# A DISCUSSION OF

# PCB TARGET LEVELS IN AQUATIC SEDIMENTS



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## Executive Summary

PCB contamination of freshwater and marine sediments is widespread, but the maximum allowable PCB sediment concentrations that would be protective of aquatic organisms have not been established. Published studies on the bioaccumulation and toxicological effects of PCBs provide guidance in determining sediment target levels that minimize adverse health effects to aquatic organisms. Complications in the analysis and interpretation of PCB data, factors affecting the distribution of PCBs in aquatic environments and their accumulation in the tissues of aquatic organisms, and background levels of PCBs must be considered in any use of sediment target levels as cleanup standards at a specific site.

PCBs are a group of 209 individual compounds (congeners) that differ in physicochemical properties, environmental behavior, and their tendency to be accumulated by aquatic organisms. Quantitative analysis of PCBs from environmental samples is difficult because environmental PCBs often differ considerably from the commercial Aroclor mixtures that are frequently used as analytical standards.

The degree of chlorination appears to be the most significant factor in determining the physical and chemical properties of PCBs. The more highly chlorinated PCBs have lower solubilities in water, lower vapor pressures, and higher octanol-water partition coefficients. These properties, along with their general resistance to degradation, make PCBs highly persistent in sediments.

Sediment characteristics (such as particle size and total organic carbon content), characteristics of the organism (such as body size, tissue lipid content, reproductive state), and the particular type of PCB are important factors in the uptake and accumulation of PCBs by aquatic organisms. The higher chlorinated PCBs are not metabolized to a significant extent by aquatic organisms and, as a result, tend to be the most persistent in their tissues.

Published data support the generalization that PCB concentrations in tissues of aquatic organisms will be greater than or equal to sediment concentrations. The large number of factors that affect the relationship between sediment PCB concentrations and tissue concentrations make more specific quantitative modeling of bioconcentration factors difficult. The toxicological literature had numerous examples of sublethal toxic effects on aquatic organisms at PCB tissue levels of less than 1 ppm and as low as 0.1 ppm. Based on these data, a tissue concentration of 0.1 ppm PCB or less would be protective of aquatic organisms. This would require a sediment target level of less than or equal to 0.1 ppm.

A review of four general approaches (EP, SLC, AET and sediment triad) to the development of sediment criteria showed that sediment target levels for PCBs ranged from 0.001 to 2.5 ppm, with most ranging between 0.01 and 1.0 ppm. These values are generally consistent with the 0.1 ppm PCB sediment target level based on bioaccumulation and toxicity data.

It is important to distinguish target levels, which are based solely on biological effects, from cleanup levels, which include consideration of other factors and correspond to "acceptable levels of impairment." Actual cleanup criteria would have to be decided on a site-by-site basis and would have to take into account the characteristics of the affected area and the specific resources at risk, as well as the potential environmental impacts of any proposed cleanup activities.

# PCB TARGET LEVELS IN AQUATIC SEDIMENTS

## Introduction

The polychlorinated biphenyls (PCBs) are environmentally persistent compounds that have a strong tendency to accumulate in aquatic sediments and tissues of aquatic organisms. An estimated 2 x $10^8$ kg of PCBs (35% of the total produced) still exists in mobile environmental reservoirs (Eisenreich 1987). Both acute and chronic toxic effects of PCBs to aquatic organisms are well-documented. Although PCB contamination of freshwater and marine sediments is widespread, criteria have yet to be established for safe levels of PCBs in sediments comparable to national water quality criteria. Decisions regarding appropriate cleanup levels at hazardous waste sites cannot be delayed until broadly applicable sediment criteria have been established. As a result, development of guideline target levels for PCB contamination in aquatic sediments that are protective of aquatic resources is an important priority for remedial action at hazardous waste sites.

The objectives of this report are twofold: First, to provide background information on PCBs that should be considered in an assessment of potential environmental risk, including aspects of the environmental chemistry and analysis of PCBs and important factors in their distribution in aquatic sediments and accumulation by aquatic organisms. The second (and primary) objective is to provide guideline target levels for PCB contamination in aquatic sediments that will be protective of aquatic resources. Preliminary sediment target levels are derived from a review of published information on the bioaccumulation and toxicological effects of PCBs on aquatic organisms and compared to the results from four recently developed approaches to the development of general sediment quality criteria. Although PCB contamination of aquatic sediments may have important implications for the health of consumers of aquatic organisms, including humans, we only address issues relating to potential injury to aquatic organisms.

## I. Background

One of the major difficulties in assessing the potential environmental risks from PCB contamination is that PCBs constitute a complex mixture of compounds with a wide range of physicochemical properties and consequent differences in environmental distribution, bioaccumulation and biological effects.

PCBs represent a group of 209 possible individual compounds (congeners) with from 1 to 10 chlorines on a biphenyl ring structure (Figure 1). Groups of PCBs with the same number of chlorines (and the same molecular weight) are referred to as homologs; each homolog group consists of isomers that differ in their patterns of chlorine substitution (Table 1) (Erickson 1985).

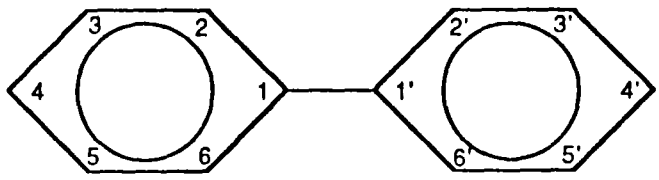


Figure 1. Structure of the biphenyl molecule (from EPA 1980).

Commercial PCB products are complex mixtures of congeners and are often described by the percent of chlorine; for example, Aroclor 1254 is a commercial formulation with 54% chlorine by weight that is composed primarily of isomers of tetra-, penta- and hexachlorobiphenyl (Eisler 1986).

The quantitative analysis of PCBs is complicated by differences between the mixtures of individual PCB congeners present in environmental samples and the commercial mixtures (Safe 1984). Selecting appropriate standards and matching peaks or patterns of chromatograms from environmental samples with those from commercial standards can be difficult and leads to inconsistencies in the results (Safe 1984; Phillips 1986; Stout 1986). Results of PCB analysis are often reported in terms of a particular reference standard (e.g., Aroclor 1254) or as total PCBs. The selection of a reference standard can make a large difference in the result (Cairns et al. 1986). As a consequence of the difficulties inherent in PCB analysis, inter-laboratory comparisons frequently demonstrate considerable variability (Eisler 1986; Stout 1986).

While the separate analysis of individual congeners would provide the most quantitatively accurate results (Safe 1984), the associated additional costs and the problem of availability of synthetic standards for each congener make it unlikely that this approach could find widespread application to waste site investigations. The quantification of PCBs by the degree of chlorination possibly would be a more immediately attainable goal in PCB analysis (Alford-Stevens 1986). The more detailed information provided by this approach would be useful in evaluating and monitoring waste sites for PCB contamination without significantly increasing the cost of the analysis. (This is currently the standard protocol for the NOAA National Mussel Watch program).

Environmental PCB mixtures may be further complicated by the potential presence of other toxic contaminants. Small amounts of highly toxic polychlorinated dibenzofurans (PCDFs) have been found as impurities in some commercial PCB mixtures (EPA 1980). In addition, PCBs may be transformed to PCDFs by heat in industrial use, sometimes in substantial amounts (Eisler 1986). It also has been suggested that PCBs could be converted to PCDFs by photochemical processes or metallic salt formations in the environment (EPA 1980).

## Factors Affecting PCB Distribution in the Environment

The distribution of PCBs in the environment is influenced by their physical and chemical properties. As a result of their highly lipophilic nature and low water solubility, PCBs are generally found to have low concentrations in water and relatively high concentrations in sediments. Individual PCB congeners have different physical properties based on both the degree of chlorination and position of chlorine substitution, although differences with degree of chlorination are usually more significant (Phillips 1986). Vapor pressure and water solubility are inversely related to degree of chlorination (Erickson 1985; Phillips 1986). Octanol-water partition coefficients, which are often used as estimators of the potential for bioconcentration, are highest for PCB congeners with the highest degree of chlorination. Solubilities and octanol-water partition coefficients range over several orders of magnitude (Table 1). Due to their higher water solubility, lower chlorinated PCBs may show a greater dispersion from a point source, while the higher chlorinated components of a PCB mixture may remain in the sediments closer to the source (Phillips 1986).

Table 1. Summary of Chemical Characteristics of PCBs and Some Commercial PCB Mixtures.

| PCB | No. of Isomers[1] | Percent Chlorine[1] | Solubility[2] (ppb) | log $K_{ow}$[3] |
|---|---|---|---|---|
| Biphenyl | 1 | 0 | 7500 | 3.89 |
| Monochlorobiphenyl | 3 | 18.6 | 1190-5900 | 4.38-4.48 |
| Dichlorobiphenyl | 12 | 31.5 | 80-1500 | 4.9-5.3 |
| Trichlorobiphenyl | 24 | 41.0 | 78-85 | 5.6-5.8 |
| Tetrachlorobiphenyl | 42 | 48.3 | 34-180 | |
| Pentachlorobiphenyl | 46 | 54.0 | 22-31 | 6.5 |
| Hexachlorobiphenyl | 42 | 58.7 | 0.95 | 6.8-6.9 |
| Heptachlorobiphenyl | 24 | 62.5 | | |
| Octachlorobiphenyl | 12 | 65.7 | | 7.1 |
| Nonachlorobiphenyl | 3 | 68.5 | | |
| Decachlorobiphenyl | 1 | 79.9 | | 8.2 |
| Commercial Mixtures | | | | |
| Aroclor 1221 | | 21 | 3500 | |
| Aroclor 1232 | | 32 | 1450 | |
| Aroclor 1016 | | 41 | 332 | |
| Aroclor 1242 | | 42 | 288 | |
| Aroclor 1248 | | 48 | 54 | |
| Aroclor 1254 | | 54 | 42 | |
| Aroclor 1260 | | 60 | 2.7 | |

[1]from EPA 1980
[2]from Chou and Griffin 1986 (range of values reported; not representative of all isomers)
[3]from Doucette and Andren 1987 (range of values reported; not representative of all isomers)

The mobility of PCBs in sediment is also a function of the chlorine substitution pattern and degree of chlorination and is generally quite low, particularly for the higher chlorinated biphenyls (Fisher et al. 1983). As a result of this low mobility, in the absence of disturbance of the sediment or bioaccumulation, even low rates of sedimentation may prevent PCBs in the sediment from reaching the overlying water via diffusion (Fisher et al. 1983). The measurement of PCB levels in sediment is complicated by their non-uniform distribution on both vertical and horizontal scales. Sedimentation rate and the depth of sediment sampled are both important factors in the determination of PCB concentrations in sediment (Pavlou and Dexter 1979; Phillips 1986). Different types of sediment sampling equipment may differ in the amount of recently deposited surface sediment retained, resulting in significant differences in concentration values (Phillips 1986).

PCB concentrations are also affected by physical characteristics of the sediment such as grain size, and total organic carbon content (Pavlou and Dexter 1979; Lynch and Johnson 1982). Fine sediments generally contain higher concentrations of PCBs than coarser sediments, probably as a result of a larger surface area (Phillips 1986). The amount of PCBs sorbed to sediments is also a function of the total organic carbon content of the sediment (Chou and Griffin 1986; Sawhney 1986). Variability in sediment PCB concentrations within segments of the Hudson River appeared to be related to both particle size and total organic content (Brown et al. 1985).

The persistence of PCBs in the environment is a result of their general resistance to degradation. The rate of degradation of PCB congeners by bacteria decreased with increasing degree of chlorination (Furukawa 1986); other structural characteristics of the individual PCBs also affected susceptibility to microbial degradation to a lesser extent. Photochemical degradation, via reductive dechlorination, is also known to occur in aquatic environments; the higher chlorinated PCBs appear to be most susceptible to this process (Sawhney 1986). In addition, reductive dechlorination of the higher chlorinated PCB congeners by anaerobic bacteria in sediments has been reported (Brown et al. 1987). The overall significance of the different degradation pathways in terms of reduction of environmental PCB levels is not clear at the present time.

## Factors Affecting the Accumulation of PCBs by Aquatic Organisms

The availability of PCBs in sediments to aquatic organisms depends on the concentrations of the specific PCBs present, physical properties of the sediment, environmental factors and characteristics of the organisms. The multitude of factors that affect the distribution and availability of PCBs in contaminated sediment and their accumulation by aquatic organisms make it extremely difficult to develop models to predict the relationship between sediment concentrations and concentrations in organisms *a priori*. In addition, the data needed to characterize all of the necessary factors are often not available, and their collection may easily exceed the costs of a direct evaluation.

PCB concentrations (lipid basis) in a polychaete and a fish species were directly correlated with sediment concentrations in a laboratory study (Shaw and Connell 1982). Uptake of specific PCB components by the organisms was a function of the product of the octanol-water partition coefficient and an empirically derived coefficient based on the pattern of chlorine substitution. Maximum uptake was observed for penta- and hexachlorobiphenyls (the primary components of Aroclor 1254); less chlorinated PCBs had smaller partition coefficients and the higher chlorinated PCBs had less favorable stereochemistry.

The elimination of PCBs from organisms is also related to the characteristics of the specific PCB congeners present. In two field studies, uptake and depuration in mussels was relatively rapid for lower chlorinated PCBs but much slower for higher chlorinated congeners (Pruell et al. 1986; Tanabe et al. 1987). In the Hudson River (New York), declines in PCB concentrations in some fish species over several years were primarily due to reduction in the lower chlorinated components (Brown et al. 1985). Differences in bioaccumulation among individual PCB congeners was reported to be the result of differences in elimination rates for juvenile soles, presumably due to differences in biotransformation metabolism (Boon and Duinker 1985).

Physical properties of the sediment, such as organic carbon content and sediment grain size, directly affect the availability and bioaccumulation of PCBs (Pavlou and Dexter 1979; Neff 1984). For example, uptake of PCBs from sediments by polychaetes was affected by the organic carbon content of the sediment; polychaetes in sediment with a low organic carbon content had a bioaccumulation factor (ratio of the concentration in tissue to the concentration in sediment) of 1.59 compared to 0.15 for high organic sediment (Rubenstein et al. 1983). Chironomid larvae from low organic sediment had a higher bioaccumulation factor than larvae from organic-rich sediment (Larsson 1984).

Connor (1984), with modifications by Breck (1985), derived an equation to predict fish/sediment concentration ratios for hydrophobic contaminants from the organic-carbon-normalized octanol-water partition coefficient. Assuming an organic carbon content of 1%,

a fish/sediment ratio of 45 is estimated from the log octanol-water partition coefficient for Aroclor 1254 (Breck 1985). For chemicals that are accumulated via the food web, the octanol-water partition coefficients may underestimate the fish/sediment ratio (Breck 1985). Fish/sediment concentration ratios were also observed to be a function of the flushing time (hydraulic retention time) of the water basin based on data from other studies (Connor 1984).

The presence of other contaminants may also influence the bioaccumulation of PCBs. For example, high concentrations of petroleum hydrocarbons may reduce the uptake of PCBs in polychaetes (Shaw and Connell 1982).

Seasonal variations in PCB accumulation by aquatic organisms have been commonly observed and may be attributed to changes in PCB availability, changes in water quality (temperature, salinity, hydrological conditions), or biological changes in the organism (Phillips 1986). In a Swedish study involving experimental ponds with PCB contaminated sediment, Larsson (1985, 1986) reported that PCB concentrations in the water varied seasonally, with higher levels in the summer and lower levels in the winter. PCB accumulation in zooplankton and fish reflected the seasonal variation in the water concentrations.

Characteristics of the organisms, including species differences, lipid content, age, rate of growth, sex and reproductive condition, may also affect bioaccumulation of PCBs. Since PCBs primarily partition into lipid compartments, differences in PCB concentration between species and between different tissues within the same species may reflect differences in lipid content (Phillips 1986). Sloan and Armstrong (1981) observed that PCB concentrations in several species of fish in the Hudson River were strongly correlated to their lipid content. In addition, accumulation of PCBs in migrant (non-feeding) fish species was related to body size (ratio of surface area to volume), regardless of species.

Reduction in PCB body burdens may take place in females of some species during egg production and spawning (Lech and Peterson 1983; Stout 1986). PCB concentrations also may be reduced through growth of the organism as a result of dilution, while the overall body burden increases or remains unchanged (Hutzinger et al. 1974; Lech and Peterson 1983).

Biotransformation of PCBs in fish and other aquatic organisms is generally slower than in mammals, and, in addition, fish appear to be less able to metabolize (and thus, excrete) the higher chlorinated PCB congeners (Lech and Peterson 1983). Consequently, fish and other aquatic organisms may accumulate a higher percentage of the higher chlorinated PCB congeners than is found in the environment.

## II. Sediment Target Levels

In this paper sediment "target levels" represent levels of PCB contamination corresponding to a threshold level of impact—that is, a level above which toxic effects have been reported. It is important to distinguish target levels, which are based solely on biological effects, from cleanup levels, which include consideration of other factors and correspond to "acceptable levels of impairment."

## Bioaccumulation and Biological Effects

One approach to determining levels of PCB contamination in aquatic sediments that would be protective of aquatic organisms involves consideration of the relationship between the concentrations of PCBs in sediment and aquatic organisms and the evidence for toxic biological effects at different tissue levels, based on published data.

Bioaccumulation of PCBs from contaminated sediments may result from uptake of PCBs from water (via respiration), direct dermal sorption or indirectly through the food web (Swartz and Lee 1980; Rubenstein et al. 1984). Diet appears to be the major pathway for PCB contamination in a number of fish species (Jensen 1982; Rubenstein et al. 1984; Thomann and Connolly 1984). Regardless of the route of exposure, PCBs in the sediment provide a significant source of contamination for aquatic organisms (Larsson 1985, 1986). Aquatic organisms living in association with PCB-contaminated sediments generally have levels equal to or greater than sediment levels (O'Connor et al. 1983; Neff 1984).

The relationship between PCB concentrations in sediments and the resulting levels in resident biota has been investigated in a number of studies (Table 2). A large scale survey of the southeast Atlantic and Gulf of Mexico coasts reported fish/sediment concentration ratios ranging from 5 to 20 for total PCBs in Atlantic croaker (Micropogonias undulatus) liver tissue (NMFS 1987). In a major field study in the Upper Mississippi River, sediment PCB concentrations of about 0.5 ppm resulted in invertebrate concentrations similar to the sediment levels. Bluegill and carp had PCB concentrations of about 0.5-1.0 ppm and 1.0-2.0 ppm, respectively (Mauck 1987). Oligochaetes in the Niagara River (New York) had PCB concentrations about 3 times higher than sediment concentrations (Fox et al. 1983). In two field studies in Puget Sound, organism/sediment ratios ranged from 0.4-7.1 in English sole (Parophyrs velotus) (Tetra Tech 1983) and from 0.6-13 in clams and 3.5-150 in shrimp (Malins et al. 1980).

Uptake by plants growing in PCB-contaminated soil does not appear to be of major importance (Pal et al. 1980; Fries and Marrow 1981). However, Mrozek et al. (1982) reported selective uptake of lower chlorinated biphenyls by salt marsh cordgrasss (Spartina alterniflora) from sediments treated with Aroclor 1254; the higher chlorinated components were not accumulated.

Sediment bioaccumulation factors (concentration in tissue/concentration in sediment) show considerable variation, but the majority of results from both field and laboratory studies support the generalization that PCB concentrations in resident organisms will equal and, in some cases, greatly exceed concentrations in the sediment. Additional information on the relationship between PCB concentrations in the sediments and in the tissues of aquatic organisms may eventually lead to predictive models. It is evident, however, that a bioaccumulation factor of 1 is more likely to underestimate than to overestimate the concentration in resident organisms, particularly organisms in higher trophic levels.

Table 2. Summary of Bioaccumulation Factors (Ratio of the Concentration in the Organism to the Concentration in Sediments) from PCB-Contaminated Sediments.

| Organism | Location | Bioaccum. Factor Mean (Range) | Reference |
|---|---|---|---|
| Marine | | | |
| Clam | Puget Sound | 5.9 (0.6-13) | Malins et al. 1980 |
| Shrimp | Puget Sound | 37 (3.5-150) | Tetra Tech 1983 |
| English sole | Puget Sound | 3.6 (0.4-7.1) | |
| Atlantic croaker | Atlantic and Gulf Coasts | (5-20) | NMFS 1987 |
| Shrimp | Lab | (1.9-3.5) | McLeese et al. |
| Polychaete | Lab | (3.8-10.8) | 1980 |
| Mussel | Lab | 0.4 | Rogerson et al. 1985 |
| Fiddler crab | Lab | (0.2-1.1) | Clark et al. 1986 |
| Polychaete | Lab | 0.15 (high organic)<br>1.6 (low organic) | Rubenstein et al. 1983 |
| Freshwater | | | |
| Oligochaete | Niagara River | about 3 | Fox et al. 1983 |
| Chironomid | Artificial ponds | 4.2 | Larsson 1984 |
| Larvae | Field | 2.9 | |
| Prawn | Lab | 1.1 | Tatem 1982 |
| Clam | Lab | 2.4 | |
| Perch | Lab | 2.7(1.5-3.9) | Seelye et al. 1982 |

While acute toxicity of PCBs appears to be relatively low (EPA 1980), a number of field and laboratory studies provide evidence of chronic sublethal effects on aquatic organisms at low tissue concentrations (Table 3). Results from chronic toxicity tests indicate that PCB toxicity is directly related to the duration of exposure (EPA 1980). Of particular note are the number of marine and freshwater fish species that experienced chronic toxicity at PCB tissue levels of less than 1.0 ppm and as low as 0.1 ppm. For example, a field study in the Baltic Sea demonstrated reduction in the viable hatch of Baltic flounder eggs at PCB concentrations greater than 0.12 ppm (Von Westernhagen et al. 1981). In another field study, Spies et al. (1985) reported an inverse relationship between PCB concentrations in starry flounder eggs in San Francisco Bay and reproductive success. Results suggested that concentrations of PCBs in the ovaries of less than 0.2 ppm could affect reproductive success. Monod (1985) also reported a significant correlation between PCB concentrations in eggs (on lipid weight basis) and total egg mortality in Lake Geneva char (at PCB concentrations ranging from 10-78 ppm lipid weight, 0.1-0.5 ppm wet weight). However,

it should be noted that, for all of the above field studies, the possible presence of other contaminants complicates the interpretation of results.

Table 3. Chronic Biological Effects Associated With PCB Concentrations

| Organism/organ | PCB Concentration | Effect | Reference |
|---|---|---|---|
| Striped bass/ gonads | 1.4 ppm FW, 2.3 ppm lipid | Reproductive failure (inference) | Ray et al. 1984 |
| Atlantic salmon/ eggs | 1.9-6.5 ppm lipid Aroclor 1254 | No effects | Zitko and Saunders 1979 |
| Baltic flounder/ ovaries | >0.12 ppm FW | Reproductive failure | Von Westernhagen et al. 1981 |
| Cyprinid minnows/ gonads | >24 ppm FW | Reproductive failure | Bengtsson 1980 |
| Rainbow trout/ whole body | 0.4 ppm FW Aroclor 1242 | Eggs with low survival | EPA 1980 |
| Rainbow trout/ eggs | 0.33 ppm FW Aroclor 1254 | Deformities and prehatch mortalities | Niimi 1983 |
| Atlantic salmon/ eggs, fry | 0.6-1.9 ppm FW (14.4-34 ppm lipid) | 46-100% mortality | Niimi 1983 |
| Starry flounder/ ovaries | 0.2 ppm FW | Reduced reproductive success; MFO induction | Spies et al. 1985 |
| Char/ eggs | 10-78 ppm lipid (0.1-0.5 ppm FW) | Egg mortality | Monod 1985 |

FW = wet weight

PCBs have also been shown to cause induction of the mixed function oxygenase (MFO) system in aquatic organisms. Induction of the MFO system, consisting of a group of microsomal enzymes important in the metabolism of xenobiotics, may result in an increased sensitivity to the toxic effects of other contaminants that undergo metabolic activation (Binder and Lech 1984). Induction of the system may also directly affect normal physiological functions such as reproduction and molting (Lee 1981; Payne et al. 1987).

Both field and laboratory studies have demonstrated MFO induction by PCBs at tissue levels within the range of environmental exposures. Addison et al. (1981) reported a dose-response relationship between PCB concentrations and MFO induction at low levels of PCB in brook trout muscle tissue (from concentrations less than 1 ppm). Contamination of lake trout gametes with PCB concentrations 1 ppm above background levels resulted in the induction of MFO enzymes in offspring (Binder and Lech 1984). Low tissue levels of PCBs induced increased MFO activity in Fundulus (1.4 ppm median effective dose)

(Binder et al. 1985). Spies et al (1984,1985) reported an inverse relationship between MFO induction and reproductive success in starry flounder.

An exhaustive literature search on the bioaccumulation and toxicological effects of PCBs was not undertaken in this review. Based on results from several studies indicating chronic toxic effects in aquatic organisms with PCB residues in tissue between 0.1 and 0.5 ppm and tissue levels that usually equal or exceed sediment levels, a sediment concentration of 0.1 ppm or less would be reasonably protective of aquatic resources . Higher sediment concentrations would most likely result in tissue levels in aquatic organisms that have been associated with chronic toxic effects.

## General Approaches to Sediment Quality Target Levels

Regulatory agencies have taken several different approaches to the problem of establishing target levels for sediment contaminants (including PCBs) that would be the equivalent of the national water quality criteria. The following section reviews four of the recent approaches and compares the values obtained for PCBs. None of these approaches has yet received widespread acceptance or application; hence, the values are still considered preliminary.

EPA TOXIC Pro-1

The sediment-water equilibrium partitioning model (EP), that was developed for the U.S. EPA Criteria and Standards division, uses a theoretical approach based on established water quality criteria (JRB 1984; Quinlan et al. 1985). Sediment target levels for non-polar organic compounds are determined by the concentration in the sediments that would result in interstitial water concentrations equivalent to the water quality criteria (for fresh water). Interstitial water concentrations are estimated from organic-carbon-normalized partition coefficients. The major limitations of the EP approach include the assumptions that equilibrium partitioning is the only significant pathway from the sediments to the biota, that equilibrium steady-state conditions exist in aquatic systems, and that the water quality criteria values determined by sediment-free bioassays have a direct application to benthic systems.

The screening level concentration (SLC) is a field-based approach developed in support of the EP model. "The SLC is an estimate of the highest concentration of a particular nonpolar organic contaminant in sediment that can be tolerated by approximately 95 percent of benthic infauna" (Battelle 1986). The SLC is derived from synoptic collections of sediment concentrations and benthic invertebrate distributions from several geographic regions. Since it does not take into account other factors that might influence distributions of benthic infauna (e.g., other contaminants, habitat variables, water depth, sediment grain size), the SLC requires a large database with a broad range of toxicant concentrations to define the influence of any one substance. One of the major limitations of this approach is that the presence of a species at a site does not necessarily imply lack of a biological effect.

The apparent effects threshhold (AET) approach uses field data (chemical concentrations in sediment) and at least one biological indicator of injury (sediment bioassays, altered benthic infaunal abundance, bioaccumulation, histopathology, etc.) to determine the concentration of a given contaminant above which statistically significant biological effects would be expected (Tetra Tech 1986). The AET approach was developed in Puget Sound to establish chemical criteria for disposal of dredged material and is also being considered by U.S. EPA Region 10 for use in establishing cleanup target levels in Puget Sound. As in the SLC approach, the AET requires a large database, and results could be strongly influenced by the presence of unmeasured, covarying toxic contaminants. In addition, AET values based

on bioassay results may not reflect chronic effects that may be of particular importance for PCBs.

The sediment triad approach uses a combination of sediment chemistry, sediment bioassays, and some measure of in situ biological effects (benthic infaunal abundance, bottom fish histopathology, etc.). Data are summarized by area (e.g., embayment). Data requirements are similar to the AET, but the approach thus far has been used only for comparisons among broad spatial areas. Results indicate chemical concentrations below which biological effects would be expected to be minimal and concentrations above which effects would be expected to be severe (Long and Chapman 1985; Chapman 1986). The triad approach, in contrast to the AET, requires the definition of "minimal" and "severe" biological effects to establish criteria.

Sediment quality (target level) values for PCBs have been determined by each of the four approaches and are presented below (Table 4). EP values for PCBs range from 0.001 ppm for dichlorobiphenyls to 0.14 ppm for hexachlorobiphenyls, based on the water quality criterion for PCBs in freshwater (0.014 ug/l) at 2% total organic carbon (TOC). SLC values for total PCBs are 0.006 ppm for freshwater and 0.086 ppm for saltwater normalized for 2% TOC. The AET values for total PCBs range from 0.13 ppm dry weight based on the Microtox bioassay to 2.5 ppm based on the amphipod bioassay; the benthic infauna test and oyster larvae bioassay both resulted in intermediate AET values of 1.1 ppm. The bioaccumulation AET value of 0.14 ppm was based on PCB concentrations in English sole muscle tissue and calculated human risk assessment; however, the mobility of English sole makes the relationship between sediment concentrations and tissue residues less direct than for the other AET values. The sediment triad approach determined that biological effects were minimal at sediment concentrations of 0.1 ppm total PCBs. It is worth noting that the EP, SLC, AET, Triad approaches have all yielded target values for PCBs that are mostly well below 1 ppm, while recognizing that none of the values have received extensive validation or application.

## Conclusions

The primary objective of this paper was to define a sediment concentration that is protective of aquatic ecosystems from PCB toxicity. The application of sediment target levels to specific systems will involve additional considerations before site-specific cleanup criteria can be established. Since the PCB target levels are based primarily on chronic effects and may affect only small portions of a population or ecosystem, the ecosystem destruction or disruption accompanying a large cleanup attempt may be greater than that associated with the PCB, even over the long term. Further, the cleanup operation itself may cause unacceptable problems. For example, a recent study by Rice and White (1987) reported that dredging to remove PCB-contaminated sediments in a Michigan river resulted in increased bioavailability of PCBs to caged clams and fish at a station 11 km downstream for at least six months.

Finally, establishing sediment target levels does not address the question of how data are to be collected to determine the extent of contamination in a specific area and how the data should be analyzed to provide numerical comparisons to the target levels (e.g., point-by-point comparisons, single-point subjective or computer-aided contouring, kriging, averages, mean values, upper confidence level limit or other statistical methods). Sediment PCB concentrations can be expected to exhibit considerable variation within an area. Different methods of data collection and interpretation can make large differences in the remedial process, even if the target level is enforced.

Table 4. Summary of PCB Sediment Quality Target Values From Four Approaches.

| | |
|---|---|
| Equilibrium Partitioning: | (JRB 1984; Quinlan et al. 1985) (µg/g dry wt @ 2% TOC) |
| Marine | 0.001[a] - 0.14[b] |
| Screening Level Concentration: | (Battelle 1986) (µg/g dry wt @ 2% TOC)(total PCBs) |
| Freshwater | 0.006 |
| Marine | 0.086 |
| Apparent Effects Threshhold: | (Tetra Tech 1986) (µg/g dry wt) (total PCBs) |
| Marine (Puget Sound): | |
| Microtox bioassay | 0.13 |
| Bioaccumulation | 0.14 |
| Benthic infauna (higher taxa) | 1.1 |
| Oyster larvae bioassay | 1.1 |
| Amphipod bioassay | 2.5 |
| Sediment Triad: | (Chapman 1986) (µg/g dry wt) (total PCBs) |
| Marine (Puget Sound): | |
| no or minimal biological effects | < 0.1 |
| major biological effects | > 0.8 |

[a] Value for dichlorobiphenyls
[b] Value for hexachlorobiphenyls

The different approaches considered in this report (bioaccumulation/toxicity, EP, SLC, AET, and Triad) arrive at sediment quality target values for PCBs in the range of 0.1 ppm or less. Since the levels developed by most of these approaches are the concentrations at which some effects were noted, it could be argued that a safety factor should be added to these values to achieve lower concentrations of PCB that would be more fully protective of the health of all aquatic organisms. However, given the widespread distribution of PCB contamination in aquatic environments (in many cases the "background" levels in an area are greater than 0.1 ppm), achieving specific target levels may not be feasible.

At this point in our understanding of the distribution, uptake and toxic effects of PCBs, a sediment concentration of 0.1 ppm appears to be a reasonable preliminary target level for use in assessing the environmental hazards from PCB contamination and the need for remedial action. Toxic effects may occur at concentrations below 0.1 ppm, particularly in

systems with sediment bioaccumulation factors much greater than 1 or as indicated by the SLC value for freshwater. It is also possible that in systems with sediment bioaccumulation factors of less than 1, sediment concentrations of 0.1 ppm may not result in detectable toxicity.

It must be recognized that this sediment target level of 0.1 ppm has a substantial, but undefined, uncertainty. The many remaining unanswered questions make it difficult and perhaps inappropriate to attempt to define a target level with more precision. It is still unclear, for example, to what extent different toxic responses may be related to the specific mixture of PCB congeners present, and whether the necessary analytical measurements to support target levels for specific types of PCBs could be routinely performed. Similarly, while it is known that the bioavailability of PCBs is affected by the physical and chemical properties of the sediments, the effects of sediment characteristics and lipid content and other characteristics of aquatic organisms on the uptake of sediment-bound PCBs by organisms have not been well-defined.

Given all the uncertainties presented above, it should be clear that even with acceptable sediment quality *target levels*, the actual cleanup *criteria* to be used must still be evaluated on a site-by-site basis.

## REFERENCES

Addison, R.F., M.E. Zinck and D.E. Willis. 1981. Time- and dose-dependence of hepatic mixed function oxidase activity in brook trout, Salvelinus fontinalis on polychlorinated biphenyl residues: implications for "biological effects monitoring". Env.Pollut. 25A: 211-218.

Alford-Stevens, A.L. 1986. Analyzing PCBs. Environ. Sci. Technol. 20: 1194-1199.

Battelle. 1986. Sediment quality criteria methodology validation: calculation of screening level concentrations from field data. Final Report. U.S. Environmental Protection Agency, Criteria and Standards Division. Battelle, Washington, DC. 60 pp. + appendices.

Bengtsson, B.-E. 1980. Long-term effects of PCB (Clophen A 50) on growth, reproduction and swimming performance in the minnow, Phoxinus phoxinus. Water Res., 14: 681-687.

Binder, R.L. and J.J. Lech. 1984. Xenobiotics in gametes of Lake Michigan lake trout (Salvelinus namaycush) induce hepatic monooxygenase activity in their offspring. Fundaml. appl. Toxic. 4: 1042-1054.

Binder, R.L., J.J. Stegeman and J.J. Lech. 1985. Induction of cytochrome P-450-dependent monooxygenase systems in embryos and eleutherembryos of the killfish Fundulus heteroclitus. Chem. -Biol. Interact. 55: 185-202.

Boon, J.P. and J.C. Duinker. 1985. Processes determining the kinetics of PCB congeners in marine organisms: a comparison between laboratory and environmental studies. Mar. Environ. Res. 17: 301-305.

Breck, J.E. 1985. Comment on "Fish/sediment concentration ratios for organic compounds." Environ. Sci. Technol. 19: 198-199.

Brown, J.F. Jr., R.E. Wagner, H. Feng, D.L. Bedard, M.J. Brennan, J.C. Carnahan and R.J. May. 1987. Environmental dechlorination of PCBs. Environ. Toxicol. Chem. 6: 579-593.

Brown, M.P., M.B. Werner, R.J. Sloan and K.W. Simpson. 1985. Polychlorinated biphenyls in the Hudson River. Environ. Sci. Technol. 19: 656-661.

Cairns, T., G.M. Doose, J.E. Froberg, R.A. Jacobson and E.G. Siegmund. 1986. Analytical chemistry of PCBs. In: J.S. Waid (ed.), PCBs and the Environment. 1: 1-46. CRC Press, Inc. Boca Raton, Florida.

Chapman, P.M. 1986. Sediment quality criteria from the sediment quality triad: An example. Environ. Toxicol. Chem. 5: 965-976.

Chou, S.F.J. and R.A. Griffin. 1986. Solubility and soil mobility of polychlorinated biphenyls. In: J.S. Waid (ed.), PCBs and the Environment. 1: 101-120. CRC Press, Inc. Boca Raton, Florida.

Clark, J.R., J.M. Patrick, Jr., J.C. Moore, and J. Forester. 1986. Accumulation of Sediment-Bound PCBs by Fiddler Crabs. Bull. Environ. Contam. Toxicol. 36:571-578.

Doucette, W.J. and A.W. Andren. 1987. Correlation of octanol/water partition coefficients and total molecular surface area for highly hydrophobic aromatic compounds. Environ. Sci. Technol. 21: 821-824.

Eisenreich, S.J. 1987. The chemical limnology of nonpolar organic contaminants: polychlorinated biphenyls in Lake Superior. In: Hites, R.A. and S.J. Eisenreich (eds.), Sources and Fates of Aquatic Pollutants. American Chemical Society. Advances In Chemistry Series 216: 393-469. Washington, D.C.

Eisler, R. 1986. Polychlorinated Biphenyl Hazards to Fish, Wildlife, and Invertegrates: A Synoptic Review. Contaminant Hazard Reviews Report No. 7, Biological Report 85(1.7). U.S. Fish and Wildlife Service. 72 pp.

Erickson, M.D.. 1985. Analytical Chemistry of PCBs. Ann Arbor Science Publishers, Ann Arbor, Michigan. 508 pp.

Fisher, J.B., R.L. Petty and W. Lick. 1983. Release of polychlorinated biphenyls from contaminated lake sediments: flux and apparent diffusivities of four individual PCBs. Environ. Pollut. 5B: 121-132.

Fox, M.E., J.H. Carey and B.G. Oliver. 1983. Compartmental distribution of organochlorine contaminants in the Niagara River and the western basin of Lake Ontario. J. Great Lakes Res. 9: 287-294.

Fries, G.F., and G.S. Marrow. 1981. Chlorobiphenyl movement from soil to soybean plants. J. Agric. Food Chem. 29: 757-759.

Furukawa, K. 1986. Modification of PCBs by bacteria and other microorganisms. In: J.S. Waid (ed.), PCBs and the Environment. 2: 89-100. CRC Press, Inc. Boca Raton, Florida.

Hutzinger, O., S. Safe and V. Zitko. 1974. The Chemistry of PCBs. CRC Press, Inc. Cleveland, Ohio. 269 pp.

Jensen, A.L.. 1984. PCB uptake and transfer to humans by lake trout. Environ. Pollut. 34A: 73-82.

Jensen, A.L., S.A. Spigarelli and M.M. Thommes. 1982. PCB uptake by five species of fish in Lake Michigan, Green Bay of Lake Michigan, and Cayuga Lake, New York. Can. J. Fish. Aquat. Sci. 39: 700-709.

JRB Associates. 1984. Initial evaluation of alternatives for development of sediment related criteria for toxic contaminants in marine waters (Puget Sound). Phase II. Development and testing of the sediment-water equilibrium partitioning approach. U.S. Environmental Protection Agency. Seattle, Washington.

Larsson, P. 1984. Transport of PCBs from aquatic to terrestrial environments by emerging chironomids. Environ. Pollut. 34A: 283-289.

Larsson, P. 1985. Contaminated sediments of lake and oceans act as sources of chlorinated hydrocarbons for release to water and atmosphere. Nature 317: 347-349.

Larsson, P. 1986. Zooplankton and fish accumulate chlorinated hydrocarbons from contaminated sediments. Can. J. Fish. Aquat. Sci. 43: 1463-1466.

Lech, J.J. and R.E. Peterson. 1983. Biotransformation and persistence of polychlorinated biphenyls (PCBs) in fish. In: D'Itri, F.M. and M.A. Kamrin (eds.), PCBs: Human and Environmental Hazards. Ann Arbor Science Publishers, Inc. Ann Arbor, Michigan. pp. 187-201.

Lee, R.F. 1981. Mixed function oxygenase (MFO) in marine invertebrates. Mar. Biol. Lett. 2: 87-105.

Long, E.R. and P.M. Chapman. 1985. A sediment quality triad: Measures of sediment contamination, toxicity and infaunal community composition in Puget Sound. Mar. Pollut. Bull. 16: 405-415.

Lynch, T.R. and H.E. Johnson. 1982. Availability of a hexachlorobiphenyl isomer to benthic amphipods from experimentally contaminated sediments. In: J.G. Pearson, R.B. Foster and W.E. Bishop (eds.). Aquatic Toxicology and Hazard Assessment: Fifth Conference. ASTM STP 766. pp. 273-287.

Malins, D.C., B.B. McCain, D.W. Brown, A.K. Sparks, and H.O. Hodgins. 1980. Chemical Contaminants and Biological Abnormalities in Central and Southern Puget Sound. NOAA Technical Memorandum OMPA-2. National Oceanic and Atmospheric Administration, Boulder, CO. 295pp.

Mauck, W. 1987. Personal communication of unpublished data. U.S. Fish and Wildfife Service, Columbia, Missouri.

McLeese, D.W., C. Metcalf and D. Pezzack. 1980. Uptake of PCBs from sediments by Nereis virens and Crangon septemspinosa. Arch. Environ. Contam. Toxicol. 9: 507-518

Monod, G. 1985. Egg mortality of Lake Geneva charr (Salvelinus alpinus) contaminated by PCB and DDT derivatives. Bull. Env. Contam. Toxicol. 35: 531-536.

National Marine Fisheries Service. 1987. Results of 1984 (Cycle-1) benthic surveillance project along the southeast Atlantic and Gulf of Mexico coasts. National Marine Fisheries Service, Southeast Fisheries Center. Beaufort, North Carolina. 125 pp + appendices.

Neff, J.M. 1984. Bioaccumulation of organic micropollutants from sediments and suspended particulates by aquatic animals. Fres. Z. Anal. Chem. 319: 132-136.

Niimi, A.J. 1983. Biological and toxicological effects of environmental contaminants in fish and their eggs. Can. J. Fish. Aquat. Sci. 40:306-312.

Pal, D., J.B. Weber, and M.R. Overcash. 1980. Fate of Polychlorinated Biphenyls (PCBs) in Soil-Plant Systems. Residue Reviews 74: 77-98.

Pavlou, S.P. and R.N. Dexter. 1979. Distribution of polychlorinated biphenyls (PCB) in estuarine ecosystems. Testing the concept of equilibrium partitioning in the marine environment. Environ. Sci. Technol. 13: 65-71.

Payne, J.F., L.L. Fancey, A.D. Rahimtula and E.L. Porter. 1987. Review and perspective on the use of mixed-function oxygenase enzymes in biological monitoring. Comp. Biochem. Physiol. 86C: 233-246.

Phillips, D.J.H. 1986. Use of organisms to quantify PCBs in marine and estuarine environments. In: J.S. Waid (ed.), PCBs and the Environment. 2: 127-182. CRC Press, Inc. Boca Raton, Florida.

Pruell, R.J., J.L. Lake, W.R. Davis and J.G. Quinn. 1986. Uptake and depuration of organic contaminants by blue mussels (Mytilus edulis) exposed to environmentally contaminated sediments. Mar. Biol. 91: 497-508.

Quinlan, E.A., P.M. Chapman, R.N. Dexter, D.E. Konasewich, C. Ebbesmeyer, G.A. Erickson, B.R. Kowalski and T.A. Silver. 1985. Toxic chemicals and biological effects in Puget Sound: Status and scenarios for the future. NOAA Tech. Mem. NOS OMA-10. Seattle: National Oceanic and Atmospheric Administration.

Ray, S., B.M. Jessop, J. Coffin, and D.A. Swetnam. 1984. Mercury and polychlorinated biphenyls in striped bass (Morone saxatilis) from two Nova Scotia rivers. Water, Air, Soil Pollut. 21:15-23.

Rice, C.P. and D.S. White. 1987. PCB availability assessment of river dredging using caged clams and fish. Environ. Toxicol. Chem. 6: 259-274.

Rogerson, P.F., S.C. Schimmel, and G. Hoffman. 1985. Chemical and Biological Characterization of Black Rock Harbor Dredged-Material. Tech. Report D-85-9, U.S. EPA Environmental Research Laboratory. Narragansett, Rhode Island. 123 pp. and appendices.

Rubenstein, N., W.T. Gilliam and N.R. Gregory. 1984. Dietary accumulation of PCBs from a contaminated sediment source by a demersal fish (Leiostomus xanthurus). Aquat. Toxicol. 5: 331-342.

Rubenstein, N.I., E. Lores, and N.R. Gregory. 1983. Accumulation of PCBs, mercury and cadmium by Nereis virens, Mercenaria merceneria and Palaemonetes pugio from contaminated harbor sediments. Aquat. Toxicol. 32:249-260.

Safe, S. 1984. Polychlorinated biphenyls (PCBs) and polybrominated biphenyls (PBBs): biochemistry, toxicology, and mechanism of action. CRC Crit. Rev. Toxicol. 13:319-393.

Sawhney, B.L. 1986. Chemistry and properties of PCBs in relation to environmental effects. In: J.S. Waid (ed.), PCBs and the Environment. 1: 47-65. CRC Press, Inc. Boca Raton, Florida.

Seelye, J.G., R.J. Hesselberg and M.J. Mac. 1982. Accumulation by fish of contaminants released from dreged sediments. Environ. Sci. Technol. 16:459-464.

Shaw, G.R. and D.W. Connell. 1982. Factors influencing concentrations of polychlorinated biphenyls in organisms from an estuarine ecosystem. Aust. J. Mar. Freshw. Res. 33:1057-1070.

Sloan, R.J. and R.W. Armstrong. 1981. PCB patterns in Hudson River fish. II. Migrant/marine species. Proc. Hudson R. Environ. Soc., Hyde Park, N.Y. 53 pp.

Spies, R.B., D.W. Rice and R.R. Ireland. 1984. Preliminary studies of growth, reproduction and activity of hepatic mixed-function oxidase in Platichthys stellatus. Mar. Environ. Res. 14: 426-428.

Spies, R.B., D.W. Rice, Jr., P.A. Montagna and R.R. Ireland. 1985. Reproductive success, xenobiotic contaminants and hepatic mixed-function oxidase (MFO) activity in Platichthys stellatus populations from San Francisco Bay. Mar. Environ. Res. 17: 117-121.

Stout, V.F. 1986. What is happening to PCBs? Elements of effective environmental monitoring as illustrated by an analysis of PCB trends in terrestrial and aquatic organisms. In: J.S. Waid (ed.), PCBs and the Environment. 1: 163-205. CRC Press, Inc. Boca Raton, Florida.

Swartz, R.C. and H. Lee. 1980. Biological processes affecting the distribution of pollutants in marine sediments. Part I: accumulation, trophic transfer, biodegradation and migration. In: R.A. Baker (ed.), Contaminants and Sediments. Vol. II: 533-553. Ann Arbor Science Publishers, Ann Arbor, Michigan.

Tanabe, S., R. Tatsukawa and D.J.H. Phillips. 1987. Mussels as bioindicators of PCB pollution: a case study on uptake and release of PCB isomers and congeners in green-lipped mussels (Perna viridis) in Hong Kong waters. Environ. Pollut. 47: 41-62.

Tatem, H.E. 1982. Bioaccumulation of PCB By Freshwater Prawns and Clams Exposed to Contaminated Dredged Material. Engineering and Scientific Research at WES, Vol. O-82-3. pp. 1-6. Vickburg, Mississippi. U.S. Army Corps of Engineers, Waterways Experiment Station.

Tetra Tech. 1985. Commencement Bay Nearshore/Tideflats Remedial Investigation, Final Report, Volume 1. EPA Report No. EPA-910/9-85-134b. Seattle: Washington State Department of Ecology and U.S. Environmental Protection Agency.

Tetra Tech. 1986. Development of sediment quality values for Puget Sound. Final Report. U.S. Army Corps of Engineers, Seattle District. Puget Sound Dredged Disposal Analysis and Puget Sound Estuary Programs. 128 pp. + appendices.

Thomann, R.V. and J.P. Connolly. 1984. Model of PCB in the Lake Michigan lake trout food chain. Environ. Sci. Technol. 18: 65-71.

U.S. Environmental Protection Agency. 1980. Ambient water quality criteria for polychlorinated biphenyls. U.S. Environ. Protection Agency Rep. 440/5-80-068. 211 pp.

Von Westernhagen, H., H.Rosenthal, V Dethlefsen, W. Ernst, U. Harms and P.-D. Hansen. 1981. Bioaccumulating substances and reproductive success in Baltic flounder, Platichthys flesus. Aquat. Toxicol. 1: 85-99.

Zitko, V. and R.L. Sanders. 1979. Effect of PCBs and other organochlorine compounds on the hatchability of Atlantic salmon (Salmo salar) eggs. Bull. Environ. Contam. Toxicol. 21: 125-130.