ASSESSMENT OF BIOACCUMULATION IN SAN DIEGO BAY

Draft Project Report

By

Steven M. Bay, Darrin J. Greenstein and Ashley N. Parks

Southern California Coastal Water Research Project

And

Catherine Q. T. Zeeman

U.S. Fish and Wildlife Service, Carlsbad Fish and Wildlife Office

July 15, 2016

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Acknowledgements

The authors would like to thank SCCWRP staff members Doris Vidal-Dorsch, Nathan Dodder, Dario Diehl, Patricia Gonzalez, and Justin Stuart for assistance with fish collection, sample processing, and data management. Thanks to Dr. Rebecca Lewison of San Diego State University for coordinating the collection of seabird eggs, and Dr. Jason Freshwater, USFWS volunteer, for many hours of assistance with processing egg samples. We also wish to thank the many participants in the Bight'13 Regional Monitoring Program for the collection and analysis of sediment and tissue samples. Special thanks to Chris Stransky and other staff at AMEC Foster Wheeler for sampling support in San Diego Bay, including extraordinary efforts to obtain benthic infauna samples and coordination of the fishing derby. Many thanks to the nearly 100 volunteer fishing derby participants that donated fish samples for this study. Additional support for sample collection and analysis was provided by the City of San Diego and Port of San Diego. Tissue and sediment chemistry analyses were conducted by Physis Environmental Labs and the City of San Diego Public Utilities Department.

This study was funded by agreements 13-075-190 and 14-032-190 with the State Water Resources Control Board (SWRCB), in coordination with the San Diego Regional Water Quality Control Board.

Disclaimer

Executive Summary

Sediment-borne bioaccumulative contaminants have the potential to impair most of the designated beneficial uses for San Diego Bay. However, the ability to assess bioaccumulation-related impacts within the Bay is constrained by limited data availability and uncertainty in the approach to use for assessment and cleanup projects. Tissue contamination data for key elements of the San Diego Bay food web, matched with sediment data, are needed to develop an improved understanding of bioaccumulation relationships within the Bay and to provide updated information needed to assess the impacts of sediment contamination on wildlife and human health.

In 2012, the California Regional Water Quality Control Board, San Diego Region (SDRWQCB), the Southern California Coastal Water Research Project (SCCWRP), and the United States Fish and Wildlife Service (USFWS), Carlsbad Fish and Wildlife Office (CFWO) developed a plan to address the bioaccumulation data needs for San Diego Bay. Funding to conduct these studies was received from the Water Board in May 2014 and January 2015. The overall goal of this project was to conduct integrated food web studies within three regions of San Diego Bay. Compilation and analysis of data from multiple studies, as well as additional sampling conducted under this project were used to accomplish three primary study objectives:

- **Describe bioaccumulation among key components of the San Diego Bay food web.** Two major contaminant exposure pathways were evaluated in the study: bioaccumulation related to feeding on sediment-dwelling organisms (benthic pathway) and bioaccumulation related to uptake of contaminants in water column-dwelling organisms (pelagic pathway).

- **Evaluate risk to wildlife from contaminant exposure.** Contaminant concentrations in the eggs and diet of five species of seabirds were examined: California least tern, Caspian tern, double-crested cormorant, western gull, and surf scoter (diet only).

- **Assess potential risk to human health resulting from consumption of San Diego Bay fish.** Tissue contamination data for several popular sport fish, including spotted sand bass, California halibut, and pacific chub mackerel, were compared to consumption advisory levels developed by OEHHA (Office of Environmental Health Hazard Assessment).

Sediment and tissue samples were obtained from three coordinated studies. Sampling was conducted in 2013 as part of the Southern California Bight Regional Monitoring Program (Bight '13) in coordination with the Regional Harbor Monitoring Program (RHMP). The Bight '13 survey also included the collection of bird eggs from five locations around the Bay. Additional sediment and tissue samples were collected in 2014 as part of a shallow water habitat survey designed to complement the 2013 sample collections. The SWHB samples were collected from water depths of 3 m or less in order to provide information on contamination and bioaccumulation patterns in areas frequently used as bird foraging areas and fish nursery grounds. Samples of five species of sport fish were collected for the study through a combination of targeted fishing and contributions from the public during a novel fishing derby.

The analyses were based on sediment contamination data from 64 stations in the Bay, selected to represent three geographical regions: North, Central, and South. Additional sampling for biota was conducted at a subset of these stations in order to obtain samples of key food web components: plankton, benthic infauna, and forage fish. A total of 209 tissue samples were analyzed for a suite of contaminants that included mercury, PCBs, DDTs, chlordanes, dieldrin, and contaminants of emerging concern (PBDE flame retardants and perfluorinated compounds).

The key findings from the study are summarized below by study objective:

Bioaccumulation among Food Web Components

- Biomagnification among food web components was evident for all major contaminant types evaluated: PCBs, DDTs, PBDEs, chlordanes, and mercury. Similar patterns among food web components were evident for most contaminant types, with the lowest concentrations occurring in the lowest trophic levels of plankton and benthic infauna (crustaceans, mollusks, polychaetes).

- The greatest bioaccumulation potential from sediment was observed for PCBs and DDTs, where all food web components had median concentrations above bay-wide sediment means.

- Median tissue mercury concentrations were below sediment levels for all trophic levels, likely reflecting a relatively low influence of local sediment mercury on tissue levels.

- Sediments in the North region of the Bay contained higher average concentrations of chlordanes, mercury, and PCBs. The concentrations of sediment DDTs were similar in the North and South, which were 2-3x higher than the concentration in the Central Bay.

- Tissue contamination in infauna appeared to follow trends in sediment concentration for PCBs and DDTs, with higher tissue PCB concentrations in the North and higher DDTs in the North and South.

- Median total PCB concentrations in fish tissue were generally highest in the Central Bay, and lowest concentrations in the South. This pattern differed from the trend seen for sediment, where the highest concentration of PCBs was measured in the North.

- The highest median PCB concentration (419 ng/g) was measured in two samples of round stingray from the South.

- Total DDT concentrations in fish were generally about ten-fold lower than PCBs. Median concentrations of DDTs were generally similar in the North and Central, which were approximately two-fold higher than the South.

- Most fish samples did not contain detectable levels of mercury, and there was little variation in concentration among species or region. The highest median concentrations were measured in round stingray (0.15 ng/g South) and spotted sand bass (0.11 ng/g Central).

- Bird eggs contained similar concentrations of DDTs and PCBs. Caspian tern eggs contained the highest median concentration of most contaminant types.

- There was little difference between regions in contaminant concentrations in California least tern eggs.

Risk to Wildlife

- Elevated concentrations of mercury in bird diets and eggs warrant further study, but the likelihood of observing measureable adverse impacts is low. Risk to adults via the diet is somewhat greater than potential impacts on embryos from egg contamination.

- Total DDT concentrations in eggs exceeds thresholds for adverse impacts on embryos of sensitive species for eggshell thinning and reduced nest productivity. Risk to adults from dietary exposure to DDTs is less. Waterbirds have intermediate sensitivity to DDTs, so the chance of detecting measurable impacts is low.

- Total PCB concentrations in eggs indicate greater potential risk to embryos of sensitive species, relative to adults (from dietary exposure). Further monitoring is warranted, but there is a low likelihood of observing measureable effects in waterbirds.

- Risk from exposure to PBDEs, chordanes, and PFCs was less than the other contaminants evaluated, and below levels potential concen.

- Some risk of adverse effects from exposure to PAHs was indicated for birds that forage on benthic invertebrates.

Risk to Human Health

- Potential human health risk due to seafood consumption was evaluated for five species of locally-caught sport fish: California halibut, pacific chub mackerel, round stingray, spotted sand bass, and topsmelt.

- PCBs are the dominant trace organic contaminant of sport fish in the Bay. DDTs, while still prevalent in fish, are usually present at much lower concentrations.

- Pacific chub mackerel and spotted sand bass tended to have the highest concentrations among the species analyzed.

- The greatest potential risk to human health appears to be associated with PCBs in fish tissue, followed closely by mercury. For both contaminants, tissue concentrations in multiple species of fish exceeded ATLs (Advisory Tissue Levels), and were within the range where consumption of no more than one meal per week is recommended by OEHHA.

- Comparison with historical data suggests that sport fish contamination levels for PCBs and DDTs have declined two to five fold over the last 15 years (depending on species). Tissue mercury concentrations have changed little over the same period. Significant human health risk remains for consumers of Bay seafood, despite these recent declines in contamination.

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Introduction

Sediment-borne bioaccumulative contaminants have the potential to impair most of the designated beneficial uses for San Diego Bay. However, the ability to assess bioaccumulation-related impacts within the Bay is constrained by limited data availability and uncertainty in the approach to use for assessment and cleanup projects. The data limitations include the lack of adequate numbers of matched sediment and tissue contamination data for important receptors (e.g., sport fish and wildlife) and other components of the food web. Such data are needed to develop the understanding of bioaccumulation relationships for interpretation of site specific data and to support development of site assessment tools to evaluate conditions with respect to sediment quality objectives (SQOs) for protection of human health and wildlife.

In 2012, the California Regional Water Quality Control Board, San Diego Region (SDRWQCB), the Southern California Coastal Water Research Project (SCCWRP), and the United States Fish and Wildlife Service (USFWS), Carlsbad Fish and Wildlife Office (CFWO) developed an integrated plan for studies to help improve understanding of contaminant transfer through San Diego Bay food webs and determine the risk to wildlife and humans from consuming contaminated fish in the bay. Funding to conduct these studies was received from the Water Board in May 2014 (Agreement 13-075-190) and January 2015 (Agreement 14-032-190). The overall goal of this project was to conduct integrated food web studies within three regions of San Diego Bay. Coordination with the 2013 Bight Regional Monitoring Survey, Regional Harbor Monitoring Program, and other studies were used to obtain bioaccumulation data for key food web components, which was supplemented by additional sampling and analyses under this project. Analyses of the data will result in an updated evaluation of sediment contamination risks to wildlife and humans for the bay, as well as providing an improved understanding of the pathways of contaminant bioaccumulation through the food web and relationship with sediment contamination.

This report provides a summary of results for three major components of the study. The first results section describes contaminant concentrations and bioaccumulation factors among key elements of the food web in San Diego Bay (Figure 1). Two major contaminant exposure pathways were evaluated: bioaccumulation related to feeding on sediment-dwelling organisms (benthic pathway) and bioaccumulation related to uptake of contaminants in water column-dwelling organisms (pelagic pathway). The second results section provides a detailed evaluation of contamination in bird eggs and diets and associated risk. Five species of birds were examined: California least tern, Caspian tern, double-crested cormorant, surf scoter and western gull. The final results section presents an evaluation of the potential human health risk from consuming sport fish from San Diego Bay. Tissue contamination data for several popular sport fish, including spotted sand bass, California halibut, and pacific chum mackerel, were compared to consumption advisory levels developed by California's Office of Environmental Health Hazard Assessment (OEHHA).

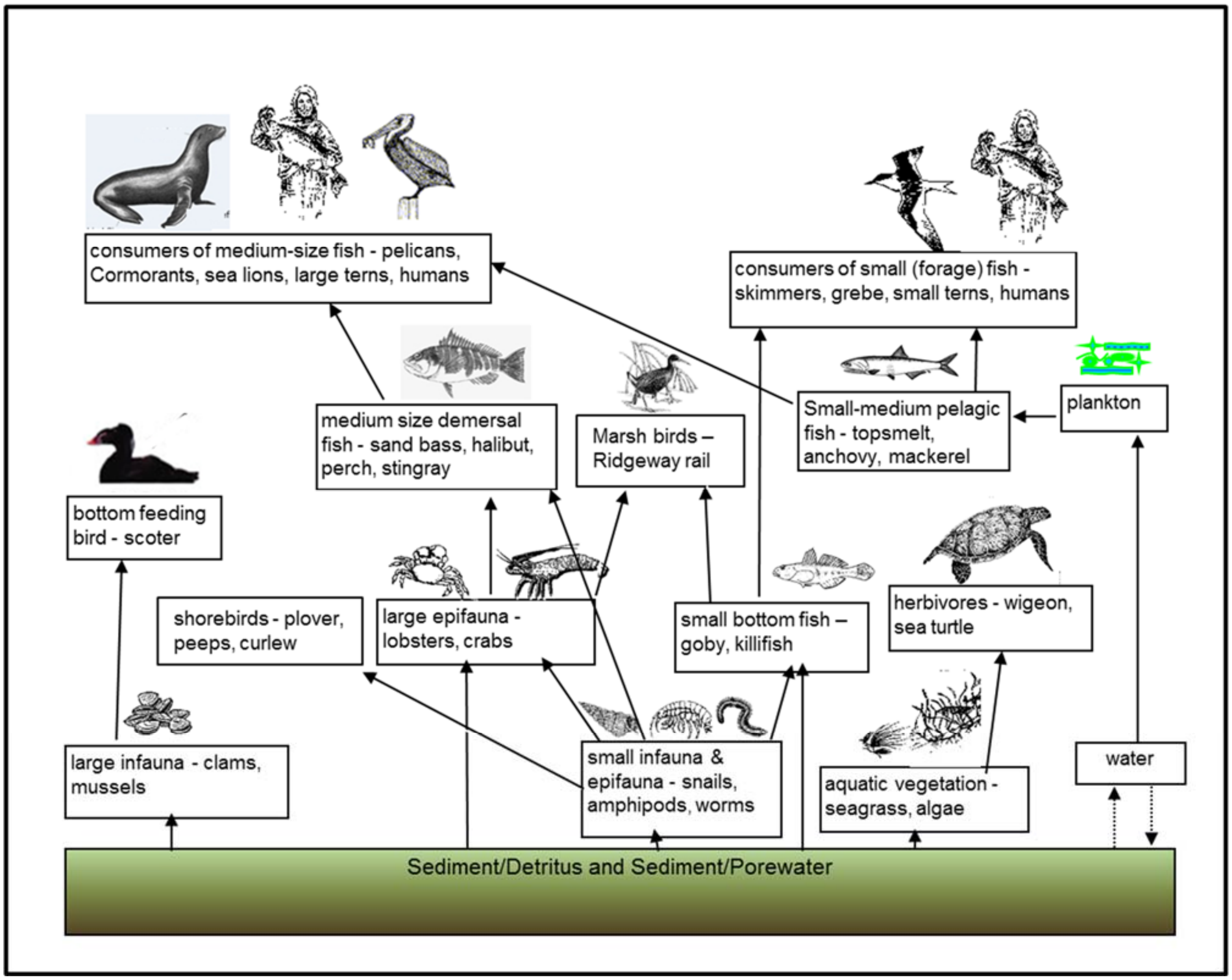

Figure 1. Conceptual food web for San Diego Bay.

Methods

Sample collection and processing

Food web components

Sediment and tissue samples were obtained from three coordinated studies conducted in 2013 and 2014. Sampling was conducted in 2013 as part of the Southern California Bight Regional Monitoring Program (Bight '13) in coordination with the Regional Harbor Monitoring Program (RHMP). The 2013 sampling was based on a stratified random design and included ports, estuaries, marinas, and other bay areas in depths of 3 meters or greater (Figures 2 and 3). The Bight '13 survey also included the collection bird eggs from five locations the Bay (Figure 3), as well as from eleven additional sites ranging from the Channel Islands to Tijuana Estuary. A detailed risk evaluation of the bird egg contamination data is presented in this report for the samples from San Diego Bay. Evaluation of egg contamination trends and risk for the entire Bight ’13 data set was conducted in a separate effort and is included in Appendix H of this report to provide context.

Sampling in 2014 was conducted as part of a shallow water habitat survey (SWHB) designed to complement the 2013 sample collections. The SWHB samples were collected from water depths of 3 m or less in order to provide information on contamination and bioaccumulation patterns in areas frequently used as bird foraging areas and fish nursery grounds. Station locations for the SWHB survey were also based on a randomized design, with the stations allocated to three strata that represented the north, central, and south regions of the Bay (Figures 2 and 3).

Eighty sediment stations were sampled during the surveys; with 50 stations sampled as part of Bight ’13/RHMP and 30 stations sampled for the SWHB survey (Tables 1 and 2). A subset of 64 randomly located sediment stations was used in this study to represent sediment contamination in the Bay. The location of tissue sampling stations was selected in order to provide at least three stations in each of the three Bay regions. In most cases, tissue samples were collected from a subset of the sediment collection locations. Sediment samples were collected using a Van Veen grab. Surface sediment from the upper 5 cm was removed from the grab and allocated to individual jars for chemical analysis. Multiple additional grabs were collected from some locations to obtain samples of benthic infauna for chemical analysis; the contents of these grabs were screened onboard the boat and sorted into separate jars by major taxon (polychaetes, mollusks, crustaceans). Fish trawls and/or seines were used at some stations to collect small and medium-sized fish for chemical analysis. The whole fish samples were wrapped in clean aluminum foil, placed in food-grade polyethylene bags, and stored on ice. All sample containers were labeled with project name, sample identification number, site location, date and time, and frozen within 24 hours of collection.

A total of 209 tissue samples, representing major components of benthic and pelagic food webs in San Diego Bay were collected and analyzed for contaminants (Figure 1, Table 3). Eggs from four bird species were analyzed, consisting of 44 samples. California least tern eggs were collected from four sites, while eggs from western gulls, Caspian terns, and double crested cormorants were obtained from single locations. Tissues from five species of sport fish, representing 23 composite or individual fish samples, were analyzed to assess potential human health risk. A total of 87 forage fish tissue samples, representing 13 species, were analyzed. Fish species included small surface feeders (e.g., anchovy, topsmelt) and medium-sized fish having a diet that included benthic organisms (e.g., black perch, round stingray, barred sand bass, spotted sand bass, and California halibut). Samples of benthic invertebrates and plankton from each of the three Bay regions were also collected in 2013 and 2014.

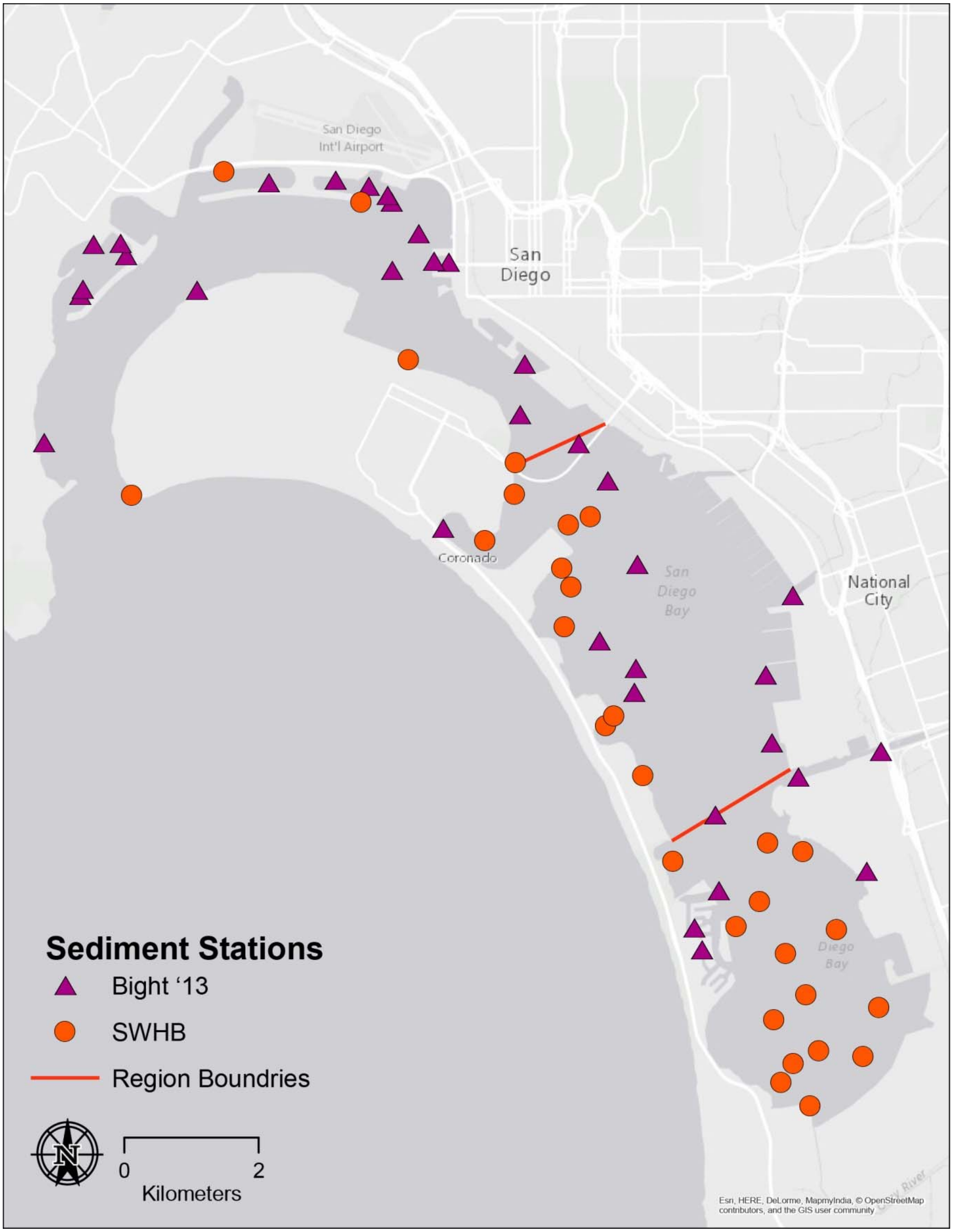

Figure 2. Sediment sample locations for Bight '13 regional survey and regional harbor monitoring program (Bight '13) and shallow water habitat survey (SWHB).

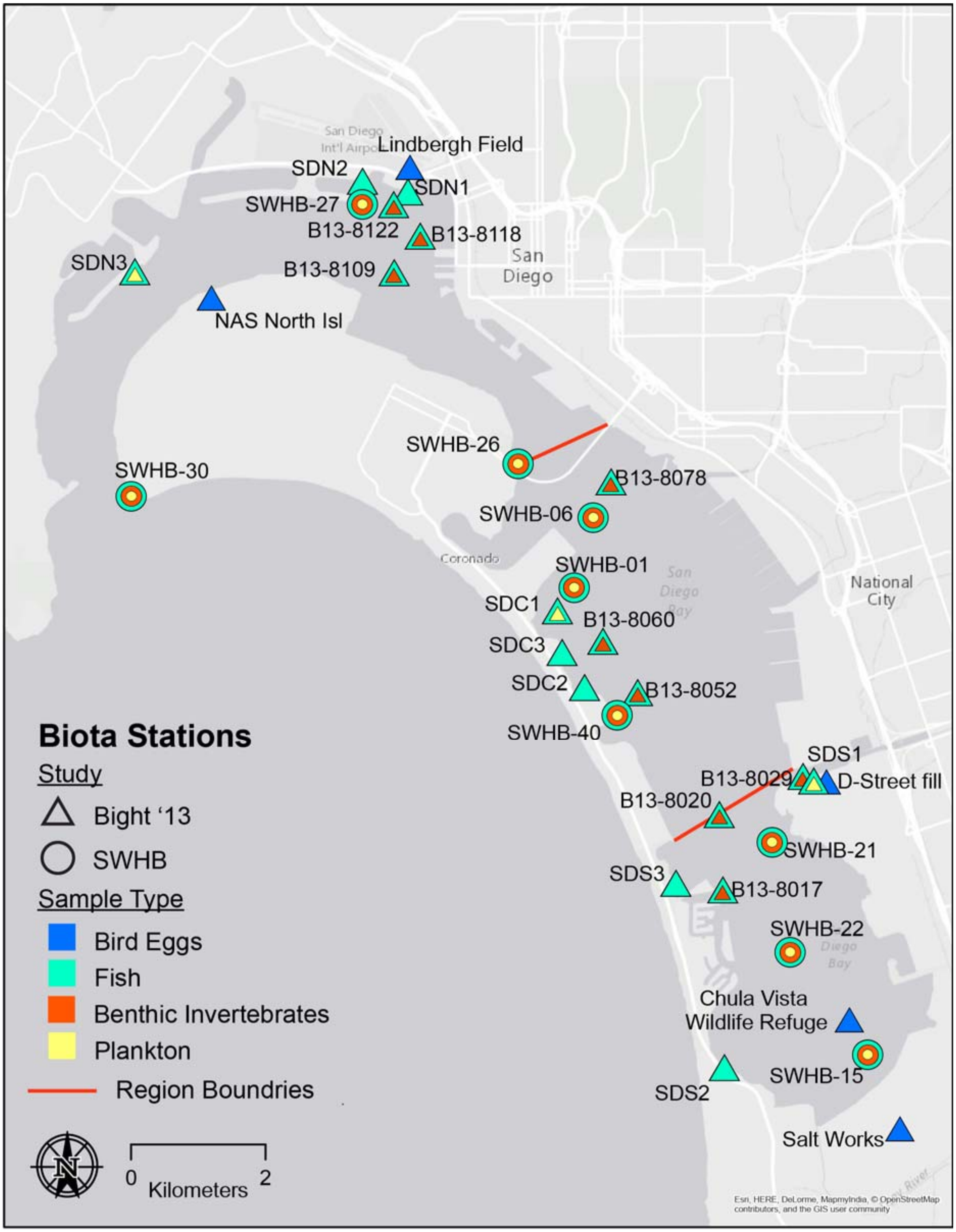

Figure 3. Tissue sample locations for Bight '13 regional survey and regional harbor monitoring program (Bight '13) and shallow water habitat survey (SWHB).

Table 1. Bight'13 stations and sample types used for analysis of bioaccumulation relationships. Sample types: S = sediment, F = fish, I = benthic invertebrate, P = plankton, B = bird egg.

Study	Location	Station	Latitude	Longitude	Type
Bight13	San Diego Bay North	B13-8085	32.691687	-117.238244	S
Bight13	San Diego Bay North	B13-8087	32.691721	-117.153217	S
Bight13	San Diego Bay North	B13-8093	32.695601	-117.162557	S
Bight13	San Diego Bay North	B13-8100	32.7024	-117.16178	S
Bight13	San Diego Bay North	NAS North Isl	32.7114626	-117.211759	B
Bight13	San Diego Bay North	B13-8102	32.711543	-117.232552	S
Bight13	San Diego Bay North	B13-8105	32.712275	-117.213967	S
Bight13	San Diego Bay North	B13-8106	32.712329	-117.232133	S
Bight13	San Diego Bay North	B13-8109	32.714963	-117.182907	SFI
Bight13	San Diego Bay North	SDN3	32.71505	-117.22385	FP
Bight13	San Diego Bay North	B13-8111	32.716092	-117.173953	S
Bight13	San Diego Bay North	B13-8112	32.71619	-117.176237	S
Bight13	San Diego Bay North	B13-8113	32.716887	-117.225212	S
Bight13	San Diego Bay North	B13-8116	32.718402	-117.2304	S
Bight13	San Diego Bay North	B13-8117	32.718569	-117.226112	S
Bight13	San Diego Bay North	B13-8118	32.719885	-117.178736	SFI
Bight13	San Diego Bay North	B13-8122	32.724148	-117.182983	SFI
Bight13	San Diego Bay North	B13-8123	32.725018	-117.183684	S
Bight13	San Diego Bay North	SDN1	32.72585	-117.1807	F
Bight13	San Diego Bay North	B13-8124	32.726301	-117.186644	S
Bight13	San Diego Bay North	B13-8127	32.726737	-117.202524	S
Bight13	San Diego Bay North	B13-8128	32.727123	-117.191922	S
Bight13	San Diego Bay North	SDN2	32.7272833	-117.1879334	F
Bight13	San Diego Bay North	Lindbergh Field	32.7291875	-117.1803969	B
Bight13	San Diego Bay Central	B13-8045	32.65155	-117.122464	S
Bight13	San Diego Bay Central	B13-8052	32.65828	-117.14434	SFI
Bight13	San Diego Bay Central	SDC2	32.65895	-117.1527333	F
Bight13	San Diego Bay Central	B13-8056	32.660613	-117.12339	S
Bight13	San Diego Bay Central	B13-8058	32.661471	-117.144097	S
Bight13	San Diego Bay Central	SDC3	32.6636667	-117.1562667	F
Bight13	San Diego Bay Central	B13-8060	32.665184	-117.149804	SFI
Bight13	San Diego Bay Central	SDC1	32.6693	-117.1569668	FP
Bight13	San Diego Bay Central	B13-8065	32.671353	-117.119134	S
Bight13	San Diego Bay Central	B13-8068	32.675472	-117.143841	S
Bight13	San Diego Bay Central	B13-8073	32.680331	-117.174759	S
Bight13	San Diego Bay Central	B13-8078	32.686723	-117.148594	SFI

Table 1. Continued.

Study	Location	Station	Latitude	Longitude	Type
Bight13	San Diego Bay South	Salt Works	32.59924	-117.102809	B
Bight13	San Diego Bay South	SDS2	32.60755	-117.1305334	F
Bight13	San Diego Bay South	Chula Vista Wildlife Refuge	32.61403	-117.11086	B
Bight13	San Diego Bay South	B13-8013	32.623601	-117.13346	S
Bight13	San Diego Bay South	B13-8014	32.626539	-117.134678	S
Bight13	San Diego Bay South	B13-8017	32.631569	-117.13084	SFI
Bight13	San Diego Bay South	SDS3	32.63298	-117.13888	F
Bight13	San Diego Bay South	B13-8018	32.63417	-117.10733	S
Bight13	San Diego Bay South	B13-8020	32.641792	-117.131413	SFI
Bight13	San Diego Bay South	D-Street fill	32.64619	-117.11455	B
Bight13	San Diego Bay South	SDS1	32.6462667	-117.1165168	FP
Bight13	San Diego Bay South	B13-8029	32.646936	-117.118238	SFI
Bight13	Sweetwater Marsh	B13-8043	32.65037	-117.105093	S

Bird eggs

Eggs were collected during the 2013 nesting season, as part of the Bight 2013 regional monitoring program. Eggs were collected during routine surveys by colony monitors with appropriate permits. All of the eggs collected were failed to hatch, which was evident by nest abandonment, crushing, or not having hatched after sufficient time for incubation. Eggs were placed in cartons and transported to the CFWO lab for processing using standard protocols adopted for studies on the impacts of PCBs in the Hudson River (Hudson River Natural Resources Trustees 2002).

At CFWO, eggs were gently cleaned with distilled water, weighed, and breadth and width were measured to the nearest 0.1 mm using a dial caliper. For those eggs that were not cracked, the volume was measured as the weight of water displaced by the egg. For those that were cracked, volume was estimated using the generic approach by Hoyt (1979) combined with species-specific data collected for this study. A scalpel, pre-rinsed with dilute nitric acid followed by distilled water, reagent grade acetone and reagent grade hexane was used to cut the shell around the equator. Contents were viewed for evidence of embryo development and malpositioning (if a chick was present) before transfer to a chemically pre-cleaned I-Chem jar.

If an embryo was present, it was further evaluated for evidence of conspicuous malformations (e.g., missing limbs, malformed beaks). To minimize potential loss of sample and cross-contamination that could result from physical manipulation, embryos were not evaluated for subtle malformations that required measurements. Once observations were recorded, egg contents were placed in a freezer for storage until they were transferred to the Southern California Coastal Water Research Project to be logged in with other Bight '13 samples, and later submitted for chemical analysis to a laboratory designated by SCCWRP. Eggs of the Caspian tern, western gull and double-crested cormorant were large enough for each egg to be an individual sample (i.e., 1 egg / sample). Because of their small size, least tern eggs were composited (2 or more eggs/sample) to produce enough material for chemical analysis. A total of 44 samples were prepared for chemical analyses (Table 3).

Eggshells were placed in cartons to dry at room temperature for a minimum of 30 days. Once dry, the thickness of each eggshell (shell + shell membrane) was measured at four points around the girth with a Starrett electronic digital micrometer fitted with a ball attachment, and with a Starrett Model 1010M dial micrometer, both accurate to 0.01 mm, and estimatable to 0.001 mm. Two micrometers were used because dial micrometer performance is affected by shell curvature (a problem with small eggs; readings may be high), while the ball of the digital micrometer may slightly dent membranes (readings may be low). The Starrett model M1010 micrometer is commonly used, so that results obtained with this micrometer were used preferentially for identifying trends in shell thickness and particularly for comparisons with results from other studies. Dried shells were then weighed, and placed in WhirlPac bags for storage at CFWO. Two different measures of shell thickness were recorded for each egg: one was thickness, as measured (in mm) with micrometers, the other was the Ratcliffe's Index (RI), as described by Burnham et al. (1984). The RI is not a direct measure of thickness, but it is estimate of eggshell density (which is affected by thickness) that is free of potential shortcomings of micrometer readings. The RI combined with direct measurements offer a weight of evidence approach to assessing eggshell thinning. The RI for each egg was computed as:

RI = dry weight of the shell (mg) / [shell length (mm) x shell breadth (mm)].

Data on egg status and measurements were provided to SCCWRP, and subsequently entered into the Bight '13 database. A total of 76 individual eggs were evaluated for condition, morphometrics, and two measures of eggshell thickness.

Table 2. Shallow water habitat survey stations and sample types used for analysis of bioaccumulation relationships. Sample types: S = sediment, F = fish, I = benthic invertebrate, P = plankton.

Study	Location	Station	Latitude	Longitude	Type
SWHB	San Diego Bay North	SWHB-26	32.68911	-117.16324	SFIP
SWHB	San Diego Bay North	SWHB-27	32.72411	-117.18791	SFIP
SWHB	San Diego Bay North	SWHB-28	32.70289	-117.18027	S
SWHB	San Diego Bay North	SWHB-30	32.68464	-117.2243	SFIP
SWHB	San Diego Bay North	SWHB-53	32.72818	-117.20972	S
SWHB	San Diego Bay Central	SWHB-01	32.6724	-117.15436	SFIP
SWHB	San Diego Bay Central	SWHB-02	32.67494	-117.15588	S
SWHB	San Diego Bay Central	SWHB-06	32.68185	-117.15135	SFIP
SWHB	San Diego Bay Central	SWHB-07	32.64702	-117.14289	S
SWHB	San Diego Bay Central	SWHB-08	32.65375	-117.14886	S
SWHB	San Diego Bay Central	SWHB-09	32.68077	-117.15484	S
SWHB	San Diego Bay Central	SWHB-10	32.68487	-117.16341	S
SWHB	San Diego Bay Central	SWHB-33	32.66704	-117.15545	S
SWHB	San Diego Bay Central	SWHB-36	32.67863	-117.16811	S
SWHB	San Diego Bay Central	SWHB-40	32.65508	-117.14755	SFIP
SWHB	San Diego Bay South	SWHB-11	32.60259	-117.11629	S
SWHB	San Diego Bay South	SWHB-12	32.61583	-117.10535	S
SWHB	San Diego Bay South	SWHB-13	32.63547	-117.13809	S
SWHB	San Diego Bay South	SWHB-14	32.61416	-117.12204	S
SWHB	San Diego Bay South	SWHB-15	32.60923	-117.10791	SFIP
SWHB	San Diego Bay South	SWHB-16	32.6175	-117.11693	S
SWHB	San Diego Bay South	SWHB-18	32.60573	-117.12089	S
SWHB	San Diego Bay South	SWHB-19	32.60828	-117.11898	S
SWHB	San Diego Bay South	SWHB-20	32.62629	-117.11212	S
SWHB	San Diego Bay South	SWHB-21	32.63798	-117.12307	SFIP
SWHB	San Diego Bay South	SWHB-22	32.6231	-117.12018	SFIP
SWHB	San Diego Bay South	SWHB-23	32.61	-117.11491	S
SWHB	San Diego Bay South	SWHB-24	32.63681	-117.11744	S
SWHB	San Diego Bay South	SWHB-25	32.63007	-117.12437	S
SWHB	San Diego Bay South	SWHB-41	32.62669	-117.12809	S

Table 2. Continued.

StudyID	Location	Station	Latitude	Longitude	Type
SWHB	North Bay	SWHB-26	32.68911	-117.16324	SFIP
SWHB	North Bay	SWHB-27	32.72411	-117.18791	SFIP
SWHB	North Bay	SWHB-28	32.70289	-117.18027	S
SWHB	North Bay	SWHB-30	32.68464	-117.2243	SFIP
SWHB	North Bay	SWHB-53	32.72818	-117.20972	S
SWHB	Central Bay	SWHB-01	32.6724	-117.15436	SFIP
SWHB	Central Bay	SWHB-02	32.67494	-117.15588	S
SWHB	Central Bay	SWHB-06	32.68185	-117.15135	SFIP
SWHB	Central Bay	SWHB-07	32.64702	-117.14289	S
SWHB	Central Bay	SWHB-08	32.65375	-117.14886	S
SWHB	Central Bay	SWHB-09	32.68077	-117.15484	S
SWHB	Central Bay	SWHB-10	32.68487	-117.16341	S
SWHB	Central Bay	SWHB-33	32.66704	-117.15545	S
SWHB	Central Bay	SWHB-36	32.67863	-117.16811	S
SWHB	Central Bay	SWHB-40	32.65508	-117.14755	SFIP
SWHB	South Bay	SWHB-11	32.60259	-117.11629	S
SWHB	South Bay	SWHB-12	32.61583	-117.10535	S
SWHB	South Bay	SWHB-13	32.63547	-117.13809	S
SWHB	South Bay	SWHB-14	32.61416	-117.12204	S
SWHB	South Bay	SWHB-15	32.60923	-117.10791	SFIP
SWHB	South Bay	SWHB-16	32.6175	-117.11693	S
SWHB	South Bay	SWHB-18	32.60573	-117.12089	S
SWHB	South Bay	SWHB-19	32.60828	-117.11898	S
SWHB	South Bay	SWHB-20	32.62629	-117.11212	S
SWHB	South Bay	SWHB-21	32.63798	-117.12307	SFIP
SWHB	South Bay	SWHB-22	32.6231	-117.12018	SFIP
SWHB	South Bay	SWHB-23	32.61	-117.11491	S
SWHB	South Bay	SWHB-24	32.63681	-117.11744	S
SWHB	South Bay	SWHB-25	32.63007	-117.12437	S
SWHB	South Bay	SWHB-41	32.62669	-117.12809	S

Table 3. Number of tissue and sediment samples analyzed for contamination.

Sample Group	Common Name	Count
Bird	California least tern	18
Bird	Caspian tern	10
Bird	Double-crested cormorant	8
Bird	Western gull	8
	Bird Total	**44**
Sport fish	California halibut	8
Sport fish	Pacific chub mackerel	3
Sport fish	Round stingray	2
Sport fish	Spotted sand bass	9
Sport fish	Topsmelt	1
	Sport Fish Total	**23**
Forage fish	Arrow goby	1
Forage fish	Barred sand bass	9
Forage fish	Black perch	2
Forage fish	California halibut	20
Forage fish	California killifish	2
Forage fish	Deepbody anchovy	11
Forage fish	Goby sp.	3
Forage fish	Northern anchovy	2
Forage fish	Round stingray	1
Forage fish	Shiner perch	6
Forage fish	Slough anchovy	10
Forage fish	Spotted sand bass	11
Forage fish	Topsmelt	9
	Forage Fish Total	**87**
Crustacea	Crustacea	13
Mollusks	Mollusks	11
Polychaetes	Polychaetes	18
	Benthic Infauna Total	**42**
Plankton	Plankton	13
	Tissue Grand Total	**209**
Sediment	Sediment	64

Sport fish

Sport fish were collected from several locations San Diego Bay in 2014 and 2015 (Figure 4). Fish were collected by rig fishing during two types of sampling events. Samples obtained in 2014 were collected by SCCWRP from a small boat that targeted specific regions of the Bay. Sport fish were collected in 2015 from a fishing derby that involved multiple volunteers from a variety of locations in the north, central, and south regions of the Bay, including shore, piers, and small vessels. Station positions were logged from either the vessel's GPS system or a hand-held GPS instrument and recorded in a field log. The whole fish samples were wrapped in clean aluminum foil, placed in food-grade polyethylene bags, stored on ice, and frozen within 24 hours of collection. The sample bags were labeled with project name, sample identification number, site location, and date and time.

A total of 137 sport fish were collected during this study. Species collected included, barred sand bass, bonefish, California halibut, Pacific chub mackerel, round stingray, sargo, shortfin corvina, spotted sand bass, yellowfin croaker, and topsmelt. A subset of species and samples were selected for chemical analysis. This subset was selected to include fish frequently consumed by the local population, and included California halibut, Pacific chub mackerel, round stingray, topsmelt and spotted sand bass (Table 4). Sample selection also considered location, so that contaminant concentrations could be compared among the North, Central, and South regions of the Bay.

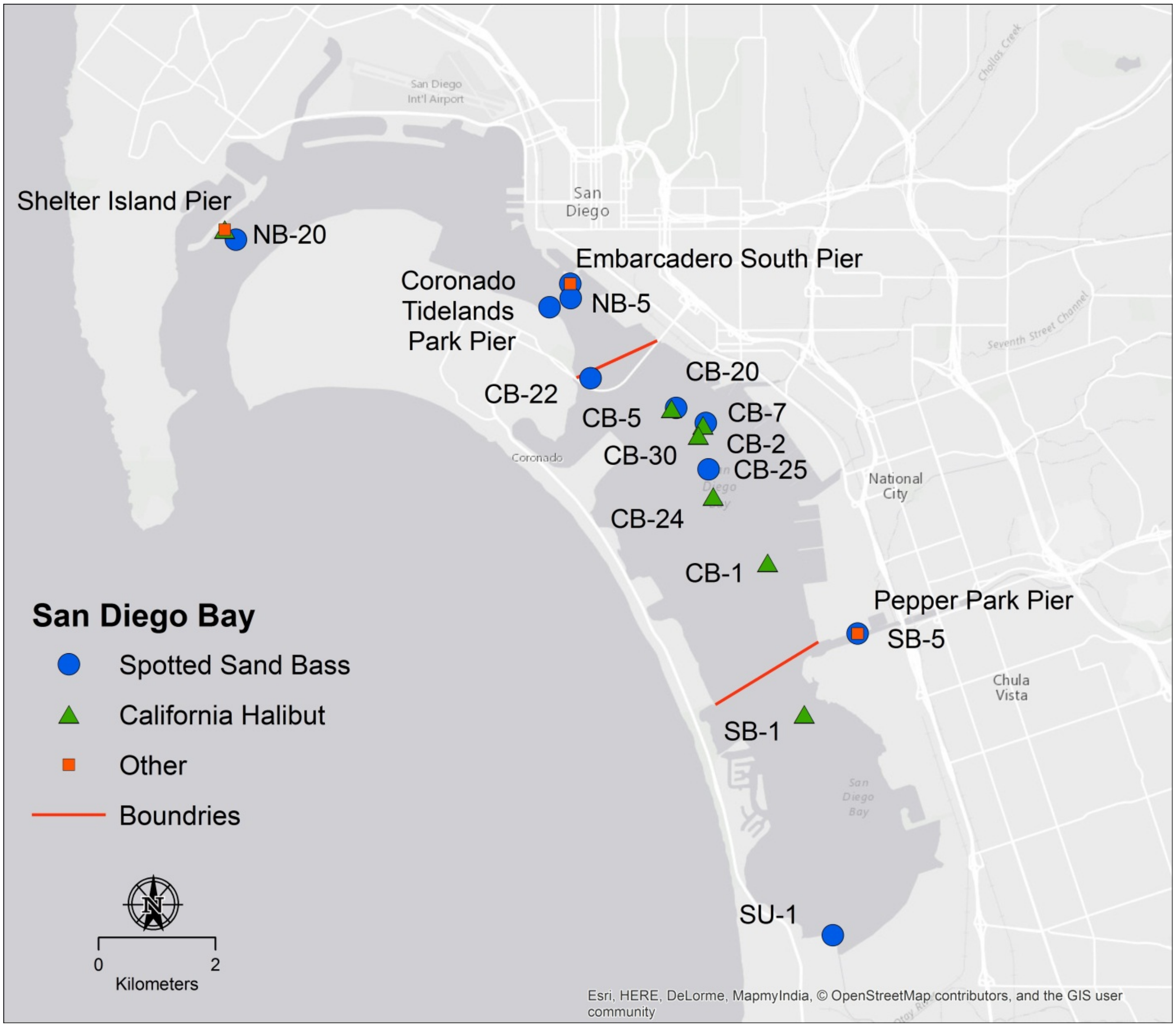

Figure 4. Sport fish collection locations.

Table 4. Location of sport fish samples collected during 2014 SCCWRP rig fishing or 2015 fish derby and used for chemical analysis.

Event	Location	Mode	Station	Date	Latitude	Longitude	Species[1]
SCCWRP	North Bay	Boat	NB-5	9/25/2014	32.70088	-117.16477	SSB
Derby	North Bay	Boat	CB-22	10/14/2014	32.688537	-117.161099	SSB
Derby	North Bay	Pier	Coronado Tidelands Park	6/6/2015	32.699481	-117.168635	SSB
Derby	North Bay	Pier	Embarcadero South	6/6/2015	32.703114	-117.164845	PCM, RR, SSB
Derby	North Bay	Boat	NB-20	12/10/2014	32.709799	-117.225872	SSB
Derby	North Bay	Pier	Shelter Island	6/6/2015	32.711349	-117.227948	CH, PCM, RR
SCCWRP	Central Bay	Boat	CB-1	10/14/2014	32.66007	-117.12877	CH
SCCWRP	Central Bay	Boat	CB-2	12/10/2014	32.68128	-117.14057	CH
SCCWRP	Central Bay	Boat	CB-20	12/10/2014	32.68399	-117.14548	SSB
SCCWRP	Central Bay	Boat	CB-7	12/10/2014	32.68167	-117.14009	SSB
Derby	Central Bay	Boat	CB-24	6/6/2015	32.6703	-117.138716	CH
Derby	Central Bay	Boat	CB-30	6/6/2015	32.6745	-117.139566	SSB
Derby	Central Bay	Boat	CB-5	7/4/2015	32.68383	-117.14635	CH
Derby	Central Bay	Boat	CB-25	7/5/2015	32.6797	-117.14145	CH
SCCWRP	South Bay	Boat	SB-1	10/14/2014	32.60269	-117.11681	SSB
SCCWRP	South Bay	Boat	SU-1	12/10/2014	32.63674	-117.12204	CH
Derby	South Bay	Pier	Pepper Park	6/6/2015	32.649231	-117.112357	SSB, TS
Derby	South Bay	Pier	SB-5	6/6/2015	32.649231	-117.112357	PCM

[1]Species codes: CH = California halibut; SSB = spotted sand bass; PCM = pacific chub mackerel; RR = round stingray; TS = topsmelt.

Chemical analysis

Chemical analyses of the samples were conducted primarily by Physis and the City of San Diego. Sediment samples were analyzed for total solids, total organic carbon (TOC), grain size, aluminum, antimony, arsenic, barium, beryllium, cadmium, chromium, copper, iron, lead, mercury, nickel, selenium, silver, zinc, polycyclic aromatic hydrocarbons (PAHs), chlorinated pesticides, pyrethroid pesticides, polychlorinated biphenyl congeners (PCBs), and polybrominated diphenyl ethers (PBDEs).

Tissue samples were analyzed for total solids, lipids, mercury, selenium, PCBs, chlorinated pesticides, and PBDEs (Table 5). Samples of bird eggs were also analyzed for perfluorinated compounds (PFCs) a compound of emerging concern.

Table 5. Analytes and laboratory analytical methods for tissue samples.

Analyte	Analysis Method	Tissue Target Reporting Limits [1]	Units
Percent Solids	SM 2540B	--	%
Lipid	Gravimetric	--	%
Mercury [2]	245.7 [3]	0.02	µg/g
Chlorinated Pesticides [4]	8270C/8270D [3]	0.5	ng/g
Polychlorinated Biphenyl (PCB) Congeners[5]	8270C/8270D PCB [3]	0.5	ng/g
Polybrominated Diphenyl Ethers (PBDEs) [6]	8270C/8270D NCI	0.2-10	ng/g

[1] Tissue minimum reporting limits are on a dry-weight basis.
[2] Reporting limits were provided by Physis Environmental Laboratories.
[3] USEPA 1986-1996. SW-846. Test Methods for Evaluating Solid Waste, Physical/Chemical Methods, 3rd Edition.
[4] Includes cis-chlordane, trans-chlordane, o.p'-DDT, p,p'-DDT, o.p'-DDD, p,p'-DDD, o.p'-DDE, p.p'-DDE, p,p'-DDMU, aldrin, BHC-alpha, BHC-beta, BHC-gamma, cis-nonachlor, trans-nonachlor, oxychlordane, DCPA (Dacthal), dicofol, dieldrin, toxaphene, endosulfan sulfate, endosulfan-I, endosulfan-II, endrin, endrin aldehyde, endrin ketone, heptachlor, heptachlor epoxide, methoxychlor, mirex, and perthane.
[5] Includes congeners: PCB-3, 5, 8, 15, 18, 27-29, 31, 33, 37, 44, 49, 52, 56, 60, 66, 70, 74, 77, 81, 87, 95, 97, 99, 101, 105, 110, 114, 118, 119, 123, 126, 128, 137, 138, 141, 149, 151, 153, 156-158, 167-170, 174, 177, 180, 183, 187, 189, 194,195, 200, 201, 203, 206, and 209.
[6] Includes congeners: BDE-17, 28, 47, 49, 66, 85, 99, 100, 138, 153, 154, 183, and 209.

Data analysis

Contamination data were summarized for both the entire San Diego Bay and by region. Three Bay regions were evaluated: North, Central, and South. Bay region boundaries are shown in Figure 2.

Egg fresh-weight correction

Results of chemical analyses are presented as fresh weight (fw)-based concentrations, which entail adjustments of analytical results for moisture loss that can occur with time between when the egg was laid and when it was collected. To obtain fw-based values, wet weight-based contaminant levels reported by the laboratory were adjusted according to methods by Stickel et al. (1973), using volume and weight measurements obtained for each egg. The extent of moisture loss from individual eggs was variable, such that adjustment factors ranged from 0.27 to 1.0. Mean adjustment factors were calculated for those samples that were composites of multiple eggs (i.e., least terns).

Wet weight (ww)-based concentrations reported by the laboratory were converted to fresh weight-based values as follows:

fw concentration = ww concentration x adjustment factor

Contaminant summarization

Results of tissue analyses for mercury are all reported as parts per million (ppm), either ug/g ww or ug/g fw (eggs). Concentrations of tissue organic analytes are reported as parts per billion (ppb), either as ng/g ww or ng/g fw (eggs). Sediment chemistry results are reported on a dry weight basis, either as ug/g dw for mercury or ng/g dw for organics. Results for other trace metals in sediment are not included in this summary, due to lack of matching data in tissue samples and limited potential to biomagnify through the food web.

Sums of organic contaminant classes (e.g., chlordanes, PCBs, DDTs, PBDEs, and PFCs) were calculated as the sum of all detected analytes within the class. In cases where all class components were nondetect for a sample, the sum value was represented by the highest detection limit of any of the class components. For seabird eggs, the sums of the 41 Bight '13 PCB congeners are approximately 17% lower than total PCB concentrations computed in other studies as sums of 90 or more congeners, sums of homologs, or by using an Aroclor standard approach. The PCB concentrations used to evaluate food chain relationships below, were adjusted upward by approximately 17% in the chapter on Wildlife Risk to enable comparisons with results of other studies and screening thresholds for adverse effects.

Quality control

Each batch of chemical analyses included several types of quality control (QC) samples to document laboratory method performance. The QC procedures followed guidelines used in the Bight'13 regional monitoring program and were consistent with Surface Water Ambient Monitoring Program (SWAMP) recommendations. The QC elements included:

- **Calibration Verification.** A new response factor or calibration curve was established for each instrumental batch. A calibration verification standard was analyzed every 12 hours to check the accuracy of the calibration. The control limit for this element was ±20% of the true value.

- **Method Blanks.** A method blank was run with each sample preparation batch (or per every 20 samples) and processed identical to the field samples. The control limit for blanks was <Reporting Limit for each analyte.

- **Sample Duplicates.** Analysis of sample duplicates was conducted at a frequency of 5% of the total sample count. The control limit for this element was a relative percent difference (RPD) of no more than 35%.

- **Matrix Spikes and Matrix Spike Duplicates.** Matrix spike and matrix spike duplicates (MS/MSD) were analyzed at a frequency of one per batch or for every 20 samples (whichever is more frequent). The control limit for MS was 50-150% recovery and the control limit for MSD was RPD ≤ 25%.

- **Certified Reference Materials or Laboratory Control Samples.** Method accuracy was evaluated through the analysis of either certified reference materials (CRM) or laboratory control samples (LCS) at a frequency of one per batch or per every 20 samples. The CRM control limit was 70-130% recovery and 50-150% for LCS.

- **Standards and Standard Recovery.** Quantification standards consisted of either isotope-labeled or structurally similar analogues to the target analytes and were included with every sample analyzed. The control limit for standard recovery was 50-150%.

All analytical data were reviewed for QC performance by the analytical laboratory and SCCWRP. QC sample results not meeting the control limits were flagged and investigated to determine the need for corrective action.

Bioaccumulation among Food Web Components

Data Analysis

Biota sediment accumulation factor

A biota sediment bioaccumulation factor (BSAF) was calculated for each tissue sample. The BSAF represents the degree of contaminant bioaccumulation in the sample, relative to the sediment. The BSAF was calculated as:

$$BSAF = C_{Tis}/C_{Sed}$$

Where:

C_{Tis} = tissue contaminant concentration (ug/g ww or ng/g ww)

C_{Sed} = sediment contaminant concentration (ug/g dw or ng/g dw)

The sediment concentration used for BSAF calculation was the average of all sediment samples from the Bay region of interest. The median of all individual values was used to represent the BSAF for each taxonomic group and region.

Normalized biota sediment accumulation factor

Normalized BSAFs were calculated to account for the effect of tissue lipid and sediment organic content on bioaccumulation in benthic infauna and fish.

The normalized BSAF was calculated as:

$$\text{Normalized BSAF} = C_{TisLipid}/C_{SedOC}$$

Where:

$C_{TisLipid}$ = tissue contaminant concentration (ug/kg lipid)

C_{SedOC} = sediment contaminant concentration (ug/kg organic carbon)

The sediment contaminant and total organic carbon concentrations used for fish and infauna BSAF calculations were based on measurements of sediment at the collection station corresponding to each tissue sample.

Bay-wide issue and sediment contamination

General trends in bioaccumulation patterns are illustrated in the following sections using box plots, which show the median, interquartile range (25^{th} to 75^{th} percentile), and individual data points.

Tissue chemistry results for the entire San Diego Bay showed that biomagnification among food web components was evident for all major contaminant types evaluated (Figure 5, Table 6). Relative to the other sample types, bird eggs (all species combined) had the highest median concentration of PCBs, DDTs, PBDEs, chlordanes, and mercury. Similar patterns were evident for most contaminant types, with the lowest concentrations occurring in the lowest trophic levels of plankton and benthic infauna (crustaceans, mollusks, polychaetes). Intermediate contaminant levels were present in fish (all species combined). PBDEs did not follow this pattern, however, with fish having similar or lower median concentrations as benthic infauna and plankton.

The greatest bioaccumulation relative to sediment was observed for PCBs and DDTs, where all food web components had median concentrations above bay-wide sediment means. Benthic infauna and plankton did not show much bioaccumulation of chlordanes relative to sediment, but the lack of detectable concentrations in many samples may have obscured some of the relationships. Median tissue mercury concentrations were below sediment levels for all trophic levels, likely reflecting a relatively low influence of local sediment mercury on tissue levels. Most of sediment mercury is present in the inorganic form, while methylated forms of mercury are the bioavailable form responsible for most of the bioaccumulation.

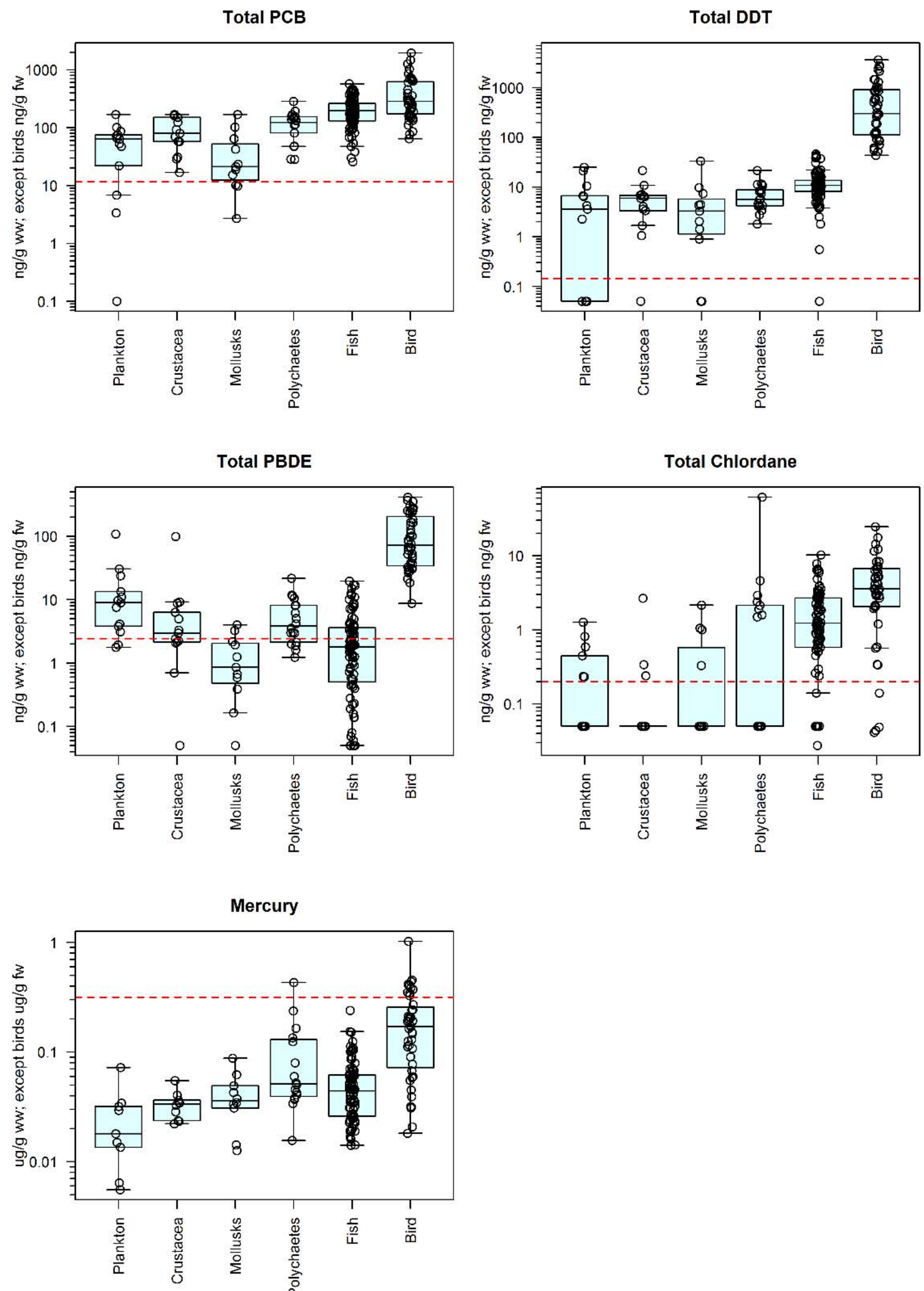

Figure 5. Summary of tissue contamination data for the entire San Diego Bay. The median is represented by the horizontal line, the box shows the interquartile range (IQR), and the whiskers flag potential outliers and extend to the furthest data point that is <1.5 x IQR from the box. Circles show individual data values. Dashed line indicates average contaminant concentration in sediment (dry weight basis).

Table 6. Bay-wide contaminant concentration summary by taxonomic group.

Group	Analyte	Min	Mean	Median	Max	StdDev	Count	Units
Bird	Chlordanes	0.04	5.03	3.56	24.64	4.92	44	ng/g fw
Bird	DDTs	43.32	686.02	301.30	3643.55	855.88	44	ng/g fw
Bird	PBDEs	8.77	123.33	71.97	413.87	110.99	44	ng/g fw
Bird	PCBs*	64.30	418.54	284.17	1950.77	393.69	44	ng/g fw
Bird	PFCs	7.54	27.63	25.84	72.53	18.39	27	ng/g fw
Bird	Mercury	0.02	0.20	0.17	1.02	0.18	43	ug/g fw
Crustacea	Chlordanes	0.05	0.29	0.05	2.66	0.72	13	ng/g ww
Crustacea	DDTs	0.05	6.07	5.97	21.48	5.48	13	ng/g ww
Crustacea	Dieldrin	0.05	0.05	0.05	0.05	0.00	7	ng/g ww
Crustacea	PBDEs	0.05	11.13	2.96	99.45	26.69	13	ng/g ww
Crustacea	PCBs	16.84	91.46	79.30	168.59	54.07	13	ng/g ww
Crustacea	Mercury	0.02	0.03	0.03	0.05	0.01	10	ug/g ww
Fish	Chlordanes	0.03	1.82	1.23	10.21	1.89	87	ng/g ww
Fish	DDTs	0.05	12.49	10.78	46.36	8.77	87	ng/g ww
Fish	Dieldrin	0.05	0.05	0.05	0.05	0.00	21	ng/g ww
Fish	PBDEs	0.05	3.17	1.79	19.53	4.20	87	ng/g ww
Fish	PCBs	25.69	211.45	197.15	570.60	112.32	87	ng/g ww
Fish	Mercury	0.01	0.05	0.04	0.24	0.04	82	ug/g ww
Mollusks	Chlordanes	0.05	0.44	0.05	2.15	0.68	11	ng/g ww
Mollusks	DDTs	0.05	6.07	3.29	33.08	9.46	11	ng/g ww
Mollusks	Dieldrin	0.05	0.05	0.05	0.05	0.00	2	ng/g ww
Mollusks	PBDEs	0.05	1.39	0.86	4.00	1.30	11	ng/g ww
Mollusks	PCBs	2.70	43.55	21.14	169.42	50.88	11	ng/g ww
Mollusks	Mercury	0.01	0.04	0.04	0.09	0.02	10	ug/g ww
Polychaetes	Chlordanes	0.05	4.40	0.05	61.61	14.34	18	ng/g ww
Polychaetes	DDTs	1.81	7.14	5.67	21.53	4.65	18	ng/g ww
Polychaetes	Dieldrin	0.05	0.05	0.05	0.05	0.00	9	ng/g ww
Polychaetes	PBDEs	1.23	6.03	3.86	21.68	5.24	18	ng/g ww
Polychaetes	PCBs	28.43	122.21	123.19	283.07	62.95	18	ng/g ww
Polychaetes	Mercury	0.02	0.10	0.05	0.43	0.11	16	ug/g ww

*calculated total PCB concentrations used for the wildlife risk evaluation are higher.

Table 6. Continued.

Group	Analyte	Min	Mean	Median	Max	StdDev	Count	Units
Plankton	Chlordanes	0.05	0.30	0.05	1.26	0.38	13	ng/g ww
Plankton	DDTs	0.05	6.15	3.59	24.95	8.19	13	ng/g ww
Plankton	PBDEs	1.77	17.54	8.96	108.27	28.58	13	ng/g ww
Plankton	PCBs	0.10	58.83	63.56	169.45	46.53	13	ng/g ww
Plankton	Mercury	0.01	0.03	0.02	0.07	0.02	9	ug/g ww
Plankton	Chlordanes	0.05	0.30	0.05	1.26	0.38	13	ng/g ww

Bioaccumulation by bay region

Sediment contaminant concentrations of trace organics varied both by contaminant type and region. Sediment PCBs were present at approximately 20-100x higher concentrations than DDTs and chlordanes (Table 7). The North region of the Bay contained higher average concentrations of chlordanes, mercury, and PCBs. The concentrations of sediment DDTs were similar in the North and South, which were 2-3x higher than the concentration in the Central Bay.

Plankton and benthic infauna

Tissue contaminant concentrations of trace organics and mercury in plankton and benthic infauna were similar (Figures 6 and 7). In addition, there was no consistent trend in relative concentration among the three types of infauna. For PCBs, mollusks had lower median contaminant concentrations than crustaceans or polychaetes. However, total DDTs concentrations were similar among infauna groups (Figure 6). Tissue contamination in infauna also appeared to follow trends in sediment concentration for PCBs and DDTS, with higher tissue PCBs concentrations in the North and higher DDTs in the North and South.

Mercury concentrations in plankton and infauna were generally quite similar, except for elevated levels in polychaetes from the Central Bay (Figure 7). Little correspondence between sediment and tissue mercury concentrations was evident. Similar tissue mercury concentrations were observed in plankton and invertebrates among regions, even though sediment mercury concentration was 2-4 fold higher in the North. Relatively high mercury concentrations were present in Central polychaetes, relative to other infauna groups. It is possible that the elevated concentrations are due to the presence of sediment in the polychaete intestine or the tube. Most of the polychaetes in the samples were likely to be sediment deposit feeders that live within a tube constructed partly of sediment. The relatively small size of the polychaetes may have increased the occurrence of contamination of the sample with sediment containing higher mercury concentrations.

Table 7. Average chemical concentrations and standard deviations (SD) for sediments collected in three regions of San Diego Bay. N = number of samples used to calculate the average. NA = data not available.

Analyte Name	N	Average	SD	Units
San Diego North				
Chlordanes	24	0.44	1.34	ng/g dw
DDTs	24	0.16	0.41	ng/g dw
Dieldrin	24	0.05 [1]	NA	ng/g dw
Mercury	18	0.44	0.52	µg/g dw
PCBs	24	21.03	23.69	ng/g dw
San Diego Central				
Chlordanes	19	0.07	0.07	ng/g dw
DDTs	19	0.07	0.08	ng/g dw
Dieldrin	19	0.05 [1]	NA	ng/g dw
Mercury	19	0.26	0.15	µg/g dw
PCBs	19	7.09	7.34	ng/g dw
San Diego South				
Chlordanes[1]	21	0.05	NA	ng/g dw
DDTs	21	0.2	0.4	ng/g dw
Dieldrin[1]	21	0.05	NA	ng/g dw
Mercury	21	0.13	0.12	µg/g dw
PCBs	21	4.95	3.5	ng/g dw

[1] All dieldrin and chlordane values were below method detection limits.

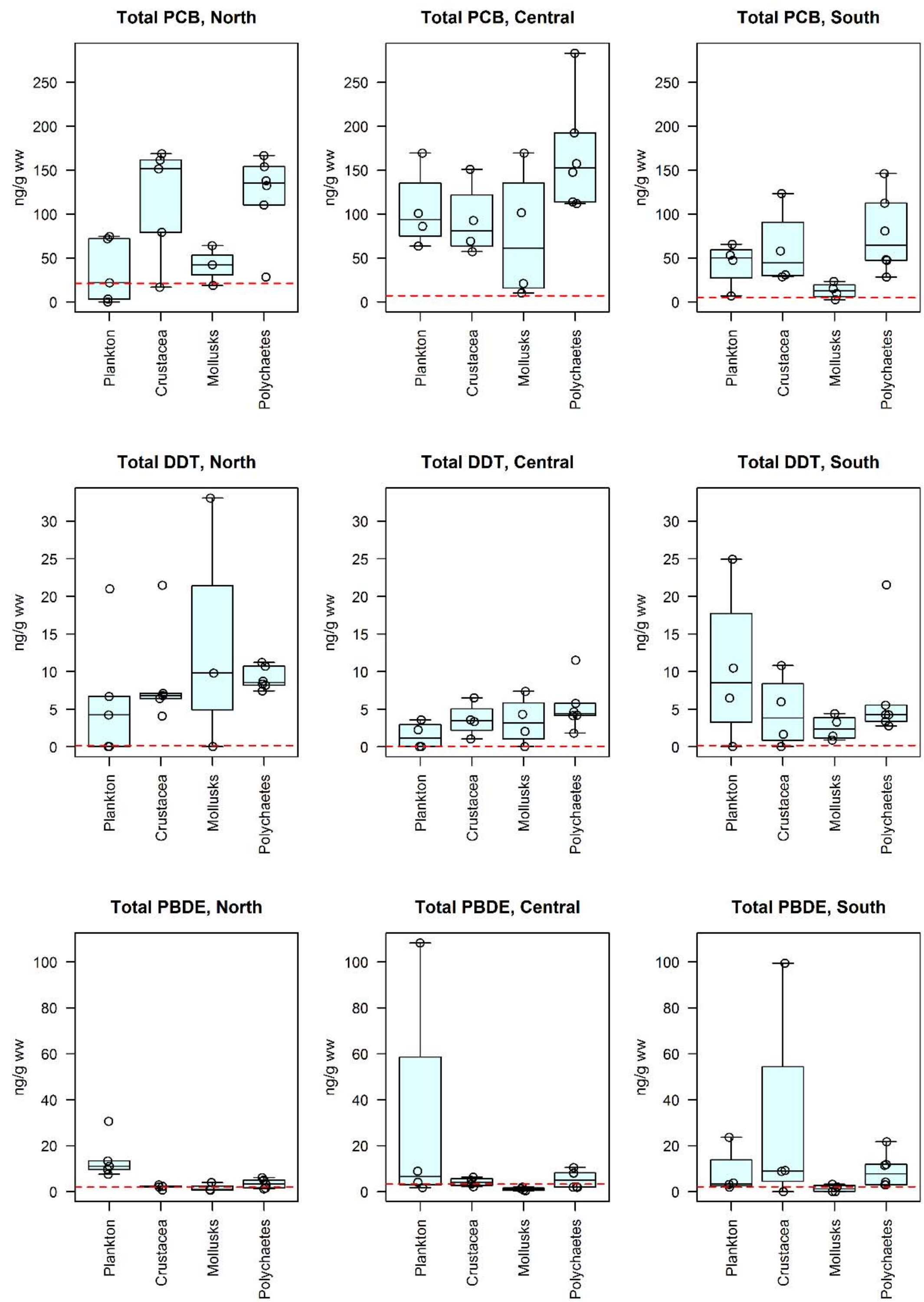

Figure 6. Concentrations of PCBs, DDTs, and PBDEs in plankton and benthic infauna by Bay region. See Figure 5 for description of plot symbols.

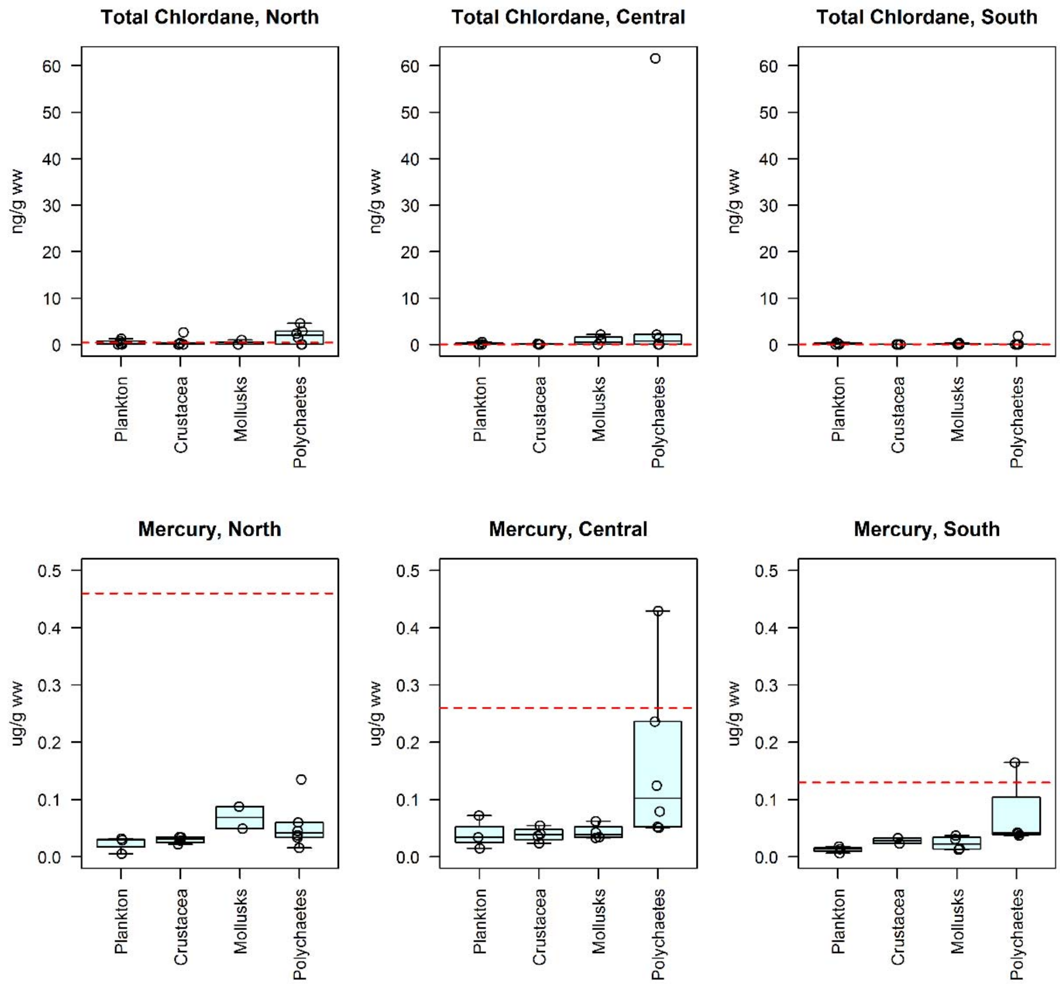

Figure 7. Concentrations of chlordanes and mercury in plankton and benthic infauna by Bay region. See Figure 5 for description of plot symbols.

Fish

The plots of fish tissue contaminants include only those species for which at least three samples were available from that region of the Bay. Data for additional species, as well as for Bay regions where fewer than three samples were analyzed, are included in Appendix A.

Median total PCB concentrations in fish tissue were generally highest in the Central Bay, with the lowest concentrations in fish from the South (Figure 8). This pattern differed from the trend seen for sediment, where the highest concentration of PCBs was measured in the North (3-5x higher than other regions).

Relative PCB concentration among species was variable and showed few consistent trends. There was a nearly eight-fold range in concentrations among species for the North and South, with less variability among species for the Central Bay (approximately 3-fold). Spotted sand bass had the highest or second-highest median PCB concentration within each region. The lowest

PCB concentrations were frequently measured in small fish that fed primarily on water column organisms (e.g., topsmelt, shiner perch, and killifish). However, relatively high PCB concentrations were also measured in some samples of small fish (e.g., Goby sp. in the North, deepbody anchovy in the Central, and northern anchovy in the South). The highest median PCB concentration (419 ng/g) was obtained for two samples of round stingray from the South (Appendix Table A3).

Total DDT concentrations in fish were generally about ten-fold lower than PCBs (Figure 8). Median concentrations of DDTs were usually similar in the North and Central, which were approximately two-fold higher than the South. Fish species having the highest median DDT concentrations were shiner perch and barred sand bass in the North, deepbody anchovy and shiner perch in the Central, and barred sand bass and slough anchovy in the South. Deepbody anchovy had the highest median concentration of any species group (28.8 ng/g in the Central). Round stingray, the fish species with the highest PCB concentration in the data set, had amongst the lowest median DDT concentration (0.6 ng/g).

Most fish samples did not contain detectable levels of mercury, and there was little variation in concentration among species or region (Figure 9). The highest median concentrations were measured in round stingray (0.15 ng/g South) and spotted sand bass (0.11 ng/g Central).

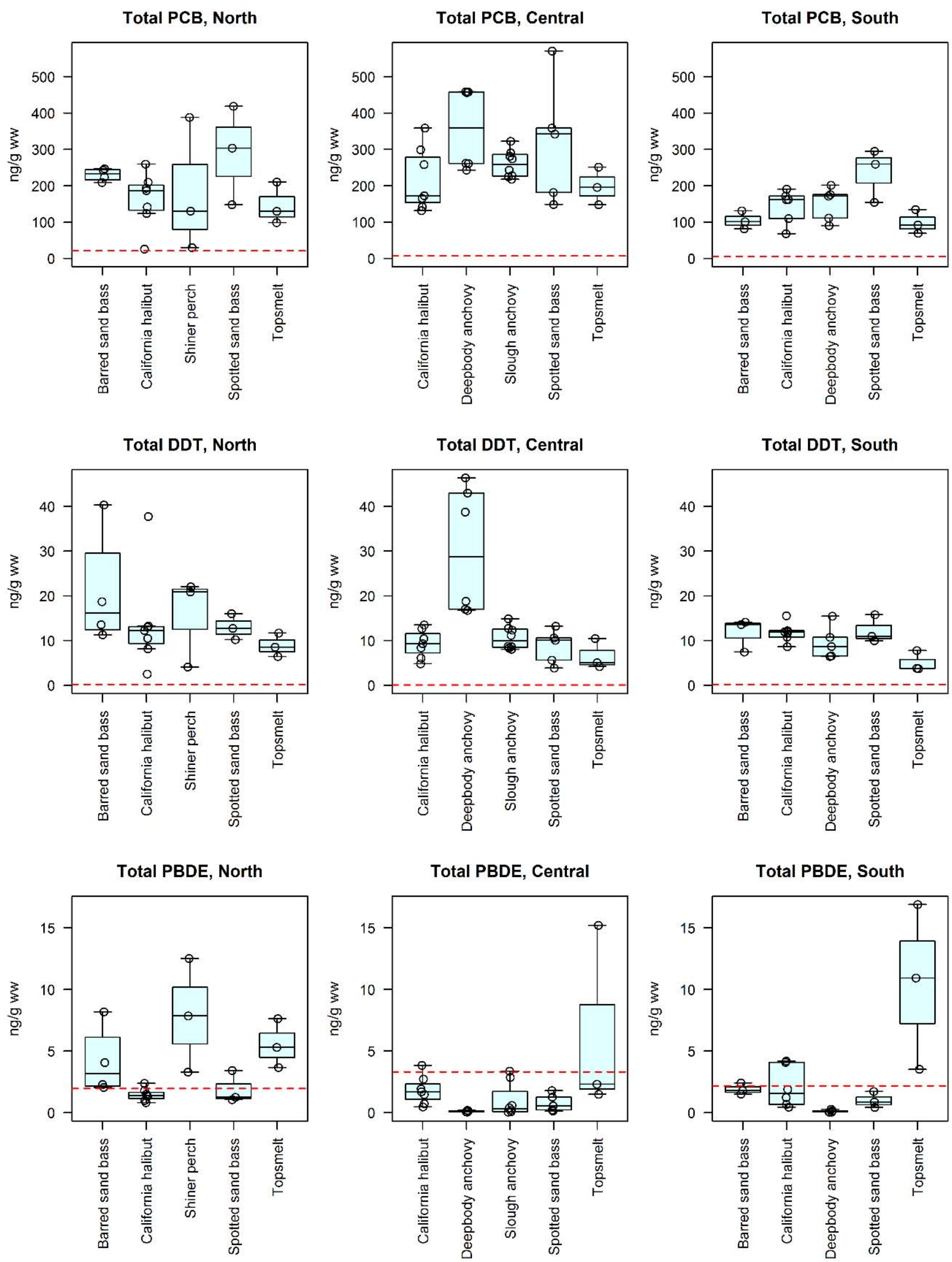

Figure 8. Concentrations of PCBs, DDTs, and PBDEs in fish by Bay region. See Figure 5 for description of plot symbols.

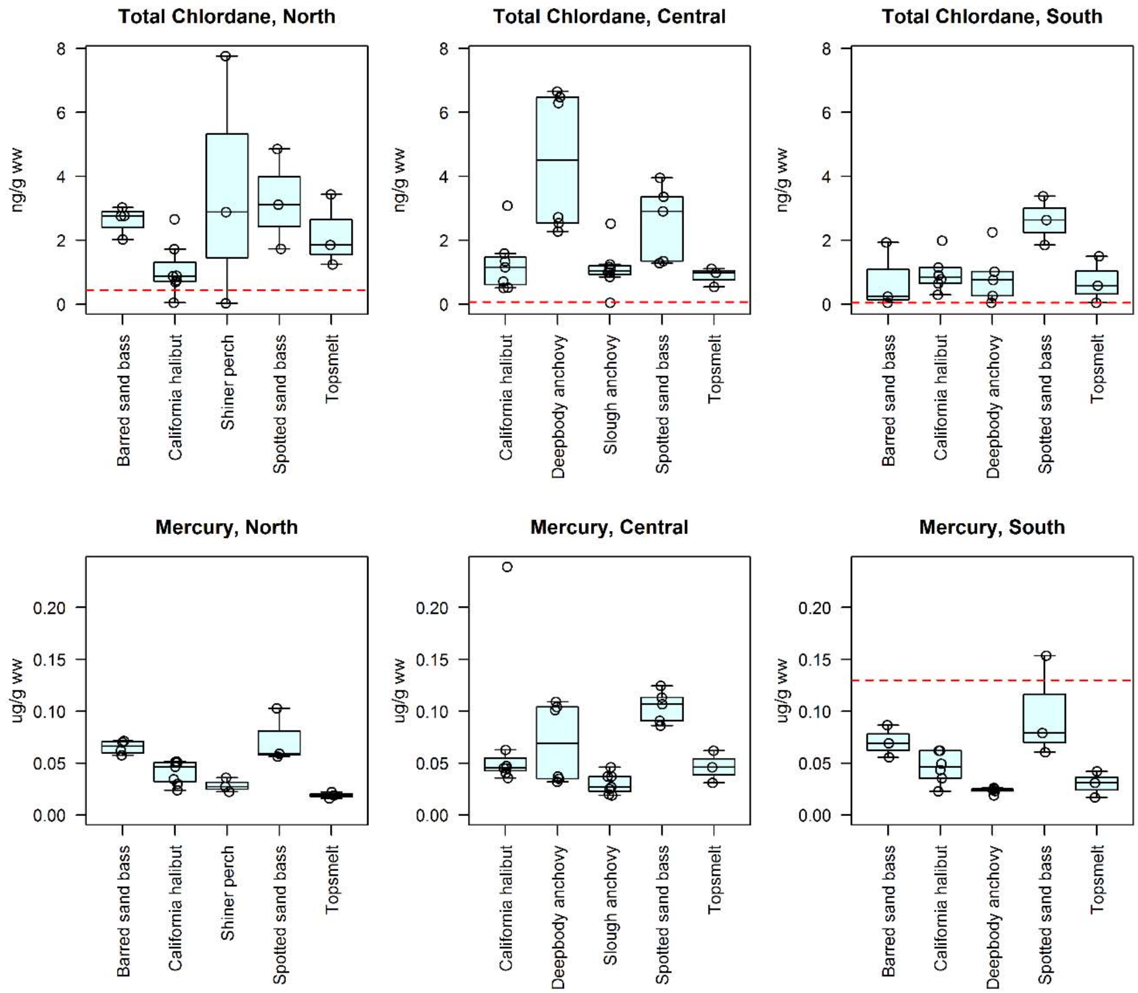

Figure 9. Concentrations of chlordanes and mercury in fish by Bay region. See Figure 5 for description of plot symbols.

Birds

Bird eggs were collected from multiple locations in the North and South; no eggs were obtained from the Central Bay, as there are no nesting areas in the Bay in this region. Regional comparisons can only be made for California least tern, as this was the only species having nesting areas in both the North and South.

Among the organic contaminants measured, concentrations of DDTs and PCBs were similar. This pattern differed from the fish, where PCBs were present at approximately ten-fold higher concentrations than DDTs. Caspian tern eggs contained the highest median concentration of each contaminant type measured, except for PCBs, which were slightly more concentrated in western gull eggs (Figures 10 and 11). Double-crested cormorant eggs had amongst the lowest concentrations of chlordanes, mercury, and PBDEs, compared to other species. Total PCBs, DDTs, and PBDEs in western gull eggs were present in higher concentrations than for California least tern eggs collected from the North region of the Bay.

There was little difference between regions in contaminant concentrations in California least tern eggs. Contaminant concentrations varied between the North and South by less than a factor of two for all contaminant types measured, with no consistent trend. These results indicate little difference in contaminant exposure among Bay regions for this species of special concern.

Contamination of bird eggs by legacy contaminants is evident throught the Southern California Bight. Analysis of over 100 bird egg samples collected during the Bight '13 regional survey detected DDTs and PCBs in virtually every sample, regardless of location (Appendix H).

Regional variation in bird egg contamination was evident for some compounds, as indicated by the data for California least tern eggs, which were collected at nine southern California locations ranging from Pt. Mugu in the north to Tijuana Estuary. Total PCB concentrations in least tern eggs were highest in San Diego Bay, relative to other locations, with approximately twice the concentration measured at northern locations (e.g., Pt. Mugu and Los Angeles Harbor). PBDEs were also highest in least tern eggs from San Diego Bay and Tijuana estuary, likely a reflection of inputs from urban stormwater discharges.

Conversely, there was little regional trend in least tern egg mercury concentrations among the Bight '13 samples, suggesting that mercury exposure is not driven by local contaminantion patterns. Total DDTs in least tern eggs were lowest in San Diego Bay and highest in samples from Los Angeles Harbor and Pt. Mugu, reflecting the influence of historical DDT contamination on the Palos Verdes Shelf.

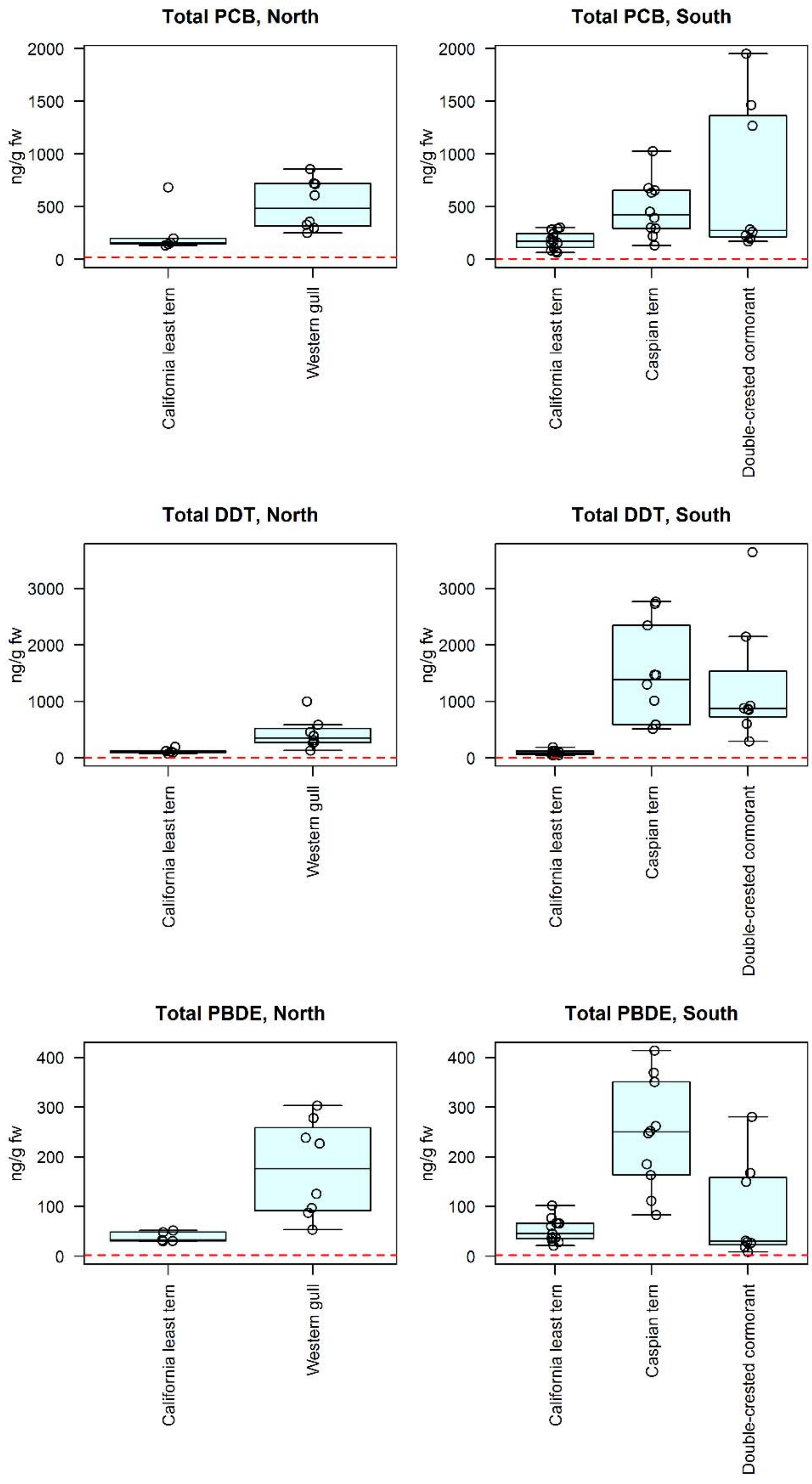

Figure 10. Concentrations of PCBs, DDTs, and PBDEs in bird eggs by Bay region. See Figure 5 for description of plot symbols.

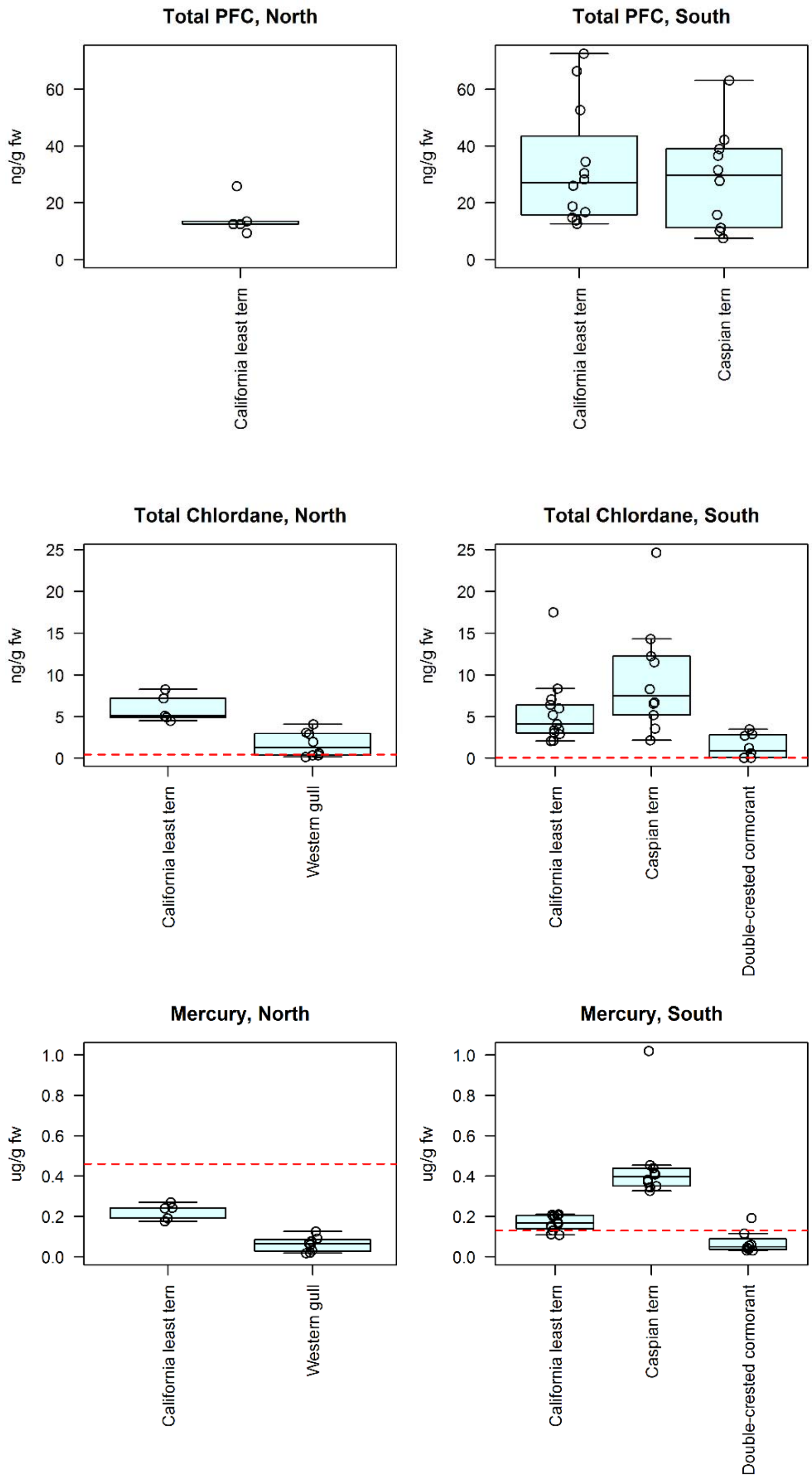

Figure 11. Concentrations of chlordanes, PFCs, and mercury in bird eggs by Bay region. See Figure 5 for description of plot symbols.

Bioaccumulation factors

Biota-sediment accumulation factors (BSAF) based on wet weight tissue concentrations and dry weight sediment concentrations were calculated for benthic infauna (Figures 12 and 13), fish (Figures 14 and 15), and birds (Figures 16 and 17). Patterns in BSAF among species, taxonomic groups, or regions are similar to those described for chemical concentrations, as the values are calculated using the same concentration values. However, calculation of BSAF facilitates comparison of bioaccumulation among contaminant types as the results are normalized to the sediment concentration. BSAF values for additional species, those where fewer than three samples were analyzed, are included in Appendix B.

Benthic infauna

Among benthic invertebrates, median BSAFs were highest for DDTs, where tissue concentrations were 12-63 times higher than sediment (Figure 12). Infauna also had relatively high BSAFs for PCBs, ranging from 2-22. Tissue mercury concentrations in infauna were less than sediment levels, resulting in BSAFs less than 1 (0.07 – 0.39, Figure 13). Bioaccumulation of PBDEs was low relative to sediment, with BSAFs ranging from 0.3 to 4. DDT BSAFs were similar among taxonomic groups, with no apparent trend. There was an apparent trend in PCB BSAFs among taxa, with mollusks having lower BSAFs, approximately one third of those calculated for polychaetes or crustaceans.

Regional differences in infauna BSAFs were evident only for DDTs. Infauna in the Central and North regions had DDT BSAFs that were approximately three times greater than South infauna. This variation suggests that there are differences in sediment characteristics among Bay regions that affect contaminant bioavailability.

Fish

Fish BSAFs varied widely among species and regions (Figures 14 and 15). For DDTs, median BSAFs ranged from 0.25 (arrow goby in South) to 411 (deepbody anchovy in Central). PCB BSAFs ranged from 6 (topsmelt and shiner perch in North) to 85 (round stingray in South). Fish BSAFs were generally three or four-fold higher than invertebrate values for the same Bay region, which is consistent with the higher trophic level of fish.

Regional variation in fish BSAFs was observed for DDTs and PCBs. Fish collected from the Central Bay tended to have higher BSAFs for both contaminant groups, with values approximately two to three times higher than fish in the other regions. The lowest DDT BSAFs were usually noted for South fish, while North fish usually had the lowest BSAFs for PCBs.

Birds

BSAFs based on bird egg contaminant data showed similar trends as described for fish, with DDT BSAFs higher than those for PCBs (Figures 16 and 17). But the magnitude of difference was much greater for bird eggs, with DDT BSAFs 50-100x higher than egg PCB BSAFs.

Egg PCB BSAFs ranged from 8 to 86, only slightly higher than those calculated for fish (6-85). DDT BSAFs were vastly different between birds and fish, however. Bird egg DDT BSAFs ranged from 435 to 6,923, approximately 10 to 50 times higher than fish BSAFs. Differences in lipid content between fish tissue and eggs likely had a major influence on these BSAFs, as DDTs and PCBs have a strong affinity for tissue lipids. Lipid normalized BSAFs, based on tissue lipid and sediment TOC normalized data were calculated in order to account for differences in matrix composition.

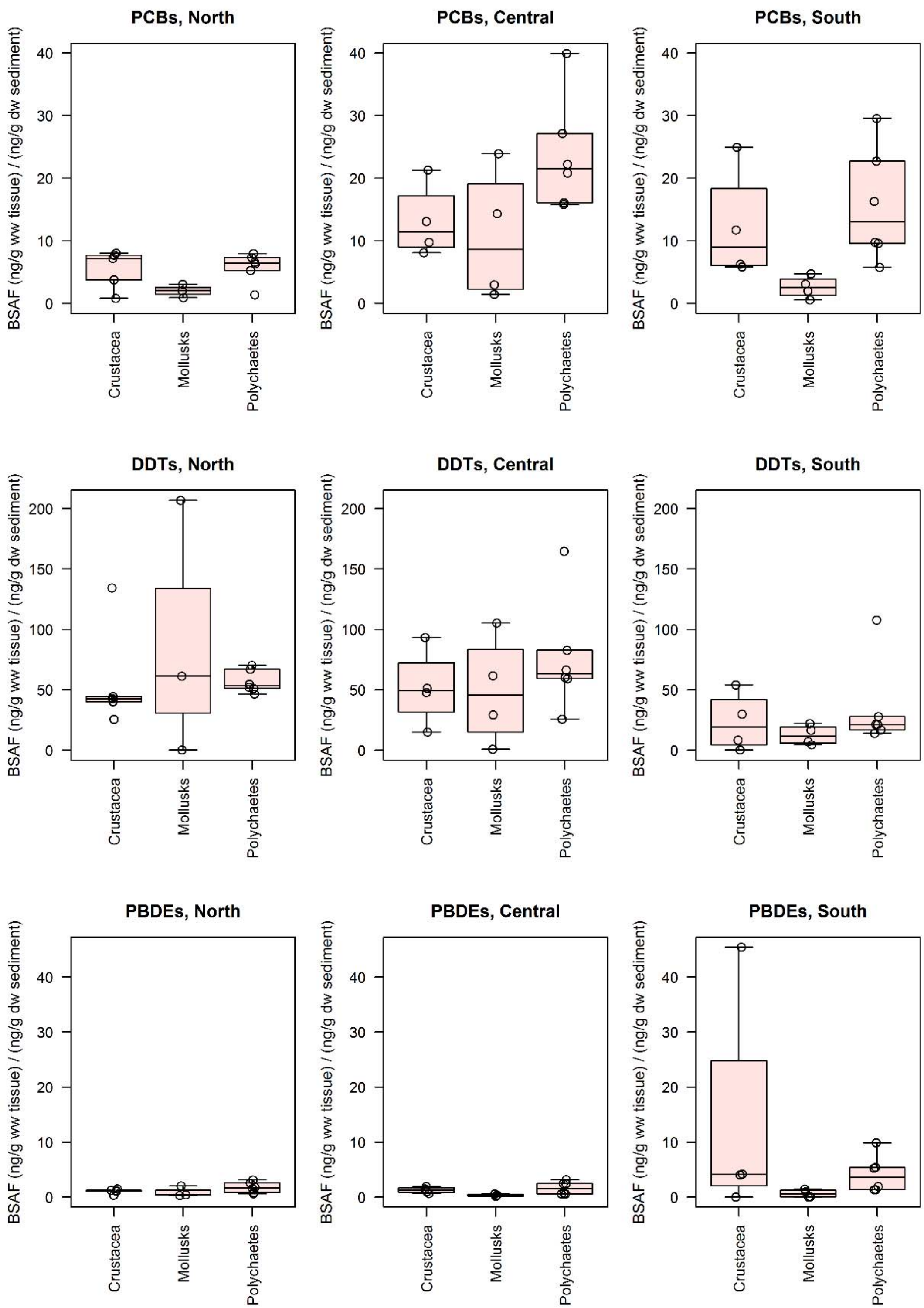

Figure 12. BSAF values for PCBs, DDTs, and PBDEs in benthic infauna. See Figure 5 for description of plot symbols.

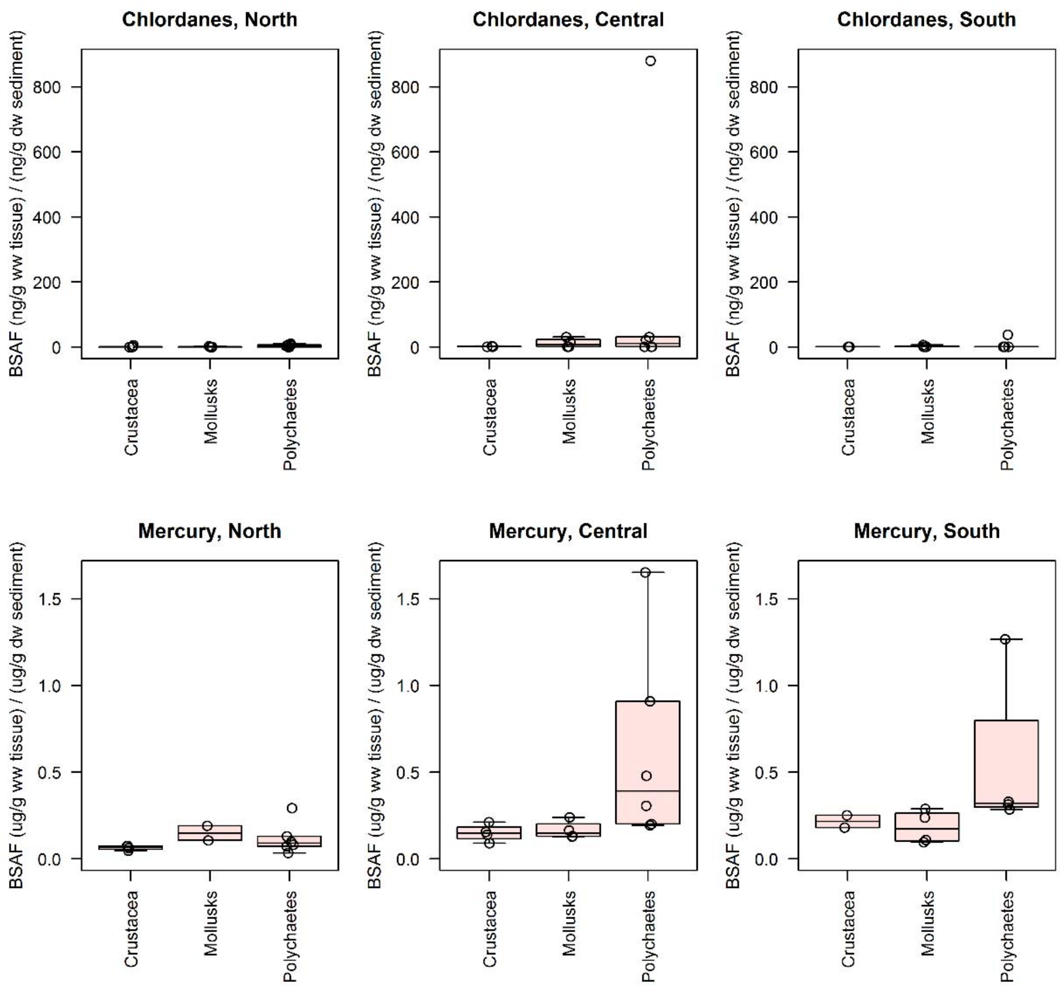

Figure 13. BSAF values for chlordanes and mercury in benthic infauna. See Figure 5 for description of plot symbols.

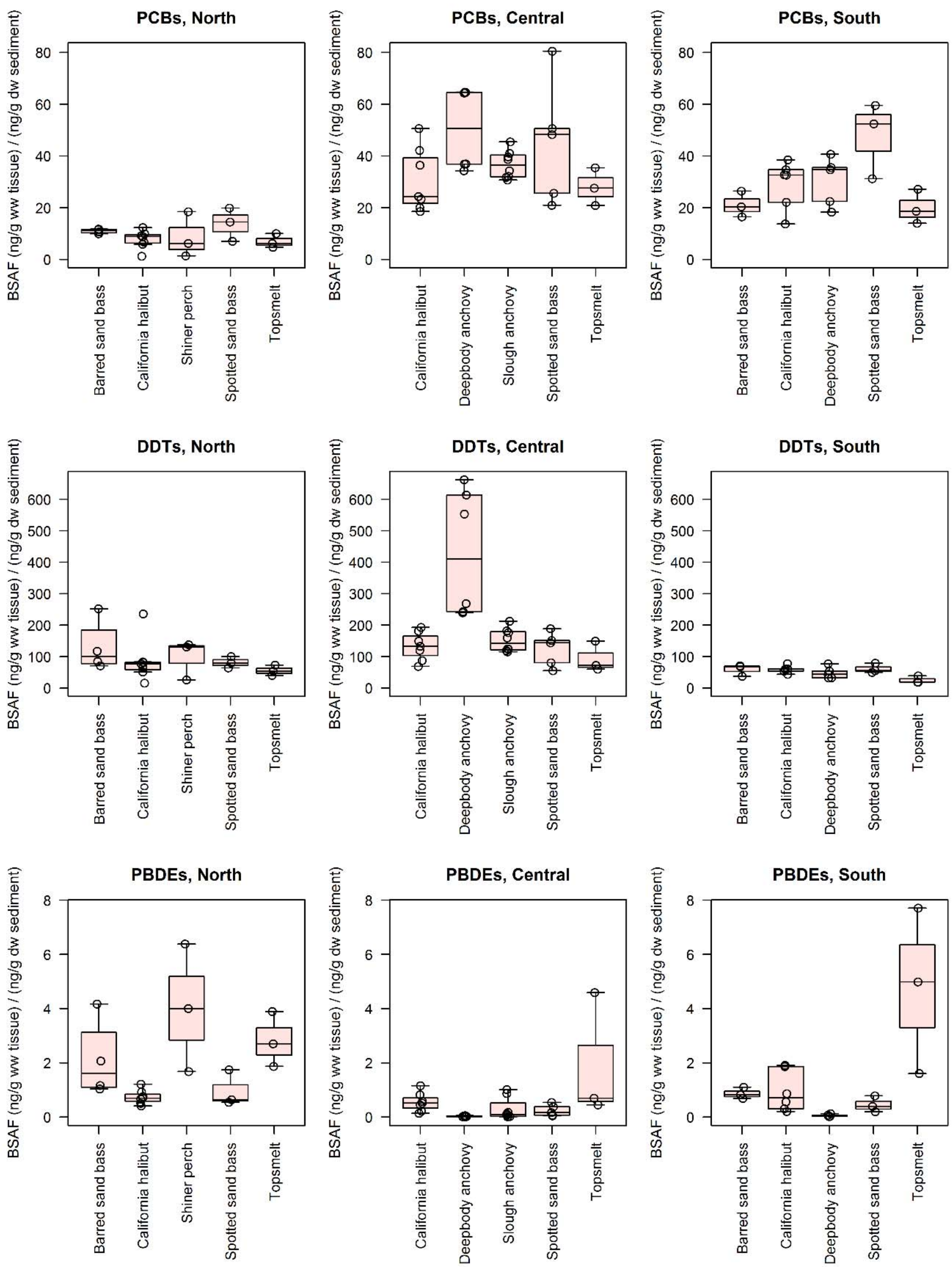

Figure 14. BSAF values for PCBs, DDTs, and PBDEs in fish. See Figure 5 for description of plot symbols.

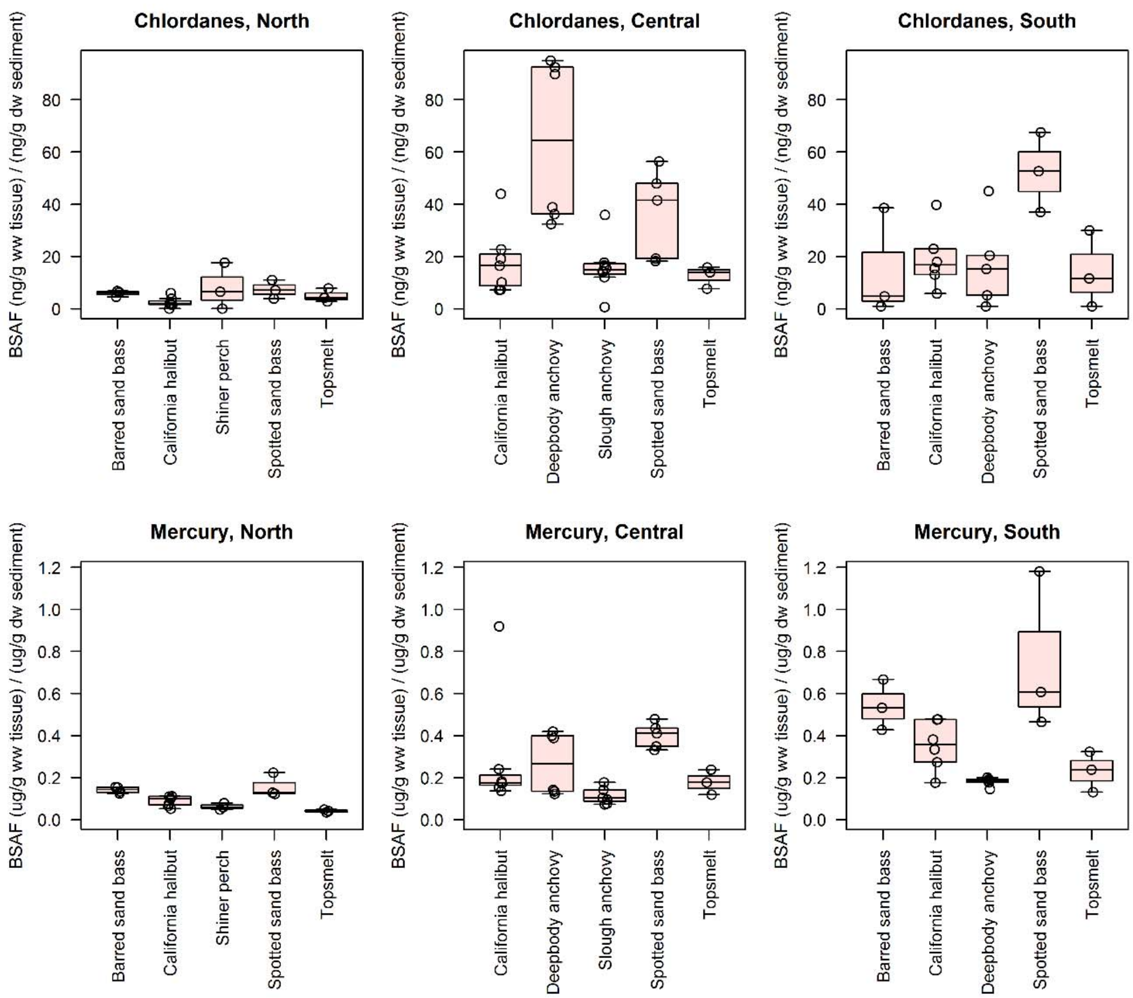

Figure 15. BSAF values for chlordanes and mercury in fish. See Figure 5 for description of plot symbols.

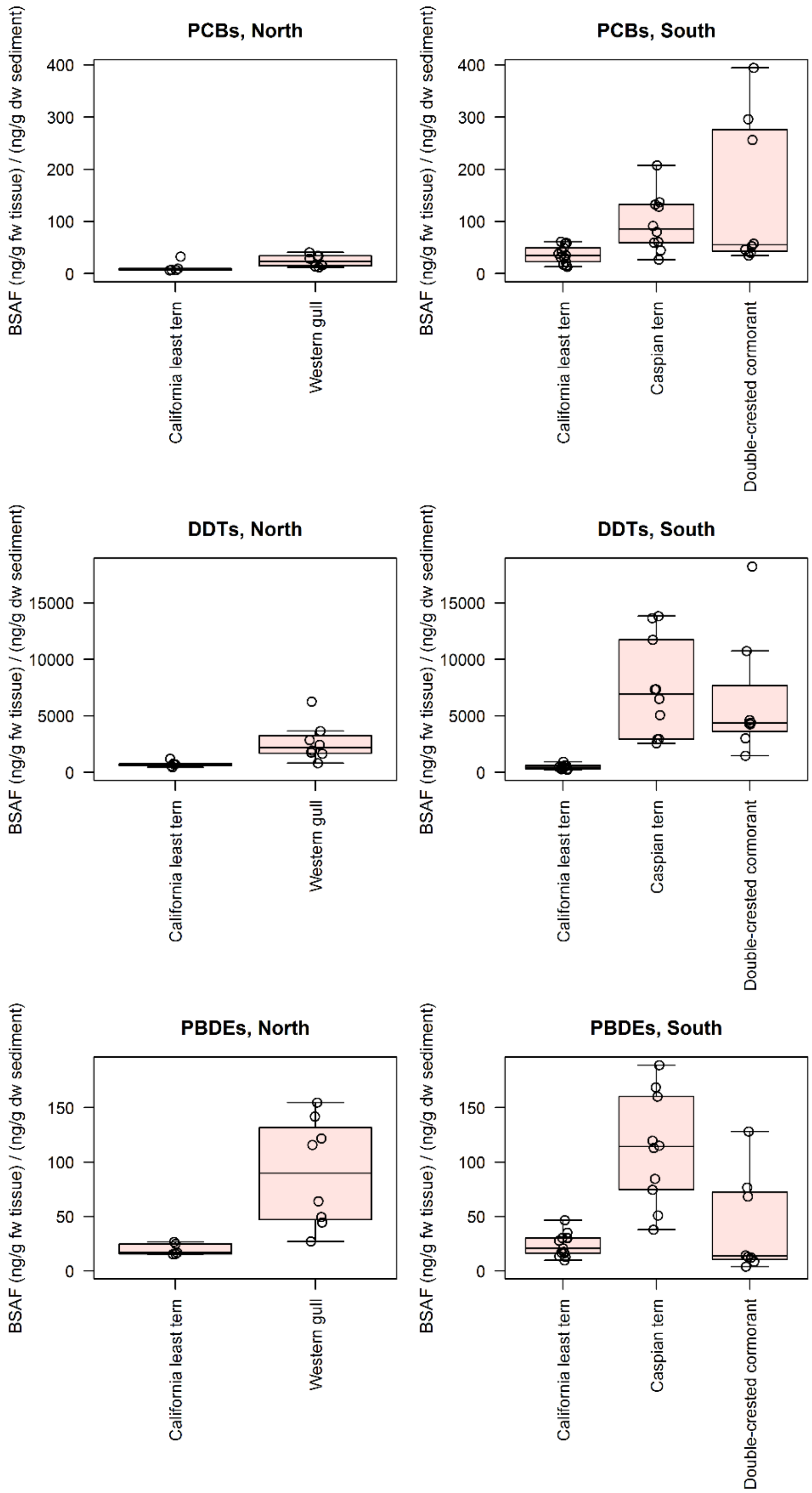

Figure 16. BSAF values for PCBs, DDTs, and PBDEs in bird eggs. See Figure 5 for description of plot symbols.

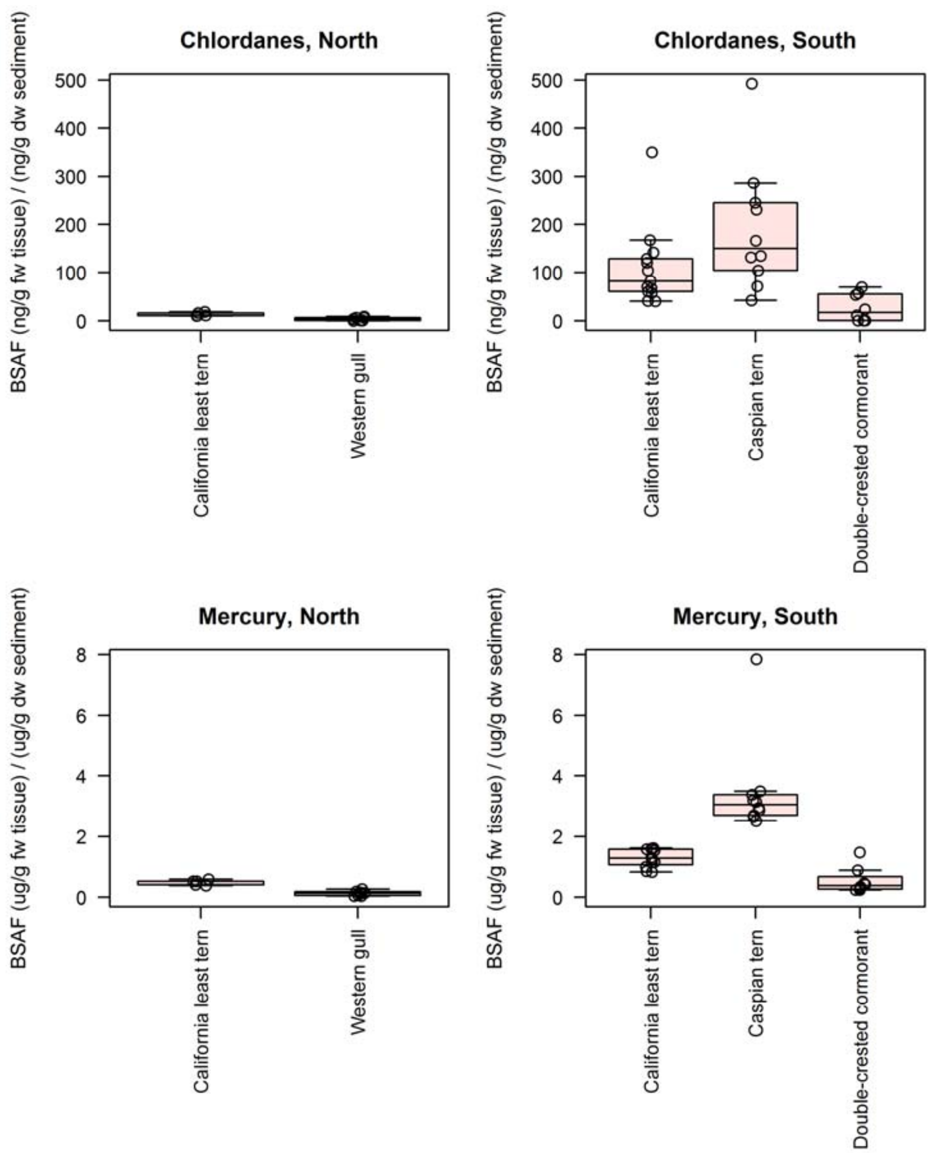

Figure 17. BSAF values for chlordanes and mercury in bird eggs. See Figure 5 for description of plot symbols.

Normalized Bioaccumulation factors

Lipid and total organic carbon (TOC) normalized BSAFs were calculated for benthic infauna (Figures 18 and 19) and fish (Figures 20 and 21). Calculation of normalized BSAFs allows for the reduction of interferences from variations in sediment TOC content and organism lipid content, which facilitates a more accurate comparison of bioaccumulation among trophic levels, species, and regions. Data for additional fish species, where fewer than three samples were analyzed, are included in Appendix C.

Benthic infauna

Among benthic invertebrates, median normalized BSAFs were still highest for DDTs, where tissue concentrations were 9-41 times higher than sediments. With the normalization to lipid and TOC content, the PBDE BSAFs increased 5 to 10-fold relative to the non-normalized BSAFs, whereas the PCB BSAFs decreased 2 to 4-fold.

Compared to polychaetes and crustacea, mollusks had 2-3 times lower median BSAFs for DDTs and PCBs in all regions, and 10-70 times lower BSAFs for PBDEs in the North and Central regions of San Diego Bay. This pattern is similar to that observed for non-normalized BSAF values with the exception of the magnitude of difference for PBDEs. PBDEs were the only contaminant to display overall regional differences as well. Crustacea and polychaetes in the North and Central regions had 2-20 times higher BSAFs compared to the South region.

Normalized BSAFs for infauna ranged from 0.4 to 41 for DDTs and PCBs and were frequently higher than expected based on bioaccumulation models. Traditionally, equilibrium partitioning models predict BSAF values in the range of 1-3. However, calculated values based on field samples are frequently outside this range. Analysis of data from Superfund sites by Burkhard et al. (2007) documented BSAF variations of several orders of magnitude. The 99^{th} percentile of those values has been proposed by Burkhard as a benchmark indicating numerically reasonable values, which corresponds to 38.9 for fish and 27.4 for invertebrates. Most of the median BSAFs for San Diego Bay benthic infauna were less than 27.4. However, three median DDT BSAF values exceeded the invertebrate benchmark. While the mechanism for these large values is uncertain, there are a few possible explanations. BSAFs assume equilibrium between sediment, pore water, and organism contaminant concentrations, but don't take into account other modes of exposure such as contaminated food or water column. Also, the sediment contaminant measurements do not account for concentration gradients that frequently occur at the sediment-water interface. Sediment chemistry samples were collected from the top 5 cm of sediment and may not represent contaminant concentrations at the sediment surface or in suspended particles near the sediment surface. Many benthic organisms, in particular crustacea and mollusks, feed at the sediment-water interface. Elevated contaminant concentrations in this top layer could have resulted in greater contaminant bioaccumulation than that expected based on bulk chemistry measurements.

Fish

Normalized BSAFs for fish continued to exhibit a wide range of variability. Median BSAFs for chlordanes ranged from 0.2 (arrow goby in South) to 62 (round stingray in South), with similar ranges for PCBs and PBDEs. Total DDTs had the largest BSAF range, from 0.06 (arrow goby in South) to 301 (deepbody anchovy in South). In general, fish BSAFs were at least two times higher than benthic BSAFs, consistent with the principles of trophic transfer. The only exception was PBDEs, where only three fish groups (barred sand bass and California halibut in North, and barred sand bass in Central) had elevated median PBDE BSAFs relative to infauna (about two-fold). There were no consistent regional differences in PBDE BSAFs for fish.

The regional differences noted for DDT BSAFs are no longer present with normalization. Regional variation for PCB BSAFs was unchanged with normalization. Central Bay fish tended to have higher BSAFs, followed by South fish and then North fish.

Some of the fish BSAFs exceeded the 99th percentile benchmark: 10 DDT BSAFs, and 1 BSAF each for chlordanes, PCBs, and PBDEs. It is not surprising that some of the BSAFs exceeded the benchmark value as the relationship between fish and sediment is more indirect than for infauna. The possibility of contaminated food sources and water column exposure providing extra inputs to the fish tissue is likely. Similar to the benthic infauna, contaminant exposure in bottom feeding fish may be influenced by concentrations at the sediment-water interface. This sediment may have higher contaminant concentrations relative to the 5 cm layer which was sampled. More investigation is needed in order to determine the cause of the relatively high normalized BSAFs.

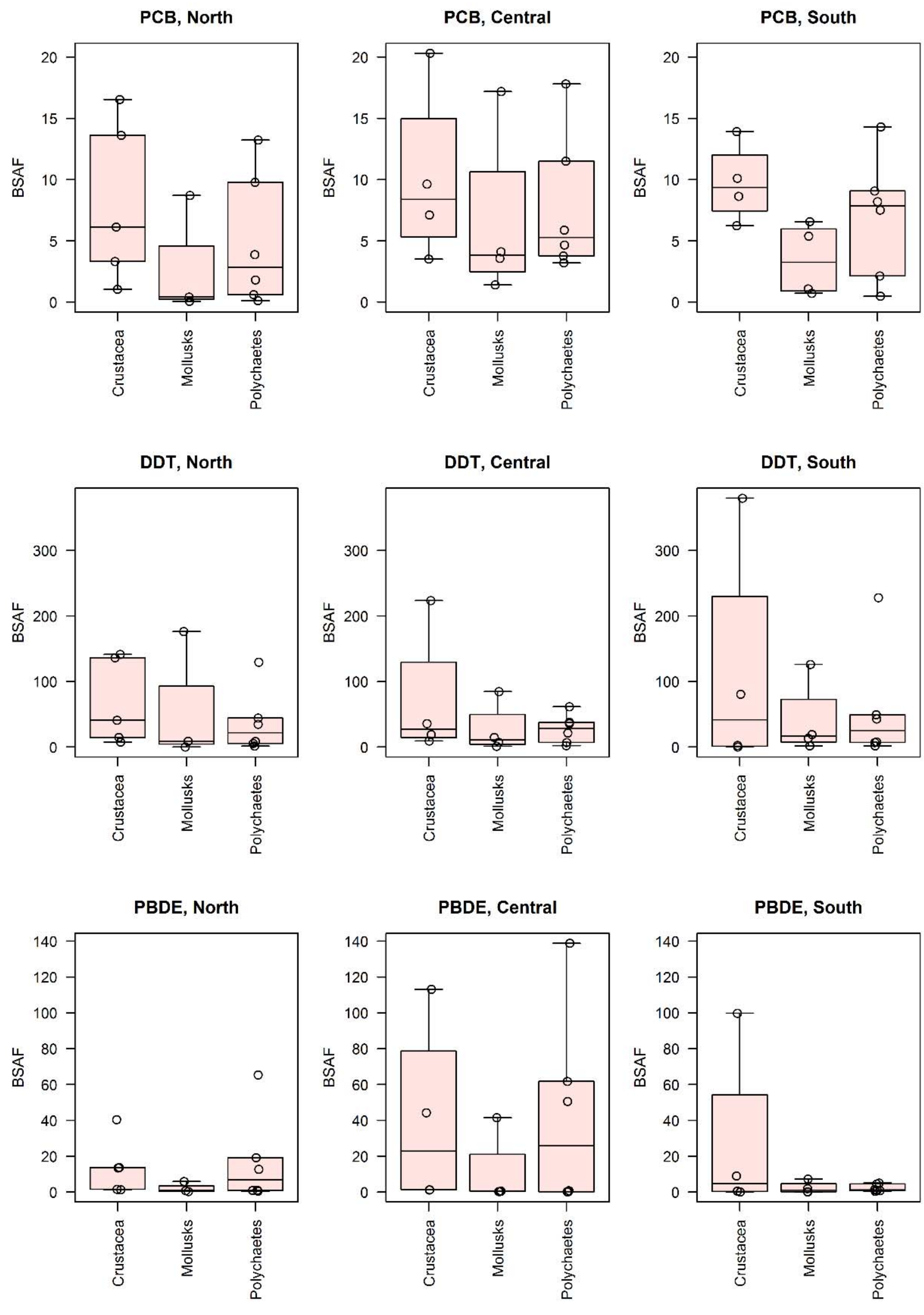

Figure 18. Normalized BSAF values for PCBs, DDTs, and PBDEs in benthic infauna. See Figure 5 for description of plot symbols.

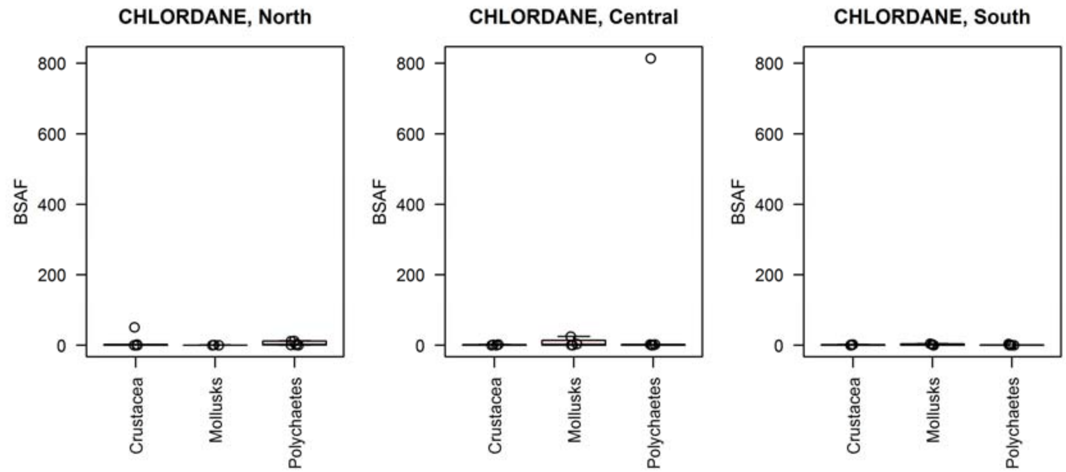

Figure 19. Normalized BSAF values for chlordanes in benthic infauna. See Figure 5 for description of plot symbols.

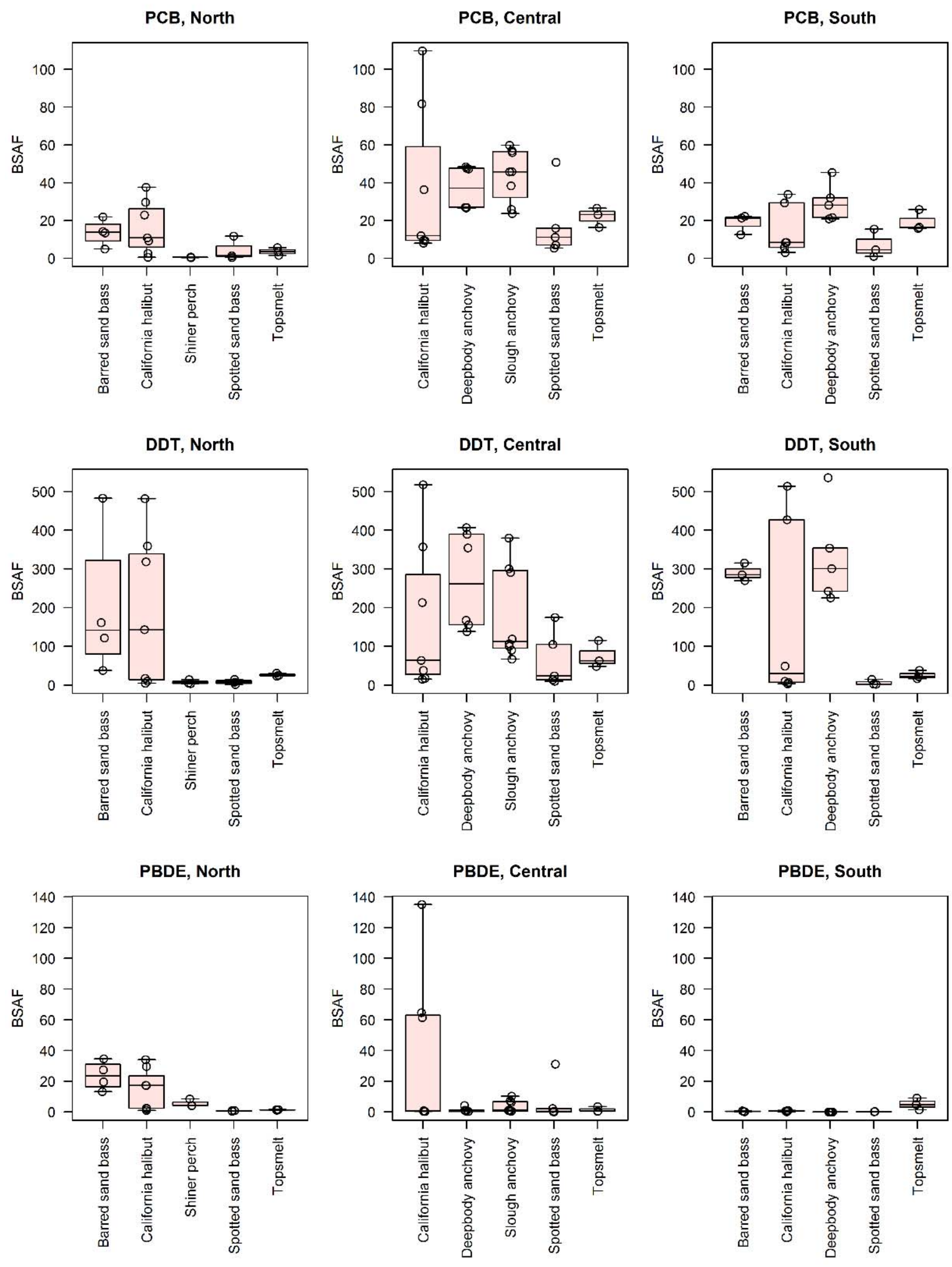

Figure 20. Normalized BSAF values for PCBs, DDTs, and PBDEs in fish. See Figure 5 for description of plot symbols.

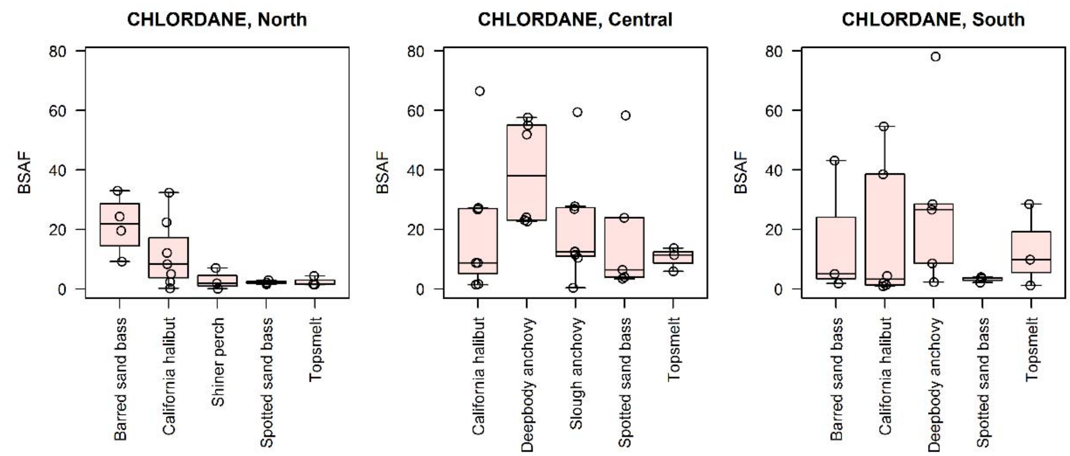

Figure 21. Normalized BSAF values for chlordanes in fish. See Figure 5 for description of plot symbols.

Risk to Wildlife

Representative Species and Exposure Factors

Five avian receptor categories are represented in this assessment. Three are piscivores that consume fish but with different preferences for fish size and/or occurrence in the water column, a fourth category is consumers of benthic invertebrates, and the fifth category is a mixed forager that is a dietary generalist, consuming pelagic aquatic biota, other birds, carrion and human refuse. Of the piscivores, one is a species of conservation concern. Four of the categories are represented by species that nest in colonies around San Diego Bay. Species present during the breeding season rely on aquatic biota from the bay during a critical period in their life cycle, during which adults may experience adverse effects from exposure to contaminants in the diet and offspring may experience adverse effects from exposure as embryos to contaminants that were transferred by the female parent to the egg (Figure 1). Receptors that forage on benthic invertebrates are represented by overwintering species including large numbers of shorebirds, which forage in intertidal habitats and waterfowl that forage in more subtidal areas (Figure 1).

Dietary exposure to contaminants by overwintering adults may adversely affect weight and body condition, which are critical for successful migration to summer breeding grounds. Wintering shorebirds and waterfowl may obtain the nutrition they need to produce eggs at migratory stopovers or at summer breeding grounds. However, birds that start migration with a nutritional deficit (poor body condition) are at risk of reduced survival during migration, and reduced breeding activity and output at breeding grounds. The representative species selected for this assessment are California least tern representing small pelagic foraging piscivores and species of conservation concern (federal and State endangered), Caspian tern representing large pelagic foraging piscivores, double-crested cormorant representing pelagic and bottom foraging piscivores, surf scoter representing bottom foraging consumers of invertebrates, and western gull representing mixed foraging carnivore (fish, invertebrates, birds and refuse).

Species characteristics translate into factors that are used to modify exposure estimates. Seasonal use may be represented by a seasonal use factor (SUF). The seasonal occurrence of receptors at San Diego Bay is noted. However, because representative species are present year-round or at least for months spanning a critical phase of their annual cycle, SUFs were assigned a value of 1.0 (no adjustment to exposure estimate), reflecting the assumption that exposure and potential for adverse effects associated with contaminant exposure while at San Diego Bay is not affected by months spent at other locations. The fraction of food obtained from the site may be represented by an area use factor (AUF). The AUF allows for the possibility that a receptor is obtaining less than 100 percent of its daily intake from the site. For example, a receptor's foraging area may extend beyond the boundaries of San Diego Bay. At 12 miles (19 km) long by 1-3 miles (1.6-4.8 km) wide, San Diego Bay is considered large enough to accommodate foraging ranges of most avian species considered in this assessment, particularly those that are present during the nesting season. The bay also provides a variety of foraging habitats, and abundances of prey species preferred by most of the receptors considered in this assessment (USDoN, SWDIV 2013). With few exceptions, a default value of 1.0 is assigned to the AUF, reflecting the assumption that while present, 100 percent of a receptor's diet is from San Diego Bay.

Daily food ingestion (FI) is species-specific, depends on body mass (hereafter referred to as body weight or BW), and is either measured or estimated using regressions relating FI to BW, depending on available information. Receptor body weights are reported as grams (g) and food ingestion is reported or estimated as grams wet weight (ww) of food per day (g_{food} ww/d). Diet composition affects the contaminant level to which a particular receptor is exposed. For example, contaminant levels in the diet of birds that prefer small pelagic fish may be lower than contaminant levels in the diet of birds that consume larger and higher trophic level fish (e.g., see Figure 1). Diet composition for representative species can be generally described but with variation in specifics, depending on what is available. Aquatic biota collected from San Diego Bay and analyzed for this assessment include invertebrates, and what is generally categorized as small benthic fish, small pelagic fish, and medium size demersal fish (Figure 1). The general diet composition for each representative species was used to select which samples of aquatic biota collected from San Diego Bay to factor into estimates of dietary contaminant levels for the receptor group the species represents. General information and assumptions relating to exposure estimates for each representative species are provided in the following profiles.

Profile for California least tern (small pelagic foraging piscivore, and species of conservation concern)

The California least tern was selected to represent small pelagic foraging seabirds, or obligate piscivores, that rely on bay resources during the nesting season. California least tern is a Federal and State Endangered species, and consequently is a species of conservation concern. The California least tern is the smallest of the North American terns and is found during its nesting season along the Pacific coast of California from San Francisco Bay, California, to Baja California, Mexico. Limited information indicates that the species migrates south to Central and South America for the winter (Massey 1971). California least terns nest in colonies on sandy dunes and flats, or similar habitats created intentionally or unintentionally as a result of human activity. Least tern nesting colonies are found all around San Diego Bay, with colonies on one side or the other in northern, central and southern segments of the bay, and at the Salt Works on the southern end. Only colonies on the eastern side of the bay are represented by eggs collected in 2013, and are of greatest concern for exposure by adults to contaminants from San Diego Bay in their diets.

The California least tern was listed as federally endangered in 1970 and state endangered in 1971, due to population declines from loss of habitat. Subsequent actions to conserve, increase and/or manage habitat have resulted in an increase in statewide numbers from 623 nesting pairs in 1973 (Bender 1974) to approximately 5,000 nesting pairs in 2013. In 2013, approximately 23% of statewide numbers nested in colonies around San Diego Bay (Frost 2014). While overall numbers have increased, the rate of nest success (as number fledged/nest) has been declining, and more so for southern colonies, such as those in San Diego Bay, than for more northern colonies (Lewison and Deutschman 2014). As a species of conservation concern, the California least tern is a high priority species for monitoring numbers and productivity, and for characterizing and managing potential threats to their conservation.

Characteristics that affect potential for exposure

The California least tern is migratory, with individuals typically arriving at nesting sites in April, and departing in late August (TDI 2009 and 2011; USDoN, SWDIV 2013). As such, they are present and rely on resources in San Diego Bay for a critical phase of their life cycle, the breeding period. Because they are present during the nesting season, contaminant exposure by both adults (via diet) and developing embryos (as concentrations in eggs) can be assessed.

California least terns are surface-feeding piscivores that forage in the bay and in nearshore waters of the open ocean (Baird 2010; Leicht 2014; Atwood and Kelly 1984), generally within 3 km of their nest sites (Atwood and Minsky 1983). They are opportunistic, eating fish that are small enough to catch (<6 cm standard length, or SL) and in less than 1 meter of water (Atwood and Kelly 1984). California least terns consume primarily young-of-the-year (YOY) of anchovies (Engraulididae) and silversides (Atherinidae, such as topsmelt), but also YOY of surfperch, sculpin, herring and others, depending on availability (Atwood and Kelly 1984; Leicht 2014; Burton and Terrill 2012; Baird 2010). As consumers of fish, least terns are upper trophic level predators (secondary carnivores) of the aquatic-based food web. Fish consumed by least terns include numerous resident species, some of which are year-round residents and others are juveniles of species that spawn in the bay and are bay-only residents prior to migrating out of the system (Allen et al. 2002). Diet composition may change from one year to the next. However, based on data from Atwood and Kelly (1984), it is assumed that least terns nesting in San Diego Bay colonies, especially ones along the eastern and southern shorelines, rely entirely on resident, juvenile, forage fish. Because of their diet and localized foraging habits during the nesting season, California least terns are likely to be exposed to elevated levels of contaminants from San Diego Bay. Because females may transfer contaminants to eggs and developing embryos, contaminant risk to both adults and embryos are evaluated in this assessment.

Direct exposure by developing embryos

Contaminant exposure by developing California least tern embryos was measured directly using chemical analysis of failed to hatch eggs.

Dietary exposure factors for adults

Adult least terns weigh between 30 and 45 g (Thompson et al. 1997). Dunning (1984) reports a mean adult body weight for least terns in Kansas of 43 g (± 2.12 g), and Massey (1971) determined that the growth of California least tern fledglings starts to level off at 35 to 40 g. Based on data from Massey (1971) specific to California least terns, 40 g may be considered the low end of adult body weights for California least terns, and as such was the value adopted for this assessment. The daily food ingestion rate for California least tern was estimated using the regression developed by Nagy (2001) for Charadriiformes (gulls, terns, shorebirds, auks). The food ingestion rate for adult California least terns was computed as:

$FI\ (g_{food}\ ww/d) = 1.914 \times (g\ BW)^{0.769}$, or
$FI = 32.65\ g_{food}\ ww/d$

Incidental ingestion of sediment is expected to be minimal to none for birds that forage in the water column, and therefore is not considered further for this species. Diet composition is assumed to be forage fish in general that are less than 6 cm SL. Of the species collected from San Diego Bay for this assessment, those less than 6 cm SL and therefore assumed to be part of the least tern diet for estimating contaminant exposure are topsmelt, slough anchovy, northern anchovy, shiner perch, black perch, killifish and gobies. Factors and assumptions used to estimate dietary contaminant exposure by California least terns while nesting in colonies around San Diego Bay are summarized below.

Species	California least tern
BW (g)	40
FI (g_{food} ww/d)	32.65
AUF (unitless)	1
SUF (unitless)	1
Assumed diet composition	topsmelt, slough anchovy, northern anchovy, shiner perch, black perch, killifish and gobies

Profile for Caspian tern (large pelagic foraging piscivore)

The Caspian tern represents large pelagic foraging seabird species. It is the largest of the tern species, and is found on every continent but Antarctica. They nest and winter along coastlines, fresh- and saltwater wetlands, estuaries, coastal bays and beaches. In North America, they breed along the Pacific coast from Alaska to Baja California (Mexico), the Atlantic Coast from Newfoundland to Florida, and the Gulf Coast from Texas to Florida. They also breed in central Canada (Northwest Territories and central provinces), and in north-central United States (USGS 2016). Their wintering range extends from southern California to South America on the Pacific coast and from North Carolina to Central America on the Atlantic and Gulf coasts (USGS 2016). The Salt Works at the southern end is the only location in San Diego Bay where Caspian terns have a nesting colony. With approximately 300 breeding pairs observed annually over decades of surveys, numbers of Caspian terns nesting at the Salt Works are sufficient to be considered continentally significant (USFWS 2006).

Characteristics that affect potential for exposure

While Caspian terns are present throughout the year in San Diego County, the species is mostly a summer visitor to San Diego Bay with greatest numbers occurring during the nesting season from March to September (SDNHM 2016; TDI 2009, 2011). As such, they are present and rely on resources in San Diego Bay for a critical phase of their life cycle, the breeding period. Because they are present during the nesting season, contaminant exposure by both adults (via diet) and developing embryos (as concentrations in eggs) can be assessed.

Similar to least terns, Caspian terns are surface-feeding piscivores that forage in nearshore waters of bays, estuaries, lakes, and the nearby ocean (Cuthbert and Wires 1999; Balz et al. 1979; Ohlendorf et al. 1985). Caspian terns are reported to have a maximum foraging range of up to 70 km, but the composition of their diets generally reflect the most locally abundant and available forage fish species near (within 25 km of) their nesting colonies (Collis et al. 2012; Roby et al.

2002). Caspian terns catch a variety of fish species with shallow plunge dives in water 0.5 to 5.0 meters deep. The diet composition and size of fish captured by Caspian terns vary with location and seasonal prey availability, but most fish are between 5 and 25 cm in length and the fish are older at 1+ yrs (Cuthbert and Wires 1999, Balz et al 1979). Caspian terns nesting in estuarine/marine habitats consume primarily silversides (topsmelt, jack smelt, grunion), surfperches, and anchovies, but also sculpin, gobies, flatfish, juvenile sharks and others, depending on availability (Collis et al. 2012). As consumers of relatively large forage fish, Caspian terns are upper trophic level predators (tertiary carnivores) of the aquatic-based food web that includes numerous fish species found in San Diego Bay. Caspian terns nesting in the San Diego Bay colony have been found to consume a large number of fish species, but topsmelt and other "unidentified atherinids" dominate along with sardines and anchovies (Ohlendorf et al. 1985; Horn and Dahdul 1998 and 1999). The Caspian tern diet is thus a mixture of bay/estuarine residents (e.g, topsmelt, anchovies other than northern anchovy, gobies) and marine species that occur in the ocean nearby, or are seasonally present in San Diego Bay (e.g., northern anchovy and California grunion). Because of their diet and foraging habits during the nesting season, Caspian terns are likely to be exposed to elevated levels of contaminants from San Diego Bay. However, not all of their food is from San Diego Bay. Using data from Horn and Dahdul (1998 and 1999), and assuming that "unidentified atherinids" are mostly topsmelt, it is assumed for this assessment that Caspian tern diet is 5 - 25 cm-long pelagic and demersal forage fish, with only 50% of their daily intake coming from San Diego Bay. Because females may transfer contaminants to eggs and developing embryos, contaminant risk to both adults and embryos are evaluated in this assessment.

Direct exposure by developing embryos

Contaminant exposure by developing Caspian least tern embryos was measured directly by chemical analysis of failed to hatch eggs.

Dietary exposure factors for adults

Adult Caspian terns weigh between 530 and 780 g, with means from 623 to 662 g (Cuthbert and Wires 1999; Dunning 1984). Absent body weight data specific to Caspian terns on the Pacific coast of North America, a low-end mean value reported by Cuthbert and Wires (1999) was selected for use in this assessment. The body weight for adult Caspian terns was thus assumed to be 618 g, which is the mean for birds from Michigan, minus one standard deviation. The daily food ingestion rate for Caspian terns was computed using the regression developed by Nagy (2001) for Charadriiformes (gulls, terns, shorebirds, auks), as follows:

FI (g_{food} ww/d) = 1.914 x (g BW)$^{0.769}$, or
FI = 268 g_{food} ww/d

Incidental ingestion of sediment is expected to be minimal to none for birds that forage in the water column, and therefore is not considered further for this species. Diet composition is assumed to be pelagic/demersal fish between 5 and 25 cm SL. Of the species collected from San Diego Bay for this assessment, those assumed to be part of the Caspian tern diet are topsmelt, slough anchovy, deepbody anchovy, northern anchovy, shiner perch, black perch, barred sandbass and spotted sandbass. Factors and assumptions used to estimate dietary contaminant exposure by Caspian tern adults while nesting in colonies on San Diego Bay are summarized below.

Species	Caspian tern
BW (g)	618
FI (g_{food} ww/d)	268
AUF (unitless)	0.5
SUF (unitless)	1
Assumed diet composition	topsmelt, slough anchovy, deepbody anchovy, northern anchovy, shiner perch, black perch, barred sandbass and spotted sandbass

Profile for Double-crested cormorant (pelagic and benthic-foraging piscivore)

The double-crested cormorant represents large seabird species that forage in mid-water depths and at the bottom of the water column. It is a colonial nesting seabird that occurs generally within sight of land on inland and coastal waterways of North American and northern Mexico (Dorr et al. 2014). Double-crested cormorants breed in Alaska, along the Pacific Coast from British Columbia to Mexico, in interior Canada and northern U.S, and along the Atlantic coast from Newfoundland to New York and Florida to the Caribbean. In the winter, those that breed in the continental interior and along the northern Atlantic coast migrate to southeastern U.S, while cormorants with breeding grounds in the west migrate to the Pacific coast, and some breeding populations do not migrate (Dorr et al. 2014). The double-crested cormorant was listed in 1978 as a California species of special concern (Remsen 1978) due to declines in numbers attributed to human disturbance, loss of nesting habitat, and pollutant impacts on survival and reproduction.

Characteristics that affect potential for exposure

The double-crested cormorant is a common visitor to San Diego County being most abundant during winter months (SDNHM 2016). San Diego Bay is host to a breeding colony that was formed in 1998 at the southern end of the bay (SDNHM 2016). The breeding colony appears to be non-migratory, with surveys showing a year-round presence of a few hundred birds, supplemented in the fall and early winter by seasonal visitors (e.g., TDI 2009 and 2011; USDoN, SWDIV 2013). The breeding population is present and relies on resources in San Diego Bay for a critical phase of their life cycle, if not the entire year. Because they are present during the nesting season, contaminant exposure by both adults (via diet) and developing embryos (as concentrations in eggs) can be assessed.

Double-crested cormorants are piscivores that forage close to shore in water less than 10 meters deep (Dorr et al 2014; Granholm 2016; Anderson et al. 2004). They forage by diving from the water surface and swimming after prey occurring near the surface to just above the bottom (Ainley et al. 1981). Foraging range for double-crested cormorants is typically less than 30 km (Dorr et al 2014), but nest sites tend to be preferentially established within 8 to 10 km of a foraging area (Granholm 2016). Cormorants can be observed in open water throughout San Diego Bay, with greatest numbers occurring during fall in the north, followed by south and south-central segments of the bay (TDI 2009 and 2011). The diet composition and size of fish captured by double-crested cormorants vary with location and seasonal prey availability. However, most fish are slow-moving, schooling, or inshore benthic species, and generally 5-15 cm in length (Dorr et al 2014; Ainley et al. 1981; Robertson 1974). Species found to be common in the diets of double-crested cormorants from colonies along the Pacific Coast include atherinids, midshipmen, shiner perch, anchovy, croaker/seabass/corbina (Ainley et al. 1981), along with benthic fish and invertebrates for birds in inshore habitats (Dorr et al. 2014; Robertson 1974; Trapp and Hanisch 2000). As consumers of fish, double-crested cormorants are upper trophic level predators of the aquatic-based food webs. Fish consumed by cormorants include numerous species that occur in San Diego Bay (Allen et al. 2002). Because of their diet and localized foraging habits during the nesting season, double crested cormorants are likely to be exposed to elevated levels of contaminants from San Diego Bay. Details on the fish consumed by double-crested cormorants from the San Diego Bay are lacking. Based on foraging behavior and food preferences, it is assumed that resident double crested cormorants, especially during the nesting season, obtain 100% of their diet from San Diego Bay and that their diet is a mix of pelagic/demersal/benthic fish 5 - 15 cm SL. Because females may transfer contaminants to eggs and developing embryos, contaminant risk to both adults and embryos are evaluated in this assessment.

Direct exposure by developing embryos

Contaminant exposure by developing double-crested cormorant embryos was measured directly by chemical analysis of failed to hatch eggs.

Dietary exposure factors for adults

The body weight of adult double-crested cormorants ranges from approximately 1,200 to 3,000 grams, depending on gender and location (Dorr et al. 2014). Along the east coast and central North America, males tend to be larger than females, and birds from northern colonies are larger than birds from southern colonies (Dorr et al. 2014). To address uncertainty about the body mass of cormorants from the San Diego Bay nesting colony, a low end value from Dunning (1984) was selected for use in this assessment. Body mass was assumed to be 1,540 g, which is a mean reported by Dunning (1984) for females from a colony in Florida, and is lower than commonly used values of 1,800 to 2,000 g in analyses of double-crested cormorant nutritional requirements (Ridgway 2010). The daily food ingestion rate for double-crested cormorants was computed using a regression and factors recommended by Ridgway (2010) specifically for cormorants during the nesting season. In this case, a regression was used to compute the mass-specific daily energy requirements in kilojoules per day (kJ/d), which was then converted to grams diet/day using an average energy density for fish consumed by cormorants (kJ/g diet) adjusted by assimilation efficiency, as follows:

FI (g_{food} ww/day) = field metabolic rate (kJ/d) ÷ metabolizable energy in diet (kJ/$g_{fish\ ww}$), where;
Field metabolic rate (kJ/d) = 16.69 (BW g)$^{0.651}$, and
Metabolizable energy (kJ/g_{fish} ww) = gross energy (5.42 kJ/g_{fish} ww) x assimilation efficiency (0.8)

For a 1,540 gram double crested cormorant;
FI (g_{food} ww/day) = 1,984 kJ/d ÷ 4.34 kJ/$g_{fish\ ww}$ = 458 ($g_{fish\ ww}$/day)

Because of foraging habits, double-crested cormorants may experience some incidental ingestion of sediment. However, exposure via ingestion of sediment is assumed to be insignificant, given that most of their foraging is in the water column. Their diet is assumed to consist of pelagic/demersal/benthic fish 5 to 15 cm SL. Medium to large benthic invertebrates (primarily crustaceans) may be consumed by cormorants as well. Of the species collected from San Diego Bay for this assessment, those assumed to be part of the cormorant diet are topsmelt, slough anchovy, northern anchovy, deepbody anchovy, black perch, shiner perch, spotted sandbass, striped sandbass, and California halibut. Unfortunately, large crustaceans were not sampled. Factors and assumptions used to estimate dietary contaminant exposure by double-crested cormorants while nesting in colonies on San Diego Bay are summarized below.

Species	Double-crested cormorant
BW (g)	1,540
FI (g_{food} ww/d)	458
AUF (unitless)	1
SUF (unitless)	1
Assumed diet composition	topsmelt, slough anchovy, northern anchovy, deepbody anchovy, black perch, shiner perch, spotted sandbass, barred sandbass, and California halibut

Profile for Surf scoter (benthic-foraging consumer of invertebrates)

The surf scoter is a large sea duck that is common along the Atlantic and Pacific coasts in the winter and breeds on fresh water boreal lakes and tundra of Canada and Alaska (Anderson et al. 2015). San Diego Bay supports continentally significant numbers of surf scoter, and perhaps the largest concentration of the species in its winter range (SDNHM 2016, USFWS 2006). Surf scoters are the most abundant avian species on San Diego Bay with peak numbers observed at any one time of >5,000 during 2006/2007 surveys and >10,000 during 2009/2010 surveys (TDI 2009 and 2011). Appreciable continental declines in numbers have been reported in the past, but there is a great deal of uncertainty in counts (Anderson et al. 2015). Results of waterbird surveys by TDI (2009 and 2011), suggest that numbers in San Diego Bay may be declining with a total of 27,357 birds observed in 2006/2007 versus a total of 14,327 birds observed in 2009/2010, but additional surveys would be required to draw any conclusions. As overwintering waterfowl, surf scoters rely on resources in San Diego Bay for maintaining weight and body condition which are critical for successful migration to summer breeding grounds. Scoters may obtain the nutrition they need to produce eggs at migratory stopovers or at summer breeding grounds. However, birds that start migration with a nutritional deficit are at risk of reduced survival during migration, and reduced breeding activity and output at breeding grounds.

Characteristics that affect potential for exposure

Surf scoters can be seen on San Diego Bay throughout the year, but peak numbers occur between late fall (November) and early spring (March). Non-migratory birds observed during summer months are believed to be non-breeding and immature birds (Anderson et al. 2015). Surf scoters are most often found in central and southern portions of the bay in water that is 1.5 to 11 meters deep (5 to 35 feet; USDoN, SWDIV 2013, SDNHM 2016, TDI 2011). They dive for prey on or near the bottom, eating mostly mussels and other mollusks (Anderson et al. 2015), but they may also consume large quantities of herring roe if present (Lewis et al. 2007). Surf scoters may also incidentally ingest sediment while foraging for food. Because of their feeding habits, surf scoters represent waterfowl that are closely linked through diet to sediment-borne contaminants in San Diego Bay. As consumers of benthic invertebrates, surf scoters are mid-trophic level predators of aquatic food web organisms that are resident in San Diego Bay. Of the avian species considered in this assessment, surf scoters are likely to encounter the highest dietary concentrations of contaminants that are bioaccumulated but not biomagnified in the food web (e.g., PAHs). Because of their diet and foraging habits, surf scoters are likely to be exposed to elevated levels of at least some contaminants from San Diego Bay. It is assumed for this assessment that overwintering surf scoters obtain 100% of their food from San Diego Bay.

Direct exposure by developing embryos

Surf scoters are not present during the breeding season, and therefore direct exposure by developing embryos was not addressed in this assessment.

Dietary exposure factors for adults

Adult surf scoters have an overall average body weight of approximately 1,153 g (Ouellet et al. 2013). However, females are smaller than males (Anderson et al. 2015), with reported average body weights of 900 to 985 g for females and 1050 to 1769 g for males. To address risks to benthic-feeding waterfowl of all sizes, the lowest mean value of 900 g for surf scoters was selected for estimating daily ingestion rates. Absent data specific to the species, a daily food ingestion rate for scoter was computed using the regression developed by Nagy (2001) for carnivorous avian species. The food ingestion rate was computed using the following equation:

FI (g_{food} ww/d) = 3.048 x (g BW)$^{0.665}$, or
FI = 281 g_{food} ww/d

Incidental ingestion of sediment was also considered for surf scoter. In studies by Beyer et al. (2008), incidental ingestion of sediments by surf scoters was equivalent to 8.6% of daily food intake, as grams dry weight. Although estimated from daily food intake, sediment is assumed to have no nutritional value, so that the estimated daily ingestion of sediment is considered to be in addition to the daily food ingestion rate needed to meet nutritional requirements. For this assessment, a factor of 0.086 was applied to a dry food (dw) food ingestion rate (g_{food} dw/d) estimated using the regression from Nagy (2001) for carnivorous avian species as follows:

SI ($g_{sediment}$/d) = 0.086 x [0.849 x (BW g)$^{0.663}$], where
SI = daily sediment ingestion = 6.6 $g_{sediment}$ dry/day

The scoter diet is assumed to consist of benthic invertebrates. Herring roe may be consumed as well, but while herring may spawn in San Diego Bay, roe was not collected for this assessment. Of the species collected from San Diego Bay for this assessment, those assumed to be part of the surf scoter diet are bivalve mollusks (mussels and clams), polychaete worms and benthic crustaceans. Factors and assumptions used to estimate dietary contaminant exposure by surf scoters while wintering in San Diego Bay are summarized below

Species	Surf scoter
BW (g)	900
FI (g_{food} ww/d)	281
SI ($g_{sediment}$ dry/d)	6.6
AUF (unitless)	1
SUF (unitless)	1
Assumed diet composition	Clams, mussels, polychaetes, and crustaceans

Profile for Western gull (mixed foraging carnivore)

The western gull represents large and abundant mixed foraging carnivore. It is a large gull found on the Pacific coast of North America from Vancouver Island to the southern tip of the Baja California peninsula (Mexico) and breeding from northern Washington state to central Baja California (Pierotti and Annett 1995). It is the largest and most numerous of the gull species that occur in San Diego County (SDNHM 2016). The western gull typically breeds in colonies on offshore islands and rocks along the coast (Peirotti and Annett 1995). However, western gulls occur throughout the year in San Diego Bay, and have nest sites at Naval Air Station, North Island, near the mouth of the bay (USDoN, SWDIV 2013). Because they are present during the nesting season, contaminant exposure by both adults (via diet) and developing embryos (as concentrations in eggs) can be assessed.

Characteristics that affect potential for exposure

Western gulls are observed throughout San Diego Bay, but with largest numbers occurring routinely in the north (especially around the bait barge), and occasionally at the Salt Works in the south (TDI 2011). Western gulls are generalist predators of marine pelagic and intertidal fish and invertebrates. They are surface foragers, and when in deeper waters they take prey only in the top 1 to 2 meters (Pierotti and Annett 1995). The composition of their diet is highly variable, and depends on their location, year and season. In general, their primary dietary items are small fish, marine invertebrates (e.g. squid, crabs, and euphausiids) in water less than 2 meters deep, and human refuse. They also consume carrion, and eggs and chicks of other birds. Based on observations of island colonies, as summarized by Pierotti and Annett (1995), fish constitute roughly 60 - 80% of the diet, invertebrates constitute 6-30% of the diet and refuse is less than 10%. The proportion of refuse may be much higher at some locations, or when the supply of other foods is poor, such as during El Nino years. Although their diet is highly varied, western gulls for the most part are upper trophic level predators of aquatic food web organisms. Fish and invertebrates consumed by western gulls include species that occur in San Diego Bay. Consequently, Western gulls are likely to be exposed to elevated levels of contaminants from San Diego Bay in their diet. However, the fraction of diet and contaminant exposure that comes from San Diego Bay is difficult to discern. Absent data specific to San Diego Bay, it is assumed that Western gulls obtain 90 percent of their daily food from San Diego Bay while nesting on North Island, and 10 percent of their daily food from upland sources. Because the gulls are present during the nesting season and that females may transfer contaminants to eggs, contaminant risk to both adults and embryos are evaluated in this assessment.

Direct exposure by developing embryos

Contaminant exposure by developing Western gull embryos was measured directly by chemical analysis of failed to hatch eggs.

Dietary exposure factors for adults

Adult western gulls have an overall average body weight of approximately 1,011 g (Dunning 1984). However, females are smaller than males (Pierotti and Annett 1995), with reported average body weights of 770 to 880 g for females and 930 to 1,136 g for males. To address risks to mixed foragers of all sizes, the lowest mean value of 770 g for western gulls was selected for estimating daily ingestion rates. Absent data specific to the species, a daily food ingestion rate for western gull was computed using the regression developed by Nagy (2001) for Charadriiformes (gulls, terns, shorebirds, auks), as follows:

$FI\ (g_{food}\ ww/d) = 1.914 \times (g\ BW)^{0.769}$, or
$FI = 317\ g_{food}\ ww/d$

Western gulls may experience some incidental ingestion of sediment, but it is assumed to be insignificant for this assessment because much of their foraging while in San Diego Bay is expected to be in the water column. Their diet, while foraging in San Diego Bay is assumed to be fish and medium to large pelagic invertebrates in the upper 2 meters of water. Pelagic invertebrates that might be consumed by western gulls were not collected as part of this study. Of the fish species collected for this assessment, those assumed to constitute the western gull diet are topsmelt, northern anchovy, slough anchovy and deepbody anchovy. Factors and assumptions used to estimate dietary contaminant exposure by western gulls while nesting in colonies on San Diego Bay are summarized below.

Species	Western gull
BW (g)	770
FI (g_{food} ww/d)	317
AUF (unitless)	0.9
SUF (unitless)	1
Assumed diet composition	topsmelt, northern anchovy, slough anchovy and deepbody anchovy

Summed concentrations of mixtures

The PCBs, PBDEs, DDT and metabolites, and chlordanes are chemical classes for which concentrations of multiple individual compounds were measured. While it was desirable to know the measured concentrations of the individual components, it was also desirable to have an estimate of the concentrations of the mixtures as a whole. DDT and its metabolites are typically found together, and consequently referred to collectively as total DDT. Studies on effects of DDT often focus on specific isomers, especially DDE. For this analysis, total DDT is used to characterize DDT exposure, and comparisons with literature-based thresholds presented as total DDT or DDE. The total DDT concentration is the sum of the concentrations of six isomers (o,p- and p,p' DDT, o,p- and p,p' DDE, and o,p- and p,p' DDD). The total chlordane concentration is the sum of the concentrations for major constituents in technical grade chlordane (α- and γ-chlordane, *cis*- and *trans*-nonachlor) and the primary metabolite oxychlordane (as applied by USEPA 1992). Concentrations of total PBDEs were computed as the sum of the detected congeners. Concentrations of total PCBs were computed as the sums of the detected congeners, further adjusted using a regression similar to that used by the National Oceanic and Atmospheric Administration (NOAA 1989 and 1993) for estimating total PCB concentrations from the sum of a fraction of the 209 possible congeners. Egg samples for this study were analyzed for 41 PCB congeners, and as such may underestimate the values for comparison with literature-based total PCB concentrations as estimated using the Aroclor standard approach, or as sums of 90 or more congeners, or sums of homologs. Data from studies by Zeeman et al. (2008) on PCBs in eggs of California least tern, elegant tern, Caspian tern, and black skimmer were analyzed to characterize the relationship between the total PCB concentration as the sum of the 41 congeners (this study), versus total PCB concentrations estimated as; (1) the sum of detected concentrations for 96 congeners, (2) the sum of ten homolog classes, and (3) as estimated using the Aroclor standard approach (Zeeman et al. 2008 study). In the earlier study, total PCB concentrations based on the sum of the concentrations of detected congeners (out of 96) was the same as for the other two measures and as such represents actual total PCB concentrations. The least squares linear regression relating the sum of 41 Bight '13 congeners to total PCBs for black skimmer, elegant tern, Caspian tern and California least tern eggs from San Diego Bay colonies (Zeeman et al. 2008) was used to estimate total PCB concentrations for this study. Total PCB concentrations were computed from sums of concentrations of Bight'13 congeners using:

Total PCBs (µg/kg fw) = 75.2 µg/kg fw + 1.17 (∑µg/kg fw Bight '13 congeners)

Toxic equivalent concentrations (TEQs) of twelve dioxin-like PCB congeners were computed for mixtures of the dioxin-like PCB congeners using Toxic Equivalent Factors (TEFs) for avian species from Harris and Elliott (2011). The concentration of each congener was multiplied by its respective TEF to produce a TCDD equivalent concentration. The TEQ is the sum of the estimated equivalent concentrations of the twelve dioxin-like congeners.

Exposure Point Concentrations (in eggs and diet)

Exposure point concentrations (EPCs) developed for this assessment include those measured in avian eggs, for direct exposure by developing embryos, and those measured in food web organisms, for dietary exposure by adults. Concentrations of all contaminants, other than mercury and PFC compounds are evaluated as mixtures, as described previously.

Concentrations of contaminants measured in avian eggs were evaluated on an egg-by-egg basis. Concentrations of contaminants measured in aquatic biota samples (and sediment) were averaged to reflect what is in the diet of the avian receptors, which is a mix of forage species and individuals within a species. Dietary EPCs used in this assessment are means (to reflect diet composition), and maxima (for worst-case scenario).

Data on concentrations of each contaminant were first grouped by taxon and then by the region of the bay from where samples were collected. Data were summarized (mean, median standard deviation, range and N) in two formats; one in which data are grouped by species only (for baywide EPCs), and the other in which data are grouped by species and bay region (for north, central and south bay EPCs).

The summary data were further grouped to reflect the mixes of species consumed by the different receptors, as described previously. The weighted average and maximum concentrations of contaminants in the mix of species consumed by each receptor were determined. Both bay wide and bay region-scale EPCs were estimated to assess risks to avian species that may forage over the entire bay or that may favor particular regions. Species-specific EPCs are used estimate daily dietary dose rates that are then used to characterize risk, as described in the following sections. The EPCs along with the types and number of samples factored into each EPCs are provided in tables summarizing daily dose and hazard quotient estimates (see Appendix F).

Exposure Estimates (concentrations in eggs and daily dose rates for adults)

Contaminant exposure was characterized two ways; as measured concentrations in eggs, and as estimated daily dietary dose rate, computed from measured concentrations in aquatic biota (EPCs; food items), combined with species-specific food ingestion rates. Each has advantages and limitations.

Concentrations in eggs are directly measured wet weight-based values, adjusted if necessary for loss of moisture in the sample (see egg processing methods). The results are fresh weight-based concentrations ($ng_{contaminant}$ /g_{egg}fw) and are directly comparable to wet weight and fresh weight-based values reported in the literature. Contaminant exposure during embryonic development is considered chronic and concentrations in eggs provide a direct measure of chronic contaminant exposure by developing embryos that can be related to adverse effects specific to embryos (e.g., malformations, edema, embryo lethality, reduced hatchability). Limitations of this approach for assessing contaminant risks are (1) not all contaminants of interest are persistent enough, and /or lipophilic or protein-philic enough to be transferred in significant amounts by the female parent to developing eggs (e.g., PAHs, numerous inorganics, perchlorate), and (2) except for uptake of some compounds from direct contact (e.g., petroleum hydrocarbons), contaminant levels in eggs result from exposure by the female parent only. Levels of some contaminants in eggs have been related to adverse effects in the parents (e.g., DDT). However, for most contaminants, characterizing such relationships quantitatively is difficult, at best. An assessment of contaminant levels in the diet provides an alternate line of evidence on contaminant exposure and risk of impacts in adult birds

Daily ingested dose is an estimate of the daily intake of a contaminant as nanograms per gram of body mass per day ($ng_{chemical}/g_{BW}$-d) by a specific receptor via one or multiple exposure routes. For reasons given earlier (conceptual model), the only exposure route considered in this assessment is ingestion, more specifically ingestion of food and in one case, incidental ingestion of sediment. Daily intake via ingestion was estimated using the following equation;

Daily Intake (ng/g_{BW}-d) = $[(EPC_{diet} \text{ x } FI_w/BW) + (EPC_{sediment} \text{ x } SI_d/BW)] \text{ x } (EMFs)$, where:

EPC_{diet} = Exposure Point Concentration of chemical in food (i.e., ng/g ww)

$EPC_{sediment}$ = Exposure Point Concentration of chemical in sediment (i.e., ng/ng dw)

FI_w = Species-specific fresh matter food intake rate (g_{food}/day)

SI_d = Species-specific dry matter sediment intake rate ($g_{sediment}$/day)

BW = Body weight of receptor (g)

EMF = Exposure modifying factors (unitless; default = 1.0 with few exceptions)

Incidental ingestion of sediment was factored into exposure estimates for one receptor only, as represented by surf scoter. As a routine forager of benthic invertebrates, surf scoter is more likely than the pelagic foragers considered in this assessment to ingest sediment. Because analytical results reported for biota are wet weight-based while results reported for sediment are dry weight-based, daily intake from diet and daily intake from sediment had to be computed separately. Once done, however, daily intake from diet is added directly to daily intake from sediment for an estimated total daily intake by ingestion. Ingestion rates, which reflect daily nutritional requirements, vary among species and with body weight. Assumptions and calculations used to derive values for FI_w and SI_d are provided with the profiles of representative species discussed earlier.

Of the EMFs, those that are most commonly used are the SUF and the AUF, as previously described (see representative species profiles). A third commonly used EMF is the fraction of ingested contaminant that is absorbed into the receptor's system (i. e., absorbance factor or ABS). Percent absorption of a contaminant from ingested food is compound-specific. For this assessment, percent absorption was assumed to be 100% (factor of 1.0) in all cases, which is a conservative default assumption used in the absence of data to support the use of a lesser value.

As described previously, both a mean and a maximum contaminant-specific EPC was developed for each of the representative species, and assuming bay-wide or regional use of the bay for foraging. All EPCs were entered into estimates of dose rates, resulting in two estimates of dietary exposure for each species (mean daily dose rate and maximum daily dose rate), given use of the entire bay or of specific bay regions for foraging. Estimated daily doses, with corresponding EPCs and assumptions are provided in tables summarizing daily dose and hazard quotient estimates (See Appendix F).

Selection of Reference Values to Characterize Risk

Data on contaminant levels in avian eggs were compared with literature-based concentrations associated with adverse effects (Table 8). As indicated previously, contaminant concentrations in eggs are reported as ng/g fw. Risks posed to adults by dietary exposure to contaminants are assessed by comparing species-specific estimated daily dose rates ($ng_{chemical}/g_{BW}$-d) with literature based reference dose rates (also as $ng_{chemical}/g_{BW}$-d), or Toxicity Reference Values (TRVs) for avian species (Table 9). A description of the process used to determine contaminant threshold concentrations in eggs and TRVs is contained in Appendix D.

Risk Characterization Protocols

Risks posed to developing seabird embryos by direct exposure to contaminants and by adults through dietary exposure to contaminants were evaluated in two phases. The first is an initial screen in which worst case comparisons are made to identify those contaminants that are of concern and those that are not. The second is a refined assessment that factors in site-specific conditions, and consideration of multiple reference values for perspective on the likelihood that adverse effects may actually occur.

Initial Screen

Study results were subjected to an initial screen to help focus the assessment, by filtering out contaminants and/or receptors for which concerns are minimal to none. For developing embryos, the maximum concentration observed in eggs of each species were compared with lowest NOAECs from Table 8. Screening was done on a contaminant-by-contaminant basis, and concentrations in eggs were evaluated in greater detail for any of the species for which the maximum concentration exceeded the lowest NOAEC.

Table 8. Screening levels used to evaluate contaminant levels in seabird eggs collected from San Diego Bay colonies in 2013.

NOAEC/LOAEC	(ng/g_{egg}fw)	Contaminant (effects)	Source
Mercury (egg hatchability, embryo mortality)			
NOAEC	300	estimated from LOAEC (egret)	Shore et al 2011
LOAEC(s)	600	5^{th} percentile all species (<lowest LOAEC for all	Shore et al 2011
	800	Low (snowy egret; field based)	Shore et al 2011
	1,300	Mid (common loon; field based)	Shore et al 2011
	3,700	High (common tern; field based)	Shore et al 2011
DDTs (productivity, eggshell thinning)			
NOAEC	200	eggshell thinning	Blus 2011
	1,000	reduced productivity	Blus 2011
LOAEC(s) – prod.	3,000	sensitive species (brown pelican)	Blus 2011
	5,000	Mid-range (e. g. double-crested cormorant & Caspian	Blus 2011
LOAEC(s) -	600	Sensitive species (pelican)	Blus 2011
	10,000	Mid-range (e. g. double-crested cormorant)	Blus 2011
Total PCBs (productivity, parental behavior)			
NOAEC	100	Adjusted LOAEC for most sensitive species	Harris & Elliott 2011
	2,300	Adjusted LOAEC for waterbirds and raptors[1]	Harris & Elliott 2011
LOAEC(s)	1,000	Most sensitive species (chickens)	Harris & Elliott 2011
	6,000	Medium sensitivity species (perching birds)	Harris & Elliott 2011
	23,000	Low sensitivity species (terns, gulls raptors)	Harris & Elliott 2011
PCB 126 (embryo lethality)			
NOAEC	0. 11	Adjusted LOAEC for most sensitive species (chickens)	
	6. 5	Adjusted LOAEC seabirds and raptors	Harris & Elliott 2011
LOAEC(s)	1. 1	Highly sensitive species (chickens)	Harris & Elliott 2011
	24. 0	Medium sensitivity species (bobwhite quail)	Harris & Elliott 2011
	65. 0	Least sensitive species (cormorant, tern, raptor)	Harris & Elliott 2011
TEQ (embryo lethality)			
NOAEC	0. 018	Adjusted LOAEC for most sensitive species	Harris & Elliott 2011
	0. 4	Adjusted LOAEC for waterbirds	Harris & Elliott 2011
LOAEC(s)	0. 18	Highly sensitive species (chickens)	Harris & Elliott 2011
	1. 0	Medium sensitivity (pigeon, pheasant, quail)	Harris & Elliott 2011
	4. 0	Low sensitivity (cormorant, heron, wood duck)	Harris & Elliott 2011
PBDEs (egg hatchability, fertility of offspring)			
NOAEC	180	Bounded NOAEC sensitive species	McKernan et al 2009
LOAEC(s)	288	sensitive species	McKernan et al 2009
PFOS (offspring survival)			
NOAEC	1,000	Adjusted LOAEC for more sensitive of two species	Newsted et al 2005
LOAEC(s)	62,000	More sensitive of two species (mallard less sensitive)	Newsted et al. 2005

1. All but most sensitive (chickens) (including terns, gull and raptors)

Table 9. Toxicity Reference Values (TRVs) - dose rates used to evaluate risks posed by contaminants in the diet to aquatic-dependent birds of San Diego Bay

NOAEL/LOAEL	(ng/g_{BW}-d)	Contaminant (most sensitive effects)	Source
Mercury (reproduction, parental behavior, productivity)			
NOAEL	4. 0	5th percentile NOAEL all species	Zhang et al. 2013
	7. 0	Sensitive species	USFWS 2003
	21	Less sensitive species (obligate piscivores – seabirds)	USFWS 2003
LOAEL	10	Most Sensitive species (ibis)	Zhang et al. 2013
	180	Mid-range (all species considered; based on mallard)	DTSC/HERD 2009
Total DDTs (based on productivity, survival and growth)			
NOAEL	9. 0	most sensitive species (pelican)	DTSC/HERD 2009
	227	less sensitive species	EPA 2007
LOAEL	27	most sensitive species	EPA 1995
	1,500	Mid-range all species	DTSC/HERD 2009
Total PCBs (based on egg production, fertility, and hatchability)			
NOAEL	90	Estimated for most sensitive species	DTSC/HERD 2009
LOAEL	1,270	Mid-range, all species (mostly non-waterbirds)	DTSC/HERD 2009
TEQ (based on egg production, hatchability)			
NOAEL	0. 0011	Estimated, for most sensitive species (chickens)	Su et al 2014
LOAEL	0. 0495	Lowest for species other than most sensitive	Su et al 2014
	0. 178	Mid-range for all species considered (incl. mallards)	Su et al 2014
PBDEs (reproductive behavior, egg quality, productivity)			
NOAEL	9. 6	Adjusted LOAEL- sensitive species[1]	Fernie et al 2009
LOAEL	96	Sensitive species[1]	Fernie et al 2009
Chlordanes (survival)			
NOAEL	160	sensitive species[2]	Stickel et al. 1983
LOAEL	7,000	Sensitive species[2]	Stickel et al. 1983
LPAHs (weight and food consumption)			
NOAEL	295	Sensitive effect in species of unknown relative sensitivity	Klasing 2007
LOAEL	4,730	Sensitive effect in species of unknown relative sensitivity	Klasing 2007
HPAHs (infertility)			
NOAEL	14.3	Sensitive effect in species of unknown relative sensitivity	Hough et al. 1993
LOAEL	1,430	Sensitive effect in species of unknown relative sensitivity	Hough et al. 1993

1. Only kestrels, mallards and terns studied (mallards and terns ≤ sensitivity of kestrels; Rattner et al. 2011)
2. Based on acute toxicity tests with multiple upland species and mallards (Eisler 1990)

For dietary exposure by adult seabirds and waterfowl, maximum concentrations of contaminants in aquatic biota were compared with risk-based dietary screening levels estimated for each representative species using lowest NOAELs in Table 9 and species-specific food ingestion rates provided with species profiles. Screening levels were computed for each contaminant and representative species as follows:

Screening level (ng/g ww) = selected TRV ÷ (FI/BW), where

Screening level = concentration in the avian diet (aquatic biota)

The selected TRV = lowest contaminant-specific NOAEL ($ng_{contaminant}/g_{BW}$-d)

FI = species-specific fresh food ingestion rate (g_{food} fw/d), and

BW = species-specific body weight (g)

The resulting initial dietary screening levels (Table 10) are conservative, in that they are based on lowest NOAELs. Also, while they are species-specific, they do not take into consideration site-specific exposure modifying factors (e.g., AUFs) and, as applied, there are no assumptions about diet composition.

Table 10. Initial risk-based screening levels for contaminants in the diet (aquatic biota) of selected avian species that forage in San Diego Bay.

Receptor category →	forager on benthic invertebrates	small piscivore pelagic	large piscivore pelagic/demersal	Large piscivore all depths & benthic	Generalist pelagic & terrestrial
Representative sp. →	surf scoter	CA least tern	Caspian tern	D-C cormorant	Western gull
Analyte ↓	Screening concentrations (ng/g ww)				
Mercury	12.8	4.9	9.2	13.5	25.2
DDTs	29	11	21	30	57
PCBs	288	110	207	303	566
PCB TEQ	0.0035	0.0014	0.0025	0.0037	0.0069
PBDEs	31	12	22	32	60
Chlordanes	513	196	369	539	1,006
LPAHs	946	362	680	993	1,855
HPAHs	46	18	33	48	90

For the initial screen, the maximum concentration of each contaminant measured in aquatic biota collected from San Diego Bay was compared with the risk-based screening levels in Table 10. Contaminants for which the maximum concentration exceeded any of the avian dietary risk-based screening levels were evaluated in greater detail as part of the refined risk characterization. Conversely, contaminants for which no samples exceeded screening levels were deemed to be of minimal to no concern, and not considered further.

Contaminants that exceed initial screening levels are defined as contaminants of potential concern (COPCs) and are evaluated further in the refined risk characterization.

Refined Risk Characterization

Only contaminants identified as COPCs from the initial screen were subject to more detailed analysis of risks to avian receptors. The refined assessment is more site-specific and/or includes consideration of the potential for adverse effects occurring.

For contaminant levels in seabird eggs, the more refined assessment simply entailed comparisons with multiple screening levels, including LOAECs to provide perspective on the likelihood that impacts on organismal functions related to survival, growth and reproduction will occur.

For risks to adults from dietary exposure, estimated dose rates were compared with TRVs using the hazard quotient approach (HQ). The hazard quotient is a unitless value computed as follows:

HQ = species-specific daily dose rate for an individual contaminant / reference dose rate (TRV) for the same contaminant.

As described earlier (*exposure estimates*), dose (intake) rates were calculated using data on contaminant levels in food web organisms from San Diego Bay, combined with species-specific assumptions about food preferences and use of San Diego Bay for foraging. The resulting dose rates are both species- and site-specific. Mean and maximum EPCs were used to derive mean and maximum dietary dose rates for each receptor exposed to each contaminant of concern in food web organisms, and given that the receptor may use the entire bay for foraging (bay-wide estimates) or limits foraging to any of three sub-regions within the bay (North, Central or South).

All of the estimated dose rates (means and maxima; baywide or by region) for each contaminant were divided by multiple TRVs from Table 9, resulting in a range of HQs for each representative species. An HQ>1 is considered to be of concern. However, the level of concern depends in part on the extent to which the HQ exceeds a value of 1 and the conservativeness of the exposure estimate, and/or the TRV. Using HQs computed for one exposure dose but with multiple TRVs provides some perspective on the potential for adverse effects to occur or be detected under field conditions.

Results and Discussion

Initial Screen

Contaminant levels detected in seabird eggs are summarized in sections on individual contaminants. However, results of the initial screen (Table E1, Appendix E) are summarized below (Table 11) and indicate that, based on potential for adverse effects in developing embryos, that mercury, DDTs, PCBs (total PCBs and TEQs), and PBDEs are contaminants of potential concern for at least one of the seabird species sampled (Caspian tern), and that PCBs are a contaminant of potential concern for all of the species sampled. Of the remaining target analytes, there are no initial determinations for chlordane (no applicable screening levels), PAHs (not analyzed in eggs), or PCB 126 which was detected infrequently but is captured in the TEQ.

Contaminant levels in aquatic biota, and how mean and maximum concentrations compared with NOAEC-based dietary screening levels for avian receptors are summarized, by taxon, in Appendix E (Tables E2a-E2f). Overall results of the initial screen (Table 12) indicate that mercury, DDT, PCBs (as total PCBs and TEQs) and PBDEs in aquatic food web organisms are contaminants of potential concern for most if not all avian receptor categories and that HPAHs are contaminant of potential concern for avian receptors that consume benthic invertebrates. Chlordane and LPAHs were below levels of potential concern for all avian receptors.

Table 11. Results of initial screen for contaminants of potential concern (identified by a ✓) in San Diego Bay based on maximum concentrations in seabird eggs.

Species →	California least tern	Caspian tern	Double-crested cormorant	Western gull
Analyte* ↓				
Mercury		✓		
DDTs		✓	✓	✓
PCBs	✓	✓	✓	✓
PCB 126*	(with TEQ)	(with TEQ)	(with TEQ)	(with TEQ)
TEQ	✓	✓	✓	✓
PBDEs		✓	✓	✓
PFOS				
Chlordanes*	uncertain	uncertain	uncertain	uncertain
PAHs*	not analyzed	not analyzed	not analyzed	not analyzed

* Samples were analyzed for chlordanes and PCB 126, but results were not amenable to screening (Table E1). PAHs are shown because they are considered in other parts of the risk assessment, but eggs were not analyzed for PAHs.

Table 12. Results of initial screen for contaminants of potential concern (identified by ✓)* for avian species that forage in San Diego Bay, based on maximum concentrations in aquatic food web organisms (fish and/or benthic invertebrates).

Receptor group (representative species) →	benthic feeding waterfowl (surf scoter)	small pelagic piscivore (California least tern)	large pelagic piscivore (Caspian tern)	large piscivore pelagic/demersal/benthic (double-crested cormorant)	generalist aquatic/non-aquatic (Western gull)
Analyte ↓					
Mercury	✓	✓	✓	✓	✓
DDTs	✓	✓	✓	✓	-
PCBs	✓	✓	✓	✓	-
PCB TEQs	✓	✓	✓	✓	✓
PBDEs	✓	✓	✓	✓	✓
Chlordane	-	-	-	-	-
HPAHs	✓	✓**	✓**	✓**	✓**
LPAHs	-	-	-	-	-

* **-** denotes no exceedances, ✓ denotes exceedance by maximum concentration.

** Exceedances occur in benthic invertebrates only

Based on the initial screen, COPCs for seabirds and waterfowl that forage in subtidal areas of San Diego Bay are mercury, DDT, PCBs (total and TEQs), PBDEs and HPAHs.

Refined Assessment

Exposure and potential risks posed to avian receptors by direct exposure by embryos (in eggs) and dietary exposure by adults to COPCs are summarized on a contaminant-by-contaminant basis in the following sections. Summary data on contaminant levels in eggs and HQ estimates are provided with the discussions. Tables showing more detail on dietary exposure and risk calculations are provided in Appendices F and G.

Mercury

Mercury concentrations in seabird eggs are summarized below (Table 13). Estimated dietary mercury concentrations and HQs for adult avian receptors are provided in Tables F1 and G1, and summarized below.

Mercury concentrations in eggs

Mercury concentrations in eggs were highest for Caspian terns, intermediate for California least terns, and lowest for cormorants and western gulls (Table 13). The most conservative screening level for mercury in eggs is 300 ng/g fw, which was exceeded by concentrations in eggs of Caspian terns only. The mean mercury concentration in Caspian tern eggs is between the NOAEC and a low-end LOAEC for avian species in general (600 ng/g fw). Concentrations in all of the Caspian tern eggs are well below a LOAEC (3,700 ng/g fw) for reproductive effects in common tern (Figure 22). Although a conservative screening level is exceeded by mercury concentrations in Caspian tern eggs, the likelihood of detecting mercury-related adverse reproductive effects in the field may be low.

Table 13. Mercury concentrations in seabird eggs collected from San Diego Bay nesting colonies, 2013

		Hg (ng/g fw)				
Species	Location	Mean	SD	N	Min	Max
California least tern	Salt Works	156	52	3	108	211
California least tern	CVWR	158	40	4	112	209
California least tern	D Street	183	33	5*	130	206
California least tern	Lindbergh	224	39	5	176	270
Caspian tern	Salt Works	451	204	10	326	1,020
Double-crested cormorant	Salt Works	71	56	8	31	192
Western gull	NAS NI	61	37	8	18	126

* Data missing for one sample

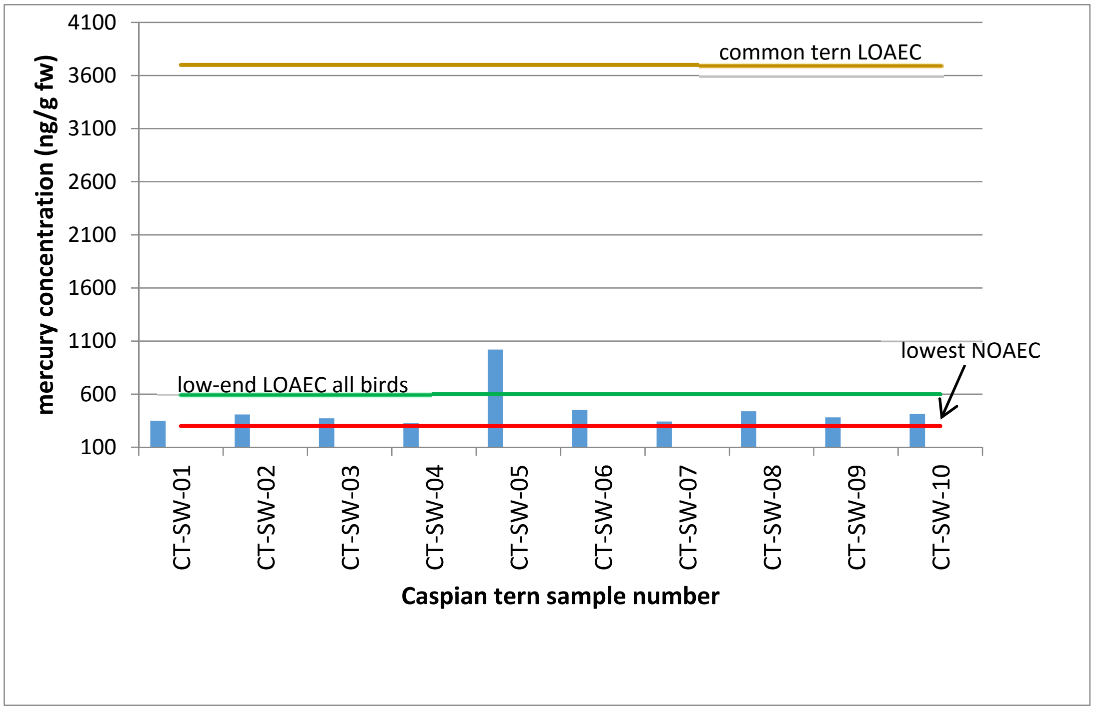

Figure 22. Mercury concentrations (ng/g fw) in individual Caspian tern eggs collected from the South San Diego Bay Salt Works, 2013.

Dietary exposure to mercury

Both mean and maximum concentrations of mercury in aquatic biota exceeded initial dietary screening levels (5-25 ng/g ww) for all of the avian receptors considered in this assessment (Table E2a). Hazard quotients estimated for mean receptor-specific EPCs and assuming bay-wide use by avian receptors (Table 14 and Table F1) appear to support observations of the initial screen. Hazard quotients derived for individual regions demonstrate no particular regional differences (Table G1).

Table 14. HQs for dietary exposure by avian species to mercury in aquatic biota from San Diego Bay, based on mean EPCs and assuming birds forage throughout the bay.

Representative species	HQ - NOAEL (lowest)	HQ - NOAEL (sensitive)	HQ - NOAEL (less sensitive)	HQ - LOAEL (most sensitive)	HQ - LOAEL (mid-range)
Least tern	6.0	3.4	1.1	2.4	0.13
Caspian tern	2.8	1.6	0.5	1.1	0.06
Cormorant	3.8	2.2	0.7	1.5	0.09
Scoter	5.4	3.1	1.0	2.2	0.12
Gull	3.5	2.0	0.7	1.4	0.08

HQs derived from TRVs for sensitive species are >1.0 for all four representative species. Comparisons between species suggest that the risk of adverse effects may be greatest for small piscivores and for waterfowl that forage on benthic invertebrates. Having NOAEL-based HQs >1.0 indicates that risks posed by mercury to avian receptors are of concern. HQs derived from LOAELs for most sensitive species are also >1.0, suggesting that avian receptors may experience adverse effects associated with mercury in their diet. However, HQs derived from a mid-range LOAEL, even those derived from maximum EPCs are all well below 1.0 (Tables F1 and G1) and therefore if there are adverse effects, they may be difficult to detect, especially under field conditions.

Seabirds such as terns are considered to be relatively insensitive to mercury toxicity (Shore et al. 2011), so that HQs derived from LOAELs for sensitive species probably overstate the risk to seabirds, and while some risk is indicated, the likelihood of observing detectable adverse effects in seabirds is low. The potential for adverse effects to occur in other more sensitive piscivorous species (e.g., egrets) was not characterized in this assessment, partly because marsh and intertidal habitats were not sampled. Results obtained with estimates for seabirds, combined with LOAELs for sensitive species, suggest that the potential for mercury to adversely affect more sensitive piscivorous species cannot be discounted.

As previously stated, HQs suggest that mercury poses some risk of adverse effects in waterfowl that forage on benthic invertebrates, and waterfowl are more sensitive to mercury toxicity than seabirds (Shore et al. 2011). However, it is noted that the TRVs for mercury are based on effects that occur from exposure during the breeding season, which the representative species (surf scoter) does not spend in San Diego Bay. For mercury, reproductive effects are most sensitive. Other effects such as impacts on growth and neurotoxicity may occur in birds exposed outside the breeding season to mercury at doses greater than those used as benchmarks in this assessment. The potential for adverse effects of mercury on benthic-feeding waterfowl cannot be completely ruled out.

Overall, LOAEL-based HQs >2.0 for surf scoter and for the least tern (a species of conservation concern), indicate that mercury in benthic invertebrates and forage fish poses some risk of adverse effects in those two representative species, and probably in other species that are more sensitive than seabirds that use the bay. Consequently, mercury is considered to be an ongoing contaminant of concern, but the likelihood of observing detectable effects in the representative species is low. Ongoing monitoring, with some additional focus on risks to especially sensitive waterbird species and species that forage on benthic invertebrates is recommended.

DDTs

DDT Concentrations in seabird eggs are summarized below (Table 15), while details on dietary concentrations and HQs for adult avian receptors are provided in Tables F2 and G2.

DDT concentrations in eggs

Results of analyses for DDT and metabolites are presented as total DDTs. The metabolite, p,p'-DDE contributed more than 97 percent to total DDTs in all of the seabird egg samples. Total DDT concentrations (ng/g fw or ppb fw) were highest for Caspian tern and double-crested cormorant, intermediate for western Gull, and lowest (below all thresholds) for California least tern (Table 15).

Table 15. Total DDT concentrations (ng/g fw) in seabird eggs collected from San Diego Bay colonies, 2013.

		Total DDTs (ng/g fw)				
Species	Location	Mean	SD	N	Min	Max
California least tern	Salt Works	133	50	3	88	187
California least tern	CVWR	101	18	4	83	120
California least tern	D-Street Fill	66	31	6	43	127
California least tern	Lindbergh field	118	46	5	71	193
Caspian tern	Salt Works	1,478	866	10	511	2,766
Double-crested cormorant	Salt Works	1,276	1,096	8	294	3,644
Western gull	NAS NI	426	270	8	128	1,003

Although DDT concentrations in western gull eggs exceed the estimated NOAEC for eggshell thinning, they are below NOAECs and LOAECs for reduced productivity in both sensitive and less sensitive species (Figure 23).

Mean DDT concentrations in Caspian tern and cormorant eggs exceeded low-end screening levels for eggshell thinning and an estimated NOAEC of 1.0 µg/g fw for reduced productivity. The maximum DDT concentration observed in one cormorant egg exceeded a level associated with reduced productivity in brown pelican, an especially sensitive species. However, for the majority of samples, DDT concentrations in individual Caspian tern and cormorant eggs are below literature-based LOAECs for eggshell thinning and reduced productivity in "less sensitive" species (including cormorants and Caspian terns). Whether shells of Caspian tern or cormorant eggs collected for this study show signs of thinning is yet to be determined. Based on comparisons with literature-based values, exceedances of NOAECs may indicate some risk of DDT-related effects in Caspian terns and cormorants nesting at San Diego Bay, but the likelihood of actually detecting impacts may be low.

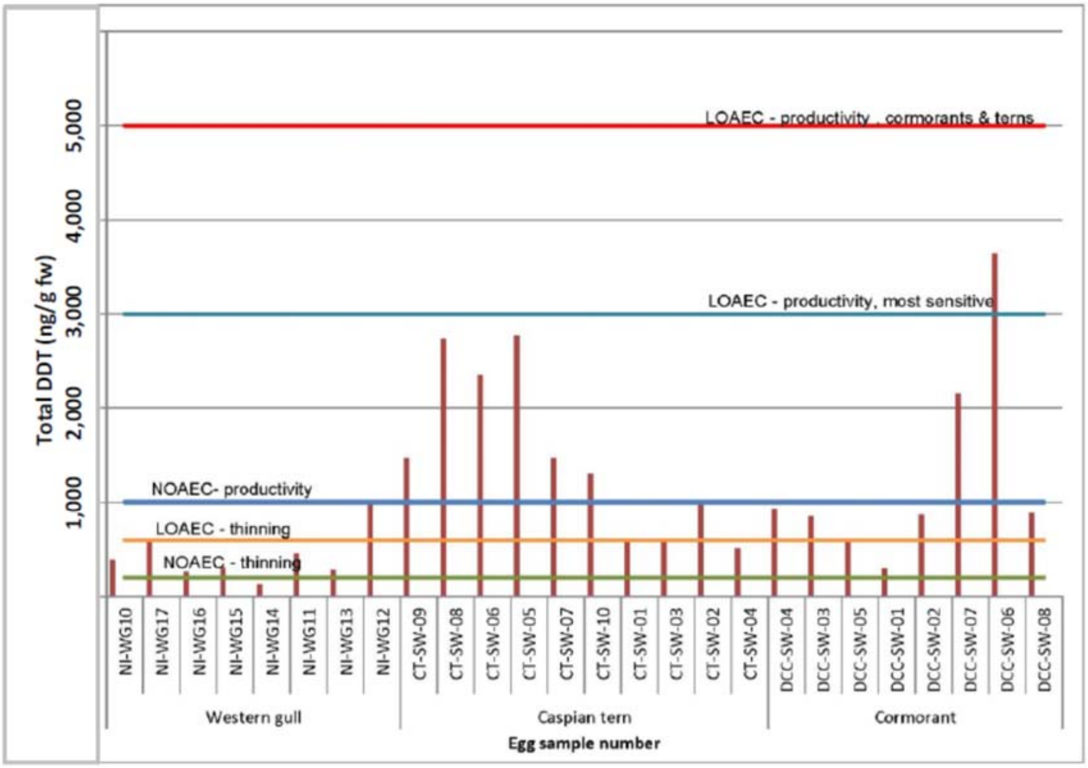

Figure 23. DDT concentrations (ng/g fw) in individual Caspian tern, double-crested cormorant and Western gull eggs collected from the San Diego Bay colonies, 2013.

Dietary exposure to DDTs

Both mean and maximum concentrations of DDT in some aquatic food web components exceeded initial dietary screening levels (11-21 ng/g ww) for two of the piscivorous avian receptors considered in this assessment (Table E2b), and mostly for the small piscivore (least tern). For receptors other than the small piscivore evaluated here, screening level exceedances are by maximum concentrations only in a few (if any) of the aquatic biota samples (Table E2). Based on the initial screen, it appears that only small piscivores are at risk of potential adverse effects from exposure to DDTs in aquatic biota from San Diego Bay.

Hazard quotients estimated for mean receptor-specific EPCs, which factor in dietary preferences, and assuming bay-wide use by avian receptors (Table 16 and Table F1) indicate that small piscivores may be at greater risk of effects than the other receptor groups. However, they also indicate that, with dietary preferences factored in to the exposure estimates, risks to even the small piscivores are below levels of concern (HQ<1.0; Table 16). Hazard quotients for the least tern might have been higher if results obtained with deepbody anchovy were included in EPC calculations for that representative species. However, presumably because of the body shape, deepbody anchovy is not a preferred prey species for the least tern (Atwood and Kelly 1984). Hazard quotients for DDTs may vary with the region in which the species are foraging, with highest occurring in the north where the HQ based on the mean EPC for least tern combined with the lowest NOAEL-based TRVs is 1.12 (Table G2). HQs computed from maximum EPCs are all less than 3.0 when based on the lowest NOAEL-based TRV, and like those computed from mean EPCs (Table 16), they are less than 1.0 when based on NOAELs for less sensitive species and LOAELs for most sensitive species.

Table 16. HQs for dietary exposure by avian species to DDTs in aquatic biota from San Diego Bay, based on mean EPCs and assuming birds forage throughout the bay.

Representative species	HQ - NOAEL (lowest)	HQ - NOAEL (less sensitive)	HQ - LOAEL (most sensitive)	HQ - LOAEL (mid-range)
Least tern	0.93	0.04	0.31	0.006
Caspian tern	0.32	0.01	0.11	0.002
Cormorant	0.43	0.02	0.14	0.003
Scoter	0.23	0.01	0.08	0.001
Gull	0.55	0.02	0.18	0.003

DDT concentrations in aquatic food web organisms from San Diego Bay may have exceeded the screening level based on the lowest NOAEL for one of the representative species, but overall the HQs that reflect the potential for adverse effects in adult birds exposed to DDT in aquatic biota from San Diego Bay are below levels of concern.

PCBs and PCB-TEQs

PCB and PCB-TEQ concentrations in seabird eggs are summarized below (Tables 17 and 18), while details on dietary concentrations and HQs for adult avian receptors are provided in Tables E2c, E2d, F3, F4, G3, and G4 with results summarized below.

PCB (total, PCB 126, and TEQ) concentrations in eggs

Based on concentration alone, PCBs are the primary contaminant of potential concern, followed by DDT, for California least tern and western gulls nesting at San Diego Bay colonies (Tables 15 and 17). For Caspian terns and cormorants, PCB concentrations are second only to DDTs. While maximum total PCB concentrations exceed a LOAEC of 1,000 ng/g fw for reproductive effects in highly sensitive species (chicken), they are below the estimated NOAEC for a sensitive reproductive effect in water birds (2,300 ng/g fw), and are well below the low-end LOAEC of 23,000 ng/g fw for embryo lethality in gulls, terns and raptors (Table 8). Alone, total PCB concentrations in sampled eggs are below levels of concern for waterbirds. However, PCBs may act in combination with other similarly acting contaminants, most notably dioxins/furans (which were not analyzed in this study) and PBDEs. In some Caspian tern and cormorant eggs, the occurrence of dioxins/furans and/or PBDEs may raise the potential for PCB-related reproductive effects to a low level of concern, in which a no effect-based screening level may be exceeded but an actual measured effect level is not.

Table 17. Total PCB concentrations (ng/g fw) in seabird eggs collected from San Diego Bay colonies, 2013.

		Total PCBs (ng/g fw)				
Species	Location	Mean	SD	N	Min	Max
California least tern	Salt Works	296	81	3	205	361
California least tern	CVWR	316	90	4	260	431
California least tern	D-Street Fill	241	99	6	150	408
California least tern	Lindbergh field	384	277	5	232	877
Caspian tern	Salt Works	636	315	10	231	1,276
Double-crested cormorant	Salt Works	927	837	8	276	2,358
Western gull	NAS NI	599	273	8	370	1,072

Concentrations of the four PCB congeners with the greatest potency for dioxin-like toxicity (77, 81, 126 and 169) were generally below the limits of detection (<0.05 ng/g ww). PCB 77 was the most frequently detected. Out of 44 samples, PCB 77 was detected in 28 samples (0.41 - 3.23 ng/g fw), PCB 126 was detected in four (0.5 - 3.3 ng/g fw), PCB 81 was detected in two (0.64 and 0.84 ng/g fw), and PCB 169 was detected in only one (0.37 ng/g fw). Observed concentrations of PCB 126 may exceed a threshold for embryo lethality in highly sensitive species (1.1 ng/g fw), but are below the estimated NOAEC (7.2 ng/g fw) for embryo lethality in waterbirds.

Of the PCB congeners with sufficient potential to cause dioxin-like toxicity to assign a TEF, PCBs 118 and 156 were detected most frequently and at highest concentrations. Concentrations of these and ten other congeners with dioxin-like toxicity were factored into estimates of total dioxin-like PCB TEQs (Table 18). Mean TEQ concentrations may be greater than the screening value for enzyme induction (0.03 ng/g fw; Harris and Elliott 2011), but are less than the threshold for embryo lethality (0.18 ng/g fw) in highly sensitive species. Based on comparisons with thresholds for highly sensitive species, TEQ concentrations in at least some individual seabird eggs may exceed levels of concern for embryo lethality (>0.18 ng/g fw). For species other than the most sensitive, TEQ concentrations in seabird eggs are below any thresholds associated with embryo lethality (Figure 24). As with total PCBs, in some small percentage of Caspian tern or cormorant eggs, the occurrence of dioxins/furans may raise the potential for dioxin-like reproductive effects to a low level of concern, in which a no effect-based screening level may be exceeded but an actual measured effect level is not.

Table 18. Dioxin-like PCB TEQs (ng/g fw) in seabird eggs collected from San Diego Bay colonies, 2013.

		Dioxin-like PCB TEQ (ng/g fw)				
Species	Location	Mean	SD	N	Min	Max
California least tern	Salt Works	0.104	0.152	3	<0.005	0.278
California least tern	CVWR	0.065	0.087	4	<0.005	0.193
California least tern	D Street fill	0.035	0.040	6	<0.005	0.107
California least tern	Lindbergh	0.035	0.039	5	<0.005	0.088
Caspian tern	Salt Works	0.100	0.103	10	0.001	0.331
Double-crested cormorant	Salt Works	0.037	0.032	8	0.001	0.082
Western gull	NAS NI	0.028	0.023	8	0.002	0.058

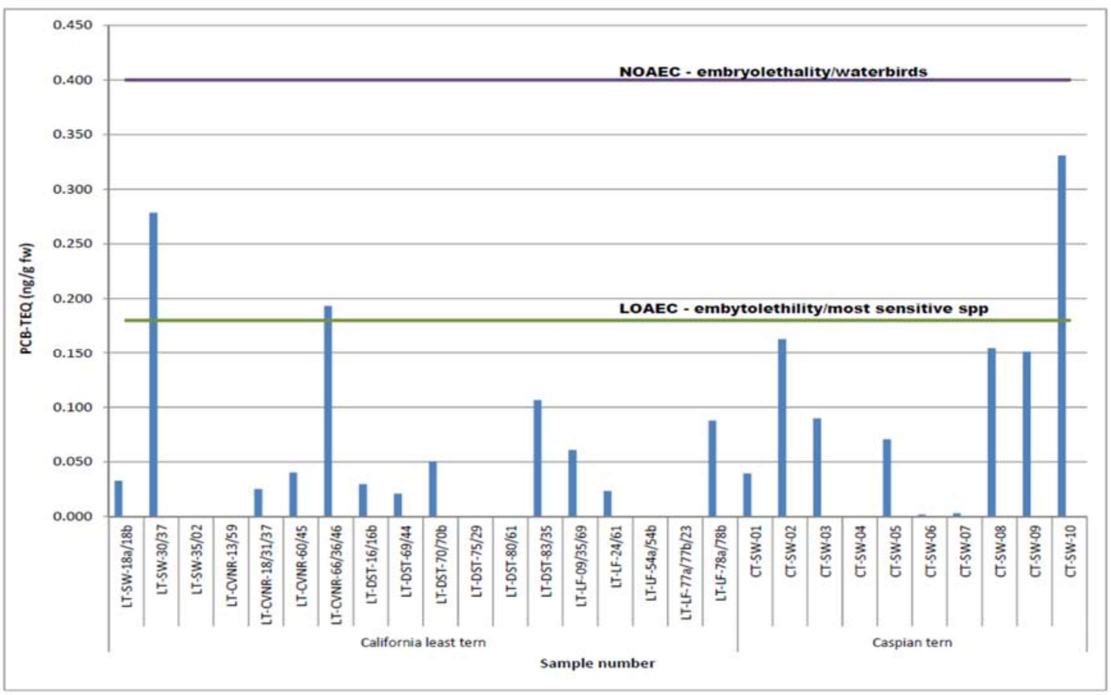

Figure 24. PCB-TEQ concentrations (ng/g fw) in individual California least tern and Caspian tern eggs collected from the San Diego Bay colonies, 2013.

Dietary exposure to PCBs (total, PCB 126, and TEQ)

Both mean and maximum concentrations of PCBs in some aquatic food web components (primarily fish) exceeded initial dietary screening levels for piscivorous avian receptors considered in this assessment (Table E2c). Based on the initial screen, it appears that piscivorous seabirds are at risk of adverse effects from exposure to PCBs in aquatic biota from San Diego Bay.

Hazard quotients estimated for mean receptor-specific EPCs, which factor in dietary preferences, and assuming bay-wide use by avian receptors as summarized in Table 19, indicate that risks posed by dietary exposure to PCBs may be elevated (i.e., HQ>1.0) for small piscivores (e.g., least tern), but are below levels of concern for other species. HQs based on maximum EPCs, combined with the NOAEL-based TRV for most sensitive species are >1.0 for four of the five representative species (excepting scoter), with HQ values ranging from 1.4 to 3.9 (Table F3). The HQ for species like the least tern raises some concern about potential for adverse effects. However, the NOAEL-based TRV used for HQ estimates is for most sensitive species (chickens), and consequently the HQs, which are all <2.0 for mean EPCs, and <4.0 for maximum EPS likely overestimate the risks to species other than chickens and other terrestrial species in general (Harris and Elliott 2011).

As discussed in "methods" the PCB concentrations reported for aquatic biota may underestimate total PCB concentrations as determined by other methods that include more than the Bight '13 congeners. As such, total PCB concentrations in fish will be approximately 1.27 times the values used for HQ estimates. As a result, estimated HQs may be higher than shown by a factor of approximately 1.27. Adjusting the results has little effect on the conclusions of this assessment. The dietary exposure to total PCBs by one or more of the receptors may pose some risk of adverse effects, however, the likelihood of detecting actual effects associated with total PCB exposure, such as impaired growth, metabolism, reproduction and behavior, in field populations is very low.

Table 19. HQs for dietary exposure by avian species to PCBs (Bight '13 congeners) in aquatic biota from San Diego Bay, based on mean EPCs and assuming birds forage throughout the bay.

Representative species	HQ - NOAEL (lowest)	HQ - LOAEL (mid-range)
Least tern	1.73	0.12
Caspian tern	0.54	0.04
Cormorant	0.71	0.05
Scoter	0.31	0.02
Gull	0.92	0.07

Hazard quotients estimated for mean receptor-specific EPCs for the dioxin-like congeners only (TEQs) indicate that risks of dioxin-like effects are elevated for all of the representative species. HQs based on mean EPCs are as high as 11 (Table 20), and HQs based on maximum EPCs are as high as 84 (Table F4).

Table 20. HQs for dietary exposure by avian species to PCB TEQs in aquatic biota from San Diego Bay, based on mean EPCs and assuming birds forage throughout the bay.

Representative species	HQ - NOAEL (lowest)	HQ - LOAEL (most sensitive)	HQ - LOAEL (mid-range)
Least tern	11	0.24	0.07
Caspian tern	6	0.12	0.03
Cormorant	9	0.19	0.05
Scoter	2	0.04	0.01
Gull	11	0.24	0.07

Like total PCBs, the high HQs for PCB TEQs were derived using a NOAEL-based TRV for most sensitive species, and as such will overestimate risk to waterbirds. HQs obtained when using the LOAEL for most sensitive species are all well below 1.0. Consequently, for those species with the higher HQs (e.g., >5.0), there may be some risk of dioxin-like adverse effects from dietary exposure to PCBs in aquatic biota from San Diego Bay. However, the likelihood of actually detecting dioxin-like effects (e.g., lethality and teratogenic effects in offspring) in field populations is low at levels where the LOAEL-based TRV is not exceeded.

PBDEs

PBDE concentrations in seabird eggs are summarized below (Table 21). Details on dietary concentrations and HQs for adult avian receptors are provided in Tables F5 and G5, with results summarized below.

PBDE concentrations in eggs

Egg samples were analyzed for fifteen PBDE congeners. Of those, BDEs 47, 99 and 100 generally contributed more than 80 percent to total PBDE concentrations in the seabird egg samples. Based on mean concentration, Caspian terns are the most exposed to PBDEs, followed by Western gull, double-crested cormorants and California least terns (Table 21). Mean total PBDE concentrations are either below or slightly exceed (Caspian terns) an estimated NOAEC (200 ng/g fw) for reproductive effects. PBDE concentrations in some individual Caspian tern, double-crested cormorant and Western gull eggs are greater than the estimated NOAEC but well below the LOAEC for reduced hatching success in a sensitive species (1,800 ng/g fw). Absent data specific to seabirds, exceedances of an estimated NOAEC may indicate a potential for reduced hatching success. However, the likelihood of actually detecting PBDE-related reductions in hatching success, especially under field conditions, is probably low.

Table 21. Total PBDE concentrations (ng/g fw) in seabird eggs collected from San Diego Bay colonies, 2013.

		Total PBDEs (ng/g fw)				
Species	Location	Mean	SD	N	Min	Max
California least tern	Salt Works	76	22	3	61	102
California least tern	CVWR	57	18	4	38	77
California least tern	D-Street Fill	37	16	6	21	67
California least tern	Lindbergh field	39	11	5	30	52
Caspian tern	Salt Works	244	110	10	83	414
Double-crested cormorant	Salt Works	89	99	8	9	280
Western gull	NAS NI	176	96	8	53	304

Dietary exposure to PBDEs

Conservative dietary screening levels for PBDEs were only occasionally exceeded by concentrations in a few species of forage fish and benthic crustaceans (Table E2e). Hazard quotients estimated for mean receptor-specific EPCs, which factor in dietary preferences, and assuming bay-wide use by avian receptors are below 1.0 (Table 22). HQs calculated from maximum EPCs, combined with the NOAEL-based TRV range from 0.4 for Caspian tern to 3.3 for Scoter (Tables F5 and G5). All HQs computed with the LOAEL-based TRV are well below 1.0. Consequently, risks associated with dietary exposure to PBDEs by seabirds and waterfowl that forage in San Diego Bay are considered to be of limited concern.

Table 22. HQs for dietary exposure by avian species to PBDEs in aquatic biota from San Diego Bay, based on mean EPCs and assuming birds forage throughout the bay.

Representative species		HQ - NOAEL (lowest)	HQ - LOAEL (most sensitive)
Least tern		0.45	0.045
Caspian tern		0.074	0.007
Cormorant		0.090	0.009
Scoter		0.208	0.021
Gull		0.115	0.012

PFCs

Samples of aquatic biota were not analyzed for PFCs, so that the assessment of risks posed by PFCs to seabirds is based on concentrations in eggs only.

PFC concentrations in eggs

Only tern eggs were analyzed for PFCs, and results are reported as the sum of the concentrations of the six PFCs targeted for this study (Table 23). Of the six PFCs, PFOS was detected most frequently and at concentrations that constituted more than 90 percent of the total. Total PFCs reported in Table 23 are essentially PFOS concentrations. Concentrations in all of the egg samples are well below the estimated NOAEC for reduced hatchability (1,000 ng/g fw) and therefore are below levels of concern.

Table 23. Total PFC concentrations in California least tern and Caspian tern eggs collected from San Diego Bay colonies, 2013.

		Total PFCs (ng/g fw)				
Species	Location	Mean	SD	N	Min	Max
California least tern	Salt Works	18.2	6.77	3	13.8	26.0
California least tern	CVWR	38.0	24.2	4	16.8	72.5
California least tern	D-Street Fill	30.9	24.1	6	4.50	66.3
California least tern	Lindbergh field	14.7	6.39	5	9.35	25.8
Caspian tern	Salt Works	28.5	17.7	10	7.54	63.2

Chlordanes

Chlordane concentrations in seabird eggs are summarized below (Table 24). Concentrations measured in aquatic biota are summarized in Table E2f. Because concentrations were all below the initial screening level, risks associated with dietary exposure to chlordanes were not considered further (i.e., no HQs were calculated).

Chlordane concentrations in eggs

Compared with other organic contaminants considered in this study, total chlordane concentrations in San Diego Bay seabird eggs are low (Table 24). Unfortunately, absent literature on effect levels in eggs, it is not possible to determine if the observed chlordane concentrations in San Diego Bay seabird eggs are sufficient to cause adverse effects.

Table 24. Total chlordane concentrations in seabird eggs collected from San Diego Bay colonies, 2013.

		Total Chlordanes (ng/g fw)				
Species	Location	Mean	SD	N	Min	Max
California least tern	Salt Works	8.86	7.64	3	3.06	17.51
California least tern	CVWR	6.03	1.82	4	4.12	8.36
California least tern	D-Street Fill	3.50	1.86	6	2.04	7.08
California least tern	Lindbergh field	6.01	1.64	5	4.50	8.27
Caspian tern	Salt Works	9.53	6.58	10	2.14	24.6
Double-crested cormorant	Salt Works	1.37	1.44	8	0.04	3.49
Western gull	NAS NI	1.68	1.54	8	0.14	4.08

Dietary exposure to chlordane

Based on the initial screen, concentrations of chlordane in aquatic biota from San Diego Bay are considered to be below levels of concern for seabirds and waterfowl that forage in the bay. It should be noted, however that there is only one study from which a dietary screening level could be derived, and therefore the NOAEL-based screening level thus derived should be used with caution. That said, maximum concentrations of chlordane in aquatic biota ranged from <0.05 to 10 ng/g ww, all of which are more than an order of magnitude lower than the dietary screening levels identified for avian receptors (196 - 1,006 ng/g ww; Table E2f) and used here for seabird and waterfowl species. Consequently, based on the available information, it appears that chlordane concentrations in aquatic biota from San Diego Bay are below levels of concern.

PAHs

Samples of seabird eggs were not analyzed for PAHs, so that the assessment of risks posed by PAHs to seabirds is based on dietary exposure only. In addition, LPAHs and HPAHs were evaluated separately to address differences in physical/chemical properties that influence fate and toxicity. Concentrations of LPAHs and HPAHs measured in aquatic biota are summarized in Table E2g. Concentrations of LPAHs in all samples were below the initial screening levels, so LPAHs were not considered further (i.e., no HQs were estimated for LPAHs). Concentrations of HPAHs in benthic invertebrates did exceed screening levels (Table E2g), and therefore are evaluated in greater detail below. Details on dietary concentrations of HPAHs and corresponding HQs for adult avian receptors are provided in Tables F6 and G6, with results summarized below.

Dietary exposure to HPAHs

Hazard quotients estimated for mean receptor-specific EPCs, which factor in dietary preferences, and assuming bay-wide use by avian receptors are below 1.0 for piscivorous birds, but approaches a value of 8.0 for waterfowl that forage on benthic invertebrates (Scoter; Table 25). HQs based on maximum EPCs are also below 1.0 for piscivores, but ~270 for species that forage on benthic invertebrates (Tables F6 and G6).

The highest NOAEL-based HQs for benthic foraging waterbirds exposed to HPAHs occur in the northern region of the bay, where the maximum HQ is ~270 and the mean is ~19 (Table G6). In the central and southern regions of the bay, maximum HQs are <3.5 and mean HQs are <1.0 (Table G6). HQs derived for HPAHs using the NOAEL-based TRV indicate that avian species that forage on benthic biota in the northern part of San Diego Bay are at elevated risk of adverse effects from dietary exposure to HPAHs.

Table 25. HQs for dietary exposure by avian species to HPAHs in aquatic biota from San Diego Bay, based on mean EPCs and assuming birds forage throughout the bay.

Representative species		HQ - NOAEL (sensitive)	HQ - LOAEL (most sensitive)
Least tern		0.34	0.003
Caspian tern		0.07	0.001
Cormorant		0.06	0.001
Scoter		7.74	0.077
Gull		0.29	0.003

However, HQs obtained for benthic feeding birds using the LOAEL-based TRV are <1.0 (for mean EPCs; bay wide or by region) and <3.5 (for maximum EPCs; baywide and by region) (Tables F6 and G6).

The TRVs for HPAHs are based on reproductive effects from exposure during the breeding season, while doses higher than the selected TRVs are associated with effects that may result from exposure outside the breeding season, such as reduced growth and impaired immune function (e.g., see Appendix D). Because surf scoters are not present during the breeding season, the elevated HQs obtained with HPAHs likely overestimate risks to that species. Avian species that forage on benthic invertebrates and are present during their breeding season may experience adverse reproductive effects from dietary exposure to HPAHs in the northern region of the bay. However, the low HQs derived from LOAEL-based TRVs suggest that the likelihood of actually detecting HPAH-related impacts in benthic foraging waterfowl in the field is low.

Uncertainty Evaluation

Uncertainties inherent in study design, data analysis and assessment protocols are identified throughout the body of this report. An awareness of uncertainties, especially those that may most affect assessment results allow risk managers to be better informed when evaluating the risk assessment conclusions. Uncertainties and some limitations of this assessment are summarized below.

Conceptual site model

The conceptual model for this assessment reflects a focus on potential worst case exposure by wildlife to specific bioaccumulative contaminants in San Diego Bay. Given the contaminants of interest, the focus is on upper trophic level avian predators of aquatic food web biota, primarily fish and benthic invertebrates, in San Diego Bay. Potential risks to herbivores or to marine mammals were not evaluated. However, the selected avian receptors are expected to experience greater levels of exposure (and risk) to the contaminants of interest than are herbivores or marine mammals.

Sampling and analyses were confined to shallow and mid-depth subtidal habitats. Consequently, risks posed by contaminants in San Diego Bay to wildlife that forage in intertidal or marsh habitats were not addressed in this assessment.

Exposure via dermal contact or inhalation pathways were not evaluated in this assessment, partly because methods are unavailable, but also because the greatest amount of exposure is expected to be via ingestion of food and, to a lesser extent, incidental ingestion of sediment. While there may be some uncertainty about exposure via dermal contact or inhalation, incorporating estimates of exposure via those routes would be difficult to make and likely have no effect on the outcome of the assessment.

Data, Contaminants of Potential Concern, and Exposure Point Concentrations

Seabird egg samples were collected in 2013 and food web samples were collected in 2013/2014. Both sets of samples provide data that are current, have sufficient numbers for statistical analyses, and food web samples include taxa that are major constituents of the diets of wildlife species that forage in shallow and mid-depth subtidal habitats of San Diego Bay. There are some uncertainties that may result in over- or underestimates of risk.

Eggs of terns and cormorants were failed to hatch, and as such data on those eggs may reflect higher contaminant loads than would be observed with randomly selected fresh laid eggs (risks may be overestimated).

Data on contaminant levels in food web organisms are considered to be representative of contaminant levels in the diets of avian wildlife foraging in San Diego Bay during the summers of 2013 and 2014. Diet composition of avian species may vary with season for both intrinsic and extrinsic reasons, and contaminant levels in aquatic biota may vary with season. Consequently. there is some uncertainty about potential seasonal or inter-annual variations in contaminant levels in the diets of avian receptors that may or may not be captured by the available data. This uncertainty is mitigated somewhat by the use of average and maximum exposure point concentrations and multiple screening levels and TRVs for bracketing risks. In addition, data are provided for a variety of aquatic species, which would allow for consideration, if desired, of changes in diet composition of receptor species.

A stratified random sampling design was used to select sample sites, which appear to have captured few if any of the most contaminated sediment sites. Whether areas with higher levels of contamination in sediment and resident benthic biota are sufficiently represented contributes to uncertainty about estimated contaminant levels in the diets of ecological receptors (e.g., surf scoter) that rely on benthic organisms for food. Both average and maximum concentrations obtained from the available data help to bracket the potential range of dietary contaminant levels and associated risks to wildlife that forage on benthic invertebrates, but risk may be underestimated by estimated averages.

Sampling and analyses for this assessment were focused on contaminants previously selected by the San Diego RWQCB, based on concerns for aquatic-dependent wildlife and humans that consume aquatic biota from San Diego Bay. This assessment evaluated wildlife risks posed by mercury, DDTs, chlordanes, PCBs, PBDEs, PFCs and PAHs in avian eggs and/or food web organisms. Risks posed to wildlife by other potential COPCs such as metals other than mercury, metalloids, and newer use pesticides were not evaluated, and as such may be underestimated.

Total PCB concentrations are reported as the sums of the 41 Bight ’13 congeners and as such underestimate total PCB concentrations (quantified using sums of more than 60 congeners or the Aroclor standard approach), the latter of which are the basis of historic data and reference values. Reported PCB concentrations in seabird eggs were adjusted for this uncertainty, while concentrations reported for aquatic biota were not. Estimated exposure point concentrations for PCBs in aquatic biota result in underestimates of dietary exposure and subsequent risks posed by total PCBs to avian receptors. For receptors that consume forage fish the difference appears to be a factor of ~1.27.

Concentrations of mixtures were computed as the sums of the concentrations of the individual constituents. The highest reported constituent detection limit was substituted for the sum when all constituents were "non-detects." This is a conservative approach that results in an estimated concentration of the mixture that may be biased high and result in an overestimate of exposure to contaminants that occur as mixtures.

Dietary exposure point concentrations were computed from summary data, which included mean, range, N, and the standard deviation for contaminant concentrations by taxon (species of fish or order of benthic invertebrates). Exposure point concentrations were computed for combinations of taxa, to reflect the dietary composition for each of the receptor categories. While it was possible to compute weighted averages, confidence limits could not be computed within the time allotted. Consequently, weighted averages and maximum values were used to represent dietary exposure point concentrations. Average concentrations are considered to be representative of contaminant levels in diets of avian receptors that forage on multiple species over areas that may encompass the entire bay, or even regions of the bay. Uncertainty about how well the average EPC represents average concentrations in receptor diets is addressed in part by considering maximum concentrations as well. Dietary exposure by avian receptors to contaminants in aquatic biota may be over- or underestimated by average concentrations, and may be overestimated by maximum concentrations. The use of maximum values increases the overall uncertainty associated with estimates of constituent intake, but makes it unlikely that exposures are underestimated.

As indicated previously, exposure point concentrations were computed from data on contaminant levels in combinations of taxa that were collected to reflect the dietary composition for each of the receptor categories. It is likely that species collected for this study did not include all that may be consumed by a particular receptor. While major dietary components appear to be represented by samples, the lack of data on other commonly consumed biota (e.g., squid consumed by gulls, and grunion consumed by Caspian terns), may result in an over- or underestimation of dietary exposure point concentrations and risk to certain receptors.

Exposure Assessment

Contaminants measured in seabird eggs are assumed to be derived, via the female parent, from aquatic biota in San Diego Bay. Contaminant levels in seabird eggs are expected to reflect levels in the diet of the female parent while present at the nesting site, but some of the parental diet may be from outside San Diego Bay. Contaminant levels measured in seabird eggs may overestimate exposure and risks posed by contaminants in San Diego Bay proper. This uncertainty is species-specific, as some avian species are expected to obtain most if not all of their food during nesting season from San Diego Bay whereas others are not (refer to species profiles for additional information).

Dietary exposure by avian receptors to contaminants was assumed to be primarily, if not entirely from San Diego Bay, depending on the species. Assumptions about foraging behavior and feeding preferences were obtained from literature, which in some cases applied to San Diego Bay, but was generally for birds at other locations. In addition, other factors used to compute daily dose rates (e.g., body weight, food ingestion rates) are based on literature values for the same or similar species at other locations. Site-specific information was used when available.

However, uncertainty about site-specific exposure factors may result in either over- or underestimation of risk specific to San Diego Bay.

Screening levels and TRVs (Toxicity Assessment)

Uncertainty is inherent in the toxicity values selected for evaluating risk. Screening levels for assessing contaminant levels in eggs and TRVs for evaluating daily dietary dose rates by adults were obtained from the literature on effect levels, which ranges from extensive for some contaminants (e.g., PCBs, DDT and mercury) to one or a few studies (e.g., PAHs, PFCs, chlordane, and PBDEs). Even with a robust data base, there are major sources of uncertainty about toxicity values. Two basic types of uncertainty relate to species differences in sensitivity and to laboratory versus field-based data. In general, the literature-based screening levels and dietary effect levels are for species other than those considered in this assessment, resulting in uncertainty about species differences in sensitivity. Identification of effect levels in eggs is typically field-based for which the presence of other contaminants is at least one complicating factor. Screening levels based on field-collected data may over- or underestimate risk, depending on interactions between co-occurring contaminants, or complications from extrinsic factors. Dietary effect levels are most often obtained from lab-based studies, but may be field-based as well. There are uncertainties about using lab-based effect levels (TRVs) to assess exposure under natural conditions, where for example uptake of the contaminant in question may be lower than occurs in the lab. Effect levels observed in lab-based studies may be from less than chronic exposures and/or dose-response relationships are not as clear as desired (e.g., effects observed at the lowest dose rate used). Depending on the data, lab-based TRVs may result in either over- or underestimates of risk from dietary exposure to a particular contaminant in the field.

Multiple toxicity values were identified for contaminants with sufficient data. The lowest toxicity values for avian eggs and TRVs were derived using conservative protocols, most notably by: focusing on sensitive effects relating to survival, growth and reproduction; working from lowest of reported effect levels; and, applying uncertainty factors when necessary for species differences in sensitivity (eggs and TRVs), lack of adequate dose-response data (eggs and TRVs), and/or exposure duration of the benchmark study (TRVs). The lowest selected toxicity values (NOAECs and NOAELs) are intended to represent values that are credible, but will overestimate risk of adverse effects. For some contaminants, most notably PCBs, the conservativeness of the lowest toxicity values used to assess risks to aquatic-dependent avian species is well established. However, toxicity values selected for contaminants with a very limited data base are used with caution and their use may over or underestimate risk.

Risk Characterization

Risk characterization using HQs allows for assessment of potential cumulative risks from exposure to multiple contaminants that exert effects through the same mode of action (e.g., PCBs, dioxins/furans and PBDEs). Samples were not analyzed for dioxins/furans and potential cumulative risks were not addressed in this assessment. Conclusions based on HQs for PCBs and PBDEs, individually may underestimate overall risks of dioxin-like effects in exposed birds.

Summary and Recommendations

This report presents a review of contaminant levels measured in seabird eggs collected in 2013 from nesting colonies around San Diego Bay, and in aquatic food web organisms collected from San Diego Bay in 2013 and 2014. The seabird eggs were collected as part of a larger investigation to assess exposure and potential risks posed to wildlife by contaminants in San Diego Bay, with a focus on avian species that are top predators in the aquatic food web, and as such are potentially the most heavily exposed to bioaccumulative contaminants. Several types of aquatic biota were collected as part of multiple investigations to characterize bioaccumulation and trophic interactions of contaminants in the aquatic-based food web of San Diego Bay. Data on contaminant levels in aquatic biota and avian eggs were used to evaluate exposure and risks associated with dietary exposure to contaminants, and for avian receptors, risk associated with *in ovo* exposure by embryos to contaminants transferred from the female parent to the egg. Risks to five types of birds, representing different feeding preferences and strategies were evaluated.

Based on previously expressed concerns by the SDRWQCB, this assessment was focused on specific bioaccumulative contaminants, those being mercury, OC pesticides (DDT and chlordanes), PCBs, PBDEs, PFCs and PAHs. Concentrations of contaminants detected in eggs were compared with literature-based values used to define thresholds associated with adverse effects in directly exposed embryos of avian species. Screening values included at least one NOAEC, above which potential risk requires further consideration, and one or more LOAECs to characterize potential for detecting contaminant-related impacts. For dietary exposure by adult birds, data on contaminant levels in aquatic biota were used to estimate species- and site-specific daily dose (exposure) rates. Daily doses were then compared with literature-based values used to define thresholds for adverse effects associated with dietary exposure to contaminants. Reference values included at least one no effect-based daily dose rate (NOAEL), above which potential risk requires further consideration, and one or more low observed effect-based daily dose rate (LOAEL) to characterize potential for detecting contaminant-related impacts.

Results are summarized as follows:

Mercury

- Mercury concentrations in at least some seabird eggs are greater than the estimated NOAEC, but below LOAECs.
- Estimates of dietary exposure by adult birds to mercury exceed NOAELs and LOAELs for most sensitive species, but are below a NOAEL for less sensitive species, which include waterbirds.
- Mercury concentrations in seabird eggs are considered to be present at levels of potential concern but the likelihood of detecting measurable effects is low.
- Mercury in aquatic biota from San Diego Bay may pose some risk of adverse effects on avian species that forage on benthic invertebrates and on small-bodied avian species that forage on pelagic fish. The likelihood of detecting measurable effects is low, but may be greater for more sensitive species. Ongoing monitoring, with additional focus on risks to especially sensitive waterbird species, and species that forage on benthic invertebrates is recommended.

DDTs

- Total DDTs concentrations in at least some seabird eggs are greater than estimated NOAECs for eggshell thinning and reduced nest productivity. They also exceed LOAECs for a highly sensitive species, but are below LOAECs for less sensitive species.
- Estimates of dietary exposure by adult birds to DDTs in aquatic food web organisms from San Diego Bay are below NOAELs, and therefore below levels of concern.

PCBs

- Total PCBs and TCDD-TEQ concentrations in at least some seabird eggs exceed LOAECs for highly sensitive species, but are below estimated NOAECs for less sensitive species, which include waterbirds. The potential for similarly-acting contaminants to increase risk still requires further consideration.
- Estimates of dietary exposure by one of the receptor species to total PCBs are greater than a NOAEL for most sensitive species. The estimated daily exposures to PCBs as TEQs for all receptors were greater than a NOAEL for most sensitive species, but less than a LOAEL, also for most sensitive species. Risks associated with dietary exposure to PCBs (as total PCBs or TEQs) may be overestimated, but are still of concern.

PBDEs

- Total PBDE concentrations in at least some seabird eggs are greater than the estimated NOAEC, but are below the LOAEC.
- Estimated daily dietary exposure by waterfowl and seabirds to PBDEs in aquatic are below NOAELs.

PFCs

- Concentrations of PFCs (primarily PFOS) in seabird eggs are well below the only readily available NOAEC.
- Aquatic biota were not analyzed for PFCs. Consequently, risks associated with dietary exposure to PFCs were not assessed.

Chlordane

- Due to lack of data on effect levels, concentrations of chlordanes in seabird eggs could not be evaluated for potential effects.
- Estimated daily dietary exposure by waterfowl and seabirds to chlordanes in aquatic biota are below the single available NOAEL.

PAHs

- Seabird eggs were not analyzed for PAHs, so potential risks posed by PAHs to developing embryos was not assessed.

- Estimated daily dietary exposure the LPAHs by waterfowl and seabirds that forage on aquatic biota from San Diego Bay are below the NOAEL.
- Estimated daily dietary exposure to HPAHs by waterfowl that forage on benthic invertebrates are greater than the NOAEL, but less than the LOAEL. Avian species that forage on benthic invertebrates, especially if present during the breeding season may be at risk of adverse effects from exposure to HPAHs.

Risk to Human Health

Data analysis

Comparison to OEHHA guidelines

Tissue contaminant concentrations were compared to Advisory Tissue Levels (ATLs) and Fish Contaminant Goals (FCGs) established by the Office of Environmental Health Hazard Assessment (OEHHA). These guidelines provide recommendations for the frequency of fish consumption and take into consideration the health benefits of a diet that includes fish (Table 26). The average tissue concentration for each species, either by region or for the entire bay, was compared to the ATLs or FCG for each contaminant type.

Table 26. Fish Contaminant Goals (FCGs) and Advisory Tissue Levels (ATLs) based on an assessment of human health risk by OEHHA (Klasing and Brodberg, 2008). All values given in ng/g (ppb) wet weight. One serving is defined as 8 ounces (227 g) prior to cooking.

Contaminant	FCG	ATL for 8 oz Serving Size (ng/g)			
		3 servings per week	2 servings per week	1 serving per week	No Consumption
Chlordanes (ng/g)	5.6	≤ 190	> 190-280	> 280-560	> 560
DDTs (ng/g)	21	≤ 520	> 520-1000	> 1000-2100	> 2100
Dieldrin (ng/g)	0.46	≤ 15	> 15-23	> 23-46	> 46
Mercury [1] (ng/g)	220	≤ 70	> 70-150	> 150-440	> 440
Mercury [2] (ng/g)	655	≤ 220	> 220-440	> 440-1310	> 1310
PCBs (ng/g)	3.6	≤ 21	> 21-42	> 42-120	> 120

[1] Women 18 to 45 years of age and children 1 to 17 years of age
[2] Women over 45 years of age and men

Temporal comparison

The average tissue contaminant concentration values from this study were compared to historical data for the same species to investigate temporal changes in potential human health risk. Comparisons were made to the contaminant data used by OEHHA to develop recent fish consumption advisories for San Diego Bay (Gassel et al. 2013); most fish for this dataset were collected in 2001. Additional historical data for San Diego Bay sport fish are available for 2008-09, as part of a SWAMP regional survey of coastal fish contamination (Davis et al., 2010). However, these data were not used in comparisons because there were few species in common with the present study and only a small sample size was available for most species (N = 2).

Tissue contamination

Contaminant analyses were conducted on 23 samples of sport fish from San Diego Bay that represented 5 commonly consumed species. Every sample contained detectable levels of mercury and PCBs (Table 27). DDTs were detected in all species, except for round stingray. Dieldrin was not detected in any of the samples. Chlordanes were usually detected at low levels in chub mackerel and spotted bass.

Species-specific variations in contaminant concentration were apparent for some chemicals. Pacific chub mackerel contained approximately 10-fold higher concentrations of DDTs and chlordanes than other species. Spotted sand bass tended to have the highest concentration of PCBs, although the highest concentration was reported for a single sample of topsmelt. Concentrations of mercury varied little among species.

Table 27. Average chemical concentrations and standard deviations (SD) for sport fish collected in San Diego Bay (whole bay). N = number of samples used to calculate the average. N below Detection = number of samples below method detection limits. NA = data not available due to small sample size.

Analyte	Species	N	N Below Detection	Average	SD	Units
Chlordanes	California halibut	8	7	0.05	NA	ng/g ww
Chlordanes	Pacific chub mackerel	3	0	1.12	0.89	ng/g ww
Chlordanes	Round stingray	2	1	0.11	0.09	ng/g ww
Chlordanes	Spotted sand bass	9	2	0.16	0.12	ng/g ww
Chlordanes	Topsmelt	1	0	0.34	NA	ng/g ww
DDTs	California halibut	8	0	1.27	0.76	ng/g ww
DDTs	Pacific chub mackerel	3	0	10.34	6.99	ng/g ww
DDTs	Round stingray	2	2	0.05	NA	ng/g ww
DDTs	Spotted sand bass	9	0	0.89	0.45	ng/g ww
DDTs	Topsmelt	1	0	3.03	NA	ng/g ww
Dieldrin	California halibut	8	8	0.05	NA	ng/g ww
Dieldrin	Pacific chub mackerel	3	3	0.05	NA	ng/g ww
Dieldrin	Round stingray	2	2	0.05	NA	ng/g ww
Dieldrin	Spotted sand bass	9	9	0.05	NA	ng/g ww
Dieldrin	Topsmelt	1	1	0.05	NA	ng/g ww
Mercury	California halibut	8	0	0.14	0.07	µg/g ww
Mercury	Pacific chub mackerel	3	0	0.11	0.09	µg/g ww
Mercury	Round stingray	2	0	0.21	0.03	µg/g ww
Mercury	Spotted sand bass	9	0	0.19	0.03	µg/g ww
Mercury	Topsmelt	1	0	0.03	NA	µg/g ww
PCBs	California halibut	8	0	14.62	8.89	ng/g ww
PCBs	Pacific chub mackerel	3	0	104	90.74	ng/g ww
PCBs	Round stingray	2	0	22.24	20.07	ng/g ww
PCBs	Spotted sand bass	9	0	28.91	12.34	ng/g ww
PCBs	Topsmelt	1	0	34.67	NA	ng/g ww

Multiple samples from more than one region of the Bay were available only for spotted sand bass and halibut. Thus, spatial patterns in tissue contamination were not evaluated for Pacific chub mackerel, round stingray, or topsmelt.

Tissue contaminant concentrations were generally similar among the North, Central, and South regions of the Bay (Table 28). The greatest regional variation was observed for PCBs in spotted sand bass, where concentrations in fish from the North and Central were 2-3x greater than the South. Mercury concentrations showed less than a 2x variation among regions. The North region of the Bay also tended to have higher concentrations of tissue DDTs, with approximately a 2x increase relative to the Central and South. Spotted sand bass also tended to have higher chlordanes in the North.

Table 28. Average chlordanes, DDTs, dieldrin, mercury and PCBs concentrations and standard deviations (SD) for sport fish collected in three different regions of San Diego Bay. N = number of samples used to calculate the average. NA = data not available.

Species	North			Central			South		
	N	Average	SD	N	Average	SD	N	Average	SD
Chlordanes (ng/g ww)									
California halibut	2	0.04	0.01	5	0.05	0	1	0.05	NA
Pacific chub mackerel	2	0.67	0.63	0	NA	NA	1	2.01	NA
Round stingray	2	0.11	0.09	0	NA	NA	0	NA	NA
Spotted sand bass	4	0.22	0.15	3	0.09	0.07	2	0.13	0.07
Topsmelt	0	NA	NA	0	NA	NA	1	0.34	NA
DDTs (ng/g ww)									
California halibut	2	2.05	1.18	5	1.03	0.52	1	0.93	NA
Pacific chub mackerel	2	6.65	4	0	NA	NA	1	17.73	NA
Round stingray	2	0.05	NA	0	NA	NA	0	NA	NA
Spotted sand bass	4	1.19	0.49	3	0.59	0.21	2	0.74	0.35
Topsmelt	0	NA	NA	0	NA	NA	1	3.03	NA
Dieldrin [1] (ng/g ww)									
California halibut	2	0.05	NA	5	0.05	NA	1	0.05	NA
Pacific chub mackerel	2	0.05	NA	0	NA	NA	1	0.05	NA
Round stingray	2	0.05	NA	0	NA	NA	0	NA	NA
Spotted sand bass	4	0.05	NA	3	0.05	NA	2	0.05	NA
Topsmelt	0	NA	NA	0	NA	NA	1	0.05	NA
Mercury (µg/g ww)									
California halibut	2	0.08	0.02	5	0.17	0.08	1	0.15	NA
Pacific chub mackerel	2	0.06	0.02	0	NA	NA	1	0.21	NA
Round stingray	2	0.21	0.03	0	NA	NA	0	NA	NA
Spotted sand bass	4	0.21	0.02	3	0.18	0.03	2	0.16	0.02
Topsmelt	0	NA	NA	0	NA	NA	1	0.03	NA
PCBs (ng/g ww)									
California halibut	2	18.17	20.92	5	13.23	4.51	1	14.44	NA
Pacific chub mackerel	2	54.93	44.99	0	NA	NA	1	202.13	NA
Round stingray	2	22.24	20.07	0	NA	NA	0	NA	NA
Spotted sand bass	4	31.06	45.93	3	36.6	10.4	2	13.06	10.38
Topsmelt	0	NA	NA	0	NA	NA	1	34.67	NA

[1] All dieldrin values were below detection limits.

Sport fish consumption risk

The average tissue contaminant concentrations of chlordanes, dieldrin, DDTs, and mercury were all below OEHHA fish contaminant goal (FCG) thresholds (Figures 25, 26, and 27). Tissue contaminant concentrations below the FCG indicate that consumption of one eight ounce meal per week over a lifetime is not expected to result in more than one additional case of cancer per one million and no significant noncancer risk (Klasing and Brodberg 2008). Mercury concentrations in most fish species were very close to the FCG. Tissue mercury in California halibut, round stingray, Pacific chub mackerel, and spotted sand bass from most regions of the Bay were within the OEHHA Advisory Tissue Level range corresponding to consumption of no more than a single serving per week for sensitive populations (>0.15-0.44 ug/g ww, Table 26).

Tissue PCBs were above the FCG for all species analyzed (Figure 26). In addition, several species had average PCBs concentrations that exceeded one or more OEHHA ATLs. The single composite sample of Pacific chub mackerel from the South exceeded the ATL for no consumption, while average PCBs in mackerel from the North were in the ATL range corresponding to consumption of no more than one serving per week (Table 28). Spotted sand bass from the North and Central regions and round stingray had average PCBs within the ATL range corresponding to no more than two servings per week. California halibut from all regions and spotted sand bass from the South had the lowest concentrations of PCBs and were in the ATL range corresponding to three servings per week.

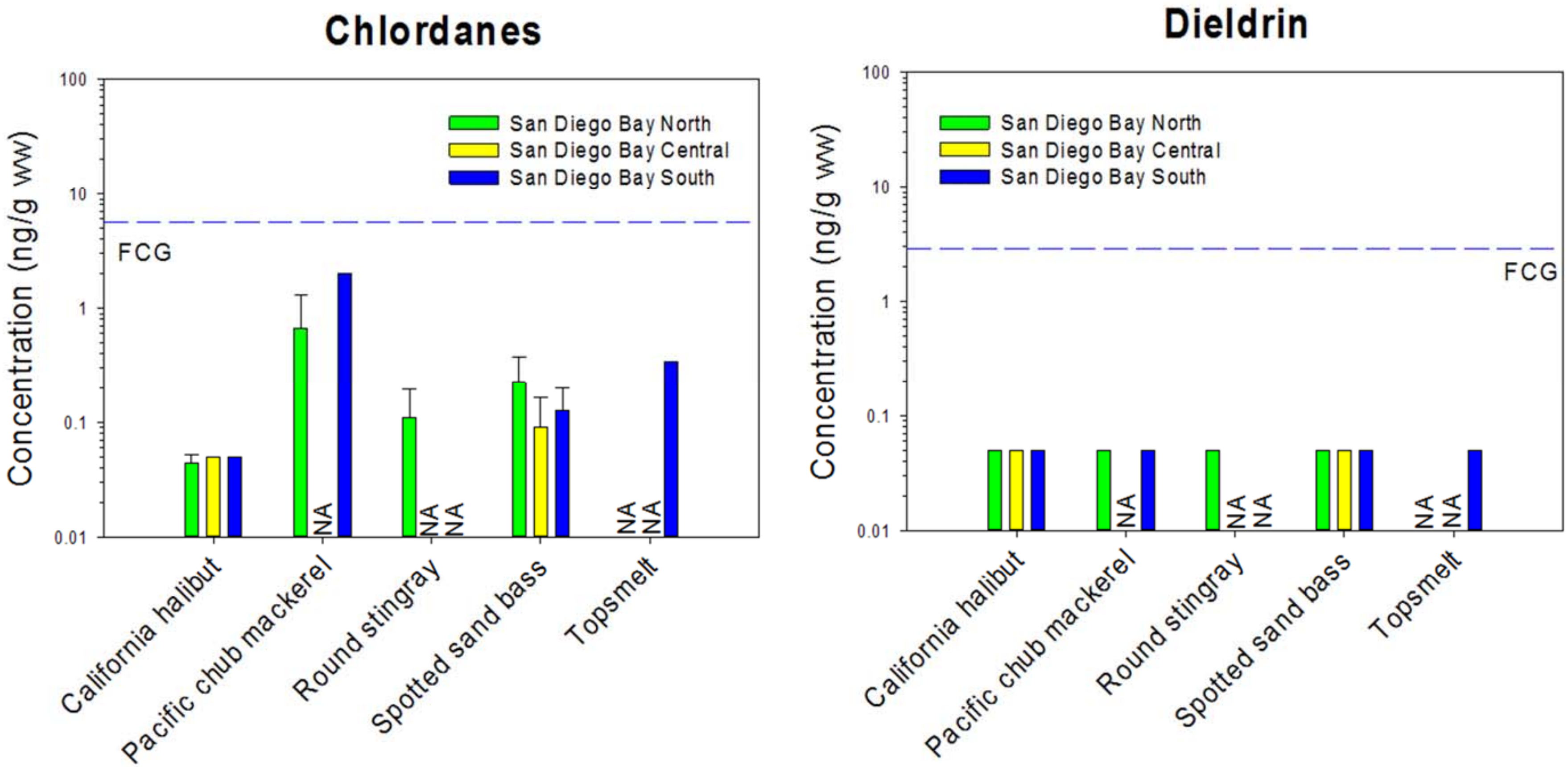

Figure 25. Average total chlordanes and dieldrin concentrations (+ standard deviation) for San Diego Bay sport fish. Dashed line represents OEHHA Fish Contaminant Goal (FCG). The no consumption Advisory Tissue Level (No Consumption ATL) for chlordanes is 560 ng/g. The No Consumption ATL for dieldrin is 46 ng/g. NA = samples were not available for the species in the region. Samples with no error bars have an N of 1 or were below detection limit for all samples. All dieldrin concentrations were below the detection limit of 0.05 ng/g ww. The detection limit for chlordanes was 0.05 ng/g ww.

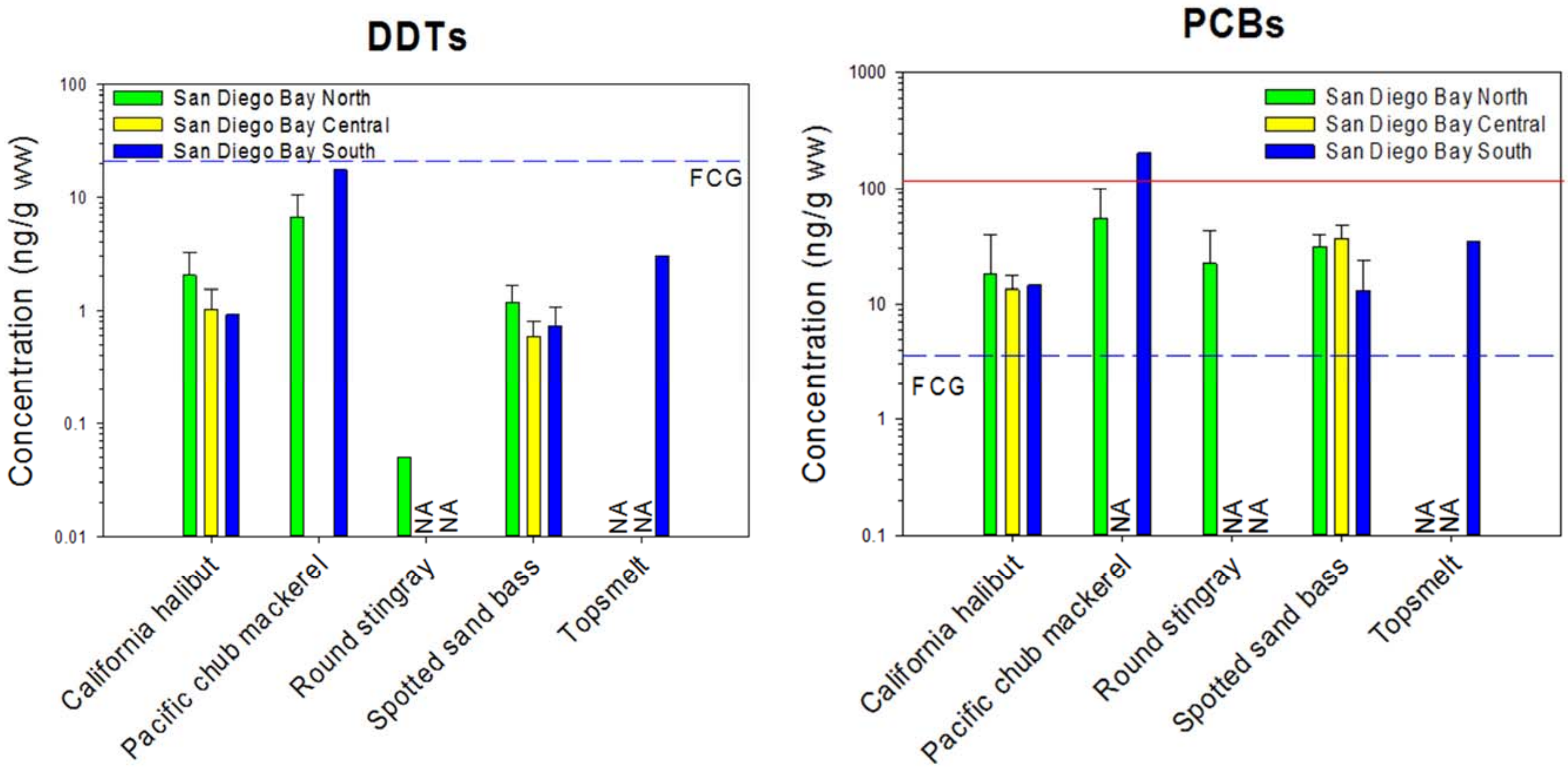

Figure 26. Average total DDTs concentrations (+ standard deviation) for San Diego Bay sport fish. Dashed line represents OEHHA Fish Contaminant Goal (FCG). The NO consumption ATL for DDTs is 2100 ng/g. The solid line represents the No Consumption ATL for PCBs of 120 ng/g. NA = samples were not available for the species in the region. DDTs detection limit is 0.05 ng/g ww. PCBs detection limit is 0.05 ng/g ww. Samples with no error bars have an N of 1.

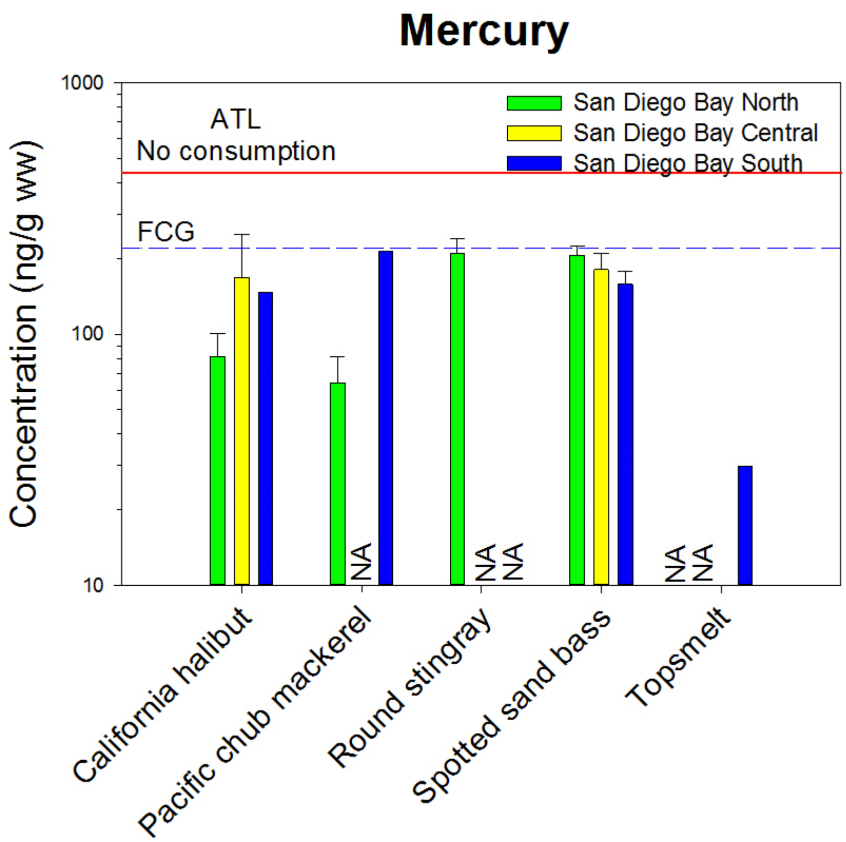

Figure 27. Average mercury concentrations (+ standard deviation) for San Diego Bay sport fish. Dashed line represents OEHHA Fish Contaminant Goal (FCG) for mercury. The No Consumption ATL for women 18 to 45 years and children 1 to 17 years of age is 0.44 µg/g. NA = samples were not available for the species in the region. Mercury detection limit is 0.00001 µg/g ww. Samples with no error bars have an N of 1.

Temporal comparison

A summary of the sport fish data used by OEHHA to develop fish advisories in San Diego Bay are shown in Table 29. These data describe tissue contaminant concentrations present in 2001 and, with the exception of California halibut, are based on the analysis of multiple composites of 5 or more fish.

Most of the tissue chlordane and dieldrin results in the OEHHA dataset are nondetect and so a temporal comparison for this contaminant is not possible. Total DDTs concentration in sport fish from the current study are approximately three to five-fold lower than measured in 2001. Little temporal change in tissue mercury relative to 2001 is evident. Mercury concentrations in 2001 are within a factor of two of those measured in the current study.

Total PCB concentrations also appear to have declined relative to 2001 for some species of sport fish. PCB concentrations in 2014-15 California halibut and spotted sand bass samples are about half of those reported for 2001 samples (Table 29). Concentrations of PCBs in Pacific chub mackerel and round stingray were similar between time periods, suggesting that these two species may be accumulating PCBs from different sources/locations than California halibut and spotted bass.

Table 29. Average contaminant concentrations in fish used to develop fish consumption advisories for San Diego Bay. N = number of composites analyzed. Fish for this study were collected in 2001, except for Pacific chub mackerel which were also collected in 2009.

Analyte Name	Species [1]	N	Total Number of Fish Analyzed	Average	Units
Chlordanes	California halibut	1	3	0	(ng/g)
Chlordanes	Pacific chub mackerel	5	34	1	(ng/g)
Chlordanes	Spotted sand bass > 12”	12	60	0	(ng/g)
Chlordanes	Spotted sand bass > 14”	1	5	0	(ng/g)
Chlordanes	Topsmelt	3	66	1	(ng/g)
DDTs	California halibut	1	3	7	(ng/g)
DDTs	Pacific chub mackerel	5	34	28	(ng/g)
DDTs	Spotted sand bass > 12”	12	60	6	(ng/g)
DDTs	Spotted sand bass > 14”	1	5	5	(ng/g)
DDTs	Topsmelt	3	66	13	(ng/g)
Dieldrin	California halibut	1	3	0	(ng/g)
Dieldrin	Pacific chub mackerel	5	34	0	(ng/g)
Dieldrin	Spotted sand bass > 12”	12	60	0	(ng/g)
Dieldrin	Spotted sand bass > 14”	1	5	0	(ng/g)
Dieldrin	Topsmelt	3	66	0	(ng/g)
Mercury	California halibut	1	3	0.2	(µg/g)
Mercury	Pacific chub mackerel	8	49	0.1	(µg/g)
Mercury	Round stingray	16	80	0.3	(µg/g)
Mercury	Spotted sand bass ≥ 12”	46	163	0.2	(µg/g)
Mercury	Spotted sand bass 12 - 14”	42	155	0.2	(µg/g)
Mercury	Spotted sand bass > 14”	4	8	0.3	(µg/g)
Mercury	Topsmelt	3	66	0	(µg/g)
PCBs	California Halibut	1	3	29	(ng/g)
PCBs	Pacific chub mackerel	5	34	89	(ng/g)
PCBs	Round stingray	16	80	15	(ng/g)
PCBs	Spotted sand bass ≥ 12”	23	113	62	(ng/g)
PCBs	Spotted sand bass 12 - 14”	22	108	61	(ng/g)
PCBs	Spotted sand bass > 14”	1	5	75	(ng/g)
PCBs	Topsmelt	3	66	127	(ng/g)

[1] Most fish were analyzed as skinless fillets. Topsmelt were analyzed as whole fish (without head, tail, and guts) with skin on.

Health risk summary

This study was successful in obtaining updated tissue contamination data for several species of sport fish commonly captured from San Diego Bay and consumed: California halibut, spotted sand bass, round stingray, Pacific chub mackerel, and topsmelt. Key findings from the data analyses are summarized below.

- PCBs are the dominant trace organic contaminant of fish in the Bay. DDTs, while still prevalent in fish, are usually present at much lower concentrations. Pacific chub mackerel and spotted sand bass tended to have the highest concentrations, among the species analyzed.

- There was little variation in tissue contaminant concentrations among the North, Central and South regions of the Bay. Within the same species, average contaminant concentrations usually varied by less than a factor of 3 among regions, with the North and Central regions containing the more highly contaminated fish.

- The greatest potential risk to human health appears to be associated with PCBs in fish tissue, followed closely by mercury. For both contaminants, tissue concentrations in multiple species of fish exceeded ATLs, and were within the range where consumption of no more than one meal per week is recommended by OEHHA.

- Comparison with historical data suggests that sport fish contamination levels for PCBs and DDTs have declined two to five fold over the last 15 years (depending on species). Tissue mercury concentrations have changed little over the same period, however, an indication that Bay sediments may not be the principal source of mercury contamination in tissue.

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Appendices

Appendix A. Food web chemistry summary

Table A1. Chemistry summary for benthic invertebrates.

Common Name	Analyte	Region	Min	Mean	Median	Max	StdDev	Count	Units
Crustacea	Chlordanes	North	0.05	0.63	0.05	2.66	1.14	5	ng/g ww
Mollusks	Chlordanes	North	0.05	0.37	0.05	1.00	0.55	3	ng/g ww
Polychaetes	Chlordanes	North	0.05	1.93	1.98	4.61	1.76	6	ng/g ww
Crustacea	Chlordanes	Central	0.05	0.10	0.05	0.24	0.10	4	ng/g ww
Mollusks	Chlordanes	Central	0.05	0.83	0.55	2.15	1.00	4	ng/g ww
Polychaetes	Chlordanes	Central	0.05	10.90	0.77	61.61	24.86	6	ng/g ww
Crustacea	Chlordanes	South	0.05	0.05	0.05	0.05	0.00	4	ng/g ww
Mollusks	Chlordanes	South	0.05	0.12	0.05	0.33	0.14	4	ng/g ww
Polychaetes	Chlordanes	South	0.05	0.36	0.05	1.90	0.75	6	ng/g ww
Crustacea	DDTs	North	4.08	9.19	6.82	21.48	6.98	5	ng/g ww
Mollusks	DDTs	North	0.05	14.31	9.79	33.08	16.97	3	ng/g ww
Polychaetes	DDTs	North	7.44	9.11	8.55	11.22	1.51	6	ng/g ww
Crustacea	DDTs	Central	1.06	3.63	3.46	6.52	2.24	4	ng/g ww
Mollusks	DDTs	Central	0.05	3.45	3.18	7.36	3.14	4	ng/g ww
Polychaetes	DDTs	Central	1.81	5.35	4.43	11.52	3.29	6	ng/g ww
Crustacea	DDTs	South	0.05	4.63	3.82	10.82	4.82	4	ng/g ww
Mollusks	DDTs	South	0.90	2.51	2.35	4.42	1.64	4	ng/g ww
Polychaetes	DDTs	South	2.79	6.96	4.26	21.53	7.20	6	ng/g ww
Crustacea	Dieldrin	North	0.05	0.05	0.05	0.05	0.00	3	ng/g ww
Polychaetes	Dieldrin	North	0.05	0.05	0.05	0.05	0.00	3	ng/g ww
Crustacea	Dieldrin	Central	0.05	0.05	0.05	0.05	0.00	2	ng/g ww
Mollusks	Dieldrin	Central	0.05	0.05	0.05	0.05	NA	1	ng/g ww
Polychaetes	Dieldrin	Central	0.05	0.05	0.05	0.05	0.00	3	ng/g ww
Crustacea	Dieldrin	South	0.05	0.05	0.05	0.05	0.00	2	ng/g ww
Mollusks	Dieldrin	South	0.05	0.05	0.05	0.05	NA	1	ng/g ww
Polychaetes	Dieldrin	South	0.05	0.05	0.05	0.05	0.00	3	ng/g ww
Crustacea	Mercury	North	0.02	0.03	0.03	0.03	0.01	4	ug/g ww
Mollusks	Mercury	North	0.05	0.07	0.07	0.09	0.03	2	ug/g ww
Polychaetes	Mercury	North	0.02	0.05	0.04	0.14	0.04	6	ug/g ww
Crustacea	Mercury	Central	0.02	0.04	0.04	0.05	0.01	4	ug/g ww
Mollusks	Mercury	Central	0.03	0.04	0.04	0.06	0.01	4	ug/g ww
Polychaetes	Mercury	Central	0.05	0.16	0.10	0.43	0.15	6	ug/g ww
Crustacea	Mercury	South	0.02	0.03	0.03	0.03	0.01	2	ug/g ww
Mollusks	Mercury	South	0.01	0.02	0.02	0.04	0.01	4	ug/g ww
Polychaetes	Mercury	South	0.04	0.07	0.04	0.16	0.06	4	ug/g ww

Table A1. Continued.

Common Name	Analyte	Region	Min	Mean	Median	Max	StdDev	Count	Units
Crustacea	PAHs	North	58.00	165.02	213.20	248.00	86.02	5	ng/g ww
Mollusks	PAHs	North	158.40	211.94	159.76	317.66	91.56	3	ng/g ww
Polychaetes	PAHs	North	106.77	2274.42	162.55	12801.74	5157.88	6	ng/g ww
Crustacea	PAHs	Central	9.80	57.99	53.26	115.63	49.02	4	ng/g ww
Mollusks	PAHs	Central	41.25	75.66	69.34	122.69	37.66	4	ng/g ww
Polychaetes	PAHs	Central	27.94	105.17	105.40	155.60	48.42	6	ng/g ww
Crustacea	PAHs	South	2.50	86.65	54.30	235.50	109.10	4	ng/g ww
Mollusks	PAHs	South	24.60	42.86	41.27	64.30	19.49	4	ng/g ww
Polychaetes	PAHs	South	32.60	135.16	135.88	200.30	57.15	6	ng/g ww
Crustacea	PBDEs	North	0.70	2.08	2.25	2.96	0.84	5	ng/g ww
Mollusks	PBDEs	North	0.59	1.82	0.86	4.00	1.90	3	ng/g ww
Polychaetes	PBDEs	North	1.23	3.45	3.30	6.17	1.92	6	ng/g ww
Crustacea	PBDEs	Central	2.16	4.20	4.14	6.36	1.85	4	ng/g ww
Mollusks	PBDEs	Central	0.39	1.06	0.96	1.93	0.68	4	ng/g ww
Polychaetes	PBDEs	Central	1.89	5.46	5.11	10.51	3.88	6	ng/g ww
Crustacea	PBDEs	South	0.05	29.39	9.02	99.45	46.90	4	ng/g ww
Mollusks	PBDEs	South	0.05	1.41	1.17	3.26	1.57	4	ng/g ww
Polychaetes	PBDEs	South	2.96	9.18	7.81	21.68	7.36	6	ng/g ww
Crustacea	PCBs	North	16.84	115.55	151.47	168.59	65.72	5	ng/g ww
Mollusks	PCBs	North	18.94	41.87	42.47	64.19	22.63	3	ng/g ww
Polychaetes	PCBs	North	28.52	121.67	135.25	166.71	49.52	6	ng/g ww
Crustacea	PCBs	Central	57.33	92.54	80.98	150.87	41.58	4	ng/g ww
Mollusks	PCBs	Central	10.18	75.59	61.38	169.42	74.66	4	ng/g ww
Polychaetes	PCBs	Central	111.85	167.71	152.53	283.07	63.94	6	ng/g ww
Crustacea	PCBs	South	28.80	60.27	44.48	123.31	44.06	4	ng/g ww
Mollusks	PCBs	South	2.70	12.76	12.50	23.35	8.74	4	ng/g ww
Polychaetes	PCBs	South	28.43	77.24	64.48	146.21	44.98	6	ng/g ww

Table A2. Chemistry summary for plankton.

Common Name	Analyte	Region	Min	Mean	Median	Max	StdDev	Count	Units
Plankton	Chlordanes	North	0.05	0.59	0.52	1.26	0.55	4	ng/g ww
Plankton	Chlordanes	Central	0.05	0.16	0.05	0.59	0.24	5	ng/g ww
Plankton	Chlordanes	South	0.05	0.20	0.14	0.45	0.19	4	ng/g ww
Plankton	DDTs	North	0.05	8.00	5.46	21.02	9.10	4	ng/g ww
Plankton	DDTs	Central	0.05	1.20	0.05	3.59	1.64	5	ng/g ww
Plankton	DDTs	South	0.05	10.49	8.49	24.95	10.55	4	ng/g ww
Plankton	Mercury	North	0.01	0.02	0.03	0.03	0.01	3	ug/g ww
Plankton	Mercury	Central	0.02	0.04	0.03	0.07	0.03	3	ug/g ww
Plankton	Mercury	South	0.01	0.01	0.01	0.02	0.01	3	ug/g ww
Plankton	PAHs	North	43.75	284.94	181.70	629.38	306.16	3	ng/g ww
Plankton	PAHs	Central	60.80	159.98	131.18	287.94	116.27	3	ng/g ww
Plankton	PAHs	South	50.31	61.17	60.09	73.10	11.43	3	ng/g ww
Plankton	PBDEs	North	7.59	14.73	10.37	30.59	10.66	4	ng/g ww
Plankton	PBDEs	Central	1.77	27.31	8.96	108.27	45.48	5	ng/g ww
Plankton	PBDEs	South	1.93	8.14	3.47	23.70	10.40	4	ng/g ww
Plankton	PCBs	North	0.10	42.15	46.96	74.57	37.06	4	ng/g ww
Plankton	PCBs	Central	3.37	84.64	86.01	169.45	60.22	5	ng/g ww
Plankton	PCBs	South	6.82	43.24	50.20	65.74	25.47	4	ng/g ww

Table A3. Chemistry summary for fish.

Common Name	Analyte	Region	Min	Mean	Median	Max	StdDev	Count	Units
Barred sand bass	Chlordanes	North	2.03	2.64	2.76	3.02	0.43	4	ng/g ww
Black perch	Chlordanes	North	0.58	3.20	3.20	5.83	3.71	2	ng/g ww
California halibut	Chlordanes	North	0.05	1.09	0.88	2.65	0.85	7	ng/g ww
Goby sp.	Chlordanes	North	1.23	1.23	1.23	1.23	NA	1	ng/g ww
Shiner perch	Chlordanes	North	0.03	3.55	2.88	7.76	3.91	3	ng/g ww
Spotted sand bass	Chlordanes	North	1.73	3.23	3.11	4.85	1.56	3	ng/g ww
Topsmelt	Chlordanes	North	1.24	2.17	1.85	3.43	1.13	3	ng/g ww
Barred sand bass	Chlordanes	Central	0.63	1.70	1.70	2.77	1.51	2	ng/g ww
California halibut	Chlordanes	Central	0.51	1.27	1.16	3.08	0.90	7	ng/g ww
Deepbody anchovy	Chlordanes	Central	2.27	4.49	4.51	6.65	2.18	6	ng/g ww
Goby sp.	Chlordanes	Central	0.45	0.45	0.45	0.45	NA	1	ng/g ww
Shiner perch	Chlordanes	Central	1.27	2.55	2.55	3.83	1.82	2	ng/g ww
Slough anchovy	Chlordanes	Central	0.05	1.11	1.05	2.52	0.68	8	ng/g ww
Spotted sand bass	Chlordanes	Central	1.28	2.57	2.91	3.95	1.20	5	ng/g ww
Topsmelt	Chlordanes	Central	0.54	0.88	0.98	1.11	0.30	3	ng/g ww
Arrow goby	Chlordanes	South	0.05	0.05	0.05	0.05	NA	1	ng/g ww
Barred sand bass	Chlordanes	South	0.05	0.74	0.24	1.93	1.03	3	ng/g ww
California halibut	Chlordanes	South	0.29	0.96	0.84	1.99	0.58	6	ng/g ww
California killifish	Chlordanes	South	0.05	0.05	0.05	0.05	0.00	2	ng/g ww
Deepbody anchovy	Chlordanes	South	0.05	0.87	0.76	2.25	0.86	5	ng/g ww
Goby sp.	Chlordanes	South	0.05	0.05	0.05	0.05	NA	1	ng/g ww
Northern anchovy	Chlordanes	South	0.05	0.05	0.05	0.05	0.00	2	ng/g ww
Round stingray	Chlordanes	South	10.21	10.21	10.21	10.21	NA	1	ng/g ww
Shiner perch	Chlordanes	South	0.14	0.14	0.14	0.14	NA	1	ng/g ww
Slough anchovy	Chlordanes	South	0.05	1.82	1.82	3.58	2.50	2	ng/g ww
Spotted sand bass	Chlordanes	South	1.85	2.62	2.63	3.38	0.76	3	ng/g ww
Topsmelt	Chlordanes	South	0.05	0.71	0.58	1.50	0.73	3	ng/g ww
Barred sand bass	DDTs	North	11.31	20.97	16.12	40.32	13.27	4	ng/g ww
Black perch	DDTs	North	6.39	13.25	13.25	20.11	9.70	2	ng/g ww
California halibut	DDTs	North	2.52	13.91	12.25	37.71	11.14	7	ng/g ww
Goby sp.	DDTs	North	10.91	10.91	10.91	10.91	NA	1	ng/g ww
Shiner perch	DDTs	North	4.08	15.67	20.87	22.05	10.05	3	ng/g ww
Spotted sand bass	DDTs	North	10.20	12.98	12.72	16.01	2.91	3	ng/g ww
Topsmelt	DDTs	North	6.45	8.90	8.55	11.69	2.64	3	ng/g ww

Table A3. Continued.

Common Name	Analyte	Region	Min	Mean	Median	Max	StdDev	Count	Units
Barred sand bass	DDTs	Central	4.82	12.94	12.94	21.06	11.48	2	ng/g ww
California halibut	DDTs	Central	4.82	9.31	9.31	13.48	3.20	7	ng/g ww
Deepbody anchovy	DDTs	Central	16.73	30.10	28.75	46.36	14.01	6	ng/g ww
Goby sp.	DDTs	Central	10.33	10.33	10.33	10.33	NA	1	ng/g ww
Shiner perch	DDTs	Central	12.48	14.38	14.38	16.27	2.68	2	ng/g ww
Slough anchovy	DDTs	Central	8.03	10.61	9.95	14.89	2.54	8	ng/g ww
Spotted sand bass	DDTs	Central	3.85	8.67	10.06	13.22	3.84	5	ng/g ww
Topsmelt	DDTs	Central	4.18	6.56	5.06	10.44	3.39	3	ng/g ww
Arrow goby	DDTs	South	0.05	0.05	0.05	0.05	NA	1	ng/g ww
Barred sand bass	DDTs	South	7.45	11.71	13.58	14.10	3.70	3	ng/g ww
California halibut	DDTs	South	8.64	11.84	11.93	15.56	2.25	6	ng/g ww
California killifish	DDTs	South	1.81	6.53	6.53	11.24	6.67	2	ng/g ww
Deepbody anchovy	DDTs	South	6.41	9.56	8.67	15.43	3.73	5	ng/g ww
Goby sp.	DDTs	South	10.62	10.62	10.62	10.62	NA	1	ng/g ww
Northern anchovy	DDTs	South	8.17	8.89	8.89	9.61	1.02	2	ng/g ww
Round stingray	DDTs	South	0.55	0.55	0.55	0.55	NA	1	ng/g ww
Shiner perch	DDTs	South	11.33	11.33	11.33	11.33	NA	1	ng/g ww
Slough anchovy	DDTs	South	6.19	18.54	18.54	30.88	17.46	2	ng/g ww
Spotted sand bass	DDTs	South	9.94	12.24	10.98	15.80	3.13	3	ng/g ww
Topsmelt	DDTs	South	3.69	5.08	3.79	7.76	2.32	3	ng/g ww
Barred sand bass	Dieldrin	North	0.05	0.05	0.05	0.05	0.00	4	ng/g ww
California halibut	Dieldrin	North	0.05	0.05	0.05	0.05	0.00	4	ng/g ww
Barred sand bass	Dieldrin	Central	0.05	0.05	0.05	0.05	0.00	2	ng/g ww
California halibut	Dieldrin	Central	0.05	0.05	0.05	0.05	0.00	3	ng/g ww
Spotted sand bass	Dieldrin	Central	0.05	0.05	0.05	0.05	0.00	2	ng/g ww
Barred sand bass	Dieldrin	South	0.05	0.05	0.05	0.05	0.00	3	ng/g ww
California halibut	Dieldrin	South	0.05	0.05	0.05	0.05	0.00	2	ng/g ww
Round stingray	Dieldrin	South	0.05	0.05	0.05	0.05	NA	1	ng/g ww
Barred sand bass	Mercury	North	0.06	0.07	0.07	0.07	0.01	4	ug/g ww
Black perch	Mercury	North	0.01	0.02	0.02	0.02	0.01	2	ug/g ww
California halibut	Mercury	North	0.02	0.04	0.05	0.05	0.01	7	ug/g ww
Shiner perch	Mercury	North	0.02	0.03	0.03	0.04	0.01	3	ug/g ww
Spotted sand bass	Mercury	North	0.06	0.07	0.06	0.10	0.03	3	ug/g ww
Topsmelt	Mercury	North	0.02	0.02	0.02	0.02	0.00	3	ug/g ww
Barred sand bass	Mercury	Central	0.06	0.06	0.06	0.07	0.00	2	ug/g ww
California halibut	Mercury	Central	0.04	0.07	0.05	0.24	0.07	7	ug/g ww
Deepbody anchovy	Mercury	Central	0.03	0.07	0.07	0.11	0.04	6	ug/g ww
Shiner perch	Mercury	Central	0.02	0.03	0.03	0.05	0.02	2	ug/g ww
Slough anchovy	Mercury	Central	0.02	0.03	0.03	0.05	0.01	7	ug/g ww
Spotted sand bass	Mercury	Central	0.09	0.10	0.11	0.12	0.02	5	ug/g ww
Topsmelt	Mercury	Central	0.03	0.05	0.05	0.06	0.02	3	ug/g ww

Table A3. Continued.

Common Name	Analyte	Region	Min	Mean	Median	Max	StdDev	Count	Units
Arrow goby	Mercury	South	0.05	0.05	0.05	0.05	NA	1	ug/g ww
Barred sand bass	Mercury	South	0.06	0.07	0.07	0.09	0.02	3	ug/g ww
California halibut	Mercury	South	0.02	0.05	0.05	0.06	0.02	6	ug/g ww
California killifish	Mercury	South	0.03	0.03	0.03	0.03	0.00	2	ug/g ww
Deepbody anchovy	Mercury	South	0.02	0.02	0.02	0.03	0.00	5	ug/g ww
Northern anchovy	Mercury	South	0.04	0.04	0.04	0.04	NA	1	ug/g ww
Round stingray	Mercury	South	0.15	0.15	0.15	0.15	NA	1	ug/g ww
Shiner perch	Mercury	South	0.01	0.01	0.01	0.01	NA	1	ug/g ww
Slough anchovy	Mercury	South	0.02	0.02	0.02	0.03	0.01	2	ug/g ww
Spotted sand bass	Mercury	South	0.06	0.10	0.08	0.15	0.05	3	ug/g ww
Topsmelt	Mercury	South	0.02	0.03	0.03	0.04	0.01	3	ug/g ww
Barred sand bass	PAHs	North	7.60	11.05	9.90	16.80	4.31	4	ng/g ww
Black perch	PAHs	North	15.02	25.79	25.79	36.57	15.24	2	ng/g ww
California halibut	PAHs	North	2.50	7.75	7.99	13.10	5.05	7	ng/g ww
Goby sp.	PAHs	North	91.80	91.80	91.80	91.80	NA	1	ng/g ww
Shiner perch	PAHs	North	19.80	23.23	24.64	25.27	2.99	3	ng/g ww
Spotted sand bass	PAHs	North	10.90	14.98	16.30	17.73	3.60	3	ng/g ww
Barred sand bass	PAHs	Central	6.20	7.50	7.50	8.80	1.84	2	ng/g ww
California halibut	PAHs	Central	1.00	9.64	9.86	23.27	8.00	7	ng/g ww
Goby sp.	PAHs	Central	50.90	50.90	50.90	50.90	NA	1	ng/g ww
Shiner perch	PAHs	Central	17.89	22.59	22.59	27.29	6.64	2	ng/g ww
Spotted sand bass	PAHs	Central	6.80	13.36	13.80	20.98	5.31	5	ng/g ww
Barred sand bass	PAHs	South	2.70	6.43	7.30	9.30	3.38	3	ng/g ww
California halibut	PAHs	South	1.00	12.87	13.63	22.45	7.04	6	ng/g ww
Goby sp.	PAHs	South	58.20	58.20	58.20	58.20	NA	1	ng/g ww
Round stingray	PAHs	South	30.30	30.30	30.30	30.30	NA	1	ng/g ww
Shiner perch	PAHs	South	21.61	21.61	21.61	21.61	NA	1	ng/g ww
Slough anchovy	PAHs	South	37.46	37.46	37.46	37.46	NA	1	ng/g ww
Spotted sand bass	PAHs	South	13.69	16.09	16.42	18.15	2.25	3	ng/g ww
Barred sand bass	PBDEs	North	2.04	4.14	3.18	8.17	2.83	4	ng/g ww
Black perch	PBDEs	North	2.86	5.02	5.02	7.19	3.06	2	ng/g ww
California halibut	PBDEs	North	0.81	1.45	1.37	2.38	0.52	7	ng/g ww
Goby sp.	PBDEs	North	4.97	4.97	4.97	4.97	NA	1	ng/g ww
Shiner perch	PBDEs	North	3.29	7.88	7.84	12.51	4.61	3	ng/g ww
Spotted sand bass	PBDEs	North	1.06	1.91	1.25	3.41	1.31	3	ng/g ww
Topsmelt	PBDEs	North	3.67	5.53	5.30	7.63	1.99	3	ng/g ww

Table A3. Continued.

Common Name	Analyte	Region	Min	Mean	Median	Max	StdDev	Count	Units
Barred sand bass	PBDEs	Central	2.92	4.97	4.97	7.02	2.90	2	ng/g ww
California halibut	PBDEs	Central	0.46	1.83	1.70	3.84	1.16	7	ng/g ww
Deepbody anchovy	PBDEs	Central	0.05	0.09	0.07	0.19	0.06	6	ng/g ww
Goby sp.	PBDEs	Central	1.98	1.98	1.98	1.98	NA	1	ng/g ww
Shiner perch	PBDEs	Central	8.81	10.29	10.29	11.77	2.10	2	ng/g ww
Slough anchovy	PBDEs	Central	0.05	0.95	0.32	3.36	1.35	8	ng/g ww
Spotted sand bass	PBDEs	Central	0.14	0.79	0.55	1.79	0.71	5	ng/g ww
Topsmelt	PBDEs	Central	1.49	6.33	2.31	15.20	7.69	3	ng/g ww
Arrow goby	PBDEs	South	19.53	19.53	19.53	19.53	NA	1	ng/g ww
Barred sand bass	PBDEs	South	1.51	1.90	1.79	2.41	0.46	3	ng/g ww
California halibut	PBDEs	South	0.45	2.09	1.56	4.19	1.67	6	ng/g ww
California killifish	PBDEs	South	0.92	2.64	2.64	4.36	2.43	2	ng/g ww
Deepbody anchovy	PBDEs	South	0.05	0.12	0.05	0.28	0.10	5	ng/g ww
Goby sp.	PBDEs	South	2.20	2.20	2.20	2.20	NA	1	ng/g ww
Northern anchovy	PBDEs	South	0.05	8.60	8.60	17.16	12.10	2	ng/g ww
Round stingray	PBDEs	South	10.08	10.08	10.08	10.08	NA	1	ng/g ww
Shiner perch	PBDEs	South	1.87	1.87	1.87	1.87	NA	1	ng/g ww
Slough anchovy	PBDEs	South	0.56	1.22	1.22	1.88	0.94	2	ng/g ww
Spotted sand bass	PBDEs	South	0.43	1.00	0.86	1.72	0.65	3	ng/g ww
Topsmelt	PBDEs	South	3.53	10.45	10.93	16.88	6.69	3	ng/g ww
Barred sand bass	PCBs	North	208.84	230.22	232.88	246.27	17.35	4	ng/g ww
Black perch	PCBs	North	47.23	239.37	239.37	431.51	271.72	2	ng/g ww
California halibut	PCBs	North	25.69	162.79	186.84	259.45	75.03	7	ng/g ww
Goby sp.	PCBs	North	265.13	265.13	265.13	265.13	NA	1	ng/g ww
Shiner perch	PCBs	North	29.95	182.71	130.05	388.14	184.81	3	ng/g ww
Spotted sand bass	PCBs	North	147.61	289.86	303.61	418.37	135.90	3	ng/g ww
Topsmelt	PCBs	North	98.53	146.09	129.81	209.93	57.46	3	ng/g ww
Barred sand bass	PCBs	Central	117.92	243.40	243.40	368.87	177.45	2	ng/g ww
California halibut	PCBs	Central	131.14	218.10	172.80	358.71	87.57	7	ng/g ww
Deepbody anchovy	PCBs	Central	243.18	356.39	358.82	458.46	111.00	6	ng/g ww
Goby sp.	PCBs	Central	327.20	327.20	327.20	327.20	NA	1	ng/g ww
Shiner perch	PCBs	Central	94.82	209.04	209.04	323.26	161.53	2	ng/g ww
Slough anchovy	PCBs	Central	217.52	260.13	258.52	322.02	37.41	8	ng/g ww
Spotted sand bass	PCBs	Central	148.49	320.29	342.16	570.60	168.40	5	ng/g ww
Topsmelt	PCBs	Central	147.89	198.17	195.48	251.15	51.68	3	ng/g ww

Table A3. Continued.

Common Name	Analyte	Region	Min	Mean	Median	Max	StdDev	Count	Units
Arrow goby	PCBs	South	98.30	98.30	98.30	98.30	NA	1	ng/g ww
Barred sand bass	PCBs	South	81.38	104.34	100.97	130.66	24.81	3	ng/g ww
California halibut	PCBs	South	67.95	143.79	161.57	190.68	45.89	6	ng/g ww
California killifish	PCBs	South	38.40	68.04	68.04	97.67	41.91	2	ng/g ww
Deepbody anchovy	PCBs	South	90.55	149.96	171.62	201.03	46.86	5	ng/g ww
Goby sp.	PCBs	South	54.22	54.22	54.22	54.22	NA	1	ng/g ww
Northern anchovy	PCBs	South	208.47	219.59	219.59	230.72	15.73	2	ng/g ww
Round stingray	PCBs	South	418.67	418.67	418.67	418.67	NA	1	ng/g ww
Shiner perch	PCBs	South	85.51	85.51	85.51	85.51	NA	1	ng/g ww
Slough anchovy	PCBs	South	197.15	199.93	199.93	202.72	3.94	2	ng/g ww
Spotted sand bass	PCBs	South	154.45	236.25	259.54	294.75	72.99	3	ng/g ww
Topsmelt	PCBs	South	69.34	98.62	92.24	134.29	32.94	3	ng/g ww

Table A4. Chemistry summary for birds.

Common Name	Analyte	Region	Min	Mean	Median	Max	StdDev	Count	Units
California least tern	Chlordanes	North	4.50	6.01	5.12	8.27	1.64	5	ng/g fw
Western gull	Chlordanes	North	0.14	1.68	1.26	4.10	1.54	8	ng/g fw
California least tern	Chlordanes	South	2.04	5.52	4.12	17.51	4.11	13	ng/g fw
Caspian tern	Chlordanes	South	2.14	9.53	7.51	24.64	6.58	10	ng/g fw
Double-crested cormorant	Chlordanes	South	0.04	1.37	0.88	3.49	1.44	8	ng/g fw
California least tern	DDTs	North	70.95	118.05	113.85	193.47	46.33	5	ng/g fw
Western gull	DDTs	North	127.73	425.62	348.53	1000.46	269.23	8	ng/g fw
California least tern	DDTs	South	43.32	92.38	86.88	187.35	40.87	13	ng/g fw
Caspian tern	DDTs	South	511.30	1477.72	1384.63	2765.57	866.31	10	ng/g fw
Double-crested cormorant	DDTs	South	293.54	1276.46	874.46	3643.55	1096.40	8	ng/g fw
California least tern	Mercury	North	0.18	0.22	0.24	0.27	0.04	5	ug/g fw
Western gull	Mercury	North	0.02	0.06	0.06	0.13	0.04	8	ug/g fw
California least tern	Mercury	South	0.11	0.17	0.17	0.21	0.04	12	ug/g fw
Caspian tern	Mercury	South	0.33	0.45	0.40	1.02	0.20	10	ug/g fw
Double-crested cormorant	Mercury	South	0.03	0.07	0.05	0.19	0.06	8	ug/g fw
California least tern	PBDEs	North	30.09	38.81	32.71	52.05	10.55	5	ng/g fw
Western gull	PBDEs	North	53.12	176.19	176.23	303.01	96.23	8	ng/g fw
California least tern	PBDEs	South	21.28	51.84	45.27	101.85	23.36	13	ng/g fw
Caspian tern	PBDEs	South	82.93	243.82	249.87	413.87	110.48	10	ng/g fw
Double-crested cormorant	PBDEs	South	8.77	88.86	29.62	280.29	99.13	8	ng/g fw
California least tern	PCBs	North	134.08	264.24	158.10	685.02	236.53	5	ng/g fw
Western gull	PCBs	North	251.85	517.52	482.90	856.91	233.87	8	ng/g fw
California least tern	PCBs	South	64.30	180.21	172.76	303.83	83.12	13	ng/g fw
Caspian tern	PCBs	South	133.57	478.90	424.03	1026.70	269.55	10	ng/g fw
Double-crested cormorant	PCBs	South	171.78	727.83	272.18	1950.77	715.49	8	ng/g fw
California least tern	PFCs	North	9.35	14.75	12.54	25.84	6.39	5	ng/g fw
California least tern	PFCs	South	12.61	32.28	27.12	72.53	20.71	12	ng/g fw
Caspian tern	PFCs	South	7.54	28.48	29.63	63.15	17.68	10	ng/g fw

Appendix B. BSAF summary

Table B1. BSAF summary for invertebrates.

Common Name	Analyte	Region	Mean	Median	Count
Crustacea	Chlordanes	North	1.43	0.11	5
Mollusks	Chlordanes	North	0.84	0.11	3
Polychaetes	Chlordanes	North	4.39	4.50	6
Crustacea	Chlordanes	Central	1.39	0.71	4
Mollusks	Chlordanes	Central	11.79	7.88	4
Polychaetes	Chlordanes	Central	155.70	11.02	6
Crustacea	Chlordanes	South	1.00	1.00	4
Mollusks	Chlordanes	South	2.40	1.00	4
Polychaetes	Chlordanes	South	7.16	1.00	6
Crustacea	DDTs	North	57.42	42.63	5
Mollusks	DDTs	North	89.41	61.17	3
Polychaetes	DDTs	North	56.93	53.44	6
Crustacea	DDTs	Central	51.79	49.46	4
Mollusks	DDTs	Central	49.22	45.49	4
Polychaetes	DDTs	Central	76.45	63.27	6
Crustacea	DDTs	South	23.15	19.12	4
Mollusks	DDTs	South	12.54	11.77	4
Polychaetes	DDTs	South	34.82	21.30	6
Crustacea	Dieldrin	North	1.00	1.00	3
Polychaetes	Dieldrin	North	1.00	1.00	3
Crustacea	Dieldrin	Central	1.00	1.00	2
Mollusks	Dieldrin	Central	1.00	1.00	1
Polychaetes	Dieldrin	Central	1.00	1.00	3
Crustacea	Dieldrin	South	1.00	1.00	2
Mollusks	Dieldrin	South	1.00	1.00	1
Polychaetes	Dieldrin	South	1.00	1.00	3
Crustacea	Mercury	North	0.06	0.07	4
Mollusks	Mercury	North	0.15	0.15	2
Polychaetes	Mercury	North	0.12	0.09	6
Crustacea	Mercury	Central	0.15	0.15	4
Mollusks	Mercury	Central	0.17	0.15	4
Polychaetes	Mercury	Central	0.62	0.39	6
Crustacea	Mercury	South	0.22	0.22	2
Mollusks	Mercury	South	0.18	0.17	4
Polychaetes	Mercury	South	0.55	0.32	4

Table B1. Continued.

Common Name	Analyte	Region	Mean	Median	Count
Crustacea	PAHs	North	0.20	0.26	5
Mollusks	PAHs	North	0.26	0.20	3
Polychaetes	PAHs	North	2.79	0.20	6
Crustacea	PAHs	Central	0.22	0.20	4
Mollusks	PAHs	Central	0.29	0.26	4
Polychaetes	PAHs	Central	0.40	0.40	6
Crustacea	PAHs	South	0.63	0.40	4
Mollusks	PAHs	South	0.31	0.30	4
Polychaetes	PAHs	South	0.98	0.99	6
Crustacea	PBDEs	North	1.06	1.15	5
Mollusks	PBDEs	North	0.93	0.44	3
Polychaetes	PBDEs	North	1.76	1.68	6
Crustacea	PBDEs	Central	1.27	1.25	4
Mollusks	PBDEs	Central	0.32	0.29	4
Polychaetes	PBDEs	Central	1.65	1.54	6
Crustacea	PBDEs	South	13.42	4.12	4
Mollusks	PBDEs	South	0.64	0.53	4
Polychaetes	PBDEs	South	4.19	3.57	6
Crustacea	PCBs	North	5.49	7.20	5
Mollusks	PCBs	North	1.99	2.02	3
Polychaetes	PCBs	North	5.79	6.43	6
Crustacea	PCBs	Central	13.05	11.42	4
Mollusks	PCBs	Central	10.66	8.66	4
Polychaetes	PCBs	Central	23.65	21.51	6
Crustacea	PCBs	South	12.18	8.99	4
Mollusks	PCBs	South	2.58	2.52	4
Polychaetes	PCBs	South	15.60	13.03	6

Table B2. BSAF Summary for fish.

Common Name	Analyte	Region	Mean	Median	Count
Barred sand bass	Chlordanes	North	6.01	6.27	4
Black perch	Chlordanes	North	7.28	7.28	2
California halibut	Chlordanes	North	2.48	2.00	7
Goby sp.	Chlordanes	North	2.80	2.80	1
Shiner perch	Chlordanes	North	8.08	6.54	3
Spotted sand bass	Chlordanes	North	7.35	7.08	3
Topsmelt	Chlordanes	North	4.94	4.20	3
Barred sand bass	Chlordanes	Central	24.29	24.29	2
California halibut	Chlordanes	Central	18.16	16.54	7
Deepbody anchovy	Chlordanes	Central	64.14	64.36	6
Goby sp.	Chlordanes	Central	6.43	6.43	1
Shiner perch	Chlordanes	Central	36.42	36.42	2
Slough anchovy	Chlordanes	Central	15.91	14.93	8
Spotted sand bass	Chlordanes	Central	36.70	41.54	5
Topsmelt	Chlordanes	Central	12.52	14.00	3
Arrow goby	Chlordanes	South	1.00	1.00	1
Barred sand bass	Chlordanes	South	14.80	4.80	3
California halibut	Chlordanes	South	19.21	16.78	6
California killifish	Chlordanes	South	1.00	1.00	2
Deepbody anchovy	Chlordanes	South	17.36	15.20	5
Goby sp.	Chlordanes	South	1.00	1.00	1
Northern anchovy	Chlordanes	South	1.00	1.00	2
Round stingray	Chlordanes	South	204.20	204.20	1
Shiner perch	Chlordanes	South	2.80	2.80	1
Slough anchovy	Chlordanes	South	36.32	36.32	2
Spotted sand bass	Chlordanes	South	52.41	52.65	3
Topsmelt	Chlordanes	South	14.20	11.60	3
Barred sand bass	DDTs	North	131.05	100.75	4
Black perch	DDTs	North	82.82	82.82	2
California halibut	DDTs	North	86.94	76.58	7
Goby sp.	DDTs	North	68.19	68.19	1
Shiner perch	DDTs	North	97.91	130.41	3
Spotted sand bass	DDTs	North	81.12	79.51	3
Topsmelt	DDTs	North	55.60	53.44	3

Table B2. Continued.

Common Name	Analyte	Region	Mean	Median	Count
Barred sand bass	DDTs	Central	184.86	184.86	2
California halibut	DDTs	Central	133.04	133.00	7
Deepbody anchovy	DDTs	Central	429.98	410.79	6
Goby sp.	DDTs	Central	147.57	147.57	1
Shiner perch	DDTs	Central	205.37	205.37	2
Slough anchovy	DDTs	Central	151.59	142.14	8
Spotted sand bass	DDTs	Central	123.87	143.71	5
Topsmelt	DDTs	Central	93.71	72.29	3
Arrow goby	DDTs	South	0.25	0.25	1
Barred sand bass	DDTs	South	58.55	67.90	3
California halibut	DDTs	South	59.22	59.66	6
California killifish	DDTs	South	32.63	32.63	2
Deepbody anchovy	DDTs	South	47.79	43.35	5
Goby sp.	DDTs	South	53.10	53.10	1
Northern anchovy	DDTs	South	44.45	44.45	2
Round stingray	DDTs	South	2.75	2.75	1
Shiner perch	DDTs	South	56.67	56.67	1
Slough anchovy	DDTs	South	92.68	92.68	2
Spotted sand bass	DDTs	South	61.21	54.91	3
Topsmelt	DDTs	South	25.40	18.95	3
Barred sand bass	Dieldrin	North	1.00	1.00	4
California halibut	Dieldrin	North	1.00	1.00	4
Barred sand bass	Dieldrin	Central	1.00	1.00	2
California halibut	Dieldrin	Central	1.00	1.00	3
Spotted sand bass	Dieldrin	Central	1.00	1.00	2
Barred sand bass	Dieldrin	South	1.00	1.00	3
California halibut	Dieldrin	South	1.00	1.00	2
Round stingray	Dieldrin	South	1.00	1.00	1
Barred sand bass	Mercury	North	0.14	0.14	4
Black perch	Mercury	North	0.04	0.04	2
California halibut	Mercury	North	0.09	0.10	7
Shiner perch	Mercury	North	0.06	0.06	3
Spotted sand bass	Mercury	North	0.16	0.13	3
Topsmelt	Mercury	North	0.04	0.04	3

Table B2. Continued.

Common Name	Analyte	Region	Mean	Median	Count
Barred sand bass	Mercury	Central	0.25	0.25	2
California halibut	Mercury	Central	0.28	0.17	7
Deepbody anchovy	Mercury	Central	0.27	0.27	6
Shiner perch	Mercury	Central	0.12	0.12	2
Slough anchovy	Mercury	Central	0.12	0.10	7
Spotted sand bass	Mercury	Central	0.40	0.41	5
Topsmelt	Mercury	Central	0.18	0.18	3
Arrow goby	Mercury	South	0.35	0.35	1
Barred sand bass	Mercury	South	0.54	0.53	3
California halibut	Mercury	South	0.35	0.36	6
California killifish	Mercury	South	0.20	0.20	2
Deepbody anchovy	Mercury	South	0.18	0.18	5
Northern anchovy	Mercury	South	0.27	0.27	1
Round stingray	Mercury	South	1.16	1.16	1
Shiner perch	Mercury	South	0.11	0.11	1
Slough anchovy	Mercury	South	0.19	0.19	2
Spotted sand bass	Mercury	South	0.75	0.61	3
Topsmelt	Mercury	South	0.23	0.24	3
Barred sand bass	PAHs	North	0.01	0.01	4
Black perch	PAHs	North	0.03	0.03	2
California halibut	PAHs	North	0.01	0.01	7
Goby sp.	PAHs	North	0.11	0.11	1
Shiner perch	PAHs	North	0.03	0.03	3
Spotted sand bass	PAHs	North	0.02	0.02	3
Barred sand bass	PAHs	Central	0.03	0.03	2
California halibut	PAHs	Central	0.04	0.04	7
Goby sp.	PAHs	Central	0.19	0.19	1
Shiner perch	PAHs	Central	0.09	0.09	2
Spotted sand bass	PAHs	Central	0.05	0.05	5
Barred sand bass	PAHs	South	0.05	0.05	3
California halibut	PAHs	South	0.09	0.10	6
Goby sp.	PAHs	South	0.42	0.42	1
Round stingray	PAHs	South	0.22	0.22	1
Shiner perch	PAHs	South	0.16	0.16	1
Slough anchovy	PAHs	South	0.27	0.27	1
Spotted sand bass	PAHs	South	0.12	0.12	3

Table B2. Continued.

Common Name	Analyte	Region	Mean	Median	Count
Barred sand bass	PBDEs	North	2.11	1.62	4
Black perch	PBDEs	North	2.56	2.56	2
California halibut	PBDEs	North	0.74	0.70	7
Goby sp.	PBDEs	North	2.54	2.54	1
Shiner perch	PBDEs	North	4.02	4.00	3
Spotted sand bass	PBDEs	North	0.97	0.64	3
Topsmelt	PBDEs	North	2.82	2.70	3
Barred sand bass	PBDEs	Central	1.50	1.50	2
California halibut	PBDEs	Central	0.55	0.51	7
Deepbody anchovy	PBDEs	Central	0.03	0.02	6
Goby sp.	PBDEs	Central	0.60	0.60	1
Shiner perch	PBDEs	Central	3.11	3.11	2
Slough anchovy	PBDEs	Central	0.29	0.10	8
Spotted sand bass	PBDEs	Central	0.24	0.17	5
Topsmelt	PBDEs	Central	1.91	0.70	3
Arrow goby	PBDEs	South	8.92	8.92	1
Barred sand bass	PBDEs	South	0.87	0.82	3
California halibut	PBDEs	South	0.95	0.71	6
California killifish	PBDEs	South	1.21	1.21	2
Deepbody anchovy	PBDEs	South	0.05	0.02	5
Goby sp.	PBDEs	South	1.00	1.00	1
Northern anchovy	PBDEs	South	3.93	3.93	2
Round stingray	PBDEs	South	4.60	4.60	1
Shiner perch	PBDEs	South	0.85	0.85	1
Slough anchovy	PBDEs	South	0.56	0.56	2
Spotted sand bass	PBDEs	South	0.46	0.39	3
Topsmelt	PBDEs	South	4.77	4.99	3

Table B2. Continued.

Common Name	Analyte	Region	Mean	Median	Count
Barred sand bass	PCBs	North	10.95	11.07	4
Black perch	PCBs	North	11.38	11.38	2
California halibut	PCBs	North	7.74	8.88	7
Goby sp.	PCBs	North	12.61	12.61	1
Shiner perch	PCBs	North	8.69	6.18	3
Spotted sand bass	PCBs	North	13.78	14.44	3
Topsmelt	PCBs	North	6.95	6.17	3
Barred sand bass	PCBs	Central	34.33	34.33	2
California halibut	PCBs	Central	30.76	24.37	7
Deepbody anchovy	PCBs	Central	50.27	50.61	6
Goby sp.	PCBs	Central	46.15	46.15	1
Shiner perch	PCBs	Central	29.48	29.48	2
Slough anchovy	PCBs	Central	36.69	36.46	8
Spotted sand bass	PCBs	Central	45.18	48.26	5
Topsmelt	PCBs	Central	27.95	27.57	3
Arrow goby	PCBs	South	19.86	19.86	1
Barred sand bass	PCBs	South	21.08	20.40	3
California halibut	PCBs	South	29.05	32.64	6
California killifish	PCBs	South	13.74	13.74	2
Deepbody anchovy	PCBs	South	30.29	34.67	5
Goby sp.	PCBs	South	10.95	10.95	1
Northern anchovy	PCBs	South	44.36	44.36	2
Round stingray	PCBs	South	84.58	84.58	1
Shiner perch	PCBs	South	17.27	17.27	1
Slough anchovy	PCBs	South	40.39	40.39	2
Spotted sand bass	PCBs	South	47.73	52.43	3
Topsmelt	PCBs	South	19.92	18.63	3

Table B3. BSAF summary for birds.

Common Name	Analyte	Region	Mean	Median	Count
California least tern	Chlordanes	North	13.66	11.64	5
Western gull	Chlordanes	North	3.82	2.87	8
California least tern	Chlordanes	South	110.33	82.37	13
Caspian tern	Chlordanes	South	190.52	150.21	10
Double-crested cormorant	Chlordanes	South	27.41	17.61	8
California least tern	DDTs	North	737.84	711.57	5
Western gull	DDTs	North	2660.14	2178.30	8
California least tern	DDTs	South	461.91	434.39	13
Caspian tern	DDTs	South	7388.60	6923.15	10
Double-crested cormorant	DDTs	South	6382.29	4372.28	8
California least tern	Mercury	North	0.49	0.52	5
Western gull	Mercury	North	0.13	0.14	8
California least tern	Mercury	South	1.29	1.29	12
Caspian tern	Mercury	South	3.47	3.04	10
Double-crested cormorant	Mercury	South	0.55	0.38	8
California least tern	PBDEs	North	19.80	16.69	5
Western gull	PBDEs	North	89.89	89.91	8
California least tern	PBDEs	South	23.67	20.67	13
Caspian tern	PBDEs	South	111.34	114.10	10
Double-crested cormorant	PBDEs	South	40.57	13.52	8
California least tern	PCBs	North	12.56	7.52	5
Western gull	PCBs	North	24.61	22.96	8
California least tern	PCBs	South	36.41	34.90	13
Caspian tern	PCBs	South	96.75	85.66	10
Double-crested cormorant	PCBs	South	147.04	54.99	8

Appendix C. Normalized BSAF summary

Table C1. Normalized BSAF summary for invertebrates.

Common Name	Analyte	Region	Mean	Median	Count
Crustacea	Chlordanes	North	10.72	0.30	5
Mollusks	Chlordanes	North	0.41	0.27	3
Polychaetes	Chlordanes	North	5.06	2.29	6
Crustacea	Chlordanes	Central	1.16	1.12	4
Mollusks	Chlordanes	Central	7.39	2.34	4
Polychaetes	Chlordanes	Central	136.67	1.45	6
Crustacea	Chlordanes	South	1.17	1.06	4
Mollusks	Chlordanes	South	2.21	1.95	4
Polychaetes	Chlordanes	South	1.06	0.52	6
Crustacea	DDTs	North	68.15	40.92	5
Mollusks	DDTs	North	61.80	8.97	3
Polychaetes	DDTs	North	37.34	21.74	6
Crustacea	DDTs	Central	71.92	27.42	4
Mollusks	DDTs	Central	26.91	10.95	4
Polychaetes	DDTs	Central	27.63	28.69	6
Crustacea	DDTs	South	115.70	41.44	4
Mollusks	DDTs	South	40.09	16.26	4
Polychaetes	DDTs	South	56.15	25.38	6
Crustacea	PBDEs	North	14.10	13.50	5
Mollusks	PBDEs	North	2.34	0.79	3
Polychaetes	PBDEs	North	16.64	6.87	6
Crustacea	PBDEs	Central	39.95	22.76	4
Mollusks	PBDEs	Central	10.62	0.39	4
Polychaetes	PBDEs	Central	42.05	25.69	6
Crustacea	PBDEs	South	27.32	4.80	4
Mollusks	PBDEs	South	2.36	1.05	4
Polychaetes	PBDEs	South	2.18	1.20	6
Crustacea	PCBs	North	8.13	6.12	5
Mollusks	PCBs	North	3.06	0.41	3
Polychaetes	PCBs	North	4.90	2.84	6
Crustacea	PCBs	Central	10.15	8.38	4
Mollusks	PCBs	Central	6.57	3.84	4
Polychaetes	PCBs	Central	7.81	5.27	6
Crustacea	PCBs	South	9.72	9.37	4
Mollusks	PCBs	South	3.44	3.23	4
Polychaetes	PCBs	South	6.94	7.84	6

Table C2. Normalized BSAF Summary for fish.

Common Name	Analyte	Region	Mean	Median	Count
Barred sand bass	Chlordanes	North	21.54	21.96	4
Black perch	Chlordanes	North	1.64	1.64	2
California halibut	Chlordanes	North	11.85	8.38	7
Shiner perch	Chlordanes	North	3.02	1.94	3
Spotted sand bass	Chlordanes	North	2.24	2.18	3
Topsmelt	Chlordanes	North	2.47	1.58	3
Barred sand bass	Chlordanes	Central	19.91	19.91	2
California halibut	Chlordanes	Central	20.16	8.78	7
Deepbody anchovy	Chlordanes	Central	39.07	38.00	6
Shiner perch	Chlordanes	Central	4.91	4.91	2
Slough anchovy	Chlordanes	Central	20.21	12.52	8
Spotted sand bass	Chlordanes	Central	19.22	6.45	5
Topsmelt	Chlordanes	Central	10.34	11.36	3
Arrow goby	Chlordanes	South	0.23	0.23	1
Barred sand bass	Chlordanes	South	16.67	5.03	3
California halibut	Chlordanes	South	16.99	3.25	6
California killifish	Chlordanes	South	0.50	0.50	2
Deepbody anchovy	Chlordanes	South	28.84	26.69	5
Northern anchovy	Chlordanes	South	0.59	0.59	2
Round stingray	Chlordanes	South	61.67	61.67	1
Shiner perch	Chlordanes	South	0.28	0.28	1
Slough anchovy	Chlordanes	South	3.58	3.58	2
Spotted sand bass	Chlordanes	South	3.23	3.53	3
Topsmelt	Chlordanes	South	13.19	9.85	3

Table C2. Continued.

Common Name	Analyte	Region	Mean	Median	Count
Barred sand bass	DDTs	North	200.88	141.34	4
Black perch	DDTs	North	7.46	7.46	2
California halibut	DDTs	North	190.65	143.16	7
Shiner perch	DDTs	North	7.71	5.20	3
Spotted sand bass	DDTs	North	7.93	7.69	3
Topsmelt	DDTs	North	26.42	25.39	3
Barred sand bass	DDTs	Central	151.52	151.52	2
California halibut	DDTs	Central	174.66	63.65	7
Deepbody anchovy	DDTs	Central	268.68	260.91	6
Shiner perch	DDTs	Central	26.92	26.92	2
Slough anchovy	DDTs	Central	181.77	112.83	8
Spotted sand bass	DDTs	Central	65.26	23.52	5
Topsmelt	DDTs	Central	75.32	62.50	3
Arrow goby	DDTs	South	0.06	0.06	1
Barred sand bass	DDTs	South	290.13	284.80	3
California halibut	DDTs	South	168.24	29.49	6
California killifish	DDTs	South	15.68	15.68	2
Deepbody anchovy	DDTs	South	331.52	300.67	5
Northern anchovy	DDTs	South	26.64	26.64	2
Round stingray	DDTs	South	3.32	3.32	1
Shiner perch	DDTs	South	22.38	22.38	1
Slough anchovy	DDTs	South	90.92	90.92	2
Spotted sand bass	DDTs	South	6.48	2.87	3
Topsmelt	DDTs	South	25.44	22.15	3

Table C2. Continued.

Common Name	Analyte	Region	Mean	Median	Count
Barred sand bass	PBDEs	North	23.69	23.49	4
Black perch	PBDEs	North	2.06	2.06	2
California halibut	PBDEs	North	14.83	17.24	7
Shiner perch	PBDEs	North	5.54	4.20	3
Spotted sand bass	PBDEs	North	0.68	0.76	3
Topsmelt	PBDEs	North	1.29	1.32	3
Barred sand bass	PBDEs	Central	58.22	58.22	2
California halibut	PBDEs	Central	37.57	0.74	7
Deepbody anchovy	PBDEs	Central	1.21	0.78	6
Shiner perch	PBDEs	Central	0.92	0.92	2
Slough anchovy	PBDEs	Central	3.43	1.08	8
Spotted sand bass	PBDEs	Central	6.76	0.46	5
Topsmelt	PBDEs	Central	1.45	0.55	3
Arrow goby	PBDEs	South	2.04	2.04	1
Barred sand bass	PBDEs	South	0.36	0.27	3
California halibut	PBDEs	South	0.56	0.54	6
California killifish	PBDEs	South	0.63	0.63	2
Deepbody anchovy	PBDEs	South	0.02	0.01	5
Northern anchovy	PBDEs	South	2.38	2.38	2
Round stingray	PBDEs	South	1.25	1.25	1
Shiner perch	PBDEs	South	0.84	0.84	1
Slough anchovy	PBDEs	South	0.10	0.10	2
Spotted sand bass	PBDEs	South	0.17	0.16	3
Topsmelt	PBDEs	South	5.05	4.75	3

Table C2. Continued.

Common Name	Analyte	Region	Mean	Median	Count
Barred sand bass	PCBs	North	13.60	13.82	4
Black perch	PCBs	North	2.40	2.40	2
California halibut	PCBs	North	16.18	10.77	7
Shiner perch	PCBs	North	0.46	0.49	3
Spotted sand bass	PCBs	North	4.46	1.29	3
Topsmelt	PCBs	North	3.55	3.44	3
Barred sand bass	PCBs	Central	29.74	29.74	2
California halibut	PCBs	Central	38.12	12.00	7
Deepbody anchovy	PCBs	Central	37.32	37.16	6
Shiner perch	PCBs	Central	4.53	4.53	2
Slough anchovy	PCBs	Central	44.01	45.73	8
Spotted sand bass	PCBs	Central	18.07	11.17	5
Topsmelt	PCBs	Central	22.00	23.11	3
Arrow goby	PCBs	South	4.55	4.55	1
Barred sand bass	PCBs	South	18.71	21.35	3
California halibut	PCBs	South	14.78	8.39	6
California killifish	PCBs	South	6.62	6.62	2
Deepbody anchovy	PCBs	South	29.60	28.16	5
Northern anchovy	PCBs	South	26.11	26.11	2
Round stingray	PCBs	South	25.49	25.49	1
Shiner perch	PCBs	South	0.81	0.81	1
Slough anchovy	PCBs	South	16.80	16.80	2
Spotted sand bass	PCBs	South	6.99	4.51	3
Topsmelt	PCBs	South	19.36	16.45	3

Appendix D. Reference Value Selection

Background on thresholds for concentrations in eggs

Contaminant levels measured in seabird eggs collected for this study were compared with levels associated with adverse effects in other studies of avian species. The amount of information on effect levels is variable, depending on the contaminant, and some may be field-based while others are laboratory-based. Effect levels may vary with species and effect, and often there are no data on effect levels for the species being studied. Consequently, contaminant levels reported for least terns, cormorants, Caspian terns and western gulls in this study were compared with both a No Observed Adverse Effect Concentrations (NOAEC) and ranges of Lowest Observed Adverse Effect Concentrations (LOAECs) for sensitive adverse effects relating to maintenance of viable populations (i.e., survival, growth and reproduction). Low ends of ranges were used for contaminants with multiple studies (and LOAECs) for an individual species, and as such are considered conservative estimates of thresholds for observed adverse effects in the species being tested. It is important to note that as derived, LOAECs are thresholds for sensitive adverse effects in chronically exposed sensitive species, and as such may overestimate risk to less sensitive species. Depending on available data, NOAECs are either known, or "bounded" (i.e., by a paired LOAEC) or estimated from a LOAEC. Bounded NOAECs were preferred, but in most cases only effect levels are reported. Consistent with approaches used by the U. S. Environmental Protection Agency (e. g., USEPA 1995), and depending on the available data, LOAECs were adjusted downward using an uncertainty factor between 2 and 10, to obtain an estimate of a NOAEC, and by another factor between 2 and 10 for uncertainty about species differences in sensitivity. The fact that NOAECs are for use to evaluate risks to waterbirds in this assessment was also considered in the selection of adjustment factors. The NOAECs thus derived, are considered conservative estimates of concentrations below which no adverse effects are expected for birds in general, or waterbirds specifically, and therefore serve as "screening levels" for identifying contaminants of potential concern. Some values are based on data from very few studies and as such may change as more data become available. Ultimately, NOAEC-based thresholds are considered screening levels below which there is no concern, but above which further consideration may be required. The LOAEC-based thresholds provide perspective on the potential for an observed contaminant concentration to be associated with a measurable adverse effect.

Background on Toxicity Reference Values (TRVs; doses)

TRVs may be No Observed Adverse Effects Levels (NOAELs) or Lowest Observed Adverse Effect Levels (LOAELs) for specific effects. NOAELS and LOAELs are obtained from laboratory or field studies in which concentrations of the contaminant in the diet are reported. Dietary NOAECs and LOAECs reported in applicable studies are converted to NOAELs and LOAELs using food ingestion rates (g_{food}/d) reported in the study (if reported) or estimated using allometric regressions from Nagy (2001).

Readily available sources of TRVs were considered for use in this assessment. Initial consideration was given to TRVs developed by the U.S. Navy and EPA Region 9 Biological Technical Assistance Group (Navy/BTAG TRVs; DTSC/HERD 2000 and 2009), followed by TRVs identified by EPA for ecological risk-based soil screening levels (Eco-SSLs; EPA 2003). Both sets were developed using a consensus process with multiple agencies, entailed comprehensive review of the available literature, and were subject to peer review. The Navy/BTAG TRVs were developed to provide consistency in the assessment of risks posed to wildlife at contaminated sites in California, while the TRVs for Eco-SSLs were developed to support ecological risk assessments in terrestrial habitats nationwide. Navy/BTAG identified two TRVs for each of several contaminants. A low end TRV (TRV-L) was developed to represent a lowest credible no adverse effect level (NOAEL), below which no observable adverse effect is expected for the wildlife receptor of concern. A high-end TRV (TRV-H) was identified as well. The TRV-H is a chronic LOAEL that falls in the middle of the range of LOAELs that were considered by Navy/BTAG and represents a daily dose rate beyond which adverse ecological effects are expected to occur. Only one TRV per receptor category was identified for Eco-SSLs. TRVs for Eco-SSLs are equal to the geometric mean of the NOAEL values for growth and reproductive effects. In cases where the geometric mean NOAEL is higher than the lowest bounded LOAEL for survival, growth or reproduction, the selected TRV is equal to the highest bounded NOAEL below the lowest bounded LOAEL. Both the Navy/BTAG and Eco-SSL TRVs have limitations. Most notably, values have been derived for only a few of contaminants considered in this assessment (i.e., mercury, PCBs and DDT), and, for some, more recent data and/or reviews provide further insight into species differences in sensitivity. In addition, studies used to derive TRVs for Eco-SSLs tend to be focused on terrestrial species. Therefore, while Navy/BTAG TRVs and Eco-SSL TRVs were the first considered, a number of alternate TRVs were derived as part of this assessment (using standard protocols) for contaminants lacking Navy/BTAG TRVs or for which there may be more recent applicable data.

The approach taken for selecting NOAELs and LOAELs was the same as that taken for identifying NOAECs and LOAECs for avian eggs. Selected TRVs include at least one NOAEL and at least one LOAEL for sensitive adverse effects relating to maintenance of viable populations (survival, growth, reproduction). The fact that LOAELs and NOAELs are being used to evaluate risks to waterbirds was considered in the selection of TRVs, and NOAELs and LOAELs from studies on waterbirds are noted if not included to provide context at least. For LOAEL-based TRVs, if multiple LOAELs were reported for a particular species exposed to a particular contaminant, the lowest of the LOAELs was considered along with lowest LOAELs for other species when deciding which to use for the risk assessment. Appropriate dose-response studies for most of the contaminants in this assessment are few in number, and LOAELs for at least two of the contaminants (PAHs and chlordanes) are from single studies. Lowest LOAELs of several are considered conservative estimates of thresholds for observable adverse effects in the species being tested, and in other species as well. The likelihood of observing adverse effects when lowest LOAEL is exceeded may still be low but increases as mid-range LOAELs (e.g., TRV-Hs) are approached. The extent to which a LOAEL based on only one study is conservative is unknown and those values are used with some caution.

For NOAELs, bounded values were preferred, but in most cases it was necessary to estimate NOAELs by adjusting a LOAEL downward by a factor between 2 and 10. Depending on the available data, additional adjustment factors between 2 and 10 were applied for uncertainty about species differences in sensitivity. Only studies using subchronic or chronic exposures were considered for TRVs, and results of those that were sub-chronic were adjusted by a factor between 2 and 10 for uncertainty about the level at which effects might occur with chronic exposure. Adopting the approach by Sample et al. (1996), studies with exposures spanning ten or more weeks were considered chronic. Studies in which exposures were less than ten weeks but spanned critical periods in development (e.g., as embryos) were also considered chronic. The resulting NOAEL-based TRVs are intended to be conservative estimates of daily doses below which no adverse effects are expected for any birds, and specifically for waterbirds. As such, the NOAEL-based TRVs serve as "screening levels" for identifying contaminants of potential concern (COPCs). It is recognized that some values are based on data from very few studies and consequently are applied with caution. Ultimately, for this assessment, NOAELs are screening levels below which there is no concern, but above which further consideration may be required. The LOAELs selected for this assessment provide perspective on the potential for an observed contaminant dose rate to be associated with a measurable adverse effect.

Screening levels (NOAECs, LOAECs, NOAELs and LOAELs) selected for this assessment are summarized in Tables D1 and D2, and the basis for selections are presented by contaminant in the following sections.

Effects and thresholds for mercury

Mercury has a high potential for bioaccumulation and is biomagnified through food chains. In birds, the highest concentrations occur in species that eat fish or other birds (Eisler 1987b). Methylmercury is the most stable form of mercury and the most toxic to wildlife. It is also the predominant form in fish (ATSDR 1999). Ingested methylmercury is readily accumulated in avian tissues, and in females, it is readily transferred to eggs. Dietary exposure to mercury may result in death from neurotoxic effects, but at relatively low dietary concentrations it adversely affects growth, development, motor coordination, behavior, and reproduction. Reproductive effects include reduced fertility, clutch size, egg hatchability, and chick survival. The reproductive effects are the most common adverse effects related to mercury concentrations in eggs.

Mercury in eggs

Effect levels for mercury in avian eggs have been most recently reviewed by Scheuhammer et al. (2007) and Shore et al. (2011), and include data for multiple species. Values selected for use in this study of seabird eggs are a combination of general guidelines, as well as LOAECs and NOAECs for piscivorous avian species, specifically snowy egret (*Egretta thula*), common loon (*Gavia immer*), and common tern (*Sterna hirundo*) (as cited by Shore et al. 2011). Few adjustments to the LOAEC were deemed necessary, given the available data, particularly on piscivorous birds. The lowest LOAEC for fish eating birds (egret) was adjusted downward by a factor of three for an estimated NOAEC. The thresholds and screening values used to assess mercury concentrations in seabird eggs for this study are summarized in Table D1 at the end of this chapter.

Mercury TRVs (dietary dose)

The Navy/BTAG (DTSC/HERD 2009) developed a TRV-L and a TRV-H for avian species based on reproductive effects in multiple studies of chronically exposed mallards (as summarized by USEPA 1995). Data from the studies on mallards may be considered the most robust on dietary doses of mercury associated with adverse effects on functions that can be related to reproductive success, such as survival, growth and reproduction (USFWS 2003). For the lowest credible NOAEL (TRV-L), Navy/BTAG used a LOAEL adopted by USEPA (1995), adjusted downward by a factor of two, resulting in a daily dose rate of 39 $ng_{mercury}/g_{BW}$-day. There was no further adjustment for species differences in sensitivity, presumably because mallards are a sensitive species. Following a re-evaluation of benchmark studies on mallards and consideration of species differences in sensitivity, USFWS (2003) recommended the use of two NOAELs for assessing risk; 21 $ng_{mercury}/g_{BW}$-day for less sensitive avian species (e.g., terns), and 7.0 $ng_{mercury}/g_{BW}$-d for potentially more sensitive species (e.g., waterfowl and marsh birds). Zhang et al. (2013) provide an updated review of data on mercury toxicity to birds, and included results of more recent studies on white ibis (*Eudocimus albus*), common loon, and great egret (*Ardea alba*). Of the species considered by Zhang et al. (2013), white ibis was most sensitive, exhibiting adverse reproductive effects as described by Frederick and Jayasena (2010) when exposed to mercury at an estimated daily rate of 10 $ng_{mercury}/g_{BW}$-d. The dose rate is a LOAEL which Zhang et al. (2013) adjusted downward by a factor of two to obtain an estimated NOAEL of 5.0 $ng_{mercury}/g_{BW}$-d for white ibis. Zhang et al. (2013) also suggested an even lower (5^{th} percentile) NOAEL for all effects in all avian species of 3.9 $ng_{mercury}/g_{BW}$-day . The lowest NOAEL recommended by Zhang et al. (2013) was used for initial screening, but all three NOAELs (two from Zhang et al. 2013 and one from USFWS 2003) were considered in follow-up risk characterization.

The TRV-H recommended by DTSC/HERD (2009) was used to evaluate potential for observable adverse effects in this assessment. As defined by Navy/BTAG the TRV-H for mercury (180 $ng_{mercury}/g_{BW}$-d) is a chronic mid-range LOAEL. The TRV-H is a LOAEL based on neurotoxicity and mortality in mallard chicks and by definition represents the mid-range LOAEL for all of the studies considered by Navy/BTAG (DTSC/HERD 2009). Compared with studies reviewed by Zhang et al. (2013) the TRV-H is in the middle of the range of lowest LOAELs for sensitive (e. g., immunological or reproductive) and/or severe effects (e. g., mortality) in all ten species for which there were data, and is 18 times greater than the LOAEL for reproductive effects in white ibis. All of the TRVs selected for assessing mercury risks to seabirds and waterfowl in this assessment are summarized in Table D2.

Effects and thresholds for DDT and metabolites

DDT is a legacy organochlorine pesticide that was manufactured and widely used between the early 1940s and 1972 for control of disease-carrying insects and insects on agricultural crops. Technical grade DDT is primarily a mix of *p,p'*-DDT (85%), and *o,p*'-DDT (15%). However, once introduced into the environment and detected in environmental media, DDT is found as a mixture with its principal metabolites DDD and DDE, and is that mixture of components that is associated with adverse environmental effects (ATSDR 2002, Blus 2011). While DDT and its metabolites are typically found together, and consequently referred to collectively as total DDT, DDE is the most common form found in avian tissues. DDT and its metabolites are persistent and have a high potential for bioaccumulation and biomagnification. Highest concentrations in aquatic food webs occur in mammals and in birds that consume fish or other birds (USDOI 1998). When ingested by birds, DDT and metabolites may accumulate in bird tissues, and they are readily transferred to eggs by the female parent. Dietary exposure by birds to DDT may result in death, reproductive impairment, reduced fledging success and eggshell thinning. In birds, the most common, and possibly most sensitive adverse reproductive effect associated with exposure to DDT, and especially its metabolite DDE, is eggshell thinning. Eggshell thinning may in turn lead to crushed eggs and nest failure.

DDTs in Eggs

Studies associating DDT concentrations in eggs with adverse effects have been conducted on numerous avian species (USDOI 1998, Blus 2011). Sensitivity to DDT can be highly variable depending on the species and the effect. USDOI (1998) and Blus (2011) provide ranges of thresholds for population-level effects and eggshell thinning with breakage in several avian species. The lowest threshold concentration for each species was used to rank species from most sensitive to least sensitive, and ranges were identified based on percentile rankings. The estimated NOAEC is based on the lowest threshold for sensitive species adjusted downward by a factor of three. Low-end thresholds (LOAECs) selected from the review by Blus (2011) for evaluating DDT levels in seabird eggs, with a focus on reduced productivity and critical eggshell thinning (18%), are summarized in Table D1.

DDT TRVs (dietary dose)

The Navy/BTAG (DTSC/HERD 2009) developed a TRV-L and a TRV-H for total DDTs (tDDT) in avian species, as derived by USEPA (1995). The TRVs were derived from results of studies on multiple species of chronically and sub-chronically exposed birds, including chickens, quail, pheasant, mallard, black duck, American kestrel, bald eagles and pelicans (USEPA 1995). The TRV-L is based on a LOAEL of 27 ng_{tDDTs}/g_{BW}-day for reduced productivity (nest success) in chronically exposed brown pelican, based on studies by Anderson et al (1975 and 1977). The brown pelican is one of the most sensitive of the bird species on which chronic exposure studies have been conducted (Blus 2011), so that adjustments for uncertainty about species differences in sensitivity were deemed unnecessary. The LOAEL for effects in pelicans was adjusted downward by a factor of three to obtain an estimated NOAEL of 9. 0 ng_{tDDTs}/g_{BW}-d (USEPA 1995).

The NOAEL-based TRV selected by USEPA (2007a) for calculating soil screening levels is 227 ng_{tDDTs}/g_{BW}-d), and is the highest bounded NOAEL lower than the lowest bounded LOAEL for reproduction, growth, and survival in studies on 22 species of birds. The selected TRV is for reduced growth in domestic chickens, an upland species, but is below LOAELs for growth, reproduction and survival for mallards, black duck, double-crested cormorants, white pelicans in addition to multiple upland and raptor species. Studies on brown pelican were not factored in to the analysis by EPA (2007a). Consequently, the TRV selected by EPA (2007a) is considered a low-end for all but most sensitive avian species.

The TRV-H recommended by DTSC/HERD (2009) was used to evaluate potential for observable adverse effects in this assessment. The TRV-H of 1,500 ng_{tDDTs}/g_{BW}-d is equal to a geometric mean of LOAELs identified by USEPA (1995) for eggshell thinning, reduced reproductive success and/or lethality in seven species of chronically exposed birds. For soil screening levels, EPA (2007a) evaluated results of dietary studies on 22 avian species, including mallards, American black duck, double-crested cormorant and white pelican. In their review, EPA (2007a) identified 112 LOAELs, and 123 NOAELs for effects on reproduction, growth and survival. The TRV-H adopted by Navy/BTAG (DTSC/HERD 2009) is greater than approximately one-third of all LOAELs, and approximately one-half of bounded LOAELs considered by EPA (2007a), and as such is clearly a mid-range LOAEL. All of the TRVs selected for assessing DDT risks to seabirds and waterfowl in this assessment are summarized in Table D2.

Effects and thresholds for total PCBs, PCB 126 and TEQs

PCBs constitute a synthetic mixture of up to 209 individual congeners. PCB mixtures were used extensively as coolants and lubricants in transformers, capacitors and other electrical equipment. PCB manufacture and uses of all kinds were banned in 1979, due to evidence of their persistence and environmental impacts. PCBs are still detected in environmental media worldwide (ATSDR 2000, Eisler 1986). PCBs are known to bioaccumulate and to biomagnify within the food chain, with highest concentrations occurring in mammals and in birds that consume fish (ATSDR 2000, Eisler 1986). Birds exposed to high levels of PCBs in the diet may exhibit signs of poisoning such as liver damage, morbidity, tremors and death. Sublethal levels of exposure disrupt normal growth patterns, reproduction, metabolism and behavior (Eisler 1986). Reproductive effects in adult birds include but are not limited to altered secondary sexual characteristics, reduced nest attentiveness, delayed egg laying, reduced clutch size and abnormal nest construction (Harris and Elliott 2011). In female birds, PCBs are readily transferred to eggs, where they may exert toxic effects on developing embryos. PCB-related effects associated with *in ovo* exposure to PCBs include deformities (embryo and hatchling), embryo lethality, cardiomyopathy, reduced hatching success, and reduced growth and survival of post-hatch nestlings (Harris and Elliott 2011, Carro et al. 2013). Sensitivity to total PCBs is highly variable depending on the species and the effect.

<u>*PCBs in Eggs*</u>

Harris and Elliott (2011) identified ranges of total PCB concentrations in eggs that are associated reduced hatching and/or fledging success (8 species), reduced productivity (3 species), and reduced parental care (2 species). Thresholds used to evaluate total PCB levels in seabird eggs from this study were selected using the low ends of ranges identified by Harris and Elliott (2011) for reproductive effects in high, intermediate and low sensitivity species. A NOAEC for all avian species (100 ng_{tPCBs}/g_{egg}ww) is based on the lowest LOAEC for highly sensitive species, adjusted downward by a factor of ten for a LOAEC/NOAEC extrapolation. An additional NOAEC that is more specific to waterbirds was identified because risk to waterbirds may be greatly overestimated by screening levels based on highly sensitive species. The estimated NOAEC for waterbirds (2,300 ng_{tPCBs}/g_{egg} fw) is based on a lowest LOAEC for terns, gulls and raptors, adjusted downward by a factor of ten for LOAEC/NOAEC extrapolation. Using NOAECs for both most sensitive species and waterbirds allows for some consideration of species that may fall between the two groups with respect to sensitivity. Multiple LOAECs were considered for this assessment. Those that were selected are the lowest in ranges identified by Harris and Elliott (2011) for high, intermediate and low sensitivity species. Thresholds used for total PCBs in this study on seabirds and waterfowl in this assessment are summarized in Table D1.

<u>*PCB 126 and TEQs in Eggs*</u>

PCB congeners with chlorines in the non- or mono-ortho position have dioxin-like toxicity, with varying potencies relative to the standard for dioxins, which is TCDD. Toxicity equivalent factors can be applied to all twelve of the congeners to estimate PCB-based TCDD equivalent concentrations (TEQs). The most potent congeners and dominant contributors to PCB's dioxin-like toxicity in avian eggs are PCB numbers 77, 81, 126 and 169. Some studies on PCB toxicity to birds have focused on individual PCB congeners, primarily PCB 126 and PCB 77. The response most often assessed with TCDD or dioxin-like PCB congeners is induction of enzymes that are mediated by the aryl hydrocarbon receptor (AhR). Enzyme induction is considered a measure of exposure that most directly relates to toxic effects observed at the organismal level, and is the basis of the assigned TEFs. At the organismal level, the adverse effects associated with TCDD or dioxin-like PCB congeners in avian eggs include embryo edema, skeletal abnormalities including beak deformities, heart malformations, and embryo lethality (Harris and Elliott 2011, Carro et al. 2013). Along with total PCBs, Harris and Elliott (2011) identified effect ranges for PCB 126 and TCDD (and TEQs). Thresholds used to evaluate PCB126 and TEQ concentrations in seabird eggs were selected using the low ends of ranges identified by Harris and Elliott (2011) for effect thresholds associated with embryo lethality for high, intermediate and low sensitivity species. Estimated NOAECs were derived using the same factors as for total PCBs, and are summarized along with LOAECs in Table D1.

Data on potential thresholds for PCB# 77 are very limited, but where available, concentrations of PCB #77 associated with adverse effects are higher than corresponding effect levels of PCB # 126.

PCB TRVs (dietary dose)

The Navy/BTAG (DTSC/HERD 2009) developed TRVs for total PCBs. The TRV-L is 90 ng_{tPCB}/g_{BW}-d and is based on a LOAEL for reduced egg production and egg fertility in chronically exposed leghorn chickens (Platonow and Reinhart 1973). The TRV-H is 1,270 ng_{tPCB}/g_{BW}-d and is a LOAEL based on reduced hatchability of eggs produced by exposed chickens (Britton and Huston 1973, as described by USEPA 1995). Specific details of how the Navy/BTAG-TRVs were derived are not provided. However, using exposure assumptions from Chiba (2014) for leghorn chickens, the BTAG TRV-L is an estimated NOAEL, based on a LOAEL adjusted downward by a factor of three. As defined, the TRV-H is a LOAEL that falls in the middle of the range of LOAELs considered by Navy/BTAG (DTSC/HERD 2009), as well as for LOAELs considered by USEPA (2000) and Su et al. (2014) for impacts on nine species of birds. The mallard is the only aquatic-dependent species represented in the analysis by EPA (2000), or in other reviews of potential avian TRVs for total PCBs (e.g., Su et al. 2014). If, as in eggs, total PCBs in the diet are less toxic to seabirds and waterfowl than to gallinaceous species (e.g., chicken and pheasant), then the TRV-H may overestimate the likelihood of observable impacts of PCB exposure on aquatic-dependent avian species, such as those considered in this assessment. Unfortunately, the ability to derive alternative LOAEL-based TRVs more specific to seabirds is hampered by lack of applicable data. Consequently, only two TRVs were selected for use in the assessment of risk posed by total PCBs to San Diego Bay seabirds and waterfowl (Table D2).

Because data on dietary effect levels for PCBs 126 and 77 are limited, risks posed by dietary exposure to these two congeners were addressed in combination with ten other dioxin-like congeners as TEQs (below).

TEQ TRVs (dietary dose)

There are no Navy/BTAG or EPA ECO-SSL TRVs for TEQs, and studies on effects of dietary exposure to specific PCB congeners are few. Most studies of dietary exposure to PCBs have entailed the use of one or more commercial Aroclor mixtures (total PCBs). Relative contributions of the dioxin-like congeners for the most common Aroclors have been roughly characterized as $ug_{TEQ}/g_{Aroclor}$ by the Canadian Council of Ministers of the Environment (CCME 2001). Using a process adopted by CCME (2001), Su et al. (2014) converted results of diet studies with Aroclor mixtures from NOAECs and LOAECs as total PCBs to NOAECs and LOAECs as TEQs. In so doing, Su et al. (2014) identified NOAECs and LOAECs as dietary TEQ concentrations for effects on growth, reproduction and/or survival in nine species of birds. Using the authors' exposure factors, NOAECs and LOAECs from Su et al. (2014) were converted to NOAELs and LOAELs for this assessment. The lowest NOAEL from the study by Su et al. (2014; 0.0011 ng_{TEQ}/g_{BW}-day) is an adjusted lowest LOAEL (0. 0060 ng_{TEQ}/g_{BW}-day). The LOAEL (and estimated NOAEL) are from a study in which chickens experienced reductions in egg production and hatchability (Su et al. 2014). The exposure duration in the benchmark study is considered borderline chronic/subchronic. As is, the estimated NOAEL is 5-times lower than the lowest LOAEL for all species and 5-times lower than the lowest NOAEL identified by Su et al. (2014) for species other than chickens. Consequently, the estimated lowest NOAEL is considered conservative and was not adjusted for uncertainty about chronic/subchronic exposure.

Two LOAEL-based TRVs from the review by Su et al (2014) were selected for assessing PCB TEQ risks to waterfowl and seabirds. For the species considered by Su et al. (2014), there is a 683-fold difference between the lowest LOAEL (for chicken) and the highest LOAEL (bobwhite quail). The lowest LOAEL for species other than chickens (0.0495 ng_{TEQ}/g_{BW}-day), and a mid-range (geometric mean) LOAEL for all species (0.178 ng_{TEQ}/g_{BW}-day) considered by Su et al. (2014) were selected for use as TRVs in this assessment. As is the case for total PCBs, the mallard is the only aquatic-dependent species represented in studies on growth or reproductive effects of subchronic or chronic exposure by birds to PCBs in the diet (e.g., Su et al. 2014; USEPA 1995; CCME 2001; USEPA 2000), and LOAELs for mallard tend to be at the upper end of the range of LOAELs for species that have been tested (Su et al. 2014; USEPA 2000). Consequently, the mid-range LOAEL selected for this assessment may overestimate the likelihood of observable impacts of PCB TEQ on waterfowl and seabirds in this assessment. The TRVs selected for assessing risks posed by dioxin-like PCB congeners (as TEQs) are summarized in Table D2.

Effects and thresholds for total PBDEs

PBDEs are flame-retardant chemicals that are added to many types of consumer products to reduce potential for burning. PBDEs came in to use in the 1970s as formulations of three products (i.e., penta-, octa-, and decabromodiphenyl ether). They are ubiquitous and persistent in environmental media (ATSDR 2004), which raised concerns that resulted in phasing out production and use of penta- and octa-BDEs in 2004/2005 and of deca-BDE in 2013 (USEPA 2014). Lower brominated PBDEs (e.g., penta-BDE) are known to bioaccumulate and biomagnify in aquatic food chains, with highest concentrations occurring in fish (e.g., Wan et al. 2008). Birds exposed to PBDE in the diet may experience changes in courtship behavior and delayed clutch initiation. Females may produce smaller eggs with thinner shells and low-weight embryos, for an overall reduction in hatching success (Fernie et al. 2009, Harris and Elliott 2011). In female birds, dietary PBDEs are readily transferred to eggs, where they may exert toxic effects on developing embryos and post-hatch offspring. Adverse effects associated with PBDE exposure by embryos include reduced pipping and hatching success, and reproductive impairments in male offspring (Fernie et al. 2009; Marteinson et al. 2010). There is limited evidence of altered blood thyroid hormone homeostasis and/or vitamin A stores, which in turn may affect development, immuno-competence, reproductive success and other physiological processes (Fernie et al. 2008, Fernie et al. 2009). However, the occurrence of thyroid involvement, especially in wild birds is under debate (McKernan et al. 2009, Harris and Elliott 2011).

PBDEs in eggs

Studies relating PBDE levels in avian eggs to adverse effects are very few in number and effect levels have only been reached in studies on American kestrel. However, results of egg injection studies by McKernan et al. (2009) and Rattner et al. (2011) indicate that exposure levels associated with adverse reproductive effects in kestrels are not sufficient to cause similar effects in chickens, mallards or common tern, and therefore that American kestrel represents a sensitive species. The thresholds considered for use in this assessment are based on reduced pipping and hatching success for American kestrel in egg injection studies by McKernan et al. (2009), and for impaired reproductive behavior, reduced clutch size and fertility of eggs produced through pairings with males exposed to PBDEs via transfer from female parent to egg (Marteinson et al. 2010; Fernie et al 2009). Results of the study by McKernan et al. (2009) provide both a bounded NOAEC (180 ng_{PBDE}/g fw) and LOAEC (1,800 ng_{PBDE}/g fw). Results of multigenerational studies suggest a LOAEC of 288 ng_{PBDE}/g fw for multigenerational reproductive effects from exposure to PBDE (via parental transfer) as an embryo (Marteinson et al. 2010). Both sets of studies appear to involve a sensitive avian species and chronic exposure. The NOAEC from the egg injection study is about one-half the LOAEC from the multigenerational study, which is consistent with dose-response relationships for chronic exposure and sensitive effects (Sample et al. 1996). Consequently, the bounded NOAEC from McKernan et al. (2009) is considered suitable as an initial screening level for sensitive effects in avian species. At 288 ng_{PBDE}/g fw, the LOAEC obtained by Fernie et al. (2009) and Marteinson et al. (2010) with embryos exposed to maternally transferred PBDEs is between the bounded NOAEC and LOAEC from egg injection studies (McKernan et al. 2009), and consequently is considered an accurate representation of a lowest LOAEC for reproductive effects in a sensitive avian species. Unfortunately, data on LOAECs for species other than American kestrel are lacking. However, results of egg injection studies by McKernan et al. (2009) and Rattner et al. (2011) indicate that the NOAECs for embryo survival and egg hatchability is greater than 5,900 ng_{PBDE}/g fw for chickens and approximately 20,000 ng_{PBDE}/g fw for mallards and common terns. Unfortunately, without actual LOAECs, it is not possible to assign a particular screening value for species other than the kestrel. The screening levels used to evaluate PBDE concentrations in seabird and waterfowl eggs for this study are summarized in Table D1, and given the available information, the screening levels are likely to overestimate risks to other species.

PBDE TRVs (dietary dose)

Studies on dietary exposure by birds to PBDEs are limited to the series by Fernie et al. (2008, 2009) and Marteinson et al. (2010) on American kestrel. All three studies are based on daily exposure by adult kestrels to 300 or 1,600 ng_{PBDEs}/g_{diet} over a period preceding and through the nesting season. Directly exposed adults, and their offspring (exposed *in ovo*) were monitored for a variety of endpoints relating to survival, growth and reproduction. The lowest concentration used is a dietary LOAEC for impacts on reproductive behaviors and success in directly exposed adults and/or their offspring (Fernie et al. 2008 and 2009, Marteinson et al. 2010). The dietary

LOAEC was converted to a LOAEL using a wet weight-based food ingestion rate of 0. 31 g_{food}/g_{BW}-d for American kestrels from USEPA (1993), resulting in a LOAEL of 96.0 ng_{PBDE}/g_{BW}-d. Unfortunately, data on NOAELs for dietary exposure to PBDEs are lacking. Consequently, the LOAEL from studies by Fernie et al. (2008 and 2009) and Marteinson et al. (2010) was adjusted by a factor of ten producing an estimated NOAEL of 9.6 ng_{PBDE}/g_{BW}-d for reproductive effects in a sensitive avian species. The TRVs selected for assessing risks posed by PBDEs in this assessment are as summarized in Table D2. Absent more robust data on PBDE dose and response in avian species, the NOAELs and LOAELs used for PBDEs in this assessment are applied with extra caution.

Effects and thresholds for PFCs (PFOS)

PFCs are a family of man-made compounds that have been used extensively since the late 1940s in industrial, commercial and consumer applications, including as water, grease and soil repellents for textiles and paper products, industrial surfactants and emulsifiers, fire-fighting foams, aids in the manufacture of non-stick coatings, and in metal plating and cleaning (ATSDR 2015). PFCs are ubiquitous and persistent in environmental media. PFCs released into aquatic environments bioaccumulate and biomagnify in food web organisms, with highest concentrations occurring in tissues of birds that eat fish or other birds (ATSDR 2015, Houde et al. 2006). Once ingested, PFCs bind to protein albumin in blood and liver. In female birds, PFCs are readily transferred to eggs, in association with the albumin (ATSDR 2015). While all of the PFCs considered in this study have been detected in a range of biota (e.g., Sedlak and Greig 2012), PFOS is the predominant PFC found in wildlife tissues (Houde et al. 2006 and 2011). Consequently, studies on the potential toxicity of PFCs in wildlife have been focused on PFOS, and for birds, only in a few species. For birds, adverse effects associated with dietary exposure to PFOS include overt signs of poisoning (ruffled appearance, reduced reaction to stimuli, loss of coordination, weakness and death), and sublethal effects such as reduced body weight and impacts on testes in males (Newsted et al. 2005, Newsted et al. 2006). Adverse effects associated with PFCs in eggs include reduced hatching success, liver pathology, and altered immune function (DeWitt et al. 2012, Molina et al. 2006).

PFCs (PFOS) in eggs

Newsted et al. (2005) evaluated effect levels in eggs of northern bobwhite quail and mallard, while Molina et al. (2006) studied effect levels for domestic chicken. The LOAEC for hatchability in chickens (Molina et al. 2006) is lower than the LOAEC for hatchability in quail or mallards. However, the study by Molina et al. (2006) used egg injection, which may overestimate toxicity, while studies by Newsted et al. (2005) were diet based and more representative of actual exposure. Using guidance from the European Commission and data from studies on northern bobwhite quail and mallard, Newsted et al. (2005) derived NOAECs and LOAECs for assessing risks posed by PFOS in eggs of top avian predators. The authors identified a LOAEC for offspring survival in the more sensitive of two species (bobwhite quail) and adjusted it downward by a factor of two for LOAEC to NOAEC extrapolation, combined with a factor of 30 for uncertainty on interspecies variation, endpoint extrapolation and laboratory-to-field extrapolation. Given the more direct applicability of the exposure route used, and the conservative uncertainty factors that were employed by Newsted et al. (2005), values derived by those authors, which were reported as μg/mL were used for this study on seabirds. Assuming a density of 1.0 g/mL for avian eggs, values from Newsted et al. (2005) were converted to a NOAEC of 1,000 ng_{PFOS}/g_{egg} ww and LOAEC of 62,000 ng_{PFOS}/g_{egg} ww (Table D1). Like PBDEs, absent more robust data on dose and response in avian species, the NOAECs and LOAECs used for PFOS in this assessment are applied with extra caution.

PFCs (PFOS) TRVs (dietary dose)

Food web samples were not analyzed for PFCs, and therefore TRVs were not developed for PFCs.

Effects and thresholds for chlordane

Chlordane is a legacy OC pesticide widely used as a pesticide for control of agricultural pests, termites, and residential lawn and garden pests. All uses of chlordane ceased in 1988 (ATSDR 1994). Technical chlordane is a mixture of more than 45 components, but mostly *cis*-chlordane, *trans*-chlordane, heptachlor, and *cis*- and *trans*-nonachlors (ATSDR 1994, Eisler 1990). Chlordane is persistent in the environment, and chlordane released into aquatic environments is bioaccumulated in aquatic food web organisms, but food web biomagnification may be low, except for marine mammals (Eisler 1990). Birds exposed to chlordanes in the diet will accumulate them, especially nonachlors and the metabolite, oxychlordane in tissues with high fat content. Adverse effects associated with ingestion of chlordanes are primarily intoxication (e.g., sluggishness, weight loss, panting and tremors) and death, believed to be due primarily to oxychlordane in the brain (Eisler 1990). In females, ingested chlordanes are readily transferred to eggs, with the nonachlors and oxychlordane dominating (e.g., Zeeman et al. 2008).

Chlordane in eggs

Unfortunately, data on chlordane effect levels in eggs are lacking (Elliott and Bishop 2011, Wiemeyer 1996). Consequently, while chlordanes are routinely detected in seabird eggs, there are no benchmarks for assessing the potential for adverse effects at observed concentrations.

Chlordane TRVs (dietary dose)

While ingestion of chlordane has been linked with lethality in numerous avian species in the field (Elliott and Bishop 2011), few studies have been conducted that provide dietary dose-response data on chlordane toxicity to birds. In most of the dietary dosing studies, the toxic responses (usually lethality) were related to chlordane concentrations in the brain (not diet) and dietary thresholds are not provided. Consequently, the TRVs used for this assessment are based on a single study by Stickel et al. (1983), in which the authors report dietary thresholds for survival in red-winged blackbirds. Stickel et al. (1983) exposed birds to chlordane mixture in the diet for 84 days over which lethality was observed among birds fed 50,000 $ng_{chlordane}/g_{diet}$ but not at 10,000 $ng_{chlordane}/g_{diet}$. The dietary concentrations were converted to daily dose rates using a food ingestion rate of 0.14 g_{food}/g_{BW}-d, based on body weight for red-winged blackbirds from Stickel et al. (1983) and the allometric regression for omnivores from Nagy (2001). The resulting estimated dose rates are a NOAEL of 1,400 $ng_{chlordane}/g_{BW}$-d and a LOAEL of 7,000 $ng_{chlordane}/g_{BW}$-d for reduced survival (LOAEL). Based on acute lethality, the sensitivity of red-winged blackbirds to chlordane in the diet is comparable to that of Northern bobwhite, Japanese quail and ring-necked pheasant and may be slightly greater than that of European starlings and mallard, with overall effect levels ranging from 200,000 $ng_{chlordane}/g_{diet}$ to 858,000 $ng_{chlordane}/g_{diet}$ (Eisler 1990). The range of dietary concentrations associated with acute lethality is fairly narrow, and might be broader if other data on more species were available. Considering data on acute lethality in multiple species, but absent chronic data for other species, the red-winged blackbird is tentatively considered an average or more sensitive species to chlordane toxicity. At twelve weeks, the exposure duration for the study by Stickel et al. (1983) may be considered borderline chronic, raising some uncertainty about effect levels under longer term exposure conditions. Given the borderline subchronic/chronic exposure duration and uncertainty about species differences in sensitivity, the NOAEL for effects on red-winged blackbirds was adjusted by a factor of nine (three for each type of uncertainty). The TRVs selected for assessing risks posed by chlordanes (as the sum of *cis*- and *trans*-chlordane, *cis*- and *trans*-nonachlor, and oxyhclordane) are summarized in Table D2.

Being based on results of one study only, the TRVs selected to assess risks posed by exposure to chlordanes are applied with extra caution.

Effects and thresholds for PAHs

Polynuclear (aka polycyclic) aromatic hydrocarbons (PAHs) are a group of compounds formed during incomplete combustion of coal, petroleum products, wood, garbage and other organic materials. They are commonly found in materials such as asphalt used for road construction, crude oil, coal, coal tar pitch, creosote, and roofing tar. Consequently, PAHs are ubiquitous in environmental media, usually attached to particles in air, soil or sediment. There are thousands of PAH compounds, each differing in the number and position of aromatic rings, and in the type and position of substituents on the basic ring structure. Environmental concerns have been focused on those that range in size from two ring compunds (e.g., naphthalene) to seven ring compounds, and with few if any substitutions (alkyl group or other radical attached to the ring). PAHs in the 2 to 7 ring size range are divided into two categories as a means to address differences in physical/chemical properties that influence fate and toxicity. Those containing 2 or 3 rings are considered LPAHs and are typically associated with adverse effects such as narcosis, gastrointestinal inflammation, anemia, poor growth and impaired kidney function. The 4 to 7 ring compounds are HPAHs, and may have the same effects as LPAHs, but also many are demonstrably carcinogenic, mutagenic, or teratogenic (Eisler 1987a).

PAHs are relatively persistent in the environment. Once in an aquatic environment PAHs may be bioaccumulated by food web organisms, but with little to no biomagnification occurring. Greatest accumulations of PAHs occur in tissues of algae, mollusks and other species that are incapable of metabolizing PAHs (Eisler 1987a). In birds, PAHs absorbed from the diet are filtered by the liver, where they are rapidly metabolized and excreted. Because of their rapid metabolism, PAHs rarely accumulate to detectable levels in avian tissues. However, PAHs may be detected in livers, and other tissues of birds that live in moderate to highly contaminated sites (Malcom and Shore 2003; Albers and Loughlin 2003). Depending on the fraction, adverse effects associated with ingestion of PAHs include gastrointestinal inflammation, anemia, impaired kidney function, poor growth, depressed immune function, and decreased egg production and fertility. PAHs absorbed by female birds may be transferred to eggs. However it is difficult to discern if the low concentrations observed in wild eggs are from maternal transfer or from application to the shell by oiled feathers of the brooding parent. Adverse effects associated with PAH exposure by embryos include embryo lethality, liver necrosis, kidney lesions, edema, poor growth and teratogenicity (Malcom and Shore 2003).

PAHs in eggs

Avian eggs were not analyzed for PAHs in this study, and therefore screening levels for PAHs in eggs were not considered further.

PAHs TRVs (dietary doses)

Dose-response studies on exposure of wildlife to PAHs in the diet are typically focused on specific constituents, generally the most toxic, selected to represent PAH mixtures as a whole. Absent data for more precise approaches, results obtained with most toxic constituents are applied to entire mixtures with no consideration of the relative toxicity of less toxic compounds (e.g., USEPA 2007b).

Studies on chronic (or subchronic) exposure by birds to PAHs in the diet are very few in number (Douben 2003). Only two potentially relevant studies for deriving LPAH TRVs were located; one study on mallards (Patton and Dieter 1980) exposed to a mixture of LPAHs (controls and two treatment levels) and one study on Japanese quail exposed to naphthalene (controls and three treatment levels) (Klasing 2007). Patton and Dieter (1980) observed no effects on survival, growth or reproduction in mallards exposed to dietary LPAHs at any of the concentrations tested, resulting no LOAECs but an unbounded NOAEC 4,000 $\mu g_{LPAHs}/g_{diet}$. Klasing (2007) monitored multiple physiological, survival, growth and reproductive endpoints in adult quail exposed over 14 weeks to naphthalene in the diet. Of the effects that are more readily related to population level impacts (survival, growth and reproduction), growth (as weight gain) was the most sensitive. Results from Klasing (2007) provide both a bounded dietary NOAEC (30,000 $ng_{naphthalene}/g_{diet}$) and LOAEC (48,000 $ng_{naphthalene}/g_{diet}$) for naphthalene effects on weight gain and food consumption by adult quail. Using the food ingestion rate measured by Klasing (2007; 0.098 g_{food}/g_{BW}-d), and assuming that naphthalene is typical of other LPAHs (EPA 2007b), the estimated NOAEL and LOAEL for effects of LPAHs on weight gain in quail are respectively 2,950 ng_{LPAHs}/g_{BW}-d and 4,730 ng_{LPAHs}/g_{BW}-d. Data on chronic dietary exposure by other species to LPAHs are lacking. Consequently, the bounded NOAEL for weight gain in quail was adjusted by a factor of ten for uncertainty about species differences in sensitivity, resulting in an estimated NOAEL of 295 ng_{LPAHs}/g_{BW}-d (Table D2).

Three potentially relevant studies for deriving HPAH TRVs for effects on survival, growth or reproduction were located; one study on Northern bobwhite quail exposed to Benz(a)anthracene (BaA) in the diet (Brausch et al. 2010), another of European starlings (adults and nestlings) exposed via gavage to 7,12-Dimethylbenz(a)anthracene (DMBA; Trust et al. 1994), and a third of two pigeon strains exposed via injection to Benzo(a)pyrene (BaP) or Benzo(e)pyrene (Hough et al. 1993). Brausch et al. (2010) observed no effects on survival, growth or reproduction in bobwhite quail exposed over two months to dietary BaA at any of the levels tested, which translates into an unbounded chronic dietary NOAEC of 11,500 ng_{BaA}/g_{food} (~2,300 ng_{BaA}/g_{BW}-d) and no LOAEC. Some exposure levels were associated with biochemical effects and impacts on the liver, however. Trust et al. (1994) reported numerous biochemical and physiological responses to European starling adults and nestlings exposed for 6 days via gavage to DMBA. At the organismal level, which may be related to population level effects, the only observed effects from short term oral exposure to DMBA were reduced growth and impaired immune function in nestlings, for which there was a NOAEL of 2,000 ng_{DMBA}/g_{BW}-d, and a LOAEL of 20,000 ng_{DMBA}/g_{BW}-d. No effects were observed in adults exposed to doses as high as 60,000

ng_{DMBA}/g_{BW}-d. The 6-day exposure duration for studies on starlings is considered acute for adults, but may be considered chronic for developing nestlings. Hough et al. (1993) reported reproductive effects in two strains of pigeon exposed over a period of five months to BaP. The study by Hough et al. (1993) entailed weekly injections at one dose rate (10,000 ng_{BaP}/g_{BW} per week or 1,430 ng_{BaP}/ng_{BW}-d) for up to five months. Observations were focused on histopathological and physiological alterations, but it was also noted that there was 100 percent infertility in female birds. Results of the study by Hough et al. (1993) are the basis of screening levels in this assessment because it entailed chronic exposure, and produced the lowest of the potential LOAELs, which is desirable for assessing risks associated with mixtures, and ensuring that risks to birds exposed during particularly sensitive stages in development (e.g., nestlings) are addressed. The LOAEL for reproductive impairments in pigeons was adjusted by a factor of ten for an estimated NOAEL. The estimated NOAEL was further adjusted by a factor of ten for uncertainty about species differences in sensitivity. Assuming BaP toxicity represents HPAHs in general (EPA 2007b), the LOAEL and estimated NOAEL for BaP were selected as TRVs for assessing risks posed to waterbirds and seabirds by dietary exposure to HPAHs in general (Table D2).

Table D1. Screening levels used to evaluate contaminant levels in seabird eggs collected from San Diego Bay colonies in 2013.

NOAEC/LOAEC	($ng/g_{egg}fw$)	Contaminant (effects)	Source
		Mercury (egg hatchability, embryo mortality)	
NOAEC	300	estimated from LOAEC (egret)	Shore et al 2011
LOAEC(s)	600	5th percentile all species (<lowest LOAEC for all species)	Shore et al 2011
	800	Low (snowy egret; field based)	Shore et al 2011
	1,300	Mid (common loon; field based)	Shore et al 2011
	3,700	High (common tern; field based)	Shore et al 2011
		DDTs (productivity, eggshell thinning)	
NOAEC	200	eggshell thinning	Blus 2011
	1,000	reduced productivity	Blus 2011
LOAEC(s) – prod.	3,000	sensitive species (brown pelican)	Blus 2011
	5,000	Mid-range (e. g. double-crested cormorant & Caspian tern)	Blus 2011
LOAEC(s) - thinning	600	Sensitive species (pelican)	Blus 2011
	10,000	Mid-range (e. g. double-crested cormorant)	Blus 2011
		Total PCBs (productivity, parental behavior)	
NOAEC	100	Adjusted LOAEC for most sensitive species	Harris & Elliott 2011
	2,300	Adjusted LOAEC for waterbirds and raptors[1]	Harris & Elliott 2011
LOAEC(s)	1,000	Most sensitive species (chickens)	Harris & Elliott 2011
	6,000	Medium sensitivity species (perching birds)	Harris & Elliott 2011
	23,000	Low sensitivity species (terns, gulls raptors)	Harris & Elliott 2011
		PCB 126 (embryo lethality)	
NOAEC	0. 11	Adjusted LOAEC for most sensitive species (chickens)	
	6. 5	Adjusted LOAEC seabirds and raptors	Harris & Elliott 2011
LOAEC(s)	1. 1	Highly sensitive species (chickens)	Harris & Elliott 2011
	24. 0	Medium sensitivity species (bobwhite quail)	Harris & Elliott 2011
	65. 0	Least sensitive species (cormorant, tern, raptor)	Harris & Elliott 2011
		TEQ (embryo lethality)	
NOAEC	0. 018	Adjusted LOAEC for most sensitive species	Harris & Elliott 2011
	0. 4	Adjusted LOAEC for waterbirds	Harris & Elliott 2011
LOAEC(s)	0. 18	Highly sensitive species (chickens)	Harris & Elliott 2011
	1. 0	Medium sensitivity (pigeon, pheasant, quail)	Harris & Elliott 2011
	4. 0	Low sensitivity (cormorant, heron, wood duck)	Harris & Elliott 2011

PBDEs (egg hatchability, fertility of offspring)			
NOAEC	180	Bounded NOAEC sensitive species	McKernan et al 2009
LOAEC(s)	288	sensitive species	McKernan et al 2009
PFOS (offspring survival)			
NOAEC	1,000	Adjusted LOAEC for more sensitive of two species	Newsted et al 2005
LOAEC(s)	62,000	More sensitive of two species (mallard less sensitive)	Newsted et al. 2005
Not done - Chlordanes (no data for screening levels) and PAHs (not analyzed in egg samples)			

1. all but most sensitive (chickens) (including terns, gull and raptors)

Table D2 Toxicity Reference Values (TRVs) - dose rates used to evaluate risks posed by contaminants in the diet to aquatic-dependent birds of San Diego Bay

NOAEL/LOAEL	(ng/g_{BW}-d)	Contaminant (most sensitive effects)	Source
		Mercury (reproduction, parental behavior, productivity)	
NOAEL	4. 0	5th percentile NOAEL all species	Zhang et al. 2013
	7. 0	Sensitive species	USFWS 2003
	21	Less sensitive species (obligate piscivores – seabirds)	USFWS 2003
LOAEL	10	Most Sensitive species (ibis)	Zhang et al. 2013
	180	Mid-range (all species considered; based on mallard)	DTSC/HERD 2009
		Total DDTs (based on productivity, survival and growth)	
NOAEL	9. 0	most sensitive species (pelican)	DTSC/HERD 2009
	227	less sensitive species	EPA 2007a
LOAEL	27	most sensitive species	EPA 1995
	1,500	Mid-range all species	DTSC/HERD 2009
		Total PCBs (based on egg production, fertility, and hatchability)	
NOAEL	90	Estimated for most sensitive species	DTSC/HERD 2009
LOAEL	1,270	Mid-range, all species (mostly non-waterbirds)	DTSC/HERD 2009
		TEQ (based on egg production, hatchability)	
NOAEL	0. 0011	Estimated, for most sensitive species (chickens)	Su et al 2014
LOAEL	0. 0495	Lowest for species other than most sensitive	Su et al 2014
	0. 178	Mid-range for all species considered (incl. mallards)	Su et al 2014
		PBDEs (reproductive behavior, egg quality, productivity)	
NOAEL	9. 6	Adjusted LOAEL- sensitive species[1]	Fernie et al 2009
LOAEL	96	Sensitive species[1]	Fernie et al 2009
		Chlordanes (survival)	
NOAEL	160	sensitive species[2]	Stickel et al. 1983
LOAEL	7,000	Sensitive species[2]	Stickel et al. 1983
		LPAHs (weight and food consumption)	
NOAEL	295	Sensitive effect in species of unknown relative sensitivity	Klasing 2007
LOAEL	4,730	Sensitive effect in species of unknown relative sensitivity	Klasing 2007
		HPAHs (infertility)	

NOAEL	14.3	Sensitive effect in species of unknown relative sensitivity	Hough et al. 1993
LOAEL	1,430	Sensitive effect in species of unknown relative sensitivity	Hough et al. 1993

1. Only kestrels, mallards and terns studied (mallards and terns ≤ sensitivity of kestrels; Rattner et al. 2011)
2. Based on acute toxicity tests with multiple upland species and mallards (Eisler 1990)

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Appendix E. Risk Characterization - Initial Screen Tables

Table E1. Maximum contaminant concentrations detected (ng/g fw) in seabird eggs collected from San Diego Bay nesting colonies in 2013, as compared with most conservative (lowest) screening levels for adverse effects.

species →	California least tern	Caspian tern	Double-crested cormorant	Western gull	lowest SL
analyte ↓					
mercury	270	1,020	192	126	300
DDTs	193	2,766	3,644	1,003	200
PCBs	877	1,276	2,358	1,072	100
PCB 126*	with TEQ	with TEQ	with TEQ	with TEQ	0.011
TEQ	0.278	0.331	0.082	0.058	0.018
PBDE	102	414	280	304	180
PFOS	72	63			1,000
Chlordane	17.5	24.6	3.5	4.1	No SL
PAH	not analyzed	not analyzed	not analyzed	not analyzed	

Due to very low detection, risk associated with PCB 126 was not evaluated separately, but in combination with other dioxin-like congeners as part of the PCB TEQ.

Table E2a Summary data on concentrations of mercury in aquatic biota from San Diego Bay, with exceedances/no exceedances (+, ++, or -) of observed effect-based dietary screening levels (NOAECs; SLs) marked for avian receptor groups represented by surf scoter (1), California least tern (2), Caspian tern (3), double-crested cormorant (4) and western gull (5).

Mercury						benthic feeding waterfowl[1]	small pelagic piscivore[2]	large pelagic piscivore[3]	large piscivore pelagic/demersal/benthic[4]	generalist aquatic/non-aquatic[5]
	N	Mean	StdDev	Max	SLs→	13	5	9	13.5	25
Arrow goby	1	46	NA	46		++	++	++	++	++
Barred sand bass	9	67	9	87		++	++	++	++	++
Spotted sand bass	11	94	30	154		++	++	++	++	++
Deepbody anchovy	11	49	36	109		++	++	++	++	++
Northern anchovy	1	35	NA	35		++	++	++	++	++
Slough anchovy	9	29	9	46		++	++	++	++	++
Black perch	2	19	7	23		++	++	++	++	-
Shiner perch	6	27	12	46		++	++	++	++	++
California halibut	20	54	45	239		++	++	++	++	++
California killifish	2	26	0	26		++	++	++	++	++
Topsmelt	9	32	16	62		++	++	++	++	++
Round stingray	1	151	NA	151		++	++	++	++	++
Brown shrimp	2	35	8	41		++	++	++	++	++
Crabs	1	12	NA	12		-	++		-	-
Crustacea	10	33	10	55		++	++	++	++	++
Mollusks	10	40	22	88		++	++	++	++	++
Polychaetes	16	99	106	429		++	++	++	++	++
Plankton	9	25	21	72		++	++	++	++	+

- no exceedances

+ screening level exceeded by maximum

++ screening level exceeded by maximum and mean

Table E2b. Summary data on concentrations of total DDTs in aquatic biota from San Diego Bay, with exceedances/no exceedances (+, ++, or -) of observed effect-based dietary screening levels (NOAECs; SLs) marked for avian receptor groups represented by surf scoter (1), California least tern (2), Caspian tern (3), double-crested cormorant (4) and western gull (5).

DDTs		Concentrations (ng/g ww)				benthic feeding waterfowl[1]	small pelagic piscivore[2]	large pelagic piscivore[3]	large piscivore pelagic/demersal/benthic[4]	generalist aquatic/non-aquatic[5]
Sample type	N	Mean	StdDev	Max	SLs →	29	11	21	30	57
Arrow goby	1	0.050	NA	0.050		-	-	-	-	-
Goby sp.	3	10.6	0.29	10.9		-	-	-	-	-
Barred sand bass	9	16.1	10.4	40.3		+	++	+	+	-
Spotted sand bass	11	10.8	3.7	16.0		-	+	-	-	-
Deepbody anchovy	11	20.8	14.8	46.4		+	++	++	+	-
Northern anchovy	2	8.9	1.0	9.6		-	-	-	-	-
Slough anchovy	10	12.2	7.1	30.9		+	++	+	+	-
Black perch	2	13.3	9.7	20.1		-	++	-	-	-
Shiner perch	6	14.5	6.7	22.1		-	++	+	-	-
California halibut	20	11.7	6.9	37.7		+	++	+	+	-
California killifish	2	6.5	6.7	11.2		-	-	-	-	-
Topsmelt	9	6.8	3.0	11.7		-	+	-	-	-
Round stingray	1	0.6	NA	0.55		-	-	-	-	-
Brown shrimp	2	2.4	1.1	3.2		-	-	-	-	-
Crabs	1	5.1	NA	5.1		-	-	-	-	-
Crustacea	13	6.1	5.5	21.5		-	+	+	-	-
Mollusks	11	6.1	9.5	33.1		+	+	+	+	-
Polychaetes	18	7.1	4.6	21.5		-	+	+	-	-

- no exceedances

+ screening level exceeded by maximum

++ screening level exceeded by maximum and mean

Table E2c. Summary data on concentrations of PCBs (Σ Bight'13 congeners) in aquatic biota from San Diego Bay, with exceedances/no exceedances (+, ++, or -) of observed effect-based dietary screening levels (NOAECs; SLs) marked for avian receptor groups represented by surf scoter (1), California least tern (2), Caspian tern (3), double-crested cormorant (4) and western gull (5).

ΣPCB congeners*		Concentrations (ng/g ww)				benthic feeding waterfowl[1]	small pelagic piscivore[2]	large pelagic piscivore[3]	large piscivore pelagic/demersal/benthic[4]	generalist aquatic/non-aquatic[5]
Sample Type	N	Mean	StdDev	Max	SLs →	288	110	207	303	566
Arrow goby	1	98	NA	98		-	-	-	-	-
Goby sp.	3	216	143	327		+	++	++	+	-
Barred sand bass	9	191	92	369		+	++	+	+	-
Spotted sand bass	11	289	132	571		++	++	++	+	
Deepbody anchovy	11	263	137	458		+	++	++	+	-
Northern anchovy	2	220	16	231		+	++	++	-	-
Slough anchovy	10	248	42	322		+	++	++	+	-
Black perch	2	239	272	432		+	++	++	+	-
Shiner perch	6	175	145	388		+	++	+	+	-
California halibut	20	176	76	359		+	++	+	+	-
California killifish	2	68	42	98		-	-	-	-	-
Topsmelt	9	148	60	251		-	++	+	-	-
Round stingray	1	419	NA	419		++	++	++	++	-
Brown shrimp	2	28	29	48		-	-	-	-	-
Crabs	1	14	NA	14		-	-	-	-	-
Crustacea	13	91	54	169		-	+	-	-	-
Mollusks	11	44	51	169		-	+	-	-	-
Polychaetes	18	122	63	283		-	++	+	-	-
Plankton	13	59	47	169		-	+	-	-	-

- no exceedances

+ screening level exceeded by maximum

++ screening level exceeded by maximum and mean

* Total PCB concentrations will be higher by a factor of ~1.27

Table E2d. Summary data on concentrations of PCB TEQs in aquatic biota from San Diego Bay, with exceedances/no exceedances (+, ++, or -) of observed effect-based dietary screening levels (NOAECs; SLs) marked for avian receptor groups represented by surf scoter (1), California least tern (2), Caspian tern (3), double-crested cormorant (4) and western gull (5).

PCB TEQs		Concentrations (ng/g ww)				benthic feeding waterfowl[1]	small pelagic piscivore[2]	large pelagic piscivore[3]	large piscivore pelagic/demersal/benthic[4]	generalist aquatic/non-aquatic[5]
Sample type	N	Mean	StdDev	Max	SLs→	0.0035	0.0014	0.0025	0.004	0.0069
Arrow goby	1	0.0064	NA	0.0064		++	++	++	++	-
Goby sp.	3	0.0099	0.0037	0.0137		++	++	++	++	++
Barred sand bass	9	0.0294	0.0110	0.0528		++	++	++	++	++
Spotted sand bass	11	0.0231	0.0317	0.1169		++	++	++	++	++
Deepbody anchovy	11	0.0645	0.0327	0.1140		++	++	++	++	++
Northern anchovy	2	0.0065	0.0000	0.0065		++	++	++	++	-
Slough anchovy	10	0.0234	0.0346	0.1125		++	++	++	++	++
Black perch	2	0.0278	0.0295	0.0486		++	++	++	++	++
Shiner perch	6	0.0155	0.0113	0.0334		++	++	++	++	++
California halibut	20	0.0435	0.0395	0.1315		++	++	++	++	++
California killifish	2	0.0067	0.0004	0.0070		++	++	++	++	+
Topsmelt	9	0.0075	0.0007	0.0086		++	++	++	++	++
Round stingray	1	0.0111	NA	0.0111		++	++	++	++	++
Brown shrimp	2	0.0064	0.0001	0.0065		-	-	-	-	-
Crabs	1	0.0063	NA	0.0063		-	-	-	-	-
Crustacea	13	0.0075	0.0016	0.0115		+	+	+	+	++
Mollusks	11	0.0064	0.0001	0.0067		++	++	++	++	-
Polychaetes	18	0.0069	0.0009	0.0097		++	++	++	++	++
Plankton	13	0.0065	0.0004	0.0075		++	++	++	++	+

- no exceedances

+ screening level exceeded by maximum

++ screening level exceeded by maximum and mean

Table E2e. Summary data on concentrations of PBDEs in aquatic biota from San Diego Bay, with exceedances/no exceedances (+, ++, or -) of observed effect-based dietary screening levels (NOAECs; SLs) marked for avian receptor groups represented by surf scoter (1), California least tern (2), Caspian tern (3), double-crested cormorant (4) and western gull (5).

PBDEs		Concentrations (ng/g ww)				benthic feeding waterfowl[1]	small pelagic piscivore[2]	large pelagic piscivore[3]	large piscivore pelagic/demersal/benthic[4]	generalist aquatic/non-aquatic[5]
	N	Mean	StdDev	Max	SLs →	31	12	22	32	60
Arrow goby	1	20	NA	20		-	++	-	-	-
Goby sp.	3	3.1	1.7	5.0		-	-	-	-	-
Barred sand bass	9	3.6	2.4	8.2		-	-	-	-	-
Spotted sand bass	11	1.2	0.9	3.4		-	-	-	-	-
Deepbody anchovy	11	0.1	0.1	0.3		-	-	-	-	-
Northern anchovy	2	8.6	12	17		-	+	-	-	-
Slough anchovy	10	1.0	1.2	3.4		-	-	-	-	-
Black perch	2	5.0	3.1	7.2		-	-	-	-	-
Shiner perch	6	7.7	4.3	13		-	+	-	-	-
California halibut	20	1.8	1.1	4.2		-	-	-	-	-
California killifish	2	2.6	2.4	4.4		-	-	-	-	-
Topsmelt	9	7.4	5.7	17		-	+	-	-	-
Round stingray	1	10.1	NA	10		-	-	-	-	-
Brown shrimp	2	1.7	1.2	2.6		-	-	-	-	-
Crabs	1	3.7	NA	3.7		-	-	-	-	-
Crustacea	13	11	27	100		+	+	+	+	+
Mollusks	11	1.4	1.3	4.0		-	-	-	-	-
Polychaetes	18	6.0	5.2	22		-	+	-	-	-
Plankton	13	18	29	108		+	++	+	+	+

- no exceedances
+ screening level exceeded by maximum
++ screening level exceeded by maximum and mean

Table E2f. Summary data on concentrations of chlordanes in aquatic biota from San Diego Bay, with exceedances/no exceedances (+, ++, or -) of observed effect-based dietary screening levels (NOAECs; SLs) marked for avian receptor groups represented by surf scoter (1), California least tern (2), Caspian tern (3), double-crested cormorant (4) and western gull (5).

CHLORDANE		Concentrations (ng/g ww)				benthic feeding waterfowl[1]	small pelagic piscivore[2]	large pelagic piscivore[3]	large piscivore pelagic/demersal/benthic[4]	generalist aquatic/non-aquatic[5]
Sample type	N	Mean	StdDev	Max	SLs →	513	196	369	539	1,006
Arrow goby	1	0.05	NA	<0.05		-	-	-	-	-
Goby sp.	3	0.58	0.60	1.23		-	-	-	-	-
Barred sand bass	9	1.80	1.18	3.02		-	-	-	-	-
Spotted sand bass	11	2.76	1.13	4.85		-	-	-	-	-
Deepbody anchovy	11	2.84	2.50	6.65		-	-	-	-	-
Northern anchovy	2	0.05	0.00	ND		-	-	-	-	-
Slough anchovy	10	1.25	1.07	3.58		-	-	-	-	-
Black perch	2	3.20	3.71	5.83		-	-	-	-	-
Shiner perch	6	2.65	2.92	7.76		-	-	-	-	-
California halibut	20	1.11	0.77	3.08		-	-	-	-	-
California killifish	2	0.05	0.00	ND		-	-	-	-	-
Topsmelt	9	1.25	0.98	3.43		-	-	-	-	-
Round stingray	1	10.21	NA	10.2		-	-	-	-	-
Brown shrimp	2	0.14	0.12	0.22		-	-	-	-	-
Crabs	1	0.05	NA	0.05		-	-	-	-	-
Crustacea	13	0.29	0.72	2.66		-	-	-	-	-
Mollusks	11	0.44	0.68	2.15		-	-	-	-	-
Polychaetes	18	4.40	14.3	61.6		-	-	-	-	-

- no exceedances
+ screening level exceeded by maximum
++ screening level exceeded by maximum and mean

Table E2g. Summary data on concentrations of PAHs in aquatic biota from San Diego Bay, with exceedances/no exceedances (+, ++, or -) of observed effect-based dietary screening levels (NOAECs; SLs) marked for avian receptor groups represented by surf scoter (1), California least tern (2), Caspian tern (3), double-crested cormorant (4) and western gull (5).

Σ HPAH		Concentrations (ng/g ww)				benthic feeding waterfowl[1]	small pelagic piscivore[2]	large pelagic piscivore[3]	large piscivore pelagic/demersal/benthic[4]	generalist aquatic/non-aquatic[5]
Sample Type	N	Mean	StdDev	Max	SLs →	46	18	33	48	90
Goby sp.	3	1.8	1.3	3.3		-	-	-	-	-
Barred sand bass	9	2.7	1.4	4.9		-	-	-	-	-
Spotted sand bass	11	3.2	1.2	5.5		-	-	-	-	-
Slough anchovy	1	11.0	NA	11.0		-	-	-	-	-
Black perch	2	7.9	9.0	14.3		-		-	-	-
Shiner perch	6	6.5	2.2	9.5		-	-	-	-	-
California halibut	20	1.7	1.1	5.0		-	-	-	-	-
Round stingray	1	3.9	NA	3.9		-	-	-	-	-
Brown shrimp	2	8.0	1.7	9.2		-	-	-	-	-
Crabs	1	24.9	NA	24.9		-	++	-	-	-
Crustacea	13	44	69	226		+	++	++	+	+
Mollusks	11	55	51	173		++	++	++	++	+
Polychaetes	18	760	2,913	12,430		++	++	++	++	++

Table E2g (continued).

Σ LPAH										
Sample Type					SLs →	946	362	680	993	1,855
Goby sp.	3	66	20	89		-	-	-	-	-
Barred sand bass	9	6	3	13		-	-	-	-	-
Spotted sand bass	11	11	3	16		-	-	-	-	-
Slough anchovy	1	26	NA	26		-	-	-	-	-
Black perch	2	18	6	22		-	-	-	-	-
Shiner perch	6	16	2	18		-	-	-	-	-
California halibut	20	9	6	20		-	-	-	-	-
Round stingray	1	26	NA	26		-	-	-	-	-
Brown shrimp	2	25	8	31		-	-	-	-	-
Crabs	1	50	NA	50		-	-	-	-	-
Crustacea	13	64	62	231		-	-	-	-	-
Mollusks	11	45	47	145		-	-	-	-	-
Polychaetes	18	78	83	372		-	+	-	-	-

- no exceedances
+ screening level exceeded by maximum
++ screening level exceeded by maximum and mean

Appendix F. Exposure Point Concentrations (EPCs), daily dose estimates, and HQs, assuming foraging is throughout San Diego Bay

Table F1. Estimated exposure point concentrations (EPCs; ng/g ww), daily dose rates and hazard quotients (HQs) for dietary mercury (Hg) exposure by representative avian species, assuming they forage throughout San Diego Bay.

Representative species	Taxa in samples	Analyte	Region	Sample N	EPC - Ave	EPC - Max		FIR (FI/BW) (gfood/gBW-d)	AUF	Daily dose - Ave (ng_{Hg}/g_{BW}-d)	Daily dose - Max (ng_{Hg}/kg_{BW}-d)		HQ - NOAEL (lowest)	HQ - NOAEL (sensitive)	HQ - NOAEL (less sensitive)	HQ - LOAEL (most sensitive)	HQ - LOAEL (mid-range)
												TRVs →	(4)	(7)	(21)	(10)	(180)
CLT	gobies (Arrow), perch (black & shiner), anchovy (slough & northern), topsmelt, CA killifish	Mercury	Baywide	30	29	62		0.818	1	23.98	51	HQ - mean	6.0	3.4	1.1	2.4	0.13
												HQ - max	13	7.2	2.4	5.1	0.28
CT	sand bass (barred & spotted), perch (black & shiner), anchovy (deepbody, northern & slough), topsmelt	Mercury	Baywide	58	51	154		0.434	0.5	11.03	33	HQ - mean	2.8	1.6	0.53	1.1	0.06
												HQ - max	8.3	4.8	1.6	3.3	0.18
DCC	sand bass (barred & spotted), perch (black & shiner), anchovy (deepbody, northern & slough), topsmelt, CA halibut	Mercury	Baywide	78	52	239		0.297	1	15.36	71	HQ - mean	3.8	2.2	0.73	1.5	0.09
												HQ - max	18	10	3.4	7.1	0.40
Scoter	Crabs, misc crustacea, mollusks, polychaetes	Mercury	Baywide	37	63	429		0.312	1	21.57	148	HQ - mean	5.4	3.1	1.0	2.2	0.12
												HQ - max	37	21	7.0	15	0.82
	Sediment									1.9	14						
Gull	Anchovy (deepbody, northern & slough), topsmelt	Mercury	Baywide	30	37	109		0.412	0.9	13.79	40.39	HQ - mean	3.5	2.0	0.66	1.4	0.08
												HQ - max	10	5.6	1.9	4.0	0.22

Table F2. Estimated exposure point concentrations (EPCs; ng/g ww), daily dose rates and hazard quotients (HQs) for dietary DDT exposure by representative avian species, assuming they forage throughout San Diego Bay.

Representative species	Taxa in samples	Analyte	Region	Samples N	EPC - Ave	EPC - Max		FIR (FI/BW) (g_{food}/g_{BW}-d)	AUF	daily dose - Ave (ng_{DDT}/g_{BW}-d)	daily dose - Max (ng_{DDT}/kg_{BW}-d)		HQ - NOAEL (lowest)	HQ - NOAEL (less sensitive)	HQ - LOAEL (most sensitive)	HQ - LOAEL (mid-range)
												TRVs →	(9)	(227)	(27)	(1500)
CLT	gobies (Arrow & spp.), perch (black & shiner), anchovy (slough & northern), topsmelt, killifish	DDT	Baywide	35	10.3	30.9		0.818	1	8.41	25.2	HQ - mean	0.93	0.04	0.31	0.006
												HQ - max	2.81	0.11	0.94	0.017
CT	sand bass (barred & spotted), perch (black & shiner), anchovy (deepbody, northern & slough), topsmelt	DDT	Baywide	60	13.5	46.4		0.434	0.5	2.92	10.1	HQ - mean	0.32	0.01	0.11	0.002
												HQ - max	1.12	0.04	0.37	0.007
DCC	sand bass (barred & spotted), perch (black & shiner), anchovy (deepbody, northern & slough), topsmelt, halibut	DDT	Baywide	80	13.0	46.4		0.297	1	3.87	13.8	HQ - mean	0.43	0.02	0.14	0.003
												HQ - max	1.53	0.06	0.51	0.009
Scoter	Crabs, crustacea, brown shrimp, mollusks, polychaetes	DDT	Baywide	43	6.50	33.1		0.312	1	2.03	10.3	HQ - mean	0.23	0.01	0.08	0.001
												HQ - max	1.15	0.05	0.38	0.007
	sediment									0.001	0.017					
WEGU	Anchovy (deepbody, northern & slough), topsmelt	DDT	Baywide	32	13.4	46.4		0.412	0.9	4.98	17.2	HQ - mean	0.55	0.02	0.18	0.003
												HQ - max	1.91	0.08	0.64	0.011

Table F3. Estimated exposure point concentrations (EPCs; ng/g ww), daily dose rates, and hazard quotients (HQs) for dietary PCB exposure by representative avian species, assuming they forage throughout San Diego Bay. PCB concentrations = Σ Bight '13 congeners, and total PCB concentrations may be greater by a factor of ~1.27.

Representative species	Taxa in samples	Analyte	Region	Samples N	EPC - Ave	EPC - Max		FIR (FI/BW) (g_{food}/g_{BW}-d)	AUF	daily dose - Ave (ng_{PCB}/g_{BW}-d)	daily dose - Max (ng_{PCB}/kg_{BW}-d)		HQ - NOAEL (lowest)	HQ - LOAEL (mid-range)
												TRVs →	(90)	(1270)
CLT	gobies (Arrow & spp.), perch (black & shiner), anchovy (slough & northern), topsmelt, killifish	PCB	Baywide	35	190	432		0.818	1	156	353	HQ - mean	1.73	0.12
												HQ - max	3.92	0.28
CT	sand bass (barred & spotted), perch (black & shiner), anchovy (deepbody, northern & slough), topsmelt	PCB	Baywide	60	226	571		0.434	0.5	49.0	124	HQ - mean	0.54	0.04
												HQ - max	1.37	0.10
DCC	sand bass (barred & spotted), perch (black & shiner), anchovy (deepbody, northern & slough), topsmelt, halibut	PCB	Baywide	80	214	571		0.297	1	63.6	170	HQ - mean	0.71	0.05
												HQ - max	1.89	0.13
Surf scoter	Crabs, brown shrimp, crustacea, mollusks, polychaetes	PCB	Baywide	43	90.3	283		0.312	1	28.3	89.0	HQ - mean	0.31	0.02
												HQ - max	0.99	0.07
	sediment									0.084	0.660			
WEGU	Anchovy (deepbody, northern & slough), topsmelt	PCB	Baywide	32	223	458		0.412	0.9	82.6	170	HQ - mean	0.92	0.07
												HQ - max	1.89	0.13

Table F4. Estimated exposure point concentrations (EPCs; ng/g ww), daily dose rates, and hazard quotients (HQs) for dietary PCB-TEQ exposure by representative avian species, assuming they forage throughout San Diego Bay.

Representative species	Taxa in samples	Analyte	Region	Samples N	EPC - Ave	EPC - Max		FIR (FI/BW) (g_{food}/g_{BW}-d)	AUF	Daily dose - Ave (ng_{TEQ}/g_{BW}-d)	Daily dose - Max (ng_{TEQ}/kg_{BW}-d)		HQ - NOAEL (lowest)	HQ - LOAEL (most sensitive)	HQ - LOAEL (mid-range)
												TRVs →	(0.0011)	(0.0495)	(0.178)
CLT	gobies (Arrow & spp.), perch (black & shiner), anchovy (slough & northern), topsmelt, killifish	PCBTEQ	Baywide	35	0.01	0.113		0.818	1	0.0120	0.0920	HQ - mean	11	0.24	0.07
												HQ - max	84	1.86	0.52
CT	sand bass (barred & spotted), perch (black & shiner), anchovy (deepbody, northern & slough), topsmelt	PCBTEQ	Baywide	60	0.03	0.117		0.434	0.5	0.0061	0.0253	HQ - mean	6	0.12	0.03
												HQ - max	23	0.51	0.14
DCC	sand bass (barred & spotted), perch (black & shiner), anchovy (deepbody, northern & slough), topsmelt, halibut	PCBTEQ	Baywide	80	0.03	0.132		0.297	1	0.0095	0.0391	HQ - mean	9	0.19	0.05
												HQ - max	36	0.79	0.22
Scoter	Crabs, crustacea, brown shrimp, mollusks, polychaetes	PCBTEQ	Baywide	43	0.01	0.012		0.312	1	0.0022	0.0036	HQ - mean	2	0.04	0.01
	sediment (not calculated)											HQ - max	3	0.07	0.02
WEGU	Anchovy (deepbody, northern & slough), topsmelt	PCBTEQ	Baywide	32	0.03	0.114		0.412	0.9	0.0119	0.0422	HQ - mean	11	0.24	0.07
												HQ - max	38	0.85	0.24

Table F5. Estimated exposure point concentrations (EPCs; ng/g ww), daily dose rates, and hazard quotients (HQs) for dietary PBDE exposure by representative avian species, assuming they forage throughout San Diego Bay.

Representative species	Taxa in samples	Analyte	Region	Samples N	EPC - Ave	EPC - Max		FIR (FI/BW) (g_{food}/g_{BW}-d)	AUF	Daily dose - Ave (ng_{PBDE}/g_{BW}-d)	Daily dose - Max (ng_{PBDE}/kg_{BW}-d)		HQ - NOAEL (lowest)	HQ - LOAEL (most sensitive)
												TRVs →	9.6	96
CLT	gobies (Arrow & spp.), perch (black & shiner), anchovy (slough & northern), topsmelt, killifish	PBDE	Baywide	35	5.27	19.5		0.818	1	4.30	16.0	HQ - mean	0.45	0.045
												HQ - max	1.66	0.166
CT	sand bass (barred & spotted), perch (black & shiner), anchovy (deepbody, northern & slough), topsmelt	PBDE	Baywide	60	3.27	17.2		0.434	0.5	0.710	3.72	HQ - mean	0.074	0.007
												HQ - max	0.388	0.039
DCC	sand bass (barred & spotted), perch (black & shiner), anchovy (deepbody, northern & slough), topsmelt, halibut	PBDE	Baywide	80	2.90	17.2		0.297	1	0.862	5.10	HQ - mean	0.090	0.009
												HQ - max	0.532	0.053
Scoter	Crabs, crustacea, brown sshrimp, mollusks, polychaetes	PBDE	Baywide	43	6.33	99.5		0.312	1	2.00	31.2	HQ - mean	0.208	0.021
												HQ - max	3.255	0.326
	sediment									0.018	0.198			
WEGU	Anchovy (deepbody, northern & slough), topsmelt	PBDE	Baywide	32	2.98	17.2		0.412	0.9	1.10	6.36	HQ - mean	0.115	0.012
												HQ - max	0.662	0.066

Table F6. Estimated exposure point concentrations (EPCs; ng/g ww), daily dose rates and hazard quotients (HQs) for dietary HPAH (PAH-HMW) exposure by representative avian species, assuming they forage throughout San Diego Bay.

Representative species	Taxa in samples	Analyte	Region	Samples N	EPC - Ave	EPC - Max		FIR (FI/BW) (g_{food}/g_{BW}-d)	AUF	Daily dose (ng_{HPAH}/g_{BW}-d)	Max daily dose (ng_{HPAH}/kg_{BW}-d)		HQ - NOAEL (sensitive)	HQ - LOAEL (most sensitive)
												TRVs →	(14.3)	(1430)
CLT	Black perch, shiner perch, slough anchovy, goby sp.	HPAH	Baywide	12	5.96	14.3		0.818	1	4.87	11.7	HQ - mean	0.34	0.003
												HQ - max	0.82	0.008
CT	sand bass (barred & spotted), perch (shiner & black), slough anchoy	HPAH	Baywide	29	4.33	14.3		0.434	0.5	0.939	3.11	HQ - mean	0.07	0.001
												HQ - max	0.22	0.002
DCC	sand bass (barred & spotted), perch (shiner & black), slough anchoy, halibut	HPAH	Baywide	58	2.77	14.3		0.297	1	0.823	4.26	HQ - mean	0.06	0.001
												HQ - max	0.30	0.003
Surf scoter	Crabs, crustacea, mollusks, polychaetes	HPAH	Baywide	43	346	12,430		0.312	1	111	3,899	HQ - mean	7.74	0.077
												HQ - max	273	2.73
	sediment									2.63	18.4			
WEGU	Slough anchovy	HPAH	Baywide	1	11.0			0.412	0.9	4.09	NA	one sample	0.29	0.003

Appendix G. Exposure Point Concentrations (EPCs), daily dose estimates, and HQs, assuming foraging is primarily in the Northern, Central and/or Southern Bay region

Table G1. Estimated exposure point concentrations (EPCs; ng/g ww), daily dose rates and hazard quotients for dietary mercury exposure by representative avian species that forage in San Diego Bay, assuming foraging is primarily in the northern, central or southern region of the bay.

Representative species	Taxa in samples	Analyte	Region	Sample N	EPC - Ave	EPC - Max		FIR (FI/BW) (g_{food}/g_{BW}-d)	AUF	Daily dose - Ave (ng_{Hg}/g_{BW}-d)	Daily dos- Max (ng_{Hg}/kg_{BW}-d)		HQ - NOAEL (lowest)	HQ - NOAEL (sensitive)	HQ - NOAEL (less sensitive)	HQ - LOAEL (most sensitive)	HQ - LOAEL (mid-range)
												TRVs →	(4.0)	(7.0)	(21)	(10)	(180)
CLT	perch (black & shiner), topsmelt	Mercury	North	8	23	36		0.818	1	18.4	29.4	HQ - mean	5	2.6	0.9	1.8	0.1
												HQ - max	7	4.2	1.4	2.9	0.2
CT	sand bass (barred & spotted), perch (black & shiner), topsmelt	Mercury	North	15	44	103		0.434	0.5	9.5	22.3	HQ - mean	2.4	1.4	0.5	1.0	0.1
												HQ - max	5.6	3.2	1.1	2.2	0.1
DCC	sand bass (barred & spotted), perch (black & shiner), halibut, topsmelt	Mercury	North	22	43	103		0.297	1	12.8	30.6	HQ - mean	3	1.8	0.6	1.3	0.1
												HQ - max	8	4.4	1.5	3.1	0.2
Scoter	Crabs, crustacea, mollusks, polychaetes	Mercury	North	13	46	135		0.312	1	17.4	55.7	HQ - mean	4	2.5	0.8	1.7	0.1
												HQ - max	14	8.0	2.7	5.6	0.3
	sediment									3.1	13.5						
WEGU	Topsmelt	Mercury	North	NA	19	22.00		0.412	0.9	7.0	8.2	HQ - mean	1.8	1.0	0.3	0.7	0.0
												HQ - max	2.0	1.2	0.4	0.8	0.0
CLT	Shiner perch, slough anchovy, topsmelt	Mercury	Central	12	34	62		0.818	1	28.1	50.7	HQ - mean	7	4.0	1.3	2.8	0.2
												HQ - max	13	7.2	2.4	5.1	0.3
CT	sand bass (barred & spotted), anchovy	Mercury	Central	25	59	124		0.434	0.5	12.8	27.0	HQ - mean	3.2	1.8	0.6	1.3	0.1

Representative species	Taxa in samples	Analyte	Region	Sample N	EPC - Ave	EPC - Max		FIR (FI/BW) (g_{food}/g_{BW}-d)	AUF	Daily dose - Ave (ng_{Hg}/g_{BW}-d)	Daily dos- Max (ng_{Hg}/kg_{BW}-d)		HQ - NOAEL (lowest)	HQ - NOAEL (sensitive)	HQ - NOAEL (less sensitive)	HQ - LOAEL (most sensitive)	HQ - LOAEL (mid-range)
	(deepbody & slough), shiner perch, topsmelt																
												HQ - max	6.7	3.9	1.3	2.7	0.1
DCC	sand bass (barred & spotted), anchovy (deepbody & slough), shiner perch, topsmelt, halibut	Mercury	Central	32	62	239		0.297	1	18.6	71.1	HQ - mean	4.6	2.7	0.9	1.9	0.1
												HQ - max	17.8	10.2	3.4	7.1	0.4
Scoter	Crustacea, mollusks, polychaetes	Mercury	Central	14	93	429		0.312	1	30.8	138.6	HQ - mean	7.7	4.4	1.5	3.1	0.2
												HQ - max	34.6	19.8	6.6	13.9	0.8
	sediment									1.8	4.5						
WEGU	anchovy (deepbody & slough), topsmelt	Mercury	Central	16	48	109.00		0.412	0.9	17.8	40.4	HQ - mean	4.4	2.5	0.8	1.8	0.1
												HQ - max	10.1	5.8	1.9	4.0	0.2
CLT	Arrow goby, killifish, anchovy (northern & slough), topsmelt, killifish	Mercury	South	10	29	46		0.818	1	23.5	37.6	HQ - mean	6	3.4	1.1	2.3	0.1
												HQ - max	9	5.4	1.8	3.8	0.2
CT	sand bass (barred & spotted), anchovy (deepbody, northern & slough), shiner perch, topsmelt	Mercury	South	18	45	154		0.434	0.5	9.8	33.3	HQ - mean	2.4	1.4	0.5	1.0	0.1
												HQ - max	8.3	4.8	1.6	3.3	0.2
DCC	sand bass (barred & spotted), anchovy (deepbody, slough, northern), shiner perch, topsmelt, halibut	Mercury	South	24	45	154		0.297	1	13.5	45.7	HQ - mean	3	1.9	0.6	1.3	0.1
												HQ - max	11	6.5	2.2	4.6	0.3

Representative species	Taxa in samples	Analyte	Region	Sample N	EPC - Ave	EPC - Max		FIR (FI/BW) (g_{food}/g_{BW}-d)	AUF	Daily dose - Ave (ng_{Hg}/g_{BW}-d)	Daily dos- Max (ng_{Hg}/kg_{BW}-d)		HQ - NOAEL (lowest)	HQ - NOAEL (sensitive)	HQ - NOAEL (less sensitive)	HQ - LOAEL (most sensitive)	HQ - LOAEL (mid-range)
Scoter	Crustacea, mollusks, polychaetes	Mercury	South	10	44	165		0.312	1	21.5	55.8	HQ - mean	5	3.1	1.0	2.2	0.1
												HQ - max	14	8.0	2.7	5.6	0.3
Scoter	sediment									7.9	4.4						
WEGU	anchovy (deepbody, northern & slough), topsmelt	Mercury	South	11	27	42.00		0.412	0.9	9.8	15.6	HQ - mean	2.46	1.4	0.5	1.0	0.1
												HQ - max	3.89	2.2	0.7	1.6	0.1

Table G2. Estimated exposure point concentrations (EPCs; ng/g ww), daily dose rates and hazard quotients (HQs) for dietary DDT exposure by representative avian species that forage in San Diego Bay, assuming foraging is primarily in the northern, central or southern region of the bay.

Representative species	Taxa in samples	Analyte	Region	Sample N	EPC - Ave	EPC - Max		FIR (FI/BW) (gf$_{ood}$/g$_{BW}$-d)	AUF	ave. daily dose (ng$_{DDT}$/g$_{BW}$-d)	Max daily dose (ng$_{DDT}$/kg$_{BW}$-d)		HQ - NOAEL (lowest)	HQ - NOAEL (less sensitive)	HQ - LOAEL (most sensitive)	HQ - LOAEL (mid-range)
												TRVs →	(9)	(227)	(27)	(1500)
CLT	Black perch, goby sp., shiner perch, topsmelt	DDT	North	9	12.3	22.1		0.818	1	10.1	18.0	HQ mean	1.12	0.04	0.37	0.01
												HQ max	2.00	0.08	0.67	0.01
CT	sand bass (barred & spotted), perch (black & shiner), topsmelt	DDT	North	15	14.9	40.3		0.434	0.5	3.2	8.7	HQ mean	0.36	0.01	0.12	0.003
												HQ max	0.97	0.04	0.32	0.01
DCC	sand bass (barred & spotted), perch (black & shiner), halibut topsmelt	DDT	North	22	14.6	40.3		0.297	1	4.3	12.0	HQ mean	0.48	0.02	0.16	0.002
												HQ max	1.33	0.05	0.44	0.01
Scoter	Crabs, brown shrimp, misc. crustacea, mollusks, polychaetes	DDT	North	16	9.5	33.1		0.312	1	3.0	10.3	HQ mean	0.33	0.01	0.11	0.002
												HQ max	1.15	0.05	0.38	0.01
	sediment									0.001	0.003					
WEGU	Topsmelt	DDT	North	3	8.9	11.7		0.412	0.9	3.3	4.3	HQ mean	0.37	0.01	0.12	0.003
												HQ max	0.48	0.12	0.16	0.003
CLT	Goby sp., shiner perch,slough anchovy, topsmelt	DDT	Central	14	10.3	16.3		0.818	1	8.4	13.3	HQ mean	0.93	0.04	0.31	0.01
												HQ max	1.48	0.06	0.49	0.01
CT	sand bass (barred & spotted), anchovy (deepbody & slough), topsmelt, shiner perch	DDT	Central	26	14.7	46.4		0.434	0.5	3.2	10.1	HQ mean	0.36	0.01	0.12	0.002
												HQ max	1.12	0.04	0.37	0.01
DCC	sand bass (barred & spotted), anchovy (deepbody & slough), halibut, topsmelt, shiner perch	DDT	Central	33	13.6	46.4		0.297	1	4.0	13.8	HQ mean	0.45	0.02	0.15	0.003
												HQ max	1.53	0.06	0.51	0.01

Representative species	Taxa in samples	Analyte	Region	Sample N	EPC - Ave	EPC - Max		FIR (FI/BW) (gf$_{ood}$/g$_{BW}$-d)	AUF	ave. daily dose (ng$_{DDT}$/g$_{BW}$-d)	Max daily dose (ng$_{DDT}$/kg$_{BW}$-d)		HQ - NOAEL (lowest)	HQ - NOAEL (less sensitive)	HQ - LOAEL (most sensitive)	HQ - LOAEL (mid-range)
Scoter	misc. crustacea, mollusks, polychaetes	DDT	Central	14	0.6	11.5		0.312	1	0.2	3.6	HQ mean	0.02	0.00	0.01	0.000
												HQ max	0.40	0.02	0.13	0.002
	sediment									0.000	0.003					
WEGU	Anchovy (deepbody & slough), topsmelt	DDT	Central	17	16.8	46.4		0.412	0.9	6.2	17.2	HQ mean	0.69	0.03	0.23	0.004
												HQ max	1.91	0.08	0.64	0.01
CLT	Arrow goby, goby sp., CA killifish, anchovy (northern & slough), shiner perch, topsmelt	DDT	South	12	8.8	30.9		0.818	1	7.2	25.2	HQ mean	0.80	0.03	0.27	0.01
												HQ max	2.81	0.11	0.94	0.02
CT	sand bass (barred & spotted), anchovy (deepbody, northern, slough), shiner perch, topsmelt	DDT	South	19	10.6	30.9		0.434	0.5	2.3	6.7	HQ mean	0.25	0.01	0.08	0.002
												HQ max	0.74	0.03	0.25	0.004
DCC	sand bass (barred & spotted), anchovy (deepbody, northern, slough), halibut, shiner perch, topsmelt	DDT	South	25	10.9	30.9		0.297	1	3.2	9.2	HQ mean	0.36	0.01	0.12	0.002
												HQ max	1.02	0.04	0.34	0.01
Scoter	crustacea, mollusks, polychaetes	DDT	South	14	5.0	21.5		0.312	1	1.6	6.7	HQ mean	0.17	0.01	0.06	0.002
												HQ max	0.75	0.03	0.25	0.01
	sediment									0.001	0.013					
WEGU	anchovy (deepbody, northern, slough), topsmelt	DDT	South	12	9.8	30.9		0.412	0.9	3.6	11.4	HQ mean	0.40	0.02	0.13	0.00
												HQ max	1.27	0.05	0.42	0.01

Table G3. Estimated exposure point concentrations (EPCs; ng/g ww), daily dose rates and hazard quotients for dietary PCB exposure by representative avian species that forage in San Diego Bay, assuming foraging is primarily in the northern, central or southern region of the bay. PCB concentrations in food web samples = ΣBight ’13 congeners, and total PCB concentrations may be greater by a factor of ~1.27.

Representative species	Taxa in samples		Region	Sample N	EPC - Ave	EPC - Max		FIR (FI/BW) (g_{food}/g_{BW}-d)	AUF	ave. daily dose (ng_{PCB}/g_{BW}-d)	Max daily dose (ng_{PCB}/kg_{BW}-d)		HQ - NOAEL (lowest)	HQ - LOAEL (mid-range)
												TRVs →	(90)	(1270)
CLT	Black perch, goby sp., shiner perch, topsmelt	PCBs	North	9	192	432		0.818	1	157	353	HQ mean	1.75	0.12
												HQ max	3.92	0.28
CT	sand bass (barred & spotted), perch (black & shiner), topsmelt	PCBs	North	15	217	432		0.434	0.5	47	94	HQ mean	0.52	0.04
												HQ max	1.04	0.07
DCC	sand bass (barred & spotted), perch (black & shiner), halibut topsmelt	PCBs	North	22	200	432		0.297	1	59	128	HQ mean	0.66	0.05
												HQ max	1.43	0.10
Scoter	Crabs, brown shrimp, misc. crustacea, mollusks, polychaetes	PCBs	North	16	91	169		0.312	1	29	53	HQ mean	0.32	0.02
												HQ max	0.59	0.04
	sediment									0.15	0.66			
WEGU	Topsmelt	PCBs	North	3	146	210		0.412	0.9	54	78	HQ mean	0.60	0.04
												HQ max	.0.86	0.06
CLT	Goby sp., shiner perch,slough anchovy, topsmelt	PCBs	Central	14	244	327		0.818	1	200	267	HQ mean	2.22	0.16
												HQ max	2.97	0.21
CT	sand bass (barred & spotted), anchovy (deepbody & slough), topsmelt, shiner perch	PCBs	Central	26	282	571		0.434	0.5	61	124	HQ mean	0.68	0.05
												HQ max	1.37	0.10
DCC	sand bass (barred & spotted), anchovy (deepbody & slough), halibut, topsmelt, shiner perch	PCBs	Central	33	268	571		0.297	1	80	170	HQ mean	0.89	0.06
												HQ max	1.89	0.13
Scoter	crustacea, mollusks, polychaetes	PCBs	Central	14	13	283		0.312	1	4	89	HQ mean	0.05	0.00
												HQ max	0.98	0.07
	Sediment									0.05	0.18			

Representative species	Taxa in samples		Region	Sample N	EPC - Ave	EPC - Max		FIR (FI/BW) (g_{food}/g_{BW}-d)	AUF	ave. daily dose (ng_{PCB}/g_{BW}-d)	Max daily dose (ng_{PCB}/kg_{BW}-d)		HQ - NOAEL (lowest)	HQ - LOAEL (mid-range)
WEGU	Anchovy (deepbody & slough), topsmelt	PCBs	Central	17	283	458		0.412	0.9	105	170	HQ mean	1.17	0.08
												HQ max	1.89	0.13
CLT	Arrow goby, goby sp., CA killifish, anchovy (northern & slough), shiner perch, topsmelt	PCBs	South	12	126	231		0.818	1	103	189	HQ mean	1.14	0.08
												HQ max	2.10	0.15
CT	sand bass (barred & spotted), anchovy (deepbody, northern, slough), shiner perch, topsmelt	PCBs	South	19	157	295		0.434	0.5	34	64	HQ mean	0.38	0.03
												HQ max	0.71	0.05
DCC	sand bass (barred & spotted), anchovy (deepbody, northern, slough), halibut, shiner perch, topsmelt	PCBs	South	25	154	295		0.297	1	46	88	HQ mean	0.51	0.04
												HQ max	0.97	0.07
Scoter	crustacea, mollusks, polychaetes	PCBs	South	14	54	146		0.312	1	17	46	HQ mean	0.19	0.01
												HQ max	0.51	0.04
	sediment									0.019	0.098			
WEGU	anchovy (deepbody, northern, slough), topsmelt	PCBs	South	12	157	231		0.412	0.9	58	85	HQ mean	0.65	0.05
												HQ max	0.95	0.07

Table G4. Estimated exposure point concentrations (EPCs; ng/g ww), daily dose rates and hazard quotients for dietary PCB-TEQ exposure by representative avian species that forage in San Diego Bay, assuming foraging is primarily in the northern, central or southern region of the bay.

Representative species	Taxa in samples		Region	Sample N	EPC - Ave	EPC - Max		FIR (FI/BW) (g_{food}/g_{BW}-d)	AUF	ave. daily dose (ng_{TEQ}/g_{BW}-d)	Max daily dose (ng_{TEQ}/kg_{BW}-d)		HQ - NOAEL (lowest)	HQ - LOAEL (most sensitive)	HQ - LOAEL (mid-range)
												TRVs →	(0.0011)	(0.0495)	(0.178)
CLT	Black perch, goby sp., shiner perch, topsmelt	PCBTEQ	North	9	0.0071	0.0278		0.818	1	0.006	0.023	HQ mean	5.3	0.12	0.03
												HQ max	20.6	0.46	0.13
CT	sand bass (barred & spotted), perch (black & shiner), topsmelt	PCBTEQ	North	15	0.0110	0.0278		0.434	0.5	0.002	0.006	HQ mean	2.2	0.05	0.01
												HQ max	5.5	0.12	0.03
DCC	sand bass (barred & spotted), perch (black & shiner), halibut, topsmelt	PCBTEQ	North	22	0.0095	0.0690		0.297	1	0.0028	0.0205	HQ mean	2.6	0.06	0.02
												HQ max	18.7	0.41	0.12
Scoter	Crabs, brown shrimp, misc. crustacea, mollusks, polychaetes	PCBTEQ	North	16	0.0066	0.0085		0.312	1	0.0020	0.0027	HQ mean	1.9	0.04	0.01
												HQ max	2.4	0.05	0.01
WEGU	Topsmelt	PCBTEQ	North	3	0.0072	0.0081		0.412	0.9	0.0027	0.0030	HQ mean	2.4	0.05	0.02
												HQ max	2.7	0.06	0.02
CLT	Goby sp., shiner perch,slough anchovy, topsmelt	PCBTEQ	Central	14	0.0076	0.0204		0.818	1	0.0062	0.0167	HQ mean	5.6	0.13	0.03
												HQ max	15.2	0.34	0.09
CT	sand bass (barred & spotted), anchovy (deepbody & slough), topsmelt, shiner perch	PCBTEQ	Central	26	0.0199	0.0902		0.434	0.5	0.0043	0.0195	HQ mean	3.9	0.09	0.02
												HQ max	17.8	0.39	0.11

Representative species	Taxa in samples		Region	Sample N	EPC - Ave	EPC - Max		FIR (FI/BW) (g_{food}/g_{BW}-d)	AUF	ave. daily dose (ng_{TEQ}/g_{BW}-d)	Max daily dose (ng_{TEQ}/kg_{BW}-d)		HQ - NOAEL (lowest)	HQ - LOAEL (most sensitive)	HQ - LOAEL (mid-range)
DCC	sand bass (barred & spotted), anchovy (deepbody & slough), CA halibut, topsmelt, shiner perch	PCBTEQ	Central	33	0.0173	0.0902		0.297	1	0.0052	0.0268	HQ mean	4.7	0.10	0.03
												HQ max	24.4	0.54	0.15
Scoter	crustacea, mollusks, polychaetes	PCBTEQ	Central	14	0.0064	0.0068		0.312	1	0.0020	0.0021	HQ mean	1.8	0.04	0.01
												HQ max	1.9	0.04	0.01
WEGU	Anchovy (deepbody & slough), topsmelt	PCBTEQ	Central	17	0.0240	0.0902		0.412	0.9	0.0089	0.0334	HQ mean	8.1	0.18	0.05
												HQ max	30.4	0.67	0.19
CLT	Arrow goby, goby sp., CA killifish, anchovy (northern & slough), shiner perch, topsmelt	PCBTEQ	South	12	0.0068	0.0083		0.818	1	0.0055	0.0068	HQ mean	5.0	0.11	0.03
												HQ max	6.2	0.14	0.04
CT	sand bass (barred & spotted), anchovy (deepbody, northern, slough), shiner perch, topsmelt	PCBTEQ	South	19	0.0100	0.0455		0.434	0.5	0.0022	0.0099	HQ mean	2.0	0.04	0.01
												HQ max	9.0	0.20	0.06
DCC	sand bass (barred & spotted), anchovy (deepbody, northern, slough), CA halibut, shiner perch, topsmelt	PCBTEQ	South	25	0.0091	0.0455		0.297	1	0.0027	0.0135	HQ mean	2.5	0.05	0.02
												HQ max	12.3	0.27	0.08
Scoter	misc. crustacea, mollusks, polychaetes	PCBTEQ	South	14	0.0064	0.0064		0.312	1	0.0020	0.0020	HQ mean	1.8	0.04	0.01
												HQ max	1.8	0.04	0.01

Representative species	Taxa in samples		Region	Sample N	EPC - Ave	EPC - Max		FIR (FI/BW) (g_{food}/g_{BW}-d)	AUF	ave. daily dose (ng_{TEQ}/g_{BW}-d)	Max daily dose (ng_{TEQ}/kg_{BW}-d)		HQ - NOAEL (lowest)	HQ - LOAEL (most sensitive)	HQ - LOAEL (mid-range)
WEGU	anchovy (deepbody, northern, slough), topsmelt	PCBTEQ	South	12	0.0075	0.0455		0.412	0.9	0.0028	0.0169	HQ mean	2.5	0.06	0.02
												HQ max	15.3	0.34	0.09

Table G5. Estimated exposure point concentrations (EPCs; ng/g ww), daily dose rates and hazard quotients for dietary PBDE exposure by representative avian species that forage in San Diego Bay, assuming foraging is primarily in the northern, central or southern region of the bay.

Representative species	Taxa in samples		Region	Sample N	EPC - Ave	EPC - Max		FIR (FI/BW) (g_{food}/g_{BW}-d)	AUF	ave. daily dose (ng_{PBDE}/g_{BW}-d)	Max daily dose (ng_{PBDE}/kg_{BW}-d)		HQ - NOAEL (lowest)	HQ - LOAEL (most sensitive)
												TRVs →	(9.6)	(96)
CLT	Black perch, goby sp., shiner perch, topsmelt	PBDE	North	9	6.14	12.51		0.818	1	5.02	10.22	HQ mean	0.52	0.05
												HQ max	1.06	0.11
CT	sand bass (barred & spotted), perch (black & shiner), topsmelt	PBDE	North	15	4.84	12.51		0.434	0.5	1.05	2.71	HQ mean	0.11	0.01
												HQ max	0.28	0.03
DCC	sand bass (barred & spotted), perch (black & shiner), CA halibut topsmelt	PBDE	North	22	3.76	12.51		0.297	1	1.12	3.72	HQ mean	0.12	0.01
												HQ max	0.39	0.04
Scoter	Crabs, brown shrimp, crustacea, mollusks, polychaetes	PBDE	North	16	2.68	6.17		0.312	1	0.85	2.06	HQ mean	0.09	0.01
												HQ max	0.21	0.02
	sediment									0.01	0.13			
WEGU	Topsmelt	PBDE	North	3	5.53	7.63		0.412	0.9	2.05	2.83	HQ mean	0.21	0.02
												HQ max	0.29	0.03
CLT	Goby sp., shiner perch,slough anchovy, topsmelt	PBDE	Central	14	3.51	15.20		0.818	1	2.87	12.43	HQ mean	0.30	0.03
												HQ max	1.29	0.13
CT	sand bass (barred & spotted), anchovy (deepbody & slough), topsmelt, shiner perch	PBDE	Central	26	2.37	15.20		0.434	0.5	0.51	3.30	HQ mean	0.05	0.01
												HQ max	0.34	0.03
DCC	sand bass (barred & spotted), anchovy (deepbody & slough), CA halibut, topsmelt, shiner perch	PBDE	Central	33	2.26	15.20		0.297	1	0.67	4.52	HQ mean	0.07	0.01
												HQ max	0.47	0.05
Scoter	crustacea, mollusks, polychaetes	PBDE	Central	14	0.46	10.51		0.312	1	0.17	3.47	HQ mean	0.02	0.00
												HQ max	0.36	0.04

Representative species	Taxa in samples		Region	Sample N	EPC - Ave	EPC - Max		FIR (FI/BW) (g_{food}/g_{BW}-d)	AUF	ave. daily dose (ng_{PBDE}/g_{BW}-d)	Max daily dose (ng_{PBDE}/kg_{BW}-d)		HQ - NOAEL (lowest)	HQ - LOAEL (most sensitive)
	sediment									0.02	0.19			
WEGU	Anchovy (deepbody & slough), topsmelt	PBDE	Central	17	1.60	15.20		0.412	0.9	0.59	5.63	HQ mean	0.06	0.01
												HQ max	0.59	0.06
CLT	Arrow goby, goby sp., CA killifish, anchovy (northern & slough), shiner perch, topsmelt	PBDE	South	12	6.66	19.53		0.818	1	5.44	15.97	HQ mean	0.57	0.06
												HQ max	1.66	0.17
CT	sand bass (barred & spotted), anchovy (deepbody, northern, slough), shiner perch, topsmelt	PBDE	South	19	3.27	17.16		0.434	0.5	0.71	3.72	HQ mean	0.07	0.01
												HQ max	0.39	0.04
DCC	sand bass (barred & spotted), anchovy (deepbody, northern, slough), CA halibut, shiner perch, topsmelt	PBDE	South	25	2.99	17.16		0.297	1	0.89	5.10	HQ mean	0.09	0.01
												HQ max	0.53	0.05
Scoter	crustacea, mollusks, polychaetes	PBDE	South	14	12.73	99.45		0.312	1	3.99	31.25	HQ mean	0.42	0.04
												HQ max	3.26	0.33
	sediment									0.02	0.20			
WEGU	anchovy (deepbody, northern, slough), topsmelt	PBDE	South	12	4.30	17.16		0.412	0.9	1.59	6.36	HQ mean	0.17	0.02
												HQ max	0.66	0.07

Table G6. Estimated exposure point concentrations (EPCs; ng/g ww), daily dose rates and hazard quotients for dietary HPAH (or PAH-HMW) exposure by representative avian species that forage in San Diego Bay, assuming foraging is primarily in northern, central or southern region of the bay.

Representative species	Taxa in samples		Region	Sample N	EPC - Ave	EPC - Max		FIR (FI/BW) (g_{food}/g_{BW}-d)	AUF	ave. daily dose (ng_{HPAH}/g_{BW}-d)	Max daily dose (ng_{HPAH}/kg_{BW}-d)		HQ - NOAEL (sensitive)	HQ - LOAEL (most sensitive)
												TRVs →	14.3	(1430)
CLT	perch (black & shiner), goby sp.	HPAH	North	6	7	14		0.818	1	5.5	11.7	HQ mean	0.39	0.00
												HQ max	0.82	0.01
CT	sand bass (barred & spotted), perch (black & shiner)	HPAH	North	12	5	14		0.434	0.5	1.1	3.1	HQ mean	0.07	0.00
												HQ max	0.22	0.00
DCC	sand bass (barred & spotted), perch (black & shiner), halibut	HPAH	North	19	4	14		0.297	1	1.1	4.3	HQ mean	0.07	0.00
												HQ max	0.30	0.00
Scoter	Crabs, crustacea, mollusks, polychaetes	HPAH	North	16	864	12,430		0.312	1	275	3,899	HQ mean	19.2	0.19
												HQ max	273	2.73
	sediment	HPAH	Central							4.9	18.4			
CLT	Goby sp., shiner perch	HPAH	Central	3	5	9		0.818	1	4.1	7.7	HQ mean	0.29	0.00
												HQ max	0.54	0.01
CT	sand bass (barred & spotted)	HPAH	Central	9	4	9		0.434	0.5	0.8	2.1	HQ mean	0.06	0.00
												HQ max	0.14	0.00
DCC	sand bass (barred & spotted), shiner perch, halibut	HPAH	Central	16.00	3	9		0.297	1	0.9	2.8	HQ mean	0.06	0.00
												HQ max	0.20	0.00
Scoter	Crustacea, mollusks, polychaetes	HPAH	Central	14	6	130		0.312	1	3.5	48.6	HQ mean	0.25	0.00
												HQ max	3.40	0.03

Representative species	Taxa in samples		Region	Sample N	EPC - Ave	EPC - Max		FIR (FI/BW) (g_{food}/g_{BW}-d)	AUF	ave. daily dose (ng_{HPAH}/g_{BW}-d)	Max daily dose (ng_{HPAH}/kg_{BW}-d)		HQ - NOAEL (sensitive)	HQ - LOAEL (most sensitive)
	Sediment									1.7	8.1			
CLT	Goby sp., shine perch, slough anchovy	HPAH	South	3	5	11		0.818	1	4.3	9.0	HQ mean	0.299	0.003
												HQ max	0.631	0.006
CT	sand bass (barred & spoted), shiner perch, slough anchovy	HPAH	South	8	4	11		0.434	0.5	0.9	2.4	HQ mean	0.062	0.001
												HQ max	0.167	0.002
DCC	sand bass (barred & spotted), CA halibut, shiner perch, slough anchovy	HPAH	South	14	3	11		0.297	1	0.9	3.3	HQ mean	0.066	0.001
												HQ max	0.230	0.002
Scoter	Crustacea, mollusks, polychaetes	HPAH	South	14	29	131		0.312	1	9.9	44.0	HQ mean	0.693	0.007
												HQ max	3.077	0.031
	sediment									0.9	3.1			
WEGU	Slough anchovy	HPAH	South	1	11	11		0.412	0.9	4.1	4.1	HQ single	0.286	0.003

Appendix H. Seabirds as Indicators of Legacy Toxicant Concentrations in the Southern California Bight

SEABIRDS AS INDICATORS OF LEGACY TOXICANT CONCENTRATIONS IN THE SOUTHERN CALIFORNIA BIGHT

Corey A. Clatterbuck[1,2], Rebecca A. Lewison[1], Nathan Dodder[3], Catherine Zeeman[4], and Kenneth Schiff[3]

1. San Diego State University, Biology Department, San Diego, California, USA

2. University of California-Davis, Graduate Group in Ecology, Davis, California, USA

3. Southern California Coastal Water Research Project, Costa Mesa, California, USA

4. US Fish and Wildlife Service, Carlsbad Fish & Wildlife Office, Carlsbad, California, USA

DRAFT - July 1, 2016

Abstract

Local and global processes concentrate anthropogenic toxicants in aquatic systems. There, toxicants enter biotic food webs that can potentially adversely affect organismal physiology and immune function at lethal and sub-lethal concentrations. Natural resource and water quality stakeholders are often tasked with monitoring the extent and magnitude of contamination in aquatic food webs. However, most monitoring efforts are limited by spatial scale, numbers of species, and number of toxicants assessed, thus are unable to consider contamination at regional scales. Here, we show the utility of seabird eggs in a regional toxicant monitoring program across an urbanized region of coastline, the Southern California Bight. We assessed the egg contents from four seabird species for four organic contaminant classes (polychlorinated biphenyls, polybrominated diphenyl ethers, DDTs, and chlordanes) and three elements (mercury, selenium, and arsenic). Results indicate toxicants are detectable across species throughout the region, and levels are steady or declining based on comparison to results from historic site-specific monitoring. While some individuals were found to have toxicants at levels above those known to cause adverse effects, on average no species met or exceeded lowest observed adverse effect concentrations (LOAECs) and eggshell thicknesses were not related to PBDE or DDT concentrations. Our results suggest that continued monitoring of legacy and more recent contaminants is useful in the Southern California Bight to inform site remediation, management, and protection of threatened wildlife in coastal systems.

Disclaimer

The findings and conclusions relating to wildlife risk in this report are those of the authors. As such, they do not necessarily represent the views of the U.S. Fish and Wildlife Service.

Introduction

Human population density continues to increase in coastal areas of California (Crossett et al. 2004). This growth has led to increased urban, suburban, agricultural, and industrial development that introduces organic contaminants and heavy metals to coastal environments via point sources, runoff, and atmospheric deposition (Elliott and Elliott 2013, Schiff et al. 2001). While production of many toxicants is banned or closely regulated, many persistent toxicants remain in coastal waters and sediments for decades and are biomagnified in aquatic food webs. At high concentrations, contaminants can reduce individual survival and reproduction, resulting in population decline, particularly for top predators (Hellou et al. 2013, Bustnes et al. 2003). Even at sublethal levels, these toxicants can impair physiological, immune, and reproductive function (Finkelstein et al. 2007, Goutte et al. 2015, Tartu et al. 2013). Many stakeholders, including ocean coastal communities, fisheries, ports, and wildlife managers, are concerned with water quality and tasked with biomonitoring in coastal food webs. These monitoring efforts are typically local in spatial extent, with a single organization or agency monitoring a single site. While site-specific monitoring is mandated and provides useful information on toxin exposure, regional monitoring is also essential to provide comparable data among sites within a geographic area and to previous studies. Regional monitoring maximizes the ability to use biomonitoring efforts to meet mandated monitoring objectives, prioritize site remediation, trace the dispersal and uptake of toxicants in marine food webs.

The Southern California Bight (SCB), which extends from Point Conception, CA to Cabo Colnett, Baja California, Mexico is oceanographically complex and has high biodiversity (Gray 1997). The SCB abuts a densely populated coastline that houses an estimated 22 million people. Numerous natural and anthropogenic toxicants, including mercury (Hotham and Powell 2000, Komoroske et al. 2012), selenium (Ohlendorf 1985, Hotham and Powell 2000), arsenic (Komoroske et al. 2011), PCBs (polychlorinated biphenyls, e.g. industrial and electrical byproducts, Fry et al. 1995, Schiff and Allen 2000, Brown et al. 2006, Jarvis et al. 2007), PBDEs (polybrominated diphenyl ethers, e.g., flame retardants, Brown et al. 2006) CHLs (chlordanes, Ohlendorf et al. 1985, Schiff and Allen 2000), and DDTs (e.g. pesticides, Risebrough et al. 1967, Ohlendorf et al. 1995, Fry et al. 1995, Schiff and Allen 2000) have been identified in wildlife in the SCB. Many of these toxicants have also been detected in SCB sediments and coastal waters (Zeng et al. 2005, Dodder et al. 2011). For some avian species, these toxicants have been directly linked to population declines in this region (Hickey and Anderson 1968, Blus et al. 1972, Gress et al. 1973).

Seabirds have been identified as effective and efficient biomonitors of coastal ecosystem health (Elliott and Elliott 2013). As top predators, seabirds generally feed at high trophic levels, which is reflected in the biomagnification of toxicants in their body tissues (Burger and Gochfeld 2002). During egg formation, birds can maternally transfer toxicants into egg contents; thus, salvaged eggs (eggs left on a colony at the end of a nesting season) collection can be a low cost and non-invasive method to assess toxicant concentrations in coastal environments. Many seabird contaminant studies, as with general biomonitoring, are limited by the cost of chemical analyses to measure multiple toxicant classes, the number of species studied, or the spatial region where samples were obtained. In this study, we assess contamination levels of six toxicant groups in four different species of seabirds that nest across the Southern California Bight. Building on decades of contaminant research in the SCB, our study objectives are to assess the extent and magnitude of contamination in the SCB, characterize the risks to seabirds from contaminants in this region, and highlight the utility the of non-invasive seabird tissues in regional contaminant biomonitoring and assessments.

Methods

Study species

We examined the egg contents of four seabird species: California least tern (*Sterna antillarum browni*), Caspian tern (*Hydroprogne caspia*), double-crested cormorant (*Phalacrocorax auritus*), and western gull (*Larus occidentalis*). The selected species differ in foraging traits, which is known to influence contaminant load. For instance, California least terns and Caspian terns are both piscivorous birds, but California least terns consume a variety of marine taxa and thought to forage within 2 km of shore during the breeding season, when this species is found in California (Fournier et al. 2016 *in review*). Double-crested cormorants are piscivorous and forage by diving. Western gulls are generalists that forage on the ocean surface as well as on marine, coastal, and terrestrial subsidies. All species are constrained by body and gape size, where smaller species (California least tern) cannot consume larger prey items, unlike larger species. Thus, each species provides a unique signal in contamination from different depths in the water column and regions of the SCB.

Egg collection and processing

Eggs were collected from the nests of these 4 species from 16 sites in the Southern California Bight (Fig 1, Table 1) during spring and summer 2013. Egg collection was executed by permitted individuals at each site in accordance with State, Federal and IACUC guidelines. All collected eggs were determined to be fail-to-hatch eggs due to nest abandonment or were taken as part of a depredation effort. Eggs were placed in cardboard cartons and transported to the US Fish and Wildlife Office in Carlsbad, CA for subsequent processing.

Eggs were processed using standard protocols for avian egg harvest for chemical analysis, embryo examination, and shell thickness determination Eggs were cleaned with distilled water, weighed, and measured for maximum length and width to the nearest 0.1mm using a analog dial caliper. We measured volume as the weight of water displaced by the egg. For cracked eggs, we estimated volume using the generic approach by Hoyt (1979). Afterwards, we sliced eggs through the equator using a scalpel pre-rinsed with dilute nitric acid, distilled water, reagent grade acetone, and reagent grade hexane. We examined egg contents for approximate embryo age and malposition, placed contents into a kilned glass jar, and stored in a -20°C freezer until subsequent chemical analysis.

We let eggshells dry at room temperature for 30 days before measuring eggshell thickness. The thickness of each eggshell (shell + shell membrane) was measured at four points around the girth using a Starrett Model 1010M dial micrometer, which is accurate to 0.01 mm, and estimatable to 0.001 mm. Although a dial micrometer is commonly used, this instrument is affected by eggshell curvature, may overestimate measurements for small eggs (i.e., California least tern). We averaged the 4 eggshell thickness measurements for each sample to derive one thickness measurement per sample. To account for errors in measuring thinner eggshells, we also calculated Ratcliffe's index, $RI = \frac{S}{L \cdot W}$, where L is the maximum shell length, W is the maximum shell width, and S is the weight of the dry shell (Ratcliffe 1970). Because a single least tern egg does not contain enough material for all chemical analyses, we combined the contents of multiple least tern eggs into composite samples until sufficient matrix was present for subsequent analyses. Least tern composite samples comprised the egg contents of 2-4 eggs collected from the same site, and we averaged resulting least tern egg morphometrics by sample.

Chemical analysis and quality assurance

The analytical methods and quality assurance/quality control (QA/QC) protocols closely followed those of the Southern California Bight Program (Dodder et al. 2016). The analytes included 41 polychlorinated biphenyl (PCB) congeners, 15 polybrominated biphenyl ether (PBDE) congeners, 7 dichlorodiphenyltrichloroethane (DDT) related compounds, 5 chlordanes (CHLs), mercury, selenium, and arsenic. The individual analytes and reporting levels are provided in Table S1. Organic contaminants and selenium were measured by Physis Environmental Laboratories (PEL; in Anaheim, CA). Mercury was measured by the Sanitation Districts of Los Angeles County (LACSD; in Whittier, CA), and the City of San Diego, CA (CSD). Selenium and arsenic were measured by LACSD only.

An elemental inter-laboratory comparison was performed prior to the analysis of field samples. A single lab performed organic contaminant analyses, so no interlab comparisons took place. Two reference materials were used: National Institute of Standards and Technology (NIST) Standard Reference Material (SRM) 1946: Lake Superior Fish Tissue, and a chicken egg homogenate containing spiked concentrations of the target elements. For both materials, all laboratories were within ± 30% of the mean value for each element.

Each lab used established EPA methods or machinery to perform toxicant and egg content analysis (Table S2). Laboratories ran a set of QC materials with the field samples, including method blanks, spiked blanks (elements only), reference materials, matrix spikes, and laboratory sample duplicates. Each QC material had associated criteria for analytical frequency and accuracy (Table S3). The success of meeting these criteria was evaluated for each contaminant class (Table S4). In all cases, the frequency success was 100%. The accuracy success was generally between 84% and 100%, except as noted (Table S4).

Statistical analysis

All statistical analysis was performed in R (R Core Team, 2015). Results from PEL were reported on a wet weight basis, in addition to percent lipid in each sample. All concentrations were standardized to unadjusted dry weight, ng/g (ppb), to account for desiccation based on differences in egg collection dates. Contaminant levels were log10-normalized to fit test and model assumptions of normality. All boxplots indicate the median (horizontal line), 1st and 3rd interquartile range (box), and 1.5 times the interquartile range (error bars).

After Wilk-Shapiro and Levene's test showed that data between species were nonparametric ($p<0.05$), we used Kruskal-Wallis ANOVAs with post-hoc Holm's correction to compare differences in toxicant concentrations among species across all sites, among species at a single site, and within a species across multiple sites. We used t-tests to assess differences in contaminant concentrations between 2 species at a single site. For samples which were non-detects for any contaminant class, we set values to 0 for statistical analysis and summary statistics and to ½ MDL for geometric means. We conducted spatial analyses for California least terns and western gulls as sample size and egg collection distribution were sufficient to allow for spatial comparison. To assess spatial relationships with toxicant concentrations within species, we used linear mixed models with latitude, distance to urban areas, and the type of collection site (e.g., designated marine protected area) as fixed effects and site as a random effect. We compared models using Akaike's Information Criterion (AIC) and described significant predictors using likelihood ratio tests.
We also considered how organic contaminant levels changed relative to eggshell thickness as both PBDEs and DDTs have been associated with decreased eggshell thickness in avifauna (Harris and Elliott 2011). Because eggshell thickness is species-specific, we did not compare eggshell thicknesses between species. Instead, we ran linear regressions to compare eggshell thickness and Ratcliffe's index to log-normalized toxicant concentrations.

Effect levels

Effect levels can be used to delineate the toxicant concentrations at which adverse effects may occur. To put our results in this context, we compare our detected contaminant levels to previously published contaminant effect levels associated with adverse effects in other avifauna (Table 3). Although effect levels vary by species and contaminant, and there are limited data available on effect levels for particular species or contaminants, the selected thresholds are ones that have been used in other studies on contaminant levels in avian eggs. Two sets of thresholds were used in this analysis: No Observed Adverse Effect Concentration (NOAEC) and Lowest Observed Adverse Effect Concentration (LOAEC). NOAEC indicates a concentration threshold where there is no concern of adverse effects and LOAEC indicates the lowest level at which adverse effects may occur. Levels between NOAEC and LOAEC suggest the toxicant merits additional consideration. Additional information on selection of effect levels is available in the supplement. We compared the range and mean for our focal species to estimates from other avian species.
Due to the variety of reported contaminant concentrations in the literature, we used the R package "OrgMassSpecR" to convert contaminant concentrations to a standardized reporting metric, ng/g fresh weight. We report both means and geometric means to ease comparisons among studies. Fresh weight concentrations are what the wet weight concentrations would be if the egg sample were fresh and before any moisture loss that occurs, especially in abandoned eggs. The adjustment

eliminates an analytical artifact that significantly affects unadjusted wet weight-based concentrations. The extent of moisture loss from individual eggs was variable, such that unadjusted wet weight-based concentrations in some eggs would be over-reported by as much as nearly four-fold. To obtain fw-based values, wet weight-based contaminant levels reported by the laboratory were adjusted according to methods by Stickel et al. (1973), using an adjustment factor equal to the ratio of the egg volume to the egg weight for each egg that was sampled. Mean adjustment factors were calculated for those samples that were composites of multiple eggs (i.e., least terns).

Current methods for PCB screening measures PCB congeners, whereas historic data and screening levels used Aroclor mixtures to examine sum PCB concentrations. More recent studies used the Aroclor approach in conjunction with the sums of homologs and/or 90 or more congeners. All three measures of total PCB concentrations are in close agreement in studies of seabird eggs for which total PCB concentrations were measured in all three ways. We use a method from Zeeman et al. (2008), who calculated a least squares linear regression formula to relate the total PCB contaminants of past studies (>90 PCB congeners) to the 41 congeners measured in this study:

$$\text{Total PCBs (ng/g fw)} = 75.2 \text{ ng/g fw} + 1.17 \left(\sum \text{lab-reported PCB concentrations ng/g fw}\right)$$

Results

1.1. Organic contaminants: Comparisons among species and locations

We detected all targeted toxicants by class in every egg sample except CHLs, and toxicant concentrations varied among species. Percent lipid in samples was not related to organic contaminant concentrations. In general, Caspian terns (CATE) had the highest concentrations of all targeted toxicants, and California least terns (CLTE) had the lowest except for mercury (Fig 2). CATE and double-crested cormorants (DCCO) had similar ($p = 0.983$) and greater amounts of PCBs ($\chi^2(3) = 35.252$, $p << 0.001$) compared to western gulls (WEGU) and California least terns (CLTE, $p = 0.983$). DCCO had similar concentrations of PBDEs as CATE ($p < 0.084$), WEGU ($p < 0.879$), and CLTE ($p < 0.084$), but all other species were different from each other ($\chi^2(3) = 40.485$, $p << 0.001$, Fig 2). WEGU and DCCO had the highest concentrations of PBDEs. There was a similar pattern in DDTs ($\chi^2(3) = 51.813$, $p << 0.001$), where WEGU were different from CATE ($p << 0.001$), DCCO ($p < 0.001$), and CLTE ($p < 0.001$), but DCCO and CATE had the highest concentrations of DDTs ($p < 0.772$, Fig 2). CHLs also differed among species ($\chi^2(2) = 37.329$, $p << 0.001$), where CHL concentrations were higher in CATE than CLTE ($p < 0.006$) and WEGU ($p < 0.001$), and CHL concentrations were higher in CLTE ($p < 0.001$) than WEGU. We did not include DCCO in CHL analyses because a high proportion (3/8) samples were non-detects.

For CLTE and WEGU, we had sufficient sample size and spatial distribution of sampling to consider contaminant levels across the region by compound class and investigate whether any of the available predictor values explained the detected variability. AIC scores of regional comparisons and toxicant are in Table 4. For CLTE, we found that marine protected area status ($\chi^2(1) = 4.622$, $p < 0.032$) and latitude ($\chi^2(1) = 4.898$, $p < 0.005$) were significant predictors of PBDE exposure, though there was no significant interaction between the two predictors ($\chi^2(1) = 0.532$, $p = 0.466$). PBDE concentrations in CLTE samples decreased about 36% per degree of latitude and were 26% lower in sites located in MPAs (Figure 3). Conversely, DDT concentrations in CLTE samples increased with latitude ($\chi^2(1) = 11.553$, $p < 0.001$) by about 45% per degree of latitude (Figure 3). No model adequately explained variation in CLTE PCB or CHL concentrations.

In WEGU, we found a significant relationship between PCB concentrations and marine protected area status where PCB concentrations were significantly higher ($\chi^2(1) = 5.106$, $p < 0.024$) by about 250% for WEGU nesting outside of an MPA (e.g., NAS North Island, Figure 4). No fixed effects or their interactions significantly predicted PBDE, DDT, or CHL concentrations in WEGU.

Two sites had sufficient sample size to examine differences in contaminant concentrations among species: Bolsa Chica and Salt Works. We sampled CATE and CLTE eggs at Bolsa Chica and CATE, CLTE, and DCCO eggs at Salt Works. At Bolsa Chica, PCB (Welch's t-test: $t=10.474$, $df = 6.66$, $p < 0.001$), PBDE ($t = 9.366$, $df = 5.20$, $p < 0.001$), DDT ($t = 8.724$, $df = 5.98$, $p < 0.001$), and CHL ($t = -5.278$, $df = 6.11$, $p < 0.002$) concentrations were higher in CATE than CLTE (Figure 5).

At Salt Works, DDT concentrations differed ($\chi^2(2) = 8.07$, $p < 0.018$) among species, where CATE ($p = 0.043$) and DCCO ($p = 0.043$) had higher concentrations than CLTE, but CATE and DCCO concentrations were similar ($p = 0.351$; Figure 6). There were no observed differences in PCB ($\chi^2(2) = 5.66$, $p = 0.059$), PBDE ($\chi^2(2) = 4.17$, $p > 0.124$), or CHL ($t = =0.264$, $df = 2.42$, $p > 0.812$) concentrations between species at Salt Works.

1.2. Organic contaminants: effect on eggshell thickness

Regressions between CLTE eggshell thickness and PBDE and DDT concentrations explained very little of the observed variability in the data (PBDEs: $F(1,52) = 2.02$, $R^2 = 0.037$, $p = 0.16$; DDTs: $F(1,52) = 3.40$, $R^2 = 0.06$, $p = 0.07$, Figure S1). For WEGU, PBDE concentrations were not significantly related to eggshell thickness ($F(1,21) < 0.003$, $R^2 < 0.0002$, $p = 0.961$, Figure S2). There was a significant but weak relationship between WEGU DDT concentrations and eggshell thickness ($F(1,52) = 5.11$, $R^2 = 0.20$, $p = 0.034$, Figure S2), which suggests OC concentration may be one of many factors contributing to variation in WEGU eggshell thickness. The relationship between PBDE and DDT concentrations and Ratcliffe's index also explained little variability in the data for CLTE (PBDE: $F(1,51) = 1.16$, $R^2 = 0.02$, $p = 0.29$; DDT: $F(1,51) = 2.53$, $R^2 = 0.05$, $p = 0.12$, Figure S1) and WEGU (PBDE: $F(1,21) = 0.10$, $R^2 = 0.004$, $p = 0.75$; DDT: $F(1,21) = 0.45$, $R^2 = 0.02$, $p = 0.51$, Figure S2).

1.3 Element contaminants: Comparisons among species and locations

We found some evidence of significant difference in element contamination among species. Mercury concentrations significantly differed ($\chi^2(3) = 71.05$, $p << 0.001$) among species in a repeated pattern of concentrations ($p < 0.05$), with greatest to smallest found in CATE, CLTE, DCCO and WEGU in that order (Fig 2). For other elements there were fewer obvious patterns, although DCCO samples were not analyzed for selenium or arsenic. Selenium concentrations were significantly ($\chi^2(2) = 26.412$, $p << 0.001$) greater in CLTE than WEGU ($p << 0.001$), but CATE and WEGU ($p = 0.086$) and CATE and CLTE ($p = 0.884$) had similar selenium concentrations. CATE and WEGU had similar arsenic concentrations ($p = 0.075$), and both CATE ($p < 0.004$) and CLTE ($p << 0.001$) had higher arsenic concentrations than WEGU ($\chi^2(2) = 27.733$, $p << 0.001$). DCCO samples were not analyzed for selenium or arsenic. Likelihood ratio tests showed that no fixed effect significantly predicted mercury, selenium, or arsenic concentrations in regional comparisons of CLTE samples. No fixed effect significantly predicted mercury samples in WEGU. We did not conduct regional comparisons of selenium and arsenic in WEGU because samples from NAS North Island were not tested for these toxicants.

There was some evidence for differences in element concentrations between species nesting at the same site that was similar to the overall between species comparisons. At Bolsa Chica, CATE harbored significantly more mercury than CLTE ($t = 4.680$, $df = 4.80$, $p < 0.006$; Figure 5), but the two species had similar concentrations of selenium ($t = 0.656$, $df = 4.54$, $p > 0.543$) and arsenic ($t = -0.928$, $df = 6.62$, $p > 0.386$). At Salt Works, mercury concentrations differed ($\chi^2(2) = 27.733$, $p << 0.001$) and were higher in CATE than CLTE ($p < 0.029$) and DCCO ($p < 0.002$), whereas mercury concentrations were similar between CLTE and DCCO ($p > 0.125$). Samples at Salt Works were not analyzed for selenium or arsenic.

1.4 Screening levels

Across the region, no species exceeded the LOAEC-based thresholds for the legacy toxicants measured on a fresh weight basis (Table 2 & 3). However, DDT concentrations were above the NOAEC threshold for eggshell thinning for the majority of individuals in all species except CLTE (Table 3). In CLTE, only one sample was above the NOAEC threshold for reduced productivity associated with DDT. Of all species, CATE had the highest proportion of individuals above NOAEC thresholds for multiple toxicants (Table 3). The majority of CATE also had DDT concentrations above the NOAEC threshold for reduced productivity associated with DDT in most sensitive species, with two individuals above the LOAEC threshold for less sensitive seabirds. Four individual CATE were above the NOAEC threshold for mercury, and two individual CATE also were above the LOAEC threshold for mercury in sensitive species. In DCCO, one individual was above the NOAEC level for PBDE and two were above the NOAEC threshold for reduced reproductive activity associated with DDT. In WEGU, eight individuals were above the NOAEC threshold for PBDEs and three were above the NOAEC threshold for reduced productivity associated with DDT. No individuals in any species were above the NOAEC thresholds for PCBs or Arsenic. Effect thresholds were not available for CHLs.

Discussion

Although seabirds have been established as sentinels of marine systems (e.g., Burger and Gochfeld 2002, Elliott and Elliott 2013), most contaminant monitoring efforts have yet to include seabirds as part of the typically studied species, a list that often includes sediment, bivalves, fish, and water quality (Dodder et al. 2011). This study confirms that salvaged eggs can be used to monitor both organic and element contaminants (Braune et al. 2002, Elliott and Elliott 2013).

Organic contaminants

Our findings confirm evidence of a continued decline in many organic contaminants (Dodder et al. 2011) in the SCB, yet these legacy toxicants persist in the SCB food web. Every sample across each of 13 sites contained congeners from each class of pollutants assessed with the exception of CHLs. In general, we found larger, piscivorous species (CATE and DCCO) had higher contaminant levels than the generalist (WEGU) and smaller (CLTE) species (i.e., in Figures 5 and 6), a finding common with previous research (Burger and Gochfeld 1997, Braune et al. 2005). While all species in this study are piscivorous, DCCO and CATE diets likely comprise larger and older fish (due to a larger gape size) and may consume higher proportions of fish in their diet, (versus other marine species like krill).

Using eggs from the two species for which samples were collected across the most sites in the SCB, California least tern (CLTE) and WEGU, we found evidence for significant spatial patterns of organic contaminant exposure. For CLTE, PBDEs increased and DDTs decreased from north to south. The observed pattern for DDTs may be explained by the location of the Palos Verdes Shelf Superfund Site, which lies just north of the Bolsa Chica nesting site. The increase in PBDEs detected in the southern CLTE colonies has not been documented previously. However, sediments in San Diego Bay harbor higher concentrations of PBDEs than elsewhere in the SCB, followed by Los Angeles Harbor and Long Beach Harbor, likely due to stormwater runoff from local sources (Dodder et al. 2011). A previous study of CLTE in San Diego Bay found demonstrably higher mean levels of PBDEs (2,210 ng/g lipid weight) than those measured here (Zeeman et al. 2008). However, both this study and existing literature confirm that the highest levels of PBDEs in CLTE were found in sites near southern San Diego Bay. Increased PBDE monitoring in San Diego Bay will better inform this spatial pattern. In WEGU, PCBs decreased from north to south. This finding reflects known patterns of PCB contamination in the SCB, where sediments in San Diego Bay harbored greater PCB concentrations than the northerly Channel Islands nesting sites (Maruya and Schiff 2009).

Element contaminants

The results from the element analyses differ both in terms of contamination levels and spatial patterns of accumulation than the organic compounds. Although piscivorous seabirds like CLTE are not thought to be sensitive to mercury at the levels reported here (Shore et al. 2011), our data suggest that CLTE had higher mercury concentrations than other monitored species. Unlike many organic contaminants, mercury is both a point and non-point source pollutant, with mercury levels varying based on local anthropogenic activity at small temporal and spatial scales as well as from the global mercury cycle (Selin 2009). Additionally, mercury is not lipophilic and, in eggs, is concentrated in albumen predominantly in the form of bioavailable methylmercury (Ackerman et al. 2013). Thus, while mercury can biomagnify in food webs, the mechanisms by which biomagnification occurs differs from organic contaminants. The increased mercury concentrations in least terns versus higher trophic species (in this study, CATE and DCCO) may be due to differences in foraging location. Breeding least terns depend on resources in nearshore surface waters where mercury concentrations are higher likely due to urban influences (but see Peterson et al. 2015) suggesting that mercury levels in CLTE may be a proxy for mercury levels in the nearshore environments in the SCB. A second explanation could be that CLTE mercury concentrations are associated with their overwintering area, as migratory CATE and DCCO in the central US and Canada have shown (Lavoie et al. 2015). This evidence suggests seabirds may have little capacity to excrete body-bound mercury via burning adipose tissue during migration, an excretion pathway that has been suggested for organic contaminants. Further study of mercury concentrations in different body tissues (e.g., feathers) would inform possible sources of mercury in these species.

While site-level data on selenium and arsenic was not available for our focal species, detected mercury concentrations were temporally variable at local scales. For instance, mean mercury concentrations decreased in CLTE nesting at the D-Street fill (948 ng/g dw) in the 1980's (Hotham and Zador 1995) but increased in CLTE nesting at Tijuana River Estuary (1010 ng/g dw) by ~300 ng/g dw compared to mean concentrations measured in eggs from 1994-1996 (Hotham and Powell 2000). Mean mercury levels in CATE (541 ng/g fw) nesting at Salt Works in San Diego Bay are also similar to those analyzed in 2005 (Zeeman et al. 2008). Adverse reproductive effects from mercury exposure, including reduced clutch size, egg viability, and egg hatchability may occur at low concentrations (e.g., between 600 and 800 ng/g fw) in sensitive species (Shore et al. 2011). Mercury levels may not have fluctuated greatly in San Diego Bay over the last 10 – 20 years, but the increase in mercury levels at Tijuana River Estuary likely merit further investigation.

Potential for adverse effects and trends over time

Screening levels can help contextualize how toxins detected compare with toxicant concentrations at which adverse effects take place and may govern management of contaminant sources. In terms of potential for adverse effects, the evidence was mixed. No species on average was at risk of adverse effects from any toxicant class, though a few individuals harbored contaminants at or above the LOAEC. Results above the NOAEC levels show that species may potentially be adversely affected by toxicants, but the specificity of the effects of toxicants at these concentrations is low. Unfortunately, little is known about the toxicity of these chemicals at low, sub-lethal concentrations, and even less is known about the additive or synergistic effects of contaminants and other stressors, including interannual periods of low food availability and climate change (e.g., Noyes and Lema 2015).

Because there has been contaminant monitoring at specific sites within the SCB, we can also compare results from this study to previously monitored sites. On average, organic contaminants were detected in lower concentrations here than those found

previously in seabird eggs in the SCB, including DDTs (424 ng/g wet weight) in WEGU nesting at NAS North Island (Jimenez-Castro et al. 1995), PCBs (512 ng/g ww), PBDEs (2550 ng/g lipid weight), and DDTs (1596 ng/g ww) in nesting CATE at Salt Works (Zeeman et al. 2008), PCBs (290 ng/g ww) and PBDEs (1320 ng/g lw) in nesting CLTE at Salt Works (Zeeman et al. 2008), and PCBs (165 ng/g ww) and DDTs (179 ng/g ww) in nesting CLTE at the Tijuana River Estuary (Hotham and Powell 2000). However, mean DDT concentrations in CLTE (764 ng/g ww) nesting at Salt Works were higher by about 400 ng/g on average, and above the maximum value of DDT concentrations in 2008 (Zeeman et al. 2008). While these contaminants continue to decline below LOAEC thresholds, they still persist at detectable levels in coastal wildlife in the SCB.
Another important comparison to contextualize detected contaminant levels in seabirds at the regional scale is to compare concentrations among sample types, e.g. sediment, bivalves, fish, and water. While important, these comparisons are challenging without a clearer understanding of the pathway by which seabirds are exposed to toxicants in a food web. Clarity on this pathway may be supported using diet, stable isotope or additional contaminant data from water, sediment and prey invertebrates and fish (Braune et al. 2002). Additional samples from seabird tissues formed at different times within the life cycle or annually, such as feathers or otoliths, can help pinpoint the geographic source of contamination (Lavoie et al. 2015). Telemetry and movement data for seabirds may provide greater explanatory power and links to observed toxicant concentrations throughout the SCB food web. Additional efforts are needed to compare the contaminant levels in the SCB across these sample types.

Contaminants and eggshell thinning

Although all four species exceeded the DDT NOAEC threshold for eggshell thinning in most sensitive species, we did not find a relationship between DDT or PBDE contamination and eggshell thickness or Ratcliffe's index in CLTE or WEGU. Values for eggshell thickness in CLTE and WEGU are similar to recent findings (Jimenez-Castro et al. 1995, Zeeman et al. 2008), and demonstrate that shell thickness for neither species have returned to thicknesses observed before DDT was in widespread use (Kiff 1994). Eggshell thinning is a concern for seabirds because it can lead to non-viable eggs and reproductive failure. There is historical evidence of eggshell thinning in the SCB associated with exposure to p,p-DDE, a metabolite of DDT converted in aquatic systems (Hickey and Anderson 1968). Avian species have varying sensitivities to DDE exposure (USDOI 1998). Since identification of the Palos Verdes Shelf Superfund Site and requisite remediation and monitoring, many avian species nesting in the SCB now lay eggs with shell thicknesses approaching pre-DDT levels (e.g., 0.367 mm for WEGU in Jimenez-Castro et al. 1995). This study supports this trend, where eggshell thicknesses approached pre-1945 thicknesses (CLTE: 0.152mm in Blus & Prouty 1979, WEGU: 0.376 mm by L. Kiff in Jimenez-Castro et al. 1995) and many species laid eggs with mean DDT concentrations between NOAEC and LOAEC (USDOI 1998). Other factors, including laying order and egg age, also influence eggshell thickness (Hunt and Hunt 1973). While we did not have access to this in our study, our regressions on eggshell thickness suggest that other predictors may explain more variability in eggshell thickness than the compounds analyzed here.

Importance of regional monitoring

Monitoring contaminants at the regional scale across taxa and sample types is essential to track health of marine systems. Seabirds are considered effective monitors of marine ecosystem health (Elliott and Elliott 2013), but few large-scale toxicant monitoring efforts include seabirds as indicator species (but see Braune et al. 2005). Here, we demonstrated the importance of including seabirds in a long-term biomonitoring program of the SCB a 400 km urbanized coastline. There are advantages to using seabird tissues to examine regional contamination patterns. Abandoned and fail-to-hatch eggs are easily sampled at low cost on seabird colonies, compared to effort needed for sampling marine sediments, macrofauna, and fish. Seabird eggs are often large enough to test for multiple contaminant classes, or can be reliably combined within site to give site-specific parameters. Due to seabirds' position atop many aquatic food webs, recorded contaminant values are biologically relevant to other wildlife, aquatic resources (e.g. seafood), and humans.

The ability to compare contaminants regionally or among sites remains limited largely due to non-standard reporting for toxicant levels, sample type and inter-laboratory variation. The inability to transpose reporting metrics presents a substantial challenge to larger scale comparative research. To address this, we have reported toxicant concentrations in four different units – dry weight, wet weight, fresh weight, and lipid weight - to enable comparisons of toxicant concentrations with future studies. While many seabird tissues can be used to assess body burdens of toxicants, concentrations in each are not comparable to each other due to differences in how each toxicant may be stored or metabolized in the body. Inter-laboratory variation in quality assurance and quality control standards will also affect the accuracy with which contaminant levels are reported. We quantified this interlaboratory variation by conducting round robin exercises with bird egg samples prior to the regional survey. Improved standardization within the monitoring community such as this will aid comparisons between local studies and scale-up to regional assessments.

Conclusions

- **Legacy contaminants remain dispersed and persistent in seabirds and the marine system in the SCB.**

Over 100 bird egg samples were analyzed as part of the Bight regional monitoring in 2013. DDTs and PCBs were detected in virtually every sample from all four bird species, regardless of location.

- **Legacy toxicant concentrations are comparable to or lower than previous studies in the SCB.**

While we cannot compare legacy contaminant concentrations at regional scales because this is the first regional survey of bioaccumulation in bird eggs, we can compare the regional concentrations in 2013 to historical site-specific studies. DDT and PCB concentrations measured in historical studies, which occurred largely in San Diego Bay and date as far back as 20 years, were typically similar to higher than what was observed in the 2013 regional monitoring.

- **Observed toxicant levels were generally lower than those that have potential for adverse effects, but synergistic or additive effects are unknown at this time.**

We used two thresholds from the literature for comparing the relative risk of regional contaminant concentration data in bird eggs: no observed apparent effects concentrations (NOAEC) and lowest observed apparent effects thresholds (LOAEC). Only 2 of 101 bird egg samples exceeded the LOAEC for any single contaminant, indicating that the probability of effects was likely low. However, many – and sometimes all samples for single species - exceeded NOAEC thresholds. The cumulative effects of multiple contaminants at these very low levels in uncertain.

- **There was no evidence of a relationship between eggshell thickness and PBDE or DDT levels in seabird eggs**

Based on the regional distribution of DDT and PBDE in bird eggs from Western gulls and California Least Terns, we did not see strong relationships with eggshell thickness. The lack of relationship may be a result of low concentrations relative to studies from the 1960's and 70's, when eggshell thinning was an important indicator of seabird population effects.

- **This study highlights the utility of seabirds as an indicator species for contaminant bioaccumulation in this region.**

The regional monitoring program was able to successfully sample, process, analyze, and assess contaminants in seabird eggs. The collaboration, coordination, and integration among sampling teams, laboratories, and managers proved that a regional monitoring program for bioaccumulation in sea birds is a viable and productive monitoring approach.

Recommendations

- **Compare toxicant concentrations in seabird eggs (this study) to toxicant concentrations in water, sediment, invertebrate, and fish from the SCB to track exposure pathways for wildlife.**

While we were able to assess the extent and magnitude of contaminants that bioaccumulate in sea bird eggs, we do not yet know how or where the contaminants came from. Future studies should investigate the trophic transfer through the coastal food web. This will be especially important for improving the sediment quality objectives indirect pathway exposure for wildlife risk.

- **Introduce additional monitoring tools, including stable isotope, telemetry, or GPS technology, to improve current understanding of exposure pathways in SCB wildlife**

One mechanism for deciphering where contaminants come from is to use tools that either track where sea birds are feeding (i.e., near sediment contaminant hot spots) or geochemical tracers of contaminants. Tools currently exist to track feeding locations, including GPS transmitters that can be attached to individual birds nesting in the Bight. Tools currently available for tracking contaminants include stable isotope chemistry. Stable isotope chemistry has been used for other elements (i.e., lead), but are more difficult for organic contaminants.

- **Continue regional monitoring efforts to detect the occurrence of emerging contaminants in coastal and marine ecosystems**

Since sea birds have proven to be a feasible and useful bioindicator for legacy (DDT, PCB) and more recent (PBDE) contaminants, sea birds can also be useful indicators for new and emerging contaminants. Bioaccumulation of new contaminants has been identified as a priority by the State Water Board's Expert Panel on constituents of emerging concern.

Acknowledgements
We thank the USFWS Analytical Control Facility for providing the egg reference material. We thank Physis Environmental Laboratories, the Sanitation Districts of Los Angeles County, and the City of San Diego for providing analytical services. Thanks to Dovi Kacev, Chris Tyson, and Matthew Savoca for input on statistical analysis. Thanks to members of the Bight committee for reviews.

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Table 1. Number of egg samples collected from each species by site. Asterisks indicate sites within a Marine Protected Area (MPA).

	California least tern	Caspian tern	Double-crested cormorant	Western gull
Anacapa Island*				5
Batiquitos Lagoon*	5			
Bolsa Chica Reserve*	8	5		
Chula Vista Reserve	4			
D-Street Fill	6			
LA Harbor				
Lindbergh Field	5			
NAS North Island				8
Pismo Beach				1
Point Mugu	9			
Salt Works	3	10	8	
Santa Barbara Island*				9
Tijuana Estuary	7			
Vandenberg	2			1
SUM:	**55**	**15**	**8**	**24**

Table 2. Summary statistics of toxicant concentrations by species. The concentrations of congeners within organic contaminant classes are summed.

Table 2. Summary statistics of egg contaminant data. All units are ng/g (ppb) basis.
Geom Mean for CHL data was calculated by replacing NDs with 1/2 MDL, 0.025.

	DRY WEIGHT				WET WEIGHT			
	CLTE	WEGU	CATE	DCCO	CLTE	WEGU	CATE	DCCO
PCBs								
Range	124 - 3042	91 - 3863	645 - 9967	1436 - 11448	33 - 833	24 - 1089	141 - 2163	207 - 2255
Mean	709.3	1235.5	3413	4795	186.8	314.7	802.4	848.8
Geom Mean	562.5	712.8	2509.3	3284.2	142.7	187.5	626.9	527
PBDEs								
Range	34 - 824	71 - 3421	330 - 2070	90 - 1645	9 - 274	18 - 2749	87 - 449	11 - 324
Mean	198	676	1131	573.9	51.7	549.1	278.2	103.5
Geom Mean	145.6	409.7	988.6	339.2	37	107.8	247	54.4
DDTs								
Range	249 - 6991	447 - 9749	2291 - 42493	3036 - 24834	57 - 1636	123 - 2749	536 - 9221	355 - 4396
Mean	1216	1444	12251	8692	297.9	549.1	2819.3	1482.2
Geom Mean	869.1	1533.6	8137.2	6975.8	220.5	403.5	2032.9	1119.3
CHLs								
Range	ND - 126	ND - 58	9 - 394	ND - 24	ND - 30	ND - 17	2 - 71	ND - 4
Mean	30.7	8	101.2	8.8	7.8	2.2	22.7	1.6
Geom Mean	21.6	0.96	56.1	1.2	5.6	0.4	14	0.4
Mercury								
Range	462 - 1666.7	66.92 - 1104.84	1210 - 4617	247 - 1280	142 - 401	17 - 274	322 - 1100	35 - 228
Mean	949.4	256	2184	482.4	243.1	73.6	537	82.2
Geom Mean	927	223.9	1982.8	411.2	235.8	58.9	495.4	66
Selenium								
Range	1883 - 3307	1480 - 2160	1566 - 6500	NA	460 - 774	367 - 656	368 - 1170	NA
Mean	2495	1752	3165	NA	609.3	495.6	642	NA
Geom Mean	2473.8	1741.6	2772.8	NA	605	490.3	586.8	NA
Arsenic								
Range	315 - 683	58 - 566	295 - 532	NA	68 - 191	15 - 155	53 - 107	NA
Mean	493.8	160.3	401.1	NA	122.3	45.25	85.7	NA
Geom Mean	484.7	136	393.6	NA	118.5	38.3	83.3	NA
Shell Thickness (mm)								
Range	0.123 - 0.169	0.331 - 0.442	0.3 - 0.371	0.328 - 0.466				
Mean	0.145	0.371	0.3413	0.4096				
Ratcliffe Index								
Range	0.573-0.804	1.567-1.954	1.279-1.682	1.773-2.123				
Mean	0.666	1.804	1.529	1.986				
Percent Lipid								
Range	4.08 - 24.8	6.75 - 13.6	5.01 - 12.3	0.77 - 6.58				
Mean	10.33	10.01	9.42	4.525				

	LIPID WEIGHT				FRESH WEIGHT			
	CLTE	WEGU	CATE	DCCO	CLTE	WEGU	CATE	DCCO
PCBs								
Range	172 - 7741	246 - 10896	1953 - 30987	4855 - 34274	26 - 685	23 - 1006	134 - 1960	172 - 1951
Mean	1819.4	3368.7	9305	18266	138.1	315.1	751.4	727.8
Geom Mean	1388	1905.8	6843.9	14033.6	109.1	183.4	592.5	461.8
PBDEs								
Range	42 - 3148	193 - 8215	1063 - 6987	548 - 4925	7 - 166	17 - 981	83 - 414	9 - 280
Mean	544.1	1777.8	3064	1983.6	37.5	171.1	261.2	88.9
Geom Mean	359	1095.4	2696.2	1449.4	28.2	99.6	233.4	47.7
DDTs								
Range	166 - 18257	1201 - 27492	5691 - 117436	13158 - 68043	49 - 1407	118 - 2538	519 - 8354	293 - 3648
Mean	3154.2	5580	33437	34981	231.5	529.3	2616.2	1280.8
Geom Mean	2144	4100.2	22193.1	29807.9	168.5	386.7	1921.2	980.8
CHLs								
Range	ND - 439	ND - 139	23 - 1414	ND - 76	ND - 25	ND - 17	2 - 65	ND - 3.5
Mean	82.5	20.9	281.1	27.5	5.8	2.2	21.3	1.4
Geom Mean	52.4	1.9	153	2.3	4.3	0.4	13.2	0.4
Mercury								
Range	309 - 5388	182 - 2988	3381 - 16587	835 - 4938	108 - 279	16 - 245	295 - 1020	31 - 192
Mean	2437	742.2	6083	2169	185.5	71.2	504.3	71.1
Geom Mean	2267.6	598.6	5407.9	1757	180.4	57.6	468.2	57.8
Selenium								
Range	3914 - 10813	3311 - 6090	3728 - 23353	NA	356 - 624	329 - 636	398 - 989	NA
Mean	6321	4655	9243	NA	487.1	470.5	591.5	NA
Geom Mean	6170.6	4596.8	7403.9	NA	483.1	464.4	557	NA
Arsenic								
Range	821 - 1761	159 - 1520	711 - 1537	NA	55 - 147	14 - 150	45 - 105	NA
Mean	1237.2	426.6	1083.3	NA	97.8	43.6	82.7	NA
Geom Mean	1209	359.1	1051	NA	94.7	36.5	79.1	NA

Table 3. Screening values for analyzed toxicants. No Observed Adverse Effect Concentration (NOAEC) are values below which no adverse effects are predicted. Lowest Observed Adverse Effect Concentration (LOAEC) are values at which eggshell thinning and/or reproductive success are impaired. Values between NOAEC and LOAEC are of concern. NOAEC thresholds for DDTs are conservative estimates for all birds. No thresholds are available for CHL data.

			Samples above threshold (sample size)				
Toxicant	Site	NOAEC (ng/g fw)	CATE	CLTE	DCCO	WEGU	Reference
PCB		2600	0 (15)	0 (55)	0 (8)	0 (23)	Harris and Elliott 2011
PBDE		200	10 (15)	0 (55)	1 (8)	8 (23)	Rattner et al. 2011
	Anacapa Isl					2 (4)	
	Bolsa Chica		4 (5)				
	NAS North Isl					4 (8)	
	Salt Works		6 (10)		1 (8)		
	Santa Barbara Isl					2 (9)	
DDT[a]		200	15 (15)	21 (55)	8 (8)	19 (23)	DOI 1998
	Anacapa Isl					2 (4)	
	Bolsa Chica		5 (5)	6 (8)			
	LA Harbor			3 (6)			
	Lindbergh Fld			1 (5)			
	NAS North Isl					7 (8)	
	Pismo Beach					1 (1)	
	Point Mugu			8 (9)			
	Salt Works		10 (10)	1 (3)	8 (8)		
	Santa Barbara Isl					8 (9)	
	TJ Estuary			1 (7)			
	Vandenberg			1 (2)		1 (1)	
DDT[b]		1000	12 (15)	1 (55)	2 (8)	3 (23)	DOI 1998
	Bolsa Chica		5 (5)				
	NAS North Isl					1 (8)	
	Point Mugu			1 (9)			
	Santa Barbara Isl					2 (9)	
	Salt Works		7 (10)		2 (8)		
Mercury		500	4 (15)	0 (52)	0 (8)	0 (23)	Burger and Gochfeld 1997
	Bolsa Chica		3 (5)				
	Salt Works		1 (10)				
Selenium		900	1 (5)	0 (29)	-	0 (15)	Ohlendorf and Heinz 2011
	Bolsa Chica		1 (5)				
Arsenic		910	0 (5)	0 (29)	-	0 (15)	DOI 1998

Toxicant		LOAEC (ng/g fw)	CATE	CLTE	DCCO	WEGU	Reference
PCB		23000	0 (15)	0 (55)	0 (8)	0 (23)	Harris and Elliott 2011
PBDE		1000	0 (15)	0 (55)	0 (8)	0 (23)	Harris and Elliott 2011
DDT[a]		10000	0 (15)	0 (55)	0 (8)	0 (23)	DOI 1998
DDT[b]		5000	2 (15)	0 (55)	0 (8)	0 (23)	DOI 1998
	Bolsa Chica		2 (5)				
Mercury		800	2 (15)	0 (52)	0 (8)	0 (23)	Henny et al. 2002
	Bolsa Chica		1 (5)				
	Salt Works		1 (10)				
Selenium		3000	0 (5)	0 (29)	-	0 (15)	Ohlendorf and Heinz 2011
Arsenic		>910	0 (5)	0 (29)	-	0 (15)	DOI 1998

[a]Thresholds for observed eggshell thinning in seabird species
[b]Thresholds for reduced reproductive activity in seabird species

Table 4. Model selection tables for CLTE spatial data. LogLik is the likelihood of the model fit. AICc is a relative measure of quality of the model with the given data. Delta is the difference between the listed model and the model with the lowest AICc. Weight is a proportional estimate of how often a model will best predict new data.

Model (**Toxicant class**)	df	logLik	AICc	delta	weight
PCBs					
MPA + (1 | Site)	4	-9.38	27.6	0	0.603
Lat + (1 | Site)	4	-10.3	29.5	1.917	0.231
Lat + MPA + (1 | Site)	5	-10.3	31.8	4.251	0.072
UrbanDist + (1 | Site)	4	-11.8	32.3	4.742	0.056
MPA + UrbanDist + (1 | Site)	5	-11.3	33.9	6.277	0.026
Lat + UrbanDist + (1 | Site)	5	-12.5	36.2	8.599	0.008
Lat + MPA + UrbanDist + (1 | Site)	6	-12.4	38.6	10.965	0.003
PBDEs					
Lat + (1 | Site)	4	13.1	-17.4	0	0.532
Lat + MPA + (1 | Site)	5	13.79	-16.3	1.046	0.315
MPA + (1 | Site)	4	11.44	-14.1	3.312	0.101
Lat + UrbanDist + (1 | Site)	5	11.17	-11.1	6.292	0.023
UrbanDist + (1 | Site)	4	9.18	-9.54	7.823	0.011
Lat + MPA + UrbanDist + (1 | Site)	6	11.65	-9.47	7.888	0.01
MPA + UrbanDist + (1 | Site)	5	10.14	-9	8.356	0.008
DDTs					
Lat + (1 | Site)	4	-1.23	11.3	0	0.764
Lat + MPA + (1 | Site)	5	-1.43	14.1	2.829	0.186
Lat + UrbanDist + (1 | Site)	5	-3.46	18.2	6.901	0.024
MPA + (1 | Site)	4	-5.49	19.8	8.512	0.011
UrbanDist + (1 | Site)	4	-5.68	20.2	8.888	0.009
Lat + MPA + UrbanDist + (1 | Site)	6	-3.69	21.2	9.902	0.005
MPA + UrbanDist + (1 | Site)	5	-6.26	23.8	12.507	0.001

Model (**Toxicant class**)	df	logLik	AICc	delta	weight
Mercury					
MPA + (1 | Site)	4	44.58	-80.28	0	0.578
Lat + (1 | Site)	4	43.86	-78.84	1.438	0.281
UrbanDist + (1 | Site)	4	42.95	-77.02	3.263	0.113
Lat + MPA + (1 | Site)	5	42.3	-73.23	7.052	0.017
MPA + UrbanDist + (1 | Site)	5	41.2	-71.03	9.252	0.006
Lat + UrbanDist + (1 | Site)	5	41.05	-70.73	9.547	0.005
Lat + MPA + UrbanDist + (1 | Site)	6	39.63	-65.31	14.97	0
Selenium					
MPA + (1 | Site)	4	32.61	-55.41	0	0.465
Lat + (1 | Site)	4	32.54	-55.27	0.142	0.433
UrbanDist + (1 | Site)	4	30.81	-51.8	3.614	0.076
Lat + MPA + (1 | Site)	5	31.08	-49.29	6.117	0.022
Lat + UrbanDist + (1 | Site)	5	28.9	-44.95	10.459	0.002
MPA + UrbanDist + (1 | Site)	5	28.68	-44.5	10.91	0.002
Lat + MPA + UrbanDist + (1 | Site)	6	27.28	-38.36	17.054	0
Arsenic					
Lat + (1 | Site)	4	30.41	-51	0	0.544
MPA + (1 | Site)	4	29.35	-48.88	2.124	0.188
UrbanDist + (1 | Site)	4	29.18	-48.54	2.461	0.159
Lat + MPA + (1 | Site)	5	30.14	-47.43	3.573	0.091
Lat + UrbanDist + (1 | Site)	5	28.06	-43.25	7.749	0.011
MPA + UrbanDist + (1 | Site)	5	27.44	-42.02	8.986	0.006
Lat + MPA + UrbanDist + (1 | Site)	6	27.23	-38.26	12.746	0.001

Table 4. Model selection tables for WEGU spatial data. CHLs are not included due to the high number of non-detects.

Model (**Toxicant class**)	df	logLik	AICc	delta	weight
PCBs					
Lat + (1 | Site)	4	-12.1	34.5	0	0.457
MPA + (1 | Site)	4	-12.3	34.9	0.43	0.368
Lat + MPA + (1 | Site)	5	-11.4	36.6	2.067	0.162
MPA + UrbanDist + (1 | Site)	5	-14.9	43.5	8.998	0.005
Lat + MPA + UrbanDist + (1 | Site)	6	-13.4	44.4	9.958	0.003
UrbanDist + (1 | Site)	4	-17.3	44.8	10.358	0.003
Lat + UrbanDist + (1 | Site)	5	-15.7	45.2	10.753	0.002
PBDEs					
Lat + MPA + (1 | Site)	5	-13.2	40.1	0	0.369
MPA + (1 | Site)	4	-14.9	40.1	0.086	0.353
Lat + (1 | Site)	4	-15.2	40.7	0.638	0.268
UrbanDist + (1 | Site)	4	-19	48.4	8.385	0.006
Lat + MPA + UrbanDist + (1 | Site)	6	-16.3	50.2	10.173	0.002
MPA + UrbanDist + (1 | Site)	5	-18.6	50.9	10.863	0.002
Lat + UrbanDist + (1 | Site)	5	-19	51.7	11.637	0.001

Model (**Toxicant class**)	df	logLik	AICc	delta	weight
DDTs					
Lat + (1 | Site)	4	-8.09	26.53	0	0.437
MPA + (1 | Site)	4	-8.16	26.68	0.145	0.406
Lat + MPA + (1 | Site)	5	-7.53	28.8	2.272	0.14
UrbanDist + (1 | Site)	4	-11.9	34.11	7.579	0.01
MPA + UrbanDist + (1 | Site)	5	-11.2	36.17	9.635	0.004
Lat + UrbanDist + (1 | Site)	5	-11.8	37.37	10.838	0.002
Lat + MPA + UrbanDist + (1 | Site)	6	-9.92	37.45	10.914	0.002
Mercury					
Lat + (1 | Site)	4	-2.05	14.45	0	0.421
MPA + (1 | Site)	4	-2.23	14.8	0.352	0.353
Lat + MPA + (1 | Site)	5	-1.18	16.1	1.653	0.184
UrbanDist + (1 | Site)	4	-5.2	20.76	6.306	0.018
MPA + UrbanDist + (1 | Site)	5	-3.64	21.04	6.589	0.016
Lat + UrbanDist + (1 | Site)	5	-4.61	22.97	8.515	0.006
Lat + MPA + UrbanDist + (1 | Site)	6	-3.44	24.48	10.024	0.003

Figure 1. Map of egg collection locations.

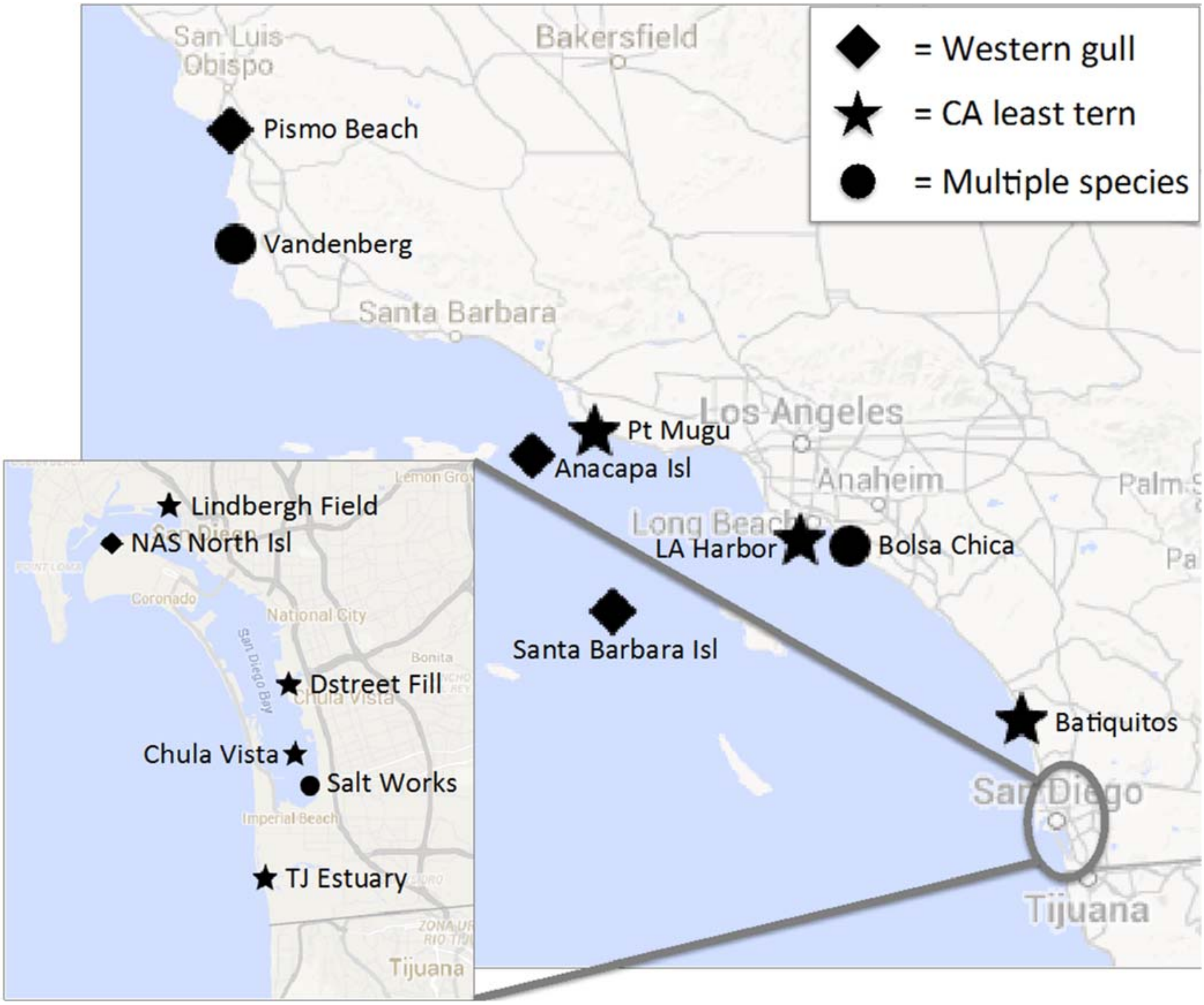


Figure 2. Toxicant concentrations by species. The concentrations of congeners within organic contaminant classes are summed by sample. Asterisks represent significant differences between species, and figures with two asterisks represent toxicant classes where all species have differing concentrations. Sample sizes are listed in parentheses below the 4-letter species ID for each toxicant class.

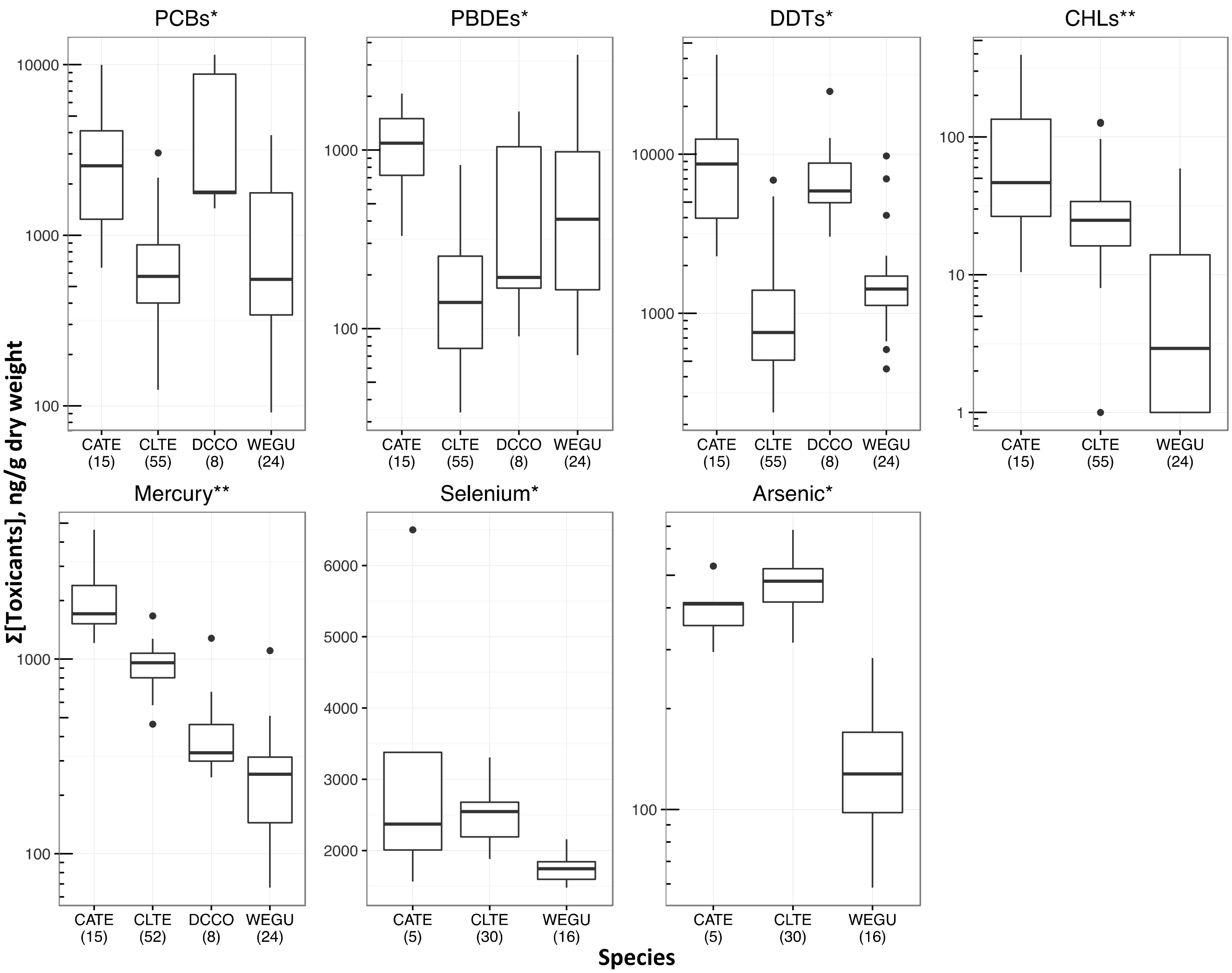


Figure 3. Latitudinal comparisons of summed toxicant concentrations in California least terns. Parentheses indicate sample size by site. Asterisks represent plots where a significant latitudinal trend is present.

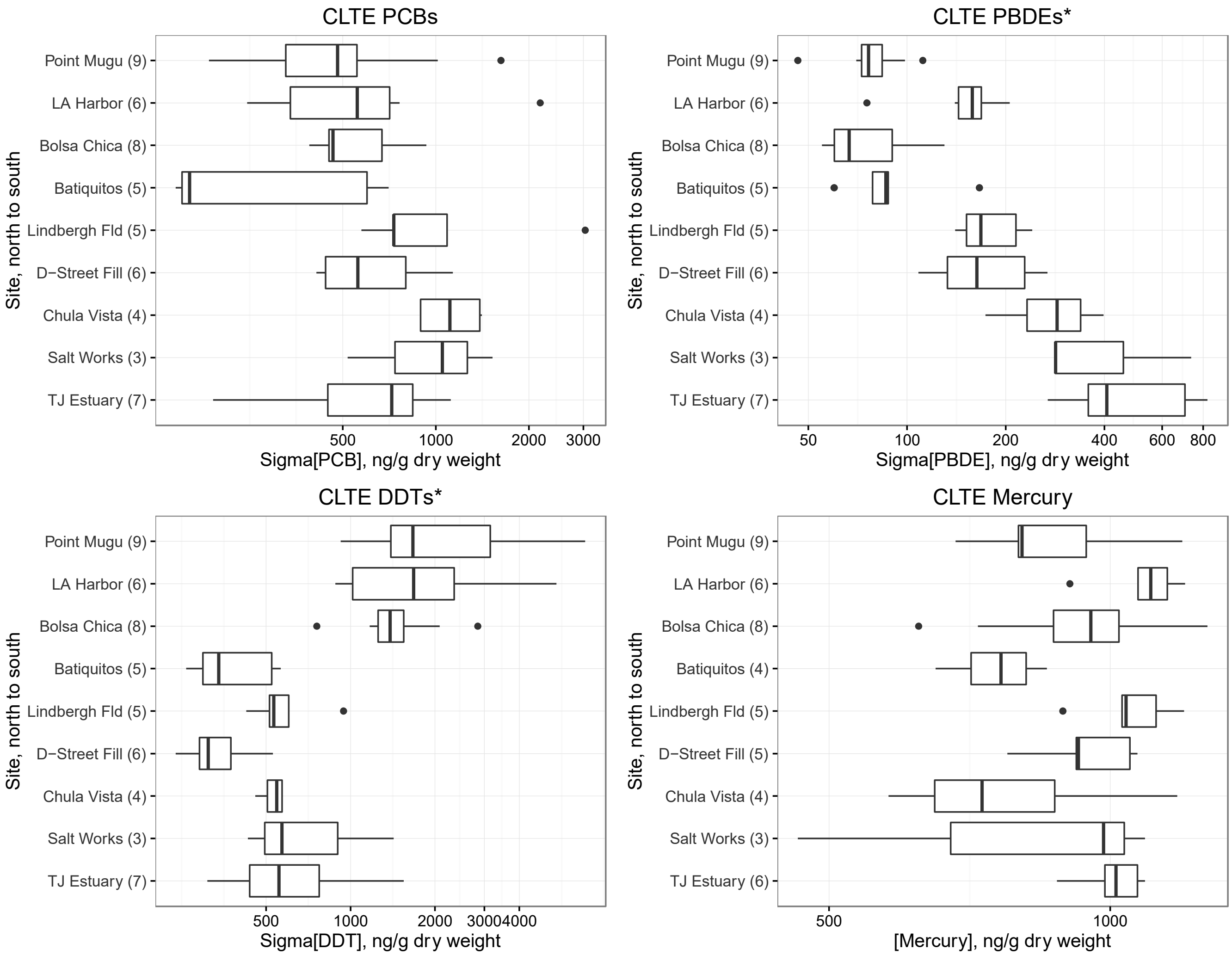


Figure 4. Latitudinal comparisons of summed toxicant concentrations in western gulls. Parentheses indicate sample size by site. Asterisks represent plots where a significant latitudinal trend is present.

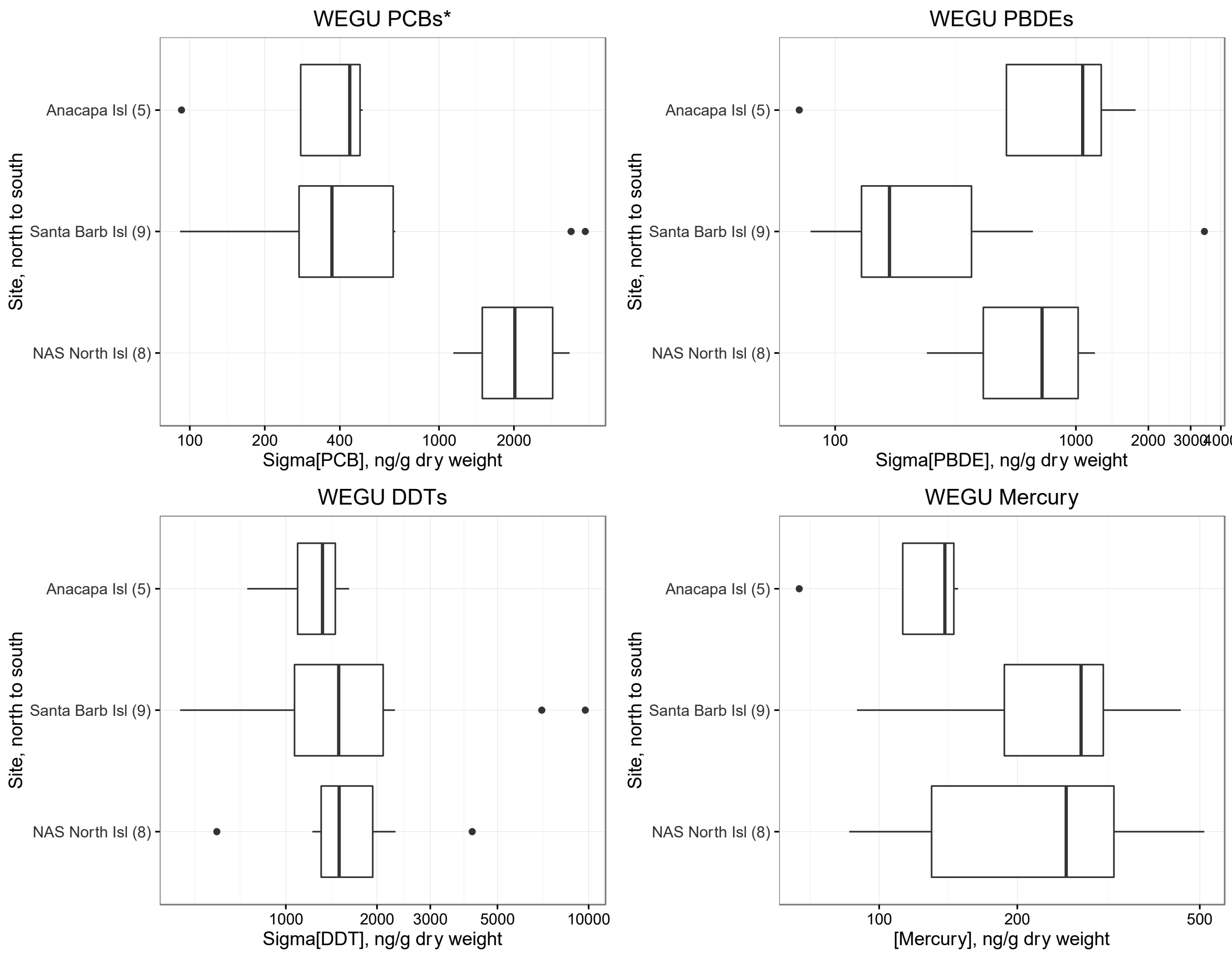


Figure 5. Toxicant concentrations in Bolsa Chica by species. The concentrations of congeners within organic contaminant classes are summed by sample. Asterisks represent significant differences between species. Sample sizes are listed in parentheses below the 4-letter species ID for each toxicant class.

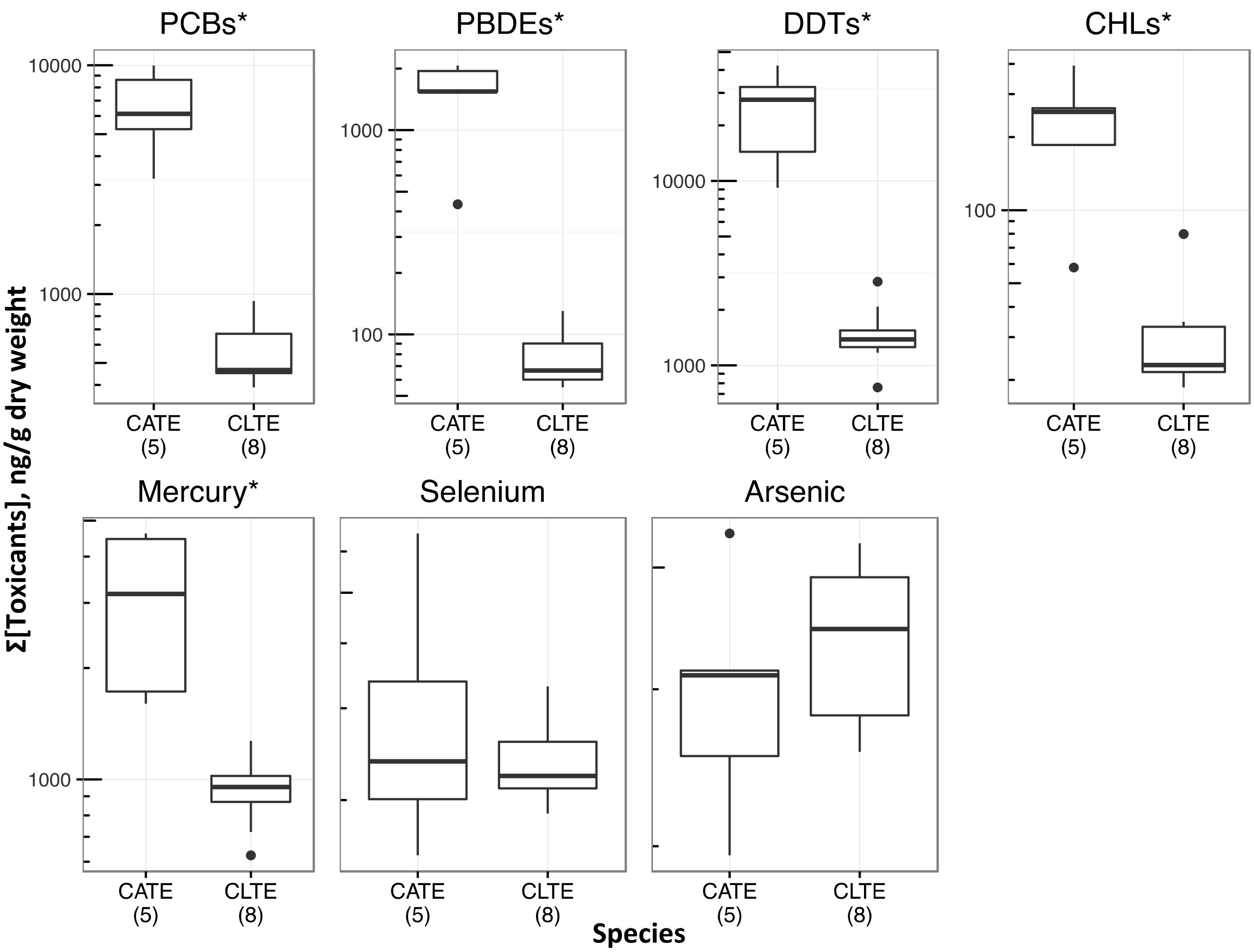
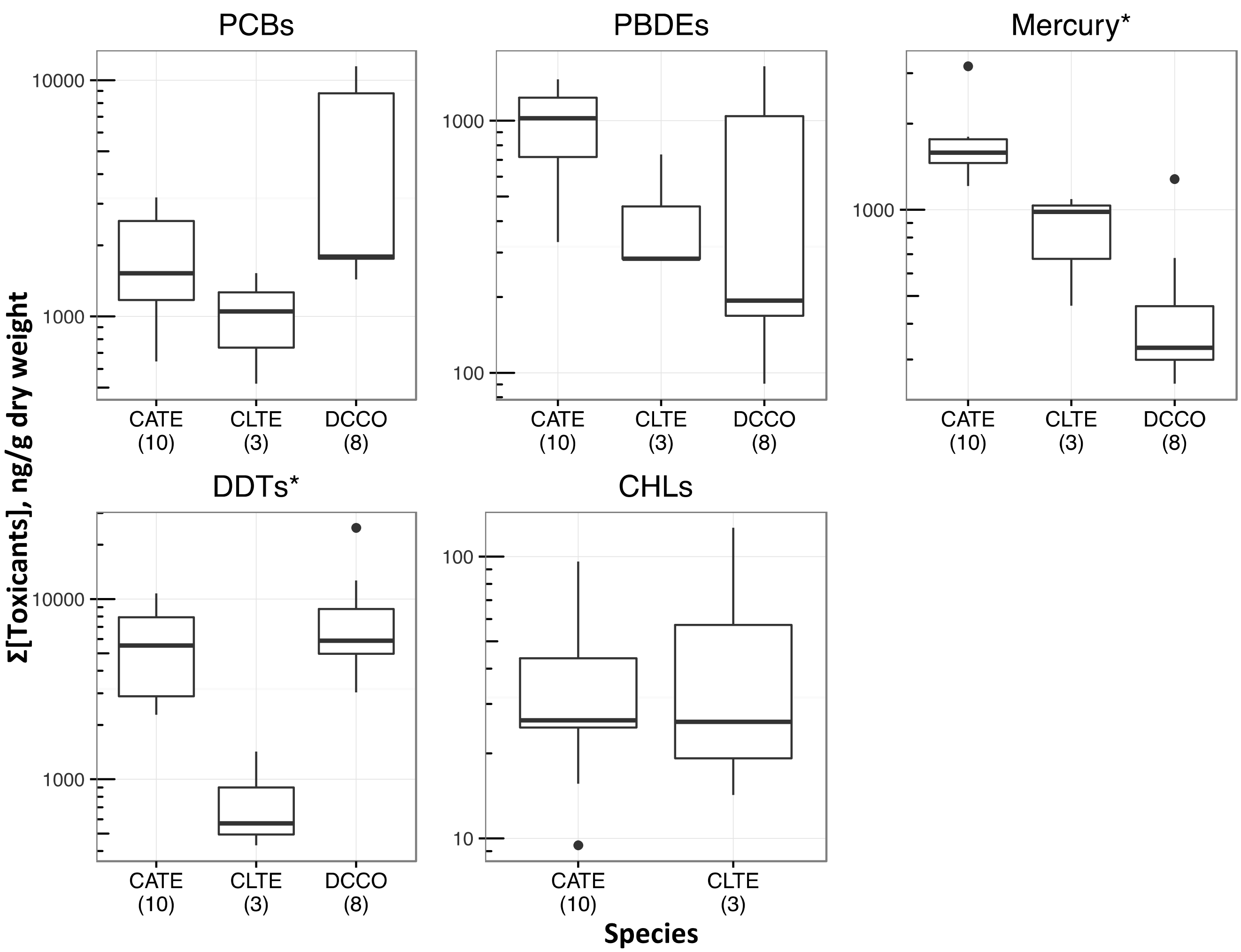


Figure 6. Toxicant concentrations in Salt Works by species. The concentrations of congeners within organic contaminant classes are summed by sample. Asterisks represent significant differences between species. Sample sizes are listed in parentheses below the 4-letter species ID for each toxicant class.



SEABIRDS AS INDICATORS OF LEGACY TOXICANT CONCENTRATIONS IN THE SOUTHERN CALIFORNIA BIGHT

Supplementary Material

Methods: Selecting screening levels

Contaminant levels measures in seabird eggs collected for this study were compared with levels associated with adverse effects in other studies of avian species. The amount of information on effect levels is variable depending on the contaminant and some may be field-based while others are laboratory-based. Effect levels may vary with species and effect, and often there are no data on effect levels for the species being studied. Consequently, contaminant levels reported for California least terns (CLTE), Caspian terns (CATE), double-crested cormorants (DCCO), and western gulls (WEGU) in this study were compared with estimated No Observed Adverse Effect Concentrations (NOAECs) and ranges of Lowest Observed Adverse Effect Concentrations (LOAECs) reported for sensitive adverse effects relating to maintenance of viable populations. Low ends of ranges were used for contaminants with multiple studies (and LOAECs) for an individual species, and as such are considered conservative estimates of "thresholds for observed adverse effects." Consistent with approaches used by the U.S. Environmental Protection Agency (e.g., USEPA 1995), and depending on the available data, LOAECs were adjusted downward using uncertainty factors between 3 and 10, to obtain an estimate of a NOAEC. The fact that the estimated NOAECs are for use to evaluate risks to waterbirds was considered in the selection of adjustment factors. The derived NOAECs are considered conservative estimates of concentrations below which no adverse effects are expected for waterbirds and as such may serve as screening levels for identifying contaminants of potential concern. Some values are based on data from very few studies and as such may change as more data become available.

1.1 Screening levels for PCBs

PCBs constitute a synthetic mixture of up to 209 congeners. Harris and Elliott (2011) identified ranges of total PCB concentrations in eggs that are associated reduced hatching and/or fledging success (8 species), reduced productivity (3 species), and reduced parental care (2 species). Thresholds used to evaluate total PCB levels in seabird eggs from this study were selected using the low ends of ranges identified by Harris and Elliott (2011) for reproductive effects in high, intermediate and low sensitivity species. The estimated NOAEC for use in this study of waterbirds is based on the lowest LOAEC for terns, gulls and raptors. Basing the estimated NOAEC on LOAECs for least sensitive species raises some concern about ensuring that potentially more sensitive species in that group are protected. Consequently, to be protective the LOAEC was adjusted downward by a factor of 3 for species differences in sensitivity, and downward by a factor of 3 again for LOAEC to NOAEC extrapolation, producing a final value that approaches an estimated NOAEC based on birds of intermediate sensitivity. Thresholds and screening levels used for total PCBs in this study on seabirds are:

NOAEC (estimated) - waterbirds (intermediate sensitivity) – 2600 ng/g fw

LOAEC - Reduced hatching/fledging or productivity - 23000 mg/kg fw

1.2 Screening levels for PBDEs
PBDEs are flame-retardant chemicals that were added to many types of consumer products to reduce potential for burning. Studies relating PBDE levels in avian eggs to adverse effects are very few in number, and the only readily available egg-based threshold is for reduced hatching success in American kestrel. Although no effect levels were reached, studies by McKernan et al. (2009) and Rattner et al. (2011) indicate that mallards and chickens are less sensitive than kestrels to PBDEs, and that the common tern, (and probably terns in general) are no more sensitive, and probably less sensitive than kestrels to PBDEs in eggs. The threshold used to evaluate total PBDE concentrations in seabird eggs for this study is a recommended threshold for reduced pipping and hatching success in American kestrel. The threshold based on American kestrel was adjusted downward by a factor of three for uncertainty about species differences in sensitivity and by a factor of three for the LOAEC to NOAEC extrapolation. The selected thresholds and screening levels for PBDEs are:
NOAEC (estimated) reduced hatching success – 200 ng/g fw
LOAEC - Reduced hatching success in a sensitive species – 1000 ng/g fw

1.3 Screening levels for DDTs
DDT is a legacy organochlorine pesticide that was manufactured and widely used between the early 1940s and 1972 for control of disease-carrying insects and insects on agricultural crops. Studies associating DDT concentrations in eggs with adverse effects have been conducted on numerous avian species (DOI 1998, Blus 2011). Sensitivity to DDT can be highly variable depending on the species and the effect. DOI (1998) and Blus (2011) provide ranges of thresholds for population-level effects and eggshell thinning with breakage in several avian species. The lowest threshold concentration for each species was used to rank species from most sensitive to least sensitive, and ranges were identified based on percentile rankings. The estimated NOAEC is based on the lowest threshold for sensitive species adjusted downward by a factor of three. Low-end thresholds used to evaluate DDT levels in seabird eggs, with a focus on reduced productivity and critical eggshell thinning (18%), are as follows:
NOAECs (estimated)-
eggshell thinning - 200 ng/g fw
reduced productivity - 1000 ng/g fw
LOAECS –
eggshell thinning in less-sensitive species - 10000 ng/g fw
reduced productivity in less-sensitive species - 5000 ng/g fw

1.4 Screening levels for mercury
Mercury naturally cycles in coastal and marine environments, but levels withing those cycles have increased due to anthropogenic activity. Effect levels for mercury in avian eggs have been most recently reviewed by Shore et al. (2011) and include data for multiple species. Values selected for use in this study of seabird eggs are a combination of general guidelines, as well as LOAECs and NOAECs for piscivorous avian species, specifically snowy egret (*Egretta thula*), common loon (*Gavia immer*), and common tern (*Sterna hirundo*) (as cited by Shore et al. 2011). Few adjustments to the LOAEC were deemed necessary, given the available data, particularly on piscivorous birds. The lowest LOAEC for fish eating birds (egret) was adjusted downward by

a factor of three for an estimated NOAEC. The thresholds and screening values used to assess mercury concentrations in seabird eggs for this study are:
NOAEC (estimated) - 500 ng/g fw
LOAECs in more-sensitive species - 800 ng/g fw (field based)

1.5 Screening levels for selenium
Selenium is an essential trace nutrient that supports beneficial metabolic functions, but it is also toxic to animals at exposure levels not much higher than those considered to be beneficial (DOI 1998). Effect levels for selenium in avian eggs have been the subject of several reviews including those by DOI (1998) and Ohlendorf and Heinz (2011). Values selected for use in this study of seabird eggs are ranges identified by DOI (1998) and Ohlendorf and Heinz (2011) as representative for species of varying sensitivities. The estimated NOAEC is based on the lowest threshold for sensitive species adjusted downward by a factor of three, producing a value comparable to background. Assuming an average moisture content of 70% for fresh seabird eggs (from Zeeman et al. 2008), the screening values used in this study for seabird eggs are as follows:
NOAEC – 900 ng/g fw
LOAEC – 3000 ng/g fw

1.6 Screening levels for arsenic
Arsenic is generally known more for its toxicity to mammals (including humans) than to birds). Although based on limited data, DOI (1998) was able to identify ranges for screening arsenic concentrations in avian eggs (as mg/kg dw). Assuming an average moisture content of 70% for fresh seabird eggs (from Zeeman et al. 2008), the screening values used in this study for seabird eggs are as follows:
NOAEC - 910 ng/g fw
LOAEC - >910 ng/g fw

Methods: Inter-laboratory Comparison

Prior to the measurement of field samples, an inter-laboratory comparison was performed to ensure the measurements between laboratories were comparable. The comparison utilized two reference materials: 1) NIST SRM 1946 Lake Superior Fish Tissue, a frozen fish tissue homogenate prepared from lake trout, and 2) frozen spiked chicken egg homogenate provided by the USFWS Analytical Control Facility. The spiked contaminant levels in the egg homogenate were set to mimic levels typically observed in other surveys (Tables X3 and X4). The exercise for metals required that laboratories obtain concentrations within 30% of the certified value (SRM 1946) or group mean (egg homogenate). The exercise for organics required measured concentrations within 40% of the certified or reference value (SRM 1946), or group mean (egg homogenate), for 70% of the target compounds with each class (organochlorine pesticides or PCBs). Both of these data quality objectives are consistent with laboratory comparability expectations for sediment (which all laboratories passed). Required reporting levels for both materials were: 20 ng arsenic/g ww, 30 ng mercury/g ww, 100 ng selenium/g ww, 0.5 ng organochlorine pesticide/g ww, and 2.5 ng PCB congener/g ww. Each material was run in triplicate by each participating laboratory.

Anonymized results are shown in tables S1 to S6, and are summarized as follows.

S7 SRM 1946 Metals

All six laboratories passed the criteria for the three metals.

S8 SRM 1946 Chlorinated Pesticides

All six laboratories passed the criteria for organochlorine pesticides.

S9 SRM 1946 PCBs

Five of six laboratories passed the criteria for PCBs.

S10 Egg Homogenate Metals

All six laboratories passed the criteria for mercury and selenium. Arsenic was problematic and required two rounds of analysis. In the first round, the arsenic results were variable, ranged from non-detect to 190 ng/g ww, and only one of six laboratories passed the criteria. The spiked concentration was 60 ng/g ww and the required reporting level was 20 ng/g ww. Differences in digestion procedures were a suspected reason. Instrumental interference was likely not a reason because all laboratories utilized ICP-MS with a collision cell. In the second round, four of six laboratories passed the criteria.

S11 Egg Homogenate Chlorinated Pesticides

All six laboratories passed the criteria for pesticides. One laboratory, while passing overall, was an outlier for 4,4'-DDT and DDMU, and these results were not included in the group mean.

S12 Egg Homogenate PCBs

The PCB exercise as a whole was rejected because a majority of laboratory values were non-detects, or did not otherwise correspond to spiked levels. Based on these results, it was not certain the material had been properly spiked.

Only laboratories passing the metal inter-calibration exercises performed measurements on the field samples. Due to uncertainty in the organics inter-calibration exercise, a single laboratory with prior experience analyzing eggs was selected to perform all organics measurements on the field samples.

Table S1. Analyte list, method detection level (MDL), and reporting level (RL) for egg samples. MDL and RL were converted from wet weight basis to ng/g dry weight assuming 75% moisture. % detect refers to the percentage of samples in which each analyte was detected.

PCBs	% detect	MDL	RL
PCB 018	0%	0.0125	0.025
PCB 028	33%	0.0125	0.025
PCB 037	23%	0.0125	0.025
PCB 044	11%	0.0125	0.025
PCB 049	41%	0.0125	0.025
PCB 052	84%	0.0125	0.025
PCB 066	92%	0.0125	0.025
PCB 070	71%	0.0125	0.025
PCB 074	85%	0.0125	0.025
PCB 077	56%	0.0125	0.025
PCB 081	3%	0.0125	0.025
PCB 087	72%	0.0125	0.025
PCB 099	98%	0.0125	0.025
PCB 101	84%	0.0125	0.025
PCB 105	89%	0.0125	0.025
PCB 110	96%	0.0125	0.025
PCB 114	26%	0.0125	0.025
PCB 118	99%	0.0125	0.025
PCB 119	0%	0.0125	0.025
PCB 123	12%	0.0125	0.025
PCB 126	6%	0.0125	0.025
PCB 128	87%	0.0125	0.025
PCB 138	100%	0.0125	0.025
PCB 149	87%	0.0125	0.025
PCB 151	54%	0.0125	0.025
PCB 153	100%	0.0125	0.025
PCB 156	72%	0.0125	0.025
PCB 157	31%	0.0125	0.025
PCB 158	97%	0.0125	0.025
PCB 167	65%	0.0125	0.025
PCB 168	86%	0.025	0.05
PCB 169	5%	0.0125	0.025
PCB 170	66%	0.0125	0.025
PCB 177	54%	0.0125	0.025
PCB 180	99%	0.0125	0.025
PCB 183	88%	0.0125	0.025
PCB 187	93%	0.0125	0.025
PCB 189	11%	0.0125	0.025
PCB 194	56%	0.0125	0.025
PCB 201	67%	0.0125	0.025
PCB 206	25%	0.0125	0.025

PBDEs	% detect	MDL	RL
PBDE 017	1%	0.0125	0.025
PBDE 028	74%	0.0125	0.025
PBDE 047	100%	0.0125	0.025
PBDE 049	68%	0.0125	0.025
PBDE 066	73%	0.0125	0.025
PBDE 071	6%	0.0125	0.025
PBDE 085	7%	0.0125	0.025
PBDE 099	95%	0.0125	0.025
PBDE 100	98%	0.0125	0.025
PBDE 138	9%	0.0125	0.025
PBDE 153	76%	0.0125	0.025
PBDE 154	75%	0.0125	0.025
PBDE 183	9%	0.0125	0.025
PBDE 190	2%	0.0125	0.025
PBDE 209	10%	0.0125	0.025

OCs	% detect	MDL	RL
Chlordane, cis-	51%	0.0125	0.025
Chlordane, trans-	6%	0.0125	0.025
DDD(o,p)	100%	0.0125	0.025
DDD(p,p)	24%	0.0125	0.025
DDE(o,p)	35%	0.0125	0.025
DDE(p,p)	100%	0.0125	0.025
DDMU(p,p)	88%	0.0125	0.025
DDT(o,p)	13%	0.0125	0.025
DDT(p,p)	19%	0.0125	0.025
Nonachlor, cis-	65%	0.0125	0.025
Nonachlor, trans-	80%	0.0125	0.025
Oxychlordane	1%	0.0125	0.025

Metals	% detect	MDL	RL
Mercury (CVAA)	100%	0.25	5
Mercury (EPA7473)	100%	2	2
Selenium	100%	0.25	12.5
Arsenic	100%	2.5	25

Table S2. Standard methods and instruments used to quantify each target class.

Target class	Method or instrument	Labs performed
PCBs	EPA8270Cm	PEL[a]
PBDEs	EPA8270Cm	PEL
OCs	EPA8270Cm	PEL
Mercury	EPA7473	CSD[b]
Mercury	CVAA[c]	LACSD[d]
Selenium	ICPMS[e]	LACSD
Arsenic	ICPMS	LACSD
Lipid	EPA160.3	PEL
Solids	SM2540D	PEL

[a]PEL = Physis Environmental Labs
[b]CSD = City of San Diego
[c]CVAA = Cold vapor atomic absorption
[d]LACSD = Los Angeles County Sanitation District
[e]ICPMS = Inductively coupled plasma mass spectroscopy

Table S3. Quality control data quality objectives. A batch was defined as not more than 20 samples. Metals includes mercury, arsenic, and selenium. For the organics, the accuracy was evaluated by individual contaminant (not the class sum). The reference material was either NIST SRM 1946: Lake Superior Fish Tissue, or a custom laboratory control material made from bird eggs. MDL = method detection limit and RPD = relative percent difference.

QC Material	Objective	Metal Criteria	Organics Criteria
Method Blank	Frequency	1/batch	1/batch
Method Blank	Accuracy	blank < 5 times MDL or blank < 5 times the minimum field concentration	blank < 10 times MDL
Spiked Blank	Frequency	1/batch	Not required
Spiked Blank	Accuracy	+/- 25% of spike value	NA
Reference Material	Frequency	1/batch	1/batch
Reference Material	Accuracy	+/- 20% of true value	+/- 30% of true value for 70% of compounds
Matrix Spike	Frequency	>= 10% of field samples	1/batch
Matrix Spike	Accuracy	+/- 25% of true value	+/- 50% of true value
Sample Duplicate	Frequency	>= 10% of field samples	1/batch
Sample Duplicate	Accuracy	< 25% RPD	< 25% RPD

Table S4. Data quality objective success rates for each contaminant class. Metals includes mercury, arsenic, and selenium. For the organics, the accuracy was evaluated by individual contaminant (not the class sum).

QC Material	Objective	PCB Success	OC Success	PBDE Success	Metal Success
Method Blank	Frequency	100%	100%	100%	100%
Method Blank	Accuracy	100%	100%	100%	85%
Spiked Blank	Frequency	NA	NA	NA	100%
Spiked Blank	Accuracy	NA	NA	NA	84%
Reference Material	Frequency	100%	100%	100%	100%
Reference Material	Accuracy	50%[1]	100%	92%	93%
Matrix Spike	Frequency	100%	100%	100%	100%
Matrix Spike	Accuracy	86%	91%	89%	86%
Sample Duplicate	Frequency	92%	92%	92%	100%
Sample Duplicate	Accuracy	82%	84%	75%[2]	94%

[1]Accuracy success was 100% if +/- 40% of the true value for 70% of the compounds, instead of +/- 30% of the true value for 70% of the compounds.

[2]Accuracy success was 82% if the RPD was < 30%, instead of < 25%.

Table S5. Summary statistics of analytes by species and site. Analyte units are ng/g dry weight.

Analyte	Species	Site	N	Mean	SD	Min	Max
PCBs	California least tern	Entire SCB	55	709.3	524.8	124	3041
PCBs	California least tern	Batiquitos	5	351.6	276.4	144	704
PCBs	California least tern	Bolsa Chica	8	575.4	204.6	390	932
PCBs	California least tern	Chula Vista	4	1143.4	289.6	892	1408
PCBs	California least tern	D-Street Fill	6	661.0	287.7	411	1136
PCBs	California least tern	LA Harbor	6	764.5	717.2	245	2175
PCBs	California least tern	Lindbergh Field	5	1233.0	1028.4	574	3041
PCBs	California least tern	Pt Mugu	9	589.4	458.7	185	1625
PCBs	California least tern	Salt Works	3	1030.7	503.0	519	1524
PCBs	California least tern	TJ Estuary	7	663.4	318.9	190	1118
PCBs	California least tern	Vandenberg	2	158.5	48.5	124	193
PCBs	Caspian tern	Entire SCB	15	3413.1	2851.3	645	9967
PCBs	Caspian tern	Bolsa Chica	5	6634.3	2696.2	3191	9967
PCBs	Caspian tern	Salt Works	10	1802.5	876.7	645	3192
PCBs	Double-crested cormorant	Salt Works	8	4794.7	596.8	1436	11448
PCBs	Western gull	Entire SCB	24	1235.5	1225.9	91	3863
PCBs	Western gull	Anacapa Isl	5	311.7	203.5	91	494
PCBs	Western gull	NAS North Isl	8	2165.2	847.9	1141	3340
PCBs	Western gull	Pismo Beach	1	362.7	NA	363	363
PCBs	Western gull	Santa Barbara Isl	9	1115.3	1438.2	176	3863
PCBs	Western gull	Vandenberg	1	370.8	NA	371	371
PBDEs	California least tern	Entire SCB	55	198.0	181.4	34	824
PBDEs	California least tern	Batiquitos	5	95.7	40.9	60	166
PBDEs	California least tern	Bolsa Chica	8	78.5	26.8	55	130
PBDEs	California least tern	Chula Vista	4	287.0	95.2	173	398
PBDEs	California least tern	D-Street Fill	6	180.2	66.1	108	268
PBDEs	California least tern	LA Harbor	6	151.4	43.2	75	206
PBDEs	California least tern	Lindbergh Field	5	183.1	42.9	140	241
PBDEs	California least tern	Pt Mugu	9	79.4	18.3	46	112
PBDEs	California least tern	Salt Works	3	433.9	261.2	282	736
PBDEs	California least tern	TJ Estuary	7	517.5	220.1	268	824
PBDEs	California least tern	Vandenberg	2	46.2	17.4	34	58
PBDEs	Caspian tern	Entire SCB	15	1130.9	540.6	330	2070
PBDEs	Caspian tern	Bolsa Chica	5	1506.6	644.9	433	2070
PBDEs	Caspian tern	Salt Works	10	943.0	390.1	330	1459
PBDEs	Double-crested cormorant	Salt Works	8	573.9	596.8	90	1645
PBDEs	Western gull	Entire SCB	24	676.0	742.5	71	3421
PBDEs	Western gull	Anacapa Isl	5	812.3	733.3	71	1772
PBDEs	Western gull	NAS North Isl	8	737.0	373.3	241	1200
PBDEs	Western gull	Pismo Beach	1	583.7	NA	584	584
PBDEs	Western gull	Santa Barbara Isl	9	615.2	1067.0	110	3421
PBDEs	Western gull	Vandenberg	1	145.3	NA	145	145

Table S5. Summary statistics of analytes by species and site. Analyte units are ng/g dry weight.

Analyte	Species	Site	N	Mean	SD	Min	Max
DDTs	California least tern	Entire SCB	55	1185.2	1224.8	238	6866
DDTs	California least tern	Batiquitos	5	396.5	137.7	259	563
DDTs	California least tern	Bolsa Chica	8	1537.8	640.0	758	2842
DDTs	California least tern	Chula Vista	4	530.5	53.7	457	572
DDTs	California least tern	D-Street Fill	6	344.7	103.5	238	528
DDTs	California least tern	LA Harbor	6	2187.0	1708.1	883	5423
DDTs	California least tern	Lindbergh Field	5	603.3	200.6	425	944
DDTs	California least tern	Pt Mugu	9	2392.1	1874.0	922	6866
DDTs	California least tern	Salt Works	3	808.1	538.5	430	1425
DDTs	California least tern	TJ Estuary	7	701.2	439.8	308	1548
DDTs	California least tern	Vandenberg	2	857.0	504.9	500	1214
DDTs	Caspian tern	Entire SCB	15	12149.7	12157.3	2282	42230
DDTs	Caspian tern	Bolsa Chica	5	25139.5	13409.4	9191	33039
DDTs	Caspian tern	Salt Works	10	5654.8	3063.0	2282	10739
DDTs	Double-crested cormorant	Salt Works	8	8683.0	7148.4	3036	24286
DDTs	Western gull	Entire SCB	24	2042.7	2122.7	447	9749
DDTs	Western gull	Anacapa Isl	5	1091.6	483.3	447	1619
DDTs	Western gull	NAS North Isl	8	1803.2	1062.3	591	4126
DDTs	Western gull	Pismo Beach	1	583.7	NA	584	584
DDTs	Western gull	Santa Barbara Isl	9	2943.2	3186.9	667	9749
DDTs	Western gull	Vandenberg	1	1673.8	NA	1674	1674
CHLs	California least tern	Entire SCB	55	30.7	26.4	0	126
CHLs	California least tern	Batiquitos	5	6.8	3.9	0	9
CHLs	California least tern	Bolsa Chica	8	31.8	20.2	19	80
CHLs	California least tern	Chula Vista	4	29.9	6.8	21	38
CHLs	California least tern	D-Street Fill	6	17.3	7.5	9	28
CHLs	California least tern	LA Harbor	6	36.5	26.9	13	82
CHLs	California least tern	Lindbergh Field	5	28.3	6.5	22	37
CHLs	California least tern	Pt Mugu	9	40.3	34.7	8	124
CHLs	California least tern	Salt Works	3	55.5	61.7	14	126
CHLs	California least tern	TJ Estuary	7	36.5	30.5	10	95
CHLs	California least tern	Vandenberg	2	16.7	10.8	9	24
CHLs	Caspian tern	Entire SCB	15	101.2	117.0	9	394
CHLs	Caspian tern	Bolsa Chica	5	230.7	122.5	58	394
CHLs	Caspian tern	Salt Works	10	36.5	25.2	9	96
CHLs	Double-crested cormorant	Salt Works	8	8.8	9.5	0	24
CHLs	Western gull	Entire SCB	24	8.0	13.0	0	58
CHLs	Western gull	Anacapa Isl	5	6.8	8.4	0	18
CHLs	Western gull	NAS North Isl	8	6.9	6.1	0	16
CHLs	Western gull	Pismo Beach	1	2.4	NA	2	2
CHLs	Western gull	Santa Barbara Isl	9	11.2	19.7	0	58
CHLs	Western gull	Vandenberg	1	0.0	NA	0	0

Table S5. Summary statistics of analytes by species and site. Analyte units are ng/g dry weight.

Analyte	Species	Site	N	Mean	SD	Min	Max
Mercury	California least tern	Entire SCB	55	949.4	205.7	463	1667
Mercury	California least tern	Batiquitos	4	759.0	88.6	650	856
Mercury	California least tern	Bolsa Chica	8	939.7	199.0	624	1271
Mercury	California least tern	Chula Vista	4	805.5	264.0	579	1180
Mercury	California least tern	D-Street Fill	5	948.2	118.5	776	1070
Mercury	California least tern	LA Harbor	6	1091.1	104.1	905	1203
Mercury	California least tern	Lindbergh Field	5	1056.0	115.5	890	1200
Mercury	California least tern	Pt Mugu	9	878.2	157.5	683	1195
Mercury	California least tern	Salt Works	3	845.7	335.6	463	1090
Mercury	California least tern	TJ Estuary	6	1010.5	78.6	877	1090
Mercury	California least tern	Vandenberg	2	1260.8	573.9	855	1667
Mercury	Caspian tern	Entire SCB	15	2183.8	1116.1	1210	4617
Mercury	Caspian tern	Bolsa Chica	5	3115.3	1439.0	1602	4617
Mercury	Caspian tern	Salt Works	10	1718.0	542.3	1210	3170
Mercury	Double-crested cormorant	Salt Works	8	482.4	349.3	247	1280
Mercury	Western gull	Entire SCB	24	276.6	213.3	67	1105
Mercury	Western gull	Anacapa Isl	5	116.7	36.4	67	149
Mercury	Western gull	NAS North Isl	8	255.3	147.3	86	512
Mercury	Western gull	Pismo Beach	1	329.6	NA	330	330
Mercury	Western gull	Santa Barbara Isl	9	286.5	92.3	170	455
Mercury	Western gull	Vandenberg	1	1104.8	NA	1105	1105
Selenium	California least tern	Entire SCB	29	2495.4	333.4	1883	3307
Selenium	California least tern	Batiquitos	4	2551.7	279.6	2199	2867
Selenium	California least tern	Bolsa Chica	8	2380.7	456.1	1883	3307
Selenium	California least tern	Chula Vista	0				
Selenium	California least tern	D-Street Fill	0				
Selenium	California least tern	LA Harbor	6	2605.0	185.7	2391	2868
Selenium	California least tern	Lindbergh Field	0				
Selenium	California least tern	Pt Mugu	9	2523.2	338.8	1983	3047
Selenium	California least tern	Salt Works	0				
Selenium	California least tern	TJ Estuary	0				
Selenium	California least tern	Vandenberg	2	2387.3	332.2	2152	2622
Selenium	Caspian tern	Entire SCB	5	3165.1	1980.4	1566	6500
Selenium	Caspian tern	Bolsa Chica	5	3165.1	1980.4	1566	6500
Selenium	Caspian tern	Salt Works	0				
Selenium	Double-crested cormorant	Salt Works	0				
Selenium	Western gull	Entire SCB	16	1751.9	198.7	1480	2160
Selenium	Western gull	Anacapa Isl	5	1677.9	139.2	1483	1842
Selenium	Western gull	NAS North Isl	0				
Selenium	Western gull	Pismo Beach	1	1689.1	NA	1689	1689
Selenium	Western gull	Santa Barbara Isl	9	1830.2	210.5	1508	2160
Selenium	Western gull	Vandenberg	1	1479.8	NA	1480	1480

Table S6. Summary statistics of analytes by species and site. Analyte units are ng/g fresh weight.

Analyte	Species	Site	N	Mean	SD	Min	Max
Arsenic	California least tern	Entire SCB	29	97.8	25.0	55	147
Arsenic	California least tern	Batiquitos	4	128.7	19.4	107	147
Arsenic	California least tern	Bolsa Chica	8	85.4	17.5	62	113
Arsenic	California least tern	Chula Vista	0				
Arsenic	California least tern	D-Street Fill	0				
Arsenic	California least tern	LA Harbor	6	114.2	21.7	80	142
Arsenic	California least tern	Lindbergh Field	0				
Arsenic	California least tern	Pt Mugu	9	84.5	18.9	55	107
Arsenic	California least tern	Salt Works	0				
Arsenic	California least tern	TJ Estuary	0				
Arsenic	California least tern	Vandenberg	2	95.5	25.6	77	114
Arsenic	Caspian tern	Entire SCB	5	82.7	25.1	45	105
Arsenic	Caspian tern	Bolsa Chica	5	82.7	25.1	45	105
Arsenic	Caspian tern	Salt Works	0				
Arsenic	Double-crested cormorant	Salt Works	0				
Arsenic	Western gull	Entire SCB	15	43.6	33.4	14	150
Arsenic	Western gull	Anacapa Isl	4	61.7	60.1	14	150
Arsenic	Western gull	NAS North Isl	0				
Arsenic	Western gull	Pismo Beach	1	33.7	NA	34	34
Arsenic	Western gull	Santa Barbara Isl	9	36.9	19.1	20	82
Arsenic	Western gull	Vandenberg	1	41.0	NA	41	41

Table S6. Summary statistics of analytes by species and site. Analyte units are ng/g fresh weight.

Analyte	Species	Site	N	Mean	SD	Min	Max
PCBs	California least tern	Entire SCB	55	236.7	127.7	105	877
PCBs	California least tern	Batiquitos	5	149.6	49.1	115	223
PCBs	California least tern	Bolsa Chica	8	205.1	47.0	152	294
PCBs	California least tern	Chula Vista	4	346.1	90.1	260	431
PCBs	California least tern	D-Street Fill	6	241.0	99.2	150	408
PCBs	California least tern	LA Harbor	6	237.2	130.3	137	482
PCBs	California least tern	Lindbergh Field	5	384.4	276.7	232	877
PCBs	California least tern	Pt Mugu	9	205.4	101.8	120	458
PCBs	California least tern	Salt Works	3	296.0	80.9	205	361
PCBs	California least tern	TJ Estuary	7	214.4	77.6	123	351
PCBs	California least tern	Vandenberg	2	109.2	5.3	105	113
PCBs	Caspian tern	Entire SCB	15	954.4	597.8	231	2368
PCBs	Caspian tern	Bolsa Chica	5	1592.1	514.1	948	2368
PCBs	Caspian tern	Salt Works	10	635.5	315.4	231	1276
PCBs	Double-crested cormorant	Salt Works	8	926.8	837.1	276	2358
PCBs	Western gull	Entire SCB	23	443.9	368.2	102	1252
PCBs	Western gull	Anacapa Isl	4	155.8	63.1	102	227
PCBs	Western gull	NAS North Isl	8	680.7	273.6	370	1078
PCBs	Western gull	Pismo Beach	1	181.5	NA	181	181
PCBs	Western gull	Santa Barbara Isl	9	420.8	435.5	137	1252
PCBs	Western gull	Vandenberg	1	171.6	NA	172	172
PBDEs	California least tern	Entire SCB	55	37.5	31.4	7	166
PBDEs	California least tern	Batiquitos	5	18.8	6.9	13	30
PBDEs	California least tern	Bolsa Chica	8	15.7	7.3	9	30
PBDEs	California least tern	Chula Vista	4	56.7	18.0	38	77
PBDEs	California least tern	D-Street Fill	6	36.5	16.0	21	67
PBDEs	California least tern	LA Harbor	6	28.9	8.2	16	38
PBDEs	California least tern	Lindbergh Field	5	38.8	10.5	30	52
PBDEs	California least tern	Pt Mugu	9	15.2	2.3	11	20
PBDEs	California least tern	Salt Works	3	76.1	22.4	61	102
PBDEs	California least tern	TJ Estuary	7	92.4	39.6	45	166
PBDEs	California least tern	Vandenberg	2	8.4	1.9	7	10
PBDEs	Caspian tern	Entire SCB	15	261.2	112.9	83	414
PBDEs	Caspian tern	Bolsa Chica	5	296.0	121.9	101	407
PBDEs	Caspian tern	Salt Works	10	243.8	110.5	83	414
PBDEs	Double-crested cormorant	Salt Works	8	88.9	99.1	9	280
PBDEs	Western gull	Entire SCB	23	171.1	210.3	17	981
PBDEs	Western gull	Anacapa Isl	4	193.3	208.4	17	435
PBDEs	Western gull	NAS North Isl	8	176.2	96.2	53	303
PBDEs	Western gull	Pismo Beach	1	146.2	NA	146	146
PBDEs	Western gull	Santa Barbara Isl	9	174.9	307.4	29	981
PBDEs	Western gull	Vandenberg	1	32.3	NA	32	32

Table S6. Summary statistics of analytes by species and site. Analyte units are ng/g fresh weight.

Analyte	Species	Site	N	Mean	SD	Min	Max
DDTs	California least tern	Entire SCB	55	225.7	228.2	47	1382
DDTs	California least tern	Batiquitos	5	76.5	15.8	61	101
DDTs	California least tern	Bolsa Chica	8	302.0	145.6	164	570
DDTs	California least tern	Chula Vista	4	106.2	18.5	88	125
DDTs	California least tern	D-Street Fill	6	70.8	30.9	47	132
DDTs	California least tern	LA Harbor	6	406.1	272.0	190	867
DDTs	California least tern	Lindbergh Field	5	129.3	50.9	78	213
DDTs	California least tern	Pt Mugu	9	453.0	363.0	190	1382
DDTs	California least tern	Salt Works	3	140.6	53.0	92	197
DDTs	California least tern	TJ Estuary	7	125.3	85.8	66	311
DDTs	California least tern	Vandenberg	2	153.6	70.2	104	203
DDTs	Caspian tern	Entire SCB	15	2594.9	2135.7	517	8303
DDTs	Caspian tern	Bolsa Chica	5	4799.5	2264.2	2335	8303
DDTs	Caspian tern	Salt Works	10	1492.6	875.5	517	2793
DDTs	Double-crested cormorant	Salt Works	8	1279.4	1098.0	294	3647
DDTs	Western gull	Entire SCB	23	527.0	566.3	118	2538
DDTs	Western gull	Anacapa Isl	4	243.7	113.9	118	369
DDTs	Western gull	NAS North Isl	8	431.1	272.6	130	1013
DDTs	Western gull	Pismo Beach	1	245.5	NA	246	246
DDTs	Western gull	Santa Barbara Isl	9	786.7	820.9	198	2538
DDTs	Western gull	Vandenberg	1	371.9	NA	372	372
CHLs	California least tern	Entire SCB	55	5.8	4.5	0	25
CHLs	California least tern	Batiquitos	5	1.4	0.8	0	2
CHLs	California least tern	Bolsa Chica	8	6.3	4.2	3	16
CHLs	California least tern	Chula Vista	4	6.0	1.8	4	8
CHLs	California least tern	D-Street Fill	6	3.5	1.9	2	7
CHLs	California least tern	LA Harbor	6	6.9	4.6	2	13
CHLs	California least tern	Lindbergh Field	5	6.0	1.6	5	8
CHLs	California least tern	Pt Mugu	9	7.6	6.8	2	25
CHLs	California least tern	Salt Works	3	8.9	7.6	3	18
CHLs	California least tern	TJ Estuary	7	6.2	4.5	2	13
CHLs	California least tern	Vandenberg	2	3.0	1.5	2	4
CHLs	Caspian tern	Entire SCB	15	21.3	21.2	2	65
CHLs	Caspian tern	Bolsa Chica	5	44.8	20.9	14	65
CHLs	Caspian tern	Salt Works	10	9.5	6.6	2	25
CHLs	Double-crested cormorant	Salt Works	8	1.4	1.5	0	3
CHLs	Western gull	Entire SCB	23	2.2	3.8	0	17
CHLs	Western gull	Anacapa Isl	4	2.0	2.4	0	5
CHLs	Western gull	NAS North Isl	8	1.7	1.5	0	4
CHLs	Western gull	Pismo Beach	1	0.6	NA	1	1
CHLs	Western gull	Santa Barbara Isl	9	3.2	5.7	0	17
CHLs	Western gull	Vandenberg	1	0.0	NA	0	0

Table S6. Summary statistics of analytes by species and site. Analyte units are ng/g fresh weight.

Analyte	Species	Site	N	Mean	SD	Min	Max
Mercury	California least tern	Entire SCB	52	185.5	43.1	108	279
Mercury	California least tern	Batiquitos	4	165.2	33.1	117	187
Mercury	California least tern	Bolsa Chica	8	180.7	39.4	125	238
Mercury	California least tern	Chula Vista	4	158.1	40.0	112	209
Mercury	California least tern	D-Street Fill	5	182.8	32.9	130	206
Mercury	California least tern	LA Harbor	6	211.7	41.8	160	255
Mercury	California least tern	Lindbergh Field	5	224.0	39.2	176	270
Mercury	California least tern	Pt Mugu	9	172.6	40.7	115	246
Mercury	California least tern	Salt Works	3	156.5	51.9	108	211
Mercury	California least tern	TJ Estuary	6	187.2	41.5	130	229
Mercury	California least tern	Vandenberg	2	228.4	71.6	178	279
Mercury	Caspian tern	Entire SCB	15	504.3	221.2	295	1020
Mercury	Caspian tern	Bolsa Chica	5	610.8	237.3	295	877
Mercury	Caspian tern	Salt Works	10	451.0	204.1	326	1020
Mercury	Double-crested cormorant	Salt Works	8	71.1	56.0	31	192
Mercury	Western gull	Entire SCB	23	71.2	50.2	16	246
Mercury	Western gull	Anacapa Isl	4	27.7	9.3	16	35
Mercury	Western gull	NAS North Isl	8	61.2	37.1	18	126
Mercury	Western gull	Pismo Beach	1	82.5	NA	83	83
Mercury	Western gull	Santa Barbara Isl	9	78.9	27.7	50	141
Mercury	Western gull	Vandenberg	1	245.5	NA	246	246
Selenium	California least tern	Entire SCB	29	487.1	63.1	356	624
Selenium	California least tern	Batiquitos	4	549.2	67.2	474	624
Selenium	California least tern	Bolsa Chica	8	453.7	56.8	356	536
Selenium	California least tern	Chula Vista	0				
Selenium	California least tern	D-Street Fill	0				
Selenium	California least tern	LA Harbor	6	501.0	62.4	426	590
Selenium	California least tern	Lindbergh Field	0				
Selenium	California least tern	Pt Mugu	9	489.6	57.4	386	545
Selenium	California least tern	Salt Works	0				
Selenium	California least tern	TJ Estuary	0				
Selenium	California least tern	Vandenberg	2	443.3	6.1	439	448
Selenium	Caspian tern	Entire SCB	5	591.5	244.5	398	989
Selenium	Caspian tern	Bolsa Chica	5	591.5	244.5	398	989
Selenium	Caspian tern	Salt Works	0				
Selenium	Double-crested cormorant	Salt Works	0				
Selenium	Western gull	Entire SCB	15	470.5	79.5	329	636
Selenium	Western gull	Anacapa Isl	4	438.7	16.4	421	454
Selenium	Western gull	NAS North Isl	0				
Selenium	Western gull	Pismo Beach	1	423.0	NA	423	423
Selenium	Western gull	Santa Barbara Isl	9	505.7	79.2	403	636
Selenium	Western gull	Vandenberg	1	328.8	NA	329	329

Table S6. Summary statistics of analytes by species and site. Analyte units are ng/g fresh weight.

Analyte	Species	Site	N	Mean	SD	Min	Max
Arsenic	California least tern	Entire SCB	29	97.8	25.0	55	147
Arsenic	California least tern	Batiquitos	4	128.7	19.4	107	147
Arsenic	California least tern	Bolsa Chica	8	85.4	17.5	62	113
Arsenic	California least tern	Chula Vista	0				
Arsenic	California least tern	D-Street Fill	0				
Arsenic	California least tern	LA Harbor	6	114.2	21.7	80	142
Arsenic	California least tern	Lindbergh Field	0				
Arsenic	California least tern	Pt Mugu	9	84.5	18.9	55	107
Arsenic	California least tern	Salt Works	0				
Arsenic	California least tern	TJ Estuary	0				
Arsenic	California least tern	Vandenberg	2	95.5	25.6	77	114
Arsenic	Caspian tern	Entire SCB	5	82.7	25.1	45	105
Arsenic	Caspian tern	Bolsa Chica	5	82.7	25.1	45	105
Arsenic	Caspian tern	Salt Works	0				
Arsenic	Double-crested cormorant	Salt Works	0				
Arsenic	Western gull	Entire SCB	15	43.6	33.4	14	150
Arsenic	Western gull	Anacapa Isl	4	61.7	60.1	14	150
Arsenic	Western gull	NAS North Isl	0				
Arsenic	Western gull	Pismo Beach	1	33.7	NA	34	34
Arsenic	Western gull	Santa Barbara Isl	9	36.9	19.1	20	82
Arsenic	Western gull	Vandenberg	1	41.0	NA	41	41

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Table S7. Metal inter-calibration results for reference material SRM 1946 Fish Tissue. All units are in ng/g ww.

Analyte	Required RL	Target Value	(+) 30% of Target	(-) 30% of Target	Lab 1			Lab 2			Lab 3			Lab 4			Lab 5			Lab 6		
					Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	Run 1	Run 2	Run 3
Arsenic	20	277	360	194	269	280	288	512	536	513	340	344	334	337	336	364	279	303	288	290	280	280
Mercury	30	433	563	303	320	319	323	554	507	522	393	337	359	431	497	466	514	489	505	400	380	370
Selenium	100	491	638	344	408	365	355	430	430	520	601	607	576	670	658	723	521	508	515	530	490	500

Table S8. Chlorinated pesticides inter-calibration results for reference material SRM 1946 Fish Tissue. All units are in ng/g ww. ND = non-detect and empty = not reported.

Analyte	Target Value	(+) 40% of Target	(-) 40% of Target	Lab 1			Lab 2			Lab 3		
				Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	Run 1	Run 2	Run 3
4,4'-DDT	37.2	52.08	22.32	27.6	30.8	31.2	26.0	28.0	27.0	30.4	33.4	29.5
2,4'-DDT	22.3	31.22	13.38	11.3	12.3	12.5	14.0	19.0	13.0	18.6	21.5	15.7
4,4'-DDD	17.7	24.78	10.62	5.7	6.2	6.2	9.8	10.0	10.0	23.6	24.9	19.2
2,4'-DDD	2.20	3.08	1.32	0.4	0.5	0.5	1.2	1.3	1.2	ND	ND	ND
4,4'-DDE	373	522.2	223.8	354.9	342.1	350.8	310	300	300	399	492	460
2,4'-DDE	1.04	1.456	0.624	0.7	0.7	0.7	1.1	1.3	1.2	1.47	ND	1.11
4,4'-DDMU				1.5	1.6	1.7	3.7	4.3	3.9	3.05	4.11	4.21
alpha-Chlordane	32.5	45.5	19.5	31.9	35.0	37.7	27.0	27.0	22.0	32.5	31.2	31.3
gamma-Chlordane	8.36	11.704	5.016	5.2	5.7	6.4	6.8	8.3	5.9	4.38	8.34	10.3
cis-nonachlor	59.1	82.74	35.46	38.8	43.7	46.8	50.0	54.0	48.0	52.2	59.2	47.2
trans-nonachlor	99.6	139.44	59.76	92.6	90.4	97.5	75.0	70.0	75.0	120	114	112
oxychlordane	18.9	26.46	11.34	11.7	12.7	14.3	24.0	20.0	19.0	24.5	23.7	20.5
dieldrin	32.5	45.5	19.5	25.6	32.0	24.4	30.0	32.0	26.0	28.1	26.4	29
Total Passing				**9**	**9**	**9**	**10**	**10**	**9**	**9**	**9**	**11**

Table S9. PCB inter-calibration results for reference material SRM 1946 Fish Tissue. All units are in ng/g ww. ND = non-detect and empty = not reported.

Analyte	Target Value	(+) 40% of Target	(-) 40% of Target	Lab 1			Lab 2			Lab 3			Lab 4			Lab 5			Lab 6		
PCB-18	0.84	1.176	0.504	ND	ND	ND				ND	ND	ND	0.415	0.391	0.353	0.35	0.35	0.29			
PCB-28	2.00	2.8	1.2	4.9	5.3	5.4	1.8	1.3	1.7	1.16	1.36	1.44	1.17	1.14	1.28	1.44	1.57	1.31	1.7	1.9	2
PCB-37										ND	ND	ND	ND	ND	0.0126	ND	ND	ND			
PCB-44	4.66	6.524	2.796	3.0	3.5	3.6	4.0	3.7	3.7	4.3	4.11	3.94	2.7	2.7	3.51	3.2	3.49	3.04	6.8	7.5	6.9
PCB-49	3.80	5.32	2.28	2.4	2.6	2.7	2.4	2.0	2.5	4	3.46	3.58	2.47	2.37	3.09	2.5	2.84	2.32	3	3.6	3.5
PCB-52	8.1	11.34	4.86	6.1	6.5	6.6	6.6	7.6	6.9	9.7	9.17	9.25	4.76	4.7	6.21	6.27	7.06	5.86	4.4	4.8	4.3
PCB-66	10.8	15.12	6.48	7.4	8.3	8.5	10.7	10.1	10.7	10.8	10.9	10.5	6.05	5.63	7.66	7.67	8.73	6.96	0	0	0
PCB-70	14.9	20.86	8.94	10.9	12.1	12.6	9.9	11.0	11.1	12.6	12.3	13.1	9.7	9.11	12.2	10.5	12.1	9.92	8.7	10	10
PCB-74	4.83	6.762	2.898	4.0	4.5	4.6	4.1	4.3	4.8	4.4	4.69	4.91	3.13	2.93	3.93	4.18	4.56	3.67	5.4	6.4	6.5
PCB-77	0.327	0.4578	0.1962	0.5	0.5	0.5	3.5	2.8	3.8	ND	ND	ND	0.201	0.202	0.257	0.35	0.41	0.45	6.6	7.8	6.5
PCB-81										ND	ND	ND	0.0669	0.0557	0.0613	ND	ND	ND			
PCB-82										ND	ND	ND	0.162	0.0798	0.145	ND	ND	ND			
PCB-87	9.4	13.16	5.64	9.0	9.6	9.7	8.9	7.6	7.6	9.6	9.86	8.45	9.15	9.02	8.3	8	8.63	7.69	8.4	11	9.8
PCB-92										ND	ND	ND	10.3	9.92	9.32	9.88	9.75	7.93			
PCB-99	25.6	35.84	15.36	21.6	23.6	23.6	28.3	24.8	24.6	26.9	24.8	27.3	27.7	26.7	25.1	25.6	25.4	20.9	25	29	30
PCB-101	34.6	48.44	20.76	35.9	36.9	37.9	32.5	33.7	34.1	35	36.2	33.6	39.5	37.7	35.6	37.7	38.1	34.1	30	36	31
PCB-105	19.9	27.86	11.94	20.1	22.1	22.2	18.7	19.3	19.4	17	16.9	18	17.3	16.8	18.2	19.6	20	16.9	14	18	17
PCB-110	22.8	31.92	13.68	20.4	22.0	22.3	23.8	22.0	23.5	22.7	19.5	20.4	26.4	25.7	24	23.2	23.9	20.3	20	22	23
PCB-114				ND	ND	ND				ND	ND	ND	1.16	1.12	1.23	ND	ND	ND	14	18	17
PCB-118	52.1	72.94	31.26	56.0	55.2	56.6	54.1	54.6	56.0	48.4	52.6	50.3	55.4	53.4	52.1	56.2	55.2	47.3	0	0	0
PCB-119										ND	ND	ND	1.22	1.15	1.11	ND	ND	ND			
PCB-123										ND	ND	ND	1.23	1.13	1.14	8.77	8.34	7.01			
PCB-126	0.380	0.532	0.228	0.7	0.8	0.7				ND	ND	ND	0.341	0.331	0.344	0.39	0.42	0.38	2.5	2.6	2.6
PCB-128	22.8	31.92	13.68	20.9	22.1	23.0	24.3	23.3	24.8	23.5	24.6	26.1	16.9	16.2	17.1	22.7	21	20.2	7.9	11	9
PCB-138	115	161	69	116.4	126.1	125.3	135.0	132.0	135.0	125	128	119	126	122	128	114	116	107	99	130	110
PCB-146	30.1	42.14	18.06	12.1	13.3	13.7				22.4	24.1	20.9	23.7	22.8	23.7	31.5	31.4	29.1	16	21	21

Analyte	Target Value	(+) 40% of Target	(-) 40% of Target	Lab 1			Lab 2			Lab 3			Lab 4			Lab 5			Lab 6		
PCB-149				21.4	23.8	24.1	28.7	27.1	26.9	24.5	26.6	28.3	32	31.2	27.5	25.6	28.4	25.4	19	21	22
PCB-151				6.5	7.1	7.3	8.5	8.8	8.3	5.33	6.61	4.94	10.5	10.3	9.11	8.71	8.96	8.05			
PCB-153	170	238	102	193.5	210.2	202.2	176.5	169.6	169.2	162	179	168	163	154	163	193	184	167	90	110	97
PCB-156	9.52	13.328	5.712	8.4	9.1	9.2	9.4	9.4	9.5	8.31	7.25	9.16	7.52	7.27	7.68	8.67	8.24	7.71	4.3	6.6	5.9
PCB-157				2.2	2.4	2.5	3.7	2.8	3.6	4.1	3.5	3.4	2.13	2.1	2.19	2.62	2.35	2.44			
PCB-158	7.66	10.724	4.596	6.0	6.9	6.6				4.98	6.22	5.57	6.73	6.36	6.76	6.99	7	6.29	89	110	110
PCB-167							6.9	7.2	6.0	ND	ND	ND	4.56	4.51	4.65	5.21	4.9	4.81	7.9	11	9
PCB-168										ND	ND	ND	0.184	0.163	0.217						
PCB-169	0.106	0.1484	0.0636	ND	ND	ND	5.4	5.5	3.9	ND	ND	ND	0.0848	0.1	0.0701	ND	ND	ND			
PCB-170	25.2	35.28	15.12	31.7	33.7	35.2	30.7	33.5	31.5	24.3	22.6	27.3	26.4	25.5	24.6	24.1	25.2	21.8	16	17	14
PCB-177				13.5	14.6	14.9	13.2	12.5	12.1	14.2	10.6	12.2	14	13.7	13.4	12.5	13.3	12			
PCB-180	74.4	104.16	44.64	79.1	84.7	85.0	75.4	72.2	74.3	76.4	73.2	71.8	78.1	76.1	74.9	71.4	75.7	71.4	10	13	12
PCB-183	21.9	30.66	13.14				21.3	21.4	20.7	19.5	21.3	18.5	23.9	23.2	21.8	18.7	20.8	17.6	11	12	12
PCB-187	55.2	77.28	33.12	57.0	62.0	59.2	55.7	53.7	54.5	53.3	55.5	50.6	67.8	66	62.3	54.2	53.8	50.7	30	42	37
PCB-189				1.6	1.6	1.7	2.5	2.3	2.9	ND	ND	ND	1.42	1.38	1.34	ND	ND	ND			
PCB-194	13.0	18.2	7.8	13.2	14.2	14.1	13.5	12.8	12.2	13.7	12.4	14.1	13.7	13.1	13.1	13.9	15	13.5	4.6	6.4	5.5
PCB-201	2.83	3.962	1.698	2.0	2.2	2.1	3.6	3.1	2.7	2.55	2.89	3.1	3.2	3.1	2.89	16.4	19.8	16.5	4.1	4.9	5.5
PCB-206	5.40	7.56	3.24	5.1	5.6	5.7	7.1	7.5	5.9	6.4	6.33	6.74	4.79	4.67	4.77	5.96	6.13	5.72	0.13	0.6	0.55
Total Passing				**23**	**23**	**23**	**24**	**23**	**24**	**25**	**26**	**26**	**25**	**25**	**29**	**27**	**27**	**27**	**10**	**16**	**14**

Table S10. Metal inter-calibration results for the spiked egg homogenate. All units are in ng/g ww. Arsenic results are from the second round of analyses.

Analyte	Required RL	Spike Level	Mean Value	(+) 30% of Target	(-) 30% of Target	Lab 1			Lab 2			Lab 3			Lab 4			Lab 5			Lab 6		
						Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	Run 1	Run 2	Run 3
Arsenic	20	60	76	98	53	104.6	86.7	92.7	69	68	65	61	56	58	88.819	103.23	105.455	69.3	70.1	64.3	62	75.3	59.8
Mercury	30	100	96	125	67	88	97	92	103	104	NA	92	90	91	99.6	98.8	99.1	118	121	114	75	77	80
Selenium	100	500	715	930	501	563	531	513	646	640	NA	836	828	877	779	833	849	772	829	830	640	620	570

Table S11. Chlorinated pesticide inter-calibration results for the spiked egg homogenate. All units are in ng/g ww. ND = non-detect and empty = not reported.

Analyte	Spike Level	Mean Value	(+) 40% of Target	(-) 40% of Target	Lab 1			Lab 2			Lab 3			Lab 4			Lab 5			Lab 6		
					Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	Run 1	Run 2	Run 3
4,4'-DDT	5	5.61	7.9	3.4	4.21	3.54		3.9	4.9	3.9	8.29	11.2	9.91	4	4.55	3.66	5.9	5.44	5.2	20	15	17
2,4'-DDT	5	7.86	11.0	4.7	7.83	6.51		9.7	10	8.9	5.83	8.15	7.14	7.64	8.11	6.92	9.75	9.22	9.27	7.7	5.3	5.6
4,4'-DDD	5	6.17	8.6	3.7	6.00	5.21		5.8	6.3	5.8	5.27	6.29	5.57	4.16	3.39	2.17	8.86	8.32	7.79	8.6	7.6	7.7
2,4'-DDD	5	6.19	8.7	3.7	3.17	2.38		5.5	6.2	6.1	5.3	6.87	5.22	5.98	6.94	5.8	8.41	8.39	8.93	7.5	6.3	6.3
4,4'-DDE	2100	1831.18	2564	1099	1984.00	1769.00		1700	1800	1800	1688	1987	1462	2400	2170	1970	2120	2350	2130	1500	1200	1100
2,4'-DDE	2	6.97	9.8	4.2	5.36	4.68		7.2	7.9	6.6	7.05	7.98	6.79	6.98	6.94	6.35	6.89	7.15	7.57	8.9	7.1	7.1
4,4'-DDMU	2	3.14	4.4	1.9	1.26	1.03		2.7	3.2	4.2	3.71	3.89	3.64	4.96	4.22	3.92	2.47	2.41	2.34	29	23	23
alpha-Chlordane	2	2.05	2.9	1.2	3.16	2.75		1.9	2.6	2.8	1.66	1.93	1.69	1.73	1.74	1.55	2.46	2.07	2.28	1.8	1.4	1.4
gamma-Chlordane	2	2.15	3.0	1.3	2.84	2.43		2.2	1.9	2.8	1.75	1.94	1.71	2.04	1.78	1.69	2.38	2.2	2.52	2.4	2	2
cis-nonachlor	20	13.22	18.5	7.9	11.59	9.38		12	13	13	8.49	13	10.4	15.2	15.4	13.3	15.2	15.1	17.7	16	13	13
trans-nonachlor	20	17.55	24.6	10.5	19.70	15.29		15	17	15	13.4	20	15	21.9	20	17.9	23	21.8	23.4	16	12	12
oxychlordane	20	21.62	30.3	13.0	20.74	18.06		15	16	19	25.2	29.8	25.4	23	20.8	19.5	24	26.1	27	22	18	18
dieldrin	20	21.03	29.4	12.6	20.96	17.76		18	19	18	14.2	20.1	15.1	19.7	19.9	18.2	31.8	31.4	34.4	21	19	19
Total Passing					**10**	**11**		**13**	**13**	**13**	**12**	**11**	**12**	**12**	**12**	**12**	**11**	**12**	**11**	**10**	**12**	**11**

Table S12. PCB inter-calibration results for the spiked egg homogenate. All units are in ng/g ww. ND = non-detect and empty = not reported.

Analyte	Spike Level	Mean Value	(+) 40% of Target	(-) 40% of Target	Lab 1			Lab 2			Lab 3			Lab 4			Lab 5			Lab 6		
					Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	Run 1	Run 2	Run 3	Run 1	Run 2	Run 3
PCB-18	none	5.88	8.2	3.5	6.48	5.69		5.6	5.2	5.1	4.38	6.4	5.42	7.78	6.99	6.87	5.69	5.45	5.3			
PCB-28	none	6.77	9.5	4.1	1.00	ND					ND	ND	ND	ND	ND	ND	ND	ND	ND	9.4	8	8.7
PCB-37	none	0.03	0.0	0.0							ND	ND	ND	0.0279	ND	0.0317	ND	ND	ND			
PCB-44	5	13.68	19.1	8.2	16.79	14.70		12.1	12.5	11.4	11	12	8.41	14.1	12.9	17.5	16.6	16	16.5	14	13	13
PCB-49	5		0.0	0.0	ND	ND					ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PCB-52	none	12.32	17.3	7.4	14.47	12.69		10.5	10.4	10.2	9.21	10.6	8.78	10.3	9.04	12.5	14.5	13.6	13.7	17	15	17
PCB-66	50	89.62	125.5	53.8	104.59	94.03		86.5	85.5	85.5	60.3	66.8	68.8	84.7	79	106	112	109	112	ND	ND	ND
PCB-70	5				ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PCB-74	50				ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PCB-77	5	42.02	58.8	25.2	ND	ND					ND	ND	ND	0.0329	0.0304	0.0381	ND	ND	ND	89	79	84
PCB-81	5										ND	ND	ND	ND	ND	ND	ND	ND	ND			
PCB-82	100		0.0	0.0							ND	ND	ND	ND	ND	ND	ND	ND	ND			
PCB-87	none	0.06	0.1	0.0	ND	ND		ND	ND	ND	ND	ND	ND	0.067	0.0616	0.0596	ND	ND	ND	ND	ND	ND
PCB-92	5		0.0	0.0							ND	ND	ND	ND	ND	ND	ND	ND	ND			
PCB-99	100	0.02	0.0	0.0	ND	ND		ND	ND	ND	ND	ND	ND	0.0255	0.0236	0.0186	ND	ND	ND	ND	ND	ND
PCB-101	100	0.29	0.4	0.2	ND	ND		ND	ND	ND	0.53	0.33	0.61	0.096	0.0874	0.0787	ND	ND	ND	ND	ND	ND
PCB-105	50				ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PCB-110	50				ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PCB-114	none		0.0	0.0	ND	ND					ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PCB-118	100	0.04	0.1	0.0	ND	ND		ND	ND	ND	ND	ND	ND	0.0377	0.0383	0.0349	ND	ND	ND	ND	ND	ND

Analyte	Spike Level	Mean Value	(+) 40% of Target	(-) 40% of Target	Lab 1		Lab 2			Lab 3			Lab 4			Lab 5			Lab 6		
PCB-119	none		0.0	0.0						ND	ND	ND	ND	ND	ND	ND	ND	ND			
PCB-123	none	0.10	0.1	0.1						ND	ND	ND	0.107	0.102	0.0966	ND	ND	ND			
PCB-126	5		0.0	0.0	ND	ND				ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PCB-128	100	1.03	1.4	0.6	1.02	1.04	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PCB-138	500	0.63	0.9	0.4	ND	ND	ND	ND	ND	1.17	1.31	1.23	0.0268	0.0281	0.024	ND	ND	ND	ND	ND	ND
PCB-146	100	0.09	0.1	0.1	ND	ND				ND	ND	ND	0.0985	0.0836	0.0939	ND	ND	ND	ND	ND	ND
PCB-149	50	0.06	0.1	0.0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.0626	ND	ND	ND	ND	ND	ND
PCB-151	none		0.0	0.0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND			
PCB-153	500	12.84	18.0	7.7	18.75	15.11	11.7	11.4	11	7.88	14.8	11.2	16.3	13.8	14.8	16.4	15.7	15.3	8.6	7.5	8
PCB-156	50		0.0	0.0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PCB-157	5	0.54	0.8	0.3	1.29	1.24	ND	ND	ND	ND	ND	ND	0.065	0.057	0.0582	ND	ND	ND			
PCB-158	none	0.03	0.0	0.0	ND	ND				ND	ND	ND	0.0365	0.034	0.0329	ND	ND	ND	ND	ND	ND
PCB-167	none		0.0	0.0			ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PCB-168	none	10.72	15.0	6.4						ND	ND	ND	ND	ND	ND	16.4	15.7	15.3	6.6	4.5	5.8
PCB-169	5		0.0	0.0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND			
PCB-170	50	12.38	17.3	7.4	18.23	14.76	10.6	10.1	9.7	10.2	8.85	7.87	20.1	16.9	16.7	14.4	14.3	13.3	9.5	7.9	7.1
PCB-177	5		0.0	0.0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND			
PCB-180	500	12.27	17.2	7.4	17.85	14.52	12.4	11.4	10.6	7.69	9.14	8.24	19.3	15.9	15.3	14.9	14.7	14.6	8.1	5.1	8.9
PCB-183	none	14.66	20.5	8.8			12	11.75	11.85	9.37	11.7	8.37	23.3	18.8	18.2	17.6	19.8	19.2	13	12	13
PCB-187	50	13.17	18.4	7.9	17.16	14.13	12.5	10.9	10.5	12.8	12.1	11.1	21.4	17.4	17.1	16.4	15.5	15.6	7.4	4.7	7.2
PCB-189	none		0.0	0.0	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND			
PCB-194	50	0.17	0.2	0.1	ND	ND	ND	ND	ND	ND	ND	ND	0.192	0.156	0.162	ND	ND	ND	ND	ND	ND

Analyte	Spike Level	Mean Value	(+) 40% of Target	(-) 40% of Target	Lab 1			Lab 2			Lab 3			Lab 4			Lab 5			Lab 6		
PCB-201	none		0.0	0.0	ND	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
PCB-206	5	11.59	16.2	7.0	18.93	16.00		12.4	10.2	9.7	5.86	6	6.6	17.7	13.5	14.2	17.3	18.8	20.5	4.3	2.4	2.6

Figure S1. Relationships between eggshell thickness and toxicant concentrations in California least tern.

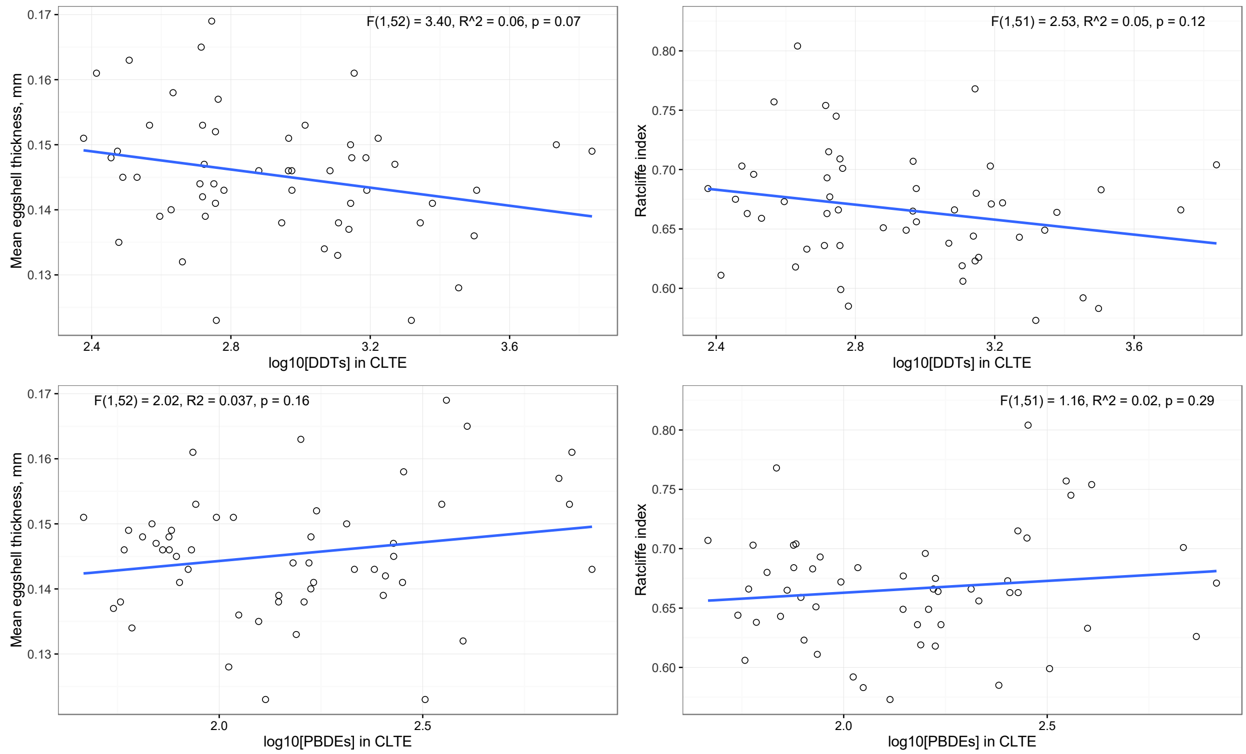

Figure S2. Relationships between eggshell thickness and toxicant concentrations in western gull.