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BY FEDERAL EXPRESS

Gayle Garman
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U.S. Environmental Protection Agency, Region I
Waste Management Division (HRM-CAN 3)
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Boston, MA 02203

Re: Comments of AVX Corporation for Inclusion in the Administrative Record for the New Bedford Harbor Superfund Site

Dear Ms. Garman:

On behalf of AVX Corporation, I am pleased to transmit with this letter the comments of AVX Corporation on EPA's proposed remediation for the Upper Estuary, Lower Harbor and Buzzards Bay. This confirms that pursuant to our conversation last week, transmittal of these comments to you today via Federal Express courier will be considered a timely submittal.

Please call me if you have any questions related to these comments. Thank you for your cooperation.

Very truly yours,

Mary K. Ryan

Mary K. Ryan

MKR/smd:4447e
Enclosures (consisting of 4 boxes)

cc: Weldon Bosworth

SDMS DocID 46649

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SUBMITTED JULY 13, 1992

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INTRODUCTION

Ten years after the New Bedford Harbor Superfund site was first placed on EPA's interim list of National Priority List ("NPL") sites in July 1982, EPA stands on the verge of making a final remedial decision for the Acushnet River Estuary, New Bedford Harbor, and Upper Buzzards Bay. During the intervening years, the level of the EPA's activity has not been matched by the level of its productivity. Time after time EPA has embarked on grandiose, ill-considered studies which have, by and large, turned out to be useless in answering the fundamental questions facing EPA. The reasons why this is so are discussed further in the overview of these comments section and in the detailed technical comments that follow.

As a result, instead of following the path EPA laid out for principled decision-making ten years ago, EPA has instead consistently accommodated its decision-making to the failings of its proposed studies. A Record of Decision based on those premises will be little more than a "house of cards" resting on the fatally flawed foundation provided by such studies as the Battelle/Hydroqual model and the Baseline Ecological Risk Assessment. After ten years and tens of millions of dollars, EPA's justification for its proposed remediation in the estuary, harbor and bay cannot withstand scrutiny and can only be considered arbitrary and capricious.

The present situation is regrettable but not irredeemable. EPA should acknowledge the weaknesses in its proffered justification for remedial action, particularly the highly speculative decision to adopt a 10 ppm Target Cleanup Level ("TCL") for the upper bay, and act accordingly to refocus its remedial efforts for New Bedford Harbor. Since the most reliable biota sampling shows that of the species of concern, only lobster continue to exceed the 2 ppm FDA level -- and that the Battelle model, which EPA insists should be used for "comparative" purposes, projects that lobster will be at or near that level in ten years or less without any remedial action -- it is time for EPA to adopt a more realistic and less drastic remedial alternative such as that proposed by AVX in 1989.

OVERVIEW

In the January 1992 Proposed Plan and the May 1992 Addendum Proposed Plan, EPA announced its intention to spend over $40 million for extensive dredging of the estuary (other than the Hot Spot) and harbor, using a Target Cleanup Level ("TCL") of 50 ppm, and to dredge and cap as yet-undefined areas of Upper Buzzards Bay to a TCL of 10 ppm. EPA's insistence on extensive dredging, and the corollary necessity of utilizing vast shoreline CDFs, does little more than move contaminated sediment a few hundred yards. In so doing, however, EPA ignore the deleterious environmental impacts of dredging and wetlands destruction necessitated by its ambitious approach.

Those consequences compel the conclusion that the capping remediation first proposed by Balsam in 1989, which would eliminate over 90% of PCB flux in the harbor by capping Upper Estuary sediments greater than 50 ppm,[1/] effectively accomplishes the same goals with far less significant

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1/ While EPA's dredging plan for the estuary and harbor has the same TCL as AVX's earlier proposal, it differs by including an additional 47 acres in the lower harbor, adding 76,000 cubic yards of dredge spoil. Draft Final Feasibility Study of Remedial Alternatives for the Estuary and Lower Harbor/Bay Volumes I-III, August 1990 ("FS-EST/H/BAY"), Vol. III, 2-26.

environmental impact.[2/] Among the reasons why capping appears more sensible are (1) the government has underestimated the resuspension of sediments and release of sediment-associated contamination associated with dredging; (2) there is good reason to be concerned that containment of contaminated sediment in harborside CDFs will cause continued PCB flux to the harbor waters due to tidal pumping, (3) there is a risk of significant PCB loss due to volatilization to the air both during dredging and during filling and storage in a CDF, and (4) the government continues to vastly underestimate costs.[3/] The disadvantages of CDFs also include their encroachment into the wetlands, problems with long term maintenance and monitoring, and the reduced potential for biodegradation.

2/ For example, the United States Army Corps of Engineers believes that [a] capping operation will release less contamination than a dredging operation although accurately quantifying the difference would be difficult." Attachment to January 25, 1990 letter from Mark Otis to Mary Sanderson, EPA Project Manager; see also FS-EST/H/BAY, Vol. III, 2-10. The technical comments below (Section VI) contain a more detailed comparison of the different remedial approaches proposed by EPA and AVX.

3/ Balsam's analysis of EPA cost estimates indicates a potential for costs to be $13 million dollars higher for the 50 ppm remediation scenario. But the upper range is virtually open-ended because of significant questions about the proposed water treatment technology. If a 10 ppm action level were chosen for the upper Bay, costs could easily quadruple. Given that costs for the Hot Spot Operable Unit have almost doubled over EPA's 1989 predictions, EPA's cost estimates cannot be considered reliable.

Although AVX believes that the extensive dredging contemplated by EPA is a serious mistake, it is even more concerned that decisions about key issues, such as the TCL for Upper Buzzard Bay, are based on faulty studies which provide no reasonable basis to conclude that a 50 ppm cleanup focused principally on the high PCB concentrations in estuary -- such as that proposed by AVX in 1989 -- will not be fully protective of human health and the environment. Even worse, because EPA's ambitious modeling effort proved far too complex to be utilized, EPA has no real way to measure any incremental benefit from the more stringent cleanup urged by NOAA (and perhaps some of the non-government environmentalist groups).

In 1989 and 1990 EPA excused its inability to document the environmental benefit of the Hot Spot cleanup on the grounds that it was only an interim action. That day is long past. EPA must prove that its decision to spend Superfund money on a particular cleanup is based on studies that meet the requirements of the NCP and other statutory and regulatory requirements. The present record provides no justification for EPA to adopt the combined remedies set forth in the Proposed Plan and its Addendum, as just a few examples illustrate:

- The assessment of ecological risk can only be considered speculative;

- The assessment of human health risk is premised on wholly-unrealistic exposure scenarios, an outdated cancer potency factor, and ignores the real world proof of lack of health effects demonstrated by the Greater New Bedford Public Health Effects Study; and, finally,

- EPA has no analytical, quantitative tool to evaluate the effectiveness of any one remedial action scenario over another, particularly with respect to Upper Buzzards Bay, because the modeling efforts by Battelle and Hydroqual are not scientifically reliable.

If EPA nonetheless adopts the proposed cleanup plan *in toto*, its action can only be characterized as arbitrary and capricious.

I. EPA HAS FAILED TO DEMONSTRATE THAT THE PROPOSED REMEDIATION IS JUSTIFIED BY A THREAT TO PUBLIC HEALTH OR THE ENVIRONMENT.

A. EPA Has Overstated Threats To Human Health[4/]

1. Application of proper risk assessment evaluation techniques illustrates the safety of applying a 50 pm PCB cleanup standard to the sediments in the Estuary, Lower Harbor and Bay.

One of the most significant remedial objectives articulated by EPA as justification for the proposed remedial action is reduction of the risk to human health. In response to these concerns, AVX's consultant Terra, Inc. ("Terra") performed an evaluation of the effect of the proposed 50 ppm cleanup level on human health risks.

The Baseline Public Health Risk Assessment prepared by E.C. Jordan for New Bedford Harbor (1989) contained a number of overly conservative exposure assumptions, which in turn led to an overestimation of risk. Terra's evaluation relied upon more realistic exposure assumptions than those utilized by E.C. Jordan with respect to such factors as sediment ingestion rate, fish consumption, the bioavailability of PCBs, the likely years of exposure for an infant or small child, and the frequency of exposure to site sediments.

4/ Although AVX has made this point before, it bears emphasis that, for a number of scientific reasons, AVX does not believe PCBs in New Bedford Harbor pose any threat to public health. These comments are not intended to make any suggestion to the contrary, but are designed to respond to EPA's stated rationale for its selection of remedial action.

In addition, Terra's evaluation utilized not only the PCB cancer potency factor factor applied by E.C. Jordan in its Baseline Public Health Risk Assessment, but also the cancer potency factor previously developed by Terra for use at this site (see Terra's 1989 submittals, which are part of the Administrative Record). Terra also took into account recent scientific evidence indicating that all pertinent PCB cancer bioassays should be reevaluated according to the new tumor pathology classification schemes now recommended and used by the National Toxicology Program. This leads to a cancer potency factor of 1.1 $(mg/kg/day)^{-1}$ (contrasted to EPA's figure of 7.7). Terra's independent risk assessment demonstrated that the excess cancer risk from sediment exposure and seafood ingestion falls within the range of risk considered acceptable by EPA. Based on its analysis, Terra concluded that the adoption of the 50 ppm cleanup guideline for the Estuary, Harbor and Bay sediments will provide a demonstrably safe remediation goal for this area.[5/]

Terra's conclusion is substantiated by the Greater New Bedford PCB Health Effects Study ("GNBHES"). "The Study ... showed that few of the residents who had participated in the study had elevated PCBs, and that the residents with the highest risk of elevated PCB's (from occupational exposure or

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5/ Terra's 1992 evaluation took into account the ongoing Hot Spot remediation.

eating harbor fish) had PCB levels within the typical range of the U.S. population." EPA Hot Spot Responsiveness Summary, p. 6. The authors of the GNBHES specifically concluded that "[a]lmost all individuals who were identified as being at the greatest risk of exposure via contaminated seafood intake had relatively low serum PCB levels." See Terra, Attachment A, p. 15. The GNBHES failed to show that there was any meaningful or significant correlation between fish consumption in New Bedford and elevated PCB levels in blood serum. Though EPA now chooses to emphasize .02 ppm as an allegedly risk-based TCL for biota levels, rather than the 2 ppm tolerance level for PCB residues used by the United States Food and Drug Administration ("FDA"), the simple fact is that there is no proof that any person has been hurt, or will be hurt, by PCBs in New Bedford Harbor fish and lobster.

2. EPA's Concerns About Fish Ingestion As A Human Health Risk Are Misplaced; In Any Event Such Concerns Can Be Addressed In A Far More Reasonable Way Than Through Closure Of The Fishery.

Although EPA repeatedly states that there is no declining trend in PCB levels and biota, more recent (and more reliable) sampling data prove exactly the opposite. Any discussion about biota sampling must start with the acknowledgement that the historical biota sampling is rife with problems that compel the rejection of the data collected by various Massachusetts agencies and laboratories and by the FDA. Because of defects

in the sampling protocol for this historical data, it is also useless as an indicator of site-wide conditions.[6/]

Based on the sampling methodology and the information regarding sample handling and analytical methodology, these historical data must be considered less reliable than the Batelle data. The Battelle data sets indicate that the average lobster muscle, winter flounder, and clam tissue PCB concentration throughout the New Bedford Harbor study area are now less than 1 ppm (Balsam, 1989). If lobster tomalley is considered part of the edible flesh, then lobster may exceed the FDA permissible level in certain areas by small amounts. Even so, Battelle has concluded that only lobster presently exceeds the FDA 2 ppm limit. (For reasons discussed below, AVX submits that the higher levels of PCBs found in the lobster tomalley should not necessitate the continued fishery closure.)

What must also be taken into account is that Balsam (Balsam 1989) has demonstrated that a 50 ppm cleanup level in the estuary alone would lead to achievement of FDA levels in fish. Id. at §3.2.4. Battelle's model also suggests that in ten years, even under a no action scenario, there will likely be no

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6/ These issues were all previously addressed by AVX in the litigation related to the Superfund site, as well as earlier Administrative Record comments. Without limiting its reliance on the full record, see generally Volume III, Sampling and Analysis for RFAs on biota sampling. (Section IV of the Overview explains what an RFA is.)

exceedance of the FDA limit; after a 50 ppm remediation, levels in lobster will drop to near or below 2 ppm (Draft Final Supplemental Feasibility Study Evaluation for Upper Buzzards Bay New Bedford Harbor RI/FS New Bedford, MA, May 1992 ("SFS"), p. 6-11). Therefore, taking into account the remediation that is ongoing in the Hot Spot area, and assuming further remediation in the harbor and bay to achieve a 50 ppm TCL, the record demonstrates that the FDA level will be achieved within approximately a 10-year period.

Given the above remediation activities, EPA has properly focused on what action, if any, it should take in the meantime while remediation is ongoing. In the Addendum Proposed Plan, EPA has asked for comments on institutional controls. According to statements made by EPA's project manager at the June 10 1992 public hearing, the institutional control issue is supposed to address whether there is a better alternative than a fishery closure. Transcript of June 10, 1992 hearing, p. 18. AVX has consistently argued that the fishery closure is an unnecessarily drastic step which has not been effective.[7/]

7/ AVX believes that the history of the closure is important evidence that EPA should take into account in revisiting the issue of institutional controls. AVX has detailed the history of the closure in its 1989 RFAs, specifically RFAs 31,100 to 31,701.

Public informational programs are often utilized in lieu of fishery closures. Fishery closures are in fact quite rare; EPA itself seems to have acknowledged in the Hot Spot responsiveness summary that the extent to which the New Bedford Harbor closure contributed to results of the GNBHES can only be speculated upon. Extensive violation of the closure regulations has been documented over the years.

It is important to remember that fishery closure is based solely on exceedence of the 2 ppm FDA level. A comparison of PCB biota levels in New Bedford to those reported in Quincy Bay highlights inconsistent treatment that has been adopted in New Bedford. After EPA documented high levels in lobster tomalley in Quincy Bay, similar to or greater than those found in New Bedford Harbor, the recommended action was immediate issuance of a human health advisory regarding consumption of lobster tomalley from Quincy Bay -- not a closure recommendation.[8/]

The approach adopted in Quincy Bay illustrates the appropriate institutional controls which EPA should recommend until the remedial action will achieve compliance with FDA

8/ AVX relies upon the information and data set forth in its RFAs Nos. 819 through 844 concerning PCB contamination in Quincy Bay. The joint defendants' October 16, 1989 comments also point out another very significant inconsistency between EPA's approach to New Bedford contrasted to Quincy Bay in EPA's use of a lower cancer potency factor (2.6 mg/kg/day) in the Quincy Bay studies.

levels. An advisory such as that issued for Quincy Bay would properly account for the fact that most PCBs (80%) are found in the lobster tomalley. AVX RFA 31,375. In fact, earlier in the history of the closure, the Massachusetts Divison of Marine Fisheries (DMF) recommended that the closure be lifted and an advisory issued warning consumers not to eat lobster tomalley because this advisory would sufficiently minimize public health risk. AVX RFA 31,376.

The general principles behind such institutional controls are that by advising fishermen and consumers about a variety of matters -- including which areas to avoid fishing in on an everyday basis, eating a wide variety of species, how to prepare and cook food to reduce fat content, and which portions of the fish not to eat -- significant risk reductions can be achieved. For example, cooking alone can decrease PCB concentrations in fish by 60 to 70 percent. See Attachment A at 17. In the case of the New Bedford Harbor Superfund Site, it is obvious that an educational program designed to educate consumers to refrain from eating lobster tomalley would provide the critical information necessary to reduce any public health risk that may exist and should be equally (if not more) effective than the closure in accomplishing that goal.

The Commonwealth of Massachusetts has previously issued such advisories with respect to concern over PCB levels in blue fish. Among other things, these specifically directed

consumers to avoid eating the darker meat because it is oilier and therefore more likely to accumulate PCBs and advised pregnant or lactating women to avoid consumption altogether. At no point did DMF close New Bedford Harbor to the taking of bluefish.

A similar approach was adopted by the State of New Jersey as a result of PCB pollution in the Hudson River. In 1983 the New Jersey Department of Environmental Protection adopted N.J.A.C. 7:15-18A, Fisheries Closures and Advisories for Striped Bass, American Eel, Bluefish, White Perch and White Catfish taken from the Northeast Region of the State. The closures applies only to two species -- American Eel and Striped Bass -- taken from specified areas. In other geographical areas, consumers were advised to "limit consumption" of those two species. Only advisories were issued with respect to bluefish, white perch and white catfish. (The text of the New Jersey regulations may be found in Attachment F.3.)

The upcoming remedial decision provides EPA and state officials with a unique opportunity to reevaluate the necessity for the fishery closure in light of planned cleanup efforts. Based on a realistic assessment of the present biota sampling data and the further reduced levels remediation will bring about, AVX believes EPA and the appropriate state officials

should conclude that the time has come to lift the fishery closure.[9/]

B. EPA's Assessment of Ecological Risk Is Totally Lacking In Scientific Validity.

The Draft Baseline Ecological Risk Assessment issued in 1990 is an unprecedented amalgam of mismatched scientific assumptions designed to buttress the inconclusive results of the earlier draft studies which have never been released to the public.[10/] The uncertainties inherent in the Ecological Risk Assessment by virtue of the techniques chosen make it useless as a predictive tool for the remedial decisions soon to be made by EPA. This is especially true because no attempt whatsoever was made to take into account the effect of the ongoing Hot Spot remediation.

9/ Although the EPA feasibility studies repeatedly note that fish and lobster are or soon will be at or near the 2 ppm FDA level, EPA has nonetheless stated that a waiver of the FDA limit may be needed. If considered necessary, AVX supports such a waiver given present and likely biota levels and the lack of risk therefrom, even at the present time.

10/ Despite repeated requests, EPA has never released the full text of the 1988 Draft Baseline Risk Assessment although it has been discussed in professional papers and its conclusions and results were presented to the Community Work Group in 1988 (See Attachment F.9.) AVX continues to request that this draft be included in the Administrative Record because the limited information concerning this draft suggests that it will show that at the time EBASCO could not substantiate any conclusion of specific quantifiable ecological risk at the site. If this is true, EPA ought to consider the draft in selecting the response action and it should be in the record.

The idea that EPA may adopt at 10 ppm TCL in Upper Buzzard Bay based on the Baseline Ecological Risk Assessment is a matter for grave concern. Moreover, as is true with the Battelle model as well, the much-vaunted Baseline Ecological Risk Assessment has never been used as originally intended to evaluate the effects of comparative remedial action as a tool in remedial decision making, nor can it be because the results of the food chain model cannot be used directly in the ecological risk assessment (Baseline Ecological Risk Assessment, p. 2-18). It is surely the height of arbitrary and capricious action to do a study that cannot be used for its intended purpose.

II. EPA'S PROPOSED SELECTION OF REMEDIAL ACTION IS ARBITRARY AND CAPRICIOUS BECAUSE THE EFFECT OF EPA'S PROPOSED REMEDIAL ACTION IS UNKNOWN.

A. The Battelle Model Cannot Be Used For Any Purpose

In the case of New Bedford Harbor Superfund Site, the linchpin in EPA's arsenal of evaluative tools was intended to be the Battelle model. This multi-million dollar, eight-year effort was expected to provide the definitive answer to every question. Instead, it is a dismal failure. Far worse than the agency's reckless undertaking of a project that was overdesigned and doomed to failure from the start, however, is the selective way in which EPA uses the model results. On the one hand, EPA lacks the fortitude to admit candidly that the model is nothing but an expensive useless toy; rather, the Agency insists that it can be used as a tool in the comparative

evaluation of remedial alternatives over a 10-year period. FS-EST/H/BAY, 2-31, 2-43. On the other hand, in the SFS, EPA proposes to select a 10 ppm TCL, but ignores the results of the model's comparative remedial action scenarios which show that there is virtually no difference between the no action, 10 ppm, and 50 ppm scenario in terms of biota responses and PCB water column levels. FS-EST/H/BAY, ES-2, 2-27, 2-28. Such conduct can only be considered arbitrary and capricious.[11/]

B. The Proposed Bay Remedy Is A Politically Expedient, Not Scientifically Justified, Cleanup

EPA now proposes a TCL of 10 ppm for the Upper Buzzards Bay portion of the New Bedford Harbor Superfund Site, but the claimed benefits of cleaning the upper bay to 10 ppm -- reducing ecological exposure to PCB contaminated sediment, reducing PCB water concentrations by reducing PCB sediment concentrations, and reducing PCB concentrations in biota by reducing PCB migration from upper bay sediments - cannot be proven for two fundamental reasons.

The first fundamental problem is that EPA lacks any basis to characterize baseline conditions. As already discussed,

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11/ As the comments above indicate, AVX believes any reliance on the Battelle Model is unwarranted. Nonetheless, because EPA insists upon using the model for comparative analysis of remedial action scenarios, AVX's comments frequently point out that EPA's proposed remediation is inconsistent with the course of action the model results suggest would be appropriate.

EPA's characterization of ecological risk cannot withstand scrutiny. The Baseline Ecological Risk Assessment is scientifically unreliable and EPA has generated no other site specific ecotoxicity data to justify its broad qualitative conclusion of incremental benefit. Problems with EPA's evaluation of human health risks have also been discussed above.

Most surprisingly, EPA has fallen down on one of the most basic tasks in the Superfund process -- collection of sufficient field data to full charactersize the nature and extent of site contamination. Because of QA/QC deficiencies in other data, EPA has relied upon only two data sets for site characterization -- the Battelle Sampling Program and the NUS/GZA Grid Sampling Program. These are simply inadequate to permit EPA to make reasonable judgments about choosing further remedial action in the upper bay.

EPA's failure to properly characterize this site is a fundamental violation of requirements for a proper remedial investigation. The NCP provides that:

> The purpose of the remedial investigation (RI) is to collect data necessary to adequately characterize the site for the purpose of developing and evaluating effective remedial alternatives.

55 Fed. Reg. 46, 8847 (1990). The following excerpt from EPA's guidance for conducting remedial investigations drives home the point:

> In general, the RI/FS must obtain data to define source areas of contamination . . . to the extent necessary to:
>
> - Determine whether, or what extent, a threat to human health or the environment exists
> - Develop and evaluation remedial alternatives (including the no-action alternative)
>
> . . .
>
> The possible uses of the data include the following:
>
> - Monitoring during implementation
> - Health and safety planning
> - <u>Site characterization</u>
> - Risk assessment
> - <u>Evaluating alternatives</u>
> - Determining the PRP
> - <u>Engineering the design of alternatives</u>

Interim Final Guidance for Conducting remedial investigations and Feasibility Studies under CERCLA, OSWER Directive 9355.3-01 (October 1988) at 2-10 (emphasis added). Site Characterization -- that is, the collection of data necessary to fully delineate the nature and extent of the contamination, including the horizontal and vertical extent of the contamination -- is a critical ingredient of any remedial investigation. <u>See</u> Guidance, Chapter 3, p. 3-1 <u>et seq</u>., particularly § 3.4.1.3.

Where is the remedial investigation for Upper Buzzards Bay? Despite the bulk of the Administrative Record dating back

to 1982, EPA relies on only two limited data sets. In the technical comments that follow, AVX has analyzed this point even further, but if there is any question that the sampling database is woefully inadequate, it can be quickly resolved by realizing that each and ever remedial alternative considered for Upper Buzzards Bay includes an extensive, complex predesign sampling plan (over 350 samples) as a component. This is not a case of "new information" or "unknown conditions" -- EPA is fully aware of PCB contamination in Upper Buzzards Bay but it has simply failed to do its homework. EPA has put the cart before the horse by selecting a remedial action, and only then doing the site characterization necessary to see where remediation should take place and what it will accomplish.

The obvious deficiencies in this data have been noted time and time again by EPA's own contractors, as demonstrated by the following:

- With respect to the deficiencies in characterization of the contamination of Upper Buzzards Bay EPA said "[I]t is important to note that efforts to accurately evaluate the extent of PCB contamination in the Upper Bay are constrained by limitations in the spatial extent of the underlying sampling programs and associated data sets." SFS, 2-9.

- EPA review of the Trustees' effectiveness evaluation ends as follows: "It should be noted that uncertainties in the estimated total acreage of Upper Bay sediments exceeding 10 ppm PCBs combined with uncertainties in the actual PCB concentrations within these areas could significantly affect model results." SFS, 4-26

In fact, EPA could not even complete a sensitivity analysis to evaluate the potential effect of the uncertainties associated with the remedial sediment volumes "due to the potential magnitude of the sediment uncertainties." SFS, 6-63. Even the Coastal Zone Management office of the Commonwealth of Massachusetts, in comments filed with EPA on March 29, 1992, stated that "[T]he lack of data which has undergone quality assurance review suggests that additional studies should be conducted prior to making decisions concerning remedial actions." CZM's approval of the 10 ppm TCL in the Bay was only tentative: "Given the limited data set," all that CZM can conclude is that the remedial action outlined in the Addendum Proposed Plan "may" help. CZM closes its comments by urging EPA to do more studies before the cleanup because otherwise the cost benefit ratios may be completely misevaluated.

NOAA itself, the major proponent of a more stringent TCL for the bay, acknowledged that "the major source of uncertainty associated with this application of the relative exposure model is concern over the inadequacy of the data base." (NOAA, p. 7). Strikingly, NOAA's effectiveness evaluation contains the following three conclusions:

- "Additional investigations into the magnitude and extent of TCB sediment contamination and the Outter Harbor could significantly alter the findings of the relative exposure model, and either reduce the effectiveness of the 10 ppm TCL, or enhance it."

- "The large amount of uncertainty resulting from a limited amount of information available on the present spacial distribution of sediment TCB contamination Upper Buzzards Bay make any conclusions speculative."

- "Without additional data to adequately define the areas of elevated concentrations and the residual contamination in background areas, a technical evaluation of the effectiveness of remediation and reducing PCB concentrations and biota is uncertain."

NOAA's last comment leads inevitably to the second major problem EPA has in trying to convince the public that there is good reason to resort to a 10 ppm TCL: without an adequate ability to characterize baseline conditions, there is no way to judge the effectiveness of any selected remedial alternative.

It is obvious that the EPA goal underlying the 10 ppm TCL cannot be simply to meet certain ARARs because accomplishment of that goal is already in sight. The Supplemental Feasibility Study acknowledges that surface water concentrations in the upper bay are already virtually in compliance with the PCB AWQC except for a <u>single</u> data point below the hurricane dike which will be remediated under the 50 ppm TCL. SFS, pp. 6-10. The Battelle model also projects that surface water will soon meet AWQC levels even without any remediation. Battelle has further concluded that only lobster presently exceeds the FDA two ppm limit and that after a 50 ppm remediation, levels in lobster will drop to near or below 2 ppm (SFS, Page 6-11). Indeed, Battelle has concluded that remediation to 10 ppm will have virtually no impact different than the proposed 50 ppm clean-up

for the lower harbor estuary in limited areas of the Bay. (SFS, Page 6-37).

It is clear, then, that what EPA really seeks to achieve with its 10 ppm TCL for the bay is some degree of unspecified incremental improvement beyond achievement of AWQC and FDA levels. The following quotes are but a sample of what has been said about EPA's uncertainty and inability to evaluate and judge the impact and effectiveness of the selected remedial alternative in meeting that goal.

- The strongest statement EPA can make about Upper Buzzards Bay as the source of PCB contamination is that "some but potentially not all of the sediment PCB contamination in Upper Buzzards Bay can be attributed to transport and deposition from the more highly contaminated sediments of the Estuary and Lower Harbor." SFS, ES-1 (emphasis added).

- Although EPA claims that the 10 ppm TCL in Upper Buzzards Bay will result in a reduction of PCB levels in biota, the magnitude of the benefits to marine biota are "somewhat uncertain." SFS, ES-3 (emphasis added).

- Although improvements in PCB residue levels in biota is the claimed justification for the more stringent bay TCL, EPA says "[i]t is difficult to draw conclusions on the relative bioavailability of the sediment bound PCB in the Bay based on the current PCB and TOC data sets alone. This is because of the spatial limitations of the PCB data set, and the lack of TOC measurements that directly correspond with the PCB measurements." SFS, p. 2-11.

- "Remediation of Upper Bay sediments to a 10 ppm action level for PCBs may reduce PCB concentrations in biota .... However, due to the complexity of PCB contaminated sediment-biota interactions, considerable uncertainty exists concerning the magnitude of possible PCB reductions in biota of concern. SFS, pp. 4-23, 24 (emphasis added).

- "In the absence of remediation of sediments to a 10 ppm action level, estimates from the Battelle Model indicate the PCB levels in certain biota in the Upper Bay may gradually drop below the 2 ppm PCB limit established by FDA. It is, however, uncertain as to whether remediation of Upper Bay sediments to a 10 ppm action level would significantly hasten this process. SFS, 4-24 (emphasis added).

- As to ecological risk, the SFS can make only this joinative conclusion: "A 10 ppm action level may therefore result in greater protection for aquatic biota in the Upper Bay." Id. (emphasis added).

- As to the results of the Trustees [sic] effectiveness evaluation which forms the basis of the 10 ppm TCL decision, EPA says this: "The results of the trustees' modeling efforts suggest that remediation of the Upper Bay areas ... to a 10 ppm action level may have somewhat limited impacts on PCB levels in biota in the overall Upper Bay region." Even that is not a sufficient qualification because EPA goes on to say that "[d]ue to the sensitivity of the Trustees [sic] model calculations to certain input assumptions, some uncertainty should be considered inherent in the model's quantitative predictions. SFS, 4-26 (emphasis added). (These factors include extent of biota exposure, how PCBs from sediments get into biota, and how much PCB contamination there is greater than 10 ppm. Id.)

- "While the average sediment PCB concentration for the Upper Bay as a whole (i.e., 1 to 2 ppm) may be close to the recommended 1 ppm sediment TCL for protection of aquatic species, the potential impact of lowering the localized areas of sediment contamination from 50 ppm to 10 ppm is difficult to quantify.... Intuitively, reducing PCB concentrations to 10 ppm in the localized areas contamination should have positive effects on marine biota." Id. (emphasis added).

The NCP does not authorize EPA to rely on "intuition" instead of science. The bottom line is that although EPA says that the additional cleanup in the upper bay will provide incremental benefits in reducing ecological exposure and reducing residual tissue levels, EPA is utterly unable to quantify the degree of improvement, relying instead on

"intuition." The claimed incremental benefit is a purely speculative one.

To persist in adopting a 10 ppm TCL on the present record is not only a violation of statutory and regulatory requirements, but it is also wholly inconsistent with the approach EPA has previously adopted at this site. When it came time to consider the Hot Spot, EPA would not wait for further studies because, it claimed, division of the site into distinct operable units was wholly appropriate. However, now that EPA faces extensive sampling deficiencies in Upper Buzzards Bay due to data quality problems with hundreds of sample points collected earlier, EPA refuses to halt its headlong impetus for remedial action by once again employing the operable unit concept as a tool.

The NCP defines an "operable unit" as follows:

> Operable unit means a discrete action that comprises an incremental step toward comprehensively addressing site problems. . . . The cleanup of a site can be divided into a number of operable units depending on the complexity of the problems associated with the site. Operable units may address geographical portions of a site. . . or may consist of any set of actions performed over time or any actions that are concurrent but located in different parts of a site.

55 Fed. Reg. 8817 (1990). See also NCP §§ 300.430(A)(1)(ii)(A) and (B), Id. at 8846.

A review of the Supplemental Feasibility Study suggests that for all practical purposes, EPA is evaluating bay remediation distinct and apart from the rest of the site. It is absolutely striking that in Section 6.0 of the SFS, the Detailed Analysis Of Remedial Action Alternatives, the analysis of a minimal no action scenario for the upper Buzzards Bay assumes a 50 ppm cleanup in the Estuary and Lower Harbor, as outlined in the FS-EST/H/BAY. For all practical purposes, then, Upper Buzzards Bay already is being treated as a separable operable unit but for inexplicable reasons, EPA refuses to acknowledge this explicitly.

EPA's handling of operable units at this site -- compare its severance of the Hot Spot into a separate operable unit when it was convenient to do so because the Battelle model and ecological risk assessments were delayed, with its refusal to take the same action with respect to Upper Buzzards Bay when sampling problems make it clear that site characterization is grossly inadequate -- can only be said to be arbitrary and capricious.

III. EPA'S ARARS EVALUATION IS FAULTY.

Compliance with federal and state law and regulations deemed applicable or relevant and appropriate (called "ARARs") is mandated by CERCLA and the NCP and is customarily evaluated as one of the nine Superfund criteria established by EPA policy. Compliance with ARARs is mentioned further in the

technical comments that follow, but there are a few critical issues related to the ARARs evaluation which should be emphasized here.

A. EPA Has Repeatedly, And Mistakenly, Identified The PCB AWQC As An "Applicable" Regulation.

In connection with its authorities under the Massachusetts Clean Waters Act (M.G.L. Chapter 21, Sections 26 et seq.), the Division of Water Pollution Control of the Department of Environmental Protection (DWPC) has promulgated the Massachusetts Surface Water Quality Standards (SWQSs) (314 CMR 4.00). These standards are used by DWPC in regulating discharges of pollutants to surface waters. DWPC is charged with limiting or prohibiting such discharges "to insure that the water quality standards of the receiving waters will be maintained or attained" (314 CMR 4.02(1)). Specific water quality criteria for defined classes of surface waters are set out in 314 CMR 4.03(4).

Federal water quality criteria also have been published by EPA pursuant to Selection 304(a) of the Clean Water Act (33 U.S.C. Section 1314(a)). Of particular relevance to the New Bedford Harbor site is EPA's "Ambient Water Quality Criteria for Polychlorinated Biphenyls" (EPA Doc. No. 440/5-80-068, October 1980). This document sets criteria for chronic and acute exposure of saltwater aquatic life to PCBs (0.030 ug/l and 10 ug/l, respectively), but EPA has repeatedly acknowledged

that the PCB AWQC is focused on determining a safe PCB residual level for human consumption, not aquatic toxicity.[12/] The AWQC for PCBs "is a scientific entity, based solely on data and scientific judgment;" it is not a water quality standard "and in itself has no regulatory effect" (EPA, 1980).

The DWPC Surface Water Quality Standards make specific reference to federal AWQCs. In Section 4.03(2) of those regulations, it is provided that DWPC "will use EPA criteria established pursuant to Section 304(a)(1) of [the Clean Water Act] in establishing case-by-case discharge limits for pollutants not specifically listed in these standards but included under the heading 'Other Constituents' in 314 CMR 4.03(4), for identifying bioassay application factors and for interpretations of narrative criteria." PCBs fall into the category of "other constituents" under these regulations, and thus, Federal AWQCs for PCBs are properly considered only "as guidance" in the application of the SWQSs for establishing pollutant discharge limitations.

12/ In 1989, AVX's RFAs also addressed some very fundamental problems in relying on the PCB AWQC even as a measure of "safe" levels in biota from the human health perspective. The PCB AWQC was published in 1980 and fails to take into account extensive new scientific information. AVX specifically incorporates herein and relies upon its 1989 RFAs concerning the AWQC, including without limitation, RFAs 30,000-30,035, as part of these comments.

EPA has erroneously identified DWPC SWQSs as "applicable" to the remedy within the meaning of Section 121(d)(2)(A)(ii) of CERCLA (42 U.S.C. Section 9621(d)(2)(A)(ii)) and erroneously has stated that the DWPC regulations incorporate the Federal AWQC for PCBs as a regulatory "standard" for Massachusetts surface waters. As AVX has previously pointed out (Balsam 1989, § 5.2.2) Massachusetts Surface Water Quality Standards are not legally "applicable" to the remedy since neither the remedial action contemplated nor the circumstances at the site "satisfy all of the jurisdictional prerequisites" of the regulatory requirements that incorporate the SWQSs, such as the discharge permit program of 314 CMR 3.10. See EPA Interim Guidance on Compliance with Applicable or Relevant and Appropriate Requirements (July 1987) at 2. Compare Section 121(d)(2)(A) and (d)(2)(B)(i) of CERCLA (42 U.S.C. Section 121(d)(2)(A) AND (d)(2)(B)(i)) (referring to AWQCs as potentially "relevant and appropriate"). Rather, as most recently stated in the SFS, Massachusetts Surface Water Quality Standards are no more than a "regulation [which] sets standards for maximum levels of contaminants that can be discharged to the surface waters of the Commonwealth." SFS at 6-11.

Further, to interpret the DWPC regulations as elevating AWQCs to the status of regulatory standards not only is inconsistent with the express language of the regulations referring to AWQCs "as guidance," but also with the mandate of the regulations that "[i]n interpreting and applying the

minimum criteria in 314 CMR 4.03(4), [DWPC] shall consider local conditions including ... temperature, weather, flow and physical and chemical characteristics ...." (314 CMR 4.02(1)). Given the comprehensive studies of this site undertaken by EPA and potentially responsible party (PRP) consultants, analysis of the surface water quality impacts of any remedial scheme without reference to other site characteristics clearly would be imprudent.

Thus AWQCs are not properly considered fixed and invariable "ARARs" for this site. Even if viewed as a guidance which should be considered, however, consideration of AWQCs does not raise questions concerning the overall protectiveness of the January 1992 proposed remedy because under either the 50 ppm cleanup scenario proffered by AVX in 1989 or EPA's 50 ppm scenario, AWQC compliance will be achieved. The proposed 10 ppm Bay cleanup does nothing that will not be accomplished by a 50 ppm cleanup.

B. The Impact of Dredging and CDF Construction Has Not Been Properly Evaluated.

Both the FS-EST/H/BAY and the SFS identify §404 of the Clean Water Act and a host of other federal and state wetlands laws, regulations and Executive Orders as location-specific ARARs. By letter of November 1, 1990 from Helen Waldorf to Mary Sanderson, the Commonwealth of Massachusetts emphasized this point as did Mr. John Darwin, one of the commentators at

the March 5, 1992 public hearing. Mr. Darwin is a member of the Fairhaven Conservation Commission and he expressed grave concern that EPA was not complying with Massachusetts wetlands laws.

Mr. Darwin's concerns are well-founded, because it is painfully clear that the impact of EPA's intrusive dredging/CDF remedies on healthy productive wetlands has not been addressed in compliance with ARARs. A common feature of wetlands regulations is a mitigation requirement but EPA's feasibility studies are devoid of <u>any</u> discussion of proposed mitigation. Moreover, EPA has failed even to quantify future potential loss due to erosion and slumping of wetlands. As the technical comments below comparing capping and dredging make clear, EPA fails to present fully, and to take into account, the extreme adverse environmental impact its proposed remediation will have on highly protected, healthy and productive wetlands.

C. <u>Sediment Quality Criteria Are Not ARARS</u>.

Sediment Quality Criteria ("SQC") are <u>not</u> federal criteria which are applicable, relevant and appropriate, or even "to be considered" in ARARS guidance. As explained below in AVX's comments on the Baseline Ecological Risk Assessment, there are many sound reasons why the concept of equilibrium partitioning (which is the basis for SQCs) is fraught with difficulty when applied to real world situations. These concerns are further

amplified in comments submitted to EPA by the Chemical Manufacturers Association, the Utilities Solid Waste Activities Group, and the National Electrical Manufacturers Association dated March 2, 1992, which are submitted as part of AVX's comments (see Attachment F.1.). It is unclear, therefore, why EPA refers to various interim SQCs in the two feasibility studies and supporting documents.

IV. THE ADMINISTRATIVE RECORD IN THIS CASE IS INCOMPLETE

As part of these comments on EPA's further proposed remediation for New Bedford Harbor, AVX expressly relies on the technical data and arguments previously made in comments submitted by AVX and the joint defendants over the course of these long administrative proceedings. AVX understands that all comments previously submitted by AVX and the joint defendants are part of the present record.

RFAs.

In addition, AVX is also submitting a substantial number of its extensive and detailed Requests for Admissions (RFAs), prepared and served on the United States and the Commonwealth of Massachusetts in June, 1990 as part of these comments. EPA has categorically stated that none of the RFAs should be considered part of the Administrative Record. Before explaining why the RFAs selected and included in Attachment E

are an integral part of AVX's comments, it should be explained exactly what "RFAs" are.

During the course of the related litigation which AVX has now settled with United States and the Commonwealth of Massachusetts, the United District Court (Young, J.) issued an order requiring that each party set forth, in the form of a request for admission, each and every fact and opinion, including expert facts and opinions, which it intended to rely upon at any trial in that action.

AVX filed and served its RFAs in June, 1990 in connection with the upcoming trial scheduled for September, 1990. This trial implicated involving not only liability issues, but also causation of natural resource injury. This latter topic addressed issues which are largely coextensive with the issues EPA is considering in connection with the selection of its remedies. To give just a few examples, AVX's RFAs explored such questions as what the field sampling showed and whether it was reliable, what is PCB fate and transport in the New Bedford Harbor system, is there natural capping, is there natural biodegradation, are there PCB toxicological impacts on fish and birds, and are wetlands impacted by PCBs. The RFAs also contained AVX's detailed critique of numerous studies upon which EPA relied as proof of natural resource injury and upon which EPA relies today as the basis for remedial decision-making. In addition, AVX RFAs introduced new data and studies,

such as studies by Dr. Louis Thibodeaux following up on his 1989 work on PCB fate and transport, as well as additional research and sampling used by AVX's experts own experts on biodegradation.

As all parties to the litigation are aware, compliance with court orders to file RFAs required a tremendous amount of time and money. It is extremely unfair to AVX Corporation to exclude this information. In effect, the government's refusal to consider the technical data contained in the RFAs in that format effectively places on AVX an unnecessary and inappropriate burden to rewrite and reformulate technical data that the government well understands. There is no question that the government is very familiar with the technical information contained in the AVX 1990 RFAs as they were required to respond and reply to them. In addition, expert depositions of both the government and AVX experts had commenced prior to the time settlement was reached in late August of 1990. Government experts, therefore, were prepared to address AVX's theories in response to cross examination and likewise, the government was prepared to cross examine AVX's experts. Although the occasional RFA may be framed in an argumentative format which the government may deem inappropriate to respond to as a comment, the government will have little trouble discerning the points of information which AVX is trying to convey through its RFAs.

The government has given two principal reasons for its categorical rejection of AVX's RFAs for inclusion in the administration record. First, EPA says that the AVX RFAs do not "form a basis for the selection of a response action." July 9, 1992 letter from Assistant Regional Council, Mark Lowe to Mary Ryan. That the government did not rely on the AVX RFAs is not the point. Rather, AVX believes that the facts and opinions set forth in the RFAs in Attachment E contain information which EPA ought to take into account. They are part of AVX comments which the government is required to consider and to which the government is required to respond. It is no more burdensome for the government to treat the RFAs as part of these comments than it has been for AVX and the public to treat the hodge-podge of studies the government has churned out since 1982 as constituting a "remedial investigation report."

Second, the government says that the key issue in the 1990 scheduled trial was liability. This ignores the significant role causation of natural resource injury played in the trial preparation. The AVX RFAs do contain extensive information relevant to the selection of response action and AVX has been selective in its inclusion of RFAs to insure that only those RFAs addressed to expert technical issues that are also at issue in selection of remedial action are being resubmitted as comments. In the summary below of RFA topics, AVX points out the obvious connection between the topics covered and the remedial action decisions currently under consideration by EPA.

AVX's right to comment is one of the public participation requirements found in CERCLA, § 113(k)(2)(B)(ii), 42 U.S.C. § 9613(k)(2)(B)(ii). In addition, in this case Judge Young expressly stated that his decision on the scope and standard of review was based on the assumption that EPA would work with the defendants to develop a full administrative record that would adequately address all of the defendants' concerns. If EPA chooses to ignore the information set forth in the attached RFAs, it will violate AVX's statutory rights and AVX reserves its right to seek redress at any appropriate time.[13/]

What follows is a summary of the technical issues covered by the RFAs included in Attachment E (though some of the RFAs included in the separate RFA attachment are described elsewhere in these comments). RFA attachments referenced in these selected RFAs are not included in Attachment E but are being separately submitted today as part of these comments; backup quality assurance/quality control ("QA/QC") documentation (including chromatograms) was previously produced to the government and is identified in the RFAs by the document control (Bates) number.

1. Volume I, Facts. The only RFAs from this volume which are included in Attachment E are RFAs 7194 to 7212. These concern Hurricane Carol of 1954, which AVX experts believe

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13/ AVX also specifically relies upon the arguments made in the joint defendants' October 16, 1989 comments about the Administrative Record. See pp. 1-34 to 1-37.

played a significant role in the fate and transport of PCBs in the upper Acushnet River estuary. The 1954 hurricane impacts also provide an excellent example of storm impacts on the hydrodynamic regime in New Bedford Harbor.[14/]

2. Volume II, Experts. This volume provides curriculum vitae to establish the credentials of AVX's expert consultants who contributed to the technical RFAs or who performed field work or laboratory analysis. It is appropriate to include such information in the record to assist the reviewer in determining the weight to be accorded AVX's comments.[15/] Volume II RFAs also briefly outline the substance of each expert's contribution to the expert RFAs.

3. Volume III, Sampling and Analysis. These RFAs review an extensive amount of the sampling data that the EPA has generated as part of the administrative proceedings since 1982. In one way or another, all of the sampling data has played a part in the remedial decisions taken to date and in the remedial decisions presently under consideration for the Estuary, Lower Harbor and Bay. In addition to critiquing specific sampling programs such as the NUS/GZAD Grid Sampling Program, the RFAs review data included in substantive reports such as the flux data relied upon by Alan Teetor in one of the government's key transport studies (cited in the FS-EST/H/BAY) or the sampling data relied upon by Russell Bellmer in one of the principal wetlands evaluation reports relied upon by the government in the same feasibility study.

Some of the data critiqued by AVX in the sampling and Analysis RFAs includes data collected prior to 1986. Although EPA has finally acknowledged the deficiencies that AVX and the other defendants have pointed out for many years (SFS, pp. 2-5, 6; Addendum Proposed Plan, p. 5, note 3), nonetheless, it is still important to be certain that a full critique of the data is in the record because key studies -- the Battelle model, the Baseline Ecological Risk Assessment, and even the human health risk assessment -- are all based on the now-rejected sampling data, as is

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14/ AVX's evaluation of Hurricane Carol was also described at length in expert affidavits filed in related insurance proceedings. Because of the significance of Hurricane Carol these affidavits are also included in the record (Attachment F.5.).

15/ For this reason, updated resumes for contributors to these 1992 comments are included in Attachment G.

the Hot Spot ROD and the FS-EST/H/BAY. As an assistance to the reader, a succinct summary of AVX's critique of the government sampling has been inserted into the RFA attachment proceedings the RFAs taken from Volume III.

The RFAs also provide the necessary background information for EPA to also review the validity of sampling data which AVX has collected and seeks to introduce into the record. Finally, the RFAs detail some of the sampling data with respect to metals and PAHs.

4. Volume IV, Fate and Transport. These RFAs contain evidence AVX intended to use at trial on the critical issue of transport and migration of sediments and contaminants at New Bedford Harbor. This is the subject of § 2.4 of the FS-EST/H/BAY. This evidence is equally relevant to the correctness of many of the conclusions drawn in the FS-EST/H/BAY, the SFS, and the numerous underlying studies.

The following are examples of the issues addressed by the Fate and Transport the RFAs which are relevant to today's remedial decisions.

- In numerous studies EPA has calculated PCB flux from the upper estuary as the basis for the baseline no action scenario. AVX's RFAs detail extensive evidence to prove EPA's estimates or measurements are wrong.

- AVX conducted its own study of PCB flux from the upper estuary, which included collection of field data. This report and the underlying data are presented in the RFAs. (To avoid duplication, RFAs which did no more than recapitulate reports are not offered as part of these comments, though the underlying report is submitted herewith. Thus, the ASA transport study report is offered, but the RFAs presenting that report are not.)

- EPA says natural capping is not a significant transport process and that it will have little or no effect in stopping PCB transport. AVX's evidence to the contrary is set forth in its RFAs.

- Further evidence concerning capping as a remediation technology is presented, including a report on a bench-scale simulation of capping effectiveness and more thin-layer sampling.

- In connection with the pilot dredging study, various government agencies collected extensive sampling information and performed a variety of toxicity testing. Among these were in situ aquatic toxicity studies or mussel physiological studies conducted in conjunction with the Pilot Dredging Monitoring Study (Appendix VII and VIII to

the Decision Criteria for Pilot Dredging and Disposal Study dated April 25, 1988), which showed no toxic effects on biota exposed to harbor waters and sediment. These conclusions are described in the RFAs. AVX also used data from the pilot dredging preoperational monitoring program to arrive at independent flux calculations; these calculations are presented in the AVX RFAs.

° AVX's review of the final report of a field study of circulation and dispersion in New Bedford Harbor is included.

5. Volume V, PCB Transformation. Although EPA has consistently rejected in situ microbial dechlorination processes as significant to remedial decision making, AVX has produced extensive evidence to the contrary not only in its 1989 comments, but also in the attached RFAs. These RFAs are limited to those which supplement the work presented by AVX as part of its 1989 comments.

6. Volume VI, Human Toxicology. No RFAs on this subject have been included because they were based on reports previously submitted to the Administrative Record; one table (Table 2) which was specifically prepared for the RFAs has been included in Attachment E.

7. Volume VII, Ecological Impact. These RFAs contain extensive information which AVX believes should be considered by the agency. The topics covered include background PCB concentrations, background on histopathological conditions in New Bedford Harbor, information on benthic species composition of local New England estuaries, information on population dynamics, and information on bioaccumulation. This latter topic includes extensive critiques of toxicity testing upon which the agency purports to rely in its study of the harbor, as well as biota sampling and wetlands evaluations. Finally, this section also includes AVX's evidence responding to EPA's contention that there have been toxic effects on birds as a result of the PCB contamination of New Bedford Harbor.

8. Volume VIII, Natural Resource Injury. AVX has included all the RFAs and attachments concerning natural resource injury (including baseline conditions showing injury from contaminants other than PCBs) only because a review of the EPA decision documents indicate that EPA is taking into account natural resource injury as a basis for its remedial action objective of improving perceived ecological risks in the harbor. The Baseline Ecological Risk Assessment defines ecological risk to include "the resource value of any species being reduced as a result of contaminant body burdens." That risk, in addition to direct risk to the ecological receptor, has been considered

in the baseline ecological risk assessment. Baseline Ecological Risk Assessment, p. 1-6. As the authors later state, "[a]nthropogenic effects can alter the resource value of the harbor (i.e., recreational, food and asthetics". Id. at 4-32. The FS-EST/H/BAY also indicates that the NOAA's Damage Assessment Report provided information used in the baseline public health risk assessment. FS-EST/H/BAY, 3-3.

9. Volume IX, Mitigation of Future Injury. No RFAs from this volume have been included because the underlying reports are already part of the Administrative Record. However, certain of the Volume IX attachments were not previously submitted (Q.IX.0012, 13, 14, 15 and 17) and are included in these comments to support AVX's earlier presentation.

10-13. Volume X-XIII. RFAs included from these volumes concern the PCB AWQC and the history of the fishery closure. Their relevance to these comments is explained elsewhere.

14. Volume XIV, Other Sources. No RFAs from this volume have been included.

Government Sampling and QA/QC Data

AVX has requested the inclusion in the Administrative Record of all QA/QC data. See correspondence between EPA Project Manager Gayle Garman and Mary Ryan dated June 29, 1992 and July 8, 1992, Attachment F.2. Experience has shown that in order to determine the validity of data of the type requested, it is imperative to review all of the backup documentation, including such items as control and quality assurance documentation, chain of custody forms, laboratory calibration information, and chromatograms.

Despite years of controversy over sampling validity, and NCP regulations which specifically list QA/QC data as the type

of information usually to be included in the Administrative Record (NCP § 300.810(a)(1), 55 Fed. Reg. 3860 (1990), EPA continues to rely upon summary data reports even though it does not have any basis whatsoever for judging the validity of that information. This is the antithesis of good science. A simple summary of sampling results, such as that set forth in Appendix A to Volume I of the FS-EST/H/BAY, is useless without further data. For example, the United States Coast Guard sampling referenced in the FS-EST/H/BAY was rejected by the court as unsuitable for use in trial as evidence. See Transcript of Pretrial Conference, February 7, 1990, pp. 58-60, Attachment F.7. The underlying QA/QC data, as well as a summary report, must be included in the record with respect to all sampling data.

The Site File.

Not all documents created or obtained by the government concerning the New Bedford Harbor superfund site are contained in the Administrative Record. Rather, EPA has excluded from the Administrative Record certain materials which it has instead segregated into what is known as the "Site File." EPA has declined to permit AVX to obtain access to the Site File. Such action on the part of EPA is arbitrary and capricious for two reasons. First, there is absolutely no question that in response to FOI Request No. 1-FOI-769-89 made by the AVX and the other joint defendants in 1989, EPA permitted AVX access to

the Site File. See attached 1989 correspondence between Assistant Regional Counsel, Charles C. Bering and Rebecca Bryant, Esquire (Attachment F.2.). AVX had no reason to expect in 1992 that EPA would reverse its previous policy; EPA is aware of its inconsistent position but nonetheless still refuses to allow AVX access.

If EPA is not required to make the Site File documents available, it will be tantamount to permitting EPA to use the Site File to "hide" documents which are highly relevant to EPA's selection of remedial action, but which show that EPA's process is arbitrary and capricious. For example, in 1989 AVX learned of, and subsequently included in the Administrative Record, a memorandum from Joseph Yustead, the project manager of NUS (the contractor who was in charge of the EPA RI/FS for a number of years) that frankly disclosed his view that "the only way that our study is worth anything would be if we can issue a ROD that establishes that 'dredging' is the only alternative!" It appears that after ten years of study Mr. Yustead's objective has been realized.

This example illustrates that, there is no way that AVX can formulate meaningful requests for document from the Site File -- it is impossible to ask for a document which you do not know

exists.[16/] Therefore, in 1989, AVX requested that the Site File also be made part of the administrative record.

Certainly at the very least AVX should be given access to the Site File so it can determine whether any of the documents contained therein are in fact relevant to the EPA decision making process. If EPA reconsiders its position on this issue, AVX requests an additional period of 15 business days to submit supplemental comments after access to the Site File is granted. If EPA does not reconsider its position, AVX believes that the burden is on EPA to specify each document, or at least each specific category of documents, which EPA claims to be exempt with reference to the particular statutory exemption or exemptions on which its claim is based and the specification of the basis for the claim of exemption. See, e.g., RCA Global Communications, Inc. v. Federal Communications Commission, 524 F. Supp. 579 (1981).

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16/ A copy of EPA's generic "Region I Superfund NPL and Removal Site File Structure" is included as Attachment F.7. to illustrate the breadth of subject matter covered by the Site File and hence, the impossibility of framing a specific request.

I. BATTELLE HYDRODYNAMIC MODEL AND HYDROQUAL FOOD CHAIN MODEL

BATTELLE HYDRODYNAMIC MODEL AND HYDROQUAL FOOD CHAIN MODEL

By the use of the results of the Battelle hydrodynamic model and the Hydroqual food chain model to form a basis for setting an action level at the New Bedford Harbor Superfund Site, the United States Environmental Protection Agency (EPA) has set a new standard for obfuscation and a nadir for misuse of taxpayer's money. Over eight years have elapsed since EPA decided the appropriate way of developing a foundation for decision-making with regard to the New Bedford Harbor Superfund Site was to embark upon an ambitious and very expensive study which involved the collection and analysis of a variety of chemical, physical and biological data to serve as a data base for developing models of the estuarine and nearshore dynamics. These models were intended to serve as comprehensive tools to evaluate the system's response to a variety of cleanup scenarios (Battelle). This included prediction of PCB flux rates within and from the estuary/harbor as well as prediction of the response by representatives of the food chain.

The results of this effort and its application to the decision-making process at the New Bedford Harbor Superfund site would be amusing, indeed, if it were not for the fact that, in spite of a number of deficiencies in the development and implementation of the models, EPA has relied upon the results of the modeling effort to make decisions that have serious and expensive ramifications. Not only did the effort take five years longer than promised and not only was it ridiculously expensive but it also contributed little more to our understanding of the problem or to our objective decision-making abilities than would have been accomplished in one day with a room full of scientists and engineers using their experience and intuition. While EPA would have us believe otherwise, it is clear from the results of the model (or should we say lack of results) that EPA's attempt at attaining their objectives in this modeling effort was a dismal failure.

A useful beginning in understanding the full extent of the model's deficiencies is gained from a comparison of what was promised in the Remedial Investigation/ Feasibility Study (RI/FS) Work Plan for the Hydrodynamic and Food Chain models (Battelle 1984) with what was ultimately delivered (Tables 1 and 2).

As one can see there were very few objectives for the hydrodynamic model achieved. Although performance on the food chain model appears to be better, one must realize that the food chain model depends upon the ability for the hydrodynamic and transport model to represent reality. So while the results of the food chain model may seem reasonable on a "stand-alone" basis it doesn't necessarily reflect what may occur in New Bedford Harbor under the scenarios EPA hypothesized.

BATTELLE HYDRODYNAMIC MODEL

Although Battelle discussed the deficiencies in their efforts (Battelle, 1990), EPA nonetheless knowingly decided that it would use these uncalibrated and unverified models to quantitatively assess the change in PCB water column and sediment concentration levels as well impact on the food chain under various remedial alternatives. Based upon the results of the model, EPA developed action levels for dredging contaminated sediments.

In the Proposed Plan, however, EPA (1992) "recognizes the limitations of the model and cautions that the model estimates should not be viewed as absolute predictions. Rather the model estimates should be used to provide a framework for assessing the relative performance of various remedial actions on a qualitative basis." Although this apology is apparently offered with the hope that it will engender a leap of faith to believe, at least partially, the model results, it still greatly overstates the model's utility, even for comparative purposes. In fact, the failure of the model to capture even the most basic known features of PCB

transport and distribution in the estuary/harbor makes it almost worthless as a tool.

The authors greatly overstate the level of model validation when they say that they are unable to "rigorously validate" the model. In fact, the model is so poorly calibrated/validated that for all intents and purposes it is uncalibrated/ unvalidated. Probably the simplest validation case for a hydrodynamic model is applicable to tidal forcing. Typical errors between predictions and observations should be 3-7% for surface elevation and 10-20% for tidal current velocities with phasing errors on the order of several to tens of minutes. Battelle's calibration exercise showed current velocities in error by 100% or more and the phasing off by several hours.

In calibrating a transport model another simple case to perform is the dilution and dispersion of a conservative tracer introduced at a known location and rate. Battelle's attempts to validate the model using ASA's (1987) dye experiment were dismal. Battelle overestimated the flushing time by at least a factor of 2. They didn't present a comparison of model predictions to the dye concentration versus longitudinal distance down estuary and hence the reader has no idea whether the model reproduced the concentration field. One might guess that it is probably not good given the concluding statement found on p. 5-95.

> "Although computed and measured concentration behavior differs significantly in many ways, the discrepancies are most likely the result of using generic hydrodynamics in the calculations and the possible influences of the unmodeled river run off event."

Given this lack of validation for this very simple case, one has little faith that when the model is applied to a significantly more complicated situation involving far more complicated tidal and wind forcing and PCB/sediment dynamics that the results are correct.

Nor do the authors provide any justification for their assumption that while the model is not calibrated/validated, it can be used as a comparative analysis tool. The model should have been applied to the historical record to show that it could reproduce the observed build up of PCBs in the sediments and correctly predict the observed distribution pattern.

Of the many specific problems associated with the development and application the model the following gives some perspective of the lack of scientific rigor and practice.

- The application of the Battelle hydrodynamic and sediment transport models (Tempest/Flescot) to marine systems has been minimal. In fact the implementation of a time varying free surface was a new feature added to the hydrodynamic model to perform this study. It is probably not surprising that the attempt to apply it to a shallow estuary was doomed to failure.

- The model as applied to the New Bedford Harbor site is effectively two dimensional, vertically averaged in the upper estuary, (north of Coggeshall Street Bridge) because only one grid layer was used in the vertical.

- The model spatial (horizontal) resolution is generally quite poor and inadequate to represent the topography in mid to upper New Bedford Harbor. The authors note that poor grid configuration and coarse resolution are the primary reasons for poor model calibration. For example, wind driven flows are poorly represented because of poor grid resolution.

- The Battelle model completely ignores transport processes within the sediment. These processes are important since mixing and sedimentation processes strongly influence the transport of PCBs from the sediments to the water column.

- The model assumes a continuously, well-mixed, 4 cm thick bed layer. This approximation overestimates the releases of PCBs to the sediments because the actual mixing times in the top 4 cm are <u>not</u> instantaneous. The technique also minimizes the natural capping effects due to sedimentation which are known to be occurring in the harbor.

- There is no definition given or procedure outlined to determine when cohesive versus noncohesive sediment erosion and deposition formulas are used.

- Calibration of both the hydrodynamic and transport models is extremely poor. As an example, errors in tidal current speeds are often larger than the maximum tidal currents. The temporal behavior of the model predicted tidal currents also show little resemblance to observations.

- The model is unable to reproduce the flushing time of New Bedford Harbor observed in a large scale dye release program (ASA 1987). Model predicted flushing times are at least twice as long as were observed in the dye study.

- The model ignored wave-current interaction during the calibration phase but included this process in the application phase. This violates one of the most fundamental principles of model application.

- Assumptions employed either in the model's governing equations or in the application have precluded the ability to represent two layer estuarine flow and stratification.

- The model ignores the Acushnet River flow and density induced flows, hence it is impossible to simulate the estuarine circulation that dominates transport in the Upper Estuary.

- The parameterization of turbulence in the model is extremely simplistic and does not account of the principal sources of turbulence generation of stratification.

- The procedure and justification for specifying the tidal conditions at the open boundary of the model domain in Buzzards Bay are not specified.

- The sequence of hydrodynamic scenarios used to drive the sediment and contaminant transport models are totally contrived. They show little resemblance to actual conditions in the area. For instance, in most analyses of this type the modeler performs a sensitivity study to help determine how many scenarios are sufficient to achieve a desired level of accuracy. In the Battelle model they have assumed that there one normal wind plus tide scenario and one storm plus tide scenario. The storm is supposedly representative of a once or twice per month event. Battelle's wind records were not selected to necessarily represent prevailing conditions at the site (p.5-128). Battelle assembles a ninety-five (95) day record incorporating 31 days of normal wind plus tides followed by one day of storm winds plus

tides repeated three times. This record is used repeatedly for the long term simulations. As one can clearly see this procedure may correctly represent the mean tide condition but does a poor job of representing the variability of wind forcing. The approach, because of its structure, does not include any events with an occurrence rate longer than one month and hence misses all the important major storm systems (northeasters, hurricanes, etc.) which likely are more significant in determining net transport.

- As a result of these greatly simplified scenarios and the computation limits (see below), Battelle's methodology for extrapolating a 95-day model run to a ten-year projection is inappropriate and results in substantial uncertainty. The procedure used to generate long term model results is not supported by any reference to the literature or analysis and, while simple and convenient, ignores the variability in environmental forcing, e.g. wind, tides, river flow rates.

- Model computational times are not cost effective, e.g. it took 90 minutes on a Cray-XMP computer to run one tidal cycle in a fully coupled mode, and have had a significant negative impact on the scope and quality of the work. At that level of resource utilization it is probably no wonder that further work with the modeling effort was suspended and EPA essentially accepted an uncalibrated model as a deliverable.

- The selection of parameters used in calibrating the sediment contaminant models are arbitrary. Model calibration as a whole is extremely poor.

- There were insufficient field data to accurately calibrate and verify the hydrodynamic and sediment and contaminant models.

- The suggestion that even though the model lacks rigorous calibration it is acceptable to use as a tool to perform comparative analyses is without support either by reference to the literature or by independent analysis presented in the report. This "trust me" attitude is entirely inappropriate either scientifically or socially when so much is at stake.

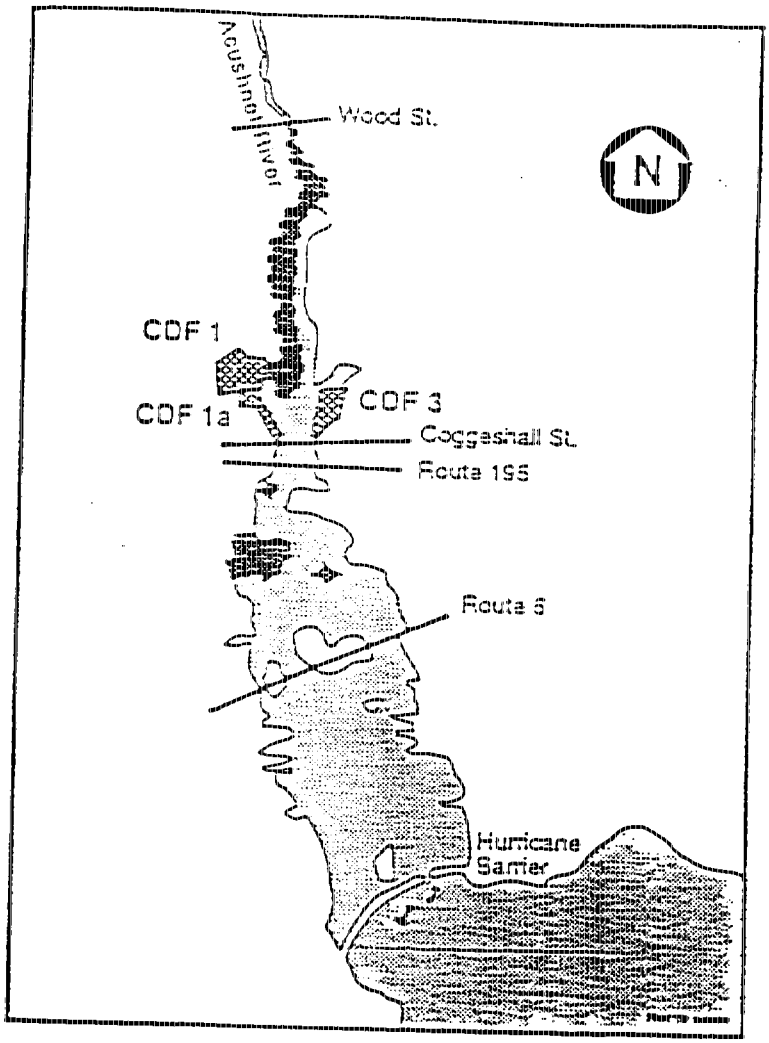
- The authors never present enough information nor provide the benefit of a sensitivity analysis to assess the concentrations predicted by the model. Where data are presented they are generally presented without regard to variability or significance of differences predicted. The lack of error bars on the figures in section 7-6.2., which summarize the substance of the report, render any legitimate comparison, <u>even a qualitative one</u>, an exercise in speculation. On the basis of what is presented, the "no action" scenario may well be as effective as any of the remediation scenarios

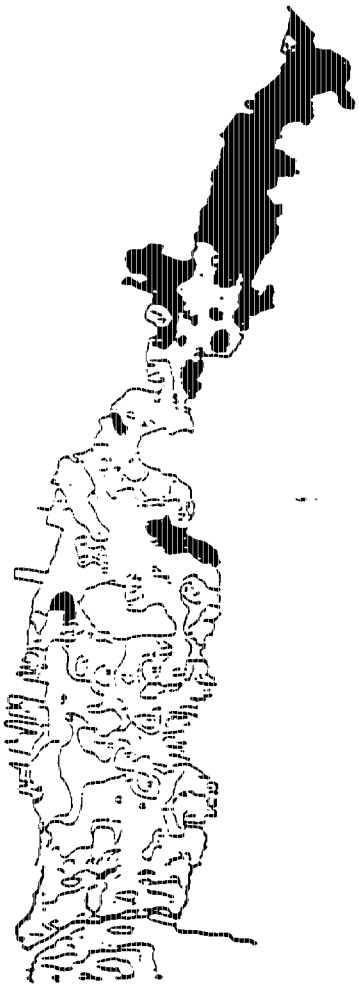
In the use of the hydrodynamic model EPA has evaluated PCB remedial action levels of 1, 10, 50 and 500 parts per million (ppm). EPA rejects the lower action levels because the volume of material, cost and environmental impact (disturbance) are large and rejects the upper level (500 ppm) because EPA feels that the remaining contaminated sediment poses a significant continuing impact (EPA 1992). Unfortunately, EPA does not evaluate the most interesting action level range: 50-500 ppm. As an example, if the action limit is raised to 100 ppm, dredging in the lower harbor (below Coggeshall Street) is unnecessary. This would save dredging 76,000 cubic yards over 47 acres or about one-third of the 232,000 cubic yards (118 acres) proposed to be dredged for a 50 ppm action level. The resulting PCB concentrations in the lower harbor, while likely higher than this 50 ppm case that Battelle modeled, would likely not be significantly increased because the areas in the lower harbor with concentrations significantly above 50 ppm are very small. In any case, EPA should redo the evaluation investigating the action level in the range of 50-500 ppm to further optimize the decision of action level.

It was very disturbing to note that the spatial location and size of areas to be dredged in EPA's preferred remedial alternative (EPA 1992, Exhibit 4, Page 15) (see figure 1) are inconsistent with the PCB surficial sediment distribution data used as input to Battelle's model (Battelle 1990, Page 5-17, Figure 5.7). This mismatch is most startling in the mid-harbor area. If in fact Exhibit 4 in the Proposed Plan is correct and Battelle's model (1990) has the wrong initial conditions, then the results of Battelle's modeling effort are of little use. If on the other hand, the sediment distribution used for Battelle's model is correct, then presumably there are some miscalculations in the Feasibility Study, upon which the Proposed Plan is based. Such fundamental inconsistencies strongly suggest that one hand (EPA) does not know what the other (Battelle) is doing.

Exhibit 4

Preferred Alternative for Estuary and Lower Harbor/Bay





Legend

CDF - Confined Disposal Facility

Areas to be dredged in estuary and lower harbor (>50 ppm PCBs)

FIGURE 1.

Left side: Areas to be dredged in estuary and lower harbor with PCB concentrations greater than 50 ppm.

Right side: Measured PCB concentrations in surficial bed sediments of New Bedford Harbor area with concentrations greater than 50 ppm (Battelle 1990).

HYDROQUAL FOOD CHAIN MODEL

The ability of the food chain model to predict the biological fate of PCB in the food chain accurately is highly dependent upon some of the results of the hydrodynamic model. This is a fundamental and fatal flaw because if the initial conditions which define the assumptions upon which the food chain model is operating are wrong, the results of the food chain model will also be wrong. In addition, there are a number of specific problems with the food chain model as well. Some of these are as follows:

- Terms having precise physiological meaning are not defined throughout the text. For example, "assimilation efficiency" is used to refer to the fraction of ingested food converted to biomass by an organism (the physiological meaning of the term) and to assimilation of contaminants. The latter process is sometimes called "chemical assimilation efficiency" in the text. The term "excretion rate", physiologically the rate of elimination of liquid waste, is used to describe what is really a depuration rate. Hence, the document is difficult to make sense of, even for a physiological ecologist. It must be impenetrable to a lay person.

- Averaging procedures applied to water column and sediment contaminant concentrations are vague and appear to be arbitrary. For example (boldface added):

  > "The data were first screened to determine specific stations or data points that would incorrectly bias an area average. This judgment was made by **visual inspection** of log normal probability distributions of the PCB and metals data from the four cruises. Data points that deviated **significantly** from the distribution indicated in the plot (i.e., values **that were either unreasonably high or unreasonably low) were not** included in subsequent data averaging." This is an extremely vague statement, with no stated and objective criteria for statistical significance. The terms **unreasonably high and unreasonably low are not defined.** What is "unreasonable" variation? One standard deviation? Two standard deviations? Points which fall outside of the 95 percent confidence interval?

- Within-cruise averages of water column and sediment toxicant levels were computed for each area. Areas were then averaged over all cruises. This

procedure is justified by the undocumented statement that the biota are not sensitive to short-term variations in exposure concentrations (page 6-13). The averaging process renders homogeneous distributions of material that obviously vary spatially, and in the case of water column contaminants, temporally. Any time variation in contaminant levels is eliminated from the model by this process. Yet seasonal variation and episodic events such as storms may exert a profound impact on the dilution/distribution of contaminants in the New Bedford Harbor area (Table 6-5; area 1).

- The data are selectively and inconsistently sieved. For example, measurements of water and sediment contaminant concentrations from station 16 in area 2 (located near a wastewater treatment plant) are higher than at other stations in area 2. On the basis that they are higher, they are not included in the area 2 average. Why not? The target apex predators, lobster and winter flounder, are motile organisms and can move within and between areas.

- Water column and sediment contaminant concentrations are not presented in the same manner and are thus difficult to compare. The water column data are given more or less in their entirety. The sediment data are not presented at all. The text states that these data are presented in Appendix I (Battelle, 1990) . However, Appendix I contains only probability plots. Without being able to examine the unaveraged data from the individual cruises, it is not possible to get a feeling for the variance, which is, presumably, large.

- Field sampling of the biota is woefully inadequate. For example, the numbers of flounder sampled are ludicrously low, less than 5 fish per age class in each area (Figure 6-17). Because so few fish were sampled, presenting Figure 6-19 (percent of total caught per age class) is totally meaningless and misleading.

- A great deal of effort (and presumably money) was expended on experiments which measured rates of chemical assimilation efficiency and excretion/depuration directly. However, the measured values (which are extremely variable) are not employed in the model but are used only as "guidance for model calibration" (page 6-63). Instead, literature values for unrelated, mostly non-marine, species (carp, sandworms, rainbow trout, goldfish and guppies) are used in the model without justification.

- The measured chemical assimilation efficiencies are not presented in the text. The reader sees only whole body concentrations. The measured rates are said to be discussed in Chapter 6-2. They are not. The chemical

assimilation efficiency rates finally appear in Table 6-23, which presents only averages. The reader has no idea of the variance.

- The measured bioconcentration factors and excretion/depuration rates for PCB do not appear anywhere in the document. Table 6-20 gives values for metals only.

- Calibration of the food chain model is, overall, not adequate. The observed and calculated PCB concentrations for mussels, crabs and polychaetes do not agree well in areas 1 and 2. Points for mussels in area 2 are missing from a number of the figures (Figure 6-28 to 6-33). On the basis of the data presented, agreement with calculated values is, presumably, poor.

- No sensitivity analysis for the various food chain model parameters is present anywhere in the document.

- A steady state assumption for toxic uptake/release is applied to the lower tropic levels of the food chain model. No bases for the assumption are cited. A voluminous literature exists on selective uptake of dissolved nutrients by phytoplankton and bacteria. If the lower trophic levels discriminate in favor of, or against, contaminants, then contaminant turnover by the biota may be faster or slower than the steady state assumption dictates. This factor could be examined in a sensitivity analysis of the food chain model.

- The food chain structures are simplistic and some of the trophic links are incorrect (Section 6-4). Both crabs and winter flounder are bottom feeders. They do not consume phytoplankton except in their larval stages, which inhabit the water column. The juvenile and adult (i.e., post-larval) stages of these organisms are simply not constructed morphologically to feed on items as small as plankton. The structure of each food chain will affect the amounts of contaminants transferred to lobster and flounder: if the model has them consume even some plankton, they are likely to accumulate less contaminant than if they consume only benthic organisms. It is not clear whether the model includes larval stages of lobster and flounder in its size/age classes.

- The well documented temperature-driven on-shore/off-shore migration of adult winter flounder is ignored in the model. It is stated that the fish do not move far from the New Bedford Harbor area. In fact, they move at least as far as Nantucket Shoals (Howe and Coats 1975) during summer, a phenomenon which is certain to alter the environmental concentration of contaminants to which they are exposed. Likewise, lobsters, while

migrating less than flounder often move sufficiently to migrate in and out of areas where the sediment is contaminated (Fogarty, et al. 1980).

CONCLUSION

In seeking to apply the hydrodynamic and food chain models to the decisions facing EPA it is clear that the problems previously identified detract significantly from the model's ability to provide answers that are useful to EPA for decision-making. The coupling of the hydrodynamic model and the food chain model was a failure, at least in part reflective of a major problems in effective coordination between the two contractors. The authors themselves felt that coordination was a major problem, noting particularly their inability to directly interface the hydrodynamic and food chain portions of the models in area 3. In fact, the models even used different spatial representation in the Outer Bay area.

Once one takes in consideration more realistic assumptions the potential seems relatively high that the body burden limits (2 ppm) that EPA used as the target in the application of the Battelle model could be reached in a relatively short time under the no action scenario. These more realistic assumptions include:

1. Battelle's hydrodynamic data set precludes the largest and potentially most important storm events which may significantly lower PCB concentrations in the sediments. In doing so it essentially ignores the very relevant question of whether the majority of the PCB transport is caused by episodic events(storms) or daily tidal processes. This is not a minor point although one might argue that as long as transport is shown, the magnitude is unimportant. This is not true, however, since the proximal cause of transport is a significant determinant of the central questions that EPA had hoped to address with their modeling effort.

2. Battelle's model consistently underestimates the flushing rate of the estuary, by at least a factor of 2.

3. The model takes no account of natural capping (sedimentation) which is known to be occurring in the harbor (Summerhayes 1977) and isolating the contaminated sediments from the water column.

4. The model ignores seasonal migrations of significant fractions of the populations of the two apex predictors, lobster and winter flounder, away from the areas highest contaminant concentration. These types of migrations serve to lower the environmental concentrations of the contaminant to which the organisms are exposed, allow them to depurate and eventually lower their body burdens of the contaminant.

In summary, this modeling effort has been a substantial waste of time and money. The results of the comparative analysis for the various remedial scenarios could have been achieved at a fraction of the cost and time of the Battelle study by using a simple box modeling approach. In addition it would have permitted far more opportunity for detailed sensitivity analysis which is so crucial for developing models that are realistic and therefore useful for decision making.

LITERATURE CITED

ASA 1987. Selected studies of PCB transport in New Bedford Harbor. ASA 86-18. Prepared by Applied Science Associates, Inc., Narragansett, Rhode Island for Ropres and Gray, Boston, Massachusetts.

Battelle, 1984, RI/FS Work Plan for modeling of the transport distribution, and fate of PCBs and heavy metals in the Acushnet River/New Bedford Harbor/Buzzards Bay system, prepared for NUS Corporation, Pittsburgh, Pennsylvania, May 1984. (Amendment of July 16, 1984)

Battelle, 1990, Modeling of the transport, distribution, and fate of the PCBs and heavy metals in the Acushnet River/New Bedford Harbor/Buzzards Bay system, prepared for ESBASCO, Boston, Massachusetts, September 21, 1990.

Conover, R.J. 1978. Transformation of Organic Matter. pp. 221-499 in: Otto Kinne (Editor). *Marine Ecology. A Comprehensive, Integrated Treatise on Life in Oceans and Coastal Waters.* John Wiley & Sons, New York.

EPA. 1992. Proposed Plan for the Estuary and Lower Harbor/Bay. New Bedford Harbor, Massachusetts.

Fogarty, M.J., D.W. Borden and H.J. Russell. 1980. Movements of tagged American lobster, Homarus Americans off Rhode Island. Fish. Bull. 78:771-780.

Howe, A.B. and P.G. Coats. 1975. Winter flounder movement, growth and mortality off Massachusetts. Trans. American Fish Society. 164:13-29.

TABLE 1

HYDRODYNAMIC AND SEDIMENT TRANSPORT MODELING

Item	Promised in Work Plan	Actual Study
Schedule	6/11/84 to 8/26/85, Final Report due 9/85	6/11/84 to 9/21/90 (5 years late!)
Contamination parameters to be modeled	Four PCB isomers and three heavy metals (copper, cadmium, and lead)	Total PCBs, no metals
Budget	$382,964	??
Fluid characteristics to be modeled	Salinity and Temperature	Salinity and Temperature not included
Turbulence	Improvements of existing K-E turbulence model	No turbulence model, Eddy viscosity assumed constant
Cohesive sediment deposition and erosion formulation	Improvements on existing formulations by Krone & Partheniades	No improvements made
Bioturbation/Bed sediment modeling	Transport model to be modified to include effect of bioturbation of sediment and sediment-sorbed within seabed	No modeling of bed sediments included
Long term simulations	Composite tidally driven base case with typical wind driven cases	Tidal base case plus one wind driven (storm) case
Tidal simulations	Assume water surface is flat and use rigid lid approximation in hydrodynamic model	Abandoned rigid lid approximation because it was incorrect and implemented time varying free surface in model

TABLE 1 (continued)

HYDRODYNAMIC AND SEDIMENT TRANSPORT MODELING

Item	Promised in Work Plan	Actual Study
Remote imagery	Proposed to use Landsat 4 and 5 imagery to calibrate circulation model	No remote sensing data used to calibrate model
Wave	Proposed to calibrate wave model using US Army Corp of Engineers wave data at hurricane barrier	No calibration of wave model
Calibration	Proposed for summer (stratified), winter (no stratification) and storm conditions	No effective calibration. Model incapable of simulating stratified conditions (no density forcing)
Three dimensional (3-D representation)	Proposed to have full 3-D simulation	Vertical gridding strategy in upper bay allows only 2-D representation

References:

1. Work Plan:
   Battelle, 1984, RI/FS Work Plan for modeling of the transport distribution, and fate of PCBs and heavy metals in the Acushnet River/New Bedford Harbor/Buzzards Bay system, prepared for NUS Corporation, Pittsburgh, Pennsylvania, May 1984.

   (Amendment of July 16, 1984)

2. Model Result:
   Battelle, 1990, Modeling of the transport, distribution, and fate of the PCBs and heavy metals in the Acushnet River/New Bedford Harbor/Buzzards Bay system, prepared for ESBASCO, Boston, Massachusetts, September 21, 1990.

TABLE 2

BIOLOGICAL FOOD CHAIN MODELING

Item	Promised in Work Plan	Actual Study
Evaluation of Existing Data	Proposed to obtain best available information for: respiration rates, growth rates, migratory habits, food assimilation efficiencies, contaminant assimilation efficiencies, contaminant bioconcentration factors, and contaminant excretion rates for species being modeled	Sparse citations from the referenced literature, most quite dated.
	Proposed detailed review of available New Bedford Harbor data, including data analyses and syntheses, as required, to assess the characteristics of extent PCB and copper contamination in the New Bedford Harbor ecosystem	Yes
Application of biological food web model to New Bedford Harbor ecosystem	Development of food web structures for the species of interest	Yes
	Detailed literatures review of prey preference, feeding habits, respiration, age-weight relationships, and migration patterns of each species	Literature review not detailed and not current
	Model application	Yes

TABLE 2 (continued)

BIOLOGICAL FOOD CHAIN MODELING

Item	Promised in Work Plan	Actual Study
Preliminary calibration of food web model using existing information	Proposed to compare model calculated and observed organism contaminant concentrations	Yes
	Proposed to adjust model as necessary	Not done
Evaluation of the 1984-1985 field and laboratory data and detailed calibration of the food web model	Proposed detailed calibration of model	Detailed calibration not done
Evaluation of effectiveness of remedial action	Proposed to apply model to various remedial scenarios	Yes
Reporting and documentation of analysis	Proposed comprehensive final report	Yea

References:

1. Work Plan:
   Battelle, 1984, RI/FS Work Plan for modeling of the transport distribution, and fate of PCBs and heavy metals in the Acushnet River/New Bedford Harbor/Buzzards Bay system, prepared for NUS Corporation, Pittsburgh, Pennsylvania, May 1984.

   (Amendment of July 16, 1984)

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   Battelle, 1990, Modeling of the transport, distribution, and fate of the PCBs and heavy metals in the Acushnet River/New Bedford Harbor/Buzzards Bay system, prepared for ESBASCO, Boston, Massachusetts, September 21, 1990.

II. BASELINE ECOLOGICAL RISK ASSESSMENT

BASELINE ECOLOGICAL RISK ASSESSMENT

THE SCIENTIFIC BASIS FOR THE ECOLOGICAL RISK ASSESSMENT IS FLAWED

The Baseline Ecological Risk Assessment (EPA 1989a) purports to evaluate the degree to which several species of concern either endemic to New Bedford Harbor or residing there temporarily are at risk due to the presence of PCB and several metals. The framework for the risk assessment is the paradigm that ecological risk can be evaluated by choosing representative species, estimating to what degree they are exposed to contaminants and estimating, through reference to direct and indirect evidence, what the toxicological result of that exposure will be.

In concept, this is an appropriate and scientifically valid approach to resolving an important issue. In practice, however, because most studies of contaminant fate and transport at Superfund sites in the marine environment are so heavily biased towards addressing human health and engineering issues, there is such an embarrassing lack of ecological data developed that even the most fundamental questions about the impact or fate of these environmental contaminants to organisms that utilize the environment cannot be addressed.

We doubt that this could be any more conspicuously obvious than in the Baseline Ecological Risk Assessment, New Bedford Harbor Site Feasibility Study. This study has taken very superficial data resulting from a few poorly designed environmental studies and a progression of concatenated assumptions and reached conclusions, that, while they have little relationship with the real world, are bestowed with artificial reality because of a superficially complex and involved analysis. These conclusions are then used as the basis for developing cleanup criteria which involve the expenditure of millions of dollars. By the time

one gets to weighing the alternatives for remediation one loses sight of the only element of the process that is factual, i.e., the extremely limited and superficial data resulting from inadequate and substandard site-specific studies on the ecology of organisms inhabiting the New Bedford Harbor area and toxicological studies conducted in the laboratory, with different species, in substantially different media than is experienced in the field.

This approach cannot be excused on the grounds that the state of the art in ecological and toxicological studies is no further advanced than what we have seen here. This is just not the case. Scientists have studied many of environmental characteristics addressed here for decades and have a variety of valuative methods and tools to develop far more knowledge and resolution of the ecological dynamics and toxicological relationships than we have seen here. Most of these methods are contained in numerous EPA Guidance Manuals (EPA/OPP 1986;1987;1988;1989)(EPA 1987;1989) and other government agency manuals (Barnthouse et al. 1986) and are practiced regularly in the laboratories of the various resource agencies around the country. For EPA or their contractors to ignore the standard of practice used in their own laboratories and described in their own guidance manuals because of bureaucratic confusion is a travesty.

While EPA admits (p. 1-7) that "the consensus among professionals concerning the most applicable methods [for ecological risk assessment] at a particular site is limited" since the methodologies for determination of ecological risk are developing, this does not give them leave to disregard the basic underlying methods for conducting a scientifically credible environmental impact study that have been developed over the last 20 years.

It is clear from reading Section 1.3 of the Baseline Ecological Risk Assessment that EPA recognized a number of the weaknesses and limitations of the risk evaluation that they presented in this Baseline Ecological Risk Assessment. EPA

characterizes this as a baseline risk assessment because it is based upon their interpretation of the existing conditions in New Bedford Harbor. EPA indicates that "the potential natural decrease in contaminant mass and concentration in the harbor due to transport and degradation through time is not considered." Nor is the impact of the ongoing remediation of the "Hot Spot" presented in any way. Although EPA indicates that there would be subsequent evaluations, presumably when the results of the hydrodynamic and food chain model are available, this is as yet an unrealized objective. Since EPA later says on page 2-18 that it is impossible to use the results from the food chain model directly in the ecological risk assessment, it appears that this objective will never be realized.

In an illustration of remarkably convoluted logic, EPA discusses (p. 1-7) that although the scientific community's realization that PCB congeners should be treated as "209 distinctly different chemical congeners" with different physicochemical characteristics including differential rates of biodegradation, uptake, volatilization, bioaccumulation and depuration, characteristics which are both not related to Aroclor characteristics and also are "not easily related to level-of-chlorination." (p.1-9), and then, although EPA realized this critical fact, it decided not to analyze PCB by congener, but by "level-of-chlorination" (homolog) arguing that congener specific analysis was too costly. EPA rationalized that analysis by homolog would "provide <u>valuable</u> additional information, and because physical behavior determining fate and transport of PCBs is relatively similar for each homolog group, quantification (and subsequent numerical modeling) by homolog was deemed a reasonable cost-effective analytical goal for the modeling program." As we subsequently learn, however, (p. 1-9): "It was later decided to model [hydrodynamic and food chain models] only <u>total</u> PCBs, and the modeling program data were later converted into total PCBs for risk assessment purposes by summing all homolog groups."

Because total PCB data was used in the Baseline Ecological Risk Assessment, "the first step in conducting a risk assessment [i.e., determining the concentration of the contaminant(s) of interest at the specified site] is not possible for PCBs at the same level of detail as for other environmental contaminants." (p. 1-10). Since many of the same analytical difficulties and uncertainties "apply equally to any toxicological studies conducted with PCBs," "various assumptions and simplifications were necessary at several points in the risk assessment." (p. 1-10) Acknowledging that it has been recognized for over ten years (Dill et al., 1982) that there is "substantial variability among congeners with regard to toxicity to aquatic organisms," (p. 1-10) EPA nevertheless failed to propose or conduct any site-specific water or sediment toxicity testing which they agree is "perhaps the best solution to this problem."

EPA concludes its discussion (p. 1-11) on the limitations and weaknesses of the ecological risk assessment by observing that "[the aforementioned] factors necessarily limits to some degree confidence in the accuracy of the risk probabilities for PCBs generated in this assessment." AVX believe that once the points discussed below are taken into account it will be seen that confidence in this risk evaluation is not only limited, but virtually eliminated.

EPA then attempts to legitimize its analysis by citing what it feels is corroborative evidence that conclusions it has reached are consistent with other studies conducted with New Bedford Harbor organisms. However, EPA has been selective in its choice of studies, for instance, failing to cite in situ aquatic toxicity studies or mussel physiological studies conducted in conjunction with the Pilot Dredging Monitoring Study (Appendix VII and VIII to the Decision Criteria for Pilot Dredging and Disposal Study, date April 25, 1988). These studies concluded that under current ambient conditions there was no toxic effect to test organisms that could be attributable to PCBs.

THE EXPOSURE ASSESSMENT IS BASED ON INFORMATION THAT IS SO INCOMPLETE AND ASSUMPTIONS THAT ARE SO SIMPLIFIED THAT THE RESULTS BEAR LITTLE RESEMBLANCE TO REALITY

According to Baseline Ecological Risk Assessment, the "extensive data base generated between 1981 and 1986 provides an accurate description of the current extent and level of contamination within the New Bedford Harbor area." However, it seems that this fact is disavowed in the Draft Final Supplemental Feasibility Study (EBASCO 1992) when EPA indicates that it will only use data from samples collected after 1985 since there was uncertain quality and inconsistency prior to that time.

In its effort to simplify the exposure analysis EPA made several compromises which, taken in concert, result in virtually eliminating any of the environmental variability that would be experienced by any species inhabiting New Bedford Harbor. In the first place EPA used a convention during data conversion which eliminated any "zero" value in the data (p 1-15) thus inappropriately biasing the average values upward. The sediment data were then log-transformed and vertically averaged for each cell, thereby dampening any natural variability in the data which would be experienced in the real world. This failure to reflect reality has the effect of relegating this effort to a hypothetical exercise.

Once this was completed, descriptive statistics were calculated and from this a probability density function developed. EPA explains (p. 2-7) that while the joint probability model upon which it bases its risk assessment assumes that the probability distribution of expected environmental concentration (EEC) and the effects distribution are normally distributed, this was not achieved in the EEC data. This problem, however, is glibly dismissed since the same log scale must be used in this risk analysis for both the EEC and the effects distributions.

While this process of data manipulation would have the desired result of simplifying the analysis, it does so at the cost of making the exposure model so fictitious that it bears little resemblance to the time- and space-varying world that would be experienced by an organism living in New Bedford Harbor. As is well accepted in the scientific community, changes in the physical-chemical environment can directly influence ecological risk by altering potential exposures (Bartell et al. 1992).

For estimating exposure of estuarine organisms to sedimentary PCB contamination, EPA relies upon the pore water concentration developed from surficial sediment concentrations via equilibrium partitioning (EqP) coefficients, using initial conditions for surficial sediment concentrations developed for the physical/chemical model and applying a partitioning coefficient based upon the tendency for PCBs to desorb from sediment particles under differing conditions of organic carbon concentration in the sediment. The fundamental problem with using the EqP approach relates to the various assumptions upon which it is based. In the first instance, the assumption of equilibrium is unlikely to occur in dynamic estuarine environments where physical processes dominate; second, organisms are not always exposed to contaminated sediments via porewater exposure as is contemplated by this hypothesis, but rather through direct ingestion; third, the bioavailability of PCB is dependent on a number of other factors and processes, other than relative fraction of organic matter (these factors include presence of humic and fluvic acids, presence of organic matter - clay complexes, solubility of the various PCB congeners, etc.); fourth, the assumption that organisms living in the water column have the same sensitivities that benthic organisms is untested; and fifth, site specific field verification of this method is lacking. The EPA's own Scientific Advisory Board (EPA 1990) was especially critical of the EqP approach, cautioning that not only is substantial verification of the assumptions of the EqP needed, but that, in the case of PCBs, the fact that the chronic ambient water is

based upon residual values rather than toxic effects makes the "entire process (of using the EqP approach) even less reliable."

Even if the EqP approach did adequately represent the exposure of benthic organisms in New Bedford Harbor, the simplifying assumptions used in the Baseline Ecological Risk Assessment, i.e., using total PCBs and only two values for the organic carbon concentration, significantly weakens the foundation of the analysis developed by EPA to make conclusions concerning the exposure experienced by organisms inhabiting New Bedford Harbor where the environmental variability of the distribution of PCB congeners and the presence of organic carbon in the sediments are substantially more complex.

EPA also discusses the relationship of the food chain model to the risk assessment although they conclude that it is "not possible to use the [food chain] model results directly in the ecological risk assessment" since there are few data available on the effect of "residue values" on biota. (p 2-18). It concludes that although dietary uptake is important to consider in order to reflect actual effects on aquatic biota "there are insufficient data to evaluate this pathway quantitatively." (p. 2-18) The inability to apply the food chain model to such fundamental issues as ecological risk is further evidence of the futility of all the overall effort and expense to develop a sophisticated model.

THE ECOTOXOCITY EVALUATION IS FLAWED BECAUSE IT RELIES ON INFORMATION THAT IS NEITHER SITE-SPECIFIC OR COMPOUND-SPECIFIC AND RELIES ON EXTRAPOLATIONS OF TOXIC EFFECTS THAT ARE UNREALISTIC

Because of the failure of EPA to focus its studies on objectives that would allow some credible evaluation of the real risks that might be experienced by organisms in and around New Bedford Harbor EPA has had to use a number of assumptions and extrapolations from studies conducted in the laboratory on unrelated species, and has used data from toxicity tests conducted under different exposure regimes with a variety of chemical compounds, many of which behave quite differently, pharmacokinetically, than specific PCB congeners.

As discussed by Bartell et al. (1992), single species laboratory toxicity assays were not designed with extrapolation [since this involves the unrealistic assumption that species in the laboratory will react to a contaminant in the same manner as those in the field] to the field as an ultimate objective .... these tests, by necessity, focus on taxa that are easily maintained under laboratory conditions and provide suitable numbers of individuals for statistical power in assay protocols." In addition, Bartell et al. (1992) observed that "testing protocols are designed to ensure that the main source of variance lies with individual response [of organisms] to the chemical, not with testing procedures. This makes for good statistics, but contributes inestimable uncertainties to extrapolating assay results to the ecological systems of interest." (Bartell et al. 1992).

EPA addresses some of these problems with the ecotoxicity assessment section. EPA acknowledges that "acute or final toxicity values are not available for PCBs in the Ambient Water Quality Criteria because acute values are often in excess of maximum solubilities; [that] minimum data criteria are not satisfied; and differing toxicities are demonstrated by the various PCB Aroclors and congeners." While EPA has referenced several laboratory bioassay studies as the basis for their

toxicity evaluation none of these bioassays have involved species that inhabit New Bedford Harbor. Furthermore, the laboratory studies conducted do not represent typical exposures in the environment where time- and space- varying exposure and numerous environmental mitigating factors may not only moderate the dose to endemic organisms but may cause a much different response as well.

In addition only a few of the studies referenced were conducted specifically to determine the maximum acceptable toxicant concentration (MATC), which is the standard chronic benchmark for survival, growth or reproduction (EPA 1986). Rather, EPA relies on studies which had other biological endpoints, ones which may or may not be representative of the same physiological response nor of the response in the environment. EPA comments that "few MATC data are available for marine organisms, and the research that has been performed is limited with respect to contaminant type and test organisms used. There are insufficient MATC data for PCBs to generate distributions for any of the taxonomic groups of interest." (p.3-6)

For those species without MATC data they have had to rely on extrapolations from existing endpoints (usually LC 50 values) from other organisms. In at least one case the organisms were not even in the same phylum, let alone marine. For example to develop a MATC curve for PCB and mollusks, two extrapolations were needed, first a relationship was established between LC_{50}s for a mysid shrimp and LC_{50}s for mollusks. Second, since there was no MATC data on mollusks, the relationship between marine fish LC_{50}s and marine fish MATCs was used to extrapolate from mollusk LC_{50}s to a mollusk MATC. Although EPA has attempted to confer credibility on this procedure and other extrapolations equally shaky by citing precedents, there is little doubt that they are far out on the scientific limb. As EPA itself, in a moment of understatement, commented "there is a large variance associated with this MATC due to the double extrapolation." Suter and Rosen (1986) observe in their study, upon which this extrapolation

method is based, that: "These extrapolations must often be made at very high taxonomic levels, and even when they can be made between closely related species, nontrivial uncertainties are involved." They also observe that: " the existing data, particularly that for chronic toxicity, cannot be said to be representative of the marine biota." EPA attempted to justify their approach by stating that, because of the limited amount of data available about the effects of PCBs and metals on marine organisms, the estimates of MATC...as used in this risk assessment have some uncertainty, which was quantified to some extent (emphasis added)." Despite this bow to unquantified uncertainty created by EPA's novel approach, the shaky foundations of the report remain.

In summary, the use of toxicity data for chemicals that have different acute/chronic ratios than PCBs, LC_{50}s corresponding to a wide range of exposure times, the use of total PCB toxicological data instead of congener data (an acknowledged compromise in scientific accuracy), and MATCs with a variety of biological endpoints, some not even ecologically meaningful, as well as taxonomic and procedural compromises that EPA has repeatedly made, further erodes the value of this evaluation and reflects negatively on the "weight of evidence" (p. 1-10) rationale upon which EPA rests its case.

THE RISK CHARACTERIZATION FAILS IN ITS ATTEMPT TO ACCURATELY QUANTIFY THE RISK TO ORGANISMS IN NEW BEDFORD HARBOR

In the section on risk characterization EPA has presented its joint probability analysis as well a number of studies that it contends corroborate and support, by weight of the evidence, the conclusions reached in the joint probability analysis. The joint probability analysis combines an expected environmental concentration (EEC) curve for each of the zones delimited in New Bedford Harbor with a MATC curve that has been derived as previously discussed. While EPA would have the public believe that this methodology is sophisticated and "state of the art" for estimating ecological risk, one must remember the very fragile foundation upon which both indices are built. Combining two uncertain and inaccurate results into a joint probability graph in no way confers any reality to the results. We must remember that the EEC curve relies upon data that has had essentially all the variability removed by averaging. There is no representation of the time- and space-varying exposure that would be experienced by any organism living in New Bedford Harbor. While we know, for instance, that estuarine species spend only a portion of their life in the harbor, and are exposed to very different levels of PCB with each tidal cycle, over each depth and between season, a regime that would moderate exposure to any environmental contaminants in New Bedford Harbor, this has not been represented.

To this unrealistic environmental exposure EPA has overlain an MATC curve, a professed measure of a chronic biological endpoint, that has been derived from laboratory toxicity data on other species and that has been extrapolated from the relationship between the LC_{50} and the MATC in some cases, for perhaps a third, not even related, species. One also notes that resultant MATC curves are log transformed because of the significantly great variability; the variability in the MATC curve for the marine fish, for instance, ranges over <u>seven orders of magnitude.</u> It is inconceivable that EPA can even propose a foundation for its

conclusions on ecological risk that has such a high degree of uncertainty, let alone hold it out to the public as a basis for rational decision making.

Lastly EPA took the results of this joint probability analysis and developed probability contours, using some method that is not explained in the report and presented this as the final result of their analysis. Although EPA cautioned (p. 4-7) that the "probability contours shown on the map indicate general trends within each zone and should not be used to assess localized differences of chronic effects" (whatever that means), they would have the public believe that the contours accurately reflect risk of adverse toxicological effects from PCBs to endemic organisms. This is indeed a separate reality!

OTHER STUDIES CONDUCTED BY EPA IN NEW BEDFORD HARBOR HAVE FAILED TO SUPPORT THE CONCLUSIONS REACHED IN THE ECOLOGICAL RISK ASSESSMENT.

EPA attempted to compare the risk assessment-derived results to the Ambient Water Quality Criteria (AWQC) and concluded (on p. 4-21) that the chronic AWQC is exceeded by the mean PCB concentration in all zones except Zone 5. EPA concludes that because of this "therefore, aquatic organisms are potentially at risk of experiencing effects due to chronic exposure to PCB contamination in all areas of New Bedford harbor north of the Hurricane barrier." Apparently, EPA has ignored that its previous conclusion (p. 3-2) that the AWQC does not derive final acute and chronic values, or that these AWQC values, whether appropriate or inappropriate, have no explicit relationship with toxicity to aquatic life.

EPA then addressed how site-specific toxicity tests provide the most realistic indication of toxicity posed by the sediment of New Bedford Harbor. However, the only test EPA cited (Hansen 1986) is seriously flawed and EPA itself concluded that, " it is difficult to separate effects due to PCBs from effects due to metals and other contaminants that may be present in the sediment. In addition, it is difficult to evaluate how closely the laboratory conditions simulated actual harbor conditions in the various tests." Aerovox previously commented (CDR Environmental Specialists, Inc. 1989) on the inability of Hansen's (1986) studies to demonstrate any causal relationship between exposure to PCB and the effects Hansen observed in his experiments.

For some reason EPA is conspicuously silent on the only toxicity tests designed for and specifically conducted to evaluate toxicity of New Bedford Harbor water on several species of test organisms. In 1987 and 1988, a series of toxicity tests that were a part of the Pilot Dredging Program's pre-operational monitoring program were conducted (Appendix VII and VIII, EPA, 1988). These standard EPA bioassay tests exposed several of species of organisms, including one of the

same organisms tested by Hansen (1986) referenced above, to ambient New Bedford Harbor water and concluded as follows: " In conclusion, we feel that despite some problems with test acceptability, this preliminary study of New Bedford Harbor water was successful. The receiving water quality was found to be generally acceptable to the organisms tested. Where effects were observed, they were marginal in nature and not believed to be significant." They also found in a followup study that only one of the test species exhibited toxicity and "that toxicity was due to factors other than PCB concentrations."

With regard to risk due to bioaccumulation, EPA falls short of establishing even the most elementary of cause and effect relationships between bioaccumulation and toxicological risk. After presenting a variety of information, some of it inaccurately, that discusses both the presence and absence of a correlative relationship between PCB and various reproductive effects reported, EPA failed to establish any conclusive evidence that bioaccumulation of PCBs causes any toxic effect. AVX previously criticized the scientific merit of the studies cited by EPA based on the investigator's inability to control for environmental variables other than PCB. In all of the studies mentioned by EPA a number of alternative explanations for the effects observed are possible.

EPA also cited the benthic survey of New Bedford Harbor conducted in 1988 (USACE 1988) as corroborating the results of the risk assessment. EPA concluded that the "results generally support the conclusions reached" (in the Baseline Ecological Risk report). AVX previously addressed the deficiencies in the benthic survey that EPA cited. Not only is the sampling design inadequate to define any significant differences that EPA references, but the indices cited, i.e., species diversity and evenness, are unspecific with regard to being able to draw any conclusions about cause and effect of PCB. In addition as EPA itself quite rightly pointed out, "habitat differences complicate any attempt to relate differences in

benthic community pattern to variation in the PCB contamination between these locations" (p.4-29).

In summary the Baseline Ecological Risk Assessment (EPA 1989a) is a complex and confusing analysis, replete with unsupportable assumptions and of little value to decision making. The scientific rigor employed in this study is not up to current standards; this is baffling since the methods to do a more defensible evaluation were readily available. The absence of site-specific ecological and toxicological studies which would have provided a more acceptable foundation for such an ecological risk assessment and the failure to develop an analysis which could be integrated with the results of the food chain model is further testimony to the misdirection and lack of coordination in the entire New Bedford Harbor Remedial Investigation.

LITERATURE CITED

Barnthouse, L.W., Suter, G.W., Bartell, S.M., Beauchamp, J.J., Gardner, R.H., Linder, E., O'Neill, R.V., and Rosen, A.E. 1986. User's Manual for Ecological Risk Assessment. Environmental Sciences Division Publication No. 2679. Oak Ridge National Laboratory, Oak Ridge, TN.

Bartell, S.M., R.H. Gardner, and R.V. O'Neill. 1992. Ecological Risk Estimation. Lewis Publishers, Chelsea, MI. 252 pages.

COR Environmental Specialists, Inc. 1989. Aquatic Toxicity and Bioaccumulation Potential of PCBs in the Marine Environment. Prepared by Ropes and Gray.

EBASCO. 1992. Draft Final Supplemental Feasibility Study for the Outer Bay. New Bedford Harbor, Massachusetts.

EBASCO. 1990. Draft Final Feasibility Study for the Estuary and Lower Harbor/Bay. New Bedford Harbor, Massachusetts.

Ecological Assessment of Hazardous Waste Sites: A Field and Laboratory Reference Document. EPA, 1989. Report No. EPA/600/3-89/013. Washington, D.C. March 1989.

Environmental Protection Agency (EPA). 1987. Acute Toxicity Handbook of Chemicals to Estuarine Organisms. Environmental Research Laboratory, Gulf Breeze, FL, April 1987. EPA/600-8-87-017.

Environmental Protection Agency/Office of Pesticide Programs (EPA/OPP). 1986. Hazard Evaluation Division. Standard Evaluation Procedure, Ecological Risk Assessment. Washington, D.C., June 1986. EPA/540-9-85-001.

Environmental Protection Agency/Office of Policy Analysis (EPA/OPA). 1988. Review of Ecological Risk Assessment Methods. Washington, D.C., November 1988. EPA/230-10-88-041.

Environmental Protection Agency/Office of Policy Analysis (EPA/OPA) 1989. Ecological Risk Assessment Methods: A Review and Evaluation of Past Practices in the Superfund and RCRA Programs. Washington, D.C., June 1989.

Environmental Protection Agency (EPA) Scientific Advisory Board. 1990. Evaluation of the Equilibrium Partitioning (EqP) Approach for Assessing Sediment Quality. EPA-SAB-EPEC-90-006.

LITERATURE CITED (Continued)

Environmental Protection Agency. 1989a. Draft Final Baseline Ecological Risk Assessment New Bedford Harbor Site Feasibility Study.

Hansen, D. 1986. Preliminary Data Report, New Bedford Harbor Project. EPA Nanagansett Laboratory; Rhode Island. Unpublished Report.

Suter, G.W. and A.E. Rosen. 1986. Comparative Toxicology of Marine Fishes and Crustaceans. NOAA. Technical Memorandum NOS OMA 30.

United States Army Corps of Engineers. 1986. Macrobenthic Survey of New Bedford Harbor.

III. THE EFFECTS OF A DREDGING REMEDY

THE EFFECTS OF A DREDGING REMEDY

After 7 years the government has proposed a remedial alternative for New Bedford Harbor Superfund site that relies primarily on removal of contaminated sediments by dredging, moving them to permanent vaults (CDFs) built on land that is now occupied by healthy wetlands and maintaining and monitoring these vaults for the foreseeable future, at least for the next 30 years.

On the surface this is a defective plan, made even more ludicrous by the fact that this is essentially the same plan that was presented, and rejected for sound cause, over 7 years ago. The government approach rationalizes the reintroduction of this plan on the basis that studies conducted in the intervening years, the so-called Pilot Dredging study (USACE, 1990), have demonstrated that the concerns expressed when the dredging plan was originally rejected are unfounded. Nothing could be further from the truth. Dredging the contaminated sediments of New Bedford Harbor was rejected in 1984 for sound reasons, reasons that were based on evidence that the excavation, handling, moving of these sediments throughout the estuary, lower harbor and Outer Bay over the next five years, and storing them in CDFs built on soft wetlands sediments would result in the release of more contaminants and potentially cause greater environmental impact than leaving the sediments in place. The intervening years have brought little, if any, new information or developments in dredging technology that would alleviate these concerns despite the government's contention that information resulting from the Pilot Dredging study provided significantly new information upon which to base informed decisions.

The Pilot Dredging study did little to resolve the compelling issues that were so controversial when the government made an attempt to "fast track" this very same solution to remediation of the New Bedford Harbor sediments in 1984. It was

obvious from the first that the Pilot Dredging study was calculated to support an already determined strategy and not to objectively address the many concerns previously expressed. For example as AVX has commented upon on numerous occasions, the choice of location for the pilot study was entirely inappropriate. Not only was there no appreciable amount of PCBs in the sediment compared to the rest of the estuary -- the average concentration of sediment PCBs in the pilot dredging area was less than 33 ppm, or less than the proposed TCL -- but also depth, hydrodynamic conditions and duration were not at all characteristic of the conditions that will be experienced under the Proposed Plan. For these reasons any attempt to extrapolate the results of Pilot Dredging study to the Proposed Plan is just wishful thinking.

One can hardly doubt that there was and still is a programmatic bias towards dredging. After spending an estimated $6,000,000 on this Pilot Dredging and Engineering Feasibility Study, we have little to show in the way of unique or innovative technology, even new dredging technology, to deal with this problem. It is obvious that the whole feasibility study was biased towards developing support and justification for a dredging alternative and not to evaluate or develop new site- specific solutions. Not only has this been a colossal misuse of money, it has also been a preposterous miscarriage of the CERCLA process.

THE ADVERSE EFFECTS OF DREDGING ARE EQUAL TO OR GREATER THAN ANY POTENTIAL EFFECTS OF LEAVING THE SEDIMENTS IN PLACE

Dredging is an intrusive remedial alternative which, by its very nature, results in disturbance of the sediments and surrounding water column and leads to a variety of potential impacts to the environment. The major effects due to dredging the sediments in New Bedford Harbor include: 1) destruction of aquatic habitat, including wetlands and the benthos; 2) resuspension of sediments and release of sediment-associated contaminants, both in the particulate, "oil", and soluble phases, with resulting decrease in water quality and potential for increased risk to marine organisms; and, 3) volatilization and release of contaminants to the air, with a resulting potential for decrease in air quality and increased risk to human health.

DESTRUCTION OF AQUATIC HABITAT

The proposed dredging will ultimately remove and displace or kill the benthic organisms living in the areas where dredging is planned. Those sessile benthic organisms living in the sediment, the infauna, will be killed by the dredging. Many of the motile "epifauna" will also be killed; some will be able to avoid the dredge. In addition, many of the endemic and migratory pelagic species utilizing the estuary will be directly impacted by the dredging or indirectly impacted through eliminating the benthos in the area as a food source. Although the area that is dredged will be subsequently recolonized by opportunistic species, it will take some time before the structure and function of the benthic community will return to the physically modified benthic habitat. EPA admits in the Feasibility Study for the Estuary and Lower Harbor/Bay (FS-E/LH/BAY) (EBASCO 1990) (p. 2-39) that, "the time required to recolonize these impacted areas is not known." Thus, EPA cannot appropriately evaluate these impacts as short or long term as it is required to do.

In intertidal and wetland areas there will be additional impacts. Although the details of the preferred remedy are insufficient to estimate the amount of intertidal habitat that will be changed it is clear that several acres of intertidal habitat will be changed into subtidal benthic habitat through dredging. This will eliminate or destroy habitat which supports shellfish beds, e.g., the soft shell clam, Mya arenaria, or which serve as important feeding areas for shorebirds and waterfowl.

The proposed plan for the estuary and harbor will also involve the destruction of several acres of saltmarsh, a valuable wetland, both by dredging and also by placement of several confined disposal facilities (CDF) in wetland areas. The ramifications of this action are only addressed casually in the FS-E/LH/Bay and there is no attempt to quantify this impact or to suggest that anything will be done to mitigate for it other some vague suggestion that the saltmarsh will be restored. This is surprising considering the high priority that has been given to the preservation of wetlands and the necessity to mitigate for them if they are taken or destroyed. If any private applicant submitted a plan which involved destruction of over three acres of valuable wetland habitat, EPA would want substantially more well thought out plans than are presented in the FS-E/LH/BAY.

The physical modification of the estuarine habitat through dredging will also result in long term indirect effects by changing circulation patterns. In areas of the upper intertidal zone, near the foot of the saltmarsh, dredging will presumably undercut the bank of the estuary and result in erosion and slumping of the edge of the saltmarsh. This process is a progressive one and without some means of stabilizing the banks, there is a substantial potential for losing great areas of saltmarsh. The overall effects of these have not been quantified or discussed by EPA, but from a qualitative perspective the changes to the hydrography and

physiography of the upper estuary may ultimately result in a different estuarine community.

In the Outer Bay, the Proposed Plan contemplates dredging and capping of some as yet unknown area to the south of the Hurricane Barrier. Not only is the areal extent unknown at this time, but the environmental impact of the proposed remedial action is hardly given lip service in the Supplemental Feasibility Study (EBASCO 1992) (SFS). Neither are the environmental impacts resulting from the capping and dredging planned for the Outer Bay compared to any potential impacts that are presently occurring. Such a flagrant disregard for even a modicum of a comparative analysis is ill-advised, arrogant and violates the carefully considered process NCP has established for remedial decision making.

RESUSPENSION OF SEDIMENTS AND RELEASE OF SEDIMENT-ASSOCIATED CONTAMINANTS

The disturbance of bed sediment by dredging and the movement of the dredged sediments, whether by barges or hydraulically through pipes will lead to increased sediment resuspension and turbidity. This will have a number of effects including release and mobilization of sediment particles as well as organics and contaminants associated with the sediments. These have been commented upon before by AVX in their critique of the Hot Spot Feasibility Study. Photographs and analysis of water column samples collected by Rizzo Associates during the USACE Pilot Dredging Study indicated levels of suspended sediment four to ten times higher than background levels at distances up to 400 feet from the dredge. (Defendants' Joint Comments on the Hot Spot Feasibility Study, New Bedford Harbor. Rizzo Associates, 1989).

During dredging of contaminated sediments it is therefore important to remove and transport the sediment with a minimum of sediment turbidity and associated

contaminant release. The major variables that can affect turbidity and contaminant release during dredging are: 1) the type of dredge used; 2) the control of the dredge to limit the amount of sediment removed and subsequently transported; 3) the period of time over which dredging is conducted; and 4) effects caused by operations and attendant equipment, e.g., barges, swing anchors, prop wash, etc. This was also observed during the Pilot Dredging Study (USACE, 1990).

The type of hydraulic dredge being proposed for use in New Bedford Harbor relies on a centrifugal water pump to create a vacuum at the dredgehead. The dredged materials will be hydraulically pumped through a pipeline to the disposal area. The typical hydraulic dredge loosens the sediments and then transports it with a quantity of water as a "slurry". A significant problem with using hydraulic dredging for contaminated sediments is the quantity of excess material that is contaminated through the process of creating the slurry. Typically, hydraulic dredging operations result in a slurry that is only 10 percent solids, the other 90 or greater percent being the water added during the dredging. In operations involving contaminated sediments, however, there may be even more dilution due to controls put in place to limit turbidity, i.e., slowing the swing speed of the dredgehead.

In addition, when working in the soft, fine sediments of New Bedford Harbor, particularly in the Upper Estuary, there is expected to be considerable "bulking" of material. This results from the disruption of previously consolidated material. Because of this the volume of material dredged, treated and subsequently stored in the CDF will initially be 40 - 50 percent greater than what was dredged from the bottom.

The primary objectives in dredging contaminated sediment are to minimize resuspension, maximize precision so "overdredging" and therefore dredged

volume is minimized, and maximize productivity so as to minimize the time that sediments are resuspended and endemic organisms exposed to contaminants released during the dredging process (Palermo, 1991). Obviously, maximizing precision and productivity are competing objectives, especially when working in a shallow estuary. The government has failed to discuss these variables in detail or to present their strategy or cost benefit analysis so it can be reviewed.

Lastly, as was observed in the Pilot Dredging Study (USACE 1990) resuspension of contaminated sediment will result from the operations of dredges in a shallow estuary. Hydraulic dredging in waters the depth of the upper estuary will require small, shallow draft barges. These barges will probably have a minimum draft of about three feet. Since much of the upper estuary is less than these depths and, in fact, over 15 acres of it in the intertidal zone, the placement, movement and operation of these barges will be severely limited by the depth of water. The dredging operation will therefore require dredging of pilot channels at least adequate for the barges to be floated in, and perhaps deep enough to allow operation of small tugs to move the barges. This will result in the need to dredge a substantial amount of uncontaminated bottom sediment and contaminate it in the process by mixing it during the dredging process.

As documented by many studies (Tramontano and Bohlen 1984; USFWS 1980) the suspended solids that results from dredging operations not only have the potential to smother benthic organisms in the vicinity when the suspended sediments settle out, but the presence of increased suspended solids level in the water column can also reduce or stop feeding by suspension feeding organisms, clog the gills of fish and invertebrates and interfere with photosynthesis of phytoplankton or submerged aquatic vegetation. The transport of these suspended sediments also spreads the contamination to other areas remote from the dredge. The potential for and magnitude of this transport has not been adequately studied by EPA.

Use of hydraulic dredging in New Bedford Harbor will also result in the entrainment of meroplanktonic larvae of some of the endemic species, e.g., the soft shelled clam and the lobster. Entrainment results from the plankton in the water getting sucked into the hydraulic lines and killed during dredging. Accentuating this effect is the fact that the estuary is relatively constricted and the dredging period will be extended, probably over several years, so there is a high probability that successive cohorts of estuarine species may experience high mortality rates through entrainment. The long term effect of this impact has not been addressed by the government.

For the same reason the hydraulic dredging operations will impede or interfere with use of the estuary as a feeding or nursery area for migratory fish. Operation of the dredges in the Upper Estuary as planned will essentially preclude use of the entire area for extended periods. Again, the long term effect of this impact has ben adequately addressed.

While it is generally true that estuarine organisms, both plant and animal have adapted to and can survive changing physical conditions and increases of turbidity, as associated for instance with storms, the dredging as planned will be a continuous insult over an extended period of time, over five years (EBASCO 1990). This is a substantially different situation than would be experienced under natural conditions where increases in turbidity due to episodic events is short term in nature.

As AVX has previously commented, and as was documented in the Pilot Dredging Study (USACE 1990), the engineering of effective and continuous turbidity controls during dredging is nearly made impossible due to the tidally-dominated, shallow estuarine environment in the Upper Estuary and Harbor or, in the Outer Bay, because of the dynamics of the near shore zone. It is probable that the activities associated with any attempt to minimize turbidity in the Upper

Estuary, for instance, will result in as much turbidity due to prop wash as would be released from unrestrained dredging.

Probably the most significant water column impact associated with dredging, however, is likely to be the release of contaminants, e.g., PCBs, PAHs and heavy metals as well as nutrients, especially free sulfides, hydrogen sulfide, and ammonia (USFWS, 1980). Among the many effects from the release of nutrients and organics, especially throughout a long duration dredging project as is presented in the FS-E/LH/Bay will be the increased biochemical oxygen demand resulting from exposure of reduced sediments to oxygenated water during dredging and discharge from the CDF. This increased biochemical oxygen demand reduces the amount of oxygen in the water column available to marine life and could have significant effects, including death, to organisms in the estuary. This impact was not addressed by the government.

As has been documented by several other studies, the release of sediment-associated contaminants through dredging will significantly increase the bioavailability and subsequent uptake and bioaccumulation of these contaminants (USFWS 1980; Dillon 1984; Rice and White 1987; Larrson 1990). While this may be an acceptable trade off in the short term, if there were subsequent benefits to the estuarine organisms in the long term, the New Bedford Harbor dredging program will continue over several years, thereby extending the period of effect. In addition, because the topography of the estuary and lower harbor will be changed, the sediment bed equilibrium that established will be modified. It is likely that this will result in increased flux of contaminants in other areas of the estuary. No attempt has been made by EPA to quantify this potential.

The process of hydraulic dredging not only will facilitate the release of contaminants from the sediment to the surrounding water at the point of dredging, it will also disperse them throughout the water used to conduct the

hydraulic dredging. Since a substantial amount of water will be necessary to conduct the hydraulic dredging, and since hydraulic dredging involves at least a 10:1 dilution, and there will be extensive mixing in the dredge slurry, such that the dredging process will daily mix millions of gallons of water with the contaminated sediment. Since the dredged sediment is no longer in equilibrium with the water in the slurry, and there is an active mechanical mixing of the sediments occurring, it is expected that many of the contaminants will be desorbed from the sediment particles and subsequently discharged or volatilized to the air either at the dredge site or during discharge of the hydraulic lines to the CDF or from the CDF before it is capped.

VOLATILIZATION AND RELEASE TO THE AIR

Once the sediments have been disturbed by dredging and the contaminants released to the dredge slurry, those contaminants that are hydrophobic, e.g., PCBss will have a significant tendency to partition back to sediment or to volatilize to the air. To the degree that this volatilization from the water surrounding the dredge location, during handling and transport through the hydraulic lines and by release from the CDF increases airborne concentrations of contaminants, any potential risk to human health or to wildlife is increased. There has been no apparent attempt to address this aspect of the Proposed Plan. This is a glaring oversight since there is no other means to weigh the benefits of the Proposed Plan to human health than to compare the relative risk associated with dredging and disposal in the CDFs. This oversight is especially remarkable since it was reported by Fowler and Hanson (1991) that there were indications from emissions modeling that there was a "potential problem associated with shoreline disposal of PCBs contaminated sediment." Despite their admonition that need for engineering controls to limit PCBs emissions "should be determined on a site specific basis" no further investigations were conducted for proposed remedial plans for either of the operable units. This, in spite of the fact that Fowler and

Hansen (1991) pointed out that their study and conclusions were only applicable for dredging in sediment of 200 ppm or less.

In a review of the potential magnitude of volatilization during implementation of the proposed remedy (Valsaraj and Rieble 1991 Attachment B) estimated that the potential volatilization during dredging would be between 146 and 2630 g/hr depending upon how well the dredging operation is controlled. Assuming that dredging, just in the estuary and lower harbor, lasts for 5 years (EBASCO, 1990) and is conducted four hours per day for 200 days each year, there is the potential for release of from 584 kg to 10,520 kg (over seven metric tons) greater than what would be released during that period if no action is taken. During a period equivalalent to the duration of dredging Valsaraj and Rieble (1991) estimate approximately 124 kg PCBs would have been released by volatilization under the no action scenario.

In addition to volatilization around the dredge head there would be additional potential for volatilization where the dredge slurry is discharged and ponded in the CDF and from the CDF before and, to a certain extent, after it is capped. Rieble (1992; Attachment C) has estimated the magnitude of that release that using the same assumptions that the dredge release rates are based upon, and equations developed by Thibodeaux (1989) for the Pilot Dredging CDF. Rieble estimated up to 480 g/hr might be released in the CDF filling operation and up to as much as 46 g/hr PCBs released from the exposed dredge spoils after dewatering. Although EPA has not described how long the CDF's will take to be filled or left open prior to capping, if one assumes two years as a reasonable approximation (the dredging will take place over 5 years or more) than one may find as much as 8400 kg or 4.2 metric tons of PCBs might be released during filling and as much as another 807 kg or PCBs released from the uncapped CDFs. It should be noted that the lower rates for PCBs concentration in the CDF effluent measured by USACE in the Pilot Dredging Study were likely due to the fact that

initial concentration of PCBs in the sediment was lower than what will be encountered in the majority of the areas to be remediated in the estuary and PCBs in the CDF effluent were net of those that had volatilized in the CDF or readsorbed to sediment or sides of the CDF structure.

It is uncertain because of the absence of data, how long dredging will be conducted in the Outer Bay or what volume of sediments will have to be disposed of in one of the three CDFs built for the Estuary and Lower Harbor remediation, or for that matter in an additional CDF built especially for the Outer Bay sediments as suggested by EBASCO (1992). Obviously, these will be additional PBCs that will be volatilized during the remediation of the Outer Bay. This has not been addressed by EPA.

As previously discussed, an additional source of contaminants being released to the system will be through flux from the CDF. Since the CDF is planned to be built in an intertidal area and will be unlined, the construction of the CDF, facilitated by the tidal pumping will likely ensure a chronic low-level release of PCBs and other contaminants. In addition to volatilization dredging would accelerate release of PCBs from sediments, as explained previously. Although there EPA has not made any effort to quantify this incremental release, it is likely to be significant.

In summary, EPA has overlooked or vastly underestimated the potential for release of PCBs during dredging and CDF disposal operations. In just reviewing these few single examples one can calculate a release of over 11 metric tons of PCBs volatilizing during the five year operation. During that same time in the absence of remedial activity only about 186 kg PCBs would have been volatilized from the upper estuary during that time.

One legitimately wonders how a decision to dredge could be made when it is clear that at least for the first several (5-10) years the result will be an incremental release of PCBs, an increase which, by the way, was not considered in Battelle's hydrodynamic model nor in the food chain model. One also legitimately wonders whether by the end of the 5-10 year period that natural capping through sedimentation and degradation of PCBs through microbial action, a mechanism which has been demonstrated by EPA, will have resulted in decreasing the release rate of PCBs at least as much as this will considered action. **EPA has not adequately answered this question!**

If EPA had made the effort to include these additional releases from volatilization in its risk assessment and included the incremental dredging-related increases in its hydrodynamic and food chain models, and included the effects of natural capping through sedimentation and degardation of PCBs by microbial action, it is likely that the results would demonstrate that there was no significant benefit to the environment gained from the incredibly expensive remediation it has proposed.

LITERATURE CITED

Dillon, T.M. 1984. Biological Consequences of Bioaccumulation in Aquatic Animals; an Assessment of Current Literature. Technical Report D-84-2, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

EBASCO. 1990. Draft Final Feasibility Study for the Estuary and Lower Harbor/Bay. New Bedford Harbor, Massachusetts.

EBASCO. 1992. Draft Final Feasibility Study Evaluation for Upper Buzzards Bay, New Bedford Harbor RI/FS. New Bedford, Massachusetts. Rice, C.P. and D.S. White. 1987. PCB Availability Assessment of River Dredging Using Caged Clams and Fish. Environmental Toxicology and Chemistry, 6:259-274.

EBASCO. 1992. Letter from Lewis Horzempa, Project Manager to Gayle Garman, EPA. Subject: Task 15 Supplemental SFS Cost Estimate.

Fowler, A.S. and M.R.Hanson. 1991. Ambient Air Monitoring of Dredging and Disposal of PCB Contaminated Sediment at a Marine Superfund Site: New Bedford Harbor, Massachusetts. Report 91-80.21. EBASCO Services Inc.

Larrson, P., L. Okla, S-O Ryding and B. Westoo. 1990. Contaminated Sediment as a Source of PCBs in a River System. Canadian Journal of Fisheries and Aquatic Sciences. 47:746-754.

Palermo, M.R. 1991. Equipment Choices for Dredging Contaminated Sediments. Remediation Journal.

Rice, C.P. and D.S. White. 1987. PCB Availability Assessment of River Dredging Using Caged Clams and Fish. Environmental Toxicology. 6:259-274.

Tramontano, J.M. and Bohlen, W.F. 1984. The Nutrient and Trace Metal Geochemistry of a Dredge Plume. Estuarine, Coastal and Shelf Science. 18:385-401.

Thibodeaux, L. J. 1989. Theoretical Models for Evaluation of Volatile Emissions to Air During Dredged Material Disposal with Applications to New Bedford Harbor, MA. Final Report No: Misc. EL-89-3. U.S.Army Engineering Waterway Experiment Station. Vicksburg, MS. 44 pp.

USACE. 1990. New Bedford Harbor Superfund Pilot Study. Evaluation of Dredging and Dredged Material Disposal. Final Report.

LITERATURE CITED (Continued)

USFWS. 1980. Impacts of Navigational Dredging on Fish and Wildlife: A Literature Review. FWS/OBS-80/07. Washington, D.C.

Valsaraj, K.T. and D.D. Rieble. 1991. A Theoretical Evaluation of the Effects of Dredging on PCB Emissions from New Bedford Harbor Sediment. report to Balsam Environmental Consultants, Inc. 33 pp.

IV. OUTER BAY PROPOSED PLAN

OUTER BAY PROPOSED PLAN

For some reason which defies logic, despite a confusing and unsuccessful attempt to justify it, EPA has decided that while 50 parts per million (ppm) is an adequate target clean-up level (TCL) within the estuary and lower harbor, that it must go to an TCL level of 10 ppm in the Outer Bay to achieve its remedial objectives. EPA indicates that this decision is necessary, even though it is obvious from recent data (Schwartz 1988) (Hillman 1987) that average PCB body burdens of flounder in the Outer Bay are generally below the FDA limit of 2 ppm PCB and average PCB body burdens of lobster in the Outer Bay appear to be at or near that level. In order to justify this more onerous TCL, EPA has relied on human health risk assessment (EBASCO 1989) in which they conclude a level of 0.2 ppm in edible tissue of aquatic organisms is require to maintain human health risk below a level of 10^{-5}. AVX has addressed the inappropriateness of this conclusion.

The "technical rationale" (EPA 1992; p.7) for the proposed remedial plan for the Outer Bay (EBASCO 1992) is a sophomoric, "back of the envelope" analytical model, the so-called "relative exposure model" developed by NOAA in its Evaluation of Effectiveness report (NOAA 1992). While AVX will not dignify this futile attempt at scientific analysis by pointing out the many shortcomings of the approach taken, suffice it to say that its use has taken the level of environmental impact evaluation back to the pre-Clean Water Act days. It is clear in the Supplemental Feasibility Study (EBASCO 1992; 4-26) that while EPA was using this relative exposure model as their "technical rationale" it recognized the many uncertainties inherent in the "model". Why EPA appears satisfied with such a flawed tool when it felt it was necessary to develop a far more sophisticated three-dimensional model for the estuary and Lower Harbor -- which cost several hundreds of thousands of dollars and tens of thousands of man hours over a six year period, which was so complex that it took 90 minutes on a Cray XMP computer to run one tidal cycle -- is incomprehensible. What is clear to AVX, however, is that is was political expediency and not any sound scientific or

engineering rationale that has caused EPA to leap to the other end of the spectrum in evaluation strategy when it came top the Outer Bay.

Before discussing a few fundamental problems with the relative exposure model it is first appropriate to point out that while EPA may appear to conclude otherwise, the development of a model, be it an analytical or a numerical model does not by itself confer any reality on or legitimacy to the results of that model. A model must be calibrated with realistic assumptions and then verified against an independent data set to acquire any credibility. The calculations made by NOAA in the so-called evaluation of effectiveness study are no more than just unfounded and simplistic speculation until EPA demonstrates their credibility by calibrating them against known data and verifying the results with other independent data. **EPA has not done that.**

In reviewing the "house of cards" upon which EPA has staked its scientific integrity, S.E.A. (Attachment D) has focused on the three fundamental points underlying NOAA's analysis: first, that PCB levels in the biota are directly related to sediment PCB concentration; second, that the sediments of the Outer Bay are the predominant source of PCB to the resident biota; and, third, that available PCB data accurately represent the present distribution patterns of PCB in the Outer Bay. S.E.A.'s evaluation of these assumptions are that even in the first order they are not met. All assumptions are shown to be either totally or partially violated. Because of this, NOAA's model is incorrect and presents no basis for a realistic or accurate assessment of alternatives. In addition, the relative absence of data on the distribution of PCB in the Outer Bay results in tremendous uncertainty in estimating scope and cost as well as effectiveness of remedial action. EBASCO (1992) has even estimated that if the nine potentially contaminated areas in Exhibit E of the SFS are confirmed to exceed 10 ppm it could result in a cost for remedial action to be $30,000,000 or $40,000,000 or more and the requirement to take more wetland in the estuary to site an additional CDF. Indeed, Balsam's independent analysis (Table 1) indicates that costs to

dredge or cap these additional areas and construct an additional CDF would bring the total estimated costs for the Outer Bay remediation to an estimated $44,707,171.

The decision by EPA to even develop a plan for remediation of the Outer Bay in the absence of sufficient data to determine is clearly premature. While the scientific uncertainty of the estimates of cleanup area and volume are discussed in the SFS (EPA 1992) and the SEA (Attachment D) review of the NOAA evaluation one does not need to be a scientist to ask the most fundamental common sense question--why does EPA feel compelled to make a remedial decision now with such an insufficiency of information on matters which have such significant economic, environmental and sociological ramifications.

It is clear from EPA's and NOAA's own words -- and certainly by the extent and comprehensiveness of EPA's proposed 'confirmatory" pre-design sampling plan (EBASCO 1992; Appendix D) -- that there is a tremendous uncertainty about the distribution of PCB in the Outer Bay (a minimum of another 300 samples are planned to be collected). In fact there is so much uncertainty that one is moved to question whether EPA's plan to cleanup certain as yet undefined parts areas of the Outer Bay is procedurally premature. Based upon the evidence of the lack of data (which is confirmed by EPA's own admission and the magnitude of the pre-design sampling) it appears that EPA is, in fact, proposing another remedial investigation. While AVX challenges the data available and the analysis used to justify further action on the Outer Bay, AVX considers that the EPA's level of understanding and knowledge of the Outer Bay is such that to go forward with a proposed plan without formally separating the Outer Bay into an operable unit and going forward with an additional remedial investigation is not only cavalier, but is regulatory extortion.

LITERATURE CITED

EBASCO 1992. Draft final feasibility study for the Outer Bay New Bedford Harbor, Massachusetts.

Schwartz, J.P. 1988. Distribution and concentration of polychlorinated biphenyls in lobster, winter flounder, and quahogs from Buzzards Bay, Massachusetts. Grant No. CX 812852-01.

Hillman, R.E., M.S. Connor, and R.A. McGrath 1987. Ratios of edible tissue to whole body PCB concentrations in flounder and lobster from New Bedford Harbor Final Report.

EBASCO 1989. Draft final baseline public health risk assessment; New Bedford Harbor Feasibility Study prepared for EPA.

EPA 1992. Addendum Proposed Plan for Upper Buzzards Bay, New Bedford Harbor Superfund Site.

NOAA 1992. Evaluation of effectiveness: relative exposure model. Report to Gayle Garman, EPA.

TABLE 1
COST ESTIMATE COMPARISON/EVALUATION
SELECTED REMEDIAL ACTION ALTERNATIVE BAY-4
UPPER BUZZARDS BAY, NEW BEDFORD HARBOR
NEW BEDFORD, MASSACHUSETTS
Cap Area C, Dredge and On-Site Disposal of Areas A & B

ACTIVITY/ DESCRIPTION	EPA COST ESTIMATE	REVISED COST ESTIMATE
I. Direct Costs		
A. Dredging Areas A & B		
1. USACE Estimate Approximately 67,000 cubic yards	$1,204,200	N/A
2. EBASCO Estimate Approximately 252,000 cubic yards	N/A	$3,325,030
B. CDF No. 1 Dike Modifications	$ 241,350	N/A
C. Construction of Additional CDF for 252,000 cubic yards	N/A	$4,850,000
D. Water Treatment	$ 335,300	$ 335,300
E. Capping of Area C		
1. USACE Estimate Approximately 160,000 cubic yards	$2,752,000	N/A
2. EBASCO Estimate Approximately 850,000 cubic yards	N/A	$14,776,000
F. Predesign Program	$ 349,600	$ 349,600
	==========	==========
SUBTOTAL I	$4,882,450	$23,635,930
II. Indirect Costs		
A. Health & Safety (at 5%) Level D Protection	$ 244,123	$ 951,364
B. Legal & Administrative (at 6%)	$ 292,947	$1,141,637
C. Engineering (at 10%)	$ 488,245	$1,902,728
D. Services during Construction (at 10%)	$ 488,245	$1,902,728
E. Turnkey Contractor Fee (at 15%)	$ 732,368	$2,854,092
	==========	==========
SUBTOTAL II	$2,245,927	$8,752,549
Contingency (at 20%)	$1,425,675	$6,477,695
		==========
TOTAL CAPITAL COSTS	$8,554,052	$38,866,174
III. Operation & Maintenance Costs (Aquatic Cap of Area C: present worth at 5% for 30 years upon completion)	$1,099,500	$5,841,000
SUBTOTAL III	$1,099,500	$5,841,000
TOTAL COST ESTIMATE OF REMEDIAL ACTION	$9,653,552	$44,707,174

V. ENGINEERING FEASIBILITY AND COST ESTIMATION

ENGINEERING FEASIBILITY AND COST ESTIMATION

The United States Environmental Protection Agency (EPA-E-LH/B) Region 1 has proposed a plan of for a preferred remedy for the Estuary and Lower Harbor/Bay (EST/LHB) portion of the New Bedford Harbor Superfund Site. The proposed plan consists of the following components:

- construction of three confined disposal areas (CDFs) within wetland areas of estuary for storage of PCB contaminated sediments that will be dredged from selected locations within the estuary and lower harbor;
- dredging of approximately 308,000 cubic yards of sediments contaminated with PCBs at concentrations greater than 50 parts per million (ppm) from selected areas within the estuary and lower harbor;
- transport (via hydraulic pumping) the dredged sediments to the CDF's where the contaminated sediments will gravity settle,
- treatment of the effluent discharged from the CDFs prior to the ultimate discharge of the effluent to the harbor. The effluent will be treated to remove PCB and heavy metals contained on the sediment remaining suspended in the effluent; and
- capping of the CDFs to prevent migration of liquids, through the cap and into the contained contaminated sediments.

In a review of the proposed plan, a number of questions have been raised that strongly imply that EPA has not adequately evaluated the feasibility of its preferred plan or has underestimated the complexity and cost of the effort. In the attached table (Table 1) costs developed by EPA in its Feasibility Study (EBASCO 1990) have been compared to costs that will result once one takes into consideration factors which EPA did not adequately take into consideration or tasks that EPA underestimated.

THE COSTS ASSOCIATED WITH DREDGING WILL BE GREATER THAN EPA HAS ESTIMATED.

There will be substantial problems associated with the performance of the dredging operations during the implementation of the proposed remedial plan, some of which were experienced by the USACE during the pilot dredging program. EPA's estimated timeframe for dredging activities does not account for a realistic production rate or sufficient down time for mechanical problems or for the need to frequently clear the dredge head of any debris encountered during the dredging activities.

EPA has estimated a cost of approximately $3,292,000 to dredge approximately 308,000 cubic yards of contaminated sediment (EBASCO 1990). The cost presented in the EPA cost tables for the dredging operation appears to include the cost associated with pumping the dredged sediments from the dredging site to the CDFs. EPA has indicated in their cost tables that a unit cost of $ 9.66 per cubic yard of sediment was used to develop the dredging cost for the preferred remedy. EPA has also indicated that the dredging is expected to require approximately 1100 days to complete (EBASCO 1990). This time frame equates to removing 280 cubic yards of sediment per day for the two dredges. This assumes that both dredges will be operating at peak performance (35 cubic yards dredged per operating hour) and that the entire four hours during high tide will be spent dredging. This assessment of the dredges efficiency does not agree with the results from the USACE Pilot Study (USACE 1990) nor does it appear to be technically feasible.

Due to the nature of the selected dredge, guide boats will be required to position the dredge prior to the initiation of any dredging operations as noted in the Pilot Dredging Study. For instance, during the Pilot Study, dredging could only be conducted in area 1 (located in the estuary in a small cove adjacent to the construction site for the Pilot Study CDF) 77 percent of the available time. Furthermore, the actual dredging rate during the Pilot Study (taking into account two passes with the dredge, averaged 26.5 cubic yards per hour). The difference in production rate (decrease from the proposed rate

to a rate that was obtained during the Pilot Study), 35 c.y./hr versus 26.5 c.y./hr would result in an increase (to an estimated 1453 days) in the time required to complete dredging in the estuary and lower harbor by an estimated 32%.

This increase in estimated dredging time assumes that both dredges are operating for the full four hours per day, and does not allow for any operational down time nor for any time associated with movement, set up and demobilization of the dredge each day. It is reasonable to expect the start up and shut down operations associated with the dredge to require one to two hours each day. Some of these operations may be performed during low tide; however, repositioning of the dredge prior to the start of dredging in a new location will have to occur during high tide due to the required draft of the dredge and the need for pilot boats to maneuver the dredge. Therefore, the effective daily operating time may be reduced from four to two or three hours per day.

Assuming the effective daily operating (dredging) time is three hours, and two dredges are available everyday, the actual time required to dredge the 308,000 cubic yards of material will be approximately (assuming a production rate of 26.5 cubic yards per hour per dredge) 1940 days. Assuming an estimated 220 days per year during which dredging may occur, the dredging operations will require approximately 9 years. This increased timeframe will result in an increase in the costs associated with the dredging due to the extended duration of the time the dredges spend on site and the additional routine maintenance requirements. Based on the issues identified above the cost associated with the dredging component of this remedy could increase from 75 to 100%. This would result in a cost increase for the dredging component of the remedy ranging from $2,500,000 to $3,300,000. These increased costs are reflected in Table 1.

THE COST ASSOCIATED WITH THE CONSTRUCTION OF THE THREE CDFS DOES NOT APPEAR TO ACCOUNT FOR THE COST ASSOCIATED WITH OBTAINING, AND PLACING ADDITIONAL MATERIAL THAT WILL BE NEEDED AS A RESULT OF THE SETTLEMENT OF THE CDF DIKES.

A review of available geotechnical information (Woodward-Clyde Consultants 1987) regarding the proposed locations for the three CDFs indicates that the native material on which the three CDFs will be constructed consists of 6 or more feet of silt with low density and poor bearing capacities. Surcharging (discharging of the dredge spoils to the CDF) of the silt material will result in the displacement and consolidation of the existing silt and subsequent failure of the dike wall. The failure of the dike walls will require additional placement of material in order to maintain the required design height and width of the dike walls. It is likely that the dike walls will settle approximately 3 to 5 feet and as a result, the cost for the construction of the three CDFs may double due to the cost associated with the extra material necessary to construct the dike walls to the required heights and the cost associated with the placement of the material. These costs have been included in Table 1.

Several additional problems were also identified concerning the constructability of the CDFs:

1) Placement of the geotextile and subsequent fill during high tide will result in a mud wave similar to the one that developed within the CDF during the Pilot Study (USACE 1990). As noted in the Pilot Study report a single CDF was constructed in the general location where CDFs 1 and 1a will be constructed. The Pilot Study CDF was designed to contain approximately 5,000 cubic yards of sediment (the Pilot Study was much smaller than either of the two proposed CDFs for this location; CDF1 or CDF1a). Due to the size of the pilot CDF it was necessary to construct a portion of the CDF below the mean high water elevation. As mentioned previously geotechnical testing of the subsurface materials present under the proposed CDF location indicated that the materials were unsuitable for structural loading. In order for the construction of the CDF to proceed, the USACE selected a woven geotextile to be placed over the unsuitable native materials. This geotextile was placed in order to reinforce the native soils. The USACE placed the geotextile immediately prior to the placement of the first lift of material used to construct the dike. The placement of the geotextile followed by the immediate surcharging of the native soils resulted in a wave of subsurface material moving away (a mud wave) from the in water portion of the dike. Due to the presence of the fill material over the geotextile, assessment of damage or displacement of the geotextile could not be performed (USACE 1990; Appendix 6). The extent that the wave of subsurface material traveled from the dike wall was not recorded.

However, the fact that material did move is significant since the proposed locations for the three CDFs is over areas of contaminated sediments that would have to be dredged.

As noted in the current proposed remedy description, the construction of the CDFs over areas of contaminated sediments will reduce the amount of contaminated sediments that would otherwise have to be dredged and placed within a CDF by approximately 147,600 cubic yards. However if construction of the CDFs results in any movement of this material from outside the footprints of the CDFs the material will require dredging and being placed within a CDF. It is expected that some if not all of the material under the CDF walls will be ultimately be displaced and require dredging. Any of these additional sediments displaced by the construction of the CDFs will have to be stored in one or more of the CDFs. Given the current volume of sediments to be dredged and the EPA's anticipated expansion factors, the storage volume of the three CDFs (EPA estimates 434,000 cubic yards) may not be sufficient to contain the extra sediment displaced by the construction of the three CDFs. Therefore, the capacity of one or more of the CDFs may have to be increased resulting in additional costs and time to complete the remedy.

2) The design of the CDFs calls for the construction of two cells. In order to meet this design requirement in the Pilot Study, the USACE installed sheet piling within the CDF effectively establishing two separate cells. If such a method is planned for the estuary and lower harbor remediation, there is a significant possibility that the use of the sheet piling will damage the geotextile that has been placed over the subsurface material and result in an increased potential for differential settlement and failure. Although no mention of any impact on the geotextiles due to the placement of this sheet piling is given in the Pilot Study report, the report does indicate that the placement of the surcharge material over the geotextile did result in tears in the geotextile material. Although it was not monitored in the Pilot Study, the installation of the sheet piling within the CDF may also have resulted in significant tearing or displacement of the geotextile that was placed beneath the CDF walls to stabilize and support the walls. It is expected that long term monitoring of the settlement and slumping (if any) of the CDF walls will have to be performed in order to evaluate any impacts resulting from the installation of the sheet piling.

3) Consolidation and differential settlement will cause the cap on the CDF to experience slumping and faulting and may ultimately cause failure in the geomembrane cover with a subsequent increase in the percolation. Increased percolation will have the effect of increasing hydraulic head within the CDF and may increase the rate of contaminant flux from the CDF. At a minimum there will be an increased requirement for O & M to maintain the integrity of the cap.

THE EPA HAS NOT INCLUDED COSTS AND ISSUES ASSOCIATED WITH SITING THE CDF'S.

Acquisition of additional land is necessary to construct the three CDFs. The CDF constructed during the Pilot Study was located on city land and posed few problems in acquiring and preparing the land. However, the selected locations for the three CDFs to be constructed as part of the preferred remedy are located on private land. Acquisition of this land will involve costs not currently contained in the EPA's cost estimate for the preferred remedy. Based on the purchase of approximately 10 acres of land along the estuary and lower harbor and current and values for this area (conversation with commercial real estate agent in New Bedford, July 8, 1992) the cost for the purchase of this land is expected to add approximately $300,000 to the EPA's cost for this remedy. Furthermore, the construction of the CDFs on this land and required long term maintenance of the CDFs (periodic replacement of the rip rap, periodic monitoring of the stability of the dikes, and periodic leachate/groundwater monitoring) will preclude the land in the vicinity of the three CDFs from other private of commercial use.

THE COSTS PRESENTED BY THE EPA FOR THE PROPOSED WATER TREATMENT SYSTEM MAY HAVE BEEN UNDERESTIMATED DUE TO THE FOLLOWING:

- **THE PRESENCE OF ORGANICS ASSOCIATED WITH SEWAGE THAT IS DUMPED INTO THE ESTUARY AND HARBOR THROUGH COMBINED SEWER OVERFLOW PIPES (CSO'S) MAY REDUCE THE TREATMENT SYSTEM'S ABILITY TO TREAT THE EFFLUENT FROM THE CDFS TO THE LEVELS NECESSARY TO ALLOW THE EFFLUENT DISCHARGE TO THE ESTUARY;**

- **THE PROPOSED TREATMENT SYSTEMS ABILITY TO TREAT SEAWATER HAS NOT BEEN EVALUATED BY EITHER BENCHSCALE OR PILOT SCALE TESTING.**

The lack of bench scale of pilot scale testing of the proposed water treatment system raises concern over the systems ability to meet the performance standards established for the system. However, the EPA has not designated nor considered an alternative treatment system. The system proposed by the EPA is the simplest and cheapest system

of those currently available to treat the type of contaminants expected. Therefore, any failure of the water treatment system will result in costs that have not been included in the EPA's estimate nor can they be currently identified given the lack of data regarding the proposed systems ability to treat the effluent from the CDFs.

The water discharged from the CDFs will require treatment prior to discharge into the estuary. As noted in the Pilot Study, a small portion of the effluent (10 to 50 gallons per minute) was treated prior to discharge back into the estuary. The levels of suspended solids in the untreated effluent discharged from the CDF ranged from 61 to 152 mg/l in samples collected from the CDF effluent (USACE 1990). In order to reduce the suspended solids content of the effluent from the CDFs to the range identified above it appears that a polymer had to be added to the discharge over the weir separating the primary and secondary cells of the CDF. A polymer was selected by the USACE based on column settling tests and added as a flocculent to facilitate settling. In column settling tests one polymer (Magnifloc 1596C) was estimated to be able to reduce TSS loads between the two CDF cells by up to 82 %. However, subsequent testing of this polymer indicated that the polymer was toxic to the native aquatic organisms present in the estuary. Therefore, this polymer may be unsuitable for use during the full scale implementation of this remedy and another must be selected. There is no information available regarding whether a non-toxic polymer can be selected, or if so what the cost implications will be.

The treatment train selected for the treatment of the effluent from the CDFs has not been evaluated under actual field conditions that may be expected during the dredging and subsequent filling activities. Due to the presence of elevated levels of organics (associated with the discharge of sewage to the harbor), and the presence of chlorine due to treatment of seawater, the capabilities of the treatment system are suspect. The lack of bench and/or pilot scale test data and data regarding other possible options precludes any assessment of the cost associated with the failure and necessity to identify, test and design a water treatment system to replace the system currently identified in the preferred remedy. Given this unknown, the cost estimates for water treatment are just

a guess; they could range substantially higher if it is conceded that treatment to the effluent discharge levels established is even possible. If it is found that a water treatment system can't be practically designed to meet the EPA criteria, then there may be a consequent increase in environmental impact.

Furthermore, the cost EPA identified in the current remedy do not appear to contain any cost for disposal and/or regeneration of the carbon expended during the treatment of the effluent from the CDFs. Due to the lack of data available to regarding the performance of the water treatment system (specifically the usage rate for the carbon) a cost for the disposal and/or regeneration of the carbon can not be estimated.

LITERATURE CITED

Woodward-Clyde Consultants. 1987. Field investigation and analytical testing, New Bedford Superfund Site, New Bedford Massachusetts.

USACE. 1990. New Bedford Harbor Superfund Pilot Study. Evaluation of Dredging and Dredged Material Disposal. Final Report.

EBASCO 1990 - Draft final feasibility study for the estuary and lower harbor/bay. New Bedford Harbor, Massachusetts.

TABLE 1
ESTUARY AND LOWER HARBOR BAY
REMEDY COST ESTIMATE

COMPONENT	QUANTITY	EPA COST ESTIMATE TOTAL COST	REVISED COST ESTIMATE TOTAL COST
Purchase of land for CDFs	10	$ 0	$300,000[1]
Construction of CDFs			
CDF 1	1	$2,771,395	$5,550,000
CDF 1a	1	$2,843,555	$5,690,000
CDF 3	1	$4,832,978	$9,700,000
Dredging	30,800	$3,292,000	$5,200,000 - $5,900,000
Water Treatment		$4,761,000	$4,800,000-??[3]
Direct Costs		$18,500,928	$31,240,000
Indirect Costs			
Health & Safety		$238,050	$240,000
Administrative (6% of Direct Costs)		$1,110,056	$1,874,400[2]
Engineering		$1,850,000	$3,100,000[2]
Services		$2,775,000	$3,100,000[2]
Contractor Fees		$2,775,000	$4,700,000[2]
Contingency (20%)		$5,265,000	$8,800,000
Present Worth (at 5% for 6 years)		$26,723,000	$39,647,553
O & M Costs		$734,000	$734,000-??[4]
Monitoring Costs		$5,817,000	$5,800,000
TOTAL ESTIMATED COSTS		$33,274,000	$46,181,553

NOTES:

1. Costs presented for purchase of land for construction of CDFs based on per acre cost of approximately $30,000 per acre. Value of land obtained during conversation with commercial realtor from New Bedford, Massachusetts, July 1992.
2. Costs for administrative, engineering, services, and Contractor Fee based on standard EPA guidelines for Superfund work. These costs can be expected to increase as the capital costs associated with the implementation of the remedy increase.
3. Cost presented for the water treatment component under the "Revised Cost Estimate" heading represent the EPA costs estimate for water treatment using the system described in the EPA feasibility study. As discussed in the attached text, this cost may be substantially greater due to the proposed systems inability to treat the contaminants at the levels anticipated in the effluent discharged from the CDFs. Refer to the attached text.
4. Cost present for the Operation and Maintenance (O & M) of the CDFs under the "Revised Cost Estimate" heading are based on the EPA cost estimate for O & M. The actual O & M costs could increase by an unquantifiable amount given the likelihood of differential settlement and the repairs necessitated by such settlement. Refer to the attached text for a brief discussion of the concerns associated with the constructability of the CDFs in the proposed locations.

VI. COMPARISON OF EPA PROPOSED PLAN FOR REMEDIATION OF ESTUARY AND LOWER HARBOR/BAY AND DREDGING AND CAPPING THE OUTER BAY AND THE IN-PLACE CONTAINMENT REMEDIAL PLAN PROPOSED BY AVX IN 1989

COMPARISON OF EPA PROPOSED PLAN FOR REMEDIATION OF ESTUARY AND LOWER HARBOR/BAY AND DREDGING AND CAPPING THE OUTER BAY AND THE IN-PLACE CONTAINMENT REMEDIAL PLAN PROPOSED BY AVX IN 1989

An evaluation was performed to determine how the EPA proposed plan for New Bedford Harbor compared to the in-place containment remedial plan submitted by AVX 1989 (Balsam 1989) using the 9 Superfund criteria. These 9 criteria are (1) overall protection of human health and the environment, (2) compliance with ARARs, (3) long-term effectiveness and permanence, (4) reduction of toxicity, mobility, or volume, (5) short-term effectiveness, (6) implementability, (7) cost, (8) state acceptance, and (9) community acceptance.

OVERALL PROTECTION OF HUMAN HEALTH AND THE ENVIRONMENT

AVX believes that its plan for in-place containment is as protective of human health and the environment as the proposed EPA plan for dredging and containment in CDFs, and probably more so. While EPA argues that in-place containment will not permanently immobilize the contaminated sediments, it is clear from our analysis that the EPA plan is even less successful at doing so. Not only is substantially more PCB potentially volatilized into air adding to a potential human health risks but the disturbance of and handling of contaminated sediment by dredging within the Estuary/Lower Harbor and Outer Bay results in significant releases of PCBs to the water column and the dispersion of surrounding areas.

While it is true that the AVX in-place containment plan would result in capping a substantial amount of the upper estuary, it is true also that dredging of the area will have the same direct result on the biota presently living there and dredging and capping in the Lower Harbor and the Outer Bay in the EPA plan greatly extends the area of potential impact to marine and estuarine species. Under the EPA plan CDFs would be constructed on wetland areas, areas that would be

permanently taken for the CDFs while in the AVX plan 19 acres of new salt marsh and 22 acres intertidal mud flats would be created.

COMPLIANCE WITH ARARs

The contemplated remedial action under both the EPA and AVX plan will generally meet existing ARARs with the caveat that activities conducted as part of the proposed plans would be permitted because this is a CERCLA site.

LONG-TERM EFFECTIVENESS AND PERMANENCE

AVX believes that its in-place containment remedial plan is as effective and permanent in the long-term as is the EPA plan. In-place containment as being an impermanent solution due to the potential for disruption of the cap, EPA has proposed a plan which, with the CDFs being built on soft wetland soils, has a very high potential for failure and at a minimum a requirement for substantial and continuing maintenance to make them perform as they are designed to. AVX believes that the potential for differential settlement, slumping and failure, as well as the potential for displacement of contaminated sediment from the CDF in mud waves has a higher potential than does failure of a cap within the Estuary. AVX also believes the effort involved in maintaining the cap and, in fact, even restoring portions of it in the event of local disruption, will be far less than should be anticipated in maintenance requirements for the CDFs.

REDUCTION OF TOXICITY, MOBILITY, OR VOLUME

Both the AVX and EPA plan focus on reducing the mobility of the contaminated sediments. In this regard, as previously discussed, it is AVX's belief that in-place containment plan does it more effectively. Many arguments that EPA presents concerning the permanency of the CDF solution are unsupported and untested. Neither the AVX nor the EPA plan will reduce the volume of contaminated

sediments nor the toxicity except insofar as those contaminated sediments are dredged and incinerated from the Hot Spot.

SHORT-TERM EFFECTIVENESS

In regard to short-term effectiveness the AVX plan is clearly more effective. It can be conducted in a shorter time frame, involves direct impact to a smaller area, and minimizes potential release of PCBs, metals and other contaminants to the surrounding waters or for increased volatilization of PCBs.

IMPLEMENTABILITY AND COST

These two criteria have been included together since they are so interrelated; likely any plan could be implemented provided sufficient money is available. The AVX plan is clearly more implementable and less costly than is the EPA plan. EPA calculates it can implement its plan over approximately a 6-year period for the cost of approximately $33.2 million dollars in the Estuary and Lower Harbor and estimated cost of about $9.6 million for the Outer Bay, for a total of about $42,800,000.

AVX, however, has estimated that the remedial plan for the Estuary and Lower Harbor may cost as much as $46 million dollars or more depending upon how much additional engineering and maintenance is required for the CDF and whether EPA is successful in finding a water treatment system that will contain the criteria that it has prescribed. While the costs involved with the Outer Bay remediation are uncertain due to the lack of data, if we assume that the areas identified by EBASCO in the Supplemental Feasibility Study need to be remediated, Balsam's estimate is that the Outer Bay remedial plan may require an additional $45 million dollars. (That Outer Bay remedy includes construction of another CDF within the estuary. AVX does not believe that the even increasing the capacity of those CDFs presently planned will be sufficient to handle the additional Outer Bay sediments). AVX estimates that the EPA total plan for the

Estuary, Lower Harbor and Outer Bay could run approximately $90 million dollars.

In regard to implementability, for the most part both plans are implementable. However, in two areas there is a deficiency of information that raises questions about the implementability of the EPA plan. Although the EPA plan for construction of CDFs and treatment of water released from the contaminated sediments may be implementable, AVX believes that costs required to meet expectations have been significantly underestimated.

STATE ACCEPTANCE AND COMMUNITY ACCEPTANCE

While it seems clear from comments to date that the State and community feels that the EPA plan is more acceptable than the AVX plan, it is not clear whether the community would feel likewise once all the facts were known. Surely, the perspective of having three huge vaults permanently sited on the shores of an estuarian wetland, vaults which need continuing monitoring and maintenance, would seem to be objectionable to the community.

In summary, AVX believes that its in-place containment remedial plan offers a plan that is less costly, is more implementable, and will result in less potential for environmental impact to organisms and less potential for impact to human health.

ATTACHMENT A

PCB Cleanup Guidelines for the Estuary and Lower Harbor/Bay Sediments: Evaluation of a 50 ppm Cleanup Level

PCB Cleanup Guidelines for the Estuary and Lower Harbor/Bay Sediments:

Evaluation of a 50 ppm Cleanup Level

July 10, 1992

TERRA, Inc.
Toxicological Evaluations, Research & Analysis

PCB Cleanup Guidelines for the Estuary and Lower Harbor/Bay Sediments:

Evaluation of a 50 ppm Cleanup Level

July 10, 1992

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PCB Cleanup Guidelines for the Estuary and Lower Harbor/Bay Sediments:

Evaluation of a 50 ppm Cleanup Level

Executive Summary

The baseline risk assessment performed by E.C. Jordan (ECJ) and submitted to USEPA by Ebasco Services, Inc. adopted overly conservative exposure assumptions to determine risks associated with direct exposure to PCB-contaminated sediments and contaminated seafood from New Bedford Harbor. These assumptions include the sediment ingestion rate, the amount of New Bedford Harbor fish consumed by area residents, the toxicokinetic (bioavailability) factors for PCBs, the number of years that an infant or small child would contact site sediments or consume fish from the area, and the frequency of exposure to site sediments. The use of these overly-conservative exposure assumptions leads to an overestimation of the risks associated with site conditions.

Using more realistic exposure assumptions where appropriate, TERRA, Inc. has performed an evaluation of the proposed 50 ppm PCB clean-up standard by determining the excess lifetime cancer risks associated with child and adult beachcomber exposures to sediments containing PCB contamination at levels corresponding to the proposed standard. To calculate these risks TERRA, Inc. applied three different PCB cancer potency factors to the dosages derived from each exposure scenario. The first value, 7.7 $(mg/kg/day)^{-1}$, represents the USEPA's conservative estimate of PCB carcinogenicity, which was based on a study using a 60% chlorine PCB mixture but which suffers from serious methodological shortcomings. Although evidence for the carcinogenicity of 54% chlorine PCB mixtures is equivocal at best, and evidence for the carcinogenicity of 42% chlorine PCB mixtures is negative, the current position of the USEPA is that this conservative value should be used to calculate lifetime cancer risks associated with exposures to PCB mixtures with lesser chlorination. Given certain experimental design flaws associated with the study upon which the USEPA slope factor is based, and the fact that all pertinent PCB cancer bioassays have been re-evaluated according to the tumor pathology classification schemes currently used and recommended by the National Toxicology Program (IEHR, 1991), the use of the

USEPA cancer potency factor provides risk estimates which are unrealistically high and lacking in credibility. Therefore, TERRA has also calculated cancer risks based on two alternative cancer potency factor values, slope factors of 1.1 and 0.18 $(mg/kg/day)^{-1}$. These cancer potency factors were derived from an animal study of a 60% chlorine PCB mixture and are therefore still extremely conservative when applied to an assessment of the risks associated with 54% and 42% chlorine PCB mixtures. The 1.1 $(mg/kg/day)^{-1}$ value derived by TERRA, Inc. was based on a consideration of Schaeffer et al. (1984) and the recent reevaluation of this study's tumor incidence as reported by the Institute for Evaluating Health Risks (1991). The 0.18 $(mg/kg/day)^{-1}$ value also was derived from these two studies, using body weight as the animal-to-human scaling factor, rather than body surface area, which was used to determine the other two cancer potency factors. TERRA, Inc. believes that either of these two values provides a more reliable estimate of carcinogenic potency than the USEPA-derived value because they are based on more recent scientific developments in areas which affect the animal-to-man extrapolations of animal cancer bioassay data.

When the risks from exposure to contaminated sediments were calculated, all risks were near or below the $1x10^{-6}$ risk level. Risks were also determined for children and adults consuming seafood contaminated with low levels of PCBs. For all individuals the highest risks were below the $1x\ 10^{-4}$ risk range. For the seafood consumption scenario, the highest risks were calculated for the adults because their residence in the area could achieve the longest duration, i.e., 30 years. When the lifetime cancer risks posed by the longest exposure duration were calculated using either of the TERRA slope factors, the risk estimates typically fell within the 10^{-5} to 10^{-6} risk range. Based on these analyses and the conservative nature of these risk estimates, the adoption of the 50 ppm cleanup guideline for Area II and III sediments would provide a remediation goal for this area that is protective of human health.

1.0 Introduction

This report begins with an evaluation of the baseline risk assessment prepared by E.C. Jordan for Ebasco Services, Inc. on PCB contamination in the Upper Estuary of the Acushnet River and the New Bedford Harbor area. TERRA, Inc. evaluated the assumptions used by E.C. Jordan to determine exposure to PCBs through direct contact with sediments and through the ingestion of contaminated seafood. Assumptions found to be overly conservative or scientifically unjustifiable have been identified and discussed. Section 2.0 addresses assumptions related to body weight and surface area for potentially exposed persons, sediment and seafood ingestion rates, PCB toxicokinetic (bioavailability) factors, exposure intervals, and exposure frequency.

Section 3.0 of this report evaluates the human health risks associated with direct contact with sediments with a PCB concentration of 50 ppm and with ingestion of contaminated New Bedford Harbor seafood. Three different cancer potency factors can be used to calculate risks from exposure to PCBs from direct sediment contact and seafood ingestion. A discussion of the scientific merit of these three factors is included. This independent risk assessment demonstrates that excess cancer risk from sediment exposure and seafood ingestion falls within the range of risks considered acceptable by the United States Environmental Protection Agency.

2.0 The Ebasco Baseline Risk Assessment

2.1 General Comments

As part of the Remedial Investigation/Feasibility Study (RI/FS) services being performed by Ebasco Services Incorporated, E.C. Jordan (ECJ) developed a baseline risk assessment for New Bedford Harbor and submitted it to USEPA. This baseline risk assessment, Draft Final Baseline Public Health Risk Assessment; New Bedford Harbor Feasibility Study; August, 1989, was based on several assumptions, including:

- The sediment ingestion rate,
- The amount of New Bedford Harbor fish consumed by area residents,
- The toxicokinetic (bioavailability) factors for PCBs,
- The number of years that an infant or small child (ages 0-5 years) would have contact with site sediments or would consume fish from the bay, and
- The frequency of visits an individual might make to Areas I and II.

ECJ used overly conservative exposure estimates in conjunction with these assumptions, and the problems associated with these assumptions are discussed in more detail below. The ECJ baseline risk assessment also utilized a USEPA cancer potency factor for PCBs which is no longer scientifically reliable for two reasons: 1) it is based on an animal study using a flawed experimental protocol, and 2) the carcinogenic potency of PCBs in rats has recently been re-evaluated based on the tumor classification scheme currently endorsed by the National Toxicology Program. Thus, a thorough evaluation of this issue based on current scientific methodologies and principles would no longer generate a slope factor that is as high as the one ECJ used. This in turn means that the PCB cleanup levels proposed by ECJ are no longer accurate.

The ECJ risk assessment divided the New Bedford Harbor area into three distinctly different areas where direct contact with sediment could occur, and referred to these as Area I, Area II, and Area III. Area I includes the area between the Wood Street and Coggeshall Street bridges. This area includes the Upper and Lower Estuary of the Acushnet River and the Cove Area. Area II includes the area between the Hurricane Barrier and the Coggeshall Street Bridge and the following specific areas: Palmer Island, Popes Island, and Marsh Island. Area III is defined as the area south of the Hurricane Barrier and includes the Fort Rodman and Fort Phoenix Beach Areas. For convenience, the TERRA assessment refers to the same areas.

2.2 Specific Comments Concerning Exposure Assessment Variables

2.2.1 Body Weight and Surface Area Assumptions

Dermal dose is a function of surface area divided by body weight. The ECJ risk assessment utilized child body weight of 10 kg and a child surface area of 6,880 cm^2. However, data on body weight and surface area for children indicate that a child with a body weight of 10 kg is not likely to have a body surface area of 6,880 cm^2. Based on children representing the 50th percentile a body weight of 10 kg would represent 1-to-2 year old infant (USEPA, 1989). In contrast, a child with a body surface area of 6,880 cm^2 is likely to be a 3-to 4-year-old child Similar ages and percentiles should have been chosen for both estimates, as the values ECJ has selected minimize the body weight parameter and maximize the surface area parameter. As a result, the dermal PCB dose calculations provided in the ECJ risk assessment report (when considered on a mg/kg body weight basis) are unrealistically inflated. If ECJ is going select children ages 3-to-4 years of age as the average child to represent the 2-to-6 year old age group, then a body weight of 15 kg would have been more appropriate (USEPA, 1989).

2.2.2 Soil/Sediment Ingestion Rates

The sediment ingestion rate used in the ECJ risk assessment, 500 mg/day, is unrealistically high. According to the analysis of LaGoy (1987), 500 mg/day represents the 99th percentile or greater for soil ingestion rates in 1- to 4-year-old children and most likely applies to children with pica behavior. In contrast, studies by Clausing et al. (1987) and Binder et al. (1986) indicate that the soil ingestion rate for small children is 100 mg/day, and the USEPA (1989) has selected 100–200 mg of soil per day as the range of soil ingestion rates for the typical child and has suggested that 200 mg/day be the exposure estimate value to use when assessing the risks posed by Superfund sites.

More realistic soil ingestion rates for young children would be 24-38 mg/day for the average case and 65 mg/day for a worst case, based on recent soil ingestion studies by Calabrese and coworkers (Calabrese et al., 1989; Calabrese and Stanek, 1991). These studies utilized aluminum, silicon, yttrium, and zirconium tracers and reported the mean soil ingestion rate for children to be 24 mg/day. By considering titanium and vanadium, which are less reliable tracers because of their high

concentrations in food, the mean soil ingestion rate was still only 38 mg/day. In either instance, the 95% upper confidence limit on the mean value was about 65 mg/day, or about eight-fold less than the 500 mg/day value used by ECJ.

2.2.3 Seafood Ingestion

The ECJ risk assessment used adult ingestion rates of 115-227 grams per meal for seafood and 115 g/meal for 0-to 5-year-old children. These rates are likely to be considerably overestimated. The USEPA has recommended 140 g/day as the 90th percentile (reasonable worst case) value for local fish consumption, based on studies of U.S. west coast fishermen (USEPA, 1989). From the FDA Total Diet Study (Pennington, 1983), adult intake of fish and seafood is approximately 20.1 g/day for men and 14.9 g/day for women, with an average adult intake of 17.5 g/day. For a 2-year-old child, the intake level of the same foods is 5.0 g/day, or some 23 times less than the 115 g value assumed by ECJ.

2.2.4 Toxicokinetic (Bioavailability) Factors

ECJ used exaggerated and inconsistent toxicokinetic (bioavailability) factors in its risk assessment report. ECJ used 50% dermal bioavailability for sediments with PCB concentrations exceeding 1% (10,000 ppm) and used 7% for "moderately contaminated" sediments. Based on studies, however, the toxicokinetic (bioavailability) factor should be 1% or less for the dermal sediment exposure to PCBs.

Information describing PCB dermal bioavailability is limited. Wester et al. (1983) found a dermal bioavailability value of 21% for PCBs containing 42% chlorine. The Wester et al. study determined the value by applying PCBs to the skin of monkeys in solvent. However, because PCBs are structurally and chemically similar to tetrachlorodibenzo-*p*-dioxin (TCDD), TCDD can be used to predict the behavior of PCBs. Poiger and Schlatter (1980) indicated that the 24-hour dermal absorption of TCDD from a soil paste applied to rat skin ranged from 0.05-2.2%, and that the percent absorbed was dependent upon the applied dose (i.e., absorption decreased with a decrease in the level of soil contamination). Shu et al. (1988a and 1988b) reported the dermal absorption of TCDD to be 1%. These studies led the USEPA to adopt a 0.5% dermal absorption factor for TCDD (USEPA, 1987).

In addition to evidence provided in studies of TCDD bioavailability, there are other reasons for adopting a dermal soil bioavailability value of 1% or less for PCBs. In animal studies, absorption was measured over a continuous 24-hour period. Humans, however, tend to wash before meals; after returning from the beach, etc., thus interrupting dermal absorption. Also, in the rat bioavailability studies, the site of chemical application was occluded to prevent interference with absorption. Furthermore, rodent skin is three to ten times more permeable than human skin (Bartek et al., 1972; Wester and Maibach, 1987).

The ECJ risk assessment incorrectly used dermal information from animal studies. First, the risk assessment averaged dermal bioavailability across species. Dermal absorption studies of PCBs cited in the ECJ risk assessment reported that bioavailability was 12% in a human skin section, 21% in monkeys, 33-54% in guinea pigs, and 55% in mice. This is consistent with the observation that dermal absorption is greater in rodent skin than in primate skin. When the results of the human (12%), monkey (21%), guinea pig (33 and 56%), and mouse (55%) studies are averaged, the average value is 30 or 36%, not 41%, as reported by ECJ. Second, if the 13% conversion factor developed in the ECJ risk assessment (page B-6) is applied to the monkey or human studies of Wester and Maibach (1987), then PCB dermal bioavailability becomes:

0.13 (conversion factor) x 0.21 (dermal absorption in monkeys) = 2.7%, and
0.013 (conversion factor) x 0.12 (dermal absorption in humans) = 1.6%,

rather than the 7% value determined by ECJ. The estimated ECJ dosages would be reduced about 4.4-fold if a 1.6% bioavailability factor were used, and about 14-fold if the USEPA-recommended value of 0.5% for TCDD were used.

ECJ also incorrectly determined the oral bioavailability value for sediments and soil. Studies of soil-bound TCDD indicate oral bioavailability is less than 50%. The USEPA has adopted a 30% gastrointestinal absorption value for PCBs in soil (USEPA, 1986). Pages B-3 through B-5 of Appendix B to the ECJ risk assessment list the pharmacokinetic studies reviewed to evaluate oral bioavailability of PCBs. Of these, five were in agreement: the minimum percent absorbed from oral doses of PCBs ranged from 85-94%, with an average of 90%. However, in the Norback et al.

(1978) study, the oral bioavailability ranged from 13-41%, with an average of 27%. ECJ chose to include this value in its estimation of oral bioavailability, although this study is contradicted by five other studies and would imply that all highly lipophilic compounds such as PCBs are poorly (and variably) absorbed in the gastrointestinal tract. As a result of including this value, the oral bioavailability is lowered by more than 10%. This in turn increases estimates of the dermally absorbed dose, since ECJ used estimates of oral bioavailability to correct for the absorbed dose by the dermal route.

Last, the authors of the ECJ risk assessment failed to consider certain confounders. All of the animal PCB studies used corn oil as a vehicle to deliver a dose. However, mineral or vegetable oils act as cathartics and may slow or decrease absorption. In addition, elimination via the bile into feces may lead to an underestimation of absorption in some of these studies, especially if the animals were monitored for several days. Thus, the oral bioavailability of PCBs when introduced in a rodent's diet is probably closer to 100% and is not as low as the 80% value ECJ has calculated.

2.2.5 Exposure Intervals

The ECJ risk assessment includes one potentially exposed group of persons up to 5 years old. However, 5 years of potential exposure is not realistic for this age group. Infants 0 to 2 years old are unlikely to spend much time, if any, unsupervised and outdoors, and children in this age group would not be allowed to spend any time at the unattractive industrial portions of the Area II shoreline. Even if taken to the beaches in Area III, the infants would spend most of their time in strollers, on blankets, etc. A more realistic exposure scenario would consider potential access to the shoreline starting at about age three. Thus, the exposure interval for small children (0 to 5 years of age) is likely to include, at most, 2 to 3 years. Furthermore, infants 0 to 2 years of age are not likely to eat fish meals. Breastmilk, infant formula, and baby foods are the most probable foods for this age group. Again the exposure interval for this age group is only 2 to 3 years.

2.2.6 Exposure Frequency

The ECJ risk assessment assumed the frequency of exposure to be 20 and 100 times per year under the moderate and conservative exposure scenarios, respectively. These numbers correspond to 1 and 5 days per week during the warmer months. Because school activities would limit play time during two of the five warmer months, it is more reasonable that 12 and 60 visits would represent exposure frequency under the moderate and conservative scenarios, respectively (i.e., 1 and 5 visits per week during the 12 nonschool weeks of summer). In addition, for several reasons, it does not seem reasonable to assume 20-100 swimming/wading events in Area I: warning signs are posted (pages 2-9 and 2-10 of the ECJ risk assessment); swimming in the mud flats is unlikely; the Acushnet River is "dirty" with brown and pungent water, oil stains, and trash; and industry in this area discourages any recreational use of these shorelines.

3.0 The TERRA, Inc. Assessment of Risks Posed by a 50 ppm PCB Sediment Remedial Action Level

3.1 Exposure Assessment

In setting PCB cleanup levels for New Bedford Harbor, it is necessary to characterize exposures to PCBs which may occur as a result of contact with sediment during beachcombing or shellfishing activities and from the ingestion of fish and shellfish caught from banned areas. Such exposures cannot be completely characterized because of insufficient available data, making assumptions necessary. However, it is important that the exposure assumptions conservatively and realistically predict exposure without grossly overestimating PCB intake.

Existing information suggests that contact with sediments containing PCBs may occur and this exposure scenario assumes that direct contact with sediment from various activities will occur. In this assessment only the risks associated with sediment exposure in Areas II and III will be considered because a remedial action has already adopted for those portions of Area I where ECJ found unacceptable risks from exposure to sediment. These exposure scenarios do not consider any PCB intake which might occur from the ingestion of clams that might be taken when a person visits one of these areas. PCB exposure resulting from ingestion of seafood is specifically addressed under the seafood ingestion exposure scenario (*see* Section 3.1.3).

3.1.1 Exposure Variables Selected For The Beachcomber Exposure Scenario

The beachcombing exposure scenario assumes sediment exposure from walking on or playing in the sediments of a sandy beach or mud flat. In Area II, older children (ages 7 to 16) and adults (ages 17 to 70) are assumed to come into contact with PCBs in sediments. In Area III, small children (3 to 6 years of age) have also been included. The assumed average body weights are 39.6 kg for older children (7 to 16 years old), 70 kg for adults, and 18 kg for young children (average for the 50th percentile body weight of male and female children ages 4–6 years old).

The fraction of a lifetime during which exposure to sediments occurs is assumed to vary with the length of time a person lives in the Greater New Bedford Harbor Area. Recent investigations by the USEPA have shown 9 and 30 years to be the 50th and 90th percentiles, respectively, for time spent at a single residence (USEPA, 1989). An exposure duration of 9 years was assumed for the older children (ages 7 to 16); this approximates the entire duration of this age group and is the duration they would live in the New Bedford Harbor area at the 50th residence percentile. For the younger children that might visit Area III (ages 3 to 7), the entire interval during which they might visit the beach under much more limited supervision than infants is only 4 years.

Exposure to sediment for all age groups was assumed to occur from mid-May to mid-September. Due to limited recreational opportunity, the number of yearly visits to Area II was assumed to be 12 and 24 visits (i.e., 1 to 2 visits per week for 12 summer weeks). The number of visits assumed to occur in Area III was 18 and 54 visits. In each case, the lower number was considered to represent an average or typical number of visits. A reasonable maximum number of visits is represented by the higher number.

Ingestion of sediment is assumed to result from incidental hand-to-mouth activity. Estimates of incidental ingestion of soil range from 24 to 65 mg/day for small children (ages 2 to 6). For older children and adults, the 50 mg/day value reported in studies by Calabrese et al. (Calabrese et al. 1989; Calabrese and Stanek, 1991) will be used. For the purpose of selecting sediment ingestion rates, soil and sediment are assumed to have an equal potential for ingestion.

The oral bioavailability of PCBs from a sand, soil, or sediment matrix has not been determined. However, the USEPA (1986b) has assumed the oral bioavailability of PCBs in soils be 30%, and this number has been adopted for the sediments at New Bedford Harbor.

For all exposure estimates, it was assumed that contact with sediment will occur as a person wearing shorts walks on the shore and digs in or plays with the sediment. The feet, hands, forearms, and lower legs make up 7%, 5.2%, 5.9%, and 12.8% of the total body surface, respectively (USEPA, 1989). If the total (30.9%) of these body surface areas comes in contact with area sediments, then the

corresponding surface areas available for contact are 5,562 cm^2 for adults, 3,955 cm^2 for older (7- to 16-year-old) children, and 2,323 cm^2 for younger children (based the average of the 50th percentile surface areas for male and female children ages 4-6 years), given that total body surface area is 18,000 cm^2 for an adult, 12,800 cm^2 for an older child, and 7,517 cm^2 for the average younger child. Estimates of soil adherence to skin range from 0.2 to 1.0 mg/cm^2 (USEPA, 1992). Due to the sandy nature of sediments present in Areas II and III, the mean value of 0.6 mg/cm^2 was selected as a conservative estimate of skin adherence for this type of sediment.

Since PCBs are lipophilic compounds, it can be expected that binding to sediment will greatly reduce their dermal bioavailability. The dermal bioavailability of PCBs in soil or sediment has not been measured, but the dermal bioavailability of TCDD in soil has been measured in rats. Data by Poiger and Schlatter (1980) indicate that TCDD absorption from a soil paste ranges between 0.05 and 2% over a 24-hour treatment period, depending upon the applied dose. Shu et al. (1988b) determined that TCDD absorption from a soil paste was approximately 1% after 24 hours. In the discussion of their results, Shu et al. (1988b) recognized the fact that the skin of the rat is more permeable to many chemicals than is human skin, and that a 1% absorption rate of TCDD would probably represent the upper limit of human dermal absorption. Because PCBs are structurally similar to TCDD and have a similar tendency to be strongly sorbed to soil (Chou and Griffin, 1986), a 0.5% dermal absorption per 12 hours is used. The twelve hour estimate represents a reasonably conservative estimate of the length of time the skin would be in contact with sediment. As described in Section 2.2.4, humans interrupt dermal absorption by washing; in rat bioavailability studies, the site of chemical application was occluded to prevent interference with absorption; and rodent skin is three to ten times more permeable than human skin (Bartek et al., 1972; Wester and Maibach, 1987).

The exposure variables discussed in this section are summarized in Table 1. These variables are used in the equations presented in the next section (Section 3.1.2), along with the assumed PCB sediment concentration (50 ppm, the proposed cleanup standard), to determine the ingestion and dermal doses of PCBs from sediment.

Table 1
Exposure Variables for the Beachcomber Scenario

Exposure Variable	Symbol	Receptor		
		Young children	Older children	Adults
Concentration of PCBs in sediment (mg/kg)	C_s			
Area II		21	21	21
Area III		4	4	4
Body weight (kg)	BW	18	39.6	70
Years exposed	YE	4	9	9 and 30
Number of exposure events per year	F			
Area II		12 and 24	12 and 24	12 and 24
Area III		18 and 54	18 and 54	18 and 54
Amount of soil ingested per exposure event (mg/event)	IR	65	50	50
Fraction of ingested PCBs absorbed	AF_{ing}	0.30	0.30	0.30
Exposed skin surface (cm^2)	SA	2,323	3,955	5,562
Amount of sediment adhering to skin (mg/cm^2)	SC	0.6	0.6	0.6
Fraction of PCBs absorbed from skin	AF_{derm}	0.005	0.005	0.005

Note: Area I was excluded from consideration in this risk assessment as a remedial action for this area has already been developed.

3.1.2 Calculation of PCB Ingestion and Dermal Absorption

The following equation is used to calculate the gastrointestinal absorption of ingested PCB from sediment:

$$I_{ing} = C_s \times IR \times AF_{ing}$$

where:

I_{ing}	is the amount of ingested PCB absorbed per exposure event (mg/exposure event)
C_s	is the concentration of 54% or 42% chlorine PCB mixture in sediment (mg PCB/kg soil)
IR	is the amount of sediment ingested per exposure event (kg/exposure event)
AF_{ing}	is the fraction of PCB absorbed (unitless)

The equation used to determine the amount of PCBs dermally absorbed from sediment is as follows:

$$I_{derm} = C_s \times SA \times SC \times AF_{derm}$$

where:

I_{derm}	is the amount of PCB dermally absorbed per exposure event (mg/exposure event)
C_s	is the concentration of 54% or 42% chlorine PCB mixture in sediment (mg PCB/kg soil)
SA	is the amount of skin surface exposed (cm^2)
SC	is the amount of sediment adhering to skin (kg/cm^2)
AF_{derm}	is the dermal absorption coefficient for PCBs (unitless)

The total PCB intake per exposure event is determined by summing the above two equations:

$$I_{total} = I_{ing} + I_{derm}$$

where:

I_{total} is the total absorbed PCB intake per exposure event (mg/exposure event)

I_{ing} is the amount of ingested PCB absorbed per exposure event (mg/exposure event)

I_{derm} is the amount of PCB dermally absorbed per exposure event (mg/exposure event)

The chronic daily PCB intake, averaged over a lifetime, is calculated using the following equation:

$$CDI_{sed} = \frac{I_{total} \times F \times YE}{BW \times YL \times DY}$$

where:

CDI_{sed} is the chronic daily intake of PCBs from exposure to sediment (mg/kg/day)

I_{total} is the total absorbed PCB intake per exposure event (mg/exposure event)

F is the frequency of events per year (events/year)

YE is the number of years a person is exposed (years)

BW is the body weight (kilograms)

YL is the number of years in a lifetime (70 years)

DY is a conversion factor (365 days/year)

Using these equations and the assumptions presented for the beachcomber scenario (Table 1), the chronic daily PCB intakes can be calculated and are presented in Table 2.

Table 2

Chronic Daily Intake Values for PCBs from Sediment

Area (Exposure Frequency, Duration)	Chronic Daily Intake (mg/kg/day)		
	Young children	Older children	Adults
Area II			
(12 events per year, 9 years)	–	6.02E-08	4.02E-08
(24 events per year, 9 years)	–	1.20E-07	8.04E-08
(12 events per year, 30 years)	–	–	1.34E-07
(24 events per year, 30 years)	–	–	2.68E-07
Area III			
(18 events per year, 4 years)	1.83E-08	1.72E-08	–
(54 events per year, 4 years)	5.49E-08	5.16E-08	–
(18 events per year, 9 years)	–	–	1.15E-08
(54 events per year, 9 years)	–	–	3.44E-08
(18 events per year, 30 years)	–	–	3.83E-08
(54 events per year, 30 years)	–	–	1.15E-07

3.1.3 Exposure Variables for Seafood Ingestion

Ingestion of locally-caught fish and shellfish represents a potential pathway of PCB exposure for persons of the Greater New Bedford Harbor area. It is well known that PCBs bioaccumulate in fish and shellfish. Although there is a potential for PCB exposure from ingestion of locally-caught seafood, such exposure is not believed to be an important PCB exposure pathway for the majority of residents in

the Greater New Bedford (GNB) Harbor area. The findings of "The Greater New Bedford PCB Health Effects Study 1984-1987" (GNBHES) indicated that among the "prevalence" group (selected randomly from the Greater New Bedford Harbor populace), the rate of consumption of locally caught fish is quite low. Only 4.2% of the prevalence group was described as catching their own fish. Serum PCB levels were also low in this group. The GNBHES also studied a smaller group of persons known as the "enrichment" group. The "enrichment" group contained more persons (35%) who reported catching their own fish than did the "prevalence" group. However, the authors also concluded that "Almost all individuals who were identified as being at the greatest risk of exposure via contaminated seafood intake had relatively low serum PCB levels."

The extent to which local fishermen may violate the Massachusetts Department of Public Health fishing closure order is not known, but a small group of local fishermen may do so. These individuals may experience relatively higher body burdens of PCBs. Again, consumption of locally caught fish was not conclusively associated with increased serum PCB levels, but the small group of local persons who catch and eat locally caught seafood (presumably from the closed part of New Bedford Harbor) can serve as the most conservative receptors in the seafood consumer exposure scenario. The following calculations apply to this worst-case scenario.

Receptors in the seafood consumer scenario were assumed to be an adult weighing 70 kg, an older child (7 to 16 years of age) weighing 39.6 kg, and a younger child (2 to 6 years of age) weighing 16.3 kg. As in the beachcombing exposure scenario, the fraction of a lifetime for exposure to fish and shellfish from Greater New Bedford Harbor was assumed to vary. Recent investigations by the USEPA have shown 9 and 30 years to be the 50th and 90th percentiles, respectively, for time spent at a single residence (USEPA, 1989). For the older child, fish consumption was assumed to occur for nine years (from ages 7 to 16). The younger child was assumed to be exposed for four years (ages 3 to 7).

The GNBHES identified seafood preferences and consumption trends among "prevalence" and "enrichment" groups, but apparently did not attempt to quantify fish and shellfish intake except to estimate the frequency of lobster consumption, amount not specified. In the absence of information specific to the Greater New Bedford area, certain assumptions were made regarding seafood intake. Reported estimates of seafood consumption vary widely for reasons which are not clear, although differences in fish and seafood consumption from one area of the United States to another may be explained by differences in culture or climate.

Several studies have addressed seafood consumption. Javitz (1980) estimated recreational and commercial fish consumption for adults in New England based on surveys conducted twice per month for an entire year. For adults, fish consumption values were reported to be 16.3 g/day and 46.5 g/day for the mean and upper 95th percentile, respectively. Data specific to children in New England were not available. However, nationwide estimates for the mean and upper 95th percentiles for children aged 0-9 years of age were 6.2 g/day and 16.5 g/day, respectively. Estimates for children aged 10-19 were 10.1 g/day and 26.8 g/day for the mean and 95th percentiles, respectively. Considerably higher estimates of fish intake have been reported in other studies. For example, Puffer (1981, as cited in USEPA, 1989) determined fish intakes for recreational fishermen in Los Angeles, California to be 36.9 g/day and 225 g/day for the 50th and upper 90th percentiles, respectively. In another U.S. west coast survey, calculated intakes of recreationally caught fish for persons fishing in Commencement Bay at Tacoma, Washington were estimated to be 23 g/day for the 50th percentile and 54 g/day for the upper 90th percentile.

In the west coast studies, the highest intake of recreationally caught fish (225 g/day) was estimated based on a survey taken in an area having a mild year-round climate. Due to considerable differences in climate between areas surveyed on the west coast and the New Bedford Harbor area, the west coast studies of recreational fish consumption were not considered applicable to the Greater New Bedford Harbor area. Data derived by Javitz (1980) specific to the New England area were considered more appropriate for estimating fish and shellfish consumption in the Greater New Bedford Harbor area. These data are also very consistent with the numbers derived from the FDA Total Diet Study (Pennington, 1983).

No quantitative information was available for adequately characterizing the fraction of the local fishermen's diet which was comprised of seafood caught in the closed areas of New Bedford Harbor. Approximately 35% of the "enrichment" group indicated that their primary source of fresh seafood was their own catch. The USEPA estimated that 20% of all seafood ingested by consumers was obtained from their own catch (USEPA, 1989). To be conservative, it also was assumed that 50% of all seafood ingested by consumers was obtained from their own catch taken from Areas I, II, III, and IV. The following calculations are based on the USEPA's reasonable estimate of 20% local fish and on a worst-case assumption of 50%.

Balsam Environmental (Balsam, 1989) has calculated the range of concentrations of PCBs expected to be found in lobster, winter flounder, and clams in Areas II, III, and IV following cleanup to the 50 ppm remediation guideline. Based on the ranges of concentrations, and using the same approach that ECJ used to calculate PCB concentrations in the edible portions of winter flounder, the geometric mean of the lower concentration range for all three species (lobster, winter flounder, and clams) was 0.114 ppm and the geometric mean of the lower concentration range for these three species was 0.17 ppm. Thus, these two PCB concentrations were used in the final chronic daily intake calculations to derive the lifetime average daily PCB intakes for seafood consumption. Since cooking appears to decrease PCB concentrations 60-70%, a factor of 0.5 (50%) was conservatively used to lower PCB concentrations in cooked lobster, winter flounder, and clams (Humphrey, 1976).

The exposure variables discussed in this section are summarized in Table 3. These variables are used in the equations presented in the next section (Section 3.1.4) to determine the intake of PCBs from New Bedford Harbor seafood.

Table 3

Exposure Variables for the Seafood Ingestion Scenario

Exposure Variable	Symbol	Receptor		
		Young Children	Older Children	Adults
Mean PCB concentration in seafood from Areas II and III if remediated to ≤ 50 ppm PCBs in area sediments.	C_{sf}	0.114 and 0.17 ppm	0.114 and 0.17 ppm	0.114 and 0.17 ppm
Amount of seafood ingested per day (kg/day)	SI	0.0062 and 0.0165	0.0101 and 0.0268	0.0163 and 0.0465
Fraction of seafood from closed areas*	FI	0.2 and 0.5	0.2 and 0.5	0.2 and 0.5
Years exposed	YE	4	9	9 and 30

* USEPA (1989)

3.1.4 Calculation of PCB Intake from Seafood

The daily intake of PCBs from ingestion of seafood is calculated using the following equation:

$$I_{sea} = C_{sf} \times CL \times SI \times FI$$

where:

I_{sea} is the amount of PCBs ingested per day from seafood (mg/day)

C_{sf} is the average concentration of PCBs in seafood (mg/kg or ppm)

CL is the fraction of PCBs remaining after cooking losses (unitless, 0.5)

SI is the amount of seafood ingested per day (kg/day)

FI is the fraction of all ingested seafood which is obtained from local catch (unitless, 0.2–0.5)

The chronic daily intake of PCBs from seafood is then calculated using the equation below:

$$CDI_{sf} = \frac{I_{sea} \times YE}{BW \times YL}$$

where:

CDI_{sf}	is the chronic daily intake of PCBs from seafood ingestion (mg/kg/day)
I_{sea}	is the amount of PCBs ingested per day from seafood (mg/day)
YE	is the number of years a person is exposed (years)
BW	is the lifetime average body weight (assumed to be 70 kg)
YL	is the number of years in a lifetime (70 years)

Chronic daily intake values for typical seafood consumption (based on a geometric mean seafood PCB concentration no lower than 0.114 mg/kg) and reasonable worst case seafood consumption (based on a geometric mean seafood PCB concentration of no higher than 0.17 mg/kg, see page 17 for discussion) were calculated using the assumptions presented in Table 3 and the above two equations. These values are listed separately for young children, older children, and adults in Tables 4A, 4B, and 4C, respectively.

Table 4A

Chronic Daily Intake Values for PCBs from Seafood: Young Children

Conditions (C_{sf}, SI, FI, YE)	Chronic Daily Intake (mg/kg/day)
0.114 mg/kg, 0.0062 kg, 0.2, 4 years	5.8E-08
0.114 mg/kg, 0.0062 kg, 0.5, 4 years	1.4 E-07
0.114 mg/kg, 0.0165 kg, 0.2, 4 years	1.5E-07
0.114 mg/kg, 0.0165 kg, 0.5, 4 years	3.8 E-07
0.17 mg/kg, 0.0062 kg, 0.2, 4 years	8.6 E-08
0.17 mg/kg, 0.0062 kg, 0.5, 4 years	2.2 E-07
0.17 mg/kg, 0.0165 kg, 0.2, 4 years	2.3 E-07
0.17 mg/kg, 0.0165 kg, 0.5, 4 years	5.7 E-07

Table 4B

Chronic Daily Intake Values for PCBs from Seafood: Older Children

Conditions (C_{sf}, SI, FI, YE)	Chronic Daily Intake (mg/kg/day)
0.114 mg/kg, 0.0101 kg, 0.2, 9 years	2.1 E-07
0.114 mg/kg, 0.0101 kg, 0.5, 9 years	5.3 E-07
0.114 mg/kg, 0.0268 kg, 0.2, 9 years	5.6 E-07
0.114 mg/kg, 0.0268 kg, 0.5, 9 years	1.4 E-06
0.17 mg/kg, 0.0101 kg, 0.2, 9 years	3.2 E-07
0.17 mg/kg, 0.0101 kg, 0.5, 9 years	7.9 E-07
0.17 mg/kg, 0.0268 kg, 0.2, 9 years	8.4 E-07
0.17 mg/kg, 0.0268 kg, 0.5, 9 years	2.1 E-06

Table 4C

Chronic Daily Intake Values for PCBs from Seafood: Adults

Conditions (C_{sf}, SI, FI, YE)	Chronic Daily Intake (mg/kg/day)
0.114 mg/kg, 0.0163 kg, 0.2, 9 years	3.4 E-07
0.114 mg/kg, 0.0163 kg, 0.5, 9 years	8.5 E-07
0.114 mg/kg, 0.0465 kg, 0.2, 9 years	9.7 E-07
0.114 mg/kg, 0.0465 kg, 0.5, 9 years	2.4 E-06
0.17 mg/kg, 0.0163 kg, 0.2, 9 years	5.1 E-07
0.17 mg/kg, 0.0163 kg, 0.5, 9 years	1.3 E-07
0.17 mg/kg, 0.0465 kg, 0.2, 9 years	1.5 E-06
0.17 mg/kg, 0.0465 kg, 0.5, 9 years	3.6 E-06
0.114 mg/kg, 0.0163 kg, 0.2, 30 years	1.1 E-06
0.114 mg/kg, 0.0163 kg, 0.5, 30 years	2.8 E-06
0.114 mg/kg, 0.0465 kg, 0.2, 30 years	3.3 E-06
0.114 mg/kg, 0.0465 kg, 0.5, 30 years	8.6 E-06
0.17 mg/kg, 0.0163 kg, 0.2, 30 years	1.7 E-06
0.17 mg/kg, 0.0163 kg, 0.5, 30 years	4.2 E-06
0.17 mg/kg, 0.0465 kg, 0.2, 30 years	4.8 E-06
0.17 mg/kg, 0.0465 kg, 0.5, 30 years	1.2 E-05

3.2 Health Risks Associated with a 50 ppm Sediment Cleanup Level

As a remedial alternative, Upper Estuary sediments with PCB concentrations in excess of 50 mg/kg would be removed by dredging. The combined remedial actions would limit future human contact to sediments containing 50 ppm or less of PCB mixtures. Sections 3.2.3 and 3.2.4 include the risks associated with direct contact with sediments and seafood ingestion in Areas II and III after all contaminated sediments have been remediated to ≤ 50 ppm.

3.2.1 PCBs Vary in Toxic Potential

In establishing clean-up levels for PCBs, ECJ and the USEPA have chosen to treat PCBs as a single entity. That is, they have assumed that the toxic potential of PCBs in a given situation is a function solely of the PCB concentration and not its congener composition. Such an approach may be expedient, but it ignores a fundamental conclusion that inevitably arises after any comprehensive review of the toxicological literature for PCBs: PCB congeners, or commercial mixtures of congeners, can differ markedly in toxicity. Specifically, this is relevant to the toxic endpoint used by the USEPA in setting clean-up standards for PCBs, i.e., the carcinogenicity of PCBs in rodent bioassays.

PCBs in commercial mixtures with chlorine contents of 42%, 54%, and 60% have been studied in several chronic rodent bioassay studies. Three bioassays of 60% chlorine PCB mixtures in rats have yielded positive results (Kimbrough et al., 1975; Schaeffer et al., 1984; Norback and Weltman, 1985). A bioassay of 54% chlorine PCBs conducted by the National Cancer Institute was interpreted as negative (NCI, 1978). While it has been suggested that the results of this bioassay might be considered as weakly positive (Morgan et al., 1981; Ward, 1985), a recent review conducted by the Institute for Evaluating Health Risks (IEHR, 1992) has confirmed that this bioassay was negative (Experimental Pathology Laboratories, Inc., 1992). A bioassay of a commercial 42% chlorine PCB mixture (Clophen A30) conducted in rats was also negative: no significant increase in carcinomas was found (Schaeffer et al., 1984). Thus, among the various commercial PCB mixtures tested in lifetime rodent bioassays, only the most highly chlorinated mixture (60% chlorine PCBs) has been shown to be carcinogenic.

Commercial PCB mixtures of 60% chlorine content (e.g., Aroclor 1260) comprised only about 12% of domestic PCB manufacture and use, and it is therefore unlikely that more than a small fraction of PCBs introduced into the environment was of 60% chlorine content. Despite this, the cancer potency factor derived from the bioassay of 60% chlorine PCBs is used by USEPA to develop clean-up guidelines for *all* PCBs, regardless of the extent of chlorination. This approach to setting PCB clean-up levels is beyond conservative; it is unreasonable.

The USEPA clean-up guidelines should recognize the differential toxicity of PCB mixtures if they are to be scientifically defensible. Environmental samples of PCB "weather" and, through processes such as microbial dechlorination, the relative congener composition may change from that in the original commercial mixture. Also, some samples may contain PCBs from more than one commercial mixture. To deal with these factors, the risks should be evaluated based upon the presence of specific congeners most responsible for PCB toxicity (USEPA, 1991). Unfortunately, those congeners have not been identified as yet, nor have the requisite dose-response relationships for them been established. Further complicating this approach, studies have shown that antagonistic as well as additive interactions occur. Until an approach can be developed which accurately combines both additive and antagonistic interactions, the reliability of the congener-specific analysis will be limited, and a direct, empirical measure of the mixture's potency for a given toxic endpoint is best for risk assessment purposes. The most reliable method for evaluating the toxic potential for PCB mixtures in environmental samples is to assess the toxic potential of the commercial mixture they most closely resemble. This approach is practical based on current knowledge and technology, unlike the approach based upon specific congeners or congener fractions. This approach is clearly preferable to the crude assumption that all environmental PCB mixtures are (or are toxicologically equivalent to) 60% chlorine PCBs, when this is clearly not the case.

It would be appropriate to develop sediment clean-up standards for PCBs which depend upon the extent of chlorination. When PCB contamination resembles commercial 60% chlorine mixtures, the clean-up standard should be based upon carcinogenic risk and the appropriate potency factor. For contamination with PCBs containing less chlorination, two approaches for determining clean-up standards are possible. One approach would be based upon an interpretation of the animal cancer bioassay data, which indicates a qualitative difference in carcinogenicity among PCB mixtures. Such a distinction has been made, for example, by the State of California which, in its Safe Drinking Water Act, recognizes only 60% chlorine PCBs as carcinogenic. Using this approach, clean-up standards for PCBs with less than 60% chlorine content would be based upon non-cancer end-points.

The second approach assumes that all PCBs are carcinogenic. Since bioassays of PCBs with average chlorine content less than 60% (i.e., 42% and 54%) were negative, the cancer potency of these congeners under this assumption must be less than that of 60% chlorine PCB mixtures. Rodricks (1984) has estimated that 42% and 54% chlorine PCBs, if carcinogenic, would have cancer potencies at least a factor of 10 less than 60% chlorine PCBs. Using this approach, cleanup levels for PCBs with less than 60% chlorine content would be calculated using potency factors at least ten-fold less than the potency factor used to set levels for 60% chlorine PCBs.

It is not unusual to recognize differential toxicity among members of a chemical class for regulatory purposes. For instance, some PAHs are regarded as non-carcinogenic and others as carcinogenic for purposes of determining risk-based acceptable concentrations. Formerly, the cancer potency factor for the most potent member of the class (i.e., benzo(a)pyrene) was used in calculating cancer risks for all carcinogenic PAHs. This approach was once considered conservative, but it now has been abandoned for the most part. Instead, different potency factors for individual PAHs have been derived in recognition of their different potencies. Another example is the polychlorinated dibenzodioxins. Dioxin toxicity varies with chlorine content, as does PCB toxicity. Toxicity equivalency factors (TEFs), based upon the degree of dioxin chlorination, are used to correct for differences in toxic potency among dioxin congeners. Such approaches would be justifiable for PCBs.

3.2.2 Potency Factors for Calculating Cancer Risks from PCBs

The USEPA Cancer Potency Factor

The USEPA cancer potency factor for PCBs, 7.7 mg/kg/day^{-1}, is based on a linearized multistage model which used combined liver carcinoma and neoplastic nodule incidence from a bioassay of 60% chlorine PCBs in rats and used body surface area as the scaling factor for dose extrapolation to humans (Norback and Weltman, 1985). One of the principal limitations of this study is that many of the rats had a liver lobe surgically removed during the course of the bioassay to study progression of hepatic lesions. Such surgical removal of the liver, termed *partial hepatectomy*, is a well-known promotional stimulus and could have influenced the outcome of the bioassay. Also, varying PCB dosages were used during the bioassay.

These two weaknesses make the Norback and Weltman (1985) study of questionable value for quantitative risk assessment purposes.

The appropriate scaling factor for animal-to-human extrapolation of carcinogen doses has been widely discussed in the scientific literature. Proposed scaling factors have exponentials of 0.66 (corresponding to extrapolation based upon body surface area) and 1.00 (corresponding to extrapolation based upon body weight). The FDA has endorsed body weight extrapolation for carcinogens, while the USEPA prefers surface area extrapolation. Dogmatic adherence to either scaling factor is difficult to justify scientifically, and the best scaling factor for an individual carcinogen is probably a function of a number of variables, including mechanism of carcinogenicity. Use of surface area as an appropriate scaling factor can be supported by studies of alkylating agents, for which carcinogenic action is thought to result from genotoxicity. Empirical observations of these agents support body surface area as the best means for extrapolation. However, for other carcinogens, including many compounds whose cancer effects appear to arise from non-genotoxic mechanisms, better animal-to-human correlations are obtained when body weight is used to normalize the dose. PCBs are included among these compounds (Allen et al., 1987).

Alternative Cancer Potency Factors

An alternative potency factor for 60% chlorine PCBs can be developed based on the tumorigenicity data of Schaeffer et al. (1984), the most defensible of three available studies. This study used a PCDF-free PCB mixture. It also was conducted for longer than required by National Toxicology Program (NTP) guidelines, and it addressed two different PCB mixtures at the same time, thus providing an unequivocal measure of potency differences. However, the accepted criteria for evaluating rodent liver lesions in cancer bioassays have changed since the Schaeffer et al. (1984) study was conducted (Maronpot et al., 1986). Recently, liver slides from this and other bioassays were evaluated using current diagnostic criteria (Experimental Pathology Laboratories, Inc., 1992). As a result, the incidence of hepatocellular carcinoma from 60% chlorine PCBs in the Schaeffer et al. (1984) study changed slightly. The updated tumor incidence data, fit to a linearized multistage model, provides for a potency factor of 1.1 $mg/kg/day^{-1}$ (based upon surface area extrapolation between rats and humans).

A potency factor of 0.18 mg/kg/day^{-1} was derived from the Schaeffer et al. (1984) dataset, using body weight extrapolation. While this value represents an accurate and scientifically defensible estimate of the cancer potency factor for 60% chlorine PCBs, it is nonetheless conservative: it is based upon the linearized multistage model and is therefore likely to overestimate the actual cancer risks at low, human exposures, particularly since 60% chlorine PCBs are non-genotoxic carcinogens in rodents.

The risks from PCBs based on these three cancer potency factors have been described in the following sections.

3.2.3 Risks Associated with Beachcombing

The risks associated with PCB intake from sediment in the beachcombing scenario were calculated using the three cancer potency factors described above and the chronic daily intake values presented in Table 2. These risks are provided in Table 5. With the exception of PCB concentration in sediment and an increased number of exposure events for the beachcomber (18 and 54 exposures per year), this exposure scenario assumes the same exposure conditions as those assumed for baseline conditions in Area I. Regardless of the cancer potency factor used to calculate risks, all post-remedial risks were still ≤ 2.1 E-06, and are risks that would be considered acceptable by the USEPA .

Table 5

Risks Associated with Exposure to PCB Contaminated Sediment

Receptor (F, YE, IR) *	Risk†		
	7.7	1.1	0.18
A. Area II Risks			
Older child			
(12/year, 9 years, 50 mg/event)	4.6 E-07	6.6 E-08	1.1 E-08
(24/year, 9 years, 50 mg/event)	9.3 E-07	1.3 E-07	2.2 E-08
Adult			
(12/year, 9 years, 50 mg/event)	3.1 E-07	4.4 E-08	7.2 E-09
(24/year, 9 years, 50 mg/event)	6.2 E-07	8.8 E-08	1.5 E-08
(12/year, 30 years, 50 mg/event)	1.0 E-06	1.5 E-07	2.4 E-08
(24/year, 30 years, 50 mg/event)	2.1 E-06	3.0 E-07	4.8 E-08
B. Area III Risks			
Young child			
(18/year, 4 years, 65 mg/event)	1.4 E-07	2.0 E-08	3.3 E-09
(54/year, 4 years, 65 mg/event)	4.2 E-07	6.0 E-08	9.9 E-09
Older child			
(18/year, 9 years, 50 mg/event)	1.3 E-07	1.9 E-08	3.1 E-09
(54/year, 9 years, 50 mg/event)	4.0 E-07	5.7 E-08	9.3 E-09
Adult			
(18/year, 9 years, 50 mg/event)	8.8 E-08	1.3 E-08	2.1 E-09
(54/year, 9 years, 50 mg/event)	2.7 E-07	3.8 E-08	6.2 E-09
(18/year, 30 years, 50 mg/event)	3.0 E-07	4.2 E-08	6.9 E-09
(54/year, 30 years, 50 mg/event)	8.8 E-07	1.3 E-07	2.1 E-08

* F = Frequency of exposure (events/year); YE = years exposed; IR = ingestion rate (mg/event).

† Risks were calculated using the three cancer potency factor values described in Section 3.2.2: 7.7, 1.1, and 0.18 $(mg/kg/day)^{-1}$.

3.2.4 Risks Associated with Seafood Consumption

The risks associated with PCB intake for the seafood scenario were calculated using the three cancer potency factors described above and the chronic daily intake values presented in Tables 4A, 4B, and 4C. These risks are listed separately for young children, older children, and adults in Tables 6A, 6B, and 6C. The highest risks were calculated for the adult consuming an average of 46.5 g of seafood per day for 30 years (Table 6C). Risks associated with this level of PCB exposure range from 1.3 E-05 to 2.2 E-06, using the 1.1 and 0.18 $(mg/kg/day)^{-1}$ cancer potency factors, respectively. For both the young and older children, the highest risk calculated was teenagers consuming an average of 46.5 g of seafood per day for 30 years (Table 6B). Risks associated with this level of PCB exposure range from 2.3 E-06 to 3.8 E-07, using the 1.1 and 0.18 $(mg/kg/day)^{-1}$ cancer potency factors, respectively.

Table 6A

Risks Associated with Consumption of Seafood: Young Children

Conditions *(C_{sf}, SI, FI, YE)	Risk†		
	7.7	1.1	0.18
0.114 mg/kg, 0.0062 kg, 0.2, 4 years	4.4 E-07	6.3 E-08	1.0 E-08
0.114 mg/kg, 0.0062 kg, 0.5, 4 years	1.1 E-06	1.6 E-07	2.6 E-08
0.114 mg/kg, 0.0165 kg, 0.2, 4 years	1.2 E-06	1.7 E-07	2.8 E-08
0.114 mg/kg, 0.0165 kg, 0.5, 4 years	3.0 E-06	4.2 E-07	6.9 E-08
0.17 mg/kg, 0.0062 kg, 0.2, 4 years	6.6 E-07	9.5 E-08	1.5 E-08
0.17 mg/kg, 0.0062 kg, 0.5, 4 years	1.7 E-06	2.4 E-07	3.9 E-08
0.17 mg/kg, 0.0165 kg, 0.2, 4 years	1.8 E-06	2.5 E-07	4.1 E-08
0.17 mg/kg, 0.0165 kg, 0.5, 4 years	4.4 E-06	6.3 E-07	1.0 E-07

* C_{sf} = concentration of PCBs in seafood (mg/kg); SI = amount of seafood ingested (kg); FI = fraction of seafood obtained from local catch; YE = years exposed.

† Risks were calculated using the three cancer potency factors described in Section 3.2.2: 7.7, 1.1, and 0.18 $(mg/kg/day)^{-1}$.

Table 6B

Risks Associated with Consumption of Seafood: Older Children

Conditions * (C_{sf}, SI, FI, YE)	Risk †		
	7.7	**1.1**	**0.18**
0.114 mg/kg, 0.0101 kg, 0.2, 9 years	1.6 E-06	2.3 E-07	3.8 E-08
0.114 mg/kg, 0.0101 kg, 0.5, 9 years	4.1 E-06	5.8 E-07	9.5 E-08
0.114 mg/kg, 0.0268 kg, 0.2, 9 years	4.3 E-06	6.2 E-07	1.0 E-07
0.114 mg/kg, 0.0268 kg, 0.5, 9 years	1.1 E-05	1.5 E-06	2.5 E-07
0.17 mg/kg, 0.0101 kg, 0.2, 9 years	2.4 E-06	3.5 E-07	5.7 E-08
0.17 mg/kg, 0.0101 kg, 0.5, 9 years	6.1 E-06	8.7 E-07	1.4 E-07
0.17 mg/kg, 0.0268 kg, 0.2, 9 years	6.4 E-06	9.2 E-07	1.5 E-07
0.17 mg/kg, 0.0268 kg, 0.5, 9 years	1.6 E-05	2.3 E-06	3.8 E-07

* C_{sf} = concentration of PCBs in seafood (mg/kg); SI = amount of seafood ingested (kg); FI = fraction of seafood obtained from local catch; YE = years exposed.

† Risks were calculated using the three cancer potency factors described in Section 3.2.2: 7.7, 1.1, and 0.18 $(mg/kg/day)^{-1}$.

Table 6C

Risks Associated with Consumption of Seafood: Adults

Conditions * (C_{sf}, SI, FI, YE)	Risk †		
	7.7	1.1	0.18
0.114 mg/kg, 0.0163 kg, 0.2, 9 years	2.6 E-06	3.8 E-07	6.1 E-08
0.114 mg/kg, 0.0163 kg, 0.5, 9 years	6.6 E-06	9.4 E-07	1.5 E-07
0.114 mg/kg, 0.0163 kg, 0.2, 30 years	8.8 E-06	1.3 E-06	2.0 E-07
0.114 mg/kg, 0.0163 kg, 0.5, 30 years	2.2 E-05	3.1 E-06	5.1 E-07
0.114 mg/kg, 0.0465 kg, 0.2, 9 years	7.5 E-06	1.1 E-06	1.8 E-07
0.114 mg/kg, 0.0465 kg, 0.5, 9 years	1.9 E-05	2.7 E-06	4.4 E-07
0.114 mg/kg, 0.0465 kg, 0.2, 30 years	2.5 E-05	3.6 E-06	5.8 E-07
0.114 mg/kg, 0.0465 kg, 0.5, 30 years	6.2 E-05	8.9 E-06	1.5 E-06
0.17 mg/kg, 0.0163 kg, 0.2, 9 years	3.9E-06	5.6 E-07	9.2 E-08
0.17 mg/kg, 0.0163 kg, 0.5, 9 years	9.8 E-06	1.4 E-06	2.3 E-07
0.17 mg/kg, 0.0163 kg, 0.2, 30 years	1.3 E-05	1.9 E-06	3.1 E-07
0.17 mg/kg, 0.0163 kg, 0.5, 30 years	3.3 E-05	4.7 E-06	7.6 E-07
0.17 mg/kg, 0.0465 kg, 0.2, 9 years	1.1 E-05	1.6 E-06	2.6 E-07
0.17 mg/kg, 0.0465 kg, 0.5, 9 years	2.8 E-05	4.0 E-06	6.5 E-07
0.17 mg/kg, 0.0465 kg, 0.2, 30 years	3.7 E-05	5.3 E-06	8.7 E-06
0.17 mg/kg, 0.0465 kg, 0.5, 30 years	9.3 E-05	1.3 E-05	2.2 E-06

* C_{sf} = concentration of PCBs in seafood (mg/kg); SI = amount of seafood ingested (kg); FI = fraction of seafood obtained from local catch; YE = years exposed.

† Risks were calculated using the three cancer potency factors described in Section 3.2.2: 7.7, 1.1, and 0.18 $(mg/kg/day)^{-1}$.

4.0 Summary

The risk assessment prepared by E.C. Jordan concerning PCB-contaminated sediment and seafood from New Bedford Harbor contained a number of overly conservative exposure assumptions, which in turn led to an overestimation of risk. These assumptions include the sediment ingestion rate, the amount of New Bedford Harbor fish consumed by area residents, the toxicokinetic (bioavailability) factor for PCBs, the number of years that a young child would have contact with site sediments or would consume fish from New Bedford Harbor, and the frequency of visits that persons may make to areas surrounding New Bedford Harbor.

A major shortcoming in the PCB risk assessment performed by ECJ is the use of cancer potency factors derived from experiments utilizing 60% chlorine PCB mixtures in predicting risks from exposure to less chlorinated mixtures. Additionally, the cancer slope factor used, while that currently promoted by the USEPA, was based on a single, methodologically flawed study and therefore is lacking in complete scientific credibility. However, alternate slope factors can be derived from experimental data which have been reevaluated in light of current scientific evidence on interspecies scaling factors and current protocols for the description of neoplastic responses. These factors would still be extremely conservative when applied to 54% and 42% chlorine PCB mixtures and so were used in this risk assessment.

Using these slope factors and more realistic exposure assumptions, an alternative risk assessment was performed. Using $\leq$ 50 ppm as the cleanup guideline for sediment concentrations, and the three cancer potency factors described in Section 3.2.2, all post-remedial risks from direct exposure to Area II and Area III sediments were determined to be near or below the $1x10^{-6}$ risk level. Likewise, the highest risks for both the young and older children from consumption of New Bedford Harbor seafood contaminated with low levels of PCBs were near or below the $1x10^{-6}$ risk level when updated slope factors were used. The highest risks were those calculated for the adults consuming an average of 46.5 g of seafood per day for 30 years. Still, this risk for this exposure scenario only ranged from $1.3x10^{-5}$ to $2.2x10^{-6}$ using the updated slope factors. Therefore, this assessment serves to illustrate the safety of applying a 50 ppm PCB cleanup standard to the sediments in Areas II and III.

References

Allen, B.C., A.M. Shipp, K.S. Crum, B. Kilian, et al. 1987. Investigation of cancer risk assessment methods. USEPA Contract 600/6-87/007a.

Balsam, 1989. PCB Assessment of Acushnet River Upper Estuary Wetland Sediments-New Bedford Harbor. Draft. March 9, 1989.

Bartek, M.J., J.A. LaBuddle, and H.I. Maibach. 1972. Skin permeability *in vivo*: Comparison in rat, rabbit, pig and man. J. Invest. Dermatol. 58:114.

Binder, S., D. Sokal and D. Maughan. 1986. Estimating soil ingestion: The use of tracer elements in estimating the amount of soil ingested by young children. Arch. Environ. Health 41:341-345.

Calabrese, E.J. and E.J. Stanek. 1991. A guide to interpreting soil ingestion studies. II. Qualitative and quantitative evidence of soil ingestion. Reg. Toxicol. Pharmacol. 13:278-292.

Calabrese, E.J., R. Barns, E.J. Stanek III, H. Pastides, C.E. Gilbert, P. Veneman, X. Wang, A Lasztety, and P. Kostecki. 1989. How much soil do young children ingest: An epidemiologic study. Regul. Toxicol. Pharmacol. 10:123-127.

Chou, S.F.J. and R.A. Griffin. 1986. Solubility and soil mobility of polychlorinated biphenyls. In: PCBs and the Environment. Waid, J.S. et al. (eds.). CRC Press, Inc. Boca Raton, Florida. Vol. I, p. 101.

Clausing, P., B. Brunekreef and J.H. van Wijen. 1987. A method for estimating soil ingestion by children. Int. Arch. Occup. Environ. Health 59:73-82.

Ebasco. 1989. Final baseline public health risk assessment; New Bedford Harbor feasibility study. Draft. August, 1989. p. 2-35 (Table 2-11).

Humphrey, H.E.B. 1976. Evaluation of changes of the level of polychlorinated biphenyls (PCB) in human tissue. US Dept.

IEHR 1991. Reassessment of liver findings in five PCB studies in rats. Pathology working group review. Institute for Evaluating Health Risks. Washington, D.C.

Javitz, H. 1980. Seafood consumption data analysis. SRI International. Final report prepared for EPA Office of Drinking Water Regulations and Standards. EPA contract 68-01-3887.

Kimbrough, R.D., R.A. Liner, J.D. Standberg, R.J. Montali and W.W. Burse. 1975. Induction of liver tumors in Sherman strain rats by polychlorinated biphenyl aroclor 1260. J. Natl. Cancer Inst. 55:1453.

LaGoy, P.K. 1987. Estimated soil ingestion rates for use in risk assessments. Risk Analysis 7:355-359.

Maronpot, R.R., C.A. Montgomery, G.A. Boorman and E.E. McConnel. 1986. National Toxicology Program nomenclature for hepatoproliferative lesions of rats. Toxicologic Pathol. 14:263-273.

Morgan, R.W., J.M. Ward and P.E. Hartman. 1981. Aroclor 1254-induced intestinal metaplasia and adenocarcinoma in the glandular stomach of F344 rats. Cancer Res. 41:5052-5059.

NCI. 1978. Bioassay of Aroclor 1254 for possible carcinogenicity. DHEW publication No. (NIH) 78-838.

Norback, D.H. and R.H. Weltman. 1985. Polychlorinated biphenyl induction of hepatocellular carcinoma in the Sprague-Dawley rat. Environ. Health Perspect. 60:97.

Norback, D.H., E. Mack, K.A. Blomquist, and J.R. Allen. 1978. Metabolic study of 2,4,5,2',4',5',-Hexa chlorobiphenyl in Rhesus monkeys. Tox. Appl. Pharmacol. 45:331.

Pennington, J.A.T. 1983. Revision of the total diet study food list and diets. J. Am. Diet. Assoc. 82:166.

Poiger, H. and C. Schlatter. 1980. Influence of solvents and adsorbents on dermal and intestinal absorption of TCDD. Environ. Toxicol. 18:477-481.

Puffer, H. 1981. Consumption rates of potentially hazardous marine fish caught in the metropolitan Los Angeles area. EPA Grant #R807 120010. As cited in USEPA (1989).

Risk Analysis 9(3):275-277.
Rodricks, J.V. 1989. Guest Editorial: Determining compliance with PCB standards

Schaeffer, E., H. Greim, and W. Goessner. 1984. Pathology of chronic polychlorinated biphenyl (PCB) feeding in rats. Toxicol. Appl. Pharmacol. 75:278.

Shu, H., D. Paustenbach, F. J. Murray, L. Marple B. Brunck, D. Dei Rossi, and P. Teitelbaum. 1988a. Bioavailability of soil bound TCDD: Oral bioavailability in the rat. Fundam. Appl. Toxicol. 10:648-654.

Shu, J., P. Teitelbaum, A.S. Webb, L. Marple, B. Brunck, D. Dei Rossi, F.J. Murray and D. Paustenbach. 1988b. Bioavailability of soil-bound TCDD: Dermal bioavailability in the rat. Fund. Appl. Toxicol. 10:335-343.

USEPA (United States Environmental Protection Agency). 1986. Development of advisory levels for polychlorinated biphenyls (PCBs) cleanup. Office of Health and Environmental Assessment. Washington, D.C.

USEPA. 1987. Drinking Water Criteria Document for Polychlorinated Biphenyls (PCBs). USEPA ECAO-Cin-414.

USEPA. 1989. Exposure Factors Handbook. Office of Health and Environmental Assessment. EPA 600/8-89/043.

USEPA. 1991. Workshop report on toxicity equivalency factors for polychlorinated biphenyl congeners/for the Risk Assessment Forum. EPA 625/3-91/020.

USEPA. 1992. New Interim Region IV Guidance. Memorandum from USEPA Region IV. Atlanta, GA. February 11, 1992.

Ward, J.M. 1985. Proliferative lesions of the glandular stomach and liver in F344 rats fed diets containing aroclor 1254. Environ. Health Perspect. 60:89-95.

Wester, R.C., D. Bucks, H.I. Maibach and J. Anderson. 1983. Polychlorinated biphenyls (PCBs): Dermal absorption, systemic elimination, and dermal wash efficiency. J. Toxicol. Environ. Health 12:511.

Wester, R.C., M. Mobayen, and H.I. Maibach. 1987. *In vivo* and *in vitro* absorption and binding to powdered stratum corneum as methods to evaluate skin absorption of environmental chemicals from ground and surface water. J. Toxicol. Environ. Health 21:367-374.

ATTACHMENT B

A Theoretical Evaluation of the Effects of Dredging on PCB Emissions from New Bedford Harbor Sediment

A THEORETICAL EVALUATION OF THE EFFECTS OF DREDGING ON PCB EMISSIONS FROM NEW BEDFORD HARBOR SEDIMENT

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ABSTRACT

This report summarizes the effects of dredging polychlorinated biphenyl (PCB) contaminated New Bedford Harbor sediment on the air emissions of PCBs. Dredging was considered only for those areas of PCB contamination between 50 and 4000 $mg \cdot kg^{-1}$. This excludes the more highly contaminated Hot Spot. Three situations were considered, PCB air emissions before dredging (base case), PCB air emissions from a suspended sediment plume generated by dredging, and PCB air emissions after re-settling of suspended solids and re-consolidation of sediment. Only the air emissions from the upper estuary were considered. The upper estuary was assumed well-mixed with an average water residence time 2.4 days.

A PCB evaporation rate from the upper estuary of 90 $g \cdot hr^{-1}$ was estimated based on an average dissolved PCB concentration of 1.6 $\mu g \cdot l^{-1}$ (Thibodeaux et al., 1990 based on data from Battelle, 1985). More recent water quality measurements by the US Army Corps of Engineers (USACE) during 1988 and 1989 (USACE, 1989) suggest a dissolved total PCB concentration of 0.54 $\mu g \cdot l^{-1}$ and a resulting evaporation rate of about 31 $g \cdot hr^{-1}$. Because the USACE data represent the most recent measurements, these data will be assumed to indicate current, base case conditions.

During dredging the emission from a suspended sediment plume in the upper estuary can be as high as 2630 $g \cdot hr^{-1}$ for an uncontrolled dredge providing high suspended solids concentrations to the water column. If the suspended sediment concentrations resulting from the dredging operation are limited to the levels observed during a pilot scale demonstration by the USACE (1989), about 146 $g \cdot hr^{-1}$ of PCBs would be evaporated from the upper estuary. If 2 cutterhead dredges are used as described by USACE (1989), however, and the sediment resuspension rate per dredge is limited to the 17 $g \cdot sec^{-1}$ estimated in that study, an average of about 57 $g \cdot hr^{-1}$ of PCBs would evaporate. During the dredging of the highest concentration areas currently under consideration (≈3,500 mg/kg PCB) about 290 $g \cdot hr^{-1}$ of PCBs would be emitted from the upper estuary above baseline conditions assuming the 17 $g \cdot sec^{-1}$ sediment resuspension rate.

After resettling of suspended solids and reconsolidation of the dredged sediment, about 0.59 $g \cdot hr^{-1}$ of PCBs would evaporate from the upper estuary. This assumes that the dredging of sediment contaminated with PCBs at concentrations greater than 50 $mg \cdot kg^{-1}$ is 99.34% efficient, consistent with the two pass dredging operations conducted during the USACE demonstration program (USACE, 1989). This estimate also assumes that the Hot Spot will be remediated to at least the same level. Post-dredging evaporative emissions from the estuary would be about 9.3 $g \cdot hr^{-1}$ if the Hot Spot were to remain.

INTRODUCTION

The objectives of this report are to estimate the emissions of polychlorinated biphenyls (PCBs) from New Bedford Harbor as a result of dredging the contaminated sediment. This remediation scheme is presently being considered by the U S Environmental Protection Agency (EPA) for those areas of the harbor and bay that have PCB sediment concentrations greater than 50 $mg \cdot kg^{-1}$. It is anticipated that the area designated as the "Hot Spot" would be excluded during this phase of remediation (Bosworth, 1991). As a basis of comparison, only the evaporative emissions from the upper estuary are considered here. Some additional evaporation will occur in the lower harbor and bay but the water column PCB concentrations in these areas are typically an order of magnitude or more smaller than the concentrations in the upper estuary (US EPA, 1990).

The history of contamination of New Bedford harbor, the possible routes of exposure by humans to PCB emissions from the sediments, and the advantages and disadvantages of capping contaminated sediments have been reported by Thibodeaux et al. (1990) and in a feasibility study by EPA (1990).

The salient aspects addressed in this report are:

(a) Estimation of steady state emission of PCBs from sediments before dredging as the "Base Case".

(b) Estimation of additional emissions of PCBs after a portion of the sediment is resuspended in the water column as a result of dredging. All areas except the "Hot

Spot" region and regions with PCB concentration less than 50 $mg \cdot kg^{-1}$ are assumed subject to dredging,

(c) Enumeration of possible PCB emission pathways during dredging and emission locales from the confined disposal facility,

(d) Estimation of emissions of PCBs from the remediated areas once the dredging operation has concluded, suspended solids have resettled and reconsolidation of the sediment has taken place.

The mixing time within the upper estuary is assumed to be short with respect to the remediation time of the sediments suggesting that evaporation can be treated as quasi-steady. The conditions for which emissions are estimated are represented in Figure 1.

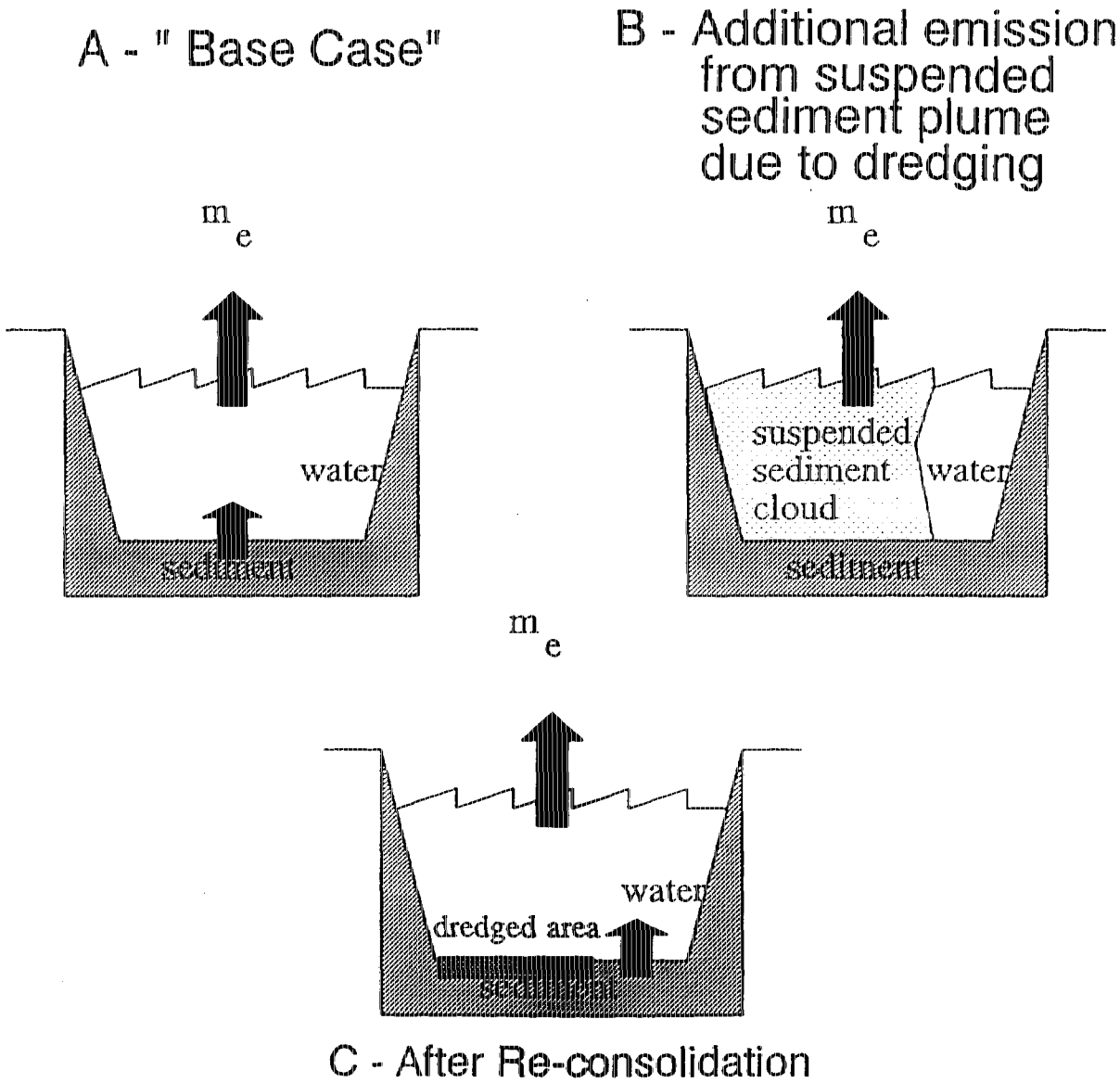

Figure 1. Sketch of conditions for which PCB emissions are calculated: (A) Base Case (B) From a Suspended Sediment Plume and (C) After Re-consolidation.

SEDIMENT AND CONTAMINANT PROPERTIES

The PCB contamination in New Bedford Harbor (NBH) was assumed to be composed 70% of Aroclor-1242 and 30% Aroclor-1254 (Thibodeaux et al., 1990). The properties of Aroclor-1242 and Aroclor-1254 are given in Table 1. The pertinent data on the NBH sediment are given in Table 2. The properties of PCBs and the transfer coefficients were obtained from Thibodeaux et al. (1990). The sediment data were obtained from Thibodeaux et al. (1990) and Bosworth (1991).

Calculation of the evaporative emissions in $g \cdot hr^{-1}$ were based on the estimated flux and the total area of the upper estuary of about 800,000 m^2. This area and the associated wetlands in the upper estuary constitute the vast majority of the sediment to be removed under the proposed dredging program. The PCB distribution in the upper estuary was reported by Thibodeaux et al. (1990) as reproduced in Figure 2. Based on this data one can conclude that a major fraction of the sediment area has sediment PCB (total) concentration of 100 - 500 ppm.

It should be noted that in certain regions, the PCB loadings on the sediment are large and that the corresponding pore water in equilibrium with the sediment will have PCB concentrations equal to the "solubility limit". The critical loading, w_{AC} for A-1242 and A-1254 are given in Table 1. In calculating PCB fluxes this limit serves as the maximum possible PCB concentration in the sediment porewater and the aqueous column above the sediment.

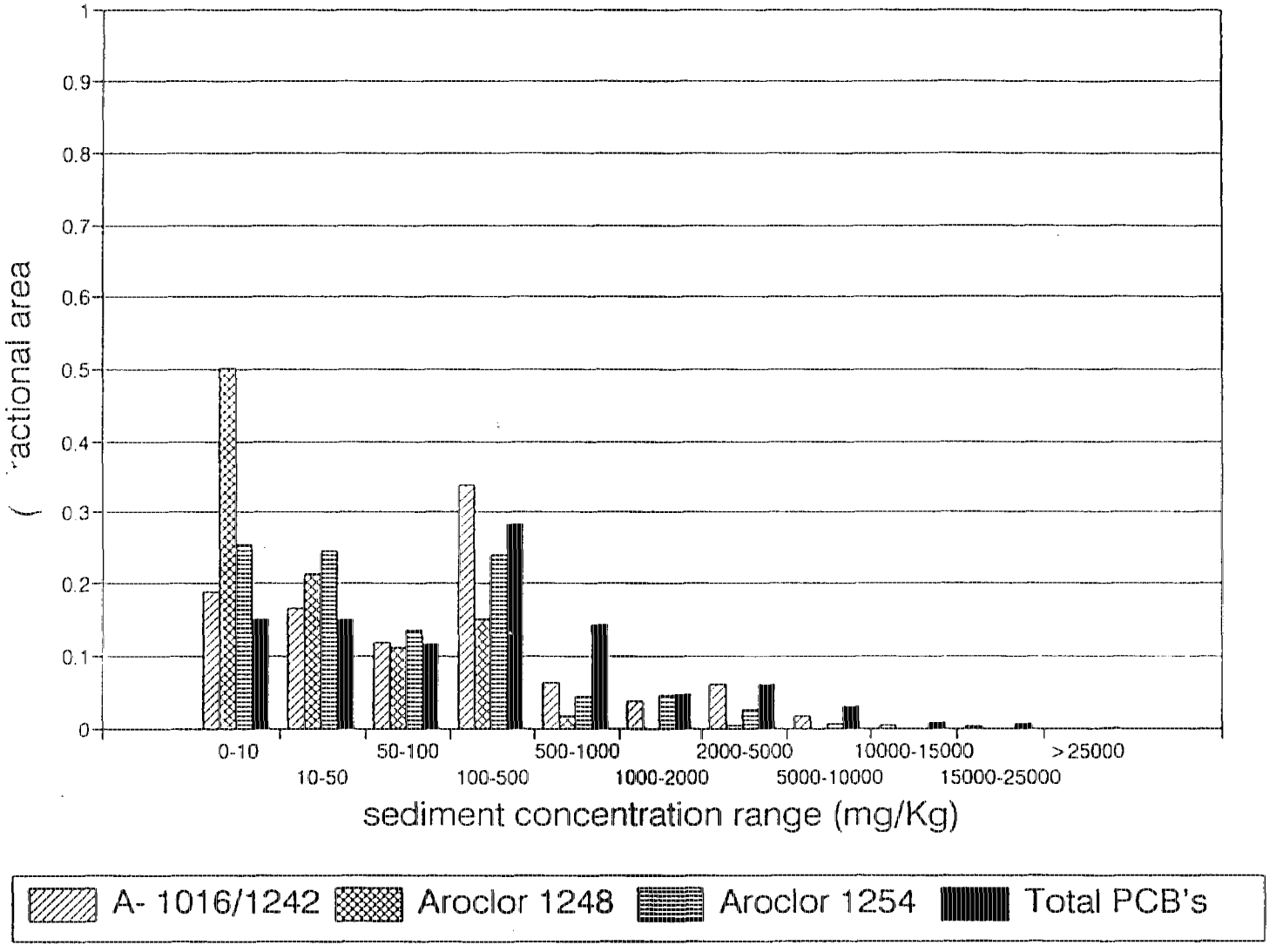

Figure 2. Sediment concentration versus fractional area of contamination at the NBH site.

TABLE 1

Selected Physico-chemical Properties Used in the Models

Property	Symbol	A-1242	A-1254
Average molecular weight	M_A	266	238
Aqueous solubility Fresh water ($mg \cdot l^{-1}$)	ρ_w	0.24	0.03
Aqueous solubility Saline water ($mg \cdot l^{-1}$)	ρ^*_w	0.088	0.012
Sediment-water partition constant ($l \cdot kg^{-1}$)	K_d	7,920	21,560
Critical loading on the sediment ($mg \cdot kg^{-1}$)	$W_{AC}=K_d\rho_w^*$	690	260
Bio-diffusion constant ($cm^2 \cdot y^{-1}$)	$D_A^{(1)}$	10	10
Bioturbation depth (cm)	y_b	10	10
Benthic boundary layer mass transfer coefficient ($cm \cdot y^{-1}$)	k_b	1.8×10^4	1.8×10^4
PCB evaporation coefficient ($cm \cdot y^{-1}$)	K_e	6.2×10^4	6.2×10^4

Note: Source is Thibodeaux et al. (1990).

TABLE 2

Sediment Data Used in the Models

Property	Symbol	Value
Dry bulk density	ρ_b	0.75 $g \cdot cm^{-3}$
Total sediment surface area to be dredged	A	310 acres
Area of upper estuary	A_s	800,000 m^2 (198 acres)
Porosity	ϵ	0.5

Note: A_s obtained from Bosworth (1991).

STEADY STATE EMISSION TO AIR FROM CONTAMINATED SEDIMENT BED (BASE CASE)

The evaporative emissions of PCBs through the air-water interface of the upper estuary under quasi-steady state conditions is given by

$$m_e = A_s K_e \left(\rho_w - \rho_w^{**}\right) \tag{1}$$

where

m_e = mass emission rate, $g \cdot hr^{-1}$

A_s = upper estuary surface area, cm^2

K_e = overall liquid phase mass transfer coefficient (evaporation coefficient) for PCB, $cm \cdot hr^{-1}$.

ρ_w = dissolved phase concentration of PCB, $g \cdot cm^3$

ρ_w^{**} = hypothetical dissolved phase concentration of PCB in equilibrium with bulk air phase concentration ($g \cdot cm^{-3}$)

The value of K_e is dependent on both the wind velocity and fetch/depth ratio of the water body. Thibodeaux et al.(1990) estimated K_e for typical conditions (wind velocity = 7 $m \cdot sec^{-1}$) at NBH and obtained a value of 1.68 $m \cdot day^{-1}$ (= 7 $cm \cdot hr^{-1}$). Another independent assessment of PCB evaporation from the NBH water column arrived at a value of K_e = 2.4 $m \cdot day^{-1}$ (Applied Science Associates (ASA), 1989).

Figure 1 (a) represents the "base case". The chemical transport pathways for PCBs from sediment to air will involve:

(i) desorption from the bed surface particles via molecular diffusion and bioturbation,

(ii) molecular diffusion through the benthic boundary layer,

(iii) movement up the water column to the interface,

(iv) volatilization through the air boundary layer into the atmosphere.

If the sediment and water columns are operating under quasi-steady conditions, however, only the water column concentration need be known to estimate the evaporative flux. The validity of the steady state (or more appropriately the quasi steady-state) approximation in estimating PCB emissions from NBH sediments has been described by Thibodeaux et al. (1990). Thibodeaux et al. (1990) reports average dissolved PCB concentrations in the upper estuary as 1.1 $\mu g \cdot l^{-1}$ for A-1242 and 0.5 $\mu g \cdot l^{-1}$ for A-1254. These estimates were based on total water column PCB measurements collected by Batelle (1985). More recently the USACE pilot study of remedial alternatives in New Bedford Harbor measured an average water column PCB concentration of 0.607 $\mu g \cdot l^{-1}$ and a background suspended solids concentration of 6.4-10.2 $mg \cdot l^{-1}$. Using 10 $mg \cdot l^{-1}$ suspended solids and assuming that the PCBs are 70% A-1242 and 30% A-1254, the 0.607 $\mu g \cdot l^{-1}$ corresponds to a dissolved PCB concentration of about 0.54 $\mu g \cdot l^{-1}$. With a suspended solids concentration of 10 $mg \cdot l^{-1}$, about 93% of A-1242 and about 82% of the A-1254 will be in the dissolved state.

Using the evaporation coefficient of Thibodeaux et al. (1990) and the concentration inferred from the Batelle (1985) data, the PCB evaporation rate constant under pre-dredging conditions is about 90 $g \cdot hr^{-1}$ or 2.2 $kg \cdot day^{-1}$. Using the more recent water quality

measurements by the USACE, the estimated evaporation rate is 31 $g \cdot hr^{-1}$ or 0.74 $kg \cdot day^{-1}$. Because the USACE data is more recent, these data and the estimated evaporation rate will be assumed to represent baseline conditions. A numerical model of the harbor reported by US EPA (1990) predicted volatization of 243 kg of PCB over 92 days (2.6 $kg \cdot day^{-1}$), in reasonable agreement with the above estimates.

ENUMERATION OF POSSIBLE EMISSION PATHWAYS DURING DREDGING AND EMISSION LOCALES FROM CONFINED DISPOSAL FACILITIES

There are several emission pathways for PCBs during the dredging process which we shall briefly consider here. Although a number of potential pathways are identified, calculations will only be attempted for emissions at the point of dredging.

Dredge Machinery Disturbance of Water Surface:

The use of a dredge machinery (ie., a barge) will likely increase the water turbulence and hence increase the intrinsic rate of PCB evaporation. The use of mechanical devices on the barge such as buckets, shovels etc. as they are being hoisted or lowered through the water surface contributes to significant point sources of emissions.

USACE (1989) conducted a pilot scale study of dredging small areas south of the estuary at the Cogeshall Street Bridge in the lower harbor/bay and also in the upper estuary. The tests were designed to estimate the resuspension of sediments and release of PCBs during dredging. A cutterhead dredge was employed. Samples were collected near the dredgehead and a modified elutriate test used to estimate the PCB release rates to water from suspended sediments. The results are summarized in Table 3.

The cutterhead dredge affected approximately a 1000 ft^2 area in an average of 4 hours of operation per day with a production rate of 35 cubic yards. It was also reported in the feasibility study (EPA, 1990) that air monitoring during dredging and disposal of sediments did increase the ambient air PCB levels above

background. Thus there is experimental evidence for PCB air emissions during dredging and disposal operations even with a single dredge affecting a small area.

Dredges such as cutterhead, matchbox and mudcat are hydraulic dredges that remove and transport sediment in slurry form. These can be assumed to enhance PCB emission rates mainly through the generation of a suspended sediment plume near the dredgehead. The activity of the dredge machinery (i.e., a barge) on the water surface will also increase the air-water interface turbulence thus affecting evaporation rate by perhaps no more than a factor of two (Thibodeaux, 1989). The increased emissions due directly to the operation of the barge in the case of hydraulic devices are therefore likely small, while the major mechanism of PCB emissions will be due to the re-suspension event.

Mechanical dredges (clamshell, dipper,ladder etc.), on the other hand will contribute to three main point sources of PCB emissions to air:

(i) the water surface under the bucket,

(ii) surface of the bucket with sediment heap, and

(iii) surface of the receiving vessel.

The first and second mechanisms are intermittent depending on the frequency of raising and lowering the bucket, while the third mechanism is a continuous one. Expressions for PCB emission rates by each of the above mechanisms can be obtained (Thibodeaux, 1989) and combined to get the total emission rate due to mechanical dredges.

TABLE 3

Estimated PCB Flux to Water During Dredging

(Pilot scale study, US ACE, 1989)

Location	Total PCB flux to water at the dredgehead (suspended + dissolved, $kg.day^{-1}$)
Lower harbor/bay	0.43
Cogeshall Street Bridge	0.30

Emissions During Discharge of Sediments :

Most hydraulic dredges are equipped with devices to pump all types of materials and compacted deposits such as clay and hardpan, to confined disposal facilities (CDFs). A CDF is often required for temporary storage or to act as an equalization basin even if ultimate disposal in a CDF is not planned. The discharge of sediment and water slurry from the end of the pipe either into water or receiving sediment delta of the CDF are point sources of PCB emissions. Equations for each of the above were derived by Thibodeaux (1989). The emissions from these processes will not be discussed further here.

Exposed Sediment Locale:

An additional VOC emission locale is the exposed sediment in a CDF. This is the delta region formed by the sediment discharged

from a pipeline or from the bucket/shovel discharge in the CDF and generally is devoid of any vegetative cover. PCBs will diffuse through the soil cover and enter the air. The estimation of these emissions is quite complicated and will not be attempted here. Equations based on a conceptual model of the process was given by Thibodeaux (1989) and are recommended.

Ponded Sediment Locale:

Another emission source is the ponded sediment locale which is the water overlying dredged material in a CDF. The water as a source of PCB emissions was considered and equations for flux derived by Thibodeaux (1989).

Vegetation Covered Sediment Locale:

The portion of the CDF that is covered with vegetation can also be a point source of PCB emission. This locale, however may have the lowest emission rate. Methods for their estimation have also been discussed by Thibodeaux (1989).

The acceptability of dredging as a remediation measure will depend not only on the extent of reduction of PCB emissions to air, but also on the cost factors involved in the number of hours required to operate the dredges and the necessity to keep the re-suspension of sediments to a minimum during dredging. Based on the pilot scale study USACE (1990) concluded that three different types of dredges used (mudcat or horizontal auger, cutterhead and matchbox dredges) were effective in removing the contaminated sediment with minimum resuspension and contaminant release. It is however, not clear whether the results of such a small pilot scale

study can adequately reflect the conditions that would exist when several dredges are operated simultaneously over a large area. EPA (1990) report 3.8 years will be required to remove 400,000 cubic yards of contaminated sediment using two cutterhead dredges. It is also anticipated that the large volume of contaminated dredged material that will have to be handled in a CDF or will have to be treated simultaneously may pose problems with respect to air emissions and contaminant migration. The calculations presented herein will focus on estimating only a portion of those emissions, those generated by contaminant release at the point of dredging.

ESTIMATION OF ADDITIONAL PCB EMISSIONS FROM A SUSPENDED SEDIMENT PLUME GENERATED BY DREDGING

One proposal for remediation of New Bedford Harbor is removal of contaminated sediments exhibiting PCB concentrations greater than 50 $mg \cdot kg^{-1}$ via dredging followed by treatment and/or disposal of the dredged material. Table 4 identifies the fraction of the total upper estuary area with average concentration within this range. Figure 3 and Figure 4 shows the distribution of A-1242 and A-1254 in the upper estuary. These tables and figures indicate the region within the upper estuary under consideration for dredging. This proposal excludes the Hot Spot in the upper estuary in which the highest contamination levels are observed.

It is assumed that the bed sediment source of contamination is removed by dredging but resuspension of some portion of this sediment can give rise to a suspended sediment plume in the water column. The concentration of suspended solids can vary considerably in a deep aqueous column. Since in the present case the mean aqueous depth is only about 90 cm, the concentration of suspended particulates is assumed uniform throughout.

The extent of resuspension will depend on the amount of material dredged or loosened from the bottom sediment and the amount actually removed by the dredge; this obviously is a function of the efficiency of the type of dredging operation. Volatile organic chemicals (VOCs) emissions will be enhanced as a result of increased solids resuspended in water and increased turbulence at

TABLE 4

Regions Proposed for Remediation (Bosworth, 1991)

Region	Details	Fractional area of contamination			Remediation Action
		A-1242	A-1254	PCB(total)	
I	w_A< 50 ppm	0.36	0.51	0.30	No dredging
II	50<w_A<4000 ppm	0.62	0.48	0.66	Dredging
	Region II Breakdown				
	75 ppm	0.12	0.13	0.12	
	300 ppm	0.34	0.24	0.28	
	750 ppm	0.06	0.04	0.14	
	1500 ppm	0.04	0.04	0.05	
	3500 ppm	0.06	0.03	0.06	
Hot spot	w_A>4000 ppm	0.02	0.01	0.04	see note

Note: w_A is the average PCB concentration (ppm is $mg.kg^{-1}$) on sediment. Hot spot is not considered for PCB emissions calculations in this phase of remediation. Fractional areas are from Thibodeaux et al. (1990). Areas contaminated with other Aroclors (in minor concentrations) are not shown in the Table.

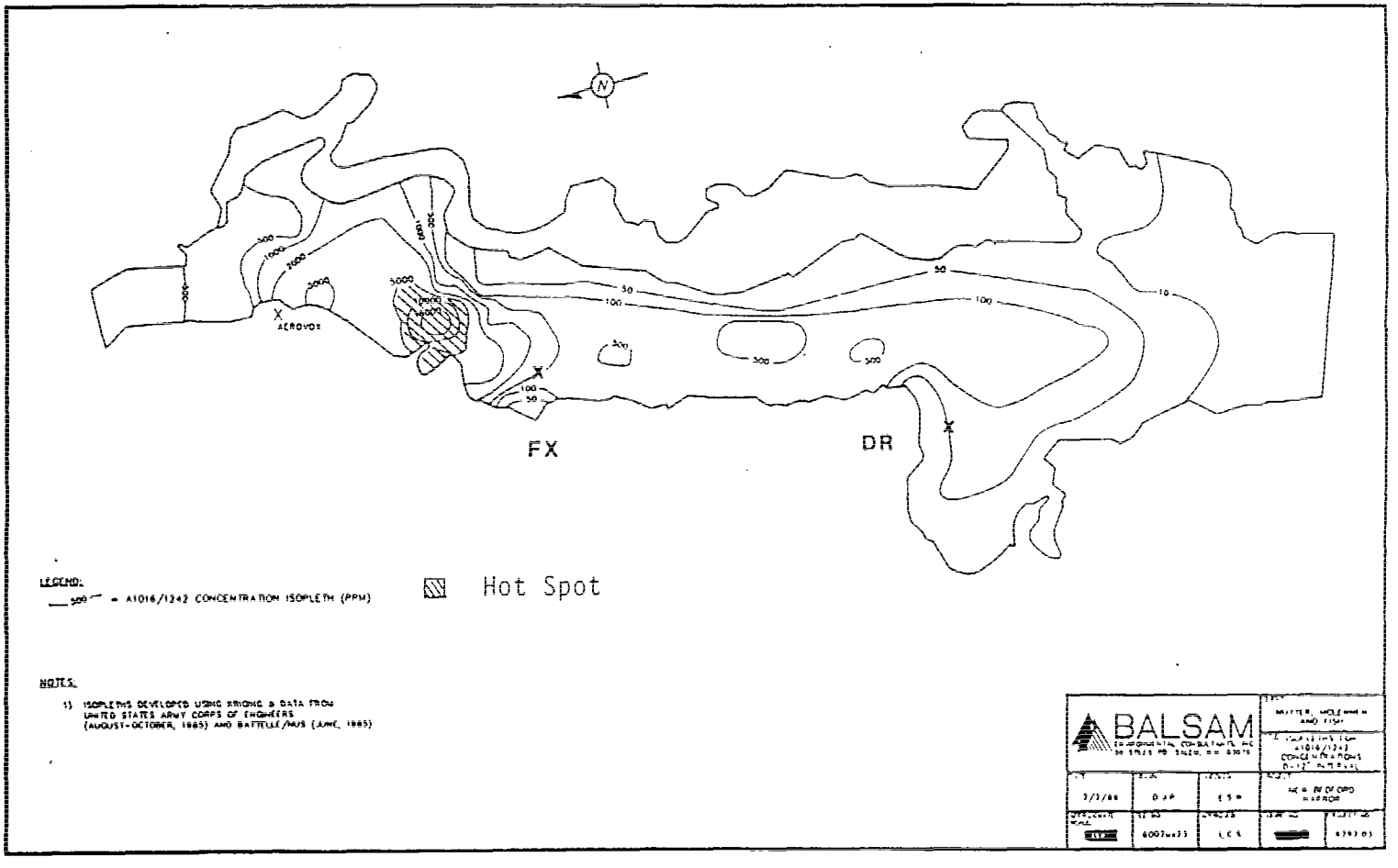

Figure 3. Isopleths of PCB Contamination (A-1242)

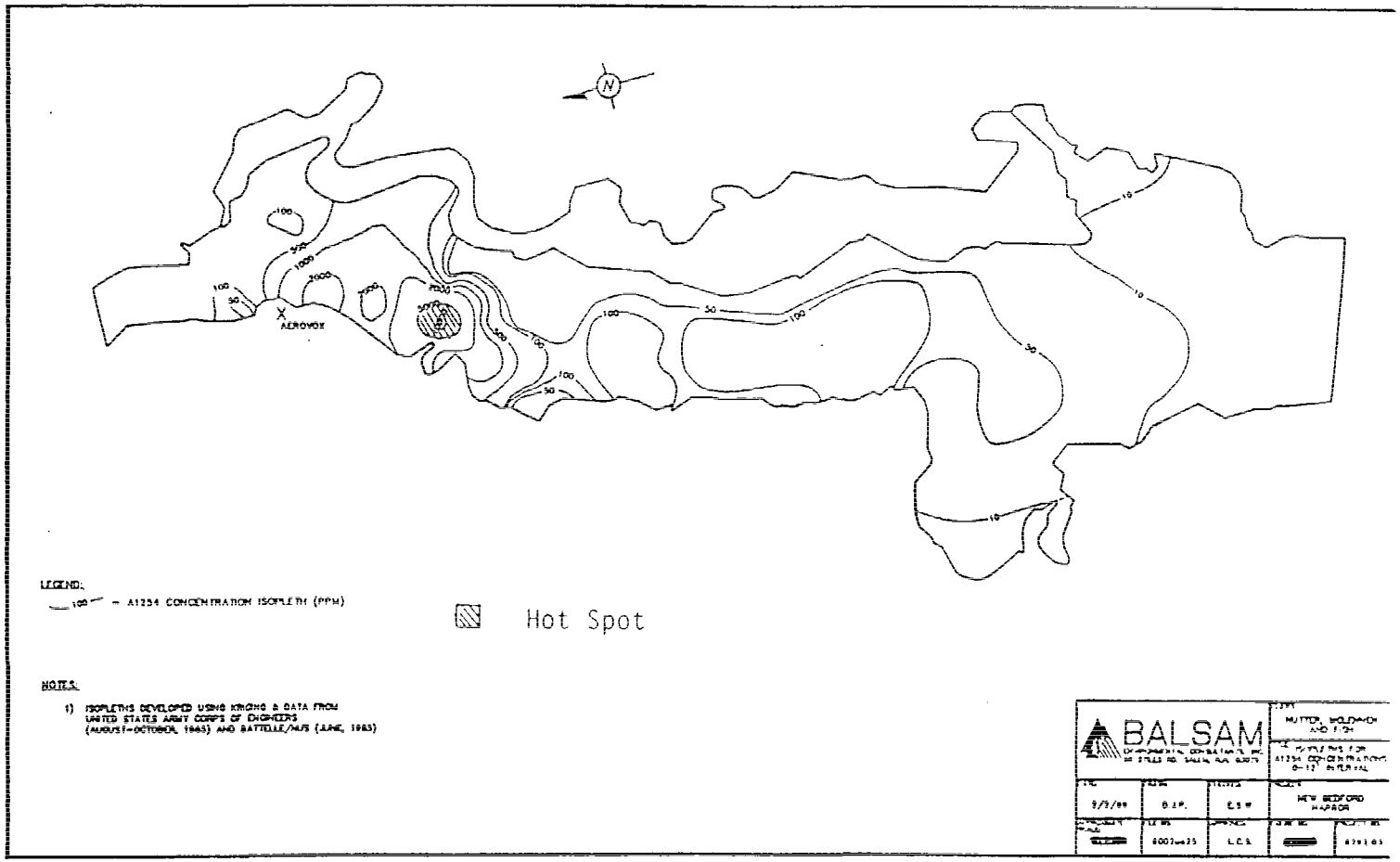

Figure 4. Isopleths of PCB Concentration (A-1254)

the air-water interface (Brannon et al., 1991). PCBs in the contaminated sediment will be released to the water column as a result of desorption of PCBs from suspended solids, and dispersal and mixing of interstitial water that contain PCBs. Desorption from the bed sediment is often thermodynamically limited by the increase in interstitial concentration. Desorption from suspended sediments widely dispersed in the water column, however, will continue to occur until essentially all of the PCBs reside in the dissolved phase. The time available for this desorption may be long for fine-grained sediments (eg. clay and silt) due to low settling velocities in water. Sandy sediments may resettle faster.

Thibodeaux (1989) has described the process of dissolution and evaporation from suspended sediment particles. Let the concentration of the suspended plume of solids be ρ_s ($kg.l^{-1}$). If the rate of PCB volatilization is small in comparison to the total mass of PCBs associated with the suspended solids, and we assume equilibrium desorption of PCBs from the solids, then an equilibrium model can be used to estimate PCB concentration in solution. The concentration in solution, ρ_w ($mg.l^{-1}$) is given by

$$\rho_w = \frac{w_A \rho_s}{K_d \rho_s + 1} \qquad (2)$$

where w_A is the PCB concentration on the suspended sediment arising from the original sediment bed ($mg.kg^{-1}$). The solubility limit applies to Equation (2), hence it is only applicable for $\rho_w < \rho_w^*$.

The most significant assumption in the model was noted above, ie., the solids suspended in the aqueous phase have sufficiently long residence times that equilibrium between the solids and water is achieved. If intraparticle diffusion of PCBs limits mass transfer from solids to water, as is the case for large particle sizes and compounds with large K_d values, equilibrium may not be achieved (Gschwend et al., 1987). In such a situation, Equation (2) would overestimate the dissolved phase concentration. We also assume that the suspended sediment plume extends over the entire upper estuary and, as a conservative assumption, that the rate of sedimentation of the dispersed material is very slow. Significant mixing and transport of suspended solids and associated contaminants will occur towards the lower harbor and bay areas, but only the evaporative emissions in the upper estuary are described here.

The emission rate of PCBs through the air-water interface is again given by Equation (1). Assuming that the air concentration of PCB is zero, so that ρ_w^{**} can be neglected and using Equation (2) in (1) we get

$$m_e = \frac{K_e A_s w_A \rho_s}{K_d \rho_s + 1} \tag{3}$$

If $K_d \rho_s >> 1$, Equation (3) reduces to

$$m_e = K_e A_s \frac{w_A}{K_d} \tag{4}$$

Thus m_e is independent of the suspended solids concentration in the limit of high suspended solids concentration. In this limit, the water is in equilibrium with the bed sediment. This is essentially true for suspended solids concentrations greater than 200 $mg \cdot l^{-1}$ for A-1254 and for suspended solids concentrations greater than 600 $mg \cdot l^{-1}$ for A-1242 ($k_d \rho_s$>=5). The USACE pilot study (USACE, 1989) demonstrated that resuspended sediment concentrations can be maintained well below these levels. Resuspended sediment plumes exhibiting these concentrations may be observed with certain dredge types, however, such as a trailing suction hopper dredge operated with overflow (Herbich and Brahme (1991)). Such a dredge should not be used to dredge contaminated sediments, but the PCB evaporation expected under such conditions was calculated to provide a basis of comparison. The expected emissions are shown in Table 5, line (a). The sediment concentrations in Table 4 are used to define an area averaged water column concentration. The suspended sediment concentration is assumed sufficiently high to not influence the water column concentration. Since the dredged sediment is assumed suspended in the aqueous phase the value of w_A on the suspended sediment was assumed to be the same as the concentration from which the suspended material arose. In calculating the evaporation rate, if $w_A > w_{Ac}$, w_A was set equal to w_{Ac}. The calculated fluxes are given in Table 5. The very high emission rates emphasize that certain types of conventional dredging techniques are inappropriate for dredging contaminated sediments.

TABLE 5

Rates of Emission of PCBs from Upper Estuary

(Total area = 800,000 m^2)

	Emission ($g \cdot hr^{-1}$)		
	A-1242	A-1254	PCB (total)
Base Case:			
	22	9	31
Additional Emission From Suspended Sediment Plume:			
(a)	2350	280	2630
(b)	114	32	146
(c)	40	17	57
(d)	215	75	290
After Re-consolidation:			
(e)	0.42	0.17	0.59
(f)	6.6	2.7	9.3

NOTE:

(a) Assuming that $\rho_s K_d >> 1$, i.e., suspended solids concentration is extremely high due to dredging.

(b) Assuming ρ_s is close to the maximum level of 2.5 x 10^{-5} $kg.l^{-1}$ observed during the US ACE pilot scale dredging.

(c) Assuming that the dredges resuspend 17 $g \cdot sec^{-1}$ of sediment (34 $g \cdot sec^{-1}$ total).

(d) Assuming that the dredges are operating in the highest concentration areas other than the Hot Spot.

(e) Assuming dredging efficiency of 99.34% and Hot Spot remediated separately.

(f) Assuming that the Hot Spot is not remediated.

If $\rho_s K_d$ is small then the flux is a function of the suspended solids concentration. Under such conditions the concentration of contaminant released to the water column will be given by Equation (2) and the flux by Equation (3). Note again that Equation (2) was derived assuming equilibrium desorption of PCBs to the aqueous phase from the suspended solids. If the amount of suspended solids as a result of dredging is known then Equation (2) can be used to estimate the amount of PCBs released to the water column.

The USACE (1989) pilot scale study contains useful information on the degree of resuspension of sediment as a result of dredging operations. For example, a single horizontal auger dredge capable of an 8 feet wide cut operating at a dredge advance rate of 13 $ft \cdot min^{-1}$ in a water depth of 3 ft gave rise to a resuspension rate of about 20 $g \cdot sec^{-1}$ at the dredgehead. This translates to a suspended sediment concentration of approximately 1.3×10^{-4} $kg \cdot l^{-1}$ near the dredgehead. The suspended solids concentration reached maximum levels of only about 2.5×10^{-5} $kg \cdot l^{-1}$ at sampling locations about 300 feet from the dredge. However when several dredges are operated simultaneously over a large area for extended periods of time the suspended solids concentration may vary from one location to the other and settling times for re-suspended sediments may also vary.

If the average suspended solids concentration in the sediment plume is assumed to be close to the maximum level that was observed during the USACE pilot scale study, i.e., 2.5×10^{-5} $kg.l^{-1}$, then Equation (3) has to be used for estimating the flux. The flux from

the upper estuary will be 114 $g \cdot hr^{-1}$ for A-1242 and 32 $g \cdot hr^{-1}$ giving a total PCB emission of 146 $g \cdot hr^{-1}$. These results are reported in Table 5, line (b).

Mass release rates of sediment resuspension were also estimated during the pilot study conducted by the USACE (1989). The most efficient dredge was a cutterhead suction dredge that released an average of about 17 $g \cdot sec^{-1}$ of resuspended sediment. Although much of this sediment may quickly settle back to the bottom, an upper bound estimate of the dissolved source strength is to assume that all of the particles remain suspended and can partition contaminants into the water column over the entire upper estuary. If the sediment is contaminated above the critical loading, w_{AC}, these contaminants may also be resuspended; i.e. the effective sediment concentration is not limited to the critical loading. USACE (1989) indicated that two dredges might be used during actual dredging operations, so 34 $g \cdot sec^{-1}$ of sediments were assumed to be resuspended. The estuary has a mean depth of 91.4 cm (ASA, 1987) suggesting a total water volume of 731,000 m^3. ASA (1987) determined the mean residence time of water in the estuary to be 2.4 days for an effective efflux rate of 3,500 $l \cdot sec^{-1}$. Assuming that the resuspension rate of sediments balances with the efflux from the assumed well-mixed estuary, 34 $g \cdot sec^{-1}$ of resuspended sediment would result in approximately doubling the suspended solids concentration in the upper estuary to about 20 $mg \cdot l^{-1}$. With 20 $mg \cdot l^{-1}$ suspended solids about 86% of the A-1242 and 70% of the A-1254 will be in dissolved form. A mass balance on the

assumed well-mixed estuary can then be used to determine the dissolved PCB concentration

$$\rho_w = \frac{w_A\ m_s\ f_w}{q_w + K_\theta\ A_s} \tag{5}$$

where:

m_s = sediment resuspension rate (34 $g \cdot sec^{-1}$)

f_w = fraction of PCB in dissolved form with ρ_s=20 $mg \cdot l^{-1}$

q_w = volumetric flow out of upper estuary (3500 $l \cdot sec^{-1}$)

This equation assumes that the estuary is well-mixed with the single input due to dissolved contaminant release at the dredgehead, and outputs via evaporation and hydraulic flushing. The equation shows that the mass of contaminant released at the dredghead is proportional to the sediment concentration. The average emission rate can be determined by using the area averaged contaminant levels included in Table 4. Line (c) of Table 5 shows the evaporation rate corresponding to the area averaged PCB concentration. The indicated emission rate would be in addition to emissions due to exposed contaminated sediment not yet dredged. Line (d) of Table 5 shows the maximum evaporation rate corresponding to the dredging of the 3500 $mg \cdot kg^{-1}$ sediment, assuming that the sediment contains 70% A-1242 and 30% A-1254. Again, the indicated emissions would be in addition to the base case emissions from exposed contaminated sediment.

An examination of the results in Table 5 indicates that air emissions of PCBs would increase during the dredging operation. On average, using well-controlled dredging techniques, Line (c) of Table 5 shows that the additional air emissions due to dredging can be maintained within a factor of 2-3 of the base case emissions. Even under these controlled conditions, however, dredging of highly contaminated sediment will result in air emissions that are an order of magnitude greater than the base case no-action emissions. Under conditions of poorly controlled dredging or when inappropriate dredging equipment is employed, even the average air emissions can be several orders of magnitude greater than the base case emissions (e.g. Line (a)). It is clear that both the degree of sediment resuspension and its areal extent play a major role in determining the average evaporative emission rate resulting from dredging.

ESTIMATION OF PCB EMISSIONS AFTER SEDIMENT RE-CONSOLIDATION

The post-dredging situation long after the suspended solids have re-settled and the sediment re-consolidated will be similar to the "base case". That portion of the harbor in which dredging was not conducted (concentrations less than 50 $mg \cdot kg^{-1}$) will continue to release PCBs at the rate estimated in the base case analysis. In addition, some contaminants will be left in the sediment after dredging. During the USACE pilot study (USACE, 1989), a hydraulic cutterhead dredge produced 16 cu $yds \cdot hr^{-1}$ while taking two passes through the contaminated region. Assuming that the two passes resulted in perfect removal of the sediment, the only loss was due to resuspension at 17 $g \cdot sec^{-1}$, or 0.66% of the production rate. This loss, combined with the 1.5% of the material contained in undredged sediment (Table 4), results in an estimated average sediment concentration 1.9% of the base case. Assuming that this translates directly into a corresponding reduction in water column concentrations and evaporative losses, the estimated evaporative losses will be 1.9% of the base case after remediation of the esturay. The estimated fluxes are reported in Table 5, Line (e).

The above calculation assumed that the Hot Spot in the estuary was also remediated by some process. If the Hot Spot were not remediated, the source of 29% of the base case evaporation would remain. This means that the total post-dredging evaporative losses would be more than 30% of the base case evaporative losses. This estimate is shown on Line (f) of Table 5.

CONCLUSIONS

(a) Figure 5 represents the total PCB flux before dredging (Base Case), the additional PCB flux from a suspended sediment cloud generated by dredging and the PCB flux after sediment re-consolidation for the proposed dredging operation. The fluxes based on the estuary area of 198 acres are given for each case. The additional flux of PCBs (total) from a suspended sediment plume extending over the entire dredged area for a very high suspended solids concentration was estimated as 2630 $g \cdot hr^{-1}$, almost 85 times the base case, no-action evaporative losses. For practical purposes this gives *the maximum emission rate possible (worst case scenario) from a suspended plume generated by dredging*. Under the best conditions as outlined by the USACE (1989), the additional evaporative emissions due to dredging would be about double the no-action base case emissions. Rapid resettling of resuspended sediments or slowed desorption kinetics of the PCBs could potentially reduce the impact of dredging still further. The post-dredging emissions, assuming separate remediation of the Hot Spot in the upper estuary would be about 2% of the base case no-action emissions.

(b) Significant dispersal of sediments can occur during dredging and possible locales for PCB emissions during dredging were identified.

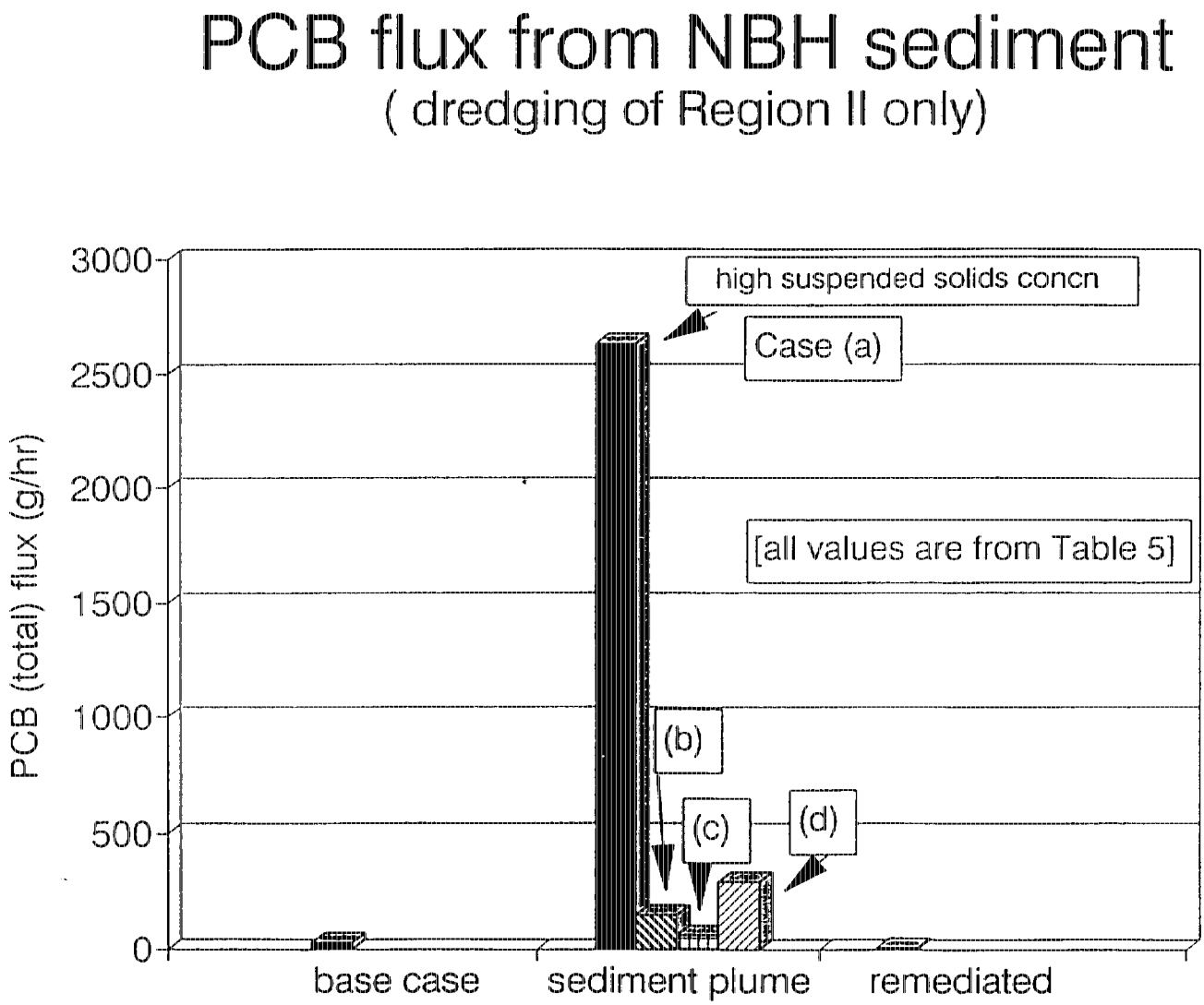

Figure 5. PCB flux from NBH Sediment: (A) Base Case, (B) From a Suspended Sediment Plume and (C) After Remediation and Re-consolidation.

(c) The results of the calculations lead to the *conclusion that dredging is a feasible operation only if adequate measures are instituted to minimize both the areal extent of sediment resuspension and PCB emissions during dredging*.

(d) There is a lack of solid information on the adequacy of the assuming equilibrium partitioning of dredging resuspended sediments and adjacent waters. Available information is for sediment-water exposure times far exceeding the few seconds or minutes that might be appropriate for resettling sediments generated at a dredgehead. Therefore *further work should be directed towards understanding the kinetics of desorption of PCBs from suspended and bed sediments* so that more refined calculations can be made of the extent of PCB emissions when a previously undisturbed sediment is dredged and suspended in water.

REFERENCES

Applied Science Associates (1989), *Estimates of PCB flux and mass balance in Upper New Bedford Harbor*, ASA 89-40, Applied Science Associates, Naragansett, RI cited in Thibodeaux et al. (1990).

Applied Science Associates (1987), *Selected Studies on PCB Transport in New Bedford Harbor*, ASA 86-18, Applied Science Associates, Naragansett, RI cited in Thibodeaux et al. (1990).

Battelle Memorial Institute (1990), *Modelling of the Transport, Distribution and Fate of PCBs and Heavy Metals in the Acushnet River/New Bedford Harbor/ Buzzards Bay System*, Draft Final Report prepared for Ebasco Services, Inc., Contract No: 4236-MOD-0019, cited in US EPA Draft Feasibility Study, Volume II (1990).

Battelle Ocean Sciences (1985) *Sampling Data for the New Bedford Harbor Superfund Project*, Duxbury, MASS, cited in EPA (1990)

Bosworth W S (1991), Personal Communication. Letter dated June 17, 1991 to L J Thibodeaux, Louisiana State University, Baton Rouge, LA.

Brannon J M; Myers T E; Gunnison D and Price C B (1991), *Nonconstant polychlorinated biphenyl partitioning in New Bedford Harbor sediment during sequential batch leaching*, Environmental Science and Technology, 25, 1082-1087 (1991).

Gschwend P M; Wu S C; Madsen O S; Wilkin J L; Ambrose R B and McCutcheon S C (1987), *Modelling the Benthos-Water Column Exchange of Hydrophobic Chemicals*, Final Report No: CR 810472-01, Environmental Research Lab., U S EPA, Athens, GA, 194 pages.

Herbich, J B; Brahme, S B (1991) *Literature Review and Technical Evaluation of Sediment Resuspension during Dredging*, Contract Report HL-91-1, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Thibodeaux, L J; Reible D D; Bosworth, W S and Sarapas L C (1990), *A Theoretical Evaluation of the Effectiveness of Capping PCB Contaminated New Bedford Harbor Bed Sediment*, Final Report, Hazardous Waste Research Center, Louisiana State University, Baton Rouge, LA, November, 92 pages.

Thibodeaux, L J (1989), *Theoretical Models for Evaluation of Volatile Emissions to Air During Dredged Material Disposal with Applications to New Bedford Harbor, MA*, Final Report No: Misc. EL-89-3, U S Army Engineer Waterway Experiment Station, Vicksburg, MS, 44 pages.

Thibodeaux, L J; Reible D D and Valsaraj K T (1991), *Quantitative Release Rates of Bed Sediment Contaminants*, Paper presented at the Contaminated Sediments Assessment Methods Workshop, May 6-8, Village of Galilee, Naragansett, RI.

US Army Corps of Engineers (1989), *New Bedford Harbor Superfund Pilot Study: Evaluation of Dredging and Dredged Materials Disposal*, US ACE- New England Division, Interim Report, June (1989), 52 pages.

U S Environmental Protection Agency (1990), *Draft Feasibility Study: Estuary and Lower Harbor/Bay, New Bedford Harbor*, Prepared by Ebasco Service Inc., EPA Contract Number: 68-01-7250, Volume II, April.