throughout the SCB indicate that the risk to these higher-order consumers is significantly linked to exposure through contaminated sediments at PVS, but is modified by the changing seasonal distribution patterns of the receptors. Risks are small in some areas but measurable at locations throughout the SCB.

5.1 Ecological Significance

Given the risk estimates, their associated uncertainties, and the risk descriptions, it was established that the PVS study area poses risks that have ecological significance. Three major criteria were used to evaluate adverse changes in the assessment endpoints:

- The nature and intensity of effects
- Spatial and temporal scales
- Potential for recovery

It is important to consider the nature and intensity of effects to distinguish ecological changes that are adverse from ecological events that are within the normal pattern of ecosystem variability. The HQ method utilized in this document relies on effect-based benchmark values from the literature that are generally protective of systems or groups of organisms. For example, USEPA water-quality criteria are designed to be protective of 95 percent of aquatic organisms, which include benthic infauna, fish, and plankton. Similarly, other effect-based benchmarks are used to protect fish, fish-eating birds, and marine mammals from sediment and dietary exposure. Literature benchmarks were generally based on survival and/or reproductive effects, though some benchmarks for marine mammals are based on immune function effects. Thus, the HQs largely reflect risks to survival and reproduction of species and have ecological significance when values consistently exceed 1, and become especially important at order-of-magnitude exceedance levels. Risks from exceedances of immune function-based benchmarks are less clear, but may exist during stressful events.

Various lines of evidence point to sediment and water-column toxicity from COPECs to benthic invertebrates at PVS. The magnitude of the SEC HQs for tDDT was greater than 1 at approximately 22 percent of PVS locations, although HQ values were less than 10. Chronic effects to benthic invertebrates as the result of porewater exposures would be expected, as HQs for tDDT were greater than 1 at 79 percent of PVS locations. Bulk sediment HQs for tPCB followed a similar pattern except with somewhat reduced exceedances for sediment tPCB than for tDDT. For porewater, less than 1 percent of PVS locations had HQs greater than 1. Risks for benthic and infaunal invertebrates at PVS are primarily from tDDT.

Fish also show evidence of exposure and risk at PVS. The percentage of fish tissue samples collected at the PVS show that exposure to tDDT occurs at levels sufficient for risk, whereas exposure to tPCB does not. HQs based on tissue residues show that this risk is likely limited to benthic fish (represented by white croaker, Dover sole, and sanddabs) from the PVS, whereas no risk is posed by tPCB. For fish tissue levels, the tDDT HQs based on chronic effects ranged from less than 1 to 8 and were highest in white croaker, followed by sanddabs, Dover sole, and kelp bass. HQs for water-column-based exposure show that no risk is posed by tPCB, but that exposure to tDDT concentrations may cause risk. That risk is most likely to occur closer to the sediment-water interface. Based on these lines of evidence, no risk from tPCB to fish occurs at the PVS or in the SCB. However, risk from tDDT, while trivial in the SCB as a whole, is possible for Dover sole and sanddabs and probable for white croaker at the PVS.

Fish species can move in and out of the PVS study area. In particular, Dover sole tend to migrate to deeper waters off the shelf. However, the basis for modeling tissue concentrations in these benthically coupled fish is an assumption of equilibrium with exposure conditions throughout the SCB. It is expected that adverse effects may be occurring, and that greater effects may be possible for other benthically coupled organisms with a similar, strong association with sediments and limited geographic range. Birds and mammals are expected to eat fish and invertebrates from the PVS as well as many other SCB locations. In addition, some of their diet is assumed to be uniformly low in COPEC concentrations (pelagic food items). Nevertheless, as seen from the bird and marine mammal exposure estimates, variable types of risks are measurable at the spatial scale of the entire SCB.

External and internal measures of risk show related but variable patterns for birds. The oral exposure model (food web exposure model; see Appendix C) indicated that oral exposures and risks are low for brown pelicans and double-crested cormorants during the breeding season, but that risk increases during other times of the year as their foraging range includes more of the PVS area. Egg data indicate continuing risk to individuals of these species as well as bald eagles and peregrine falcons nesting on the Channel Islands. In combination with site-specific field studies, the weight-of-evidence from these analyses indicates continuing risk from tDDT and tPCB throughout the SCB for the bird species evaluated in this ERA. On the whole, tDDT are associated with greater risk for birds than are tPCB in the SCB.

Adult female sea lions also show evidence of risk from the COPECs from external and internal pathways. Modeled oral exposures indicate seasonal differences, with exposure to PVS area fish being most important to the population in the fall. In comparing estimates of tissue body burdens and oral exposure model estimates, it is not clear whether tDDT or tPCB pose the greater risks to SCB sea lion populations. In both cases, the risks appear to be measurable, but small. As expected, nursing sea lion pups receive high exposure to tDDT and tPCB from adult females via transfer through the milk and these exposures are related to the seasonal foraging areas of the mother. This higher exposure results in greater risk to pups than adult females. However, this risk is associated with probable impaired immune function in sea lion pups from tPCB and tDDT exposure, and population effects would only be expected during stressful events (e.g., disease outbreak or food shortages during El Niño periods).

Each of the taxonomic groups with assessment endpoints (marine infaunal and benthic macroinvertebrate communities, fish, birds and mammal) represents a different spatial scale, temporal scale of response times, and degree of COPEC exposure for assessing risk. They were selected for the risk assessment to determine if adverse effects would be pronounced in the relatively short term or long term, owing to exposure to these two highly persistent COPECs. Contamination of the PVS and SCB ecosystem by tDDT and tPCB has been largely related to historical releases. Although measured contaminant levels in sediments and fish tissue decreased significantly after the early 1970s, levels do not appear to have changed significantly since the mid-1980s. The results of this risk evaluation indicate that existing concentrations of contaminants in the environment continue to pose a risk of adverse effects on portions of this ecosystem and can be expected to persist for many years. Future risks may be expected to change, as related to the burial, transport to deep water, or re-exposure of contaminated sediments. The time frame and direction of that change is, as yet, unknown.

The documentation of ecological risk throughout the SCB food web from tDDT and tPCB demonstrates that the Ecological Management Goals (Section 2.4.1) have not been met. COECs identified from the ERA include both tDDT and tPCB.

5.2 Receptor-Specific Risks

A summary of risk by receptor group follows:

- **Sediment Infauna and Benthic Macroinvertebrates:** There are risks from waterborne, bulk sediment, and sediment porewater tDDT to populations and community structure at PVS. There are lower levels of risk from tPCB from all media.

- **Fish:** There are risks from water- and sediment-borne tDDT to populations at PVS. There is no risk from tPCB from all media.
- **Birds:** There is continuing risk to individuals and populations of seabirds and raptors from tDDT in diet along the SCB coast near PVS, and at Santa Catalina Island for bald eagles. There are lower levels of risk in all areas from tPCB in diet.
- **Marine mammals:** There are seasonally variable, measurable but small risks from tDDT and tPCB in diet to individuals and populations near PVS and in areas of highest population densities on the Channel Islands. There is higher risk to pups resulting from higher exposure through the milk. Also seasonally variable as pup exposures are a result of oral exposures to females from their diet.

5.3 Data Gaps

The following data gaps have been identified through the ERA process for the PVS and SCB study areas:

- Fish tissue concentrations for COPECs need to be analyzed throughout the Channel Islands. SCB-wide samples of market squid should be analyzed for COPEC concentrations.
- Local, SCB-derived relationships between fish fillet concentrations and whole-body concentrations of COPECs are necessary to fully utilize the extensive fillet chemistry database for ecological risk applications.
- More recent tissue chemistry data are needed for birds and marine mammals. These data include whole-body and egg DDT and PCB measurements for seabirds and blubber and/or blood DDT and PCB measurements for sea lions.

Filling these data gaps would enhance our ability to assess and predict ecological risks to higher-order consumers in the SCB. The refinement of the food web exposure model would improve our ability to predict dietary risks to birds and marine mammals. Although filling these data gaps would decrease the uncertainty associated with some model parameters, lack of these data does not preclude the use of the exposure model in its current state. The current food web exposure model can be used to back-calculate preliminary sediment clean up goals. In addition, it allows for examination of sediment quality change scenarios for their effects on ecological risk. Specifically, the model can be run with new sediment quality contours based on capping scenarios. Resultant fish concentrations and dietary exposure values can then be predicted to show relative changes to the marine food web resulting from future changes in PVS sediment quality.

SECTION 6

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APPENDIX A

Database

APPENDIX A

Database

A.1 Introduction

One of the goals for this Ecological Risk Assessment (ERA) and the EcoRisk exposure model was to summarize data from as many different sources as was practical, collected during the years 1990 through 2003 for all media and receptors, except birds, for which data from 1985 to 2003 were collected. Although there was a desire to carefully and fully characterize the Palos Verdes Shelf (PVS) study area for exposure to resident invertebrates and fish, data were also needed to characterize Southern California Bight (SCB)-wide exposure for wider-ranging, higher order consumers (marine birds and mammals). Hence, sediment, fish, water, avian, and mammalian datasets were compiled to be representative of both the PVS and greater SCB areas. Data sources by medium are listed in Table 2-2.

A list of data files is provided on compact disc (CD) with the final copy of the PVS ERA. This is a comprehensive list of the raw data used to make risk determinations for the selected ecological receptors at the PVS. These data were also used as inputs for the food web exposure model (Appendix C). Data files are provided as Microsoft Excel 97 files with accompanying metadata as separate worksheets.

A.2 Sediment Database

The sediment database consists of data from Anderson et al. (1998), Fredette et al. (2002), LACSD (1991-2001), Lee (1994), NOAA (1994), NOAA (unpublished), Noblet et al. (2002), SAIC (2003), SCCWRP (1999-2000), Schiff and Gossett (1997), and USACE (2000). These files can be found on the accompanying data CD in the "Sediment" directory.

Differing degrees of data reduction were performed on each of these datasets to create common fields in the master sediment database (the "SEDIMENT.xls" file). Replicates from the same sample location, within a dataset, were averaged (geometric means) to produce a representative value for each location. For sample concentrations qualified as "non-detect", half the reported detection limit was used. If samples were qualified as "non-detect" and no detection limit was provided, half the reported value was used. See Table A-1 (all tables are located at the end of this section) for a complete list of files containing sediment data, and Table A-2 for metadata associated with these files.

File name: SEDIMENT.xls

Description: Contains reduced DDT and PCB sediment concentrations for the SCB, latitude, longitude, sampling agency, core/layer depth, sampling date, QA qualifiers, MDLs, and miscellaneous comments. This file is the sediment database used by CH2M HILL for the PVS ERA.

***File name*:** Anderson et al 1998 raw data.xls

Description: Sediment samples were collected during 20 separate sampling periods between July 30, 1992 and August 22, 1997. Surface sediment samples were collected using a 0.1 m^2 Young-modified Van Veen grab sampler. tDDT and tPCB concentrations were reported. Only a subset of the available tDDT and tPCB samples from the Bay Protection Toxics Cleanup Program (BPTCP) database were incorporated into the CH2M HILL surface sediment (0-15 cm) database. The samples selected from the BPTCP database were chosen because they represented mid-harbor areas. tDDT and tPCB surface sediment concentrations were calculated by summing the measured DDT isomer and PCB congener concentrations.

File name: Fredette et al raw data 2002.xls

Description: Contains tDDT and tPCB sediment concentrations from the Army Corps of Engineers pilot capping study from the A-III borrow site. tDDT concentrations were reported as the sum of p,p'-DDD, DDE, and DDT. tPCB concentrations were reported as the sum of Arochlors (1016, 1221, 1232, 1242, 1248, 1254, and 1260). These data were incorporated into the CH2M HILL surface sediment (0-15 cm) database.

***File name*:** LACSD 1990-2001 raw data.xls

Description: Sediment cores were collected by LACSD in 1991, 1993, 1995, 1997, and 1999. Of the six DDT isomers, only p,p'-DDE concentrations were reported. These cores sampled sediment to at least 79 cm below the sediment-water interface, and concentrations from multiple strata were reported. However, data were reduced to fit into a 0-15 cm surface layer and a 15-30 cm deep layer by calculating the average sediment DDE concentrations over those layers using percent moisture as a weighting factor. These data were also incorporated into the CH2M HILL sediment database.

***File name*:** Lee 1994 raw data.xls

Description: USGS performed a multi-step, integrated investigation of the continental shelf, slope, and basin adjacent to the Palos Verdes Peninsula to map the distribution of DDT, PCBs, and other chemical and physical properties in the sediment (1992-1993). Vibracores, box cores, and gravity cores were collected during different studies. Vibracores were collected to characterize native sediment (underlying the effluent-affected sediment), whereas box and gravity cores taken at approximately 60 stations provided the basis for mapping the effluent-affected sediment layer. Box and gravity cores from 37 stations were analyzed for DDT and PCBs. These data were incorporated into the CH2M HILL sediment database as tDDT and tPCB; the data were reduced to fit into a 0-15 cm surface layer and a 15-30 cm deep layer by calculating the average sediment DDE concentrations over those layers using percent solids as a weighting factor.

***File name*:** NOAA 1994 raw data.xls

Description: Data were collected in 1990 and 1991 by gravity core from surface sediments along the California coastline. tDDT and tPCB surface sediment concentrations were calculated by summing the measured DDT isomer and PCB congener concentrations, respectively. These data were incorporated into the CH2M HILL surface sediment (0-15 cm) database.

File name: NOAA (unpub) raw data.xls

Description: Data were collected for the bioeffects assessment program from San Pedro Bay, Los Angeles Harbor, Long Beach Harbor, Anaheim Bay, Alamitos Bay, and Huntington Harbour. These data were used to characterize the magnitude and relative spatial extent of toxicant-associated bioeffects in these nearshore areas, to determine relationships between concentrations and mixtures of sediment-associated toxicants and the occurrence and severity of bioeffects, and to distinguish more severely impacted sediments from less severely impacted sediments. These data were collected in 1984, 1985, 1989, 1990, and 1991; however, only those data collected during 1990 and 1991 were incorporated into the CH2M HILL surface sediment (0-15 cm) database because data collected before 1990 were not considered to represent current conditions. tDDT and tPCB surface sediment concentrations were calculated from measured DDT isomer and PCB congener concentrations.

***File name*:** Noblet et al 2002 raw data.xls ("Bight 98")

Description: In the summer of 1998, surface sediments from 290 sites in the SCB (between Point Conception and the U.S.-Mexico international border) were collected using a 0.1 m^2 Young-modified Van Veen grab sampler. tDDT and tPCB concentrations were reported. These data were incorporated into the CH2M HILL surface sediment (0-15 cm) database.

***File name*:** SAIC (2003) raw data.xls

Description: Two cores were collected during the March 2002 survey of the pilot capping cells at the PVS. The cores were collected from uncapped locations between the cells and the Y-shaped diffuser on the JWPCD outfall. Cores were collected using a box core, and subsamples were taken for DDE analysis. Only 4,4'-DDE sediment concentrations were reported (no other isomers). The average DDE concentration of the subsamples was used as an approximation of tDDT in the surface sediment layer (0-15 cm). These data were incorporated into the CH2M HILL surface sediment (0-15 cm) database.

***File name*:** SCCWRP 1999-2000 raw data.xls

Description: SCCWRP collected box cores in June, 1997, as part of a study to assess contaminant concentrations in Santa Monica Bay sediments. DDT isomer and PCB congener concentrations were measured in these sediment samples. tDDT and tPCB were calculated by summing the associated isomers and congeners, for the surface sediment layer (0-15 cm). These data were incorporated into the CH2M HILL surface sediment database.

***File name*:** Schiff and Gossett 1997 raw data.xls ("Bight 94 Pilot Project")

Description: In the summer of 1993, surface sediments from 261 sites in the SCB (between Point Conception and the U.S.-Mexico international border) were collected using a 0.1 m^2 Young-modified Van Veen grab sampler and a Teflon or HDPE scoop or syringe. tDDT and tPCB concentrations were reported. Samples analyzed by the Pace Laboratory were removed from the dataset. The remaining data were incorporated into the CH2M HILL surface sediment (0-15 cm) database.

***File name*:** USACE (2000) raw data.xls

Description: Contaminant sediment analyses were performed on nearshore sediment samples as part of the sediment surveys performed for an Army Corps of Engineers feasibility study. Samples were collected with a Vibra-core from five nearshore locations in Portuguese Bend and one location in Smuggler's Cove. 4,4'-DDD, 4,4'-DDE, and 4,4'-DDT concentrations, as well as tDDT calculated from these isomers, was reported for the upper, middle, and lower subsamples from each core. The upper subsample was incorporated into the CH2M HILL surface sediment (0-15 cm) database.

A.3 Fish Tissue Database

The fish tissue database consists of data from Anderson et al. (1998), OEHHA (2001), Costa et al. (1994), LACSD (1990-2001), Hansen and Associates (2000), Connolly and Glaser (1994), SMBRP (1992), and Noblet et al. (2002). These files can be found on the accompanying data CD in the "Fish" directory.

Differing degrees of data reduction were performed on each of these datasets to create common fields in the master fish database (the "FISH.xls" file). Replicates from the same sample location, within a dataset, were averaged (geometric means) to produce a representative value for each location. For sample concentrations qualified as "non-detect", half the reported detection limit was used. If samples were qualified as "non-detect" and no detection limit was provided, half the reported value was used. See Table A-3 for a complete list of files containing avian data, and Table A-4 for metadata associated with these files.

File name: FISH.xls

Description: Contains DDT and PCB fish tissue concentrations for the SCB, latitude, longitude, sampling agency, species, gender, tissue type, sampling date, QA qualifiers, MDLs, and miscellaneous comments. This file is the fish tissue database used by CH2M HILL for the PVS ERA.

File name: Anderson et al 1998 raw data.xls

Description: Contains tPCB and tDDT fillet concentrations for a small set of fillet samples from white croaker. These data were selected to represent mid-harbor locations, from a larger dataset collected as part of the Bay Protection Toxics Cleanup Program (BPTCP). Fillet concentrations were converted to whole body concentrations (see Appendix C) and used as EPCs in the exposure model. These data were incorporated into the CH2M HILL fish tissue database.

File name: OEHHA 2001 raw data.xls

Description: Contains tPCB and tDDT fillet concentrations for several fish species, including kelp bass and white croaker, from the Coastal Fish Contamination Project. Data were converted to whole body concentrations (see Appendix C) and used as EPCs in the exposure model. These data were incorporated into the CH2M HILL fish tissue database.

File name: Costa et al 1994 raw data.xls

Description: These data were queried from the FIS_MRG ("Merge Data") database file that was produced as part of the NRDA data CD, following the completion of the Natural Resources Damage Assessment for the Southern California Bight. The NRDA data CD accompanied the Southern California Bight Expert Reports, provided by the Trustees of the NRDA settlement (US Dept of Commerce, NOAA, US Dept of the Interior, US Fish and Wildlife Service, California State Lands Commission, California Dept of Fish and Game, and the California Dept of Parks and Recreation). The Merge Data contains reduced (merged) dual column data, calculated tDDT and tPCB data, percent Moisture or percent Solid, and percent Lipid. The lab results are listed as one record per analyte and LabID. Data were converted to whole body concentrations (see Appendix C) and used as EPCs in the exposure model. These data were incorporated into the CH2M HILL fish tissue database.

File name: LACSD 1990-2000 raw data.xls

Description: Fish trawls were conducted as part of the NPDES permit for the Joint Water Pollution Control Plant (JWPCP). Trawls were conducted at three non-contiguous sampling zones. tDDT and tPCB concentrations in muscle tissue (i.e., fillet) from white croaker, kelp bass, and Dover sole were reported. Data were converted to whole body concentrations (see Appendix C) and used as EPCs in the exposure model. These data were incorporated into the CH2M HILL fish tissue database as geometric means. They are categorized by species and sampling location. The sampling zones were converted to an appropriate mid-point location for use in the EcoRA exposure model.

File name: Hansen and Associates 2000 raw data.xls

Description: A three-part study was conducted over the course of one year to determine whether consumption of white croaker would be harmful to anglers fishing on and around the PVS. White croaker fillet data collected during this effort were converted to whole body concentrations (see Appendix C) and used as EPCs in the exposure model. These data were incorporated into the CH2M HILL fish tissue database.

File name: Connolly and Glaser 1994 raw data.xls

Description: These data were queried from three files in the "Connolly" sub-directory as part of the NRDA data CD, which accompanied the Southern California Bight Expert Reports, provided by the Trustees of the NRDA settlement (US Dept of Commerce, NOAA, US Dept of the Interior, US Fish and Wildlife Service, California State Lands Commission, California Dept of Fish and Game, and the California Dept of Parks and Recreation). This dataset was produced by merging the three files contained within the "Connolly" sub-directory. The files were merged on the "ID" field. Only relevant species (Dover sole, kelp bass, and white croaker) were retained in the dataset. Data duplicated in other datasets (i.e. LACSD) were removed before processing. Merged and reduced data were converted to whole body concentrations (see Appendix C) and used as EPCs in the exposure model. These data were incorporated into the CH2M HILL fish tissue database. The following are descriptions of the original three files (PHYS.PRN, PCB.PRN, and PEST.PRN):

> *PHYS.PRN*: File consists of one table containing sample collection and conventional data on invertebrates and fish. Table contains ID number, agency (agency responsible for data collection), segment (segment within

Bight), latitude, longitude, species, species code, tissue type, date, percent lipid, percent water, and flag (indicates whether field or lab data). Bottom of file contains a key for the species, tissue, and flag columns. A value of -9 in the date or segment fields indicates the date/data is unknown or not available. A value of -999 in other fields indicates that data was not reported or not available. A negative value other than -999 in the concentration fields indicates that the measured value was below the analytical limit of detection. The value given is the limit of detection.

PCB.PRN: File consists of one table containing tissue concentrations of Aroclor for aquatic invertebrates and fish. Units are nanograms/gram (ng/g) wet weight. Table contains sample ID numbers, Aroclor, and tPCB concentrations. The values of -9 and -999 represent the same values as in PHYS.PRN.

PEST.PRN: File consists of one table containing concentrations of DDD, DDE, DDT and tDDT for aquatic invertebrates and fish. Units are ng/g wet weight. The values of -9 and -999 represent the same values as in PHYS.PRN.

File name: SMBRP 1992 raw data.xls

Description: Data from the Santa Monica Bay Restoration Project (SMBRP) were originally compiled as part of the NRDA report "Southern California Bight Damage Assessment Food Web/Pathways Study" (HydroQual, Inc. 1997). Fillet data from this dataset were converted to whole body concentrations (see Appendix C) and used as EPCs in the exposure model. These data were incorporated into the CH2M HILL fish tissue database.

File name: Noblet et al 2002 raw data.xls ("Bight '98")

Description: The primary objective of this study was to determine the extent of area on the southern California shelf with contaminant levels of concern. Whole fish composites of sanddab guild species were collected. tDDT and tPCB concentrations were reported. Whole body concentrations were used as EPCs in the exposure model (see Appendix C). These data were incorporated into the CH2M HILL fish tissue database.

A.4 Water Database

The water database consisted of one file of aqueous DDT and PCB concentrations from Zeng and Tran (2002). These files can be found on the accompanying data CD in the "Water" directory. PCB and DDT data fields were merged to produce one file, from the original tables provided by SCCWRP. No data reduction was performed. See Table A-5 for metadata associated with these files.

File name: WATER.xls

Description: Data were collected to investigate the fate of historically discharged DDTs and PCBs in the coastal environment of the SCB. Water samples were collected at different depths from the sea floor, filtered, and analyzed for DDT isomers and PCB congeners. Aqueous concentrations of reported tDDT and tPCB were used as EPCs in the risk analysis. These data were incorporated into the CH2M HILL database, and constitute the water database used for the PVS ERA.

A.5 Avian Database

The avian tissue database consists of data from the PVS NRDA data CD and several other studies (Anderson et al., 1975; Anderson et al., 1977; Elliot and Norstrom, 1998; Elliot et al., 1996a; Elliot et al., 1996b; Elliot et al., 1989; Garcelon 2000; Garcelon and Thomas, 1997; Garcelon et al., 1989; Gress 1994; Gress et al., 1973; Jarman 1993a; Jarman 1993b; Jehl 1973; Lamont et al., 1970; Nelson 1990; Ohlendorf et al. 1982; Ohlendorf et al. 1985; Peakall et al. 1983; Wiemeyer et al., 1984; Wiemeyer et al., 1993). The AVN_INF and AVN_MRG files were supplied on the PVS NRDA data CD. A second set of files (Avian Analytical and Avian Sample) containing data from the studies listed above, were created following the format provided in the AVN_INF and AVN_MRG files. A fifth file (Bird Location Query) contains latitude and longitude data for sample data provided in the AVN_INF files. All five files can be found on the accompanying data CD in the "Avian" directory.

Differing degrees of data reduction were performed on each of these datasets to create common fields in the master avian database (the "AVIAN.xls" file). For sample concentrations qualified as "non-detect", half the reported detection limit was used. In a few cases where only the DDE concentration was given, tDDT was calculated by dividing the DDE concentration by 0.87 because DDE in the SCB makes up approximately 87 percent of the tDDT. tDDT, when not given, was also calculated in some samples by summing the concentrations of the individual DDT isomers. See Table A-6 for a complete list of files containing avian data, and Table A-7 for metadata associated with these files.

File name: AVIAN.xls

Description: Contains DDT and PCB avian tissue concentrations for the SCB, species name, location, tissue type, sampling date, QA qualifiers, MDLs, and miscellaneous comments. This is the final dataset used in the ERA to determine tissue concentrations for avian receptors within and outside the SCB. Egg concentrations were used as EPCs in the risk analysis. This file is the avian tissue database used by CH2M HILL for the PVS ERA.

File name: Avian Sample Data_Raw.xls

Description: Contains the sample information (e.g., species, year, and location) for each sample in the "Avian Analytical.xls " file. These data were input from literature studies (see list above) reporting contaminant levels in tissues of the avian receptors, and includes values within and outside the SCB. These data were incorporated into the CH2M HILL avian tissue database.

File name: Avian Analytical Data_Raw.xls

Description: These data were input from literature studies (see list above) reporting contaminant levels in tissues of the avian receptors, and includes values within and outside the SCB. The contaminant values are contained in this file. These data were incorporated into the CH2M HILL avian tissue database.

File name: AVN_INF_Raw.xls

Description: This file contains the information (e.g., species, year, and location) associated with each avian sample in the NRDA dataset (AVN_MRG_Raw.xls). There is also information on several fish, sea lion, and whole-body avian samples that were collected as food items for the raptors (bald eagles and peregrines). These data were incorporated into the CH2M HILL avian tissue database.

File name: AVN_MRG_Raw.xls

Description: Contains the analytical data for avian samples collected during the NRDA process. Generally, these are egg samples, but some other tissues (e.g., liver, brain) are also included. Additionally, several fish, sea lion, and whole-body avian samples that were collected as food items for the raptors (bald eagles and peregrines) are included. These data were incorporated into the CH2M HILL avian tissue database.

File name: Bird Location Query_Rev.xls

Description: Longitude and latitude values for each sample location were input to enable the spatial analysis of the avian tissue data. These data were incorporated into the CH2M HILL avian tissue database.

A.6 Sea Lion Database

The sea lion tissue database consists of data from DeLong et al. (1973) and Gilmartin (1976). These files can be found on the accompanying data CD in the "Sea Lion" directory. See Table A-8 for metadata associated with these files.

File name: SEA LION.xls

Description: Contains DDE and PCB California sea lion tissue concentrations (and units) for the SCB, pregnancy condition, and sampling year. This is a limited dataset taken directly from DeLong et al. (1973) and Gilmartin et al. (1976). Although blubber samples were collected and evaluated in 1991 during the NRDA process, the values reported for DDTs and PCBs in this study were not used in the current ERA. These data are currently being re-evaluated by NOAA because certain quality control measures were not met (DeLong 2002). However, because the 1991 blubber concentrations are known to be one order of magnitude lower than those recorded in 1970 and 1972 (DeLong 2002), these 1970s data were used to estimate current (i.e., 1991) blubber concentrations in sea lions. The estimated blubber concentrations were used as EPCs in the risk analysis. These data were incorporated into the CH2M HILL sea lion tissue database.

TABLE A-1
File Names for Sediment Data Files
Palos Verdes Shelf Ecological Risk Assessment

File Name	Type
Anderson et al 1998 raw data.xls	raw data
Fredette et al 2002 raw data.xls	raw data
LACSD 1990-2000 raw data.xls	raw data
Lee 1994 raw data.xls	raw data
NOAA (unpub) raw data.xls	raw data
NOAA 1994 raw data.xls	raw data
Noblet et al 2002 raw data.xls	raw data
SAIC (2003) raw data.xls	raw data
SCCWRP 1999-2000 raw data.xls	raw data
Schiff and Gossett 1997 raw data.xls	raw data
SEDIMENT.xls	**CH2M HILL database**
USACE 2000 raw data.xls	raw data

TABLE A-2
Sediment Metadata for Raw Sediment Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Field	Description/Acceptable Values
Schiff and Gossett (1997)	Log Number	Ten digit SCCWRP code
	Station Number	SCCWRP station designation
	NAD 83 LAT	Latitude (degrees, minutes, seconds) reported as NAD 83 datum
	NAD 83 LONG	Longitude (degrees, minutes, seconds) reported as NAD 83 datum
	Parameter Code	Analyte of interest (OPDDE, PPDDE, OPDDD, PPDDD, OPDDT, PPDDT)
	QA Batch	Alphanumeric code that links samples to QA
	Lab Rep	Two or three digit code (S1-original sample; S2-duplicate sample analysis)
	Qualifier	blank - not qualified; nd - not detected; < -estimated, below reported detection limit
	MDL	Method detection limit
	Result	Quantified value for each sample (blank if ND)
	Units	ng/g DW
	Month	Month of sampling
	Day	Day of sampling
	Year	Year of sampling
	QA Code	blank - not qualified; Q - for Station 964 only
	Lab Code	Laboratory doing analysis (HY=Hyperion Treatment Plant; LA = County Sanitation Districts of Los Angeles County; OC = County Sanitation Districts of Orange County; SC = SCCWRP; SD = City of San Diego, Metropolitan Wastewater Department.
	Comments	Sample-specific notes
NOAA (1994)	Site Code	Char(05); 5 letter code site identifier
	STATION	Char(08); An acronym assigned by the responsible labs to distinguish samples within a site.
	Other Codes Used	Char(10); Original 3 letter codes identifying sites used by NS&T participating laboratories
	Month	Char(01); Month
	Day	Char(02); Day
	Year	Char(04); Year
	Depth (m)	Numerical; Sediments were usually sampled from ship or boat(meters)
	Depth (ft)	Numerical; Sediments were usually sampled from ship or boat(feet)

TABLE A-2
Sediment Metadata for Raw Sediment Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Field	Description/Acceptable Values
	Longitude	Numerical; Actual longitude at time of sampling; decimal degrees(decimal degrees)
	Latitude	Numerical; Actual latitude at time of sampling; decimal degrees(decimal degrees)
	Sample Colelction Lab	Char(20); NMFS/Regional Lab acronym/city
	Analyzing Lab-Organics	Char(20); Lab responsible for organic analyses
	Moisture Content	Numerical; Percent moisture of sediments(Percent)
	TOC	Numerical; Total organic carbon(µg/g dry weight)
	Grainsize	Numerical; Amount of fines <64 microns(% fine grain)
	PCB8	Numerical; PCB IUPAC NO. 8(ng/g of dry weight)
	PCB18	Numerical; PCB IUPAC NO. 18(ng/g of dry weight)
	PCB28	Numerical; PCB IUPAC NO. 28(ng/g of dry weight)
	PCB44	Numerical; PCB IUPAC NO. 44(ng/g of dry weight)
	PCB52	Numerical; PCB IUPAC NO. 52(ng/g of dry weight)
	PCB66	Numerical; PCB IUPAC NO. 66(ng/g of dry weight)
	PCB77	Numerical; PCB77 IUPAC NO. 77 extracted and quantified separate of PCB110(ng/g of dry weight)
	PCB11077	Numerical; PCB IUPAC NO. 110 or PCB IUPAC NO. 77; PCB 110 is the dominant congener(ng/g of dry weight)
	PCB101	Numerical; PCB IUPAC NO. 101(ng/g of dry weight)
	PCB105	Numerical; PCB IUPAC NO. 105(ng/g of dry weight)
	PCB118	Numerical; PCB IUPAC NO. 118(ng/g of dry weight)
	PCB126	Numerical; PCB IUPAC NO. 126(ng/g of dry weight)
	PCB128	Numerical; PCB IUPAC NO. 128(ng/g of dry weight)
	PCB138	Numerical; PCB IUPAC NO. 138(ng/g of dry weight)
	PCB153	Numerical; PCB IUPAC NO. 153(ng/g of dry weight)
	PCB170	Numerical; PCB IUPAC NO. 170(ng/g of dry weight)
	PCB180	Numerical; PCB IUPAC NO. 180(ng/g of dry weight)
	PCB187	Numerical; PCB IUPAC NO. 187(ng/g of dry weight)
	PCB195	Numerical; PCB IUPAC NO. 195(ng/g of dry weight)
	PCB206	Numerical; PCB IUPAC NO. 206(ng/g of dry weight)
	PCB209	Numerical; PCB IUPAC NO. 209(ng/g of dry weight)
	OPDDE	Numerical; 2,4'DDE(ng/g of dry weight)
	PPDDE	Numerical; 4,4'DDE(ng/g of dry weight)

TABLE A-2
Sediment Metadata for Raw Sediment Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Field	Description/Acceptable Values
	OPDDD	Numerical; 2,4'DDD(ng/g of dry weight)
	PPDDD	Numerical; 4,4'DDD(ng/g of dry weight)
	OPDDT	Numerical; 2,4'DDT(ng/g of dry weight)
	PPDDT	Numerical; 4,4'DDT(ng/g of dry weight)
	PGM	Char(02); BS refers to the National Benthic Surveillance Project
NOAA (unpublished)	Station Number	Station number
	Site Name	Descriptive station name
	Organizational Identification No.	NOAA internal reference number (1-138)
	Leg Number	1-5
	Longitude	Longitude, in decimal degrees
	Latitude	Latitude, in decimal degrees
	Month	Month sampled (7 - 10)
	Day	Day sampled (1-31)
	Year	Year sampled (1992 - 1993)
	Depth (meters)	0.5 - 82
	OPDDD (ng/g dry weight)	o,p'-DDD concentration
	PPDDD (ng/g dry weight)	p,p'-DDD concentration
	OPDDE (ng/g dry weight)	o,p'-DDE concentration
	PPDDE (ng/g dry weight)	p,p'-DDE concentration
	OPDDT (ng/g dry weight)	o,p'-DDT concentration
	PPDDT (ng/g dry weight)	p,p'-DDT concentration
	PCB8 (ng/g dry weight)	PCB congener #8 concentration
	PCB18 (ng/g dry weight)	PCB congener #18 concentration
	PCB28 (ng/g dry weight)	PCB congener #28 concentration
	PCB44 (ng/g dry weight)	PCB congener #44 concentration
	PCB52 (ng/g dry weight)	PCB congener #52 concentration
	PCB66 (ng/g dry weight)	PCB congener #66 concentration
	PCB87 (ng/g dry weight)	PCB congener #87 concentration
	PCB101 (ng/g dry weight)	PCB congener #101 concentration
	PCB105 (ng/g dry weight)	PCB congener #105 concentration
	PCB118 (ng/g dry weight)	PCB congener #118 concentration

TABLE A-2
Sediment Metadata for Raw Sediment Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Field	Description/Acceptable Values
	PCB128 (ng/g dry weight)	PCB congener #128 concentration
	PCB138 (ng/g dry weight)	PCB congener #138 concentration
	PCB153 (ng/g dry weight)	PCB congener #153 concentration
	PCB170 (ng/g dry weight)	PCB congener #170 concentration
	PCB180 (ng/g dry weight)	PCB congener #180 concentration
	PCB187 (ng/g dry weight)	PCB congener #187 concentration
	PCB195 (ng/g dry weight)	PCB congener #195 concentration
	PCB206 (ng/g dry weight)	PCB congener #206 concentration
	PCB209 (ng/g dry weight)	PCB congener #206 concentration
Lee (1994)	STUDY	USGS SED
	FIELD_ID	A combination of STATION, SAMPTOP, and SAMPBOT
	STATION	101-B5, 102-B1, 106-B1, 108-B2, 109-W1, 109-W2, 111-B1, 113-B1, 114-B1, 115-B2, 117-B4, 120-B1, 121-B1, 122-B1, 123-W2, 124-B1, 125-B2, 127-B1, 128-B1, 130-B1, 131-W1, 132-B1, 136-B1, 137-B1, 138-B2, 139-B2, 141-W1, 143-B1, 146-B1, 147-B3, 148-B1, 149-B1, 153-B1, 155-B2, 156-B1, 157-W1, 159-B1, 160-B1, 163-B1, 166-B1, 169-B1, 171-B1, 173-B1, 174-B1, 177-G2, 179-G3, 181-G2, 187-B1
	SUBCORE	DDTI DDT-1
	SAMPTOP	Depth (cm) of top of sample layer
	SAMPBOT	Depth (cm) of bottom of sample layer
	FRACTION	Empty field
	LAT	Latitude, in decimal degrees
	LON	Longitude, in decimal degrees
	SAMPDATE	month/day/year
	SAMPTIME	0000 - 2359
	W_DEPTH	Water depth of sample station
	UNITS	ng/g dry
	ANALYTE	Total_DDT or Total_PCB
	VALUE	measured value
	LAB_QUAL	NR (no result); blank (quantifiable result)
LACSD Core Samples 1991-1999	Year	1991, 1993, 1995, 1997, 1999
	Month	July (7); blank

TABLE A-2
Sediment Metadata for Raw Sediment Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Field	Description/Acceptable Values
	Day	Numerical day (1-31); blank
	Station	05C, 0C, 1C, 2C, 3BC, 3C, 4C, 4CD, 5C, 5CD, 5DC, 6A, 6B, 6BC, 6C, 6CD, 6D, 6DC, 6Z, 7C, 7CB, 7CD, 75C, 75CD, 8B, 8BC, 8C, 8CB, 8CD, 8DC, 85C, 85CD, 9BC, 9C, 9CD, 10C
	Water Depth (m)	30, 40, 50, 60, 70, 100, 150, 300, 500
	Sed Depth	depth of top of layer in centimeters; 1-85
	Parameter	%MOISTURE P,P'-DDE, SPECIFIC GRAVITY
	Sign	blank (detected); < (less than detection limit)
	Value	measured result of analyses
	Unit	ug/kg, mg/kg, %, Sp Gr (specific gravity)
	Average meas density (1989)	average sediment density for 1991 samples
	1989 density rep1 (g/cc)	sediment density measurement rep1
	1989 density rep2 (g/cc)	sediment density measurement rep2
	1989 density rep3 (g/cc)	sediment density measurement rep3
Anderson et al. (1998)	BPTCP Station #	Sample collection location
	Station Name	Station name
	Lat N	Latitude (degrees.minutes.seconds)
	Long W	Longitude (degrees.minutes.seconds)
	Date	month/day/year
	Analyte	tDDT (tDDT) or tPCB (tPCB)
	Result	Measured value
	MDL	Method detection limit
	Units	mg/kg DW (dry weight)
	TOC (%DW)	Total organic carbon (% dry weight)
SCCWRP (1999-2000) DDTs	Sediment Core	ID number for each sediment core
	Year	Sample collection year
	Layer	Sediment layer (cm)
	Dry wt,g	Dry weight of sample
	Total ng/g	tDDT concentration (dry wt); Sum of detected values
	o,p'-DDE	Analyte-specific sediment concentration; "," denotes concentration below detection limit
	p,p'-DDE	Analyte-specific sediment concentration; "," denotes concentration below detection limit

TABLE A-2
Sediment Metadata for Raw Sediment Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Field	Description/Acceptable Values
	o,p'-DDE	Analyte-specific sediment concentration; "," denotes concentration below detection limit
	p,p'-DDE	Analyte-specific sediment concentration; "," denotes concentration below detection limit
	o,p'-DDE	Analyte-specific sediment concentration; "," denotes concentration below detection limit
	p,p'-DDE	Analyte-specific sediment concentration; "," denotes concentration below detection limit
SCCWRP (1999-2000) PCBs	Sediment Core	ID number for each sediment core
	Year	Sample collection year
	Layer	Sediment layer (cm)
	Dry wt,g	Dry weight of sample
	Total ng/g	tDDT concentration (dry wt); Sum of detected values
	18	Sediment concentration of PCB congener 18; '<' denotes concentration below detection limit
	28	Sediment concentration of PCB congener 28; '<' denotes concentration below detection limit
	52	Sediment concentration of PCB congener 52; '<' denotes concentration below detection limit
	49	Sediment concentration of PCB congener 49; '<' denotes concentration below detection limit
	44	Sediment concentration of PCB congener 44; '<' denotes concentration below detection limit
	37	Sediment concentration of PCB congener 37; '<' denotes concentration below detection limit
	74	Sediment concentration of PCB congener 74; '<' denotes concentration below detection limit
	70	Sediment concentration of PCB congener 70; '<' denotes concentration below detection limit
	66	Sediment concentration of PCB congener 66; '<' denotes concentration below detection limit
	101	Sediment concentration of PCB congener 101; '<' denotes concentration below detection limit
	99	Sediment concentration of PCB congener 99; '<' denotes concentration below detection limit
	119	Sediment concentration of PCB congener 119; '<' denotes concentration below detection limit
	87, 81	Sediment concentration of PCB congener 87, 81; '<' denotes concentration below detection limit

TABLE A-2
Sediment Metadata for Raw Sediment Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Field	Description/Acceptable Values
	110, 77	Sediment concentration of PCB congener 110, 77; '<' denotes concentration below detection limit
	151	Sediment concentration of PCB congener 151; '<' denotes concentration below detection limit
	123	Sediment concentration of PCB congener 123; '<' denotes concentration below detection limit
	149	Sediment concentration of PCB congener 149; '<' denotes concentration below detection limit
	118	Sediment concentration of PCB congener 118; '<' denotes concentration below detection limit
	114	Sediment concentration of PCB congener 114; '<' denotes concentration below detection limit
	153	Sediment concentration of PCB congener 153; '<' denotes concentration below detection limit
	168	Sediment concentration of PCB congener 168; '<' denotes concentration below detection limit
	105	Sediment concentration of PCB congener 105; '<' denotes concentration below detection limit
	138	Sediment concentration of PCB congener 138; '<' denotes concentration below detection limit
	158	Sediment concentration of PCB congener 158; '<' denotes concentration below detection limit
	126	Sediment concentration of PCB congener 126; '<' denotes concentration below detection limit
	187	Sediment concentration of PCB congener 187; '<' denotes concentration below detection limit
	183	Sediment concentration of PCB congener 183; '<' denotes concentration below detection limit
	120, 167	Sediment concentration of PCB congener 120, 167; '<' denotes concentration below detection limit
	177	Sediment concentration of PCB congener 177; '<' denotes concentration below detection limit
	156	Sediment concentration of PCB congener 156; '<' denotes concentration below detection limit
	157, 200	Sediment concentration of PCB congener 157, 200; '<' denotes concentration below detection limit
	180	Sediment concentration of PCB congener 180; '<' denotes concentration below detection limit
	169	Sediment concentration of PCB congener 169; '<' denotes concentration below detection limit

TABLE A-2
Sediment Metadata for Raw Sediment Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Field	Description/Acceptable Values
	170	Sediment concentration of PCB congener 170; '<' denotes concentration below detection limit
	201	Sediment concentration of PCB congener 201; '<' denotes concentration below detection limit
	189	Sediment concentration of PCB congener 189; '<' denotes concentration below detection limit
	194	Sediment concentration of PCB congener 194; '<' denotes concentration below detection limit
	206	Sediment concentration of PCB congener 206; '<' denotes concentration below detection limit
Noblet et al. (2002)	Station ID	SCCWRP 4 digit code for sampling stations
	Depth	Depth samples collected
	Latitude	Latitude where sample was collected
	Longitude	Longitude where sample was collected
	%TOC	Sediment total organic carbon, as percent
	totSedDDT	Sediment DDT concentration, reported as total
	totSedPCB	Sediment PCB concentration, reported as total
	Units	ng/g; ng/g dry; ug/kg
Sediment.xls	Database	Working version of database; Version 3.0 is most current
	Descriptive Name	Descriptive name of data subset (Lee, 1994; NOAA, 1994; etc…)
	Agency/Institution	Agency or Institution name associated with a subset of data.
	Agency/Institution Sample ID	Sample ID carried over from original dataset
	Latitude	Latitude of sample, in decimal degrees
	Longitude	Longitude of sample, in decimal degrees
	Datum	Coordinate system (NAD 83) for Latitude and Longitude
	Day	Day sample collected
	Month	Month sample collected
	Year	Year sample collected
	% Water	% water in sample
	% OC	% OC measured in sample
	Core Horizon	Surface (0-15 cm) or Deep (15-30 cm) sediment layer
	Sample Size	Applicable for locations where multiple samples were collected/co-located

TABLE A-2
Sediment Metadata for Raw Sediment Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Field	Description/Acceptable Values
	Sampling Depth (m)	Water depth at sample location
	Parameter	Analyte of interest
	Result	Concentration
	Units	Units for concentration
	MDL	Method detection limit
	QA Qualifier	Quality assurance qualifier codes for samples (nd = non-detect)
	Comments	Notes/Comments carried over from original dataset or inserted during data reduction steps.
Fredette et al., 2002	Sample ID	Army Corps of Engineers sample identification number (VC00-A3-05; VC00-A3-07)
	Easting	Sample location coordinates
	Northing	Sample location coordinates
	Date	Sample collection date
	Sample Type	Matrix samples (sediment)
	Core Horizon	Sediment depth sampled (0-15 cm)
	n	Number of samples in composite (5)
	Parameter	Analyte name (tDDT, tPCB)
	Result	Measured concentration
	Units	Units for Result (mg/kg DW)
	MDL	Laboratory reporting limit
	Qual	QA qualifier (ND = non detect)
	Comments	Sample specific information
SAIC (2003)	Station ID	Unique sampling location identification number (LUU07, SUU019)
	Month	Month of sample collection (3)
	Year	Year of sample collection (2002)
	Core depth	Segment of core (cm) from which samples were collected (0-4; 4-8)
	Parameter	Analyte name (4,4'-DDE; dry weight unit; wet weight unit; water content; specific gravity)
	Result	Results of analysis
	Units	Units of measurement (mg/kg DW, g/cm^3, %)
	Latitude	Latitude, in decimal degrees

TABLE A-2
Sediment Metadata for Raw Sediment Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Field	Description/Acceptable Values
	Longitude	Longitude, in decimal degrees
USACE 2000	Station	US Army Corps of Engineers station ID
	Core Section	U = upper core; M = middle of core; B = bottom of core
	Analyte	Chemical analyzed; tDDT = sum of isomers
	Units	Units of measurement (mg/kg)
	Result	Chemical concentration; ND = non-detect
	QA Qualifier	U = non detect
	MDL	Method detection limit (0.02 mg/kg for DDT isomers)
	Datum	Coordinate system (NAD 83) for Latitude and Longitude
	Latitude	Latitude in decimal degrees
	Longitude	Longitude in decimal degrees

TABLE A-3
File Names for Fish Tissue Data Files
Palos Verdes Shelf Ecological Risk Assessment

File Name	Type
Anderson et al 1998 raw data.xls	raw data
Connolly and Glaser 1994 raw data.xls	raw data
Costa et al 1994 raw data.xls	raw data
FISH.xls	**CH2M HILL database**
Hansen & Associates 2000 raw data.xls	raw data
LACSD 1990-2001 raw data.xls	raw data
Noblet et al 2002 raw data.xls	raw data
OEHHA 2001 raw data.xls	raw data
SMBRP 1992 raw data.xls	raw data

TABLE A-4
Fish Tissue Metadata for Raw Sediment Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Worksheet	Field	Description/Acceptable Values
SMBRP 1992	Raw data	HQI_Segment	Segment of coastline delineated by HydroQual (Connolly & Glaser, 1994)

TABLE A-4
Fish Tissue Metadata for Raw Sediment Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Worksheet	Field	Description/Acceptable Values
	Raw data	Station	Abbreviated station name (DANAPOIN = Dana Point; EL SEGU = El Segundo; HERMOSA = Hermosa Beach; HYPRION = Hyperion; MALIBU = Malibu; MARIDREY = Marina del Rey; POINDUME = Point Dume; PVERNTH = Palos Verdes North; PVERSTH = Palos Verdes South; SMONICA = Santa Monica; WHITPOINT = White Point)
	Raw data	Station#	Numerical equivalent of Station (196-206)
	Raw data	Latitude	Latitude in decimal degrees
	Raw data	Longitude	Longitude in decimal degrees
	Raw data	Species	WHITCROA = White croaker
	Raw data	HQI_Species#	Numerical equivalent of Species (54)
	Raw data	Tissue	MUSL = Muscle
	Raw data	Month	no data (-999 represents missing data)
	Raw data	Day	no data (-999 represents missing data)
	Raw data	Year	1990
	Raw data	Lipid_%	Reported as % (-999 represents missing data)
	Raw data	WaterContent_%	Reported as % (-999 represents missing data)
	Raw data	opDDD	Reported as ng/g wet weight (-999 represents missing data)
	Raw data	ppDDD	Reported as ng/g wet weight (-999 represents missing data)
	Raw data	opDDE	Reported as ng/g wet weight (-999 represents missing data)
	Raw data	ppDDE	Reported as ng/g wet weight (-999 represents missing data)
	Raw data	opDDT	Reported as ng/g wet weight (-999 represents missing data)
	Raw data	ppDDT	Reported as ng/g wet weight (-999 represents missing data)
	Raw data	tDDT	Reported as ng/g wet weight (-999 represents missing data)
	Raw data	A1016	Reported as ng/g wet weight (-999 represents missing data)
	Raw data	A1221	Reported as ng/g wet weight (-999 represents missing data)
	Raw data	A1232	Reported as ng/g wet weight (-999 represents missing data)

TABLE A-4
Fish Tissue Metadata for Raw Sediment Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Worksheet	Field	Description/Acceptable Values
	Raw data	A1242	Reported as ng/g wet weight (-999 represents missing data)
	Raw data	A1248	Reported as ng/g wet weight (-999 represents missing data)
	Raw data	A1254	Reported as ng/g wet weight (-999 represents missing data)
	Raw data	A1260	Reported as ng/g wet weight (-999 represents missing data)
	Raw data	tPCB	Reported as ng/g wet weight (-999 represents missing data)
OEHHA 2001	Raw data	Sample ID	Unique sample identifier assigned by lab
	Raw data	Station Number	Sample collection station identifier assigned by lab
	Raw data	STATION NAME	Sample collection station name
	Raw data	MISC STATION INFO	Additional station information
	Raw data	Data Reporting Year	Year 1=1999; Year 2=2000
	Raw data	SPECIES CODE	Code for fish species (KB, SPN, SSD, WC)
	Raw data	SPECIES NAME	Common name for collected fish species (Kelp bass, Splitnose rockfish, Speckled sanddab, White croaker)
	Raw data	SKIN PREP	S = prepared skin on; N = prepared skin off
	Raw data	LATITUDE	Latitude in degrees and decimal minutes
	Raw data	LONGITUDE	Longitude in degrees and decimal minutes
	Raw data	COLLECTION DATE	Date sample collected
	Raw data	Special Treatment	Comment field for special handling or analysis
	Raw data	N (Number per sample)	Number of fish included in analytical result; when N=1 tissue from one fish was prepared/analyzed; when N>1, a composite of tissue from that number of fish was prepared and analyzed.
	Raw data	>25%	Y = there was greater than 25% difference between the largest and smallest fish in this composite
	Raw data	% Moisture MLML	Percent moisture
	Raw data	ANALYSES	MO = Arsenic, cadmium, mercury, selenium, and organics; &; M = Arsenic, Cadmium, Mercury and Selenium; EM = Arsenic, Cadmium, Copper, Chromium, Lead, Mercury, Nickel, Selenium, Silver and Zinc
	Raw data	PWATF	% water in fillet samples measured by Water Pollution Control Lab

TABLE A-4
Fish Tissue Metadata for Raw Sediment Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Worksheet	Field	Description/Acceptable Values
	Raw data	PLIPF	% lipid in fillet samples measured by Water Pollution Control Lab
	Raw data	tPCB	tPCB = sum 1248, 1254 & 1260; if all 3 are ND, then value entered is 5=1/2 the lowest MDL; ng/g WW
	Raw data	tDDT	tDDT = Sum ddeop, ddepp, dddop, dddpp, ddtop, ddtpp; if all 3 are ND, then value entered is 1=1/2 the lowest MDL; ng/g WW
Noblet et al. 2002	Noblet et al Stations	StationID	Unique station identification number
		Depth	Station depth (m)
		Latitude	Station location latitude in degrees and decimal minutes
		Longitude	Station location longitude in degrees and decimal minutes
	Noblet et al ChemDDT Raw data	StationID	Unique station identification number (2147, 2189-92, 2194-5, 2197-8. 2200-2, 2204-20, 2233, 2242, 2244, 2254, 2256, 2262, 2267-8, 2274, 2280-98, 2302-3, 2306, 2312, 2314, 2317, 2319-20, 2341, 2343-62, 2364-5, 2368, 2372, 2374-5, 2377-87, 2393-4, 2396-8, 2400-8, 2411-13, 2417-19, 2426, 2434, 2444, 2446-8)
		CompositeID	Unique identification number for whole-fish composites (1116-18, 1120-21, 1123, 1125, 1127-31, 1133-34, 1136, 1138-42, 1144, 1147, 1149, 1151-52, 1154-55, 115991963-64, 1181, 1183-85, 1187, 1189-90, 1192-95, 1197-1200, 1203-04, 1208, 1210-12, 1216-17, 1219-22, 1224-25, 1227, 1230-31, 1235-36, 1238-40, 1243-44, 1263, 1267, 1269, 1271-72, 1277-78, 1280-82, 1284-86, 1289-90, 1292-94, 1296-1304, 1307, 1309-11, 1313-16, 1318, 1320, 1322-23, 1325-26, 1328, 1345, 1347-50, 7311981943, 804981938-40, 813981946)
		Species	Latin names (Citharichthyes sordidus, Citharichthyes stigmaeus, Citharichthyes xanthostigma, Eopsetta exilis, Paralichthyes californicus)
		Age Class	0, 1, 2
		Total FishDDT	Whole fish DDT concentrations
		% OC	Percent organic carbon (sediments)
		totSedDDT	tDDT (sum of the isomers) measured in sediment samples
		qry_DDTtot.Units	Units for totSedDDT
		totDDT/foc	OC-normalized sediment tDDT concentration
		TotalFish DDT	tDDT (sum of the isomers) measured in whole fish
		Lipid	Percent lipid in fish tissue
		TotDDT Fish/Lipid	Lipid-normalized tDDT concentration in whole fish

TABLE A-4
Fish Tissue Metadata for Raw Sediment Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Worksheet	Field	Description/Acceptable Values
	Noblet et al ChemPCB Raw data	StationID	Unique station identification number (2147, 2189-92, 2194-5, 2197-8. 2200-2, 2204-20, 2233, 2242, 2244, 2254, 2256, 2262, 2267-8, 2274, 2280-98, 2302-3, 2306, 2312, 2314, 2317, 2319-20, 2341, 2343-62, 2364-5, 2368, 2372, 2374-5, 2377-87, 2393-4, 2396-8, 2400-8, 2411-13, 2417-19, 2426, 2434, 2444, 2446-8)
		CompositeID	Unique identification number for whole-fish composites (1116-18, 1120-21, 1123, 1125, 1127-31, 1133-34, 1136, 1138-42, 1144, 1147, 1149, 1151-52, 1154-55, 115991963-64, 1181, 1183-85, 1187, 1189-90, 1192-95, 1197-1200, 1203-04, 1208, 1210-12, 1216-17, 1219-22, 1224-25, 1227, 1230-31, 1235-36, 1238-40, 1243-44, 1263, 1267, 1269, 1271-72, 1277-78, 1280-82, 1284-86, 1289-90, 1292-94, 1296-1304, 1307, 1309-11, 1313-16, 1318, 1320, 1322-23, 1325-26, 1328, 1345, 1347-50, 7311981943, 804981938-40, 813981946)
		Species	Latin names (Citharichthyes sordidus, Citharichthyes stigmaeus, Citharichthyes xanthostigma, Eopsetta exilis, Paralichthyes californicus)
		Age Class	0, 1, 2
		tFishPCB	Whole fish PCB concentrations
		TOC	Total organic carbon (sediments)
		qry_TOC.Units	Units for TOC
		totSedPCBs	tPCB (sum of the isomers) measured in sediment samples
		qry_PCBtot.Units	Units for totSedPCBs
		totPCB/foc	OC-normalized sediment tDDT concentration
		tFishPCB	tPCB (sum of the congeners) measured in whole fish
		Lipid	Percent lipid in fish tissue
		TotPCBFish/lipid	Lipid-normalized tDDT concentration in whole fish
		BSAF (col M / col J)	Lipid-normalized tPCB whole-fish concentration divided by organic carbon-normalized sediment tPCB concentration
Anderson et al. 1998	Raw data	BPTCP Station#	Bay Protection Toxics Cleanup Program (BPTCP) station number (40020, 49001, 49002, 49003)
		Station Name	Name equivalent of Station #
		Lat N	Latitude in degrees-minutes-seconds
		Long W	Longitude in degrees-minutes-seconds
		Date	MM/DD/YYYY
		Species	White croaker
		# in Composite	5 skin-on fillets per composite

TABLE A-4
Fish Tissue Metadata for Raw Sediment Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Worksheet	Field	Description/Acceptable Values
		tDDT (mg/kg WW)	tDDT
		tPCB (mg/kg WW)	tPCB
		%lipid	Percent lipid
Hansen and Associates 2000	Raw data	Location	Sample collection location (BunkerJune, BunkerMar, BunkerWhJune, LongPt, WhitesMar)
		Location Code	Code for sample collection locations (BP_ML, BW_ML, LP_ML, WP_ML)
		Date	Sample collection data
		tDDT (mg/kg WW)	tDDT concentration
		tPCB (mg/kg WW)	tPCB concentration
		DDTlipid (mg/kg WW)	Lipid-normalized tDDT
		PCBlipid (mg/kg WW)	Lipid-normalized tPCB
		%DDT lipid	Percent lipid associated with samples analyzed for tDDT
		%PCBlipid	Percent lipid associated with samples analyzed for tPCB
		Comments	Information about samples
Costa et al. 1994	Raw data	Sample #	Sample number (KB1 - KB145)
		Common Name	Kelp bass
		Scientific Name	*Paralabrax clathratus*
		Location	Sample collection location (Catalina, Dana Point, Dana Point or San Clemente, Laguna, PV, PV - Rocky Pt, San Clemente, San Mateo, San Onofre, <Blank>)
		Latitude	Latitude in decimal degrees
		Longitude	Longitude in decimal degrees
		Parameter	Analyte (tDDT, tPCB)
		% Lipid	Percent lipid in fillet
		Result (ng/g WW)	Analyte concentration
Connolly and Glaser 1994	Raw data	Sample ID	Sample identification number (1125-1155; 1193-1199; 1275-1304; 2076-2122)
		Latitude	Latitude in decimal degrees
		Longitude	Longitude in decimal degrees
		Year	1990; 1991
		Tissue Type	MUSL (fillet)
		% Lipid	Percent lipid in fillet

TABLE A-4
Fish Tissue Metadata for Raw Sediment Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Worksheet	Field	Description/Acceptable Values
		% Water	Percent water in fillet
		Species	Kelp bass, Dover sole, white croaker
		Analyte	tDDT; tPCB
		Result	Analyte concentrations (ng/g WW)
		Comments	Sample-specific notes (Segment refers to HydroQual segments from Connolly and Glaser, 1994)
LACSD 1990-2001	Raw data	Survey	Collection date (1990-2001)
		Zone	Collection zone (Zone1, Zone2, Zone3)
		Replicate	Sample replicate number (1-10)
		Common_Name	Common fish name (Dover sole, kelp bass, white croaker)
		Parameter	Analyte name (Total Detectable DDT, Total Detectable PCB, Total Lipid)
		#_in_Composite	Number of fillet samples in each composite (1, 5, 6, 7, 10)
		Qualifier	QA qualifier (blank, ND)
		Result	Measured concentration
		Result_Units	Units for Result (%, MG/KG, UG/KG)

TABLE A-5
Metadata for Water Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Field	Description/Acceptable Values
Zeng and Tran 2002	Sample ID	Station location identificaion number (0C-1M, 3C-1M, 5C-1M, 6B-1M, 6C-1M, 6C-20M, 6C-2M, 6C-35M, 6C-5M, 6D-1M, 7C-1M, 9C-1M
	Sample Type	Water
	Sampling Depth (m)	Depth of water from which samples were collected (25, 29, 40, 55, 58, 59, 149)
	Parameter	Analyte measured (tDDT, tPCB)
	Month	Month during which samples were collected (Jan-March; June-July)
	Year	Year during which samples were collected (1997)
	Results	Analyte concentration (ng/L)
	Comments	Sample-specific information

TABLE A-6
File Names for Fish Tissue Data Files
Palos Verdes Shelf Ecological Risk Assessment

File Name	Type
Avian Analytical Data_Raw.xls	Raw data
Avian Sample Data_Raw.xls	Raw data
AVIAN.xls	**CH2M HILL Database**
AVN_INF_Raw.xls	Raw data
AVN_MRG_Raw.xls	Raw data
Bird Location Query_Orig_Raw.xls	Raw data

TABLE A-7
Metadata for Avian Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Field	Description/Acceptable Values
Bird Location Query	STUDY	Avian
	CATALOG	024, 025, 028, 029, 031, 036, 038, 047, 048, 052, 053, 057, 065, 076, 078, 081
	FIELD_ID	Identification number given at sample collection
	MATRIX	Type of tissue sample collected (Avian egg, Avian yolk, Brain, Fat, Liuver, Muscle, Other, Serum, Whole Blood, Whole Body)
	SPEC_CODE	Numerical code for species (07, 08, 11, 12)
	SPEC_COM	Species common name (Blad Eagle, Brown Pelican, Double-crested cormorant, Peregrine falcon)
	SPECIES	Latin species name (peregrinus, auritius, leucocephalus, occidentalis)
	LAT	Latitude in decimal degrees
	LON	Longitude in decima degrees
	SPECLOCCOD	Specific location code (AB, AI, BR, CP, DC, EE, GI, HB, HI, HP, LC, LP, MI, MR, PR, RR, SA, SB, SR, TR, VB, WC, WE, WZ, <blank>); see NRDA metadata files (AVN_INF or AVN_MRG) for complete listing; abbreviated location description
	GENLOCCOD	General location code (02-09); see NRDA metadata files (AVN_INF or AVN_MRG) for complete listing.

TABLE A-7
Metadata for Avian Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Field	Description/Acceptable Values
	Location Description	Full text name for sample locations
	Web Address for LAT and LON	Internet site where location data were found
AVN_MRG	See specific files for metadata	
AVN_INF		
Avian Sample Data_Raw	STUDY	Avian
	CATALOG	<blank>
	LAB-ID	Identification number given at the lab
	FIELD_ID	Identification number given during sample collection
	MATRIX	Tissue type analyzed
	SPEC_CODE	Numerical equivalent for species
	SPEC_COM	Species common name (BP = brown pelican; BE = bald eagle; PF = peregrine falcon; DCC = double-crester cormorant)
	SPECIES	Latin species name
	LAT	Latitude in decimal degrees
	LON	Longitude in decimal degrees
	SPECLOCCOD	Specific location code (see secondary table, below)
	GENLOCCOD	General location code
	COUNTY	County where sample collected ()
	STATE	State where sample collected ()
	SAMPVOL_ML	<blank>
	SAMPWT_G	<blank>
	SAMPDATE	1969-2000
	EGGVOL_ML	<blank>
	EGGLEN_MM	<blank>
	EGGBRTH_MM	<blank>
	EGGWT_G	Egg weight (grams)
	SHELLWT_G	<blank>
	CONTWT_G	<blank>
	FRESHWT_G	<blank>
	SHELTHK_MM	Shell thickness (mm)

TABLE A-7
Metadata for Avian Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Field	Description/Acceptable Values
	EMBSTAGE	Embryo developmental stage (addled; large embryo)
	EGGCOND	Egg condition (broken, intact)
	EGGABNORM	<blank>
	EMBLEN_MM	<blank>
	EYEDIAM_MM	<blank>
	EYEPIGM	<blank>
	FEATHER	<blank>
	INCUB_ID	<blank>
	CLUTCH_ID	<blank>
	BODY_COND	emaciated, excellent, fair, good, thin, very good,
	AGE	1, 2, 3, 4, adult
	% lipid	Percent lipid
Avian Analytical Data_Raw	STUDY	Avian
	QC_DESIG	tissue sample type (blood, bm, brain, carcass, egg, liver, membrane, plasma, serum, sm, wm, yolk, yolklip, yolksacs)
	LAB_ID	ID given at the laboratory
	INJECTID	<blank>
	FIELD_ID	ID given during collection from the field
	BATCH	1, 2
	COLUMN	<blank>
	UNITS	Units of measure (%, mg/kg DW, mg/kg lipid, mg/kg lipid basis, mg/kg lipid wt, mg/kg WW)
	ANALYTE	Chemical analyzed (numbers represent PCB congener numbers)
	BOOLEAN	QC qualifier (<, ND, NM)
	VALUE	Measured result
	LAB_QUAL	<blank>
	DV_QUAL	<blank>
	REP_PREF	<blank>
	ANA_ORDER	<blank>
	Reference	Literature citation for dataset
	Comments	Sample specific information
	VALUE	values in mg/kg

TABLE A-7
Metadata for Avian Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Field	Description/Acceptable Values
	Value_new	converted values (in ppb)

TABLE A-8
Metadata for Sea Lion Data Files
Palos Verdes Shelf Ecological Risk Assessment

Dataset	Field	Description/Acceptable Values
Sea Lion	Year	Year samples collected (1970, 1972, 1991)
	Condition	Pregnancy condition/term (Premature Parturient; Full-term Parturient)
	Gender	Female
	Mean Age	Mean age from adult sampled (years)
	p,p'-DDE	Analyte concentration
	PCB	Analyte concentration
	Units	Units for measured concentration (mg/kg W)

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APPENDIX B

Life-History Summaries for Representative Species

APPENDIX B

Life-History Summaries for Representative Species

B.1 Brown Pelican

The California brown pelican (*Pelecanus occidentalis californicus*) is one of six recognized subspecies of brown pelicans. Unlike other brown pelicans, it has a bright-red gular pouch during the courtship and egg-laying period.

Distribution

The California brown pelican ranges from as far north as Vancouver Island, BC, to Colima, Mexico in the south. During breeding season the range is limited to the Channel Islands, Islas Los Coronados, south to Isla Isabella, and Islas Tres Marias off Nayarit, Mexico and Isla Ixtapa off Acapulco, Guerrero, Mexico.

Body Size and Weight

One captive adult from Florida weighed 3.3 kg at the minimum of its range (Schreiber 1976, in CalEPA 2001). Weight at fledging is often greater than that of adults. This allows for extra energy reserves until the young become proficient at feeding themselves (U.S. Fish and Wildlife Service [USFWS] 1983).

Diet Composition

Brown pelicans feed in the vicinity of the Palos Verdes Shelf (PVS). The brown pelican feeds mostly on fish, but occasionally feeds on crustaceans, carrion, and young of its own (Zeiner et al. 1990a). Northern anchovies make up most of the pelican diet (80 percent) with pelican population trends fluctuating based on anchovy abundance (Bonnell and Dailey 1993). In late spring, anchovies occur in spawning schools; in summer, post-spawning anchovies occasionally form large near-surface schools in the daytime near the coast, peaking in abundance in October.

Food Consumption Rate

Food consumption by seabirds of the SCB is lowest in January and February and in June and July, although the brown pelican is noted as one of two species that cycle the most carbon through the ecosystem. Brown pelicans are estimated to consume about 0.02 mg $C/m^2/d$ in the early fall. Their annual consumption is about 16,500 to 18,000 tons (Briggs et al. 1987, as cited in Bonnell and Dailey 1993).

Water Consumption Rate

No information was available.

Sediment Consumption Rate

Brown pelicans primarily feed on pelagic and water-column prey; therefore, sediment ingestion is expected to be negligible.

Habitat Requirements

Foraging

The offshore zone within 30 to 50 km of the colony is critical to pelicans for food supplies, especially when young are being fed (Anderson et al. 1980, as cited in USFWS 1983). Pelicans are usually found within 20 km of the shore and in waters less than 150 m deep (Briggs et al. 1983). Pelicans prefer fronts with sharp thermal gradients though water may not be the warmest. Geographic and hydrographic preferences include nearshore freshwater and 13-17° C translucent neritic water (Baird 1993).

Nesting

Habitat among the colonies in the SCB differs. Anacapa has relatively dense shrubby vegetation, whereas the islands located farther south along the Pacific coast of Baja California are more xeric and more sparsely vegetated (USFWS 1983). What they do have in common, however, is steep, rocky slopes utilized for nest sites. In the SCB, pelicans use large, bulky sticks to make nests that they line with grasses and forbs and build on the ground or in brush (Gress 1970, as cited in USFWS 1983). When fewer nesting materials are available, as on the more southern islands, nests are not as bulky and more are built on the ground (USFWS 1983).

Brown pelicans are colonial nesters and require sites free from mammalian predators and human disturbance. Sites must also supply an adequate and consistent food supply. Islands such as San Martin and Todos Santos are no longer used as nesting sites because of increased human disturbance.

Roosting and loafing sites include offshore rocks and islands; river mouths with sand bars; and many breakwaters, pilings, and jetties along the coast. Año Nuevo Island and Southeast Farallon Island are important roosts for pelican populations. When left undisturbed, major roosting areas on islands near the colonies have the best opportunity to become nesting areas if appropriate conditions exist. When the breeding season is finished, the center of activity shifts from the breeding sites to the roosting sites (USFWS 1983).

Movements

Home Range

No information was found on home range.

Foraging Range

Foraging generally occurs in neritic zones (Sydeman et al. 2001). During breeding season in the Channel Islands, they are most often found within 20 km of the breeding islands. From June to October, they are common along the coast in southern California within 30 km of shore but are regularly seen out to 175 km (Briggs et al. 1981, as cited in Zeiner et al. 1990b). The presence of the offshore islands extends the range of foraging pelicans farther from the mainland, and pelicans off the coast of San Diego have been seen as far out as 190 km (Briggs et al. 1987).

In 1978 and 1979, the primary foraging area during the early part of the breeding season was in the Santa Barbara Channel, whereas in 1980 the primary foraging area was centered in the area between Santa Barbara and Anacapa Island and in Santa Monica Bay in February and March (Gress 1981).

During daylight hours, pelican populations are most widely dispersed at sea. Estimated ratio of pelicans ashore versus pelicans at sea in southern California during daylight hours was about 1:4. During the nesting season, brown pelicans are most numerous within 20 km of the nesting islands. One radiotagged juvenile in south Monterey Bay traveled from 10.2-49.4 km/day (Croll et al. 1986).

Migration

After breeding as early as mid-May, individuals leave colonies in the Channel Islands and Mexico and disperse along the entire California coast. Most return to breeding colonies by March and April (Zeiner et al. 1990a). Population peaks occur from September to October with populations at their lowest from December to March (Briggs et al. 1987).

Dispersal

Population Density

Distributions at sea often mirror numbers on land. In summer and fall, major concentrations were seen over the shelf from Cordell Bank to Monterey Bay off Morro Bay, in Santa Barbara Channel from Anacapa to Santa Barbara Island, and from San Clemente Island to San Diego (Briggs et al. 1981, as cited in Briggs et al. 1987).

Population Dynamics/Survival

Breeding pelicans were noted on Anacapa Islands as far back as 1898. Population estimates were sketchy and unreliable but indicated that breeding populations fluctuated over time. Breeding on Santa Barbara Island can be documented back to 1911, although nesting on the island was probably sporadic as Anacapa serves as the main breeding island. Breeding was also noted in the early 1900s on Santa Cruz Island and Prince Island, offshore from San Miguel (Willet 1910 and 1912; Howell 1917, as cited in USFWS 1983). This nesting was considered irregular and confined to few nests. Prince Island has not served as an active nesting site since at least the early 1960s.

From the 1950s to the 1970s, a slow continuous decline occurred in breeding pelican populations (USFWS 1983). In 1968, a seabird breeding survey was conducted on the Channel Islands and Los Coronados. Pelicans were found breeding only on West Anacapa Island and were limited to about 100 pairs. Additionally, the nesting colony was abandoned before the young pelicans could have fledged. This was the first indication of population decline and reproductive problems. In 1969, out of 1,272 nests, at most four young fledged from the Anacapa colony (USFWS 1983). Chemical analyses of eggs that year indicated high levels of DDT and DDE (Risebrough 1972; Risebrough et al. 1970; Keith et al. 1971; Blus et al. 1972, as cited in USFWS 1983). While reproductive success was still low in 1972, two breeding colonies—one on Scorpion Rock (an islet offshore of Santa Cruz Island) and one on Anacapa Island—showed significant improvement over 1969-1971. While breeding on Anacapa was continuous from 1969-1980, colonies on Scorpion Rock were observed only in 1972, 1974, and 1975 (Gress 1981). Breeding continued to increase until a food shortage from

1976 through 1978 occurred. Breeding again increased in 1979, when more pairs produced young in the SCB than in any year since 1968. From 1978 to 1980, breeding on Anacapa Island took place at six different locations. Colonies were found at the west end of the island (2), at the northeast end (3), and at Camel Peak. From 1969 to 1980, productivity ranged from 0.002 to 0.88, with the number of pairs ranging from 76 to 2,244 (Gress 1981).

In 1980, brown pelicans were documented as nesting on Santa Barbara Island. Colonies were found in three locations including Signal Peak and Lower Bluff, both located on the southwest side of the island, and Sutil Island, a small islet about 1 km southwest of Signal Peak (Gress 1981). Populations as of the late 1980s ranged from 56,000 to 80,000 at sea and 10,000 on land from May to December, dropping to 5,000 to 6,000 at sea from December to March, when population size in the area is at its lowest. Reproductive trends generally correlated with food abundance thereafter. In 1989-1991, there were 5,340 breeding pairs on West Anacapa and another 618 on Santa Barbara, indicating that the population has clearly recovered (Power 1994).

Human disturbance is another factor affecting breeding success. As human activity increases, in general, pelican productivity declines in the disturbed areas. Especially in the early stages of breeding, pelicans will easily abandon nests when disturbed (USFWS 1983).

Reproduction/Breeding

Historically, brown pelicans in the SCB have nested on Anacapa Island, Santa Barbara Island, Scorpion Rock at Santa Cruz Island, and San Miguel Island. Anacapa and Los Coronados, historically, have been the largest and most consistent brown pelican colonies in the SCB (Anderson and Gress 1983, as cited in USFWS 1983). San Martin used to support a breeding population but has been inactive since 1972 (Anderson and Keith 1980, as cited in USFWS 1983). Since 1968, the major SCB colonies have been found on West Anacapa Island and Isla Coronado Norte. From 1969 through 1981, breeding pairs in the SCB ranged from 330 to 3,510 (USFWS 1983). Breeding colonies on Anacapa and Los Coronados are not year-round residents. Breeding occurs on the Channel Islands, including Anacapa, Santa Barbara, and Santa Cruz, from March to early August. During the nesting season, brown pelicans are most numerous within 20 km of the nesting islands. Breeding populations make up an estimated 6.2 percent of the total brown pelican population throughout its range in western North America during typical breeding years, based on an estimate from 1983 (USFWS 1983).

Pelicans begin to breed when they reach 3 to 5 years of age with females breeding at younger ages than males. Since 1969, the earliest breeding on Anacapa was initiated in early January, while the latest nesting was initiated in mid-May; the 1980 Santa Barbara Island colony began in late December.

Pair bonds are formed at the nest site, and egg laying begins one to two weeks after the beginning of courtship and nest building. A three-egg clutch is typical, with an incubation period of about 30 days. The California subspecies usually fledge at about 13 weeks. It is believed that re-nesting rates of the brown pelican are low.

A critical fledging area offshore West Anacapa was set aside by the California Department of Fish and Game in 1979. The regulation restricts, between January 1 and October 31, all activity offshore an area that encompasses the colony sites used by pelicans in 1979-1982 seaward to a depth of 20 fathoms.

B.2 Double-Crested Cormorant

Double-crested cormorants (*Phalacrocorax auritus*) are part of a world-wide order of seabirds that includes pelicans, boobies, tropicbirds, and frigatebirds. They are members of the family Phalacrocoracidae and can be found throughout North America in both fresh water and marine habitats.

Distribution

The double-crested cormorant is found all around North America. They inhabit marine and inland waters. They breed locally in interior from Alaska, Manitoba, and Newfoundland south to Mexico and the Bahamas. They winter mainly on coasts, north to Alaska and southern New England.

Body Weight and Size

The average adult weight for the double-crested cormorant in the SCB is 2.23 kg (Ainley and Boekelheide 1990, as cited in Glaser and Connolly 2002). They can be 19-40 inches long (Landsborough Thomson 1964); however, they are generally 33 inches in length.

Diet Composition

The cormorant feeds mainly on fish, but also on crustaceans and amphibians. Consumers of fish and squid spend most of their time feeding in neritic areas (Briggs and Chu 1987, as cited in Daily et al. 1993). Diet preferences for double-crested cormorants are rather narrow because they eat mostly schooling prey that occur from surface to near-bottom with no relief (aggregating prey that occurred above the substrate) (Ainley et al. 1981). This results in exact feeding locations being a function of the local oceanographic conditions that determine fish abundance (Ainley and Boekelheide 1990, as cited in Glaser and Connolly 2002). Substrate preference of double-crested cormorant prey was dominated by rocky reef habitat (54 percent), followed by mid-depth to surface depths in water (23 percent), flat areas near rocks (22 percent), and flat sand or mud (1 percent) (Ainley et al. 1981).

Cormorants forage throughout a wide range to find food. In Rail and Chapdelaine (1998), diets of cormorants living in the estuary of the St. Lawrence River, Quebec, were analyzed. Diet, in percent by volume, consisted of sand lance (1.8), gunnels (13.1), capelin (37.2), rainbow smelt (1.9), Atlantic herring (11.1), flatfishes (13.1), sculpins (2), other fish (18.5), shrimps (0.2), other crustaceans (0.1), and other food (1). Based on this diet, double-crested cormorants eat small schooling fish as well as benthic fish.

Samples from material regurgitated by chicks on Prince Island in 1976 included common mid-water fishes that inhabit littoral waters, particularly kelp beds (Hunt et al. 1981, as cited in Glaser and Connolly 2002). One study in New Hampshire found a diet consisting of cunner (59.7 percent), winter flounder (15.5 percent), sculpin (2.4 percent), rock eel (0.3 percent), American eel (6.6 percent), pollock (9.8 percent), and mummichog (5.8 percent) (Dunn 1975, as cited in CalEPA 2001). In the Farallon Islands, shiner surfperch made up 78.6 percent of the breeding season diet (Boekelheide et al. 1990).

Food Consumption Rate

No information was found.

Water Consumption Rate

No information was found.

Sediment Consumption Rate

Although some benthic prey are consumed, double-crested cormorants primarily feed on pelagic and water-column prey; therefore, sediment ingestion is expected to be negligible.

Habitat Requirements

Foraging

Double-crested cormorants generally forage in neritic zones and estuaries (Sydeman et al. 2001). Cormorants pursue their prey underwater, often to great depths (maximum depth is about 20 m) feeding throughout the water column (Bonnell and Dailey 1993). They prefer water less than 9 m deep with rocky or gravel bottom but may catch fish as deep as 22 m (Zeiner et al. 1990a).

Nesting/Breeding

Double-crested cormorants nest on vertical cliff faces, flat ground, and even in trees and require undisturbed nest sites beside water on islands or mainland. The double-crested cormorant nests in discrete colonies, sometimes in trees, but most often on the ground on islands. They prefer the shoulder of hillsides, higher slopes, and summits of islands for their colony sites (Boekelheide et al. 1990). The nest must be within 8 to 16 km of dependable food supply (Palmer 1962, as cited in Zeiner et al. 1990b) and be large enough to hold up to four chicks, who grow as large as their parents before they leave the nest for good (U.S. Geologic Survey [USGS] 2002).

Movement

Home Range

No information was found on home range.

Foraging Range

The cormorant usually forages within 8-16 km of the roost or nest colony (Zeiner et al. 1990a) and is known to forage in the SCB all year (Bonnell and Dailey 1993). Geographic and hydrographic preferences include neritic, nearshore, freshwater, and littoral-benthic zones (Dailey et al. 1993). Cormorants forage nearshore, and those nesting on the Channel Islands may even fly to the mainland coast to feed (Ainley, personal communication, in Sowls et al. 1980, as cited in Bonnell and Dailey 1993). In the p,p'-DDE and total PCB bioaccumulation model for birds in the SCB presented by Glaser and Connolly (2002), foraging for juveniles (from fledgling to 4 years of age) was considered to occur anywhere within the SCB. However, from 4 years and older, foraging was limited to 50 km around the nest site during breeding season. During the non-breeding season, foraging was considered to occur anywhere in the SCB.

Migration

Some double-crested cormorants are year-long residents of the California coast. Summer residents of the mountains and northeastern plateau are absent from about November to March, and presumably migrate west and south to lowlands, especially along the coast, where the population increases in winter (Zeiner et al. 1990a). The double-crested cormorants that inhabit the Channel Islands are known as mostly sedentary with populations moving from offshore islands to inshore channels in the non-breeding season (September to March) (Hunt et al. 1981, as cited in Glaser and Connolly 2002).

Dispersal

The double-crested cormorant seems to have a pattern of significant movement during the first year, although trends indicate a southerly movement (Boekelheide et al. 1990).

Population Dynamics/Survival

Historically, the double-crested cormorant was a common breeder on the Channel Islands. At Santa Barbara Island in 1939, approximately 2,000 individuals began nesting (Sumner 1939, as cited in Hunt 1980) whereas only 67 pairs nested there in 1977. The largest breeding colonies were located on Prince, Santa Barbara, and Anacapa Islands; now only remnant populations exist (Power 1994). Complete reproductive failure occurred in double-crested cormorant populations on Anacapa Island in 1969 (Gress et al. 1973, as cited in Peakall 1994). In 1975 to 1977, breeding pairs were found on Prince Island (San Miguel), West Anacapa, Santa Barbara, and Sutil Island (Hunt 1980). Productivity of double-crested cormorants in the SCB dropped significantly by the 1970s and did not increase again until the 1980s. In the mid-1990s, however, p,p'-DDE levels were still elevated and eggshell thinning was observed. Concentrations of contaminants in eggs on Santa Barbara Island were lower than those found on Anacapa Island. In 1992, eggs were collected on both Anacapa Islands and Santa Barbara Islands.

The decline was most likely attributable to several factors, including the disappearance of Pacific sardines through overfishing, reproductive failure caused by DDT, and increased human disturbance (Power 1994). In 1992, eggshell thinning was found at very severe levels in double-crested cormorants although less thinning was found in double-crested cormorant eggs from more northern locations (Kiff 1994).

This maximum age for this cormorant recorded from nature is 17 years (Clapp et al. 1982).

Reproduction/Breeding

The breeding season lasts from April to August (Hunt et al. 1981, as cited in Glaser and Connolly 2002). Double-crested cormorants have a mean clutch size of 2.3, ranging from 2 to 7 eggs in the SCB (Ainley and Whitt 1974; Ainley et al. 1974; Frame 1972; Ayers 1975, as cited in Baird 1993).

Double-crested cormorants were breeding on West Anacapa Island in the north bluffs around 1980. In 1979, they nested in three subcolonies, but only two subcolonies were observed in 1980. From 1969 to 1980, productivity on Anacapa ranged from 0 to 1.2 with pairs ranging from 3 to 78 (Gress 1981).

B.3 Western Gull

Western gulls (*Larus occidentalis*) are divided into two subspecies by geographical region. The northern race (*L. o. occidentalis*) inhabits central California and in areas along the northern coast; the southern race (*L. o. wyamni*) extends from central California to Baja California. They are members of the family Laridae and are limited along the Pacific Coast of North America.

Distribution

Western gulls (*Larus occidentalis*) are abundant, yearlong residents of coastal California, including San Francisco Bay and the Channel Islands. They occupy coastal islands, cliffs, harbors, bays, river mouths, and garbage dumps. Western gulls are the only seabird species breeding in the Southern California Bight area. According to Briggs et al. (1987), their population size averages 5,000 to 10,000 during spring to summer (breeding) and peaks at 25,000 to 50,000 during the winter (January and February).

Western gulls occur in neritic waters and stay close to their colonies during April through August (Briggs et al. 1987). Generally, they are rarely found greater than 25 km seaward of the shelf break.

Body Size and Weight

On the average, adult males are heavier than females. The average weight measured on Santa Barbara Island, California, for adult males was 981 grams; while the adult female average weight was 769 grams (Pierotti and Annett 1995). The arithmetic average of the adult female and male western gull is 875 grams.

Diet Composition

Seabirds are among the top consumers in the Southern California Bight area, along with marine mammals, and are responsible for removing approximately 14 to 29 percent of various fish stocks (Schaefer 1970; Robertson 1972; Furness 1978; Furness and Cooper 1982; Zeiner et al. 1990a).

The western gull's diet is composed mainly of anchovies, rockfish (*Sebastes*), flatfish, Pacific sauries (*Coloabis saira*), midshipmen (*Porichthys*), cephalopods, euphausids, and offal (Zeiner et al. 1990a; Baird 1993). They forage over open water using aerial dives (subsurface seize) and on the water surface by dipping. They also forage in intertidal areas for invertebrates, on nesting islands for small avian prey (auklets, petrels, gull chicks, and eggs), and scavenge in a variety of habitats, including dumps, harbors, beaches, and near fishing boats. Diet composition studies based on Santa Barbara Island revealed its diet as consisting of 85 percent small fish and more than 10 percent cephalopods during the breeding season (Pierotti and Annett 1995).

Food Consumption Rate

A food consumption rate in kg-dry food/kg-body weight/day was obtained using the allometric equation developed by Nagy (1987). This rate was converted to a wet weight food consumption rate of 0.257 kg-wet weight food/kg-body weight/day assuming water contents of 75 percent and 82 percent for fish and marine invertebrates, respectively.

Water Consumption Rate

No information was found. As a seabird, the water ingestion of western gulls is expected to be negligible, with basic water requirements being met through the diet.

Sediment Consumption Rate

No information was found. As western gulls forage in the top 1-2 m of the sea surface, sediment ingestion is assumed to be negligible or absent.

Habitat Requirements

Foraging

Oceanic productivity determines the abundance and distribution of seabird prey and, therefore, the abundance and distribution of seabirds. The majority of seabirds forage in the highest ocean productivity areas, which are:

- Waters overlying the continental shelves
- areas of convergence, where water of different properties comes together and mixing and turbulence occur
- areas of upwelling, where cold, rich, deep water replaces surface water

Upwelling, a localized phenomenon along the Southern California Bight coast, plays a crucial role throughout the year for seabird foraging. During late winter, large numbers of dinoflagellates concentrate in the upwelling areas. They furnish food for larvae of northern anchovies - one of the western gull's most common prey items - which spawn in the zone of upwelling at this time (Briggs et al. 1987). In spring and early summer, rockfishes and flatfishes spawn in the upwelling zones and are found in the majority of seabird diets at this time (Briggs et al. 1987). The greatest extent and persistence of upwelling in the Southern California Bight area occurs in April through early June. One of the major centers of upwelling is Point Arguello-Point Conception. Storms can influence the survival of larval prey fish and thus can negatively affect seabird foraging success, productivity, and chick growth (Lasker 1981; Baird 1990).

Western gulls spend much time on the water after feeding in flocks, and thus are also vulnerable to sea surface pollutants.

Breeding/Nesting

The western gull tends to nest in well-drained rocky areas in crevices free of predators and protected from prevailing wind. Western gulls usually nest next to a large object (e.g. rock, bush) for protection from predators and adverse weather conditions. They typically nest in colonies during the breeding season (January through June). Parents forage away from the nesting areas. Chicks are cared for up to 6 months before both parents and chicks abandon their territory in the early fall (Pierotti and Annett 1995).

Movement

Home Range

Western gulls can forage up to 80 km from the colony, though 20 km is more common (Pierotti and Annett 1995). The home range size depends on the location of a reliable food source as well as a dietary preference of individuals.

Territory

Inter-nest distances have been reported from 16-24 m at Moss Landing (Pierotti 1976; Pierotti and Annett 1995). The same adult may occupy a territory in consecutive years.

B.4 Peregrine Falcon

Peregrines are a species of the order Falconiformes, family Falconidae, which includes 39 species of falcons. The peregrine falcon (*Falco peregrinus*) is cosmopolitan, meaning that the species is found around the world, from the Arctic to the South America. The subspecies found in the Eastern United States is anatum, and referred to as the American Peregrine Falcon.

Distribution

The peregrine has an almost worldwide distribution excluding Antarctica, New Zealand, and Iceland. Nesting populations have been established in the Arctic, and as far south as Tasmania, South Africa, and the Falkland Islands.

Body Size and Weight

Peregrine falcons differ in size according to sex. In general, males are 13 inches in length while females average 19 inches. Average wingspan is about 40 inches. Average adult weights of peregrine falcons are 1,000 g (females) and 680 g (males) (Ratcliff 1993, as cited in Glaser and Connolly 2002).

Diet Composition

Peregrine falcons feed on gulls, auklets, and other water birds within the SCB (Hunt 1994). Waterfowl (50 percent), shorebirds (10 to 12 percent), small gulls (10 to 15 percent), small land birds (20 percent), and small mammals (2 to 3 percent) made up the breeding diet based on remains in the nest from June to July in Alaska (Cade et al. 1968, as cited in CalEPA 2001), although on the Channel Islands diets consist only of birds (Kiff 1980). In the winter, the most important prey species are the California gulls (37 percent) and the Cassin's auklet (22 percent). The rest of the winter diet is made up of various birds including western gull (6.8 percent), eared/horned grebe (3.8 percent), marbled godwit (2.5 percent), mourning dove (6.8 percent), rock dove (3.6 percent), red phalarope (2.8 percent) and black-bellied plover (1.5 percent), among others. In spring, diet consists of rock dove (12.7 percent), gulls (16.8 percent), Cassin's auklet (11.3 percent), xantus' murrelet (3.1 percent), other auklet/murrelet species (4.9 percent), mourning dove (5.3 percent), band-tailed pigeon (1 percent), red phalarope (5.1 percent), common raven (3.1 percent), European starling (8.8 percent), and western meadowlark (2 percent), among others (Hunt 1994). Proportion of diet on an energy basis (proportion wet weight*energy content; standardized to a total of 1) in the Channel Islands include western gull

(0.096), California gull (0.094), Heermann's gull (0.022), Bonaparte's gull (0.022), Cassin's auklet (0.18), other waterbirds (0.26), resident land birds (0.19), and migrant land birds (0.13) (Glaser and Connolly 2002). In Cape Peninsula, South Africa from 1989 to 1995, columbidae made up 85 percent of all prey species while sturindae accounted for 8.2 percent, apodidae for 1.9 percent, and ploceidae for 1.1 percent of prey species (Jenkins and Avery 1999).

During its first year, the peregrine falcon may travel large distances. By the age of 2, they are considered resident on the island from which their eggs were sampled. Adults are present in their aeries on the Channel Islands throughout the year (Hunt 1994). Contaminant data in eggs have been collected on San Miguel, Santa Rosa, Santa Cruz, and Anacapa Islands. These populations are considered capable of feeding on bird populations from any of these four islands, but unlikely to feed on prey captured on the mainland or on Santa Barbara, San Nicholas, Santa Catalina, or San Clemente Islands (Glaser and Connolly 2002).

Because of their size difference, male and female peregrines take different prey species. This is known as resource partitioning and eliminates competition between the pair, a common feature in medium to large raptors.

Food Consumption Rate

No information was found.

Water Consumption Rate

No information was found.

Sediment Consumption Rate

Peregrines in the Channel Islands feed primarily on seabirds; therefore, no sediment ingestion is expected.

Habitat Requirements

Foraging

Foraging area usually consists of wooded areas, marshes, open grasslands, coastal strands, and bodies of water (Pacific Coast American Peregrine Falcon Recovery Team [PCAPFRT] 1982). In the Channel Islands, the shoreline serves as the primary foraging area.

In a study by Jenkins and Benn (1998) in South Africa, it was suggested that habitat use, defined in terms of land use, was random and that habitat composition of each of the home ranges seemed to depend largely on the location of the territory. While hunts may be concluded over a specific habitat type because prey are numerous or vulnerable, falcons do not necessarily make very frequent trips to these foraging areas, or spend long periods of time in them (Jenkins and Benn 1998).

Breeding/Nesting

Peregrine falcons nest on coastal and insular sea cliffs as well as inland cliffs usually near water. Tree nesting is virtually unknown (PCAPFRT 1982). The most preferred sites are sheer cliffs 150 feet or more in height. These cliffs often have small caves or overhung ledges large enough to hold three or four full-grown nestlings.

Peregrine falcons no longer nest along the coast from Santa Barbara south to the Mexican border because of development on the sea cliffs (Banks 1969, as cited in PCAPFRT 1982).

Movement

Home Range

Home range in Sonoma County was estimated at 320 km^2 (79,040 acres) (Zeiner et al. 1990a). On Cape Peninsula in South Africa, the average home range of males and females combined was 123 km^2 and ranged from 89.7 to 192.1 km^2. The average daily range for all four birds is 22.8 km^2 (Jenkins and Benn 1998). This study also concluded that activities of each falcon were centered on nest cliffs, and only a small number of distant locations accounted for most of their home range. In the Rocky Mountains, home range included the area encompassed by a radius up to 23 km from cliff nests (Zeiner et al. 1990a).

Migration

Migrant peregrine falcons are occasionally found in the Channel Islands in winter. Migrants occur along the coast and in the western Sierra Nevada in spring and fall. Some are resident breeders and remain near the nest cliff year round (PCAPFRT 1982); while other individuals, breeding farther north, migrate into California for winter. True migration does not occur in the resident peregrine falcon populations, although some winter movement may occur (PCAPFRT 1982).

Foraging Distance

Nineteen nests averaged 5.3 km from the nearest foraging marsh, and 12.2 km from the nearest marsh over 130 ha in area (Porter and White 1973, as cited in CalEPA 2001). In South Africa, maximum distance traveled from the nest cliff was 9.2 km for females and 11.1 to 16.4 km for males. Average distance traveled per trip was also assessed and resulted in ranging trips between <1 km to >80 km in length (Jenkins et al. 1998, as cited in CalEPA 2001). Male and female adults in Texas during winter averaged a foraging distance of 7.5 to 28 km based on 8 birds (Enderson et al. 1995, as cited in CalEPA 2001). According to personal communications with Kirven, foraging flights may extend as many as 19.2 km from the nest during nesting season (Jurek 1989).

Dispersal

Immature peregrines from California range more widely than adults and occasionally travel south toward northern Mexico and north toward central Oregon. Peregrines, however, typically return to the area of their original nest site, although some establish territories hundreds of miles from where they fledged. No peregrines banded in California have been located outside the state in adult plumage (Walton, personal communication, as cited in Jurek 1989).

Population Density

Peregrines are territorial during the breeding season. Even in areas where nests are most numerous, nesting pairs are usually more than 1 km, and often much farther, apart. This ensures adequate food for all nesting pairs and their offspring.

Cade (1971) reported that mean spacing between nests was 9.7 km along Alaskan Rivers (Zeiner et al. 1990a).

Population Dynamics/Survival

Peregrine falcons were present in the SCB in the 1940s but disappeared by the early 1950s. Pre-1940 population estimates are at a minimum of 20 to 30 pairs (Kiff 1980). Eggs collected during the years immediately following the introduction of DDT averaged 19 percent thinner than pre-1947 eggs (Anderson and Hickey 1972, as cited in Kiff 1994). By 1970, only two active nest sites were confirmed, and it was believed that the total number of active nest sites in the entire state was less than five (Herman 1971). After the re-introduction program in the 1970s, re-introduced birds began to reproduce in the SCB in the mid-1980s (Glaser and Connolly 2002). The first breeding on San Miguel Island occurred in 1988 (Walton, personal communication, as cited in Kiff 1994). Peregrine populations on the Channel Islands have been increasing on an annual basis (Hunt 1994) and are considered to have returned to pre-DDT densities, although natural production is low because of residual DDT effects (Garcelon et al. 1989).

Peregrine falcon nesting at Langara Island, British Columbia, was only slightly affected by DDT, and populations fluctuations from 1968 to 1989 were based on prey abundance (Nelson 1990).

Documented cases of mortality include shooting, collecting, and removal of young from nests for falconry as well as DDT impacts (Kiff 1980).

Reproduction/Breeding

Peregrine falcons in the SCB area historically nested on Los Coronados, San Clemente, Santa Catalina, San Nicolas, Santa Barbara, Anacapa, Santa Cruz, Santa Rosa, and San Miguel Islands (Kiff 1980). Nesting currently occurs on San Miguel, Santa Rosa, Santa Cruz, Anacapa, and Santa Barbara Islands (Walton 1997, as cited in USEPA 2000). Active nesting sites are also found along the coast north of Santa Barbara (Zeiner et al. 1990a). In 1975, observed pairs in California fledged a minimum mean of 1.75 young per pair (Thelander 1975).

Peregrines pair for life and a territory is fiercely defended. Peregrines breed from early March to late June in California (Jurek 1989). The size of a clutch ranges from three to seven eggs, with the eggs being laid at two- or three-day intervals. Incubation lasts between 28 to 35 days and is performed by both parents. The young fledge between 35 and 42 days of age. A second clutch may be laid if the eggs from the first one are destroyed or removed early in the season (Zeiner et al. 1990a). In resident birds, pair-bonds stay strong year-round.

B.5 Bald Eagle

Bald eagles (*Haliaeetus leucocephalus*) have two subspecies: the southern and the northern bald eagle. The northern bald eagle is larger and heavier. The adult bald eagle has distinctive coloring with a white head and a brownish-black or dark-brown body. Juveniles are less distinctive with variations in plumage ranging from entirely black to mottled with buff or buff-white until the brown adult plumage is attained (Snow 1973).

Distribution

Bald eagles are restricted to North America and are found primarily along the coasts of North America and inland lakes and rivers from the Gulf of Mexico to the Arctic. Southern bald eagles nest primarily in the estuarine areas of the Atlantic and Gulf coasts from New Jersey to Texas and the lower Mississippi Valley, northern California south to both coasts of Baja California, central Arizona, and New Mexico. Their winter range is about the same, although some eagles may migrate farther north after breeding season (Snow 1973).

Body Size and Weight

Average wing measurements for ten Alaskan male eagles was 24.1 inches, while average wing size of six Alaskan females was 25.54 inches (Snow 1973). Of seven adults captured in Alaska, the average weight was 4.8 kg. An immature female weighed 5.6 kg (Chura and associates 1967, as cited in Snow 1973). In general, female eagles are larger than males. According to Stalmaster (1987), average adult bald eagles weigh about 5 kg for females and 4 kg for males (Glaser and Connolly 2002).

Diet Composition

Bald eagles in the SCB feed primarily on fish in the pelagic zone and have a relatively localized feeding range near Santa Catalina Island, generally offshore from Palos Verdes. These near-shore fish most likely accumulate much of their contamination from around Santa Catalina Island. While fish represent the greatest proportion of the diet, birds and mammals were seasonally important near Santa Catalina Island between December 1991 and July 1993. Sea lion carrion was the predominant mammal consumed, although feral goats were also consumed (Garcelon et al. 1994a and b). It is also important to note that while fish make up the majority of the diet, sea lion carrion serves as the greatest source for p,p′-DDE to the eagles. It is estimated that between 91 and 93 percent of total contaminant dose to the bald eagle for total PCBs originates in the SCB (Glaser and Connolly 2002).

A study in 1982 by Jurek tracked the translocated eaglets from 1980 to 1981. During one observation a recent fledgling was seen with a feral piglet carcass. Eagles were also observed eating carcasses of goats and seal or sea lion carcass on the shore (Jurek 1982).

On Santa Catalina Island, diet composition varied by month from December 1991 to November 1992. In December 1991 and January 1992, diet consisted of 80 percent fish and 20 percent invertebrates; in February and August of 1992, it consisted solely of fish. In March, April, May, and June, the diet consisted of fish, birds, mammals, and invertebrates at varying percentages. Once again in July, the diet was made up merely of fish (97.2 percent) and invertebrates (2.8 percent). September had a diet of fish (85.7 percent) and mammals (14.3 percent) while October and November both had a mix of fish, birds, and mammals but no invertebrates (Garcelon 1994a). From February 1993 to June 1993, diets included no invertebrates. In February, fish (50 percent) and mammals (50 percent) made up the diet, whereas in May and June fish made up the entire diet. March diets consisted of 81.8 percent fish, 9.1 percent birds, and 9.1 percent mammals, while April diets consisted of 90.2 percent fish and 9.8 percent birds (Garcelon 1994b). If averaged over the year, diet consisted of fish (77.2 percent), birds (8.9 percent), mammals (7.9 percent), and invertebrates (5.8 percent).

Food Consumption Rate

Retflavi (1965) calculated food consumption for nestlings at three-fourths of an ounce (21.3 g) per day. Once the eaglets were ready to leave the nest, food consumption had increased to three to four pounds (1.36-1.81 kg) a day.

Water Consumption Rate

No information was found.

Sediment Consumption Rate

Although some benthic fish prey are consumed, bald eagles primarily feed on pelagic and water-column fish prey or carrion; therefore, sediment ingestion is expected to be negligible.

Habitat Requirements

Foraging

Habitats are generally based on nesting requirements, but foraging requirements include a nearby waterbody and available trees for roosting. Foraging typically occurs in open water bodies including estuaries. Wintering bald eagles generally concentrate in aquatic habitats and, if food is sufficient, they use these areas over nearby habitat (Stahlecker and Smith 1993).

Bald eagles in the SCB feed primarily on fish in the pelagic zone and have a relatively localized feeding range near Santa Catalina Island, generally offshore from Palos Verdes. A study performed in 1980 and 1981 on the bald eagles on Santa Catalina Island indicated that the eagles were frequently found in the canyon areas of the island with oak woodland habitat in addition to the shoreline. These canyon areas were heavily populated by feral goats, and carcasses were often available there. Frequented canyon areas included Silver Canyon, Grand Canyon, and Coffeepot Canyon areas near the south of the island, about 7 km from their release site (Jurek 1982). These woodlands are composed of scrub oak (*Quercus dumosa*), with toyon (*Heteromeles arbutifolia*) and Catalina cherry (*Prunus lyonii*).

Breeding/Nesting

Habitat requirements usually focus on nesting territories. Nesting sites vary tremendously and depend on the tree species available in a specific area. Nesting has been known to occur in Channel Island (probably *Quercus tomentella*) and Catalina cherry (*Prunus ilicifolia*) (Jurek 1988). Typically nest sites occur within a half mile of water. Trees chosen for nesting are generally the largest or the stoutest in the area, and nests are often found between 10 and 15 feet from the top of the tree (Snow 1973). On San Juan Island in Washington, bald eagles nested in Douglas firs (Retflavi 1965, as cited in Snow 1973). Nesting sites must also offer a clear flight path to a point on a beach or a river and an open view of the surrounding area. In general, eagles almost always nest in live trees, although the tops may be dead. An additional characteristic of most nesting locations is the availability of good perch trees within the vicinity of the nest.

Movement

Home Range

Mean winter home range of 12 radiotagged adults in Colorado was 311 km^2 (Harmata 1984, as cited in Stahlecker and Smith 1993). Grubb et al. (1989) reported a mean seasonal home range of 401 km^2 for immature eagles based on inter-roost movements in Arizona (Stahlecker and Smith 1993). Home ranges for eagles at Besnard Lake during breeding season ranged from 10 to 15 km^2 (Gerrard et al. 1980, as cited in Gerrard et al. 1992). Eagles on the San Juan Islands in Washington averaged a home range of 30 km^2 (Retflavi 1965, as cited in Gerrard et al. 1992).

Foraging Range

Preferred foraging habitats are rivers, lakes, and estuaries. Bald eagles in the Channel Islands forage within 5 km of the nest (Garcelon 1994b, as cited in Glaser and Connolly 2002).

A 1982 study indicated that eagles stayed within 4 km of the hatch site during the first 5 weeks after fledging. Of the 1981 release group, one eagle was found with two of the 1980 birds some 7 km from the release site (Jurek 1982).

Foraging distances were recorded for the three colonies located on Santa Catalina Island in 1992. The foraging area used by eagles at Seal Rock extended from a point 0.3 km northeast of Seal Rock to Church Rock, located at the northwest end of the territory, and covered a linear distance of about 3 km. The pair breeding at the West End territory foraged along a linear distance of 4.5 km. On the north side of the island, the foraging area extended from the western tip of the island, approximately 2 km to the southeast, and on the south side of the island extended 2.5 km to the southeast. The third territory, Pinnacle Rock, had a foraging area that covered a linear distance of 3.5 km and extended northwest from Pinnacle Rock (Garcelon et al. 1994a). Non-breeding birds were also found in three different areas: Twin Rocks, 4.8 km northwest of the City of Avalon; Torqua Springs, 2.5 km northwest of Avalon; and China Pointe, 7.3 km west of Avalon.

Migration

The bald eagle is fairly common as a local winter migrant at a few favored inland waters in southern California (USFWS 1986; Zeiner et al. 1990a). Most birds that breed in the Pacific region probably winter in the vicinity of their nests. Some move short distances to lower elevations or inland food sources. Bald eagles in the Channel Islands are non-migratory (Garcelon et al. 1994b, as cited in Glaser and Connolly 2002). Immature birds tend to move south earlier and travel farther south than adults and tend to move north later in the spring than adults (Snow 1973).

Dispersal

Juveniles disperse in random directions from the nest sites. Fledging, young eagles remain in the vicinity of their nests from 3 to 6 weeks before dispersing northward to western Canada and Alaska (Jackman et al. 1987, as cited in Jurek 1988), although bald eagles on Santa Catalina Island are resident throughout the year.

Population Dynamics/Survival

The southern bald eagle population has been declining since World War II at an accelerated rate. In 1972, the population of eagles was estimated at 750 breeding pairs in the continental United States (Knoder 1972, as cited in Snow 1973).

Bald eagles were common residents on the islands prior to the 1950s. In 1920, seven active nests were present on Santa Cruz Island. The last confirmed nesting of an eagle on the Channel Islands was in 1949; by 1960, there were no eagles left (Kiff 1980). A reintroduction program began in the mid-1980s on Santa Catalina Island, and natural reproduction did not successfully occur until 1993. Eggshell thickness from Santa Catalina Island between 1987 and 1993 averaged 12.5 percent thinner than pre-1947 eggs. Attempts to reintroduce the bald eagle to San Clemente Island were not successful (Power 1994).

In 1970, Coon did a study examining the fatalities of 76 bald eagles. Sixty-two percent of these eagles died from being shot. Other serious threats to eagle populations include toxicity from environmental contaminants, such as organochlorine pesticides and PCBs, and loss of habitat due to development and tree cutting (Snow 1973).

Bald eagles are believed to live 30 years or longer in the wild, and even longer in captivity.

Reproduction/Breeding

Historically, bald eagles nested on Santa Catalina Island, Santa Cruz, Anacapa, San Miguel, Santa Rosa, Santa Barbara, San Nicolaus, San Clemente, and Los Coronados islands (Kiff 1980). As of 1992, three active bald eagle nesting territories were found on Santa Catalina Island. These territories included Seal Rock, located 4.5 km SE of the City of Avalon; West End, located 0.5 km from the NW end of the island; and Pinnacle Rock, located 4.3 km southwest of Avalon.

The breeding season for bald eagles begins in February and lasts through July, with pairs establishing long-term bonds. Clutch size is typically two eggs, although occasionally three may be laid. The incubation period lasts for 35 days with both parents incubating and caring for the young when they hatch. During the first 3 to 4 weeks after hatching, one parent will always remain at the nest. In Alaskan populations, about 35 percent of successful pairs produce two young annually, whereas only 3 percent of pairs produce more than one eaglet annually in the Great Lakes region (Sprunt et al. 1973, as cited in Snow 1973). Previous reports suggest that eagles on the Channel Islands did not lay replacement clutches when nest sites were disturbed (Kiff 1980).

Behavior/Social Organization

Relationships with non-breeding eagles are indifferent and tolerant. Several eagles may gather in the same location to feed or for other common interests. Juvenile bald eagles have not been observed acting with hostility toward another of their own species (Snow 1973).

During the 1982 study by Jurek, all six eaglets of the 1981 release roosted at night together in oak woodlands within a few weeks after release.

B.6 California Sea Lions

California sea lions (*Zalophus californianus*) are the most abundant pinnipeds in California waters. Males have dark brown fur, nearly black when wet, while the female's is often light brown or tan.

Distribution

This species is mainly found in California's coastal waters but can also be found in the Galapagos Islands and Japan. It ranges on the Pacific coast from British Columbia south to California to the coast of Mexico, possibly to Tres Marias Islands.

Body Size and Weight

Average adult weight is 100 kg for females and 300 kg for males (Antonelis et al. 1990, as cited in CalEPA 2001). Males may measure up to 2.4 meters, with females measuring only up to 1.8 meters (San Diego Natural History Museum [SDNHM] 2000).

Diet Composition

Pinnipeds feed extensively in kelp bed habitats. Diet is diverse, varying by season and location. Common prey in California are Pacific whiting, anchovy, market squid, and shortbelly rockfish. North of the breeding range, diets shift to those species that are locally and seasonally abundant. Studies on diet in the Southern California Bight (SCB) indicate that in summer, sea lions primarily consume juvenile Pacific whiting, juvenile rockfish, market squid, northern anchovy, and jack mackerel. In autumn, they move northward and consume seasonally abundant stocks of Pacific whiting, squid, and anchovy in central California (Bonnell and Ford 1987). According to the National Marine Fisheries Service (NMFS 1997), the most commonly consumed prey in the Channel Islands consists of market squid, Pacific whiting, shortbelly rockfish, jack mackerel, chub mackerel, and northern anchovy, while sea lions in Central California near the Farallon Islands consume Pacific whiting and rockfish. In the summer, rockfish and squid are found near shore (Baird 1993) and most likely influence sea lion foraging areas. A study by Lowry et al. (1990) looked at food habits of the California sea lion at San Clemente Island from 1981 to 1986. Diet compositions consisted of market squid (7.9 percent), Pacific whiting (9.4 percent), rockfish (9.2 percent), jack mackerel (11.8 percent), pacific mackerel (4.6 percent), northern anchovy (24.7 percent), pelagic red crab (10.2 percent), blacksmith (5.1 percent), octopus (2.5 percent), squid (1.1 percent), and other fish (13.3 percent).

California sea lion pups rely solely on the mother's milk for approximately the first seven months of their life and generally begin eating solid food after the first 200 days (Boness et al. 1991). During the first year of life, juvenile sea lions accumulate all their contaminants through nursing (Glaser and Connolly 2002). It has been estimated that the stellar sea lion transfers about 80 percent of its DDT to its first-born pup through lactation (Lee et al. 1996, as cited in Glaser and Connolly 2002). Body burden concentrations of p,p'-DDE in California sea lion pups spike within the first year during lactation. Over the next year a dramatic decrease occurs, and at about three years of age, the body burden steadies out and begins to increase slowly over the next several years until the sea lion reaches sexual maturity and begins to nurse its first pup. At this point, the body burden

drops dramatically and stays relatively low for the remainder of the sea lion's life (Glaser and Connolly 2002).

Food Consumption Rate

Costa (1986) used the water flux method to determine total food consumption of several lactating females in 1983 and 1984. Food consumption during foraging bouts ranged from 11.6 percent of the body mass per day in 1983 to about 10.4 percent in 1984. Assuming a body weight of 100 kg for female sea lions (Antonelis et al. 1990), a food consumption rate of 0.104-0.116 kg/kg/day can be calculated.

The milk ingestion rate of the sea lion pup is highest in the first two months at 0.088 g/g-day and can be as low as 0.03 g/g-day over a period of seven months. The average male and female milk ingestion rate of the sea lion pup over the first seven months after birth is 0.0488 g/g-day (see Appendix C for detailed descriptions and calculations).

Water Consumption Rate

Freshwater requirements are met through metabolic processes and food consumption (Pilson 1970 and Harrison and Kooyman 1968, as cited in Zeiner et al. 1990b).

Sediment Consumption Rate

Although some benthic prey are consumed, California sea lions primarily feed on pelagic and water-column prey; therefore, sediment ingestion is expected to be negligible.

Habitat Requirements

Foraging

Foraging occurs in groups of various size, increasing with the size of prey schools. Typical dives last from 3 to 7 minutes and may last as long as 20 minutes. California sea lions are capable of dives to 250 meters (m)—under experimental conditions (Zeiner et al. 1990b). In a study by Antonelis and others (1990), California sea lions were typically found in waters with an average depth of 323 m and ranging from 18 to 1556 m. Fifty-six percent of individual sea lions were found northwest of San Miguel Island, while 22 percent were found south of the island. In general, California sea lions tend to be located over the continental shelf in the shallower waters of the neritic zone (Antonelis et al. 1990).

In the summer breeding period, foraging occurs in the eastern Santa Barbara Channel, near Anacapa Island and southward over the eastern rim of the Santa Cruz Basin, the island shelves to 200 m depth contour around San Miguel, San Nicolas, and Santa Barbara Islands, and over the Tanner and Cortez Banks. From July to September, areas most intensely used include the island shelves around the northern island chain, San Nicolas, Santa Barbara Island, and the central and western Santa Barbara Channel, and the Santa Rosa-Cortes Ridge. In late autumn and early winter, populations shift to the east and south with increased use of waters near Santa Cruz, Santa Catalina, the San Clemente and Coronados escarpments, Huntington Flats, Tanner and Cortes Banks, the northern rim of the San Nicolas Basin, and the eastern Santa Barbara Channel, especially in the vicinity of the Ventura Shelf. During winter, use of the waters near Santa Cruz, San Clemente, Santa Barbara, and San Nicolas Islands continues, while use of Tanner and Cortes Banks, Santa Cruz and Santa Monica Basins, and eastern Santa Barbara Channel decreases. Increased use

occurs at the Santa Rosa-San Nicolas Ridge and western Santa Barbara Channel (Bureau of Land Management [BLM] 1981).

Reproduction/Breeding

Females give birth and nurse on land. Mating takes place both in water and on land on all substrates (Peterson and Bartholomew 1967 and Heath 1981, as cited in Zeiner et al. 1990b) although preferred breeding sites include flat, open, sandy beaches. Haul-out and breeding sites are preferable if they are near a food supply, they have easy access to water, and human disturbance is minimal (Zeiner et al. 1990b).

Movement

Home Range

No information was found on home range. However, during autumn and winter, sea lions congregate on Año Nuevo Island, an island of 6.5 hectares (ha), with a density of up to 13,000 animals. On some of the more crowded beaches, only about 0.6 m^2 is available per animal (Nowak 1995).

Breeding territories on the Channel Islands are generally arranged one deep along the beach, have access to the water, are not separated by topographical features, and are about 10 to 15 m wide. In rocky areas, territories averaged 130 m^2. Each territorial male averages 16 adult females (Nowak 1995).

Foraging Distance

The average foraging range for female adults between June and July at San Miguel Island was 54.2 km (Antonelis et al. 1990). Adult females spend about 11 months of the year foraging within several hundred kilometers of the colony on San Miguel Island while, after the first year, males forage over great distances, traveling hundreds or even thousands of kilometers up the west coast of North America. King (1983) noted that the California sea lion is rarely found farther than 16 km out to sea. Typical foraging trips lasted an average length of 73.4 hours with a mean time spent ashore between foraging trips of 50.1 hours (Antonelis et al. 1990). Adult males do not forage during the 2 to 6 weeks they spend defending a territory and breeding (Zeiner et al. 1990b).

Migration

Males migrate southward to the Channel Islands and Mexico for the breeding season and north to central and northern California, Oregon, Washington, and British Columbia after breeding in late summer (August–September), while females and pups remain near breeding rookeries (little is known about female migration). The northern migration of the males peaks in September along central and northern California (SDNHM 2000). Año Nuevo Island and the Farallon Islands provide major haul-out grounds for males throughout the year (Zeiner 1990b). During summer, sea lions are most abundant near their rookery islands and in north and central waters of the SCB. Non-lactating females that remain in the SCB disperse away from the rookery during winter and spring (Melin et al. 2000). During the winter, populations shift eastward to the waters around Santa Catalina and San Clemente Islands and southward to Tanner and Cortes Banks (Bonnell and Ford 1987).

Dispersal

Pups remain with the females near the rookeries after the breeding season.

Population Dynamics/Survival and Densities

California sea lions are the most abundant pinnipeds of the SCB, representing 50 to 93 percent of all pinnipeds on land and about 95 percent of all sightings at sea (Bonnell and Ford 1987). In the mid-1800s, sea lions were very abundant along the coast of California and Baja California (Scammon 1874, as cited in Baird 1993). Hunting limited populations drastically and, by the 1930s, populations were largely restricted to Mexican waters surrounding the Baja California peninsula. From 1927 to 1938, population counts at the Channel Islands ranged from 1,134 to 3,696 (O'Shea and Brownell 1998). Population growth and range expansion in the 1930s led to recolonization of islands in the SCB (Bartholomew and Boolootian 1960, as cited in Baird 1993).

During summer breeding season, peak numbers on land occur in late June. During the 1982 breeding season, over 51,000 California sea lions were counted on land in the SCB, of which 52 percent inhabited San Miguel Island (Bonnell 1982, as cited in Baird 1993) and 40 percent occupied San Nicolas Island (Stewart and Yochem 1984, as cited in Baird 1993). The total SCB summer breeding population was estimated at 63,000 in 1982 (Bonnell and Ford 1987). The California population has been growing at almost 12 percent per year since 1983, and in 1986 was estimated to contain 87,000 animals (Boveng 1988, as cited in Baird 1993). Summer population counts statewide resulted in the following numbers: 4,378 (1980) 11,209 (1982), 14,300 (1995). Fall counts in the 1980s ranged from 10,334 to 24,348 with 1995 counts of 16,900 (NMFS 1997). Summer counts in 1995 on Año Nuevo Island (6,745) accounted for 60 percent of the total mainland count (NMFS 1997). At the Channel Island breeding rookeries, the minimum count for the period of peak abundance was 81,300 sea lions. In 1994, the California sea lion population along the west coast was estimated between 161,066 and 181,355 (Barlow et al. 1995, as cited in NMFS 1997). Greatest abundance at sea occurs in late autumn and early winter with approximately equal numbers of animals on land (BLM 1981). Resident populations in southern California consist of females, pups, and juveniles (NMFS 1997).

One captive individual lived for 30 years (Nowalk 1995).

In the Gulf of California, although poaching for meat and oil of California sea lions occurs, populations were still 35 percent higher in 1983 than in 1966 (Le Boeuf et al. 1983, as cited in Nowalk 1995). The subspecies in the Sea of Japan may be extinct already owing to fishing practices. The population in the Galapagos is doing well, but numbers have declined after recent El Niño events.

Reproduction/Breeding

Sea lions breed in the Channel Islands and Mexico from May to August. Breeding colonies have been established in several areas of the Bight, particularly in the northern Channel Islands such as San Miguel Island. Main breeding islands include San Nicolas, San Miguel, Santa Barbara, and San Clemente (NMFS 1997); San Miguel and San Nicolas serve as the primary breeding colonies. Pups are born primarily at Point Bennett on the west end of the island and on nearby Castle Rock of San Miguel Island. On San Nicolas, pups are born along

a 6-km section of shoreline on the winward west side of the island. On Santa Barbara Island, the small coves on the north, west and south sides of the island support rookeries; and Seal Cove and Mail Point serve as breeding rookeries on San Clemente Island (BLM 1981). Females rarely give birth north of San Miguel Island (Zeiner et al. 1990b).

Females are ready to breed at 3 years of age, while males take up to 5 years (SDNHM 2000). Birth of a single pup usually occurs sometime between mid-May and late June (Odell 1975, as cited in Glaser and Connolly 2002) after a year-long gestation period. Although pups may be nursed for up to a year, pups older than 200 days may begin to ingest solid food (Boness et al. 1991).

Behavior/Social Organization

During non-breeding season, sexual segregation occurs but not completely. Individuals form temporary dominance hierarchies based on size, with smaller animals being forced from favorable resting sites. California sea lions, however, are highly gregarious, packing themselves tightly together even when empty space is available (Nowak 1995).

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APPENDIX C

Food Web Exposure Model

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Food Web Exposure Model

C.1 Introduction and Background

The food web exposure model described in this appendix was constructed to evaluate sediment contamination effects throughout the Southern California Bight (SCB). The model is an integral part of the effects and exposure characterization of the Ecological Risk Assessment (ERA). In particular, the model is designed to characterize exposure to wide-ranging species of birds and marine mammals, for which ecological risk is associated with the quality of sediments. Changes in the distribution or exposure of chemicals of potential ecological concern (COPECs) in SCB sediments will yield predictions of changes in exposure and ecological risk to fish, birds, and marine mammals.

C.2 Rationale

The results of the model may be used for studies involving the prediction of ecological risk. First, an analysis designed to predict the effects of sediment quality on the ecological risk to selected receptors has provided input into the Baseline ERA for the Palos Verdes Shelf (PVS) and the SCB. The results of the model yield estimated whole-body tissue concentrations for various fish species and estimated average dietary quality for birds and mammals. The dietary concentrations served as exposure-point concentrations (EPCs) in the exposure and risk characterization sections of the ERA.

Second, the model will serve as a tool for future investigations on the effects of changing sediment quality in the PVS and SCB study areas. Model results are based on a knowledge of current, steady-state SCB-wide sediment quality for DDTs and PCBs. The effects of broad-scale degradation, loss, or burial of contaminants over time or as a result of sediment remediation techniques (such as capping) could be estimated using the model. New values may be assigned to discrete sediment areas, and the resulting changes to PVS and SCB-wide ecological risk can then be predicted as part of new steady-state conditions.

C.3 Methods

The basic concept of the model is to take advantage of the relatively high sample density for surface-sediment quality in the PVS and SCB continental shelf to create an SCB-wide sediment quality dataset that can be used to predict bioaccumulated dichloro-diphenyl-trichloroethane (tDDT) and total polychlorinated biphenyls (tPCB) in fish. The fish serve as primary dietary components for marine birds and mammals. Average dietary exposure from fish (and squid) is the basis for modeled ecological risk to higher-order consumers (taking into account their home range, dietary composition, and consumption rates). Risks will vary by modeled or assigned changes to underlying sediment concentrations. The basic, conceptual model structure is shown in Figure C-1. Specific model inputs and outputs are shown in Figure C-2.

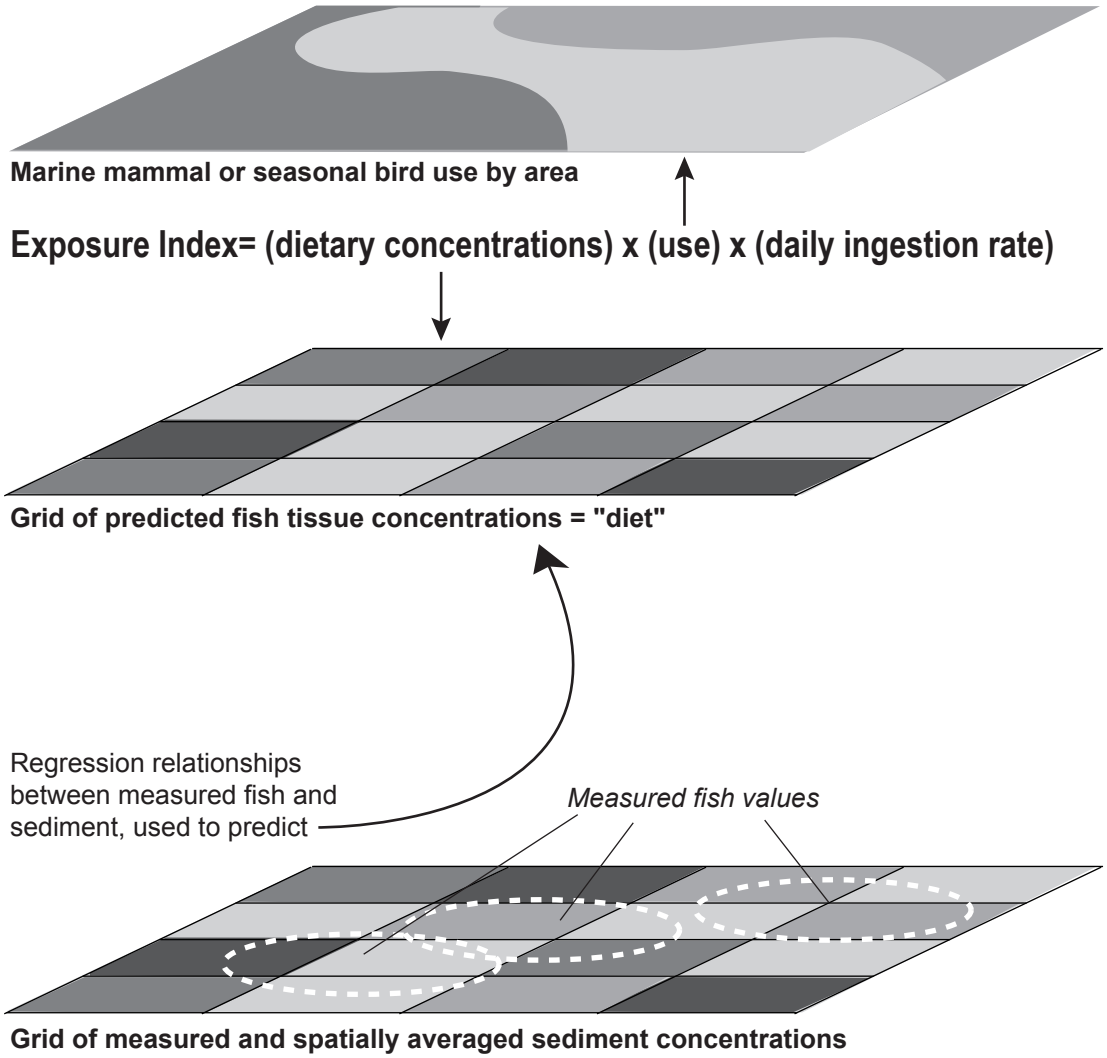


Figure C-1
Conceptual Structure of Food Web Exposure Model
Palos Verdes Shelf Ecological Risk Assessment

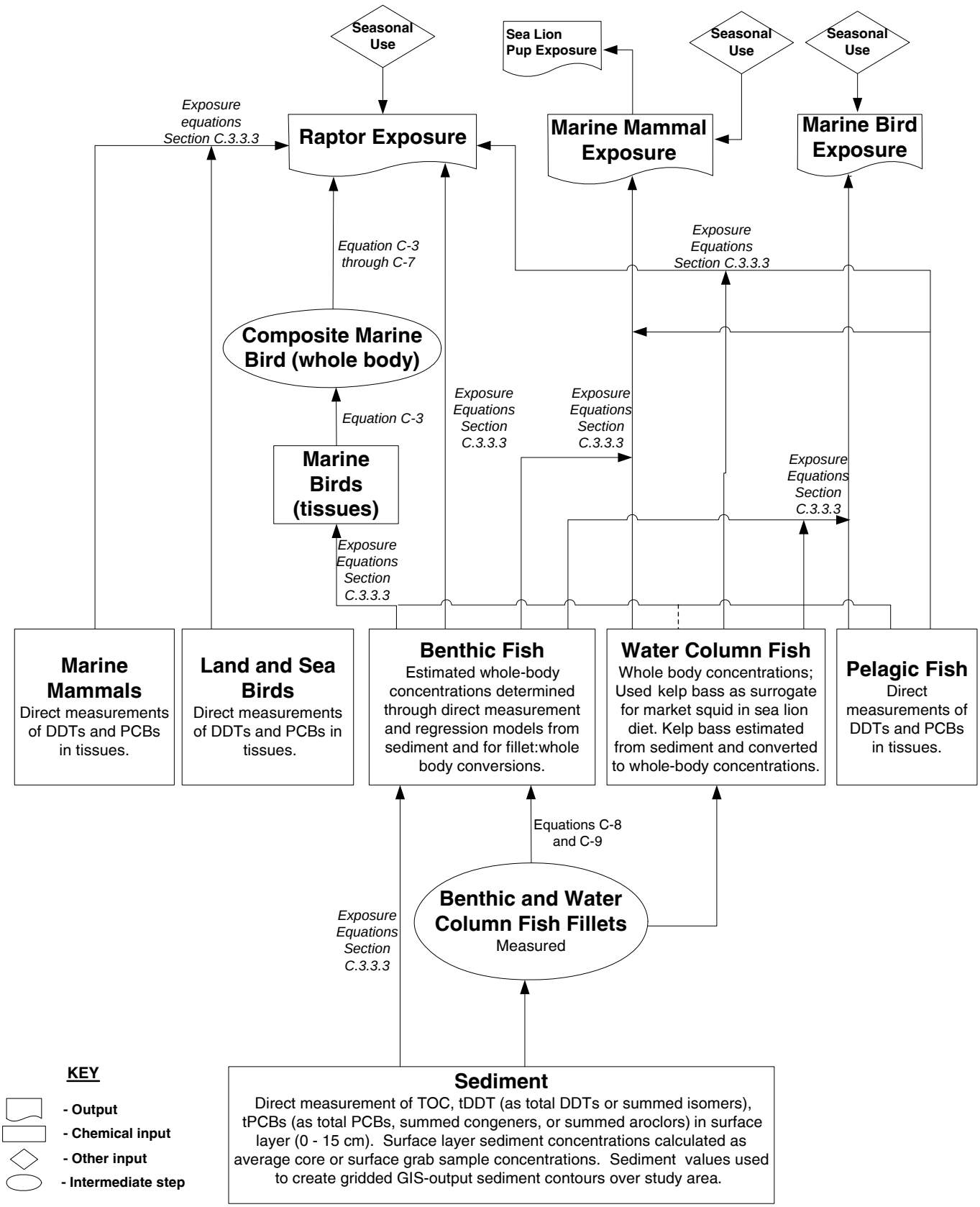


Figure C-2
Conceptual Design of Food Web/Pathways Exposure Model
Palos Verdes Shelf Ecological Risk Assessment

C.3.1 Assumptions

The underlying assumption of the model is that current, steady-state SCB-wide sediment and fish tissue quality can be characterized by media-specific concentrations of DDTs, PCBs, and total organic carbon (TOC) as analyzed from samples collected from 1990 through 2001 by a variety of agencies and institutions (summarized below). Further, it is assumed that DDT and PCB levels in benthic species of fish (white croaker, sanddab species, and Dover sole) and kelp bass are significantly correlated to contaminant concentrations in the co-located sediment. This general relationship has already been shown statistically for PCBs and DDTs in sanddabs (SCCWRP 2002) and demonstrated graphically for the other species, including kelp bass (MSRP 2002). In contrast, it is assumed that such common bird and marine mammal dietary items as pelagic fish are wide-ranging throughout the SCB and that their tissue concentrations are not tied to underlying sediments.

Whole-body fish tissue concentrations of PCBs and DDTs are necessary to predict dietary risk to higher-order consumers. Most of the existing database for DDTs and PCBs in SCB fish is derived from human health risk-related sampling and is, therefore, composed of concentration values from fillets. Predictions of tissue concentrations for white croaker, dover sole, and kelp bass were converted from fillet to whole-body concentrations to facilitate the evaluation of ecological risk. In contrast, whole-body tissue concentration estimates are available for mixed sanddab species as well for various pelagic species (MSRP 2002).

Risks to marine birds are characterized by exposure estimates for brown pelicans and double-crested cormorants. Risks to raptors are assessed by exposure estimates for bald eagles and peregrine falcons. Risks to marine mammals are characterized by exposure estimates for adult female California sea lions and nursing California sea lion pups. Dietary items are assumed as follows:

- Brown pelicans: fish
- Double-crested cormorants: fish
- Bald eagles: fish, mixed seabirds, and sea lion carcasses
- Peregrine falcons: land birds (resident and migratory) and mixed seabirds
- Adult female California sea lions: fish
- Nursing California sea lion pups: milk

Dietary information for all receptors and the proportions of prey in surrogate diets are summarized in Table 3-4 and in Section 3 of this report. Receptor life histories, including diet, are detailed in Appendix B.

Risks to marine birds and mammals were evaluated through characterization of sediment quality in coastal zones of the SCB to a total depth of 200 meters (m). The evaluation area was limited to those locations, as representative of sea lion foraging depths to 200 m (BLM 1981). The water depth for modeling of brown pelican exposure was limited to 150 m, and the water depth for modeling of double-crested cormorant exposure was limited to 22 m (Zeiner et al. 1990).

C.3.2 Estimating Methods

Exposures for birds and sea lions were estimated as the geographic distribution of the receptor (as percentage of time at a certain location) times the estimated tissue concentration of COPECs in the prey species in that location times the daily dietary intake. Resulting exposure values were expressed in standard dosage units, as mg of contaminant/kg body weight consumed per day.

The exposure model for peregrine falcons and bald eagles required that whole-body seabird concentrations be estimated from fish. This was accomplished using the western gull as a representative seabird to develop a "composite seabird" that varies relative to western gull tDDT and tPCB concentrations. This provides a more realistic estimate of the tDDT and tPCB concentrations in the bird component of the peregrine and bald eagle diet.

To estimate tDDT and tPCB body burdens in the western gull, the transfer of these contaminants from sediment to fish to western gull (whole-body, mg/kg) was modeled. The sediment-to-fish model components were the same as those developed for the other receptors and are described later in this section. Western gulls were assumed to eat a diet of 100 percent fish with proportions of prey in the surrogate diet of 50 percent pelagic fish, 5 percent white croaker, 10 percent kelp bass, and 25 percent mixed species of sanddabs. The western gull life history, including diet, is detailed in Appendix B. To estimate uptake from prey to western gull whole-body tissue, models developed in HydroQual, Inc. (1997) were used as described below.

The body burden of the adult western gull can be described by the following toxicokinetic model (HydroQual, Inc. 1997):

$$\frac{dv}{dt} = \alpha \ C \ u_{prey} - (k + G) \ v \qquad \text{[Equation C-1]}$$

Where

$\frac{dv}{dt}$ = change of concentration of tDDT or tPCB over time (mg/kg-d)

α = assimilation efficiency (unitless)

C = consumption rate of the western gull (kg food/kg fresh body weight-day)

u_{prey} = concentration of contaminant in prey (mg/kg)

k = rate of elimination (1/d)

G = growth rate of the western gull (1/d)

v = concentration of tDDT or tPCB in the western gull (mg/kg fresh body weight)

The integral form of the model is:

$$v(t) = \frac{\alpha \ C \ u_{prey}}{k + G}\left[1 - e^{-(k+G)t}\right] \qquad \text{[Equation C-2]}$$

Where

$v(t)$ = concentration of tDDT or tPCB in the western gull at time t (mg/kg fresh body weight)

α = assimilation efficiency (unitless)

C = consumption rate of the western gull (kg food/kg fresh body weight-day)

u_{prey} = concentration of contaminant in prey (mg/kg)

k = rate of elimination (1/d)

G = growth rate of western gull (kg fresh body weight/kg fresh body weight-day)

v = concentration of tDDT or tPCB in the western gull (mg/kg fresh body weight)

Equation C-2 indicates that as the exposure time increases, the body burden of the western gull will attain an equilibrium, and the steady-state concentration of tDDT or tPCB in the western gull would then be:

$$v_{ss} = \frac{\alpha \ C \ u_{prey}}{k + G} \qquad \text{[Equation C-3]}$$

Where

v_{ss} = concentration of tDDT or tPCB in the western gull at steady-state condition (µg/g fresh body weight)

α = assimilation efficiency (unitless)

C = consumption rate of the western gull (kg food/kg fresh body weight-day)

u_{prey} = concentration of contaminant in prey (mg/kg)

k = rate of elimination (1/day)

G = growth rate of western gull (kg fresh body weight/kg fresh body weight-day)

v = concentration of tDDT or tPCB in the western gull (mg/kg fresh body weight)

Once it becomes an adult, the incremental change in body weight typically would remain unchanged, other than temporary seasonal variations. The increase in the incremental growth rate would increase the volume and body mass, which would dilute the tissue concentration of contaminants; on the other hand, a decrease in incremental growth rate would decrease the body mass, which in turn would increase the concentration of contaminant in the body (assuming no change in the amount of contaminant in the body). For the purpose of modeling the body burden of contaminants in the adult western gull, the incremental growth rate is assumed to be negligible (i.e., equal to zero).

Parameter values used in the models, including those specific for tDDT and tPCB, are outlined in Table C-1.

TABLE C-1
Input Parameters used for Estimating tDDT and tPCB Concentrations in Western Gulls
Palos Verdes Shelf Ecological Risk Assessment

Parameter	Value	Unit	Source
α	tDDT = 0.75 tPCB = 1.0	Unitless	HydroQual, Inc. (1997)
C	0.257	kg/kg-day	Table 3-3 (Section 3.1.4.1)
u_{prey}	Modeled concentrations of the western gull prey (i.e., fish)	mg/kg	Modeled in the current Ecological Risk Assessment
k	tDDT = 0.00219[a] tPCB = 0.00438	1/day	HydroQual, Inc. (1997)
G	0[b]	1/day	None

[a] Based on the allometric equation presented in HydroQual, Inc. (1997) using an average adult body weight of 875 grams.

[b] The growth rate is assumed to be zero because the incremental growth rate for the western gull is negligible once it reaches adulthood (Pierotti and Annett 1995).

For determining peregrine falcon exposure, whole-body western gull concentrations were estimated throughout the major foraging areas of the western gull within the SCB (Figure 3-7). Unlike peregrines which breed at several locations within the SCB, bald eagles currently only breed on Santa Catalina Island (Garcelon 2000). Therefore, bald eagles are most likely to prey on western gulls breeding and/or resting on Santa Catalina Island. Western gulls generally forage within 80 km of the colony, with distances within 20 km being common (Pierotti and Annett 1995). Ultimately, the foraging distance depends on availability of reliable food sources and preferences of individual birds. To account for these possible differences, whole-body concentrations of western gulls for use in the bald eagle exposure model were estimated in a 50-km (midpoint between 20 and 80 km) radius of Santa Catalina Island.

Once the whole body concentration of tDDT or tPCB in the western gull is obtained, exposure to the peregrine falcon or bald eagle from the seabird prey source is estimated by calculating the dose from the concentration of tDDT or tPCB in the western gull and the ingestion rate of the peregrine or bald eagle.

A "composite seabird" was constructed that integrates whole-body concentrations of dichlorodiphenyldichloroethylene (DDE) and PCBs from several prey species, normalized to the western gull whole-body concentration. This composite seabird model made it possible to calculate peregrine exposure and partial bald eagle exposure relative to western gull whole-body concentrations as mg/kg wet weight (WW). This is important because future monitoring efforts of peregrine and bald eagle dietary exposure could focus on collection of western gulls to reflect ingestion of a wide variety of seabirds and other waterbirds.

To estimate dietary exposure for tPCB and tDDT from avian prey to peregrine falcons and bald eagles, when whole-body concentrations and dietary proportions are known for all prey items, exposure is calculated as:

Exposure (mg/kg WW) = $$\left(\frac{\sum_{i=1}^{n}(C_i P_i)}{\sum_{i=1}^{n}(P_i)}\right)$$ [Equation C-4]

Where

n = number of prey items (bird species)

C_i = concentration in prey item i (mg/kg WW)

P_i = proportion of diet for prey item i (unitless)

To construct the "composite seabird," the fraction of mean whole-body COPEC concentrations of various prey species, relative to the western gull whole-body concentration, was calculated as $F_i = (S_i/W)$.

Rearranging this equation gives:

$$\mathbf{S_i = (F_i)(W)}$$ [Equation C-5]

Where

W = western gull whole-body concentration (mg/kg WW)

S_i = whole-body concentration of species i (mg/kg WW)

F_i = proportion of whole-body concentration of species S_i to the western gull

Rearranging and substituting F_iW in Equation C-5 for C_i in Equation C-4 yields:

$$\mathbf{C_s} = \left(\frac{\sum_{i=1}^{n}(F_i P_i)}{\sum_{i=1}^{n}(P_i)}\right)$$ [Equation C-6]

Where

C_s = whole-body concentration (tDDT or tPCB) in the composite seabird (mg/kg WW).

The following composite seabird equations were developed for peregrines and bald eagles using the seabird and waterbird dietary proportions and contaminant concentrations detailed in HydroQual, Inc. (1997) and presented in Tables C-2 and C-3. See HydroQual, Inc. (1997) for methods and assumptions associated with mean 4,4'-DDE and PCB concentrations in seabirds and waterbirds:

$$\text{tDDT } C_{s\ (\text{peregrine falcon})} = 0.563*(W)$$ [Equation C-7]

$$\text{tPCB } C_{s\ (\text{peregrine falcon})} = 0.486*(W)$$ [Equation C-8]

$$\text{tDDT } C_{s\ (\text{bald eagle})} = 0.458*(W)$$ [Equation C-9]

$$tPCB\ C_{s\ (bald\ eagle)} = 0.477*(W) \qquad \text{[Equation C-10]}$$

These equations are considered to be accurate, if the same prey species are used as those used to develop the constants in Equations C-7 through C-10. If other prey species are used, it is likely the exposure model will be neither accurate nor precise.

TABLE C-2
Dietary Proportions and Whole-Body 4,4'-DDE and PCB Concentrations in Seabird and Other Waterbird Prey of Peregrine Falcons
Palos Verdes Shelf Ecological Risk Assessment

Prey	Proportion of Diet (energy basis)	Contaminant Levels (mg/kg wet weight)	
		4,4'-DDE	PCB
Western gull	0.096	4.0	0.90
California gull	0.094	2.9	0.90
Heermann's gull	0.022	2.9	0.90
Bonaparte's gull	0.022	2.9	0.90
Cassin's auklet	0.18	2.2	0.42
Other waterbirds[a]	0.26	1.2	0.39

Source: Reproduced from Table 5-3 in HydroQual, Inc. (1997).
[a] Species include grebes, shearwaters, waterfowl, shorebirds, and phalaropes.

TABLE C-3
Dietary Proportions and Whole-Body 4,4'-DDE and PCB Concentrations in Seabird and Other Waterbird Prey of Bald Eagles
Palos Verdes Shelf Ecological Risk Assessment

Prey	Proportion of Diet (energy basis)	Contaminant Levels (mg/kg wet weight)	
		4,4'-DDE	PCB
Western gull	0.033	8.3	2.3
Other gulls[a]	0.013	5.4	1.3
Other waterbirds[b]	0.085	1.7	0.6

Source: Reproduced from Table 5-7 in HydroQual, Inc. (1997).
[a] Heermann's and California gulls.
[b] Species include western grebes, sooty shearwaters, Brandt's cormorant, Cassin's auklet, and Xantus' murrelet.

Seabirds make up 68 percent of the peregrine diet and migratory (13 percent) and land (19 percent) birds make up the remaining 32 percent. The fall distribution of modeled seabird concentrations are included in the peregrine exposure model and the other birds are represented as constants. Therefore, peregrine exposure is depicted across the foraging range of the western gull.

For the bald eagle, seabirds make up only 13 percent of the diet, with fish (81 percent) and marine mammals (6 percent) making up the remaining portion of the diet.

Bald eagles forage within 5 km of their nesting site (Garcelon 1994a and 1994b) and therefore, feed on fish in this localized area surrounding Santa Catalina Island. In contrast, the eagle seabird and sea lion prey forage over large areas, which exposes bald eagles to contaminants from areas across the foraging range of the prey species. As described, seabird body burdens were estimated for a 50-km radius around Santa Catalina Island. For sea lions, it was assumed that bald eagles were most likely to prey on sea lion carcasses from animals using the island as a resting/loafing area. Therefore, body burdens in sea lions (methods described below) were estimated within a 54-km radius of Santa Catalina Island based on the foraging distance of female sea lions during the breeding season. This is a conservative estimate because adult and juvenile males are generally observed on and near the island (Le Boeuf 2002) and males likely forage across further distances than females during the breeding season. Because the individual species in the three components of the bald eagle diet represent three different foraging areas, the 95 percent UCL of the geometric mean was calculated from the body burden distributions modeled for seabirds and sea lions and was used in the bald eagle exposure model as a constant. The fish component was modeled over the 5 km foraging area of the bald eagle.

Sea lion diets are assumed to be composed of 50 percent mixed pelagic species, 20 percent kelp bass, 5 percent white croaker, and 25 percent mixed species of sanddabs.

To estimate the effect of food chain transfer of tDDT and tPCB, a mass-balance model was used to propagate the concentration of tDDT and tPCB from fish to the adult female sea lion, which ultimately exposes sea lion pups to tDDT and tPCB through milk. For exposure to the bald eagle, a mass-balance model was used to estimate adult male body burdens of tDDT and tPCB from the fish diet. The body burden of the adult female and male sea lion can be described by the following toxicokinetic model (HydroQual, Inc. 1997):

$$\frac{dv}{dt} = \alpha \ C \ u_{prey} - (k + G) \ v \qquad \text{[Equation C-11]}$$

Where

$\frac{dv}{dt}$ = change of concentration of tDDT over time (mg/kg-d)

α = assimiliation efficiency (unitless)

C = consumption rate of the adult sea lion (kg food/kg fresh body weight-day)

u_{prey} = concentration of contaminant in prey (mg/kg)

k = rate of elimination (1/d)

G = growth rate of the sea lion (1/d)

v = concentration of tDDT in the sea lion (mg/kg fresh body weight)

The integral form of the model is:

$$v(t) = \frac{\alpha \ C \ u_{prey}}{k + G}\left[1 - e^{-(k+G)t}\right] \qquad \text{[Equation C-12]}$$

Where

$v(t)$ = concentration of tDDT or tPCB in the sea lion at time t (mg/kg fresh body weight)

α = assimilation efficiency (unitless)

C = consumption rate of the adult sea lion (kg food/kg fresh body weight-day)

u_{prey} = concentration of contaminant in prey (mg/kg)

k = rate of elimination (1/d)

G = growth rate of the sea lion (kg fresh body weight/kg fresh body weight-day)

v = concentration of tDDT or tPCB in the sea lion (mg/kg fresh body weight)

Equation 12 indicates that as the exposure time increases, the body burden of the sea lion will reach an equilibrium, and the steady-state concentration of tDDT in the adult female sea lion would then be:

$$v_{ss} = \frac{\alpha \ C \ u_{prey}}{k + G} \qquad \text{[Equation C-13]}$$

Where

v_{ss} = concentration of tDDT or tPCB in the sea lion at steady-state condition (mg/kg fresh body weight)

α = assimilation efficiency (unitless)

C = consumption rate of the adult sea lion (kg food/kg fresh body weight-day)

u_{prey} = concentration of contaminant in prey (mg/kg)

k = rate of elimination (1/day)

G = growth rate of the sea lion (kg fresh body weight/kg fresh body weight-day)

v = concentration of tDDT or tPCB in the sea lion (mg/kg fresh body weight)

Once a sea lion becomes an adult, the incremental change in body weight typically would remain unchanged (HydroQual, Inc. 1997), other than temporarily seasonal variations. An increase in incremental growth rate would increase the volume and body mass, which would dilute the tissue concentration of contaminants; on the other hand, a decrease in incremental growth rate would decrease the body mass, which in turn would increase the concentration of contaminant in the body (assuming no change in the amount of contaminant in the body). For the purpose of modeling the body burden of contaminants in the adult sea lion, the incremental growth rate is assumed to be negligible (i.e., equal to zero).

The final result of equation C-13 is a wet weight body concentration of tDDT or tPCB for adult male or female sea lions (depends on excretion rate used). Once the whole-body concentration of tDDT or tPCB in the adult female sea lion is obtained, the concentration of

tDDT or tPCB in the milk of lactating female sea lions can be estimated using the relationship presented in HydroQual, Inc. (1997):

$$v_{milk} = \left[\frac{1 + K_{MLA} x_{L_m}}{1 + K_{LA} x_L} \right] v_{ss} \qquad \text{[Equation C-14]}$$

Where

v_{milk} = concentration of tDDT or tPCB in female sea lion's milk (mg/kg)

K_{MLA} = the equilibrium partition coefficient between the milk lipids and the whole body non-lipid fraction (unitless)

X_{LM} = the lipid fraction of the milk

K_{LA} = the equilibrium partition coefficient of tDDT or tPCB between the lipid and the aqueous phase of the sea lion (unitless)

X_L = the lipid fraction of the sea lion

v_{ss} = concentration of tDDT or tPCB in the sea lion at steady-state condition (mg/kg fresh body weight)

Finally, exposure of tDDT or tPCB to sea lion pups is estimated by calculating the dose from the concentration of tDDT or tPCB in milk and the milk ingestion rate of the pup. A weighted average of the milk ingestion rate over the first 7 months of the sea lion pup's life was derived from published data on sea lion pup body weights and milk intake rates for months 1, 2, 3, and 7 postpartum (Table C-4). Body weights and milk intake rates for months 4, 5, and 7 postpartum were interpolated from the available data. From these, milk ingestion rates were calculated as grams of milk ingested per gram of body weight per day. The average milk ingestion rate for male and female pups over the 7 month period (0.0488 kg/kg BW/d) was used in the model.

Parameter values used in the exposure models, including those specific for tDDT and tPCB and those specific for adult males and females, are presented in Table C-5.

TABLE C-4
Average Milk Intake Rate by Sea Lion Pups [a]
Palos Verdes Shelf Ecological Risk Assessment

Month Postpartum	Body weight (kg)		Milk intake rate (g/d)		Normalized milk intake rate[b] (g/g-d)	
	Male	Female	Male	Female	Male	female
1st	8.79	7.76	773 [c, d]	631 [c, d]	0.088	0.081
2nd	10.32	9.12	681 [c, d]	600 [c, d]	0.066	0.066
3rd	22.7	15	768 [e]	772 [e]	0.034	0.051
4th	22 [f]	18 [f]	845 [g]	801 [g]	0.0388	0.0456
5th	26 [f]	21 [f]	891 [g]	868 [g]	0.0344	0.0412

TABLE C-4
Average Milk Intake Rate by Sea Lion Pups [a]
Palos Verdes Shelf Ecological Risk Assessment

Month Postpartum	Body weight (kg)		Milk intake rate (g/d)		Normalized milk intake rate[b] (g/g-d)	
	Male	Female	Male	Female	Male	female
6th	30 [f]	25 [f]	938 [g]	935 [g]	0.0313	0.0381
7th [h]	33	28	1000 [i]	1000 [i]	0.0303	0.0357
Average	21.8	17.6	845	801	0.0462	0.0513
Average of male and female pups					0.0488	

[a] Source: Oftedal et al. (1987); values taken from this reference are those reported for 1982; 1984 values were not included because those were collected during an El Niño year.
[b] Calculated by dividing the milk intake rate by the body weight
[c] Values reported in Oftedal et al. (1987); developed using age-specific data for the sea lion pups one month and two month postpartum
[d] No statistically significant difference was found between the age groups (Oftedal et al. 1987)
[e] Estimated using the equation presented in Oftedal et al. (1987)
[f] Estimated by linear extrapolation using the body weight data from the first, second, third, and the seventh month postpartum for males and females, respectively
[g] Estimated by linear extrapolation using the milk ingestion data from the first, second, third, and the seventh month postpartum for males and females, respectively
[h] Pups older than the first 200 days may begin to ingest solid food (Boness et al. 1991); thus, milk ingestion rate is not estimated beyond the seventh month postpartum
[i] Value reported in HydroQual, Inc. (1997)

TABLE C-5
Input Parameters Used for Estimating tDDT and tPCB Concentrations in Sea Lion Pups
Palos Verdes Shelf Ecological Risk Assessment

Parameter	Value	Unit	Source
α	tDDT = 0.75 tPCB = 1.0	unitless	HydroQual, Inc. (1997)
C	0.116	kg/kg-day	Table 3-3 (Section 3.1.4.1)
u_{prey}	Modeled concentrations of the sea lion prey (i.e., fish)	mg/kg	Modeled in the current Ecological Risk Assessment
k	0.00045 for males 0.01 for lactating females	1/day	HydroQual, Inc. (1997)
G	0[a]	1/day	None
K_{MLA}	DDT = $1x10^6$ PCBs = $1x10^7$	unitless	HydroQual, Inc. (1997)
X_{LM}	0.4057[b]	unitless	
K_{LA}	DDT = $1x10^6$ PCBs = $1x10^7$	unitless	HydroQual, Inc. (1997)
X_L	0.30	unitless	HydroQual, Inc. (1997)

[a] Growth is assumed to be zero because the growth curve presented in HydroQual, Inc. (1997) for lactating female adult sea lions indicates that the incremental growth of adult sea lions is negligible.

[b] Based on the weighted average lipid fraction of sea lion milk over the first 7 months postpartum (32 percent for months one and two and 44 percent for months three through seven; HydroQual, Inc. 1997).

The brown pelican, double-crested cormorant, and sea lion diets are composed wholly of fish, as are portions of the bald eagle diet. Additionally, seabird and marine mammals prey of bald eagles and seabird prey of peregrines have diets composed wholly of fish. Therefore, measured fish and sediment data were used to develop site-specific bioaccumulation models describing sediment-to-fish transfer of tDDT and tPCB.

Chemical concentration data used as the basis for the model consist of spatially distributed sediment and fish tissue values as summarized from 1990 to 2003. Data sources are listed in Table 2-3. Surface sediment data (grab samples or 0-15 cm core sections) were plotted by location and spatially averaged for concentration in the coastal areas within the SCB bounded by dry land and the 200-m depth bathymetric contour. GIS-based techniques for spatial averaging (kriging) are discussed in the GIS section below. Summaries of sediment and fish tissue sample values used in the model are shown in Tables 2-7 and 2-10.

It was necessary to establish predictive relationships between sediment and benthic fish tissue concentrations to create the model. Pelagic fish are wide-ranging and assumed not to be significantly linked to underlying sediment quality. Because the home ranges of the benthic species was unknown (Allen 2002), several possible areas were assigned to each fish sampling location. Spatially averaged sediment concentrations corresponding to the areas under several potential home ranges (centered on the fish sample location) were compared to the associated fish concentrations using simple linear regressions with log-transformed data. All species tested showed significant, positive linear regressions between sediment concentrations of tPCB or tDDT and corresponding fish tissue. All regression relationships were established using log-transformed values to improve normality. In some cases, the best relationships required sediment normalization to TOC and/or fish tissue normalization to lipid concentrations.

The species varied as to which potential home range yielded a regression relationship that explained the greatest degree of variability. Based on the variability of tissue concentrations of COPECs, kelp bass, and white croaker were estimated to have home ranges with diameters of 2,000 m and 10,000 m, respectively. Sanddabs were evaluated from point source sediment samples as was presented in Bight 98 reports (SCCWRP 2002).

The regression analysis results are given in Table C-6. All predicted values were back-transformed to yield geometric mean fillet concentrations for given sediment values.

Sanddab guild values consist of cumulative results based on longfin sanddab, Pacific sanddab, speckled sanddab, California halibut, and slender sole (SCCWRP 2002).

TABLE C-6
Linear Regression Relationships for Predicting Dietary Concentrations: Fish Fillet Concentrations as Predicted from Surface Sediment Concentrations.
Palos Verdes Shelf Ecological Risk Assessment

Analyte	Fish Species	Regression Equation[a, b]	r^2; n; range[c]
DDTs	White croaker	logDDT(F)=0.214 + 0.750(logSedDDT)	0.766; 220; 10,000m
DDTs	Kelp bass	logDDT(F/%lipid)=-0.657 + 0.319(log(SedDDT/TOC))	0.366; 153; 2,000m
DDTs	Sanddab guild	logDDT(WB)=3.416+1.02(log(SedDDT/TOC))	0.641; 76; <0.0001[d]
DDTs	Composite pelagic species	0.0147 (WB)	Constant value[d]
DDTs	Market squid	0.0147 (WB)	Constant value[d]

TABLE C-6
Linear Regression Relationships for Predicting Dietary Concentrations: Fish Fillet Concentrations as Predicted from Surface Sediment Concentrations.
Palos Verdes Shelf Ecological Risk Assessment

Analyte	Fish Species	Regression Equation[a, b]	r^2; n; range[c]
PCBs	White croaker	logPCB(F/%lipid)=-0.438 + 0.888(logSedPCB)	0.638; 199; 10,000m
PCBs	Kelp bass	logPCB(F/%lipid)=-0.83 + 0.301* (logSedPCB/TOC)	0.313; 153; 2,000m
PCBs	Sanddab guild	logPCB(WB)=2.905 + 0.706* (log(SedPCB/TOC))	0.505; 34
PCBs	Composite pelagic species	0.0077 (WB)	Constant value[d]
PCBs	Market squid	0.0077 (WB)	Constant value[d]

[a] All regressions are statistically significant, P < 0.001 except for PCBs in kelp bass.
[b] F = fillet concentrations, mg/kg, wet-weight basis; WB = whole body concentrations, mg/kg, wet-weight basis; S = sediment concentrations, mg/kg, dry-weight basis.
[c] Range denotes the home range diameter (over which sediment concentrations were averaged) that produced the best linear fit.
[d] No home range estimated for sanddab guild species, squid, or pelagic fish.
% lipid = Lipid % of fish tissue <1.9 for white croaker and 4.2 for kelp bass

Pelagic fish and market squid were assumed to be uniformly constant throughout the SCB. Pelagic species tissue concentrations were computed as the geometric mean value for combined whole-body concentrations in mackerel, Pacific bonito, and barracuda from the SCB (HydroQual, Inc. 1997).

The California sea lion has a varied diet, of which 10 percent consists of market squid. To assess dietary exposure to the California sea lion, it was necessary to have measured or estimated contaminant levels in its various diet items. Recent body burden data were not available for market squid in the SCB. The most recent data, from 1981, consisted of only three samples (Mearns et al. 1991; Table C-7). Because of these extremely limited data, pelagic fish were used as a surrogate for the market squid portion of the sea lion diet. These fish were selected as surrogate species because both the market squid and fish are water column feeders. Additionally, their diets are similar. Both species feed on crustaceans, such as the pelagic red crab, and on small fishes like anchovies, sardines, surfperch, and queenfish. Pelagic fish and market squid are both water column feeders that forage over wide ranges, though market squid are short-lived and concentrated along the continental shelf (PFMC 1998).

TABLE C-7
Tissue Sample Data from the SCB
Palos Verdes Shelf Ecological Risk Assessment

Species	Analyte	Year	Number of Samples	Mean Tissue Concentration (mg/kg WW)	Range
Market Squid	DDT	1980-1981	3	0.01	0.006 – 0.031
	PCB	1980-1981	3	0.01	0.003 – 0.24

Source: Mearns et al. (1991)

Three of the regression predictions for fish from sediment were based on databases of fillet concentrations. Unfortunately, whole-body to fillet contaminant concentration relationships are lacking for SCB species. In general, whole-body concentrations of organochlorine

contaminants are greater than those in muscle tissue, given the higher lipid content of the whole body and the strong lipid affinity of the bioaccumulated compounds within certain tissues. A literature search was conducted to compile comparative data for seven freshwater species. Literature sources and species used are summarized in Table C-8. Predictive regression models yielding whole-body contaminant concentrations as estimated from fillet measurements were established for both tDDT and tPCB. Species used in the regressions included catfish, carp, largemouth bass, smallmouth bass, walleye, brook trout, and lake trout. The species span a large range in lipid content of fillets versus whole body. For example, lake trout tissues are essentially identical at approximately 10-15 percent lipid in both whole body and fillets, whereas walleyes have 1.5 percent lipid in fillet vs. 6 percent lipid in whole-body samples (Michigan DEQ 2002). It is recognized that organochlorine partitioning between muscle and other body tissues varies among species, sex, and age (e.g., Bevelhimer et al. 1997). It is also known that lipid normalization is likely to provide the same concentrations among tissues within any given fish, indicating that the organochlorine contaminants are almost totally lipid-associated (Baker and Toaspern 2002; Hebert and Keenleyside 1995). Despite these known sources of variability, the pooled species, sexes, and ages from Table C-8 data provided useful, predictive, log-log regressions as given below and shown in Figure C-3. The relationships were used to predict log, whole-body tissue concentrations and were then back-transformed to yield geometric mean, whole-body tissue concentrations. This portion of the model should be replaced with species-specific conversion relationships, should they become known.

TABLE C-8
Sources of Data and Species Used in Developing Fillet to Whole-Body Tissue Concentration Relationships for DDTs and PCBs
Palos Verdes Shelf Ecological Risk Assessment

Data and Source	Species of Fish	Number of Samples
DDTs, PCBs: Mierzykowski and Carr (1998), Bevelhimer et al. (1997)	Smallmouth bass	38
DDTs, PCBs: Mierzykowski et al. (1993), Bevelhimer et al. (1997)	Largemouth bass	33
DDTs, PCBs: Michigan DEQ (2002)	Lake trout	20
DDTs, PCBs: King et al. (1997)	Carp	2
DDTs, PCBs: Mierzykowski and Carr (1998)	Brook trout	5
DDTs: Mierzykowski et al. (1993), PCBs: Bevelhimer et al. (1997)	Bullhead	14

The pooled fish equations for converting fillet to whole-body concentration are:

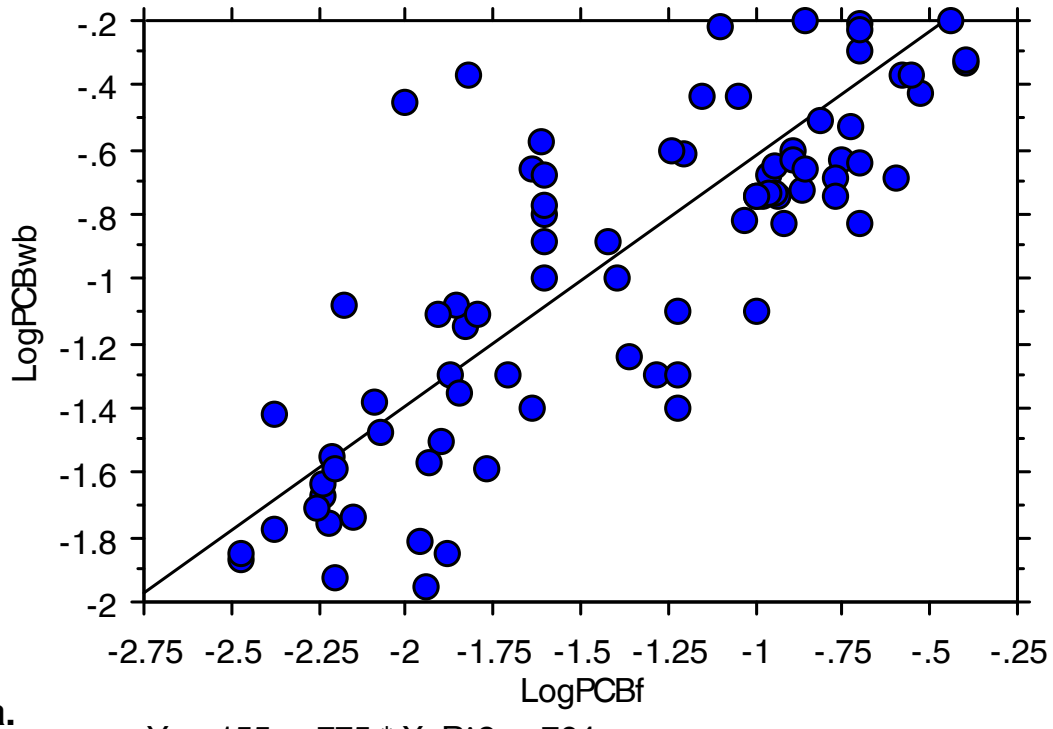
$$\text{LogDDT (WB)} = 0.138 + 0.845 * \log(F) \qquad r^2 = 0.844,\ n = 51 \qquad \text{[Equation C-15]}$$

$$\text{LogPCB (WB)} = 0.155 + 0.775 * \log(F) \qquad r^2 = 0.764,\ n = 106 \qquad \text{[Equation C-16]}$$

Where

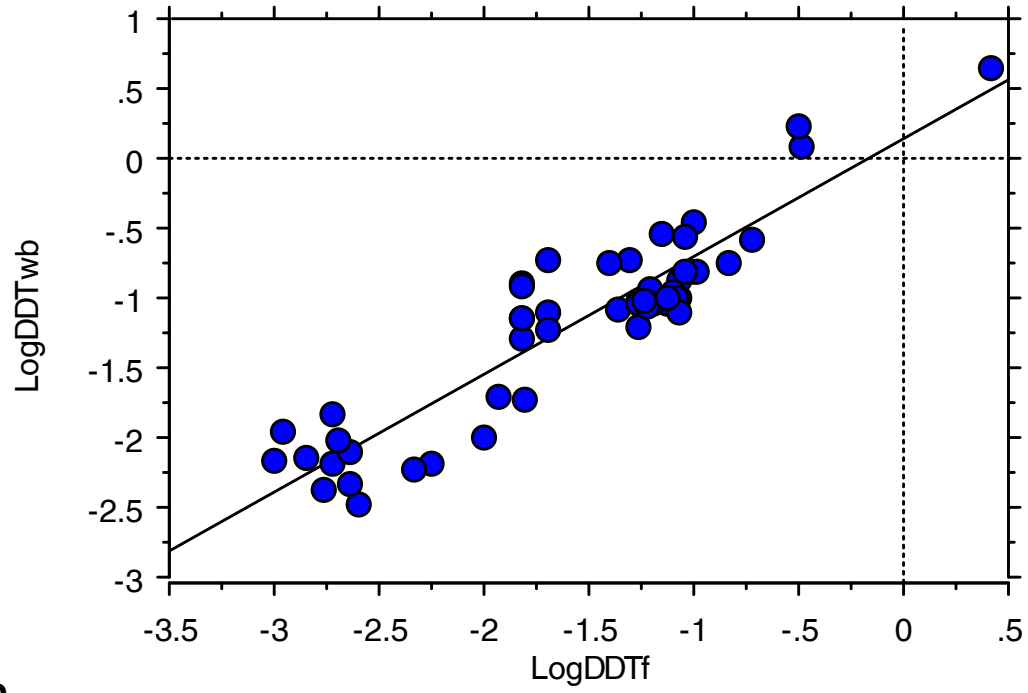
F = fillet (mg/kg WW)

WB = whole-body tissue contaminant concentrations (mg/kg WW)



a.

Y = .155 + .775 * X; R^2 = .764



b.

Y = .138 + .845 * X; R^2 = .844

Lines are linear regression lines of best fit. Data compiled from six freshwater species (Table C-8).

Figure C-3
Whole Body Tissue Concentrations (mg/kg WW) Predicted from Fillet Concentrations, for Total PCBs (a) and Total DDTs (b)
Palos Verdes Shelf Ecological Risk Assessment

The foraging ranges of marine birds and mammals were established through a search of recent literature and conversations with local SCB-area researchers (as discussed in Section 2). Ranges used to establish overlap with dietary exposure estimates are shown in Figures 3-4, 3-7, and 3-11. Differing distributions based on season or breeding condition of the birds or mammals were incorporated wherever possible.

C.3.3 GIS

C.3.3.1 Methods Used to Produce Gridded Data Representing Contaminant Distributions in Sediment

To process the data electronically, it was necessary to interpolate the measured concentrations of DDTs and PCBs in shallow sediment onto a regularly spaced grid. This process of interpolation is referred to as gridding. The grid spacing used was 100 m. The resulting grids contain an array of values, each representing the average, interpolated concentration over a 10,000 square-meter (m^2) area.

Sediment data were generally sparse in areas around the Channel Islands and the offshore shallow banks. It was necessary to assign estimated sediment concentrations in these areas for the gridding program to produce meaningful data. Two methods were used to develop estimates of sediment concentrations in these areas. In areas where fish tissue samples were available, the sediment concentrations were inferred from areas where similar fish tissue concentrations were found. In areas where no fish tissue values were available, the sediment concentrations were assumed to decrease with distance from the coastline and from the sewer outfall, and concentrations were assigned consistent with this assumed trend.

The data were gridded using Surfer version 7 surface mapping system software (Golden Software, Inc., Golden Colorado). The Inverse Distance to a Power gridding method was used to produce the gridded data. With Inverse Distance to a Power, data are weighted during interpolation such that the influence of one point relative to another declines with distance from the grid node. Weights are assigned to data points through the use of a weighting power that controls how the weighting factors drop off as distance from a grid node increases. The greater the weighting power, the less effect points far from the grid node have during interpolation. As the power increases, the grid node value approaches the value of the nearest point. For a smaller power, the weights are more evenly distributed among the neighboring data points. Weighting factors used in gridding the data ranged from 3 to 6.

The large differences in concentrations and data density between the PVS and the offshore islands presented problems during gridding. The relatively large concentrations and high data density on the PVS tended to overwhelm the small concentrations and low data density near the islands. This resulted in concentrations around the islands being overestimated. To mitigate this problem, the full data set was broken up into several smaller data sets. Each of these smaller data sets was gridded individually in Surfer; then these individual grids were combined. This method produced a gridded data set that more closely represented the measured data.

The presentation of sediment quality as contoured values requires making choices concerning the boundaries of the areas to be contoured. In the case of PVS, the high sample density allowed us to divide the shoreline from the area immediately offshore that was significantly more contaminated. Inshore, less-contaminated stations were used as a boundary because they represent actual observations of less-contaminated sediment, but if

not separated, were "overwhelmed" by the contouring program that would otherwise depict highly contaminated sediments all the way to the shore. The use of this technique has produced straight lines along the PVS shoreline for all tDDT and tPCB sediment quality figures. In addition, for the greater PVS and SCB areas, Los Angeles and Long Beach harbors were contoured for sediment concentrations inside the breakwater separately from all areas outside the breakwater. For all sediment quality figures, the seaward contouring boundary was set at the 200-m depth contour, except for the immediate PVS area where the contour boundary was set deeper to accommodate the availability of sampling results.

C.3.3.2 Data Use and Sources

The following datasets were used to generate geographic distributions (i.e., foraging, breeding areas) and sediment concentration contours. Together, these data elements were used to determine exposure by calculating the overlap in areal use and COPEC distribution:

- UCSD's bathymetry data
- Fish locations
- Sediment sample locations and concentration data
- Brown pelican foraging areas
- Brown pelican breeding areas
- Cormorant foraging area
- Cormorant breeding area
- Western gull foraging area (representative of peregrine falcon exposure)
- Bald eagle foraging area
- Sea lion use by area

C.3.3.3 GIS Processing

All data were projected/developed using the UTM coordinate system Zone 11 North American Datum 1983 with units of meters.

Sediment Concentration Layer Development

Two sediment concentration layers (0-15 cm and 15-30 cm) for DDTs, PCBs and TOC were developed by CH2M HILL in Surfer and imported into the GIS as floating-point GRIDs with 100-m grid cell size.

Relating Sediment Concentration to Fish Concentration for DDT and PCB

We related the tDDT, tPCB, and TOC sediment concentration layers in the GIS to calculate an average concentration for each unique fish sample location. There were 105 unique locations in the fish sample data. GIS macros were created to process each location. We "looped" through these locations buffering each location by a radius of 500 m, 1,000 m, and 2,000 m. Using overlay analysis we calculated the average concentration for tDDT, tPCB, and TOC for each location and buffer size. We stored these averages as attributes in the GIS for fish sampling.

Modeling the Spatial Distribution of Bird and Mammal Exposure

We developed GIS macros and layers to model the spatial distribution of tDDT and tPCB exposure for birds and mammals using ESRI's GRID module and grid mathematics. We used information on percentage use areas for sea lions at three periods throughout the year: December through May, June through August, and September through November. For

birds, we developed pelican, cormorant, western gull, and bald eagle foraging areas, and also developed breeding areas for pelicans and cormorants.

Using the sediment concentration/fish concentration modeling results, we developed a spatial distribution of DDT and PCB fish concentrations based on the GIS layers in the sediment for three fish species:

- Sanddab DDT Concentration = exp10(3.416 + 1.02 * log10((Sediment DDT Layer/ Sediment TOC Layer)))
- Sanddab PCB Concentration = exp10(2.905 + 0.706 * log10(Sediment PCB Layer / Sediment TOC Layer))
- Kelp Bass DDT Concentration = exp10(0.138 + 0.845 * [log10(exp10(-0.657 + 0.319 * log10(Sediment DDT Layer/ Sediment TOC Layer)))] * 4.2)
- Kelp Bass PCB Concentration = exp10(0.155 + 0.775 * [log10(exp10(0.830 + 0.301 * log10(Sediment PCB Layer/Sediment TOC Layer)))] * 0.42)
- White Croaker DDT Concentration = exp10(0.138 + 0.845 * LOG10(exp10(0.214 + 0.750 * LOG10(Sediment DDT Layer))))
- White Croaker PCB Concentration = exp10(0.155 + 0.775 * [LOG10(exp10(0.438 + 0.888 * LOG10(Sediment PCB Layer)))] *1.9)

For birds and mammals, diet GIS layers were produced based on the percentage of each fish species consumed:

- Pelican DDT Diet = (0.20 * Kelp Bass DDT Concentration) + (0.80 * 0.0147)
- Pelican PCB Diet = (0.20 * Kelp Bass PCB Concentration) + (0.80 * 0.0077)
- Cormorant DDT Diet = (0.05 * White Croaker DDT Concentration) + (0.3 * Kelp Bass DDT Concentration) + (0.15 * Sanddab DDT Concentration) + (0.5 * 0.0147)
- Cormorant PCB Diet = (0.05 * White Croaker PCB Concentration) + (0.3 * Kelp Bass PCB Concentration) + (0.15 * Sanddab PCB Concentration) + (0.5 * 0.0077)
- Bald Eagle DDT Diet = (0.50 * Kelp Bass DDT Concentration) + (0.10 * Sanddab DDT Concentration) + (0.21 * 0.0147) + (0.06 * 49.9) + (0.13 * 13.10 * 0.458)
- Bald Eagle PCB Diet = (0.05 * Dover Sole PCB Concentration) + (0.50 * Kelp Bass PCB Concentration) + (0.05 * Sanddab PCB Concentration) + (0.21 * 0.0077) + (0.06 * 31.24) + (0.13 * 5.37 * 0.477)
- Western Gull DDT Diet = (0.05 * White Croaker DDT Concentration) + (0.1 * Kelp Bass DDT Concentration) + (0.15 * Sanddab DDT Concentration) + (0.7 * 0.0147)
- Western Gull PCB Diet = (0.05 * White Croaker PCB Concentration) + (0.1 * Kelp Bass PCB Concentration) + (0.15 * Sanddab PCB Concentration) + (0.7 * 0.0077)
- Sea Lion DDT Diet = (0.05 * White Croaker DDT Concentration) + (0.2 * Kelp Bass DDT Concentration) + (0.25 * Sanddab DDT Concentration) + (0.5 * 0.0147)

- Sea Lion PCB Diet = (0.05 * White Croaker PCB Concentration) + (0.2 * Kelp Bass PCB Concentration) + (0.25 * Sanddab PCB Concentration) + (0.5 * 0.0077)

Sea lion exposure GIS layers were produced by multiplying the sea lion diet layer by the percent use layer by the amount of fish ingested per day:

- Sea Lion DDT Exposure Sep. – Nov = (Sea Lion DDT Diet * Sea Lion Use Sep. - Nov) * 0.116
- Sea Lion PCB Exposure Sep. – Nov = (Sea Lion PCB Diet * Sea Lion Use Sep. - Nov) * 0.116

These calculations were repeated for the other two sea lion use area layers that represented the varying times of the year:

- Sea Lion DDT Exposure Jun - Aug = (Sea Lion DDT Diet * Sea Lion Use Jun - Aug) * 0.116
- Sea Lion PCB Exposure Jun - Aug = (Sea Lion PCB Diet * Sea Lion Use Jun - Aug) * 0.116
- Sea Lion DDT Exposure Dec. - May = (Sea Lion DDT Diet * Sea Lion Use Dec. - May) * 0.116
- Sea Lion PCB Exposure Dec. - May = (Sea Lion PCB Diet * Sea Lion Use Dec. - May) * 0.116

Sea lion pup exposure GIS layers were produced by multiplying the adult sea lion exposure layer by the uptake to adult tissue, the transfer factor from adult tissue to milk, and the amount of milk ingested per day by the pups.

- Sea Lion Pup DDT Exposure Jun - Aug = [Sea Lion DDT Exposure Jun - Aug] * (0.75/0.01) * 1.35233 * 0.0488

- Sea Lion Pup DDT Exposure Sep - Nov = [Sea Lion DDT Exposure Sep. – Nov] * (0.75/0.01) * 1.35233 * 0.0488

- Sea Lion Pup DDT Exposure Dec. - May = [Sea Lion DDT Exposure Dec. - May] * (0.75/0.01) * 1.35233 * 0.0488

- Sea Lion Pup PCB Exposure Jun - Aug = [Sea Lion PCB Exposure Jun - Aug] * (1/0.01) * 1.35233 * 0.0488

- Sea Lion Pup PCB Exposure Sep-Nov = [Sea Lion PCB Exposure Sep. – Nov] * (1/0.01) * 1.35233 * 0.0488

- Sea Lion Pup PCB Exposure Dec. - May = [Sea Lion PCB Exposure Dec. - May] * (1/0.01) * 1.35233 * 0.0488

Pelican and cormorant exposure values for tDDT and tPCB were calculated for both foraging and breeding areas:

- Pelican DDT Exposure = (Pelican DDT Diet * 0.15)
- Pelican PCB Exposure = (Pelican PCB Diet * 0.15)
- Cormorant PCB Exposure = (Cormorant PCB Diet * 0.18)
- Cormorant DDT Exposure = (Cormorant DDT Diet * 0.18)

Western gull body burden layers were produced for the foraging areas by multiplying the diet by the amount of fish ingested per day and by the uptake of tDDT or tPCB from the diet.

- Western Gull DDT Body Burden = (Western Gull DDT Diet * 0.257) * (0.75/0.0219)

- Western Gull PCB Body Burden = (Western Gull PCB Diet * 0.257) * (1/0.00438)

Peregrine exposure layer produced by multiplying the western gull body burden by the composite bird ratio and by the amount of bird prey ingested per day.

- Peregrine DDT Exposure = [(0.68* Western Gull DDT Body Burden * 0.563) +(0.19*0) + (0.13*0.33)] * 0.193
- Peregrine PCB Exposure = [(0.68* Western Gull PCB Body Burden * 0.486) +(0.19*0) + (0.13*0.26)] * 0.193

For bald eagles, it was necessary to estimate tDDT and tPCB concentrations for the seabird and marine mammals portions of the diet. The western gull (body burden equations provided above) and sea lions (body burden equations provided below) were used as surrogates for these portions of exposure. Sea lion body burdens were calculated for the foraging areas around Santa Catalina Island by multiplying the diet by the amount of fish ingested per day and by the uptake of tDDT or tPCB from the diet. Western gull body burden values were calculated within a 50 km radius of Santa Catalina Island and sea lion body burden values were calculated within a 54 km radius. The 95 percent UCLs from these body burden distributions were used in the bald eagle diet equations (above).

- Sea Lion DDT Body Burden = (Sea Lion DDT Diet * 0.116) * (0.75/0.00045)
- Sea Lion PCB Body Burden = (Sea Lion PCB Diet * 0.116) * (1/0.00045)

The bald eagle foraging area layer was used to develop the bald eagle exposure values:

- Bald Eagle DDT Exposure = (Bald Eagle DDT Diet * 0.13)
- Bald Eagle PCB Exposure = (Bald Eagle PCB Diet * 0.13)

C.3.4 Calibration and Statistical Error

The simple linear regressions forming the basis of the model include documented errors in predictive ability. They are given as the r^2 values in Table C-6. The gridding function in GIS, key assumptions, use of weighting factors, and the need for multiple data sets are described above.

Measured PVS study area fish tissue concentrations (e.g., Figures 3-2 and 3-3) can be compared to predicted values for fish as a means of checking the model's accuracy. Table C-9 shows maximum and central tendency for both data sets. The comparison indicates that, for white croaker and kelp bass, highly contaminated individual fish were not reproduced as maximum values in the model (with the exception of tPCB for kelp bass). Maximum and 95 percent UCLs are presented in Table C-9 to illustrate these differences. However, that is an expected characteristic, common to all such models. Differences between mean values showed the model underestimated mean tDDT and tPCB in white croaker, and overestimated mean tDDT and tPCB in kelp bass.

Measured SCB tDDT and tPCB concentrations in western gulls and adult sea lions can be compared to predicted western gull and adult sea lion values as a means of checking the model's accuracy. To verify our model, we converted our tDDT and tPCB concentrations in sea lions from whole-body concentrations to blubber (shown in parentheses below) concentrations, assuming all tDDT and tPCB accumulates in the blubber [a reasonable assumption given that the log K_{ow} of DDT is 6.0 and that for PCBs is 7.0 (HydroQual, Inc.

1997). A lipid content of 30 percent for sea lions was used for this conversion. Table C-10 shows mean, 95 percent UCL, and maximum values for both data sets. Both sea lion and western gull concentrations were overestimated in their respective models, though mean and 95 percent UCL estimates for gulls were very similar to the measured data.

TABLE C-9
Comparisons Between Measured and Predicted Fish Fillet Concentrations
Palos Verdes Shelf Ecological Risk Assessment

		Geometric mean		95% UCL		Maximum	
Analyte	**Species**	**Sampled**	**Modeled**	**Sampled**	**Modeled**	**Sampled**	**Modeled**
tDDT	White croaker	6.86	1.65	14.86	5.25	86.72	29.11
tDDT	Kelp bass	0.562	1.02	1.52	1.27	15.22	2.82
tPCB	White croaker	1.08	0.78	1.69	2.99	8.64	9.73
tPCB	Kelp bass	0.20	0.46	0.66	0.558	4.14	0.99

Note: Sampled data were converted from fillet (Table 2-7) to whole-body concentrations using regression equations C-15 and C-16.

TABLE C-10
Comparisons Between Measured and Predicted Western Gull and Sea Lion Concentrations
Palos Verdes Shelf Ecological Risk Assessment

		Geometric Mean		95% UCL of the Geometric Mean		Maximum	
Analyte	**Species**	**Sampled**	**Modeled WW (LW)**	**Sampled**	**Modeled WW (LW)**	**Sampled**	**Modeled WW (LW)**
DDT	Sea lion	37.67	25.52 (85.07)	836.55	49.49 (164.97)	1589.40	2820 (9398)
DDT	Western gull	7.34	7.39	14.36	13.10	19.3	814.03
PCB	Sea lion	12.29	19.26 (64.20)	[a]	31.24 (104.13)	227.81	222 (740)
PCB	Western gull	1.54	3.04	3.10	5.37	3.2	40.28

Notes:
[a] The 95 percent UCL of the geometric mean exceeded the maximum concentration in the measured data, and is therefore not considered an appropriate measure of the data.
Sea lion data represent lipid weight concentrations measured in tissue samples from carcasses collected on Santa Catalina Island in 1992-93 and 2000 (Costa et al. 1994 and Le Boeuf 2002). Western gull data represent whole-body wet weight concentrations from gulls collected on Santa Catalina Island in 1992-93 (Costa et al. 1994).
WW = wet weight
LW = lipid weight

C.4 Results

All results predicting marine bird and mammal dietary exposure levels in the SCB are presented in Sections 3 and 4 of the ERA.

Steady-state, SCB-wide surface-sediment concentrations for tPCB, tDDT, and total organic carbon (TOC) were created based on data collected between 1990 and 2003 and are presented in the ERA as Figures 2-2 and 2-3, 2-4 and 2-5, and 2-6, respectively for the PVS study area and the SCB.

Seasonal distributions of brown pelicans, double-crested cormorants, bald eagles, and sea lions from locations throughout the SCB are shown in Figures 3-4 and 3-11 of the ERA.

Dietary exposure estimates, based on predicted fish concentrations, receptor distributions, and daily intake, are depicted in Figures 3-5, 3-6, 3-8, 3-9, 3-10, and 3-12 through 3-15 of the ERA.

C.5 Discussion and Conclusions

The model appears to be a useful tool for evaluating ecological risk to fish, birds, and marine mammals that are potentially affected by sediment contaminants in the SCB. The high density of underlying sediment samples accurately depicts current conditions. The apparent small home ranges of resident fish aid in establishing predictive relationships between surficial sediment and tissue concentrations in co-located samples. The result is an ability to tie sediment contamination by tDDT and tPCB throughout the SCB to contamination in the dietary items of birds and marine mammals. Any changes in the spatial distribution and degree of contamination of surface sediments can be predicted to affect the dietary exposures of higher-order consumers of the SCB.

One area for improvement in predictive ability in the model would be the incorporation of the development of whole body-to-fillet relationships for organochlorine compounds in SCB fish species of the model. Those relationships await future analytical chemistry results.

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APPENDIX D

Summaries for Bird and Mammal Toxicity Studies

APPENDIX D

Summaries for Bird and Mammal Toxicity Studies

D.1 Summaries

Compound: Aroclor 1242
Form: not applicable
Reference: Bleavins et al. 1980
Test Species: Mink
Body weight: 1.0 kg (USEPA 1993)
food consumption: 0.137 kg/d (Bleavins and Aulerich 1981)
Study Duration: 7 months (during a critical life stage = chronic)
Endpoint: reproduction
Exposure Route: oral in diet
Dosage: four dose levels: 5, 10, 20, and 40 mg/kg; 5 mg/kg = LOAEL
Calculations:

$$LOAEL: \left(\frac{5\,mg\ Aroclor1242}{kg\ food} \times \frac{137\,g\ food}{day} \times \frac{1kg}{1000\,g} \right) / 1kg\ BW = 0.685\,mg/kg/d$$

Comments: Because all Aroclor 1242 dose levels produced total reproductive failure, and the study considered exposure over 7 months including critical life stages (reproduction), the lowest dose was considered to be a chronic LOAEL. A chronic NOAEL was estimated by multiplying the chronic LOAEL by a LOAEL-NOAEL uncertainty factor of 0.1.
Final NOAEL: 0.069 mg/kg/d
Final LOAEL: 0.69 mg/kg/d

Compound: Aroclor 1242
Reference: Britton and Huston 1973
Test Species: White leghorn hens
BW: 1 kg (assumed)
Food Consumption
Rate: 0.06 kg/day (assumed)
Endpoint: reproduction
Doses: 0, 5, 10, 20, 40, and 80 mg/kg
Duration: 2 weeks
Calculations: *NOAEL* = (10 mg/kg * 0.06 kg/d)/1 kg = 0.6 mg/kg/day
LOAEL = (20 mg/kg * 0.06 kg/d)/1 kg = 1.2 mg/kg/day

Comments: Assumed body weight of 1 kg and food consumption rate of 0.06 kg/d. NOAEL and LOAEL calculated based on egg hatchability after 12 weeks.

Compound: Aroclor 1248
Reference: McLane and Hughes 1980
Test Species: Screech owl
BW: 0.181 kg (Dunning 1993)
Food Consumption Rate: 0.025 kg/day (Pattee et al. 1998)
Endpoint: reproduction
Doses: 3 mg/kg
Duration: 2 breeding seasons
Calculations: *NOAEL* = (3 mg/kg * 0.25 kg/d)/0.181 = 0.41 mg/kg/day
Comments: No detectable effect on screech owl production.

Compound: Aroclor 1248
Reference: Barsotti et al. 1976
Test Species: Rhesus monkey
BW: 5 kg (from study)
PCB Consumption Rate: 0.43 mg/d based on 2.5 mg/kg (from study)
Endpoint: Reproduction, resorption
Doses: 0, 2.5, 5 mg/kg
Duration: 14 months
Calculations: *LOAEL* = (0.43 mg PCB/day)/5kg = 0.087 mg/kg/day
Comments: Rate of resorption was higher in monkeys fed 2.5 mg/kg than the control monkeys.

Compound: Aroclor 1248
Reference: Lowe and Stendell 1991
Test Species: American kestrel
BW: 0.13 kg (Elliott et al. 1997)
Food Consumption Rate: 0.05 kg/d (from study)
Endpoint: reproductive performance
Doses: 3 mg/kg
Duration: Through reproduction
Calculations: *NOAEL* = (1.15 mg/kg/d) * 0.1 = 0.115 mg/kg/day
LOAEL = (3 mg/kg * 0.05 kg/d)/0.13 kg BW = 1.15 mg/kg/day
Comments: Uncertainty factor of 10 applied to calculate NOAEL. Effects include shell thickness, shell length, shell breadth, shell size, shell weight, and shell thickness index.

Compound: Aroclor 1254
Reference: Platonow and Reinhart 1973
Test Species: Chicken
BW: 1.5 kg (USEPA 1988)
Food Consumption Rate: 0.106 kg/d (estimated using the allometric equation from USEPA 1988)

Endpoint: Reproduction, egg production, and female fertility
Doses: 5 mg/kg
Duration: 39 weeks
Calculations: *LOAEL* = (5 mg/kg * 0.106 kg/d)/1.5 kg BW= 0.35 mg/kg/day
Comments: Egg production and fertility were significantly different in hens fed 5 mg/kg in the last 14 weeks from the hens fed the control diet.

Compound: Aroclor 1254
Reference: Custer and Heinz 1980
Test Species: Mallard
BW: 1.25 kg (Dunning 1993)
Food Consumption Rate: 0.0169 kg/d (Nagy 1987)
Doses: 0, 25 mg/kg
Duration: Through reproduction
Calculations: *NOAEL* = 2.5 mg/kg/day * 0.1 = 0.25 mg/kg/day
LOAEL = (25 mg/kg * 0.0169 kg/d)/1.25 kg BW= 2.5 mg/kg/day
Comments: None.

Compound: Aroclor 1254
Reference: McCoy et al. 1995
Test Species: Oldfield mouse (*Permyscus poliontus*)
Body weight: 0.014 kg (from Silva and Downing 1995)
food consumption: assumed comparable to that reported by Linzy (1987) for white-footed mice (*Peromyscus leucopus*): 0.135 g food/g BW/d or 1.9 g/animal/d
Study Duration: 12 months (>1 yr and during a critical life stage = chronic)
Endpoint: reproduction
Exposure Route: oral in diet
Dosage: one dose level:
5 mg/kg = LOAEL
Calculations:

$$LOAEL: \left(\frac{5mg\ Aroclor1254}{kg\ food} \times \frac{1.9g\ food}{day} \times \frac{1kg}{1000g} \right) / 0.014kg\ BW = 0.68\ mg/kg/d$$

Comments: Aroclor 1254 at 5 mg/kg in the diet reduced the number of litters, offspring weights, and offspring survival. Because the study considered exposure over 12 months, including critical life stages (reproduction), this dose was considered to be a chronic LOAEL. A chronic NOAEL was estimated by multiplying the chronic LOAEL by a LOAEL-NOAEL uncertainty factor of 0.1.
Final NOAEL: 0.068 mg/kg/d
Final LOAEL: 0.68 mg/kg/d

Compound: Aroclor 1254
Reference: Aulerich and Ringer 1977

Test Species: Mink
BW: 1.0 kg (USEPA 1993)
Food Consumption Rate: 0.137 kg/d (Bleavins and Aulerich 1981)
Study Duration: 4.5 months (during a critical life stage = chronic)
Endpoint: reproduction
Exposure Route: oral in diet
Dosage: three dose levels: 1, 5, and 15 mg/kg; NOAEL = 1 mg/kg.
Calculations:

$$NOAEL: \left(\frac{1mg\ Aroclor1254}{kg\ food} \times \frac{137\,g\ food}{day} \times \frac{1kg}{1000\,g} \right) / 1kg\ BW = 0.137\,mg/kg/d$$

$$NOAEL: \left(\frac{5mg\ Aroclor1254}{kg\ food} \times \frac{137\,g\ food}{day} \times \frac{1kg}{1000\,g} \right) / 1kg\ BW = 0.685\,mg/kg/d$$

Comments: Because Aroclor 1254 at 5 and 15 mg/kg in the diet reduced the number of offspring born alive and the study considered exposure over 4.5 months days including critical life stages (reproduction), the 5 mg/kg dose was considered to be a chronic LOAEL and the 1 mg/kg dose was considered to be a chronic NOAEL.
Final NOAEL: 0.14 mg/kg/d
Final LOAEL: 0.69 mg/kg/d

Compound: Aroclor 1254
Reference: Heinz et al. 1980
Test Species: Ring dove
BW: 0.155 kg (Terres 1980)
Food Consumption Rate: 0.0743 kg/d (from study)
Endpoint: Neurotransmitter levels
Doses: 0, 1, 10, 100 mg/kg
Duration: Through reproduction
Calculations: *NOAEL* = (1 mg/kg * 0.0743 kg/d)/0.155 kg BW = 1 mg/kg/day
LOAEL = (10 mg/kg * 0.0743 kg/d)/0.155 kg BW = 2.5 mg/kg/day
Comments: Significant difference between dopamine and norepinephrine levels between birds on the 10-mg/kg diet and the controls. No difference was seen at the 1-mg/kg level.

Compound: Aroclor 1254
Reference: Elliott et al. 1997
Test Species: American kestrel
BW: 0.13 kg (from study)
Food Consumption Rate: N/A
Endpoint: Porphyrinogenic effects
Doses: 7 mg/kg/day (average daily dose)
Duration: 12 weeks

Calculations: *LOAEL* = 7 mg/kg/d (from study)
Comments: Only one dose tested for the chronic study.

Compound: Aroclor 1254
Reference: Hallgren et al. 2001
Test Species: Rat
BW: 0.35 kg (USEPA 1988)
Food Consumption Rate: 0.028 kg/d (estimated using the allometric equation from USEPA 1988)
Endpoint: Endocrine (thyroid)
Doses: 10 mg/kg/d
Duration: 2 weeks
Calculations: *LOAEL* = 10 mg/kg/day
Comments: None.

Compound: DDE
Reference: McLane and Hall 1972
Test Species: Screech owl
BW: 0.181 kg (Dunning 1993)
Food Consumption Rate: 0.25 kg/d (Pattee et al. 1988)
Endpoint: reproduction (eggshell thinning)
Doses: 2.8 mg/kg
Duration: 2 breeding seasons
Calculations: *NOAEL* = (0.39 mg/kg/d) * 0.1 = 0.039 mg/kg/d
LOAEL = (2.8 mg/kg * 0.25 kg/d)/0.181 kg = 0.39 mg/kg/d
Comments: 10 mg/kg dw DDE in diet caused 13.3 percent decrease in shell thickness. Uncertainty factor of 10 applied to calculate NOAEL.

Compound: DDE
Reference: Mendenhall et al. 1983
Test Species: Barn owl
BW: 0.466 kg (Johnsgard 1988)
Food Consumption Rate: 0.0625 kg/d (Johnsgard 1988)
Endpoint: reproduction
Doses: 3.0 mg/kg DDE
Duration: 2 years
Calculations: *NOAEL* = (0.4 mg/kg/d) * 0.1 = 0.04 mg/kg/d
LOAEL = (3.0 mg/kg * 0.0625 kg/d)/0.466 kg = 0.4 mg/kg/d
Comments: Uncertainty factor of 10 applied to calculate NOAEL. Severe eggshell thinning from DDE exposure; breeding success impaired.

Compound: DDT (technical)
Reference: Ottoboni et al. 1977

Test Species: Beagle dog
BW: N/A
Food Consumption Rate: N/A
Endpoint: reproduction
Doses: 0, 1, 5, 10, mg/kg/day
Duration: Multiple generations
Calculations: *NOAEL* = 10 mg/kg/day
Comments: No statistically significant differences among control and DDT-treated dogs in any of the reproductive variables, with the exception of age at puberty of the females.

Compound: DDT
Reference: Bernard and Gaertner 1964
Test Species: Mouse
BW: 0.03 kg (USEPA 1993)
Food Consumption Rate: 0.0055 kg/d (estimated using the allometric equation from USEPA 1988)
Endpoint: Reproduction, fertility
Doses: 100, 200, 300, 600 mg/kg DDT
Duration: 60 days
Calculations: *LOAEL* = (300 mg/kg * 0.0055 kg/day)/0.03 kg BW = 55 mg/kg/day
Comments: Female fertility was reduced in mice fed 300-mg/kg diets.

Compound: DDT
Reference: Heath et al. 1969
Test Species: Mallard
BW: 1 kg (Heinz et al. 1989)
Food Consumption Rate: 0.1 kg/d (Heinz et al. 1989)
Endpoint: Reproduction
Doses: 0, 2.5, 10, 25, 40 mg/kg
Duration: Through reproduction
Calculations: *NOAEL* = (10 mg/kg * 0.1 kg/d)/1 kg BW = 1 mg/kg/day
LOAEL = (25 mg/kg * 0.1 kg/d)/1 kg BW = 2.5 mg/kg/day
Comments: Reduced duckling survival occurred at 25 mg/kg as well as significant thinning of eggshells. No thinning or reduced survival was observed at 10 mg/kg.

Compound: DDT
Reference: Del Pup et al. 1978
Test Species: Mouse
BW: 0.03 kg (USEPA 1993)
Food Consumption Rate: 0.0055 kg/d (estimated using the allometric equation from USEPA 1988)

Endpoint: Reproduction, neonate survival
Doses: 0, 100 mg/kg DDT
Duration: Two generations
Calculations: *LOAEL* = (100 mg/kg * 0.0055 kg/day)/0.03 kg BW = 18.3 mg/kg/day
Comments: Neonate survival was reduced at 100 mg/kg after 20 weeks when compared to controls.

Compound: DDT
Reference: Laug et al. 1950
Test Species: Rat
BW: 0.35 kg (USEPA 1988)
Food Consumption Rate: 0.028 kg/d (estimated using the allometric equation from USEPA, 1988)
Endpoint: Hepatic cell alterations
Doses: 0, 1, 5, 10, 50 mg/kg DDT
Duration: 15-27 weeks
Calculations: *LOAEL* = (5 mg/kg * 0.0055 kg/day)/0.03 kg BW= 0.4 mg/kg/day
Comments: Hepatic cell alterations were noticed at 5 mg/kg, but not at 1 mg/kg.

Compound: 2,2′-DDT
Reference: Wren et al. 1971
Test Species: Rat
BW: 0.35 kg (USEPA 1993)
Food Consumption Rate: 0.028 kg/d (estimated using the allometric equation from USEPA 1988)
Endpoint:
Doses: 0, 1, 10, 20, and 40 mg/kg 2,2′-DDT
Duration: Two generations
Calculations: *LOAEL* = (40 mg/kg * 0.028 kg/day)/0.35 kg BW = 3.2 mg/kg/day
Comments: None.

Compound: 4,4′-DDT
Reference: Davison and Sell 1974
Test Species: Mallard
BW: 0.94 kg (final body weight from study)
Food Consumption Rate: 0.08 kg/d (from study)
Endpoint: Reproduction, 20 percent eggshell thinning
Doses: 0, 2, 20, 40, 200 mg/kg
Duration: Through reproduction
Calculations: *NOAEL* = (20 mg/kg * 0.08 kg/d)/0.94 kg BW = 1.7 mg/kg/day
LOAEL = (40 mg/kg * 0.08 kg/d)/0.94 kg BW = 3.4 mg/kg/day
Comments: 20 percent eggshell thinning occurred at 40 mg/kg.

D.2 References

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