

Appendix C: Summary of an ERA at a Hypothetical Sediment Site

This section evaluates ecological risk for a hypothetical case study within the study design framework defined in Step 4. In practice, only final data that have met specified DQOs, are applied to Step 4 decision rules to determine corresponding actions for various result scenarios. The worst-case scenario is assumed for the hypothetical case study, where site sediment chemical concentrations, toxicity and tissue bioaccumulation are negatively impacted compared to the reference site, indicating unacceptable “relative” risk. Absolute risk is then evaluated for bioaccumulation using toxicity reference values (TRVs) obtained from the literature for each assessment endpoint. The site is then re-examined on a station-by-station basis by augmenting the decision rules with the TRV results. This approach provides critical information on the spatial extent of at-risk areas for remedial design to risk managers and decision makers. PCBs and copper are examined as common Navy sediment COPCs in the following hypothetical study.

Hypothetical Site Setting

The hypothetical site consists of shoreline sediments located in close proximity to a historical Naval Air Station. COPCs enter the system through outfall pipes and road-runoff as land-based point- and non-point sources, respectively. The assessment endpoints are healthy communities of benthic infauna, as well as forage fish and piscivorous birds. The measurement endpoints are amphipod survival, polychaete survival and growth, and bioaccumulation of COPCs in field-collected fish tissue. These endpoints address both acute and chronic toxicity as well as bioaccumulation in the lower food web, which is subsequently modeled for higher-order receptors (i.e., birds).

Surface sediments were collected and tested for COPCs, 10-day acute toxicity using the amphipod *Rhepoxynius abronius*, 28-day survival and growth using the polychaete *Neanthes* sp., and 28-day in-situ bioaccumulation in forage fish (e.g., top smelt). Prior to sampling, the site was divided into three subareas based on existing sediment grain size data, geographic location and proximity to specific land-based contaminants. Resident forage fish were collected for each subsite and reference area by trawl. Ten fish per subsite were each analyzed for whole body COPC concentrations. For surface sediments, each subsite was sampled in a grid pattern that provided uniform sample density, resulting in a slightly unbalanced study design of 10, 12 and 14 sediment stations for the three subsites due to differences in area. Although stations were regularly spaced, the grid placement was made without *a priori* assumptions (at random) so that subsite stations could be treated as replicates in the statistical analyses. This sampling design also provides optimum coverage of the area with contiguous information for subsequent remedial design, if needed.

A total of 10 reference stations were selected throughout the same harbor that were previously shown to be unimpacted using agreed-upon Tier 1 screening criteria (in this case > 69.5%

amphipod survival and no COPC concentrations > NOAA ER-Ls). Reference stations also were selected for sediments that spanned the range of grain size expected for site sediments (based on field validation data collected in Step 5). More importantly, reference stations were selected based on a similar expected outcome of measurement endpoint test results as compared with the test site in the absence of effects from COPCs (e.g., no other confounding factors – sulfides, salinity).

Results and analysis of relative and absolute risk for chemical exceedances, toxicity and fish tissue bioaccumulation follow. Bioaccumulation was evaluated using field-collected fish that were local to the site and had a known exposure route to COPCs in site sediment. Alternatively, bioaccumulation can be estimated using either laboratory tests or modeling using either empirical or mechanistic approaches. Laboratory tests determine the extent of bioaccumulation by organisms, usually at lower trophic levels (e.g., clams, polychaetes, forage fish). This method accounts for bioavailability from sediment to organisms, but typically some form of modeling is needed to predict COPC concentrations in higher trophic levels (e.g., piscivorous fish, birds, mammals). Empirical modeling is commonly used in routine assessments due to its simplicity compared to mechanistic approaches, which rely on trophic transfer in the food web. Direct measure in field specimens is the preferred method (EPA 1996, 2000) as it produces less uncertainty in the final result.

Lines-of-Evidence Results Summary

In this example, mean copper concentration in sediment for one of the three subsites was 90 mg/kg (ppm) dry weight (2% TOC), and was significantly elevated ($p < 0.05$) compared to the corresponding reference site mean of 28 mg/kg, even when normalized to organic carbon. The reference site had a lower range of TOC (e.g., mean = 1.2%), so all COPC sediment concentrations were normalized to organic carbon (e.g., mg COPC/kg organic carbon) prior to inference testing (applying to decision rules).

Mean copper concentration in fish tissue was not elevated compared to reference tissue. Three other metals, chromium, zinc and lead also were significantly elevated in sediment but all concentrations were less than corresponding Tier 1 screening criteria (e.g., NOAA ER-Ls), and were therefore excluded from further analysis. Additionally, mean total PCB measured as the sum of 22 individual congeners were elevated in both sediment and fish tissue for the same subsite. Mean total PCB sediment concentration was 0.3 mg/kg dry weight, at 3% TOC; and the mean whole body PCB concentration was 0.29 mg/kg wet weight. The PCB mixture consisted of a variety of congeners, but was dominated by Aroclor 1254. In this example, the mean PCB concentration in resident fish (smelt) collected in the affected subsite was used as the concentration in prey. However, these data could have been obtained from the literature or modeled from site sediment data (EPA 2000). No other COPC exceedances were observed for

this or the remaining two subsites. The final COPC list was limited to copper and PCBs, and risk characterization was performed for the only subsite affected.

For the same subsite, mean amphipod survival of 57% was not statistically lower than the mean reference survival of 75%; however, it was lower than a guideline value of 69.5% that was used in other local regulatory programs. Polychaete survival also was not statistically different between test and reference areas; however, growth was significantly impaired at the test site (mean of 2.36 mg/day vs. 5.22 mg/day for reference).

Although commonly used, standard inference tests are often not applicable to percentile toxicity data (which range from 0-100%), since results are often not normally distributed and fail test assumptions, even after they are transformed. It is therefore appropriate to consider other methods to evaluate toxicity data, including non-parametric methods (e.g., rank-sum tests) or non-statistical methods, such as use of local guidelines.

Copper Toxicity

In this case study, the mean copper concentration of 60 mg/kg was above the Tier 1 screening criterion (34 mg/kg) and it was elevated compared to the reference mean, so it was retained as a COPC. If copper is one of many COPCs in sediment, toxicity test results must be interpreted as the effects of the overall mixture thereby the toxic effects of the copper must be estimated. There are several approaches to do the estimation including:

1. If a wide range of copper concentrations were measured at the site, the highest concentration that was not toxic can be used as an acceptable sediment copper concentration for the site. This is similar to the AET approach used for Puget Sound sediments (Barrick et al. 1988).
2. If concentrations are not wide-ranging (homogeneous), spiked sediment toxicity tests using COPCs may be useful, although there are many difficulties with implementing and interpreting spiked sediment toxicity tests.
3. Instead of laboratory bioaccumulation, critical body residues of resident fauna can be measured. In either case, results can be compared to critical concentrations for similar species in the ERED (<http://www.wes.army.mil/el/ered/>) database or from the scientific literature. Clams, which are filter and deposit feeders, are efficient bioaccumulators of copper, although other abundant or ecologically important organisms may be more appropriate. However, interpretation of the significance of tissue residues is problematic and should be interpreted with caution; they should not be used as the only approach for assessing risks from sediment copper.

If the concentration of copper in site sediments was much higher than the Tier 1 screening criterion, for example greater than 390 mg/kg, and toxicity testing indicated sediment toxicity, then it may be best to forego further studies and assume the site poses significant risk.

For this case study, copper concentrations were examined on a station-by-station basis. Where copper was both elevated compared to Tier 1 screening criteria and the reference mean, and significant impacts were observed for either amphipod survival or polychaete growth, the station was identified as having “unacceptable risk”. Bioaccumulation of copper was not significantly elevated in fish collected at the subsite, so potential effects from bioaccumulation to assessment endpoints were not evaluated. A discussion on the bioaccumulative effects of copper follows as a representative approach to evaluate significantly elevated site COPCs. Although copper is considered a bioaccumulative constituent by many agencies and guidance documents, it generally does not biomagnify in higher trophic levels (most vertebrates), such as methyl-mercury or PCBs. If there are adverse effects at a site, they usually occur to benthic biota such as bivalve mollusks, and occur relatively close to the contaminated site (there are fewer far-field effects). Copper is usually not a problem for birds and mammals, except at highly contaminated sites or unusual exposure scenarios.

Bioaccumulative Effects of Copper

Copper tissue residues in laboratory organisms (or resident fauna) can also be compared to the critical body residue (CBR) concentrations shown to cause toxic effects in similar organisms. A search of the ERED database shows a variety of CBR values for copper and other COPCs. The type of effect and tissue analyzed in the study should correspond with data from the literature. The species analyzed in literature studies are rarely the same as those encountered at the site, but the same criteria for selecting measurement endpoints for assessment endpoints can be used to select appropriate CBRs from the literature. However, habitat and feeding regime may not be as important for tissue residues, since tissue residues already reflect the effects of differences in bioavailability and uptake mechanisms. The tissue residues can be compared to CBRs from the literature to determine whether there are adverse effects from copper in tissues. However, since copper is a metal that can be sequestered in non-toxic forms in some species under certain exposure regimes, other supporting lines of evidence should also be used to establish the significance of tissue residues to adverse effects.

PCB Risk Characterization

PCBs are lipophilic, highly stable and degrade slowly in the environment, resulting in high concentrations in sediment and organisms even at low concentrations in water. PCBs also readily biomagnify from lower to higher trophic levels, making them some of the chemicals of greatest concern for bioaccumulative effects.

PCBs consist of a class of 209 congeners, and are usually found as complex mixtures of Aroclors in contaminated sediments. Information on bioaccumulation and toxicity exists for Aroclor mixtures of PCBs, but much of the data available are for individual congeners. Risk characterizations can be made for each congener when warranted; however, evaluation of total PCBs or Aroclor mixtures is sufficient in many cases.

For the case study, piscivorous birds, are the highest trophic level receptors potentially impacted by PCBs. Piscivorous birds are exposed to PCBs primarily from ingestion of contaminated prey – in this case, forage fish. Risk characterization for these birds requires three pieces of information: 1) PCB concentration in forage fish exposed to contaminated sediment; 2) an assessment endpoint representative of all piscivorous birds at the site; 3) a corresponding toxicity reference value (TRV) for the chosen endpoint. In this case, the Belted Kingfisher serves as a representative piscivorous bird with a corresponding TRV obtained from Sample et al. (1996).

Forage fish were collected at the contaminated subsite, and the mean whole body PCB concentration was 0.29 mg/kg wet weight. Although measuring field-collected specimens is the preferred method, estimates can also be made from literature values if field data are unavailable. The simplest approach is to use existing biota-sediment accumulation factors (BSAFs) from the literature to estimate PCB concentrations in fish from measured PCB concentrations in sediment. BSAFs are defined by EPA (1995) as the ratio of a lipid-normalized COPC concentration in tissue to its organic carbon-normalized concentration in sediment. For nonpolar organics, BSAFs are calculated as:

$$\text{BSAF} = (C_t/F_t) / (C_s/F_{oc})$$

Rearranging to calculate tissue concentration (C_t) from a BSAF gives:

$$C_t = (\text{BSAF} * (C_s/F_{oc})) * F_t$$

Where:

C_t is concentration in the organism (check basis for wet or dry weight);

F_t is the lipid fraction in tissue;

C_s is the COPC concentration in sediment (usually dry weight); and

F_{oc} is the organic carbon fraction in the sediment.

There are other approaches to modeling bioaccumulation to aquatic organisms, including both empirical and mechanistic models. The choice of methods and models will depend on the scope of the study, selected endpoints, and data available (both in the literature and from the site). A discussion of some of these models can be found in EPA (2000).

Determining Bioaccumulation Effects to Piscivorous Fish. The concentration of PCBs in fish measured from field-collected specimens can be compared directly to PCB tissue concentrations

shown to cause adverse effects in other studies. Appropriate species from the site should be designated as surrogate species, and compared to species from the literature that are closest in habitat, trophic status and taxonomic level. For highly mobile and wide-ranging species such as fish, PCB tissue concentrations may not reflect the worse case assumptions of all feeding within the contaminated site. However, measures from field-collected specimens will reflect the proportionate use of the site area for relatively immobile species. It is especially important to ensure proper study design and statistical analyses when using field-collected data.

A search of the ERED database yielded a number of NOEDs for PCB concentrations in whole bodies of several fish species and life stages. For salmonids, NOED levels for mortality range from 2.0 mg/kg (whole body) for rainbow trout to 30 mg/kg (whole body) for Atlantic salmon. Channel catfish (*Ictalurus punctatus*) NOEDs for mortality ranged as high as 14.3 mg/kg (whole body). For a comparison to low and high effect levels, in rainbow trout a 1.3 mg/kg PCB concentration in the whole body resulted in 10% mortality (Hogan and Brauhn, 1975), and for the Coho salmon, a whole body concentration of 645 mg/kg PCB resulted in 100% mortality. Data for fish that are forage species include NOED concentrations as high as 230 mg/kg (whole body) for the fathead minnow.

For this example, the 0.6 mg PCB/ kg body weight we measured in our forage fish was well below the adverse effects levels cited above. We conclude that there is no risk to fish from bioaccumulation of PCBs.

Determining Bioaccumulation Effects to Piscivorous Birds. Assessing bioaccumulation effects to piscivorous birds usually involves modeling PCB accumulation through the food chain via prey items, which are typically forage fish.

The measured concentration for forage fish was used as the estimated concentration of PCB in site forage fish. This can be compared to the TRV for the concentration of PCB in prey eaten by the Belted Kingfisher. The no observed adverse effects level (NOAEL) for food for the Belted Kingfisher is 0.355 mg of Aroclor 1254/ kg food (Sample et al. 1996). The concentration of PCB in tissues from forage fish at the site (0.29 mg PCB/kg body weight) is below the TRV, so we conclude that there is no risk to piscivorous birds from the bioaccumulation of PCBs.

If the assessment endpoint at the site is not the Belted Kingfisher or some other species with published TRVs (there may be several important species at the site which need to be considered as assessment endpoints), then TRV calculations can be performed using methods described in Sample et al. (1996). These usually include adjustments for body weight and food consumption rates for specific species. Alternatively, TRVs can be derived for contaminant dose per body weight per day (e.g., mg PCB/kg·day) and compared to ingestion rates based on allometric formulas (EPA 1996, 2000). TRVs based on ingestion rates can then be applied to other assessment endpoint species (e.g., other sea birds), once their body weight is known. The

following ingestion rate formula (EPA 2000) also includes adjustments for the relative bioavailability of a COPC (BF) and the proportion of the study area (site) relative to the home range of the assessment endpoint (EF) (i.e., site use factor). Bioavailability (BF) and site use (EF) are set at one (1) using the most conservative assumptions; however, these assumptions are usually unrealistic and increase model uncertainty. However, information on site use and COPC bioavailability are often difficult to obtain. Therefore, these factors should be adjusted only when the resulting TRV (in this case the IR_t) indicates unacceptable risk and the required information is available. The COPC ingestion rate for any species can be estimated from a generic food web model (EPA 2000) as:

$$IR_t = \text{SUM} [(C \cdot I \cdot BF)_i \cdot EF] / BW$$

Where:

IR_t = total rate of COPC ingestion (mg COPC/kg body weight-day) (in wet weight)

C_i = COPC concentration in each medium (e.g., food, water) as mg/kg (wet weight)

I = the rate of ingestion in each medium (e.g., mg food/day)

BF = the relative bioavailability factor of the chemical from each medium (unitless)

EF = the proportion of the study area relative to the home range of the species (unitless)

BW = body weight of assessment endpoint (kg)

Final Risk Characterization

Final risk characterization is based on results for the decision rules applied from Step 4 (relative risk) and absolute risk evaluated using - in this case - TRVs. Following the example used for copper above, all COPC results should be evaluated on a station-by-station basis. This allows identification of problem areas within the subsite and ranking of these areas relative to each other and the reference site. Evaluation by station also identifies any potential “hot-spots” that may have been previously concealed when the subsite was examined as a single area. Application of decision rules could be made by comparing each site station result to a 95th or 99th predictive interval based on grouped reference site data for each parameter (e.g., COPC concentration, toxicity result, COPC tissue concentration) (see Step 4 for a discussion of appropriate statistical methods).

For the hypothetical case, each station is examined to ensure that COPCs, toxicity and bioaccumulation (both relative to reference and literature-based TRVs, where either indicates significant impact). This information is then used to identify affected stations for the remedial design based on relative location (to each other and onshore site contaminants) and degree of impact.